ELASMOBRANCH BYCATCH IN THE CANADIAN NORTHWEST ATLANTIC AND ARCTIC ADJACENT SEAS: COMPOSITION, BIOGEOGRAPHY, AND MITIGATION

by

Aurélie Cosandey-Godin

Submitted in partial fulfillment of the requirements
for the degree of Doctor of Philosophy

at

Dalhousie University
Halifax, Nova Scotia
March 2015

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

ABSTRACT

BACKGROUND: Industrial fishing has profoundly changed the biological state of oceans and seas. While the direct impacts of overfishing on target stocks are being increasingly addressed, unwanted bycatch and discarding of non-target species remains a key challenge of contemporary fisheries management. Excess bycatch mortality is particularly threatening for intrinsically vulnerable species, such as sharks, skates, and rays (Class Chondrichthyes).
QUESTION: In order to address these pressing concerns we ought to know where bycatch occurs in time and space, and how it can be solved through spatial-temporal management and modification of fishing technique.
METHOD: These questions were addressed by developing novel geostatistical methods to track and quantify bycatch "hotspots" in the Northwest Atlantic and adjacent Arctic seas. These tools were applied to analyse (i) bycatch patterns of Greenland shark (*Somniosus microcephalus*), an emerging species of concern in expanding Arctic fisheries; (ii) bycatch of large sharks in pelagic longline fisheries; and, (iii) to estimate total discards of all elasmobranch species in Eastern Canadian fisheries. Additional research also clarified the role of circle hooks in reducing post-release mortality for sharks and field-tested electropositive metals as bycatch deterrents in longline fisheries.
RESULTS: Areas of high bycatch were concentrated in southern Canadian waters bordering the United States: upper Bay of Fundy, Georges Bank, Browns Bank, and Emerald Basin, however northern latitudes were associated with bycatch of deep-water species — which are in need of greater attention. These patterns were driven by both species' abundance and fishing intensity. In 2012, total discard amounts were estimated at 3250 mt (2722-3849, 95% credible intervals) for sharks and 1772 mt (1642-1911) for skates and rays. Total bycatch of large sharks was higher than previously assumed in pelagic longline fisheries. For bycatch mitigation circle hooks reduced post-release mortality, but electropositive shark repellents were not effective in reducing bycatch in pelagic longline fisheries.
CONCLUSION: This dissertation clearly demonstrates the high spatiotemporal variability and inherent complexity of bycatch and supports the use of geostatistical models and fine-scale spatial management for elasmobranch conservation.

LIST OF ABBREVIATIONS AND SYMBOLS USED

Symbol	Description
BPUE	Bycatch-Per-Unit-Effort
CITES	Convention on International Trade in Endangered Species of Wild Fauna and Flora
CPUE	Catch-Per-Unit-Effort
CRDT	Constrained Refined Delaunay Triangulation
CSRS	Canadian Spatial Reference System
DFO	Canadian Department of Fisheries and Oceans
DIC	Deviance Information Criterion
EEZ	Exclusive Economic Zone
GAM	Generalized Additive Model
GAMM	Generalized Additive Mixed Model
GLM	Generalized Linear Model
GLMM	Generalized Linear Mixed Model
GMRF	Gaussian Markov Random Field
GRF	Gaussian Random Field
HMS	Highly Migratory Species
ICCAT	International Commission for the Conservation of Atlantic Tunas
IUCN	International Union for Conservation of Nature
INLA	Integrated Nested Laplace Approximations
MCMC	Markov Chain Monte Carlo
MSC	Marine Stewardship Council
MSE	Mean Squared Error
MSY	Maximum Sustainable Yield
MUR	Multiscale Ultrahigh Resolution
NAD83	North American Datum of 1983
NAFO	Northwest Atlantic Fisheries Organization

Continued on next page

Continued from previous page

Symbol	Description
NASA	National Aeronautics and Space Administration
NOAA	National Oceanic and Atmospheric Administration
RFMO	Regional Fisheries Management Organization
SPDE	Stochastic Partial Differential Equation
SST	Sea Surface Temperature
UTM	Universal Transverse Mercator

ACKNOWLEDGEMENTS

To my supervisors, Boris Worm and Joanna Mills Flemming. Thank you for supporting me for all these years. You allowed me see more of this beautiful world, from Australia to Germany, Norway, and the U.S. where I explored new ideas. Boris, thank you for sharing the art of scientific literature. This is one of your many talents and I am privileged to have learned from the best. Joanna, thank you for thinking outside of the box and daring the impossible. It is astonishing that a few years ago, I barely knew the difference between a mean and a median. This amazing personal growth started with you. You were the one who planted the INLA seed.

Elias Teixeira Krainski; without you my dissertation would have been completely different. I learned so much and I am truly blessed to have had the chance to study with you. Thank you to Havard Rue for your warm welcome to your lab. My research would not have been the same without you kindly accepting my six month invasions to Norway. To the INLA team, you all influenced my work and made my time in Norway special.

To my WWF-Canada sponsors, Bettina Saier, Tonya Wimmer, Jarrett Corke, Daniela Diz, and Robert Rangeley. Thank you for supporting me during both my Masters and PhD journeys. It was an amazing privilege to be involved with the Panda.

Many thanks to Steven E. Campana, Margaret Treble, Sylvain Hurtubise, Peter Comeau, Todd Inkpen, and Carol Ann Peters at Fisheries and Oceans Canada, for your support, advice and suggestions throughout this dissertation.

To my labmates, colleagues and friends at Dalhousie and co-authors, thank you for having shared your knowledge and made this dissertation so much better.

Katie Lo and Edel; we were the best trio of all times. These years at Dalhousie would not have been the same without you two. Brendal Davis and Don McCrimmon, thank you for all the support you two have given me and so much more. Finally, to my mom, dad, sisters, and extended family in Canada and Switzerland, and closest friends, I dedicate this dissertation to you for your endless support and love over the years. You have been with me through every step of the journey, I could not have done it without you. Merci de tout mon coeur.

CHAPTER 1

INTRODUCTION

1.1 Global Fisheries, Bycatch, and Elasmobranch Conservation

Since the mid-1950s, industrial fishing has become a global enterprise affecting all corners of our oceans (*Halpern et al.*, 2008) from coastal environments (*Jackson et al.*, 2001) to deeper waters that were once out of reach to the industry (*Morato et al.*, 2006). Wild capture fish production has even outpaced the worlds population growth (*FAO*, 2014) and is recognized as one of the foremost drivers of ecological change in the oceans (*Pauly et al.*, 2002; *Jackson et al.*, 2001). Among the documented effects of industrial fishing are: the removal of large, long-lived fish (*Myers and Worm*, 2003; *Baum et al.*, 2003), the degradation and loss of critical habitats (*Turner et al.*, 1999; *Thrush and Dayton*, 2002), the simplification of marine food-webs (*Pauly*, 1998), and evolutionary and population structure changes (*Law*, 2000; *Hutchings and Fraser*, 2008). Historically, fishing has rarely been sustainable (*Pauly et al.*, 2005; *Lotze and Worm*, 2009; *Lotze et al.*, 2010). Excessive fishing along with other anthropogenic activities have resulted in severe marine biodiversity loss (*Worm et al.*, 2006). In light of a growing world population, the importance of sustainable fisheries and their supporting ecosystems is at the core of the global food-security debate. Recent efforts to restore marine ecosystems have shown some regional successes (*Worm et al.*, 2009), supporting the idea that these negative trends are at least partly reversible (*Worm et al.*, 2006; *Pauly et al.*, 2002). However, to increase marine biodiversity, exploitation rates need to be further reduced and fisheries need to be managed in a more ecologically sensitive manner. The previous dogma of fisheries management — based on a single-species approach and aiming at maximizing yields

is slowly being replaced by a new paradigm of ecosystem-based fisheries management (EBFM) — focusing on multiple species, habitats, and ecosystem integrity (*Garcia et al.*, 2003). A core element of EBFM is related to bycatch, or non-target catch, some of which may be landed and sold, but much of which is discarded. These interrelated issues of bycatch and discarding are the central focus of this dissertation.

Bycatch is an inherent part of any fishing activity and represents a significant proportion of global marine catch (*Hall et al.*, 2000; *Kelleher*, 2005). The Food and Agriculture Organization of the United Nations (FAO) estimates global discard rates to be some 7.3 millions tonnes (t) per year (*Kelleher*, 2005, Fig. 1). However, quantifying global discards is challenging and previous estimates have exceeded 20 million t, or about a quarter of global marine capture every year (*FAO*, 2011, 2014). There is no single internationally-agreed definition of the term "bycatch"; interpretations vary according to the socio-economic and regulatory context (*FAO*, 2011). As a generic term, bycatch usually refers to that part of the catch that is not the main target species or species assemblages (*Alverson et al.*, 1994), more broadly, it includes all unwanted, unmanaged, and discarded catch (*Davies et al.*, 2009). For some marine taxon, particularly air-breathing marine megafauna (seabirds, marine mammals, and sea turtles) bycatch is the predominant threat to their conservation and the leading cause of population decline (reviewed in *Lewison et al.*, 2014). For other vulnerable species, it may be difficult to differentiate bycatch from targeted catch, this is the case for many elasmobranch species, a group of 1150 cartilaginous fish that include sharks, skates, and rays. In many parts of the world, these species are captured in mixed fisheries and are sold on the market (*Vannuccini*, 1999).

According to the International Union for Conservation of Nature (IUCN) one in four elasmobranch species are now threatened with extinction, placing them among the most at-risk vertebrate groups studied to date (*Dulvy et al.*, 2014). The collapse of many teleost fish stocks coupled with an increasing demand for shark products (primarily shark fins for Asian markets) has resulted in a shift to and drastic increase in fishing efforts toward cartilaginous fish. Global elasmobranch catches reported to FAO tripled since the 1950s with a peak tonnage of 893,000 t in 2000 (*FAO*, 2014). Elasmobranch catch represents a small proportion of total wild fish capture, and, in comparison to the latter, often has low commercial value (with the notable exception of shark fins (*Clarke et al.*, 2007)). This context results in a general lack of resources and funds allocated to the regional

and international management of these species (*FAO*, 2009a). In light of many well-documented population declines (e.g., *Baum et al.*, 2003; *Dulvy et al.*, 2008; *Ferretti et al.*, 2008; *Robbins et al.*, 2006; *Musick et al.*, 2000; *Clarke et al.*, 2013) and the precarious management situation of these species, the FAO developed the International Plan of Action for the Conservation and Management of Sharks (IPOA Sharks) in 1999 (*FAO*, 1999). Since then, a number of activities have been undertaken at both the international and regional level to address and improve the understanding of shark biology, use and management (e.g. *Techera and Klein*, 2011; *Camhi et al.*, 2008; *Worm et al.*, 2014). However, effective management regulations for cartilaginous species are still largely insufficient or lacking in many parts of the world.

This dissertation tackles questions related to both statistical and applied conservation facets of fisheries bycatch related to cartilagenous fish. The overarching goals are to (i) introduce and apply new statistical approaches better suited to bycatch data which can more effectively identify spatiotemporal risk; (ii) advance research on spatiotemporal patterns of elasmobranch bycatch; and, (iii) evaluate mitigation measures in higher-risk fisheries. The focus is on elasmobranch bycatch in Canadian fisheries operating in the Northwest Atlantic and Arctic adjacent seas. However, results are relevant to bycatch problems in other regions and, more generally, the methods applied have wide applications in wildlife conservation and spatial management.

1.2 Elasmobranch Species in the Canadian Northwest Atlantic and Arctic Adjacent Seas

Identifying, quantifying, and prioritising the extent of bycatch and discards is an important step toward mitigating the environmental impact of commercial fishing (*Kirby and Ward*, 2014). Globally, such essential information is often lacking for elasmobranch species, but published estimates reveal that bycatch of sharks, for example, typically exceeds reported landings by a factor of 3 to 4 (*Worm et al.*, 2013; *Clarke et al.*, 2006). In Canada, estimates have been provided for some selected species, for instance, pelagic sharks in the Scotia-Fundy region (*Campana et al.*, 2011a) or winter skate in the Gulf of St-Lawrence (*Benoît*, 2006), but information remains fragmented and often specific to Fisheries and Oceans Canada (DFO) operational regions i.e., Maritimes, Quebec, Gulf, Newfoundland and Labrador, Central and Arctic, and Pacific. Hence, the cumulative risks of different fishing

sectors remained largely unknown, particularly for deep-water and northern populations.

An estimated 40 elasmobranch species (shark, skate, and rays) can be found in the Atlantic and Arctic adjacent seas in Canadian waters (*Corke*, 2012). Thorny skate (*Amblyraja radiata*) is the only species that still supports a large and active targeted fishery, while a few other species are sometimes captured and landed as part of mixed-fisheries, for example, skate species in groundfish fisheries and larger sharks in pelagic longline fisheries. However, the bulk of elasmobranch catches are discarded at sea (*DFO*, 2007b). Unregulated and unreported discarding, limited at-sea monitoring, insufficient taxonomic reporting, and limited biological information have been the common problems for elasmobranch conservation in Canada's Exclusive Economic Zone (EEZ) (*Cosandey-Godin and Worm*, 2010), and elsewhere. Several species have now been considered to be listed under Canadas Species at Risk Act, notably porbeagle *Lamna nasus*, spiny dogfish *Squalus acanthias*, basking shark *Cetorhinus maximus*, blue shark *Prionace glauca*, shortfin mako *Isurus oxyrinchus*, winter skate *Leucoraja ocellata*, Barndoor skate (*Dipturus laevis*), smooth skate (*Malacoraja senta*) and thorny skate, but while legal listings are pending, the problem of bycatch and discards remains largely unaddressed. Figure 1.1 shows detailed maps of the region that was covered by this dissertation.

1.3 Structure of the Dissertation and Statement of Co-Authorship

This dissertation is divided into five chapters in addition to an abstract, introduction and conclusion. Each chapter corresponds to an individual manuscript that was written for publication in a scientific journal. Therefore each chapter follows the normal structure of scientific papers, with an abstract, introduction, methodology, results and discussion. All co-authors contributed to these manuscripts through revisions, comments, advice on research design and interpretation of analyses. In addition, Campana, S. E., Carlson, J. K., Wang, J. H. and Kulka, D. provided additional data or material used in the respective research project. Teixeira Krainski, E. and Bolin, D. also provided important technical support on programming and modeling.

The dissertation is divided into two main parts: 1) bycatch modeling (chapters 2, 3, and 4) and 2) bycatch mitigation (chpt. 5 and 6). Chapter 2 is an analysis of Greenland shark (*Somniosus microcephalus*) bycatch in the Canadian gillnet fishery in Baffin Bay,

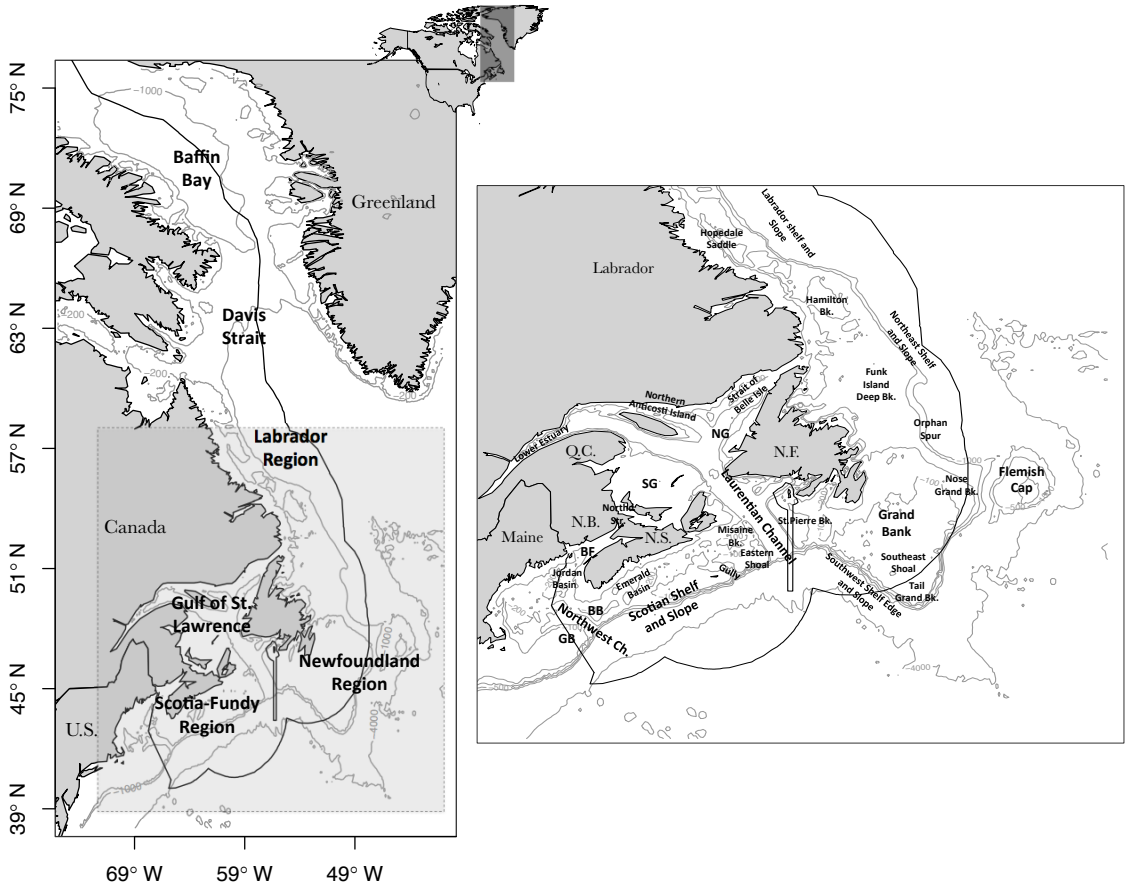


Figure 1.1: Maps showing the study region and specific names and areas of interest. Q.C.: Quebec, N.B.: New-Brunswick, N.L.: Newfoundland, SG: South Gulf Region, NG: North Gulf Region, BB: Browns Bank, GB: George Bank, BF: Bay of Fundy, Bk.: Bank, Northd. Str.: Northumberland Strait, Ch.: Channel.

an Arctic adjacent sea. This chapter introduces and applies new geostatical models that are then used in subsequent chapters. Chapter 3 presents an analysis of pelagic shark bycatch hotspots in the swordfish and tuna longline Canadian fisheries. Chapter 4 analyzes 17 years (1996-2012) of Canadian at-sea observer data and quantifies total discards and cumulative impacts across fisheries on elasmobranch species in the Northwest Atlantic and Arctic adjacent seas.

Bycatch mitigation is primarily focused on pelagic longline fisheries. The Canadian swordfish and tunas longline fishery captures the most pelagic sharks of all Canadian fisheries, hence there was a desire to quantify uncertainties around bycatch estimates and to further investigate plausible spatial and gear mitigation options in this fishery. Following this motivation, Chapter 5 is a meta-analysis on the effect of circle hooks as a mitigation

tool to reduce shark bycatch and discard mortality in pelagic longline fisheries and finally, chapter 6 is an experimental trial of an electropositive shark repellent technology conducted on the Canadian swordfish fleet.

The journal and the publication status of each chapter's paper at the time of submission of this dissertation were as follows:

Cosandey-Godin, A, Teixeira Krainski, E., Worm, B. and Mills Flemming, J. 2014. Applying Bayesian spatio-temporal models to fisheries bycatch in the Canadian Arctic. *Canadian Journal of Fisheries and Aquatic Sciences*. 72: 1-12. (Chapter 2)

Cosandey-Godin, A, Teixeira Krainski, E., Bolin, D., Mills Flemming, J., Campana, S. E., Worm, B. Predicting shark bycatch in a pelagic longline fishery. Submitted to *Conservation Biology* 14 November 2014, currently Under Review. (Chapter 3)

Cosandey-Godin, A., Teixeira Krainski, E., Kulka, D., Mills Flemming, J., Worm, B. In prepation for *Fish and Fisheries*. Elasmobranch Discards in the Northwest Atlantic and Arctic Adjacent Seas: Composition and Biogeography. (Chapter 4)

Cosandey-Godin, A., Carlson, J. K., Burgener, V. 2012. The effect of circle hooks on shark catchability and at-vessel mortality rates in longlines fisheries. *Bulletin of Marine Science*. 88(3):469-483. (Chapter 5)

Cosandey-Godin, A., Wimmer, T., Wang, J. H., Worm, B. 2013. No effect from rare-earth metal deterrent on shark bycatch in a commercial pelagic longline trial. *Fisheries Research*. 143: 131-135. (Chapter 6)

1.3.1 Data Accessibility

R scripts and datasets (similar to original data as per data agreement) are available on the R-INLA website (www.r-inla.org) and personal GitHub repositories (https://github.com/GodinA).

CHAPTER 2

APPLYING BAYESIAN SPATIO-TEMPORAL MODELS TO FISHERIES BYCATCH IN THE CANADIAN ARCTIC

2.1 Abstract

Understanding and reducing the incidence of accidental bycatch, particularly for vulnerable species such as sharks, is a major challenge for contemporary fisheries management. Here we establish Integrated Nested Laplace Approximations (INLA) and Stochastic Partial Differential Equations (SPDE) as two powerful tools for modeling patterns of bycatch through time and space. These novel, computationally fast approaches are applied to fit zero-inflated hierarchical spatio-temporal models to Greenland shark (*Somniosus microcephalus*) bycatch data from the Baffin Bay Greenland Halibut (*Reinhardtius hippoglossoides*) gillnet fishery. Results indicate that Greenland shark bycatch is clustered in space and time, varies significantly from year to year, and that there are both tractable factors (number of gillnet panels, total Greenland halibut catch) and physical features (bathymetry) leading to the high incidence of Greenland shark bycatch. Bycatch risk could be reduced by limiting access to spatio-temporal hotspots or by establishing a maximum number of panels per haul. Our method explicitly models the spatio-temporal correlation structure inherent in bycatch data at a very reasonable computational cost, such that the forecasting of bycatch patterns and simulating conservation strategies becomes more accessible.

2.2 Introduction

High levels of bycatch are a predominant problem in many fisheries around the world and contribute greatly to broader concern about overfishing (*Kelleher*, 2005; *Davies et al.*, 2009). Bycatch commonly refers to the part of the catch that is not a legal target of the fishery; it may be retained and landed but is often discarded (dead or alive) (*FAO*, 2011). It tends to be particularly problematic for long-lived marine megafauna such as sharks, marine turtles, seabirds, and marine mammals (*Lewison et al.*, 2004a; *Hall et al.*, 2000). Under the ecosystem approach to fisheries management, a core objective is to reduce and eliminate bycatch (*Pikitch et al.*, 2004; *Garcia et al.*, 2003). One of the first steps in addressing bycatch issues is to identify and prioritize key conservation and management areas (*Kirby and Ward*, 2014). These priority areas are often referred to as *hotspots* and are locations where bycatch patterns indicate abnormally high risk (e.g. *Huang and Yeh*, 2011; *Cambiè et al.*, 2012; *Roe et al.*, 2014). Detailed information on bycatch patterns and their drivers can help establish effective spatial management, such as time-area closures and spatially explicit gear restrictions and catch quotas. These tools are increasingly used in marine resource management to better integrate multiple and often divergent economical and environmental objectives (*Dunn et al.*, 2011; *Douvere*, 2008). However, such regulations require some understanding of the spatio-temporal dynamics of the system.

Bycatch data are most often collected by at-sea observer programs and comprised of the presence and absence (either count or weight) of non-target species georeferenced by fishing location. They typically also contain information concerning the target species, vessel and gear specification, fishing effort, and environmental information like depth and sea surface temperature. Like many fishery datasets, bycatch data are characterized by complicated statistical features, such as excess of zeros, non-linearity and non-constant variance structure, and spatio-temporal correlation (*Ciannelli et al.*, 2008). These characteristics violate the assumptions underlying basic statistical techniques such that more sophisticated models are required. Statistical tools that explicitly model the sources of zero observations (*Martin et al.*, 2005) are commonly used in fisheries science and custom practice in bycatch studies (e.g. *Minami et al.*, 2007; *Barlow and Berkson*, 2012; *Murray and Orphanides*, 2013).

To deal with non-linearity and non-constant variance, bycatch data are often modeled

with well-established techniques used in catch-per-unit-effort (CPUE) standardization and stock assessments. These methods include generalized linear models (GLMs) (e.g. *Megalofonou et al.*, 2009; *Jannot and Holland*, 2013), generalized additive models (GAMs) (e.g. *Minami et al.*, 2007; *Murray and Orphanides*, 2013) and, to a lesser extent, generalized linear mixed models (GLMMs) (e.g. *Trebilco et al.*, 2010) and generalized additive mixed models (GAMMs)(*Bjorge et al.*, 2013). For a detailed discussion of these models in fisheries research see *Venables and Dichmont* (2004).

Other techniques rely on matching animal telemetry and fleet distribution data to infer spatial overlap and derive bycatch predictions (e.g. *Harden and Willard*, 2012; *Roe et al.*, 2014; *McClellan et al.*, 2009). These techniques offer advantages over fishery-dependent data but are limited to species with available tracking information. Bayesian hierarchical models have also been proposed for estimating bycatch probabilities (*Gardner et al.*, 2008; *Sims et al.*, 2008; *Moore and Read*, 2008), but have not been widely adopted, likely due to high computational costs, and complex estimation routines. However, these methods do represent very powerful approaches for dealing with complex ecological datasets with multiple sources of uncertainty (*Cressie et al.*, 2009), and are readily used in other areas of fisheries science (e.g. *Rivot et al.*, 2008; *Yu and Leung*, 2010; *Harley and Myers*, 2001).

Hierarchical models can simplify complex interactions by allowing parameters to vary at more than one level via the introduction of random effects. The expected value of the response is then expressed conditional on these random effects. Mixed models, such as GLMMs and GAMMs are examples, the first being a fairly straightforward extension of linear regression (*Venables and Dichmont*, 2004). *Trebilco et al.* (2010) model seabird bycatch in the eastern Australian tuna and billfish pelagic longline fishery using GLMMs with a random effect for each fishing vessel. *Ortiz and Arocha* (2004) standardized CPUE indices of billfish bycatch in the Venezuelan tuna longline fishery with all year-interactions treated as random effects. Hierarchical approaches are very well suited to nested data, such as bycatch data, where for example, fishing sets are sampled from a trip, sampled from a vessel, which is part of a larger fleet. In this setting, errors associated with both the data and uncertainties about the ecological process are included, which results in more robust statistical inference (*Cressie et al.*, 2009; *Wikle*, 2007). The advantages of using hierarchical Bayesian models emerge more so as complexity increases, when for example, spatio-temporal variability needs to be modeled explicitly (*Cressie et al.*, 2009).

The Bayesian framework also offers the advantage of providing full inference, such that model parameters and uncertainty can be quantified, which has great utility in applied conservation (*Wade*, 2000; *Wintle et al.*, 2003).

The ad-hoc approach often taken in bycatch studies is to model space using a non-random factor variable, like a $5° \times 5°$ grid cell (e.g. *Brodziak and Walsh*, 2013) or geographical fishing boundaries (e.g. *Bjorge et al.*, 2013; *Barlow and Berkson*, 2012). Others commonly include one or two geographic coordinates (latitude or/and longitude) in their models (e.g. *Yeh et al.*, 2013; *Jiménez et al.*, 2009; *Orphanides*, 2010). Representing latitude and longitude as continuous variables offers the advantage that the data are not isolated in separate units, but these variables are still only incorporated into models as fixed effects and, as such, do not include spatial dependency. When fishing locations are georeferenced in space and the main research questions revolve around spatial predictions, the most appropriate statistical approaches are geostatistics based-models, which intrinsically incorporate the first law of geography: *Everything is related to everything else, but near things are more related than distant things* (*Tobler*, 1970). Hierarchical Bayesian models extend the concept of multilevel structure to include a spatial random effect (Gaussian Random Field — GRF). This random field is a stochastic process indexed in space that essentially represents all spatially explicit processes that may have an effect on the bycatch pattern. This is the real advantage of these models; they are built to approximate and include uncertainties with the entire bycatch phenomena as opposed to only that associated with discretely observed data. In so doing, bycatch hotspots can not only be rigorously identified, they can also be better forecasted for management planning (*Clark et al.*, 2001).

Hierarchical Bayesian models have traditionally relied on Markov chain Monte Carlo (MCMC) simulation techniques, which are computationally expensive and technically challenging, consequently limiting their use. However, a new statistical approach is now readily available, namely INLA via the R-INLA package (http://www.r-inla.org). INLA methodology and its powerful application to modeling complex datasets has recently been introduced to a wider non-technical audience (*Illian et al.*, 2013). As opposed to MCMC simulations, INLA uses an approximation for inference, and hence, avoids the intense computational demands, convergence and mixing problems sometimes encountered by MCMC algorithms (*Rue and Martino*, 2007). Moreover, included in R-INLA, the SPDE approach (*Lindgren et al.*, 2011) is another statistical development that models GRFs much

faster (similar to kriging) as well as constructs flexible fields which are better adept to handle datasets with complex spatial structure (*Lindgren*, 2013). This is often the case with fisheries data since fishermen tend to target particular fishing grounds, resulting in clustered spatial patterns and large regions without any values. Together, these new statistical methods and their implementation in R, allows scientists to fit considerably faster and more reliably complex spatio-temporal models (*Rue et al.*, 2009).

The aim of this paper is to analyze bycatch data using hierarchical Bayesian spatio-temporal models fitted using these two novel techniques. We present an analysis of Greenland shark bycatch in the Canadian gillnet fishery in Baffin Bay. We demonstrate how our approach can yield answers to the ubiquitous questions behind bycatch studies, that is, to: (1) identify times and areas of higher bycatch risk (which may give insight into the species biology, i.e. suitable habitat); (2) identify environmental and fisheries drivers affecting bycatch rates; and (3) identify plausible mitigation measures. In this paper, we first fully describe the Greenland shark bycatch data, briefly discuss relevant aspects of INLA and SPDE, and specify all models that were investigated. We then address model selection, inference, and goodness of fit. Details on the final model are provided in the Appendix A and all R-codes and datasets are available on the R-INLA website. In closing, we discuss future opportunities for the INLA framework in relation to bycatch studies.

2.3 Materials and Methods

2.3.1 Greenland Shark Bycatch Dataset

Baffin Bay and Davis Strait are two large basins between Nunavut's Baffin Island and Greenland that connect the Arctic and Atlantic Oceans. These regions sustain the only large-scale commercial fisheries in Canada's Arctic. A Greenland halibut (*Reinhardtius hippoglossoides*) fishery in Baffin Bay began in 1996 as a small exploratory fishery but has been expanding greatly since 2001 (*DFO*, 2014b). Greenland sharks are commonly caught as bycatch and discarded (*MacNeil et al.*, 2012). Currently, estimates of their biomass, productivity, and fishing mortality are unknown, which limits the ability to predict fishing impact on the Greenland shark population. As a result there is concern that the species could become overexploited due to presumed low productivity; hence a precautionary approach to their management is advocated (*Davis et al.*, 2013). Investigating bycatch data, one of the only available sources of information for the species, can provide insights

into the shark's biology and help establish appropriate management efforts.

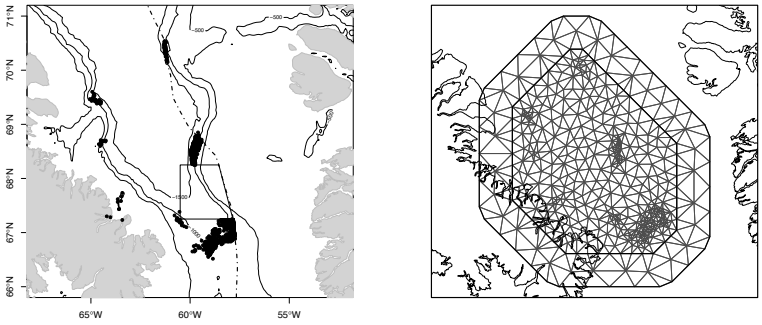
At-sea observers are assigned to monitor the Greenland halibut fishery (*DFO*, 2014b). Observers record location of the net (at the start of the haul), target species (R. hippoglossoides) and weight captured, number of gillnet panels, mesh size, soak duration, depth at which the net was set, and bycatch weight and count per haul. Count information has only recently (since 2008) been recorded for pelagic species, including Greenland sharks.

The observer dataset used for this case study represents 79.2% of the total fishing effort (number of hauls) of the Baffin Bay gillnet fleet over the 4 year period (2008-2011). Data was collected by the Newfoundland and Labrador based observer company and obtained from the Canadian Department of Fisheries and Oceans.

Gillnet vessels tend to be relatively small in size (65 feet) and are allowed to carry a maximum of 500 nets per trip (*DFO*, 2014b). Fishermen usually set several gillnets a day with on average about 40-50 panels in each gillnet (*DFO*, 2014b). Since 2007, a large portion of the Greenland shelf 11 750 km in the southeast was closed to fishing for the protection of narwhal and deepsea corals (*DFO*, 2007a). Gillnet effort is concentrated in areas to the north and south of the closure (Figure 2.1). The fishery has expanded over the years, for example, the northeast of Baffin Bay (71°) is a new fishing ground where the Greenland shelf projects into Canadian waters and provides suitable depths for halibut fishing.

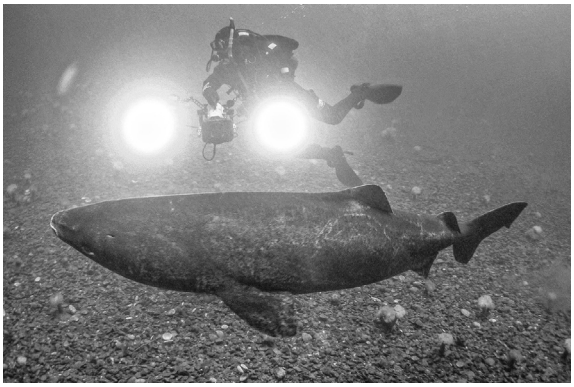
There were 1647 hauls from 26 trips, spanning 2008 to 2011 with all observations occurring between the months of July and November. Three hauls were discarded because there was no information on the weight of Greenland halibut captured; none of these hauls had shark bycatch. Another 3 hauls were inside the closure or outside the 200 nautical mile Canadian Exclusive Economic Zone (EEZ) and hence, were also disregarded. In addition, 25 hauls had recorded weight of Greenland sharks but no associated counts. The majority of these (21 records) were less than 300kg. For each of these entries we assumed a count of one, which makes sense given the average weight of a shark in the others hauls (127 kg). Hereafter, a total of 1641 hauls were included in the analysis (Table 2.1). We refer to each haul as a fishing geolocation, 147 (9%) of which recorded bycatch of Greenland shark; counts varied from 1 to 40 sharks per haul.

In addition to the observed data, ocean depth was extracted from the National Oceanic and Atmospheric Administration (NOAA) global relief model ETOPO1 (*Amante and*



(a) Map of Baffin Bay with isobaths (meters)

(b) Constrained refined Delaunay triangulation



(c) *Somniosus microcephalus* ©Jeffrey Gallant

Figure 2.1: Spatial distribution of the observed Greenland halibut gillnet fishing hauls, location of the fishing closure (solid line), Canadian Exclusive Economic Zone (dashed line), mesh used to calculate the Gaussian (Markov) Random Field in the SPDE approach, and image of a free swimming Greenland shark.

Eakins, 2014). Spatial data were projected from latitude/longitude locations on the surface of a sphere into locations on a plane using NAD83 (CSRS) / UTM zone 19N, a coordinate reference system suitable for use in Canada between 66W and 72W. The relationship between shark bycatch and four variables of potential interest: duration (soaktime in hours and decimal minute), number of gillnet panels, total catch of Greenland halibut (in metric tonnes, mt) and the bathymetry (meters, m) were included in the models. Exploratory analysis revealed no clear relationships between Greenland shark bycatch and these covariates, with the exception of some evidence of a nonlinear relationship with bathymetry (Figure 2.2).

	year	hauls with bycatch	hauls	weight (kg)	counts
1	2008	46	411	5027	66
2	2009	34	378	8765	83
3	2010	20	374	4435	21
4	2011	47	478	92425	400

Table 2.1: Summary of the dataset showing the total number of hauls with bycatch (> 0), number of observed hauls, weight (kg) and counts of Greenland shark bycatch per year.

2.4 Hierarchical Spatio-temporal Model Structure

Similar to a GLM framework, the response, in our case, the observed Greenland shark bycatch (count) at a particular fishing geolocation and time is assumed to have a distribution that belongs to the exponential family and the parameters of the family (ϕ) are linked to a structured additive predictor η through a link function $g(\cdot)$ such that $g(\phi) = \eta$ where the linear predictor η is defined, in our case, as follows:

$$\eta = \beta_0 + \beta_1 Duration + \beta_2 Ngillnet + \beta_3 TC.tspp + \sum_{k=1}^{K} f_k(Bathymetry)\mathbf{w_k} + f(\cdot). \quad (2.1)$$

β_0 is the intercept, β_1, β_2 and β_3 are the (linear) regression coefficients for our covariates: duration or soaktime in hours and decimal minutes (Duration), number of gillnet panels (Ngillnet) and catch of Greenland halibut in mt (TC.tspp), respectively. The function f_k is the sum of smooth functions defining the random effect of bathymetry (m), where regression coefficients vary with bathymetry values (K values) and $\mathbf{w_k}$ is a vector of known bathymetry values defined for each of the fishing geolocations. This is equivalent to a smooth function used in generalized additive models (GAM). $f(\cdot)$ is a semi-parametric function defining the spatio-temporal random effect included in the model (Table 2.2). All of these components form the (non observable) latent field defined as $\theta = \{\beta_0, \beta, f\}$ where β and f are the covariates and smooth functions included in the linear predictor with their appropriate priors (ψ).

The latent field is characterized by a joint normal (Gaussian) multivariate distribution with mean 0 and precision matrix $Q(\psi)$, i.e., $\theta \sim N(0, Q^{-1}(\psi))$. Each observation y_i depends on a linear combination of a subset of the elements of θ defined as:

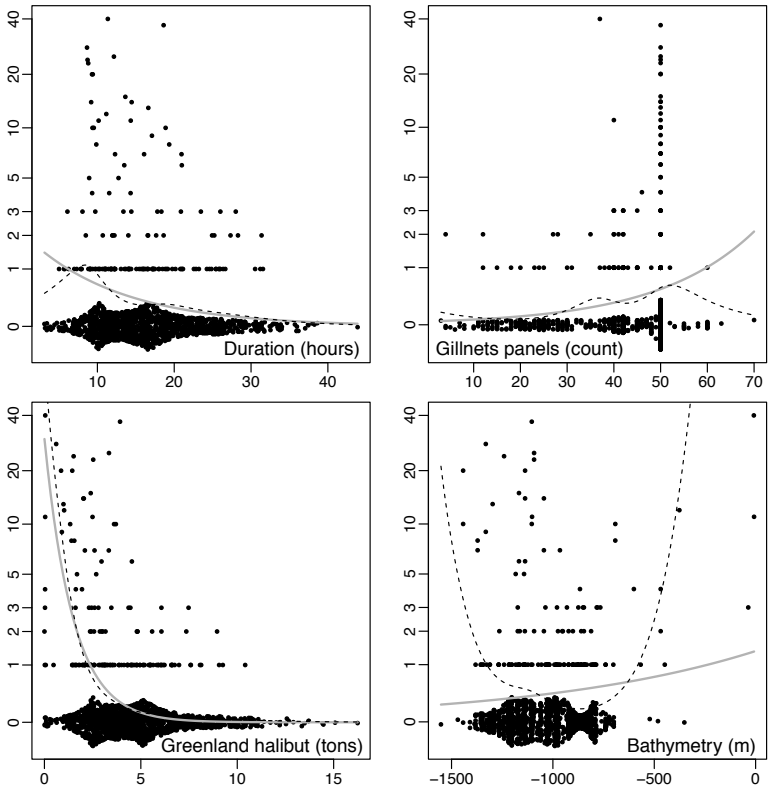


Figure 2.2: Relationship between Greenland shark bycatch counts and covariates of interest. Results of fitting both a Poisson regression (solid line) and a Generalized Additive Model (hatched line) are shown.

$$y_i|\theta,\psi \sim p\left(y_i\middle|\sum_j A_{ij}\theta_j,\psi\right). \tag{2.2}$$

The term A_{ij} is the generic element of an observation matrix A defined by the SPDE approach. Each y_i is independent and identically distributed given the latent field θ. Simplistically, the SPDE method allows one to fully evaluate the continuous GRF as a *discretely* indexed random process, i.e., a Gaussian Markov Random Field (*Lindgren and Rue*, 2013) and it does so by subdividing the domain D (the area of the ocean where the fishing fleet is active) into triangular tiles, creating an index mesh (*Lindgren and Rue*, 2013). Further built-in R-INLA commands are then used to construct the observation matrix A that extracts the values of the spatio-temporal random field at the measurement locations and time points used for the parameter estimation. The likelihood is linked to the latent field through η^* ($\eta^* = \mathbf{A}\eta$) as:

Identifier	Spatio-Temporal Structure
m0	None
m1	Constant: $f(s,t) = f(s)$ is a Matérn correlation structure with $\nu = 1$, where scale and variance need to be estimated
m2	Different each year: $f(s,t)$ is a Matérn correlation structure like $m1$ but with a different realization every year
m3a	Correlated in consecutive years: $f(s,t)$ is a combination of $m1$ with additional correlation structure between neighbouring years
m3b	Exchangeable correlation between years: $f(s,t)$ is similar to $m3a$ but correlation structure between years is repeated (i.e., the correlation between years 1 and 2 is the same as that between years 1 and 4)
m4	Different for each month: similar to $m2$ but the time index is month rather than year
m5a	Correlated in consecutive months of the same year: same structure to $m3a$ for month
m5b	Exchangeable correlation between months of the same year (e.g., 2009-07 has the same correlation with 2009-08 than with 2009-09)

Table 2.2: Spatio-temporal correlation structures considered.

$$p(\mathbf{y}|\theta,\psi) = \prod_{i=1}^{\mathbf{n}} p(y_i|\eta^*,\psi). \tag{2.3}$$

Since the Greenland shark bycatch data are counts characterized by many zeros, we evaluated Poisson and negative binomial distributions, as well as their zero-inflated versions. R-INLA offers different forms of zero-inflation, namely Type 0, 1, and 2. In a nutshell, Type 0 is a hurdle model which treats the response variable as being in a "perfect-state" where the probability of bycatch can only be positive i.e., does not include zero, whereas Type 1 and 2 are mixture models (2 being an extension of 1, that allows for additional zero probability), which describe the probability of being in an "imperfect-state" where positive events (e.g., bycatch) may occur, but are not certain and as such, include both zero and non-zero values. For more details refer to *Martin et al.* (2005) and the R-INLA website. Table 2.2 lists all spatio-temporal correlation structures evaluated in our models. Please see *Cameletti et al.* (2012) and the R-INLA website for further documentation on these latent random field models.

Inference, Goodness of Fit, and Prediction

All analyses were performed using R (*R Core Team*, 2013), specifically the R-INLA package (*Rue et al.*, 2009). The INLA procedure, in accordance with the Bayesian approach,

calculates the marginal posterior distribution of all random effects and parameters involved in the model. There are different options offered in R-INLA with which to approximate the posterior marginal distributions, we used the most accurate one, the Laplace (*Martins et al.*, 2013). We also used the default and recommended settings for priors (*Held et al.*, 2010). These priors are vague priors or approximations of *non-informative* priors, which have little influence on the posterior distributions, hence results are mostly derived from the data (similar to a frequentist approach). Prior sensitivity tests were conducted on the final best model. Further information on priors is provided in the Appendix A.

To use the SPDE approach, the first step is to create a mesh on which the GRF is to be built; this mesh defines the spatial domain of interest (Ω) (Figure 2.1(b)). This is straightforward in R-INLA, but still requires some tuning. The mesh function creates by default a constrained refined Delaunay triangulation (CRDT) for the set of spatial locations provided i.e., uses the observed fishing locations as the triangle nodes. If desired, the mesh could be derived from another sets of points, for example, a regular grid. However, using the discrete fishing locations offers precision and efficiency. With CRDT, smaller size triangles can be defined in areas that have been sampled (fished) and larger ones in areas with no information (no fishing). This saves computational costs and increases the accuracy of the spatial field where there is fishing. Best mesh designs have more regular shape triangles and include some outer extension to avoid the "boundary effect" (increased variance at borders) (*Lindgren and Rue*, 2013). Different mesh designs were evaluated to investigate their effects during model selection.

Best candidate models were selected based on deviance information criterion (DIC) (*Spiegelhalter et al.*, 2002). Further model selection was performed on the 3 best models using cross validation; 10 samples, each with 1100 observations were randomly drawn from the dataset (total of 1641 observations) and fitted with each model, the remaining 541 observations were used for validation. DIC values and the mean squared error (MSE) using the log of the positive observed values and the linear predictor were calculated for each cross validation. Note that MSE were only computed for the positive counts (bycatch of shark) since it is bycatch events that we are interested in predicting accurately.

A final model was chosen for model inference and prediction of bycatch hotspots for 2008-2011. Estimated bycatch can be predicted over the whole spatial domain (Ω) determined by the mesh (Figure 2.1(b)). However, in our case study, there are large areas

of Baffin Bay that are not fished. To avoid predicting bycatch in areas where fishing is unlikely, we first created a lattice of 1 km $\times$ 1 km grid cells over Ω and selected only the cells within a 5 km radius around our observed fishing locations. Note that these numbers (1 km and 5 km) are arbitrary; we could choose a larger or smaller grid cell or radius since we can predict bycatch everywhere in Ω. However, predictions on finer grids and larger areas of Ω will be more computationally intensive.

Bathymetry for each 1 km $\times$ 1 km grid cell was extracted from the NOAA ETOPO1 raster. Since Greenland halibut catch also fluctuated in space and time, we interpolated catch for each grid cell using a weighted mean, where the weights are proportional to $exp(-dist/a)$, where a is equal to 0.5 km, that is inversely proportional to the distance. Spatial predictions of the expected bycatch for each year were then obtained by running the final model with the mean values of the other covariates (duration=15.94 and Ngillnets=43.33), and the grid values of the bathymetry and halibut catch. Results were then plotted on the 1 km $\times$ 1 km grid.

2.5 Results

2.5.1 Model Selection

DIC results of all models tested are presented in Table 2.3. Models ran from less than 10 seconds to 45 minutes on an apple darwin 10.8.0 (64-bit) platform with 2.4 GHz Intel Core 2 Duo Processor. Models fitted with a more elaborate spatio-temporal random field performed better than those with no random field (m0) or one that is constant over time (m1), irrespective of the likelihood. Models with spatial correlation among years (m3a and m3b) did not perform as well as the others, which treated years as independent.

	m0	m1	m2	m3a	m3b	m4	m5a	m5b
poisson	2630.46	1400.51	1171.36	1176.61	1177.74	1144.19	1139.26	1143.51
zpoisson0	1678.52	1394.45	1375.65	1380.46	1376.68	1376.66	1370.33	1368.60
zpoisson1	1665.22	1321.45	1178.56	1180.23	1180.35	1151.52	1153.06	1137.73
zpoisson2	1859.05	1352.43	1186.43	1189.52	1187.79	1160.64	1165.65	NA
nbinom	1442.40	1297.25	1182.62	1183.76	1181.94	1149.13	1157.12	1163.13
znbinom0	1484.98	1415.29	1389.23	1391.14	1390.26	1385.27	1388.60	1400.54
znbinom1	1442.79	1298.10	1185.19	1186.59	1184.92	1154.65	1160.51	1168.08
znbinom2	1448.39	1292.72	1172.73	1174.23	1173.77	1153.38	1159.47	1168.11

Table 2.3: DIC values for all models tested. Poisson and negative binomial with 0, 1, 2 are referring to the zero-inflated types. Note that model m5b with zero-inflated Poisson 2 had a very large DIC suggesting that the model was inappropriate, hence shown as NA.

Correlation among months was very high (around 90%), such that m5 models were very similar. For further model selection, m2, m4, and m5a were investigated. DIC and MSE results from these models validation simulations are shown in Figure 2.3. All models with the exception of their Type 0 zero-inflated Poisson and negative binomial versions have similar DIC values (Figure 2.3). However, MSE results show that the zero-inflated Type 2 Poisson and negative binomial likelihood performs better with all 3 models i.e., MSE values are smaller. However, when these models were tested with different mesh designs with more triangulations and different boundaries, the Poisson likelihoods DIC values were very sensitive (large fluctuations with zero values), whereas the negative binomial likelihoods were stable irrespective of the mesh used. For this reason, the zero-inflated Type 2 negative binomial likelihood and the most parsimonious mesh (simplest boundary and lesser number of triangulations) were selected.

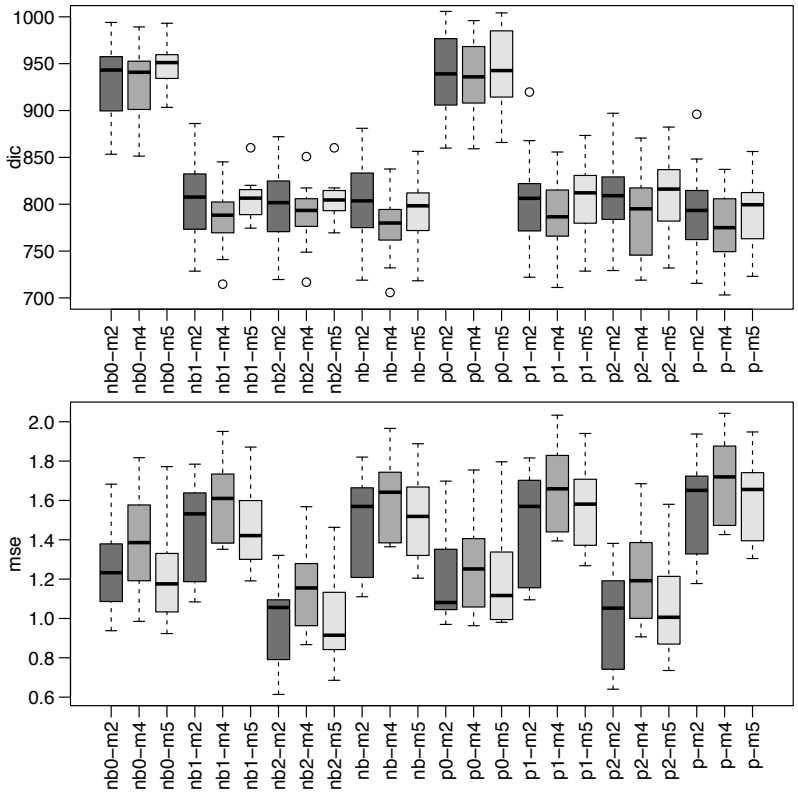


Figure 2.3: Shown are deviance information criterion (DIC) and mean squared error (MSE) box-plots under different likelihood families and spatio-temporal structures, considering different samples of the data. Grey shades refer to m2, m3, and m5 models, respectively. Note that m5 refers to the m5a model.

All 3 models provide similar prediction performance as seen from the MSE boxplots in

Figure 2.3. Model m4 is not a practical model for predictions purpose, since each month is an independent realization of the random field. Under m2 we have a common spatial pattern for months within each year and no correlation between years. Under m5a we have a high correlation between consecutive months, within each year. Note that this may have arisen due to observed hauls in consecutive months but very close in time. For this reason and because it is a more simple model, we selected m2 as our final model. This model includes a different realization of a spatial correlation structure every year. The model ran for less than 6 minutes on our platform. Inference, posterior means, and simulations of fitting model m2 with a zero-inflated negative binomial (Type 2) model are presented below. Additional details on the structure of this final model can be found in the Appendix A.

2.5.2 Inference and Prediction

Results for all parameters are shown in Table 3.3 and Figure 2.4. Duration of the hauls (soaktime) had no significant effect. The number of gillnet panels was positively correlated with the amount of bycatch. For every 10 panels, the expected bycatch increased by approximately 30%. Total halibut catch was negatively correlated, such that higher expected bycatch was associated with hauls where less Greenland halibut was captured. Shallower waters ($<$1000 m) were associated with higher bycatch, whereas deeper waters ($>$1000 m) were associated with less bycatch, however the credible intervals were large (Figure 2.4). The spatial correlation range (nominal range) was 175 kilometers with 95% credible interval [75,292]. The posterior mean of the overdispersion parameter (n) was 2.03 and the 95% credible interval was [0.98, 3.25]. Since this overdispersion parameter was significantly greater than zero, we can conclude that there was evidence that bycatch was clustered. The posterior mean of α was 0.76 with 95% credible interval [0.4,1.19]. This indicated that the extra probability of zero was also significantly dependent on the linear predictor i.e., greater values of the linear predictor resulted in less zeros.

Predicted versus observed bycatch of Greenland shark are shown in Figure 2.5. The model was able to predict with more accuracy small to medium bycatch events (1-10 sharks per haul) but underestimated rare catch of large numbers of sharks ($>$20 sharks). This is expected since the zero-inflated negative binomial distribution is a generalization of the Poisson distribution, which assumes a variance equals to the mean, such that larger means are associated with larger errors. The mean and standard deviation of the yearly spatial

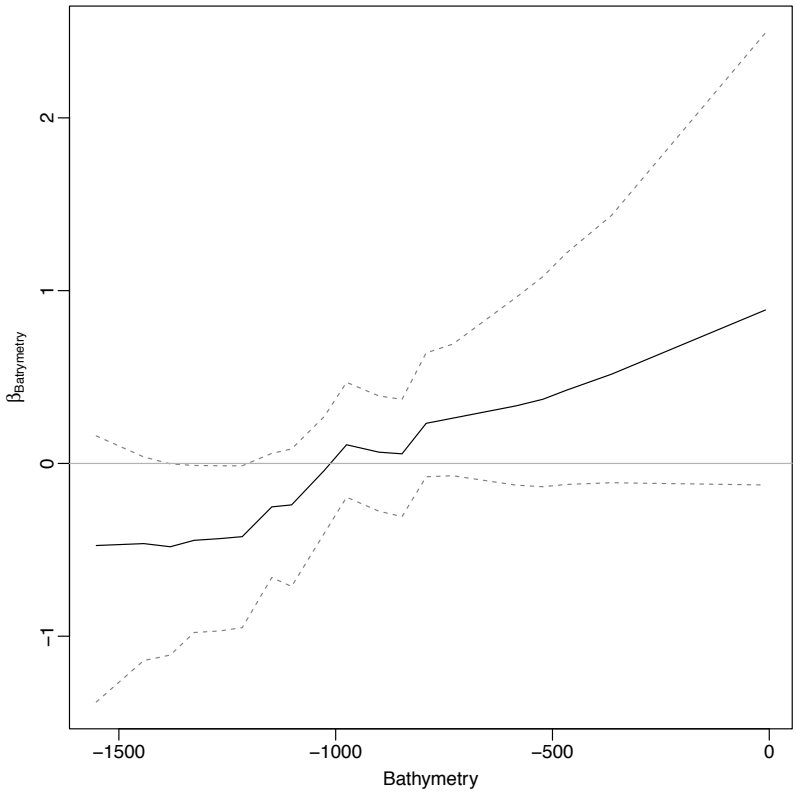


Figure 2.4: Mean and the 2.5% and 97.5% quantiles for the posterior distribution of the bathymetry smoothed regression effect.

random field are shown in Figure 2.6. The final model included a different realization of the GRF every year, and, accordingly, the spatial effects differ from year-to-year. The southeastern part of Baffin Bay has always been negatively correlated with Greenland shark bycatch while the waters near the coast (fished in 2009, 2011) and the northeastern fishing ground (fished in 2010, 2011) were positively correlated with Greenland shark bycatch. Standard deviation patterns for the spatial random field are driven by the amount of information; there is reduced uncertainty where fishing occurred. Prediction of bycatch risk (expected mean counts) of Greenland shark is shown in Figure 2.7. Each year is different, but high levels of bycatch remains the same throughout the year i.e., no month effect. Shallower waters observed in 2011 and the northernmost area in 2011 were associated with nearly 100 times higher bycatch risk, calculated as the exponential differences between the average linear predictor values of these regions $[exp(3.1-(-1.5))]$.

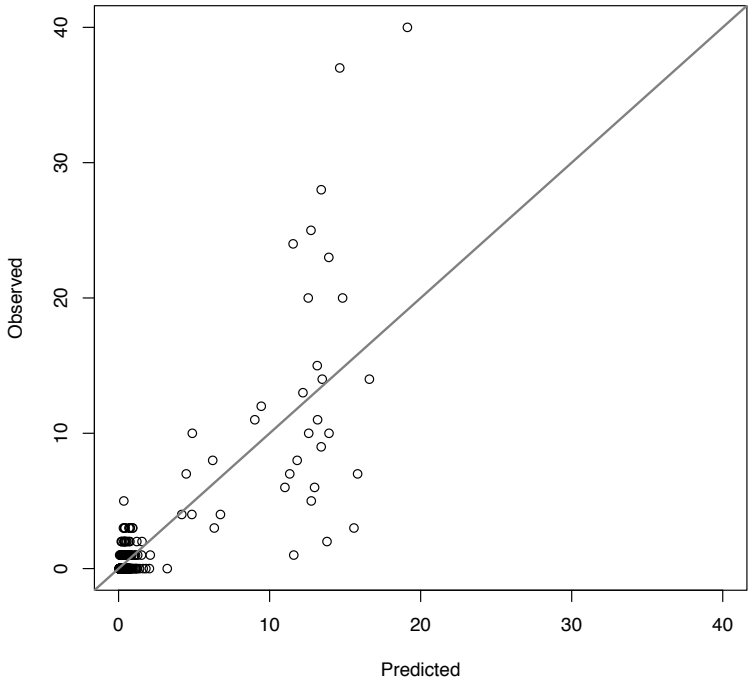
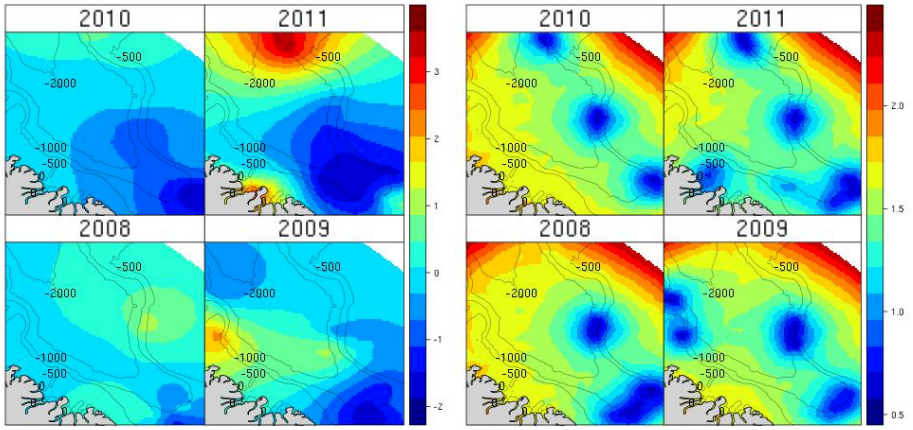


Figure 2.5: Observed versus predicted Greenland shark bycatch (counts) of the final model including a negative binomial zero-inflated Type 2 likelihood and m2 spatio-temporal structure.

2.6 Discussion

The objective of this study was to detail a computationally efficient and statistically powerful approach to analyzing spatially explicit bycatch data and provide an example of its application using the case study of Greenland shark bycatch in Canadian gillnet fisheries in Baffin Bay. The first part of this discussion is concerned with the biological results and management applications, while the second part, focuses on the model and modeling approach, and future applications to bycatch studies.

Our results provide novel insight into the pattern of Greenland shark bycatch in gillnet fisheries. There is evidence that fishing locations less than 175 km apart are spatially correlated, hence sharing similar underlying processes (e.g., oceanographical, biological, fisheries). In our case, most of the fishing clusters are spatially dependent, with the exception of the southernmost and northernmost fishing clusters. This suggests that spatially explicit bycatch management could potentially be considered for these two



(a) Mean (b) Standard deviation

Figure 2.6: Summary of the spatial random effect (Gaussian Random Field) every year included in the final model with negative binomial zero-inflated Type 2 likelihood and m2 spatio-temporal structure.

regions. Greenland shark bycatch patterns are distinct from year to year but hotspots remain the same for the fishing season. Given that the fishing season is currently short, limited by the ice-free season during the summer months, it is sensible that bycatch hotspots remain constant throughout the year. Maps show clear evidence of bycatch hotspots (100 times higher risk), particularly in shallower areas in coastal fjords near Broughton Island as well as to the northeast of Baffin Bay. Some of these coastal areas (Scott Inlet and Sam Ford Trough) may be nursery grounds for the species (unpublished data, N.Hussey), which may explain the higher incidence of bycatch in these areas. The northeast hotspot, on the other end, appeared to be associated with larger (>200 kg) individuals (*Hussey et al.*, 2014), which were present across a broad geographic area in the region (*Campana et al.*, 2013). The northeast hotspot was only present in 2011, although fished as well in 2009. This area may be of some biological importance to the species, but future bycatch monitoring and tracking studies are required here. Greenland shark bycatch is clustered; where bycatch occurs, it is more likely to catch more than one shark in the same haul, which supports recent tracking data suggesting that Greenland sharks have schooling behaviour (unpublished data, N.Hussey). Note that we assume the count of a single animal in hauls with observed bycatch weight but missing count information, such that this "clustering" parameter is most likely larger than estimated.

Given the increasing commercial fishing interest in the region (*Christiansen et al.*, 2014)

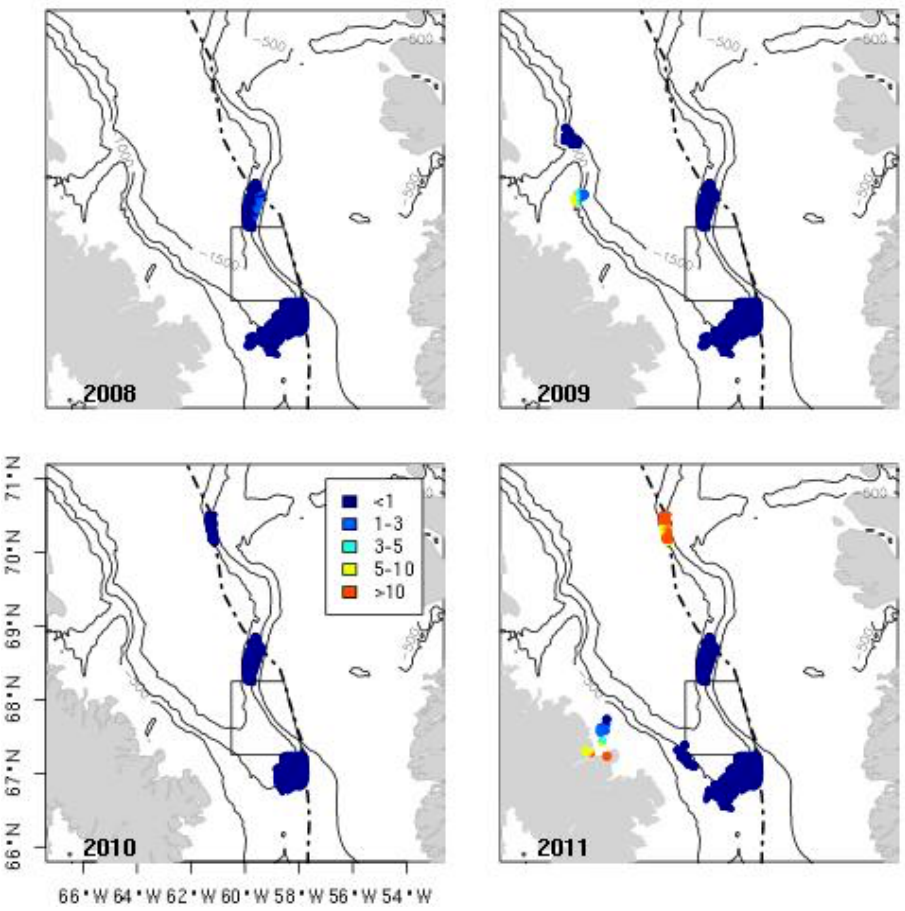


Figure 2.7: Simulated posterior means of the relative risk of Greenland shark bycatch (counts) in the Baffin Bay Greenland halibut gillnet fishery for the years 2008-2011.

and the recent expansion of gillnet fisheries, and high post-release mortality associated with gillnets (usually >70 % e.g. *Manire et al.*, 2001; *Thorpe and Frierson*, 2009), precautionary management which aims at minimizing bycatch of Greenland shark is timely. Our modeling approach allows forecasting of bycatch patterns on a year-by-year basis at a high spatial resolution. In our case study, hauls set in shallower waters, with more gillnet panels, and less halibut yield, resulted in more shark bycatch. Hence sensible bycatch management options could be to limit fishing in hotspots (e.g., using time-area closure or spatially explicit discard quotas), set a maximum number of gillnet panels per haul, or limit fishing to deep waters (>1000 m). The relationship with low halibut yield associated with higher shark bycatch is less intuitive. Greenland halibut may display some avoidance behaviours when confronted with higher abundance of predators, but more likely, habitat conditions along the Baffin coast (e.g., temperature, salinity) are not as favourable for Greenland

Halibut as they are for Greenland sharks. The above is by no means an exhaustive list of mitigation options, but highlights those that can be directly inferred from our model results. We caution the readers that the analysis is based on a limited dataset with only 4 replicated years. The data was statistically challenging with overdispersion, zero-inflation, and discontinuous temporal and spatial domains i.e, patchy fishing pattern, and not all fishing grounds were fished every year. Our case study was not attempting to describe the full cost-benefit of bycatch mitigations, but rather to present an effective modeling scheme for inference and predictions, which could eventually be used to quantitatively assess the effect of spatially-explicit management scenarios.

Our analysis captures the variations in fishing effort as well as the correlation with bathymetry and fishery variables, in addition to the spatio-temporal random effect. This spatio-temporal random field was necessary to rigorously capture the heterogeneity of Greenland shark bycatch. In general, bycatch studies always find that spatial and temporal factors are very influential factors for bycatch distribution, while other factors are of lesser importance (e.g. *Megalofonou et al.*, 2009; *Winter et al.*, 2011). Hotspots are sensitive to the structure of the data; *Viana et al.* (2012) showed that omitting the hierarchical and spatial and temporal structure in the modeling pattern of discards in the Irish Sea greatly affect the ability to characterize hotspots. This strongly indicates that there is a need to better incorporate space (and time) within the modeling scheme when analyzing bycatch data. However, there has been a slow progress in this area, most likely due to the unique difficulties arising from the spatial structure of fisheries data and the large computational burden of fitting complex spatio-temporal models.

The proposed Bayesian hierarchical spatio-temporal models are extremely powerful and flexible, specifically when the focus of the study is to quantify the spatial magnitude and uncertainty of this process. This is because the latent field is explicitly specified in the hierarchical model, such that information can be directly extracted from the model. To our knowledge, no applied bycatch studies to date have included such random processes in order to better capture the spatial and temporal correlation in the data (but see *Sims et al.*, 2008; *Viana et al.*, 2012). This is somewhat surprising considering the fact that the use of GRFs is very popular in the field of epidemiology (*Best et al.*, 2005) where the research interest is analogous to bycatch studies: describing the variation of a biological process over space and time.

However, it is our hope that INLA, SPDE, and their R interface will allow fisheries bycatch scientists to explore these sophisticated models, which can now be more easily fitted. INLA avoids the vast computational demands of MCMC methods and SPDE allows to directly model geo-referenced data rather than gridding the data first, which necessarily results in a significant loss of information. Moreover, probability maps can easily be obtained from the posterior distribution and results are more intuitive and interpretable (i.e., the probability is explicit) for non-statisticians than p-values (*Wade*, 2000). Unfortunately, many fisheries receive very little observer coverage, mostly because of the expense associated with such programmes. Hence, observer-based bycatch datasets are often of poor quality and maps of raw bycatch rates can be misleading (*Sims et al.*, 2008). In such cases it is especially important to include the hierarchical structure and spatio-temporal heterogeneity in order to better estimate total bycatch and hotspots locations for the fleet of interest. The framework we developed here allows for this, and can be extended to complex situation where multi-species bycatch occurrences are of interest (*Sims et al.*, 2008). Moreover, since the computational cost is low and time constraints not an issue, scientists can now simulate a large number of cost-benefit scenarios with which to investigate best management options.

R-INLA is continuously evolving, greatly extending the scope of models available to applied scientists. These methods have now been applied in health (*Bessell et al.*, 2010; *Li et al.*, 2012; *Musenge et al.*, 2012; *Wilking et al.*, 2012), and climate research (*Cameletti et al.*, 2012), ecology (*Johnson et al.*, 2011; *Haas et al.*, 2011; *Holand et al.*, 2013; *Illian et al.*, 2013), and recently in marine ecology research (*Muñoz et al.*, 2012). The package interface resembles the *glm* function in R, such that scientists already familiar with these common tools should find it straightforward to use R-INLA. An advantage of the Bayesian approach is that prior knowledge can be incorporated into the model in order to get more precise posterior distributions. Instead of using non-informative priors (default priors in R-INLA), user-friendly commands for prior specificity are currently undergoing development (H.Rue, pers.comm.). Moreover, there is great interest in bycatch studies in combining datasets from different sources (e.g., oceanographic data) to better predict high incidental catch. Each subset of data may be described by a different family of distributions. Fitting models with different likelihoods is nearly impossible with most popular R packages, but is implemented in R-INLA. Please see *Martins et al.* (2013) for a complete overview of

INLA's new features.

The models that we have proposed here are not yet widely used in fisheries and aquatic sciences. To this end we have demonstrated the merits of using Bayesian hierarchical spatio-temporal models, for bycatch studies in particular, but have gone further to present an effective way of fully implementing these models (R-INLA). Bycatch studies tend to have a strong ecological focus, seeking insight into the causal drivers of bycatch frequency and spatial distribution. However, increasing demand for mapped products for fisheries spatial planning is requiring scientists to look into new statistical techniques which can explicitly include the inherent uncertainties associated with the observations, sampling, models, and parameters, such that accurate predictions of bycatch can be obtained and included in decision-making. Under a Bayesian hierarchical model, all uncertainties e.g., resulting from sampling bias and uncertainty due to the variability across space (and time) are incorporated and as such it is straightforward to obtain posterior predictive probability distributions for non-sampled areas as demonstrated in our case study. By making available both the data and the code necessary to implement the models for our particular application, we hope to facilitate the uptake of these powerful approaches by marine scientists.

Acknowledgements

We acknowledge DFO, especially Margaret Treble who provided in-depth details on the Greenland halibut fishery and Todd Inkpen and Carol Ann Peters who supplied the observer data. We are greatly indebted to Havard Rue for hosting A.C.G. and for generous advice and suggestions throughout this project. This research was supported by the Natural Sciences and Engineering Research Council of Canada (NSERC) through an Industrial Postgraduate Scholarship to A.C.G., a Discovery Grant to J.M.F. and a Canada Strategic Network Grant (NETGP 375118-08) to J.M.F. via the Ocean Tracking Network. In addition, A.C.G. received funding from WWF-Canada and E.T.K. received funds from the Research Council of Norway.

CHAPTER 3

PREDICTING SHARK BYCATCH HOTSPOTS IN A PELAGIC LONGLINE FISHERY

3.1 Abstract

High levels of bycatch and discards of marine megafauna in pelagic longline fisheries are a growing conservation concern around the world. Prioritizing mitigation measures in areas of high bycatch incidence, or "hotspots", has been proposed as a key tool for the conservation of a number of vulnerable bycatch species, such as sharks. However, it has proven difficult to reliably track the occurrence and extent of bycatch hotspots in time and space, especially so for highly migratory species, and given limited scientific observer coverage. This research provides the first geostatistical model-based estimates of shark bycatch hotspots in a pelagic longline fishery. We used new statistical approaches to fit hierarchical Bayesian models that account for correlation in both time and space. Models were parameterized from scientific observer data (5% of total effort) and then used to predict total bycatch occurrence across all fished locations. We further quantified the probability of exceeding a certain level of bycatch for each location. In our case study, the Northwest Atlantic Canadian pelagic longline fishery (2003-2013), we identified clearly delineated hotspots of porbeagle (*Lamna nasus*), shortfin mako (*Isurus oxyrinchus*), and blue shark (*Prionace glauca*) bycatch. Results shows that blue shark bycatch increased since 2003, whereas porbeagle and shortfin mako shark bycatch remained fairly constant. Total bycatch estimates were higher than previously assumed, particularly for porbeagle and shortfin mako sharks. Hotspots were spatially aggregated in three key areas, Browns

Bank, Emerald Bank and Emerald Basin, on the Scotian Shelf. These results imply that spatial management could be a viable method to reduce catch rates and overall mortality of sharks in longline fisheries. Our approach provides a new and practical tool to inform decisions on location, extent, and duration of plausible spatial bycatch mitigation measures. This method can easily be extended to investigate multi-species bycatch risk and cumulative impacts of different fisheries.

3.2 Introduction

Pelagic longlines are used extensively throughout all oceans, and may cover more surface area than any other method of fishing or hunting, with total fishing effort around 1.4 billion hooks in the year 2000 (*Lewison et al.*, 2004b). These fisheries are often targeting highly migratory species (HMS) such as tunas, billfishes, swordfish, and sharks. It is now widely accepted that sharks are more vulnerable to overexploitation than other fishes, as their maximum population growth rate (r_{max}) is less than half that for teleosts, on average (*Hutchings et al.*, 2012). Most scientific assessments demonstrate that shark populations, particularly pelagic species have declined to low levels (e.g. *Baum et al.*, 2003; *Ferretti et al.*, 2008), raising concerns about possible extinctions (*Dulvy et al.*, 2014).

Motivated by a growing public awareness, some management advances have been made, for example, most Regional Fisheries Management Organizations (RFMOs) have banned the previously widespread practise of "shark-finning" and prohibited the landing of some of the most vulnerable species (e.g.,bigeye thresher, oceanic whitetip, hammerhead, and silky sharks caught in association with fisheries managed by the International Commission for the Conservation of Atlantic Tunas, ICCAT). In addition, several species were recently listed under the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). However, these measures have not reversed declines in most regions (*Clarke et al.*, 2013), and average mortality estimates exceed the rebound potential of most species (*Worm et al.*, 2013).

At this point, there is little doubt that high levels of bycatch and discarding represent the most significant threat to sharks, and indeed other elasmobranch species (skates and rays), according to the IUCN (*Dulvy et al.*, 2014). For example, longline fishery discards of blue sharks (*Prionace glauca*) in the North Atlantic were estimated to exceed nominal catch more than two-fold (*Campana et al.*, 2009b). Such additional mortality can impair current

recovery strategies and change the perspective on species conservation status. For instance, the Northwest Atlantic porbeagle (*Lamna nasus*) population is classified as Endangered by the IUCN Red List of Threatened Species (2006). The population suffered severe declines in the 1960s and 1990s from extensive directed fisheries and, despite some signs of recovery, the stock remains small and well below maximum sustainable yield (MSY) (*Campana et al.*, 2012). Recovery time for this species is highly susceptible to changes in human-induced mortality, including bycatch (*Campana et al.*, 2012).

How can unsustainable bycatch levels be addressed? Conventional fisheries management approaches often focus on total catch, effort, or gear restrictions. Spatial management is a complementary approach, that is sometimes used to meet broader ecosystem objectives, or to minimize socio-economic impacts (*Keefe et al.*, 2014). Temporary spatial restrictions ("time-area closures"), for example, have a long history in protecting species during vulnerable life stages, for example in spawning aggregations (e.g. *Rijnsdorp et al.*, 2012) may be used to reduce conflicts among competing sectors (e.g. *Findlay et al.*, 2003). Such temporary closures are also becoming popular in addressing bycatch problems (*Catchpole and Cadrin*, 2014), even for HMS species that move extensively (e.g. *Hobday and Hartmann*, 2006; *Jensen et al.*, 2010; *Grantham et al.*, 2008). For examples, temporary closures in pelagic longline fisheries off Baja California, Mexico contributed to an estimated 12–22% increase in the local abundance of striped marlin (*Kajikia audax*) (*Jensen et al.*, 2010). Time-area closures often work best in combination with other mitigation approaches such as individual vessel bycatch quotas, gear-modifications, fleet communication, and strong participation by the fishing industry (*Keefe et al.*, 2014). For instance, bycatch mortality of albatross and petrel species were reduced to near-zero in the area of the Southern Ocean managed by the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR)(*Croxall*, 2008). This was achieved by a comprehensive mitigation programme which included several fishing gear modifications as well as seasonal closures (*Waugh et al.*, 2008). Where seasonal closures were not implemented, it took an additional 4 years to achieve equivalent reduction in bycatch (*Croxall*, 2008).

However, compared to other HMS, particularly seabirds and marine turtles, such spatial management has not received much attention for sharks caught on longlines (but see *Grantham et al.*, 2008; *Petersen et al.*, 2009). Identifying spatial and temporal patterns of high interaction rates (*hotspots*) would also provide important information on shark

distribution in vast areas of our oceans for which we have little information. Indeed, information on habitat preferences remains scarce for many HMS species (*Pikitch et al.*, 2008). Most longline fisheries have very limited scientific observer coverage, often in the order of 5–10% of the total fishing effort. Thus, reliable information on actual bycatch levels is sparsely distributed in space and time, making it challenging to track the occurrence and extent of bycatch hotspots in a statistically sound fashion.

Here we make use of hierarchical Bayesian modeling techniques that include a spatio-temporal random effect which accounts for processes that may have had an effect on shark bycatch rates in space or time (*Cosandey-Godin et al.*, 2014). We further introduce a method to quantify the probability of exceeding a certain level of bycatch, providing a framework to identify zones of interest for bycatch management and decision making. We analyzed the Northwest Atlantic Canadian pelagic longline fishery to identify bycatch hotspots of porbeagle, shortfin mako (*Isurus oxyrinchus*) and blue shark, which comprised over 99% of the total shark catch in this fishery. These three species are the most common large shark species in Atlantic Canada (along with Greenland shark in northern regions, *Somniosus microcephalus*). For 2010, *Campana et al.* (2011a) estimated that this fishery alone accounted for 58% of porbeagle, 70% of shortfin mako, and 99% of blue shark discards in Atlantic Canadian waters. Porbeagle and shortfin makos are valuable bycatch and are sometimes landed, whereas all blue sharks are discarded at sea, due to a lack of market (*Campana et al.*, 2011a). As directed fisheries have ceased, the sole ongoing threat identified for these three populations in Canadian waters is fishing mortality due to bycatch. There are strong incentives to reduce bycatch of sharks in this fishery, particularly when fishing for swordfish, since swordfish licences were granted a Marine Stewardship Council (MSC) certification in spring 2012 (*DFO*, 2013).

The objectives of this research were to 1) provide an annual estimate of total shark bycatch for the Canadian longline fishery, 2) identify times and areas of high incidence of pelagic shark catch in the longline fishery and 3) present and apply a novel statistical technique to the analysis of hotspots, which has potentially wide applications in marine spatial planning and management.

3.3 Materials and Methods

3.3.1 Datasets

Fishermen logbooks and at-sea scientific observer data from the Canadian pelagic longline fishery were provided by Fisheries and Oceans Canada, Scotia-Fundy Region for the years 2003-2013 on a per-set basis. A longline set consists of the setting and hauling in of a 50-100km fishing line, typically carrying approximately 1000 baited hooks, and is the most basic form of data aggregation. This particular longline fishery primarily targets swordfish (*Xiphias gladius*) and, to a lesser extent, albacore (*Thunnus alalunga*), bigeye (*T. obesus*), and yellowfin (*T. albacares*) tunas (*Paul and Neilson*, 2009). Fishing extends from Georges Bank south of Nova Scotia to the Flemish Cap, east of Newfoundland, and occurs primarily from May through November (Figure 3.1). Fishermen logbooks contain information on area fished, species landed (which may include some non-target species), fishing effort, vessel and gear specifications. Logbooks provide the most complete census of all fishing activities, however anything that is not landed (i.e. discards) is not recorded. At-sea scientific observers fill this gap and provide detailed and independent information on all catches, including both landings and discards. However, observers sample only a random subset of fishing trips, averaging about 5% of total days fished (*DFO*, 2013), and varying between 1-10%. Here, the observed fishing sets are used to model and predict 1) total shark bycatch in the logbook dataset (total fishing effort) and 2) bycatch hotspots.

In both the observer and logbook datasets, records with missing spatial information or apparent errors (e.g., zero values, on land, or northeast of Greenland) were removed. After data cleaning, the final dataset available for analysis was comprised of 887 observed sets and 20,522 logged sets for the years 2003-2013. Based on the number of sets, there is an average of 4.3% observer coverage per year. We used the number of sharks captured as the dependent variable, however, for some fishing sets only weights of caught sharks were recorded. For these entries and for each species separately, missing counts were estimated based on the average count-to-weight ratio calculated from the remaining sets. For blue, porbeagle, and mako sharks, respectively, 63, 22, and 15 missing data entries were completed with an averaged observed weight of 40.88 kg, 23.65 kg, and 30.28 kg per shark. In total, 1,421 porbeagle sharks, 28,850 blue sharks, and 1,767 shortfin mako sharks were recorded in the observer dataset for the whole time-series (2003-2013). Blue sharks were captured on nearly all fishing sets (96%), while porbeagle and mako sharks

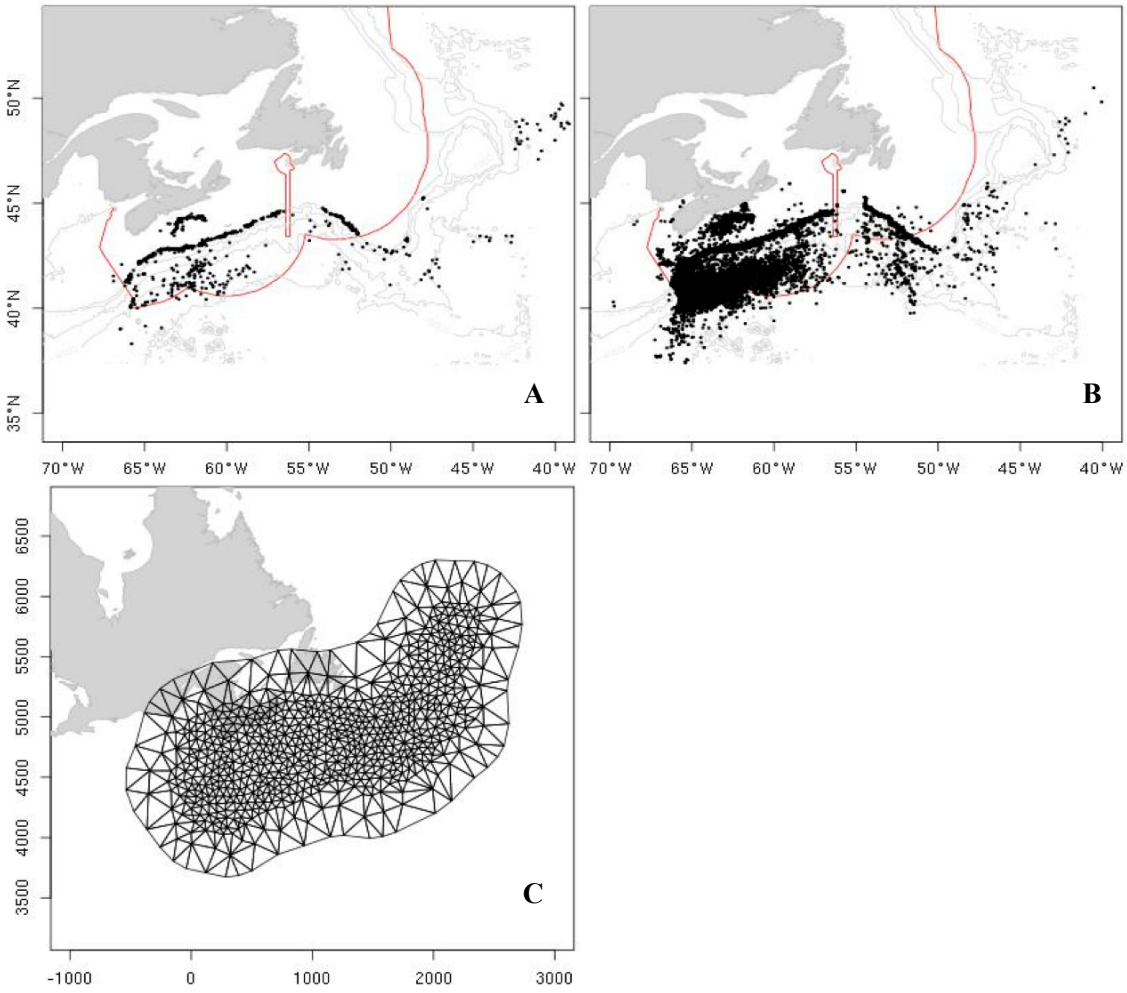


Figure 3.1: Map of Canadian Atlantic waters showing the locations of observed (A) and total fishing effort (B) in the Canadian pelagic longline, 2003-2013 and the mesh (C) used to build the random field in the spatial models. Note that the latter map shows the projected coordinates in km. The red boundary indicates the 200 nautical mile Canadian Economic Exclusive Zone (EEZ).

were captured less frequently: on 30% and 53% of the fishing sets, respectively.

In order to comprehensively model bycatch patterns, a total of 11 predictor variables were investigated (Table 3.1). Spatial catch data were projected from latitude/longitude locations on the surface of a sphere into locations on a plane using NAD83 (CSRS) / UTM zone 20, a coordinate reference system suitable for use in Canada between 66W and 60W. Targeted species were explicitly indicated in the observer dataset, but not in the logbook data. Thus, for the logbook dataset, the main target species was estimated based on weight composition of the catch; the largest catch was assumed to indicate the targeted species: Swordfish (SWO), Albacore tuna (ALB), Yellowfin tuna (YFT), or Bigeye tuna (BET). Note however that the observer dataset did not report any targeted fishing sets for ALB, and for this reason all tuna species were treated under a combined tuna category. Two additional predictor variables were obtained from external sources. Bathymetry was extracted from the National Oceanic and Atmospheric Administration (NOAA) global relief model ETOPO1 (*Amante and Eakins*, 2014) and sea surface temperature (SST) from the NASA Multi-Scale Ultra-High Resolution (MUR) dataset. NASA MUR provides daily SST at the highest spatial resolution currently available (0.01 degrees in longitude-latitude coordinates). Based on previous studies of this particular fleet, SST appears to be a reasonable predictor of shark bycatch, particularly for blue sharks (*Carruthers et al.*, 2011; *Carruthers and Neis*, 2011).

3.3.2 Bycatch per Unit of Effort Standardization Models of the Observer Dataset

To analyze trends in shark bycatch rates, we used hierarchical Bayesian spatio-temporal models that have recently been introduced as an efficient and powerful tool for analyzing bycatch data (*Cosandey-Godin et al.*, 2014).

As in the more commonly utilized generalized linear modeling (GLM) framework, the response is assumed to belong to the exponential family and the parameters of the family (ϕ) are linked to a structured additive predictor (η) through a link function. For each species, the response, bycatch count ($y_{i,t}$) made at each observed fishing location (i) at time (t) is assumed to be dependent on a latent field ($x_{i,t}$) which is defined by the linear predictor (η) and comprised of the set of covariates (z_k) and the spatio-temporal random field (ξ) with appropriate priors (ψ). The latent field is characterized by a joint Gaussian multivariate distribution with vector mean 0 and precision matrix $Q(\psi)$, i.e.,

Predictor	Abbreviation	Type	Description
Date	date	Continuous	day-month-year per fishing sets, 2003-2013
Longitude	x	Continuous	NAD83/UTM zone 20 projection in km
Latitude	y	Continuous	idem
Targeted species	tspp	Categorical	Swordfish (SWO) and Tunas unspecified (UNS)
Amount of targeted species	tsppK	Continuous	Metric tons of targeted species kept and landed per fishing set
Tonnage	tonnage	Categorical	Class registered tonnage of the vessel, 1-4 (i.e., size of the boat)
Number of hook	hookn	Continuous	Number of hook set on the line
Time	time	Continuous	Start time of the set in hours decimal minute (local time)
Soaktime	duration	Continuous	Time difference between the end of the set time and the beginning of the haul time in hours decimal minute
Bathymetry	bathy	Continuous	NOAA global relief model ETOPO1 in km
MUR-SST	sst	Continuous	NASA MUR-SST in degree celsius

Table 3.1: List of covariates considered in the bycatch-per-unit-of-effort (BPUE) modeling of the observer dataset of shark bycatch counts in the Canadian pelagic longline fishery.

$x_{i,t} \sim N(0, Q^{-1}(\psi))$. The continuous random spatial process is introduced in the model using a discretely indexed field, see *Cosandey-Godin et al.* (2014) and references for further details. This random field accounts for the spatial correlation through a Matérn covariance function, which depends on distances between points (i.e., fishing set locations).

The first step in constructing the spatial random field is to create a triangulated mesh which defines the spatial domain of interest. We constructed non-convex boundaries around the entire set of fishing locations, including both the observer and logbook datasets (Fig. 3.1). The largest allowed triangle edge length was set at 100 km in the inner boundary and 500 km in the exterior boundary of our mesh. The latter boundary serves as a computational buffer (e.g., eliminates the issue of increased variance at the boundary). Different mesh designs were tested for sensitivity. Each observation $y_{i,t}$ depends on a linear combination of a subset of the latent field, $\eta^*_{i,t} = \mathbf{A_t}\eta$, where $\mathbf{A_t}$ is a matrix for time t which defines the mapping between the mesh and the observations (fishing set locations). The likelihood is hence defined by:

$$p(\mathbf{y}|\mathbf{x_{i,t}}, \psi) = \prod_{\mathbf{i=1}}^{\mathbf{n}} p(y_{i,t}|\eta^*_{i,t}, \psi). \tag{3.1}$$

The joint posterior distribution of $x_{i,t}$ and ψ is given by the product of the likelihood, the spatio-temporal random field density, and the hyper-parameter distribution prior ($p(\psi)$). This framework allows us to explore different random fields and to develop dynamic models where ξ evolves according to some time-varying processes, for example an autoregressive process. Different sets of covariates and spatio-temporal random fields were tested with several likelihoods, specifically Poisson, Negative Binomial, and their zero-inflated forms as the data tended to be overdispersed and had many zero observations for some species. All analyses were performed using R (*R Core Team*, 2013), specifically the R-INLA package (r-inla.org) (*Rue et al.*, 2009). In accordance with the Bayesian approach, R-INLA calculates the marginal posterior distribution of all random effects and parameters involved in the model. We used the default and recommended settings for priors; these priors are approximations of *non-informative* priors, which have little influence on the posterior distributions. Each shark species was modeled separately. Table 3.2 lists and provides a brief description of all models considered. Best candidate models were selected based on the deviance information criterion (DIC) (*Spiegelhalter et al.*, 2002).

Id	z_k	ξ	*L.nasus*	*P.glauca*	*I.oxyrinchus*
m0	$\beta_{tspp} + \beta_{tsppK} + \beta_{tonnage} + \beta_{hookn} + \beta_{time} + \beta_{duration} + \beta_{bathy} + \beta_{sst}$	none	2287.68	7493.70	3191.45
m1	idem	Constant: $f(s,t) = f(s)$	2217.82	7377.05	3066.31
m2	idem	Different each year: $f(s,t)$	2006.42	7040.09	2865.26
m2.1	$\beta_{tsppK} + \beta_{hookn} + \beta_{duration} + \beta_{bathy} + \beta_{sst}$	$f(s,t)$	1891.15	7033.88	2880.27
m2.2	hookn as offset, $\beta_{tsppK} + \beta_{duration} + \beta_{bathy} + \beta_{sst}$	$f(s,t)$	1887.18	7034.45	2876.76
m2.21	idem, $\beta_{duration} + \beta_{bathy} + \beta_{sst}$	$f(s,t)$	1884.96	7036.46	2879.24
m2.22	idem, $\beta_{bathy} + \beta_{sst}$	$f(s,t)$	1883.33	7052.81	2877.71

Table 3.2: List of bycatch-per-unit-of-effort (BPUE) models considered. Each model included a different linear predictor defined by the sets of covariates (z_k) and spatio-temporal random field structure (ξ). DIC values are presented for each species.

3.3.3 Prediction on the Total Fishing Effort Using the Logbook Data

For each species a final model was chosen for inference and subsequent prediction of total bycatch. For prediction, bathymetry and SST data were extracted for each logbook fishing set location. Other potential covariates, such as number of hooks or duration of the set were not available from the logbook data, and hence fixed at the median value computed from the observer data (number of hooks=1080 and soaktime=8.25 hours). The annual total bycatch per species, along with confidence intervals, was estimated by computing summary statistics on 1000 independent and identically distributed (i.i.d.) samples from the posterior distribution.

3.3.4 Hotspots and Excursion Maps

To visualize spatial bycatch patterns and hotspots, the final predictive model for each species was used to make predictions at the mesh node locations rather than at each logbook fishing locations. The rationale for this approach was that results for each fishing location are in point-form and hence, cannot be directly interpreted over the entire space. Simulation on the mesh nodes provides an efficient means with which to represent spatial patterns over space and time.

Again, fishing-related covariates were fixed at the observed median values to represent average fishing conditions. Bathymetry was extracted from the ETOPO1 raster layer and SST values were interpolated over time (year) from the observed and logbook datasets extracted using an inverse distance-weighted average. Interpolation was necessary to average the observed and logbook SST values which were originally extracted at the dd/mm/yy level. The distance was set at 50km: half the maximum size of the mesh triangulation. Different distance values were tested for sensitivity. We found this approach more suitable for representing the actual SST in the vicinity of the fishing activity over the 10 year period. Yearly maps of the expected total bycatch (mean linear predictor) and standard deviations were plotted overlayed with the Eastern Canadian coastline for reference.

We were further interested in finding areas where the expected bycatch exceeded a fixed level of bycatch ("an acceptable level"). In this instance, the mean and standard deviations of the linear predictor do not provide all information needed. As a first step, it is necessary to define what is an acceptable level of bycatch. In the absence of any standards, we evaluated different thresholds and calculated the probability of catching 2 times, 5 times,

and 10 times more sharks per fishing set as the total observed average number (for all years and the entire region). For shortfin mako (mean= 1.99 sharks/set) and porbeagle (mean= 1.60) sharks, the value were rounded and set at 4, 10, and 20 sharks per set and for blue shark (mean= 32.53), rounded values gave thresholds of 66 and 132 per set (the upper threshold of 330 sharks yielded too few observations).

To quantify these hotspots, we applied a newly established statistical method developed by (*Bolin and Lindgren*, 2014). The probability of exceeding the fixed thresholds at each location (mesh node) was estimated, and locations where the marginal probability was, say, at least $\alpha = 0.95$ identified locations of interest. However, these locations were not yet defining a hotspot; there is no clear joint statistical interpretation of these locations. To define a hotspot, we instead defined the *largest* area, such that with probability $\alpha = 0.95$, the threshold was exceeded at all locations in the area. This largest area was what we defined as hotspot. *Bolin and Lindgren* (2014) defined these areas in terms of distributional properties (based on the fitted models). Let ξ be a random field and u some level of interest (e.g., 10 sharks/set). A hotspot, or excursion set, is denoted by $E^{+}_{u,\alpha}(\xi)$, and is defined as the largest area such that with probability $1-\alpha$, the level u is exceeded at all locations in the area. The sets are indexed by the error probability α, and results can be easily mapped using the so-called excursion function:

$$F^{+}_{u}(\mathbf{s}) = 1 - inf\{\alpha; \mathbf{s} \in E^{+}_{u,\alpha}(\xi)\}. \tag{3.2}$$

The function takes values between zero and one, and each set $E^{+}_{u,\alpha}$ can be retrieved as the $1-\alpha$ excursion set of the function. For example $E^{+}_{u,0.1}$ is obtained as the locations where $F^{+}_{u}(\mathbf{s}) \geq 0.9$. The excursion functions were calculated using the Quantile Correction (QC) method, which is the most cost-effective (accuracy versus computational time) method for handling the non-Gaussian likelihoods used in our models. See *Bolin and Lindgren* (2014) for further details.

Since our models included a spatio-temporal structure (ξ), there are at least three types of hotspots map (excursion sets) that can be generated:

1. Individual hotspots map per time, in our case by year.

2. One summary map showing combined hotspots over the whole time-period i.e., combined largest areas where bycatch threshold was exceeded at least in one year.

3. One summary map showing only hotspots that were consistent over the whole time-period, i.e., largest area where the bycatch threshold exceeded every time.

All excursions sets were evaluated and results were projected from the mesh on a 4x4 km lattice grid.

3.4 Results

3.4.1 Model Selection

For all shark species, models that assumed a negative binomial likelihood performed best according to DIC and are shown in Table 3.2. Models that assumed a different realization of the random field every year ($m2$) performed much better than models without a random field (base model $m0$) or with just one field applied across the time serie ($m1$). Note that models with a finer temporal resolution (i.e. monthly versus yearly), or with other time-varying processes (e.g., autoregressive process over years), were also tested but did not provide a better fit according to DIC.

Of the available covariates in the observer data, target species (tspp), tonnage, and time of day (indicating the start time of the set) were found not significant. In contrast, bathymetry, SST, and the weight of the targeted species (tsppK) were significant for all shark species. In addition, set duration was significant for blue shark and the number of hooks (hookn) was significant for both blue and mako sharks. It is common practice when standardizing catch rates in pelagic longline fisheries to model effort, i.e. the number of hooks, as an offset. Effort is hence added to the outcome from the linear predictor as opposed to incorporated via an explanatory variable (*Maunder and Punt*, 2004). As seen in Table 3.2, including hooks as an offset performed better (lower DIC) for both porbeagle and mako sharks, but not for blue sharks, although the DIC difference between the two candidate models was small. Note that the DIC score for $m2$ for mako sharks was lower than for other models. However, tspp and tonnage had large standard deviation and non-significant β, which strongly indicates that the model was inadequate (results not shown).

Since one primary goal of this research was to predict total bycatch in the logbook data (using total fishing effort), it was important to consider the similarities between covariates available in both datasets. The *main* targeted catch is not defined in the logbook dataset.

The targeted species is inferred from catch composition, but this method assumes that the most common landed species is, in fact, the intended target (see Method section). Because of these issues and since tsppK was a weak predictor i.e., 95% credible region included 0 for all shark species, this covariate was omitted and tested in $m2.21$. Moreover, duration was only significant for blue shark species i.e., shorter soaktime were associated with more bycatch, hence we also tested $m2.22$ model which included only bathymetry and SST. This latter model provided a better fit for porbeagle shark and shortfin mako sharks compared to $m2.21$ model. Hence, we choose model $m2.21$ for blue shark and $m2.22$ for both porbeagle and shortfin mako sharks for model inference and prediction.

3.4.2 Inference and Prediction

Table 3.3 shows that for all three species of sharks, shallower bottom depth is associated with larger bycatch rates. This relationship is less pronounced for shortfin mako sharks and blue sharks as compared to porbeagle sharks. For both blue sharks and shortfin mako sharks, each additional 1km in depth decreased bycatch risk by 20% and 16%, respectively whereas for porbeagle sharks, it decreased expected bycatch by 63%. Each additional 1 degree in SST, lowered the expected bycatch rate of porbeagle and blue sharks by 16% and 13%, respectively, while shortfin mako shark bycatch was expected to increase by 22%. Soaktime had a mild effect on blue shark bycatch; each additional 1 hour of soaktime decrease the expected bycatch by 6%.

The posterior mean of the negative binomial overdispersion parameter (n) was 0.91, 1.99, and 1.45 for porbeagle, blue, and shortfin mako sharks respectively (Table 3.3). For all of these species, this parameter was significantly greater than zero, indicating that more bycatch is occurring than expected, i.e., bycatch is spatially aggregated. This clustering parameter was more pronounced for blue sharks and mako sharks than porbeagle sharks. The spatial correlation range, which defines the distance beyond which the spatial dependence between observations is unsubstantial, was similar for all species: for porbeagle sharks 156.86km with 95% credible interval [80.58; 239.85], blue sharks 170.78km [105.10; 240.20] and mako sharks 180.94km [107.54; 260.10]. Note that the size of the area where the fleet has been active is about 1700 km by 2350 km.

Figure 3.2 shows predicted versus observed bycatch for each species as derived from the best models of the observer data. In general, the models were able to predict with more accuracy small bycatch events but tended to underestimate rare catch events involving

Parameters	Mean	$Q_{0.025}$	$Q_{0.975}$
L.nasus			
Bathymetry (km)	-0.9895	-1.3030	-0.6976
SST (°C)	-0.1712	-0.2618	-0.0822
β_0	-4.0493	-5.7486	-2.3478
n	0.9139	0.6787	1.2136
P.glauca			
Bathymetry (km)	-0.2237	-0.3119	-0.1356
SST (°C)	-0.1422	-0.1775	-0.1070
Soaktime (hrs dec.min)	-0.0604	-0.0937	-0.0270
β_0	-0.9309	-1.5888	-0.2757
n	1.9953	1.7647	2.2483
I.oxyrinchus			
Bathymetry (km)	-0.1721	-0.3225	-0.0242
SST (°C)	0.2039	0.1390	0.2706
β_0	-10.2890	-11.5904	-9.0382
n	1.4887	1.1399	1.9195

Table 3.3: Posterior estimates, mean and 95% credibility interval, of model parameters. Note that bathymetry values were inverted (positive) in the models, such that a negative β signifies a decrease in expected bycatch with increasing depth.

very large numbers of sharks. This is expected since the negative binomial likelihood is a generalization of the Poisson distribution, which assumes a variance equal to the mean, such that larger means are associated with larger variability.

For predictions of total bycatch at all fishing locations (logbook data), the number of hooks (offset) was set at 1080 and soaktime at 8.25 hours; these median values were estimated from the observer dataset. Observed and total bycatch estimates and confidence intervals for the year 2003-2013 are shown in Figure 3.3. Total bycatch varied from year to year, but with substantial uncertainty, particularly for the expected bycatch of porbeagle sharks. It is apparent that blue shark bycatch has increased over the time period, whereas total bycatch appears to have remained fairly constant for porbeagle and shortfin mako sharks. The year 2012 is an outlier with respect to total bycatch estimates for mako sharks. This year was also associated with much higher observed bycatch. According to our estimates an average of 7377 [829;27653] porbeagle, 59819 [20266;124544] blue sharks, and 8060 [1907;35001] shortfin mako sharks were captured per year between 2003-2013. Based on the estimated weight:count ratio from the observer dataset, these numbers are equivalent to 174.5 t, 2445.4 t, 261.5 t of porbeagle, blue, and mako, respectively. From

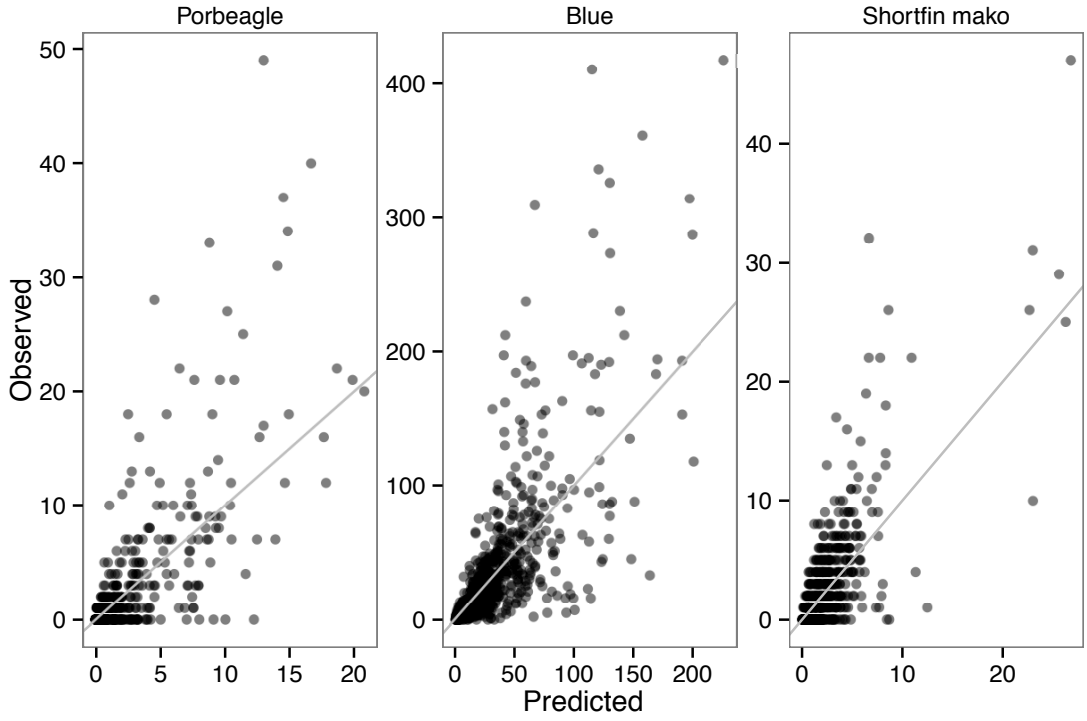


Figure 3.2: Goodness of fit for the at-sea observer dataset bycatch-per-unit-of-effort (BPUE) models. Observed versus predicted bycatch for the 3 main shark species captured in the Canadian pelagic longline fishery, 2003-2013

2003-2013, the fleet has landed on average 1327 t of directed species (swordfish and tunas) per year.

3.4.3 Hotspot Regions and Excursion Maps

Since our final models included SST and a random field component that was replicated in time (year), the mean and standard deviation of the expected bycatch varied from year-to-year, and consequently the hotspots (excursion sets). Yearly maps of the expected bycatch (mean linear predictor) are shown in Fig. 3.4. Mean expected bycatch values varied from 0.08 to 88.6 mako sharks per set, 1.3 to 218.0 blue sharks per set, and 0.001 to 65.7 porbeagle sharks per set. Areas of high bycatch rates were not constant over time and varied greatly for both blue and shortfin mako sharks. However, for porbeagle sharks, these areas showed a more uniform pattern over time; high bycatch rates were found on the Scotian shelf, south and south-east of Nova Scotia. Note that porbeagle bycatch was not observed in 2013 and as such, our model was not predicting any high values. For shortfin mako sharks, there was an apparent outlier in 2012, when a large area southeast of Nova

Scotia was associated with high bycatch rates for this species.

Patterns of variability were similar among the species and from year to year. Not surprisingly, the estimated standard deviation was smaller for areas with more information, such as along the shelf, as opposed to offshore areas that received considerably less observer coverage (Fig. 3.5).

For each species, yearly excursions maps showed similar hotspots patterns for all bycatch exceedence levels tested, but areas associated with 2 times more than the average shark bycatch were larger than those that had 5 times, or 10 times more bycatch than average (results available in supplemental material). Fig. 3.6 presents the summary of the combined hotspots over the whole time-period (combined largest areas where bycatch threshold was exceeded for at least one time) for a bycatch threshold set at 5 times the average i.e., 10 sharks per set for porbeagle and shortfin mako and 165 per set for blue sharks. For porbeagle shark, hotspots were associated with the Emerald Basin, particularly southeast of the northern basin near Emerald Bank. Another important area was Browns Bank, just north of the Northeast Channel and Georges Bank. These areas have been consistent over time, however hotspots north of these two main areas were only present in 2003, 2005, and 2010. For blue sharks, the main hotspot was concentrated on Emerald Bank, between the shelf break and east of Emerald Basin. This area is mainly identified due to high bycatch levels observed since 2009 in this area. For shortfin mako sharks, Browns Bank appeared to be also an important hotspot, most prominently in 2012, but also in other years. Hotspots were present in this region in 2007, 2009, and 2011-2013. Although some hotspots were visible in most years, no hotspots were completely consistent over the whole time-period.

3.5 Discussion

The goals of this study were to derive statistically sound estimates of annual shark bycatch for the Northwest Atlantic Canadian pelagic longline fishery, and to quantify the probability of exceeding different bycatch levels through space and time. Our results illustrate a powerful application of newly developed spatio-temporal modeling techniques, and derive well-defined spatial patterns, along with total bycatch estimates for the three main shark species captured in this fishery. We were able to show that bycatch tends to be spatially aggregated in a few hotspots, suggesting that spatial management could be a solution for reducing encounter rates and limiting total mortality for at least some shark species in

pelagic longline fisheries. Further simulation studies would be necessary to quantify the biological and socio-economic effect of applying spatial management; our approach could be extended to incorporate such factors. We further note that this method is not limited to the study of bycatch; it could be useful in any analysis of spatio-temporal patterns in fisheries or wildlife management situations and in informing decisions on size, location, and timing of spatial management interventions, such as closed areas.

Spatial and temporal variation in bycatch patterns arises from a complex and often poorly understood set of interactions between fishers, targeted and bycatch species, and the environment. Disregarding such variation can lead to the failure of spatial management measures. A well-known example is the New England multispecies sink gillnet fishery, which implemented closures to protect harbour porpoises (*Murray et al.*, 2001). Mortality of harbour porpoises was nearly 3 times the acceptable level allowed by the U.S. Marine Mammal Protection Act. In response, a large area (Mid-Coast) was closed during the month of November in 1994, when bycatch peaked. However, because of high seasonal variability, high levels of bycatch sometimes occurred earlier in the year and led to a failure in protecting harbour porpoises (*Murray et al.*, 2001). Temporally and spatially explicit modeling is needed to quantify bycatch patterns and their variability, however this is limited by statistical challenges and scarce information. The random fields included in our models account for all processes that may have had an effect on the bycatch pattern, but that are unknown (unobserved). Inter-annual variability in bycatch patterns was pronounced, necessitating the use of yearly random fields to appropriately capture the spatial processes influencing bycatch rates. The prediction maps in Fig. 3.4 illustrate the inherent complexity of managing bycatch over space and time. Models with even finer temporal resolution (monthly or weekly) would be better suited to inform fine-scale spatial management, but did not provide a better fit to the data, possibly due to limited temporal observer coverage. Fishing occurs from May to November, but is mostly concentrated during the months of July to September. The overall mean of the observer effort reflects this pattern, however there is large variability in the amount of observer coverage received each month. Scarce observer information also affects our uncertainty estimates in other ways, particularly in offshore areas, which hold most tuna-targeted sets (Fig. 3.5). Tuna-targeted sets received only about half the observer coverage compared with swordfish-targeted sets i.e., averaged 2.6% versus 5.7% . Additional observer data with finer stratification (by area, season,

target) would improve the ability to capture finer spatial/temporal variability, improve estimates and avoid any bias.

BPUE relationships with SST and bathymetry observed in our models were consistent with available knowledge of species biology and fishing practices. Both porbeagle and blue sharks are known to favor cooler temperate waters and are mostly associated with swordfish-targeted set, which are primarily fished later in the season in shallower and cooler waters (*Campana et al.*, 2011b; *Carruthers and Neis*, 2011). In contrast, shortfin mako sharks are more often captured in association with Gulf Stream waters (warmer and offshore waters), mostly when the fleet's effort concentrates on tunas, which occurs earlier in the season (*Campana et al.*, 2005, 2011b). SST as opposed to temperature at depth has previously been shown to be a poor predictor of porbeagle catch (*Campana and Joyce*, 2004). The apparent SST relationship here (i.e., cooler waters increase expected bycatch) is likely describing more the relationship of porbeagle bycatch with swordfish-targted sets. The main targeted species (swordfish versus tunas) was not a significant covariate and was omitted in our final models. Considering the discrepancies between the logbook and observer datasets (the main targeted species is identified from catch composition rather than being explicitly specified), others who work on this particular fleet suggested that it would be better to infer target species from fishing practices (*Carruthers et al.*, 2011), as done here.

Our average total bycatch estimates were approximately 2x, 3x, and 5x larger for blue, porbeagle, and mako sharks, respectively, compared to a previous estimate in this fishery *Campana et al.* (2011a). Although there are large uncertainties associated with these estimates, higher bycatch levels are concerning, considering the conservation status of some of these species. These three species of sharks are commonly captured in pelagic longline fisheries active in the Northwest Atlantic (*Cortés et al.*, 2009) and are managed by the International Commission for the Conservation of Atlantic Tunas (ICCAT). The Northwest Atlantic porbeagle population has been assessed as Endangered by the IUCN Red List of Threatened Species (2006). Recovery to a preliminary target of 20% of unfished biomass for female spawners, is estimated to take several decades, but conservation efforts could be jeopardized by excess bycatch and discard mortality, as well as unregulated fishing (*Campana et al.*, 2012). The North Atlantic stock of shortfin mako sharks is considered Vulnerable (IUCN, 2006). The species has high-value meat and fins and as such is often

landed as opposed to discarded at sea (*Campana et al.*, 2005). As a precautionary measure, the ICCAT Working Group recommended that fishing mortality should not be increased until more reliable stock assessments are available (*ICCAT*, 2012). Blue sharks, on the other hand, are the most common and most productive pelagic shark in the North Atlantic. ICCAT population assessments suggest that the stock is above MSY and that fishing mortality is probably sustainable, however, the data and models remain highly uncertain (*ICCAT*, 2009). These species have different management needs; investigating hotspots patterns at different exceedance levels may help in the development of species-specific conservation solutions. For example, for threatened species, resource managers could investigate lower bycatch thresholds than those for species of less concern. Bycatch limits could eventually be informed by robust reference points that determine an acceptable level of bycatch that might avoid negative population impacts (*Moore et al.*, 2013). This method can also easily be extended to identify time-area combinations shared by several species and to investigate cumulative fisheries impacts.

Hotspots of these three sharks were associated with shallower waters within the Canadian EEZ (Fig. 3.6). Porbeagle shark hotspots were consistently found on Browns Bank and Emerald Basin, a similar relationship was discussed by *Campana et al.* (2011a). Browns Bank and Emerald Bank were important areas for shortfin makos and blue sharks, respectively, however not consistently throughout time. These areas are all significant ecological areas of the Scotian Shelf; they are associated with high productivity, biodiversity and are important reproductive grounds (*DFO*, 2014a). For example, Browns bank is a spawning and nursery area for some commercially important species (e.g., lobster, scallop, and haddock; *Frank and Simon*, 1998; *Pezzack et al.*, 2001). Just south of Browns Bank, Georges Bank has been identified as a mating ground for porbeagle sharks (*Campana et al.*, 2012), which could explain, in part, the higher occurrence of the species in the area. Similarly, the Gulf Stream and adjacent waters have recently been proposed as a key winter feeding ground for several large pelagics, including blue sharks (*Campana et al.*, 2011b). Consistently high bycatch rates of porbeagle throughout the 10-year period suggest that spatial management may be more applicable for this species, whereas blue and mako sharks may require different approaches, for example near real-time spatial management (*Hobday et al.*, 2010) or the establishment of a fleet communication programme (*Gilman et al.*, 2006a). Nonetheless, higher bycatch rates of these species over time indicate that

these areas may be of some biological significance to these species and may be warrant further consideration.

Currently, in the Canadian pelagic longline fishery, shark bycatch landings are monitored and managed with quota allocations (*Cosandey-Godin and Worm*, 2010; *DFO*, 2013). In addition, the fleet practices live release, uses circle hook (mandatory since 2012), as well as de-hooking and line-cutting kits to release sharks (*DFO*, 2013) and reduce post-release mortality (*Cosandey-Godin et al.*, 2012). The fleet has also participated in experimental shark deterrent studies (*Cosandey-Godin et al.*, 2013) and is currently involved in post-release mortality studies. Despite efforts from longline fishermen to avoid shark catch (*Carruthers and Neis*, 2011) and mitigate discard mortality, no solutions are yet available to address the high level of bycatch in this fishery. Time/area closures can be simple and enforceable regulations, but can lead to unintended consequences (e.g., *Worm et al.*, 2003). Effective spatial management requires an understanding of the plausible consequences of fishing effort displacement, socio-economic impacts, and effects on targeted and other bycatch species. This paper does not address these questions, but our modeling approach could easily be extended to analyze these effects and simulate management scenarios. At present, our models cannot predict bycatch in the future (e.g., next year), since each year is currently treated as an independent replicate, and moreover, the kept weight of the main targeted species was a weak predictor and omitted in the final models. Any attempt to evaluate spatial closures in this fishery would have to include, at a minimum, the distribution of the target species, and would probably require more refined temporal analysis. The models were nonetheless suited to quantify total annual bycatch and overall hotspots pattern for the year 2003-2013. Using the true fishing effort values at each fishing location, as opposed to the median values, would have provided more accurate bycatch estimates, however this information was not available at the time of this research.

Fisheries scientists and managers are increasingly under pressure to meet diverse conservation goals while maximizing societal gains from commercial harvest. In order to meet these objectives, a suite of complementary management measures is required (*Worm et al.*, 2009); in many cases these will include fine-scale spatial-temporal management approaches, ideally evolving to near real-time fisheries management. Such elaborated management was once perceived too complex to be successfully implemented, but experiences off the east coast of Australia indicate that this is possible (*Hobday et al.*, 2010). Since

2003, dynamic spatial zoning has replaced the spatial static closure approach and been successful in protecting and managing a highly migratory species, the critically endangered southern bluefin tuna (*Thunnus maccoyii*) in the eastern Australia longline fishery (*Hobday et al.*, 2010). Nonetheless, this approach required a large amount of information from electronic tags to infer habitat preferences and fine scale oceanographic models to update management (*Hobday and Hartmann*, 2006). While this may not yet be feasible for many bycatch species around the world, our modeling approach provides a mean with which to infer complex spatio-temporal patterns from limited data, and offers a new way of defining hotspots.

The excursion method developed by *Bolin and Lindgren* (2014) and applied here to the definition of bycatch hotspots provides a novel way to explore and define spatial patterns that may have wider application in marine spatial planning. It is conceivable that managers are interested in finding areas where a particular process exceeds a certain threshold or acceptance level, or where it differs significantly from a baseline reference. A typical example comes from pollution studies, where scientists and managers may focus on areas where toxicity levels exceed a fixed threshold set by regulatory agencies. Similar approaches may apply to marine ecosystem management, which includes for example the effects of fisheries, shipping, pollution, and their cumulative impacts on marine biodiversity. Methods such as those represented here may become more widely used as management approaches become more refined in space and time.

Acknowledgements

We gratefully acknowledge the help of colleagues at the Canadian Department of Fisheries and Oceans (DFO), especially Peter Comeau who supplied the observer dataset. Many thanks to Toshio Michael Chin of the Jet Propulsion Laboratory for supplying the MUR-SST data. We are greatly indebted to Havard Rue for hosting A.C.G. and for generous advice and suggestions throughout this project. This research was supported by the Natural Sciences and Engineering Research Council of Canada (NSERC) through an Industrial Postgraduate Scholarship to A.C.G., and co-funded by WWF-Canada, NSERC Discovery Grants to J.M.F. and B.W. and a Canada Strategic Network Grant (NETGP 375118-08) to J.M.F. via the Ocean Tracking Network. In addition, E.T.K. received funds from the Research Council of Norway.

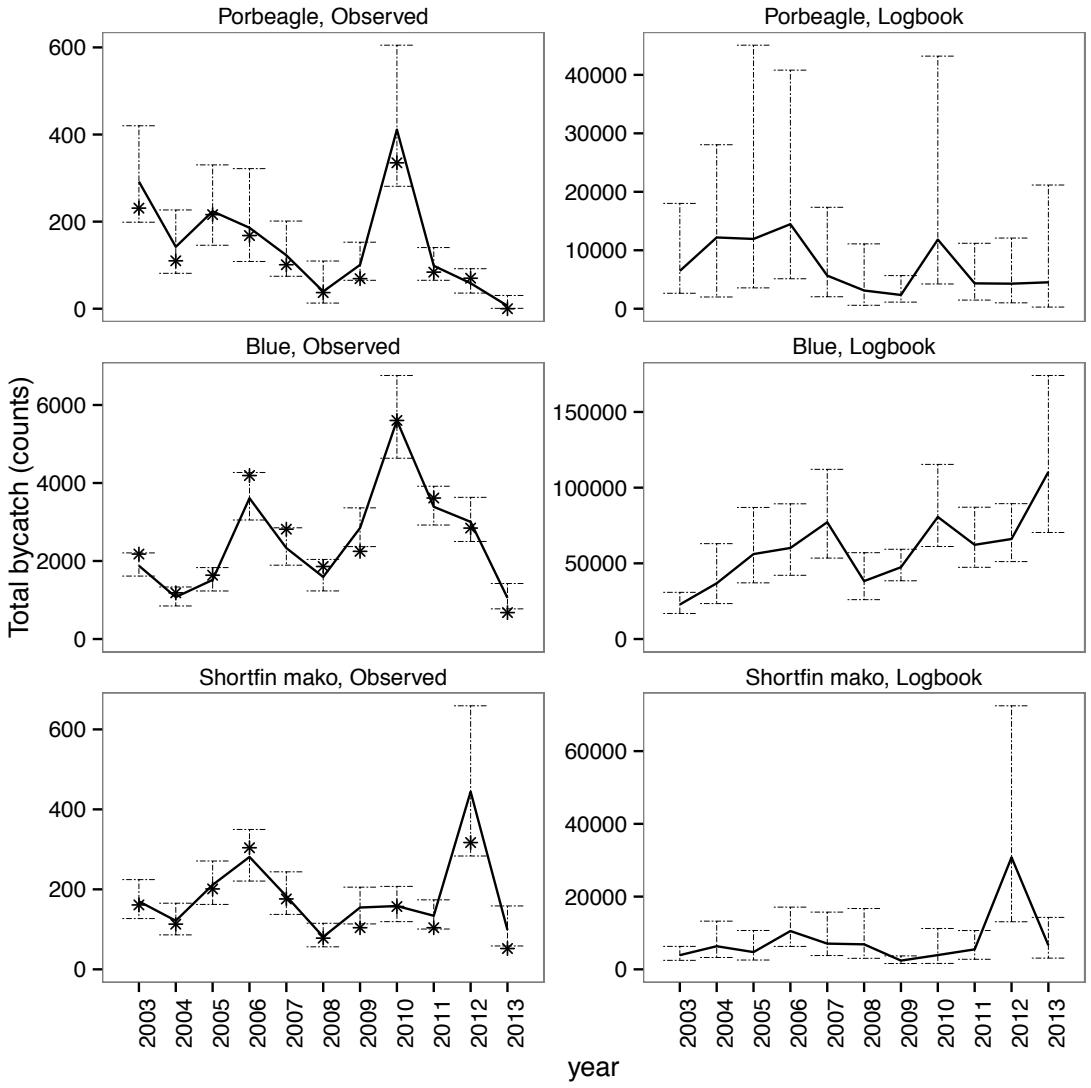


Figure 3.3: Observed and total expected bycatch and confidence intervals obtained from resampling for the 3 main shark species captured, 2003-2013. The asterisk represent observed values.

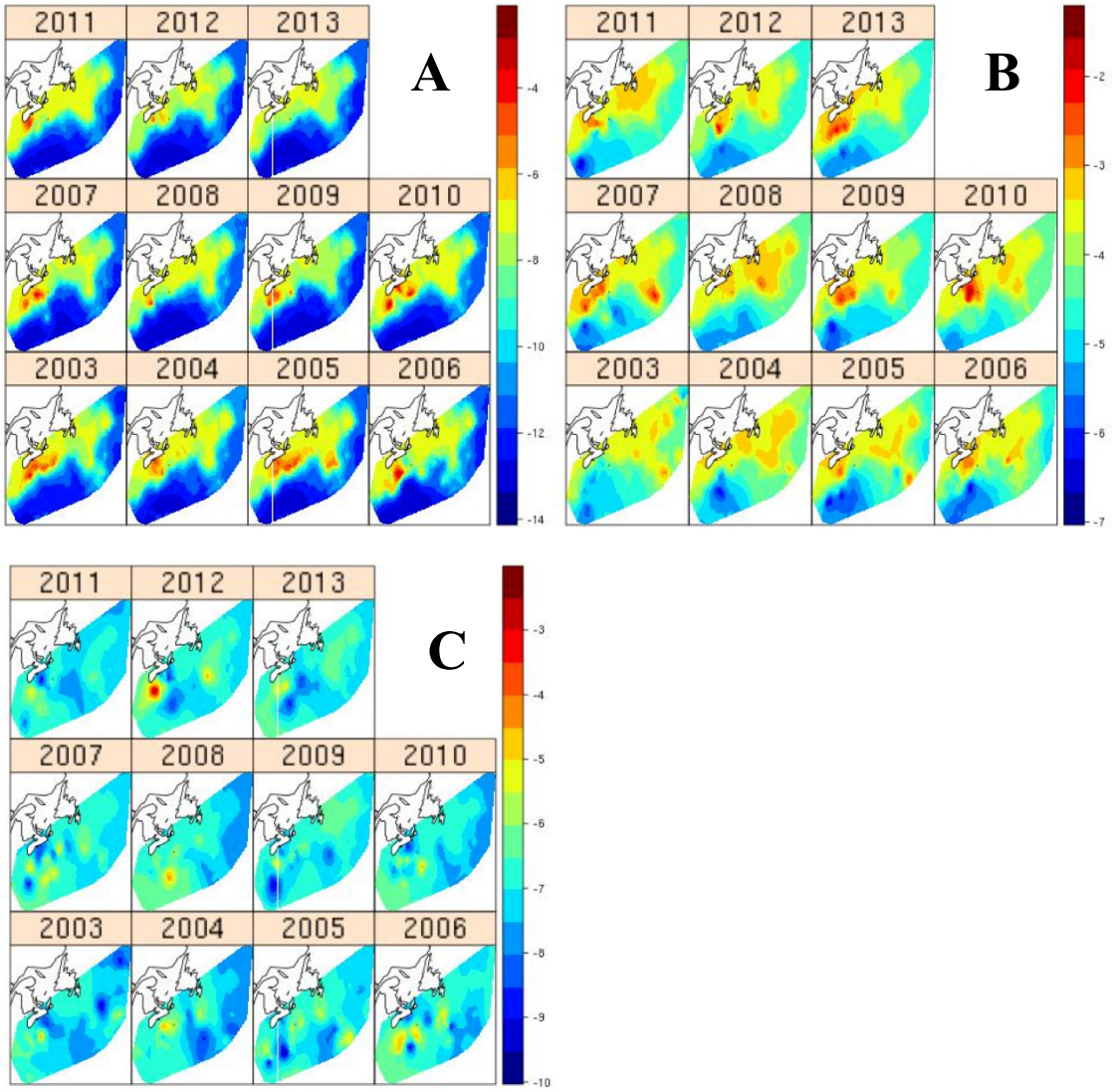


Figure 3.4: Annual maps showing the expected shark bycatch on a log scale for porbeagle shark (A), blue shark (B), and shortfin mako (C). Fishing covariates were fixed at the median observed value. The number of hooks was set at 1080 and soaktime (included only in the blue shark model) at 8.25 hours.

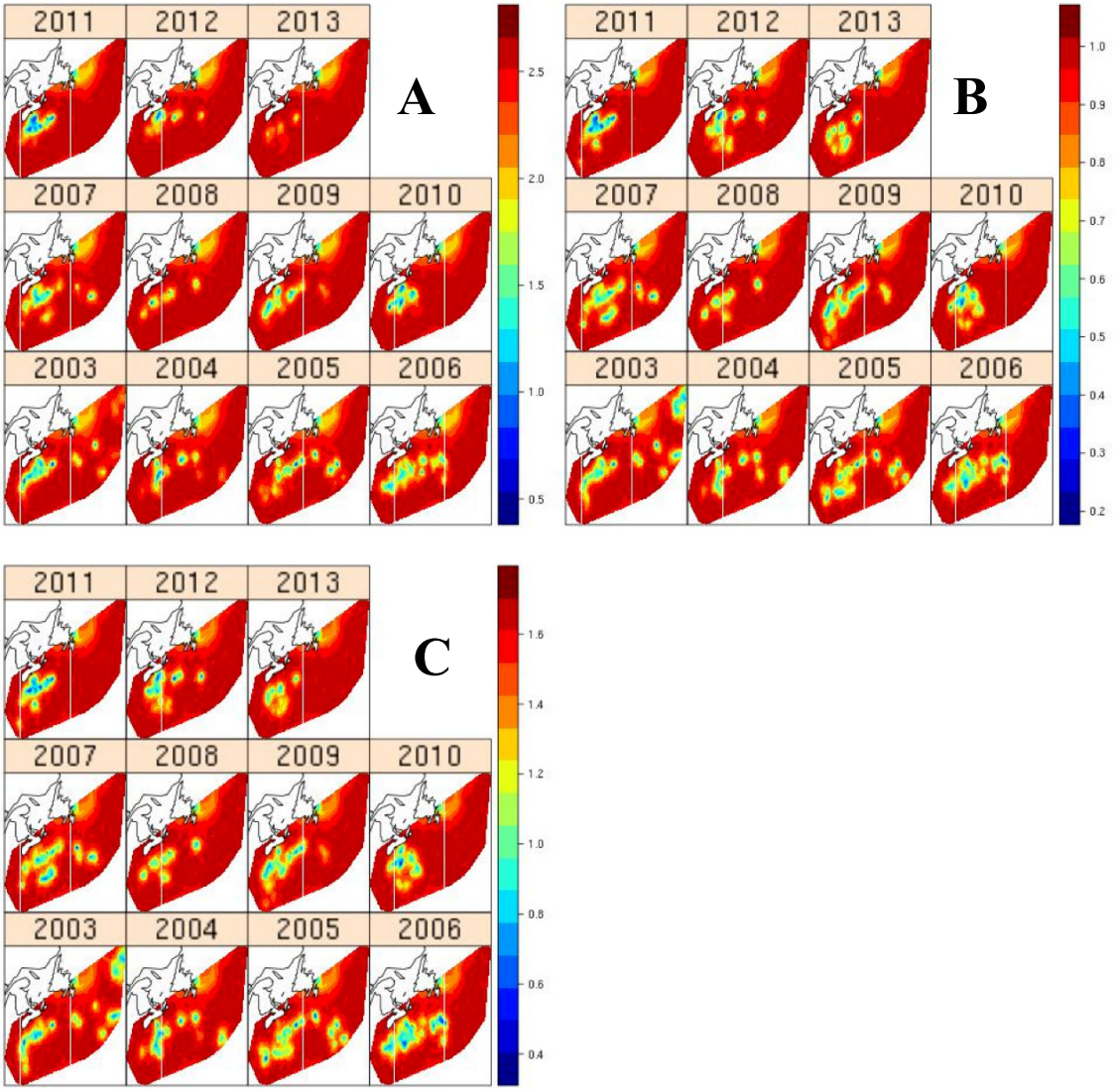


Figure 3.5: Yearly maps showing the standard deviation of the linear predictor for porbeagle shark (A), blue shark (B), and shortfin mako shark (C).

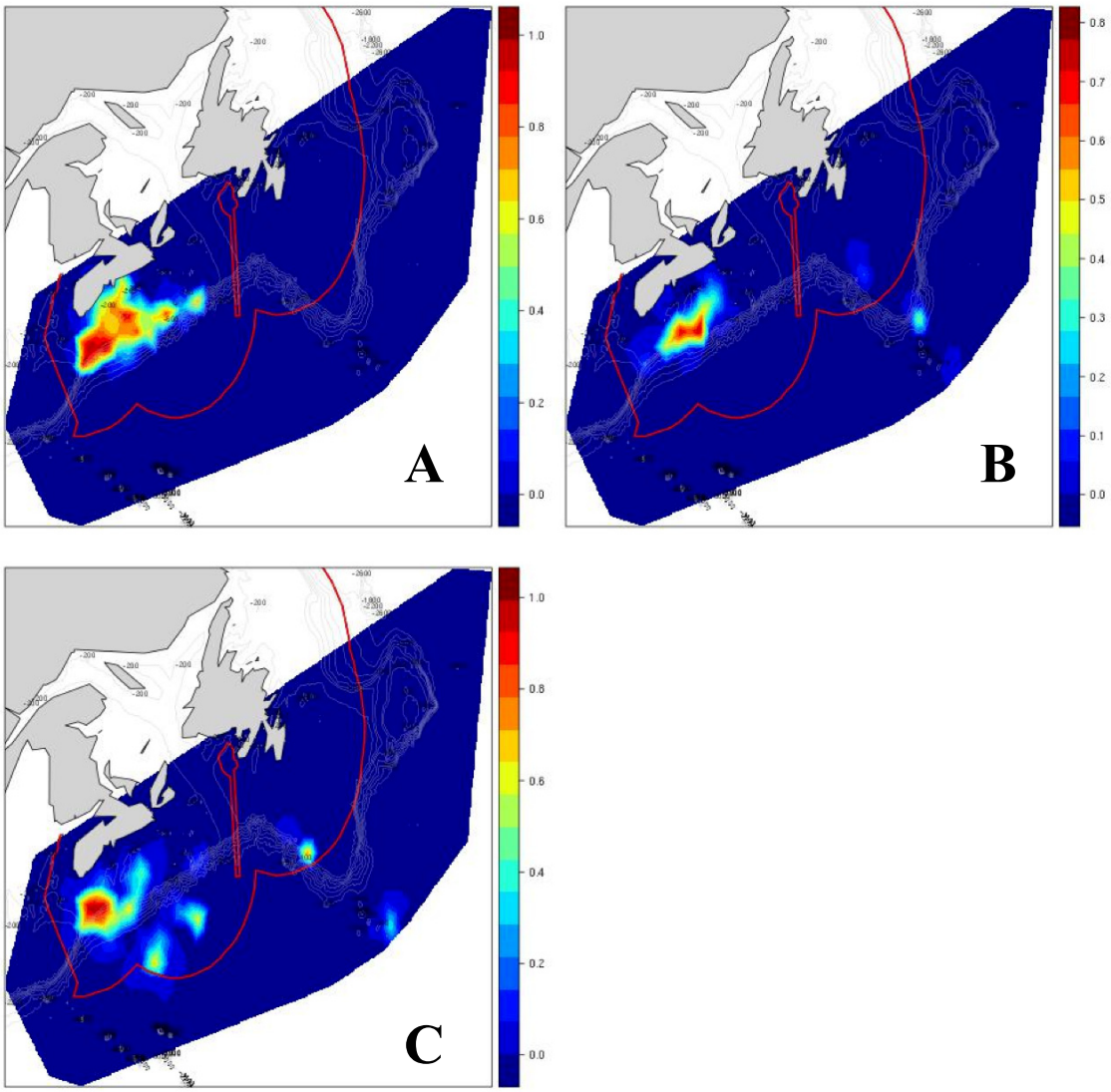


Figure 3.6: Excursions functions for catching 5 times the average number of sharks per fishing set in the Canadian pelagic longline fishery, 2003-2013. A) Porbeagle shark (Lamna nasus), 10 sharks/set B) Blue shark (Prionace glauca), 165 sharks/set C) Shortfin mako (Isurus Oxyrinchus), 10 sharks/set. The red line indicates the 200 nautical miles Canadian Exclusive Economic Zone and isobaths were plotted for reference.

CHAPTER 4

ELASMOBRANCH DISCARDS IN THE CANADIAN NORTHWEST ATLANTIC AND ARCTIC ADJACENT SEAS: COMPOSITION AND BIOGEOGRAPHY

4.1 Abstract

Overfishing is the primary threat to elasmobranch fishes (sharks, skates and rays) around the world. Management of these species is insufficient in most regions, and as a result many populations have declined to a fraction of their historical biomass and geographical range. While quota systems may restrict landings of some species, discard mortality is often overlooked and poorly quantified. This has limited our ability to set conservation priorities and mitigation options. To address these gaps, we utilized novel Bayesian geostatistical models to simultaneously analyze time-series of both the probability of discards (presence/absence) and the expected weight of discards (catch rates). These models were fitted to 17 years (1996-2012) of Canadian at-sea observer data from the Northwest Atlantic and adjacent Arctic seas. By also considering total fishing effort (as obtained from Canadian commercial data - including fishermen logbooks and dockside monitoring) we further quantified total discards by fishery sector so as to ultimately identify discard hotspots. The majority of elsmobranch discards occurred in southern Canadian waters bordering on those of the U.S. and were concentrated in a few hotspots, notably Georges Bank, Browns Bank, and the upper Bay of Fundy. These patterns correlate with both species abundance and fishing effort. In 2012, total discards were estimated

at 3250 mt (2722-3849, 95% credible intervals) of selachii and 1772 mt (1642-1911) of batoids. Deep-water species represented 3-5% of total elasmobranch discards and were primarily concentrated at more northern latitudes, notably the tail of the Grand Banks — in international waters, Davis Strait and Baffin Bay. Frequency maps illustrate that species-specific identification is particularly problematic in these regions, suggesting that discard levels of deep-water species are likely underestimated. The geostatistical models applied in this study are a powerful means with which to analyze at-sea observer data in order to offer proxie estimates of species distribution and abundance. This may have wide application to species that are not readily sampled by traditional scientific surveys, for instance pelagic and deep-water species.

4.2 Introduction

Elasmobranchs — the sharks, skates, and rays are a relatively small class of cartilaginous fish ($\sim$1150 species) that have, in less than 50 years, become the most threatened group of marine vertebrates (*Dulvy et al.*, 2014). Extinction risk is strongly related to the species' life-history traits (*Hutchings et al.*, 2012). As a group, elasmobranchs have low fecundity, slow growth rates, late sexual maturity and long reproductive cycles, which makes them more vulnerable to overexploitation, even when incidentally captured at low levels (*Hoenig and Gruber*, 1990; *Smith et al.*, 1998; *Frisk et al.*, 2001). Vulnerability increases with depth, such that deep-water species, about half of known elasmobranch fish, are even more vulnerable than their shallower congeners (*Garcia et al.*, 2008). Today, the estimated annual global average exploitation rate of many shark species exceeds average rebound rates by 1.5 to 3% (*Worm et al.*, 2013), which explains the well-documented and global widespread decline of many large sharks (*Baum and Worm*, 2009; *Ferretti et al.*, 2010; *Baum et al.*, 2003; *Ferretti et al.*, 2008). Targeted fisheries as well as indirect fishing — namely, bycatch and associated discarding are the primary threats to these fishes (*Dulvy et al.*, 2014). Globally, about 28 populations are regionally extirpated due to overfishing, a great number of these are from the skate family (*Rajidae*) (*Dulvy et al.*, 2000, 2003, 2014). Elasmobranchs represent a low proportion of global marine catch, generally have low commercial value (with the notable exception of fins), and as such, when not targeted, are often readily discarded (*Stevens et al.*, 2000). These characteristics explains why, in many countries, these species still lack adequate resources and appropriate management

attention (*FAO*, 2006).

In Atlantic Canada, an estimated 40 elasmobranch species have been reported (*Corke*, 2012). Thorny skate (*Amblyraja radiata*) is currently the only actively targeted species in Atlantic Canada, primarily from the Grand Banks of Newfoundland (*Simpson et al.*, 2012b). The majority of other species are discarded at sea and are largely unrestricted and unmanaged. In Canada as in other parts of the world, discard mortality is thought to be the primary threat to non-commercial cartilaginous fish. The impacts of discarding often remains overlooked until the population has fallen to low levels (*Brander*, 1981; *Casey and Myers*, 1998; *Dulvy et al.*, 2000). In Canada, discard estimates are available for individual species caught in some regions or fisheries. However, despite providing important details, information remained fragmented by departmental regions and a comprehensive understanding of discard levels across fisheries and over the entire basin of waters where Canadian fleets are active remained incomplete and unknown for deeper-water species. This paucity of information has limited the ability to set management and conservation priorities for many elasmobranch species.

The purpose of this research was meant to address these gaps and provide new information on discarded elasmobranch species in the Northwest Atlantic and Arctic Canadian waters, with a focus on deep-water species. The primary goals were to identify fine-scale areas of cumulative impacts of discards across fishing sectors for the main elasmobranch species discarded in Canadian waters. We used geostatistical Bayesian models which include time-serie and combined information of both the probability of discards (presence/absence) and the expected weight of discards (catch rates) to answers four main questions: (1) Where is bycatch highest by species and fishery sectors? (2) Where are there discards hotspots in Canadian waters? (4) How much total discards by species and fishery sectors? and (5) Where are the gaps in existing data?

4.3 Materials and Methods

4.3.1 Data sources

A wide range of commercial fishing activities are conducted in the waters off Atlantic Canada. These activities are licenced by Fisheries and Oceans Canada (DFO) and regulations and monitoring varies for each fishing regulatory regime which are generally

specified by species sought, type of gear, and management area (Northwest Atlantic Fisheries Organization, NAFO, fisheries statistical unit areas and divisions) (*Halliday and Pinhorn*, 1962).

At-sea observer data were provided by each DFO Regional Office (Gulf, Quebec, Scotia-Fundy, and Newfoundland and Labrador), for the year 1996-2012. The commercial fisheries dataset which includes information retrieved from both fishermen logbooks and dockside monitoring was provided by DFO statistical branch in Ottawa, again for the year 1996-2012. The commercial dataset contains information on area fished, kept portions of targeted species and other species landed (may include elasmobranch species), fishing effort, and vessel and gear type specifications. This data set provides the most complete census of the fishing activities in Canadian waters and was used to estimate total discards. However, discarded catches are not recorded. The most comprehensive information on discards is obtained from at-sea observer monitoring programme. The Atlantic Canadian International Observer Program was introduced in 1978, shortly after the extension of the Canadian 200 nautical miles exclusive economic zone (EEZ) to monitor international vessels fishing in Atlantic Canadian waters, but was later expanded to include the coverage of the Canadian fleet (for a detailed summary, see *Kulka and Waldron* 1983). The program now covers a portion of most fisheries, including invertebrates, groundfish, small and large pelagic and is administered on an annual basis by each DFO region. Trained sea-going observers collect a broad range of data that include: estimated weight of species captured, both retained and discarded, detailed fishing locations, and effort and gear specifications for every fishing set/haul. Annual observer coverage, most commonly expressed as a percentage of sea days fished, ranged widely across fisheries from less than 1% for many pot and trap fisheries to $> 80\%$ for shrimp trawl fisheries.

4.3.2 Data preparation

The sampling unit used in these analyses is a fishing set or haul, hereafter referred to simply as a *set*. Characteristics of each set were recorded in both the observer and commercial datasets, however these datasets did not include a field indicating the fishery. For both datasets, fishing set were therefore stratified into fleet sectors with similar characteristics based on the species sought and type of fishing gear used — this is commonly done in such large-scale analysis (e.g., *Benoît and Allard*, 2009; *Gavaris et al.*, 2010; *Campana et al.*, 2011a). Several types of fixed and mobile gear are used in Canadian waters. In

total, 55 observed fishing gear configurations were categorized into one of 11 major categories: *Trawlb* — include all types of bottom otter side or stern trawl, twin or triple trawl, shrimp trawl and Danish/Scottish seine (which behave like bottom trawls), *Trawlm* — all midwater trawl, *Gillnet* — set or fixed, *Line* — bottom, drift, or fixed longline, *Hook* — hand-bait and troll line, *Jigger*, *Dredge* — scallop, clam, or unspecified dredge, *Seine* — purse, *Trap* — all covered or uncovered trap or pot, *Harpoon* — mechanical and electrical, and *Diver*. Species sought (main targeted species) were categorized into 7 broader categories: *Groundfish* (n=32 species), *Large pelagic* (n=7), *Bivalve* — bivalve unspecified, scallop spp. and quahog, *Small pelagic* (n=3) — Atlantic mackerel, herring, and capelin, *Crab/Lobster* (n=7), *Invertebrate* ("Invert", n=3) — sea urchins, squid, and seacucumber spp., and *Shrimp* (n=2) — *Pandalus borealis* and *P. montagui*. Hence, in total, the observer dataset had 26 individual fleet sectors e.g., Groundfish-Line, groundfish trawls, Large pelagic-Line, etc. See Appendix C for details species and gear included under each category. Note that several additional fishery sectors were available from the commercial dataset, but did not received any observer coverage.

All entries with missing information on species sought or fishing gear were removed if the missing information could not be retrieved from other similar records (e.g., from the same fishing trip). Similarly, all incomplete and/or obvious errors in the geolocation — latitude and longitude of the fishing sets were removed from the dataset (less than 10% of the data). Complete geolocation was necessary for the spatial analysis used in this research project. To further investigate overall patterns, all shark and unidentified shark species in the observer dataset were summed under the order *Selachii* and all skate and rays, and unidentified skate and rays species were summed under the superorder *Batoid*. After data cleaning, a total of 580,111 fishing sets were available from the observer dataset, with an annual average of around 34,000 sets. Selachii discards were recorded in 37,109 sets (6.4%) whereas batoid discards were observed on 221,743 sets (38.2%). Over 3 millions fishing sets were available from the commercial dataset.

4.3.3 Analysis

For each of the main species captured as well as for the overall selachii and batoid order and superorder, two-parts time-serie Bayesian geostatistical models were built to investigate the effect of fleet sector, bathymetry, temporal trends, and spatial patterns on 1) the probability of discards (presence/absence) and 2) the expected weight of discards (catch rates). All

analyses were performed using the R-INLA package (r-inla.org) (*Rue et al.*, 2009). Models were designed with the aim to compare results among species as well as to synthesize information over time and space. Best models were selected based on deviance information criterion (DIC).

The response (y) was stratified by fleet sector, time (year-month) and summarized by tiles (details below). For the presence/absence of discards, y — the number of observed fishing units with discards over the total observed fishing units were assumed to follow a binomial distribution: $Prob(y) = \binom{n}{y} p^n (1-p)^{n-y}$. Hence, the mean and variance of y are therefore given as $\mu = np$ and $\sigma^2 = np(1-p)$. The probability of p is linked to the structured additive predictor (η) by $p(\eta) = \frac{exp(\eta)}{1+exp(\eta)}$.

For the weight of discards, y — the mean weight, was assumed to follow a gamma-distribution with density $\pi(y) = \frac{b^a}{\Gamma(a)} y^{a-1} exp(-by)$, where $a > 0$ is the shape parameter and $b > 0$ is the inverse scale parameter. The mean and variance is hence given by $E(y) = \mu = a/b$ and $Var(y) = 1/\tau = a/b^2$, where τ is the precision parameter and μ the mean. The response was linked to the linear predictor by a log-link $\mu = exp(\eta)$.

For both models, the structured additive predictor (η) was of the following form:

$$\eta = \beta_{fleet} + \beta_{bathy} + f(time, model = rw1) + f(poly, model = spde). \tag{4.1}$$

Where β_{fleet} are the fixed effect for each fleet sector included, β_{bathy} is the mean of the bathymetry, $f(time, model = rw1)$ is a one-dimensional random walk function describing the temporal trend expressed as year-month, and $f(poly, model = spde)$ is a Matérn covariance function which accounts for the spatial autocorrelation and is included in the model via the SPDE (stochastic partial differential equations) approach described in *Cosandey-Godin et al.* (2014). Species sought weight landed (a proxy of fishing effort) is often a common choice for scaling observed discard ratio to total fishing effort. We tested several models which included the later, but, at the level of this analysis, this variable was poorly correlated with the presence/absence and weight of elasmobranch discards and consequently was omitted from the final models. Fishing effort was accounted for by the number of fishing sets.

To maximize computational time and since a large percentage of the data were zero observations (no discards), the data was summarized using Dirichlet tessellations — a

form of partitioning (*Okabe et al.*, 2000). This method was chosen because it is compatible with the SPDE approach and build on the strength of the latter. A first step in the SPDE approach is to create a triangulated spatial grid or 'mesh' which defines the spatial domain of interest which is then used in the models to estimate the spatial random field (*Cosandey-Godin et al.*, 2014). A mesh was designed for each species and customized according to the locations of discards (using latitude/longitude). In doing so, we created meshes with finer triangulation where discards occurred. This design allows to better capture the local properties of the process (spatial autocorrelation) as well as minimize computational time. The set of mesh nodes were then used to compute the Dirichlet tessellation (i.e., around each node) and, the number of occurrences (presence/absence model) or the weight of discards (weight model), total number of fishing units and bathymetry were summarized by fleet and time (defined by year-month) for each tile. Bathymetry was extracted from the National Oceanic and Atmospheric Administration (NOAA) global relief model ETOPO1 (*Amante and Eakins*, 2014). An example of a mesh and Dirichlet tessellation is presented in Appendix C.

For each species or group of species, only the fisheries that explained more than 4.5% of the discard occurrence and at least 1% of the weight were included in the analysis. In general, this selection resulted in explaining more than 95% of the observed discards per species. Consequently, different fisheries are included in each species model (see Fig. 4.2). Species-misidentification is a known issue in the observer dataset (e.g., *Benoît*, 2006; *Gavaris et al.*, 2010). Based on the distribution and published knowledge about the biology of the species, discard occurrences in northern latitudes i.e., North of Newfoundland and Labrador were considered highly unlikely. Hence, to minimize model biases all discard records north of latitude 52°N were treated as zeros for porbeagle (*Campana and Joyce*, 2004), shortfin mako (*Campana et al.*, 2005), blue shark (*Campana et al.*, 2011b), basking shark (*Campana et al.*, 2008), winter skate (*Simon et al.*, 2003) and Barndoor skate (*Simon et al.*, 2009). These occurrences of elasmobranch species were nonetheless included and analyzed under the broader category of selachii and batoid.

4.3.4 Estimates of Discards on Total Fishing Effort

For each of the main species captured as well as for the overall selachii and batoid order and superorder, we further quantified total discards and identify areas of higher discards. To do so, the commercial data was first summarized as y using the same species-specific

mesh and tessellation used in the modeling of observer data.

Each model (binomial and gamma) was fitted using the INLA approach, and 1000 independent distributed samples were drawn from the joint posterior distribution of all parameters in these models to estimate the probability of: 1) presence/absence of discards (*probs*) and 2) mean discarded weight ($weight_{mean}$). For each sample, total estimates of discarded weight were then computed by: $probs * weight_{mean} * logbook_n$, where $logbook_n$ is the total number of fishing units recorded in the commercial dataset per fleet, time (year-month), and tiles. These samples were further summarized by mean, quantiles, or standard deviations, according to the level of interest. Final results were presented by fleet, time, and over space.

The commercial dataset was corrected for obvious discrepancies where lower fishing effort was recorded compared to the observer dataset in the northern region i.e., Baffin Bay and Davis Strait (NAFO Div.0AB). This is a known issue for this DFO region; commercial data have often been incomplete, particularly in the 1999-mid 2000s. The region was newly fished and catch records needed to be compiled across DFO regional databases (M.Treble, personal communication, 10/29/2013). To rectify this problem, we added the observed fishing effort difference (by fleet sector, time, and tiles) to the commercial dataset. Since 2002, all directed Greenland turbot (*Reinhardtius hippoglossoides*) fishing in Baffin Bay has received 100% at-sea observer monitoring *DFO* (2014b). However, observer coverage is much lower in Davis Strait ($< 20\%$), hence total fishing effort might be underestimated for this particular region *DFO* (2014b).

4.4 Results

4.4.1 Descriptive analysis

4.4.1.1 Composition

A total of 44 species and groups of elasmobranch were observed during 1996-2012 (fishery observer records). For this period, the total weight observed was nearly identical for selachii and batoids: about 6,407 mt and 7,142 mt respectively (Figure 4.1). The great majority of the discards were attributed to a few species, namely, thorny skate, which comprised more than 50% of the batoid discards, followed by winter skate and unidentified skate species. For selachii, blue shark, Greenland shark, and spiny dogfish represented about 85% of the total observed discards (Fig. 4.1).

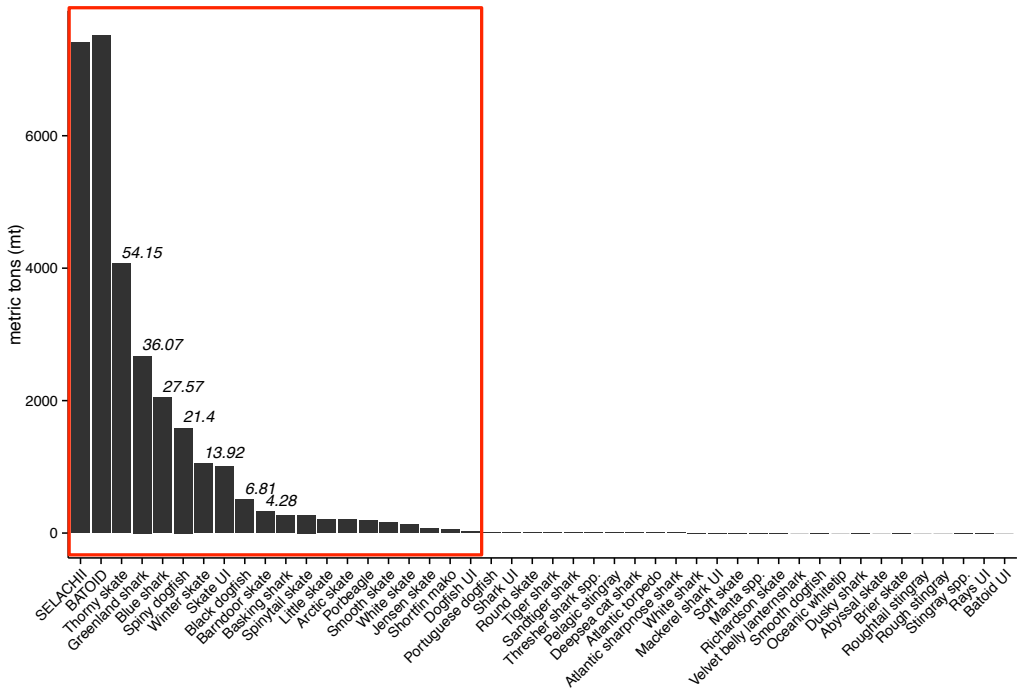


Figure 4.1: Total observed weight of elasmobranch discards in metric tons (mt) per species from 1996-2012. The numbers at the top of the histograms indicate the percentage that each species contributed to the overall order Selachii or superorder Batoid. The red box indicates the species or group of species that were analyzed. UI: unidentified.

4.4.1.2 Fleet effect

Figure 4.2 shows the effect of individual fleet sectors included in each species' models. Note that for visualization purposes, all β were plotted on the same figure, but values are model-specific. Of the 26 fleet sectors observed, few contributed to the bulk of elasmobranch discards. In general, between 3 and 4 sectors were responsible for the majority of the discards (Fig. 4.2). For selachii, large pelagic line i.e., swordfish and tunas pelagic longlines was, by far, the most common source of both the presence and weight of discards for porbeagle, shortfin mako, and blue sharks. Discards events occurred 7 times more often than for the groundfish sector and discarded weight tended to be nearly 35 times larger e.i., captured larger individuals or a larger number of individuals. The remaining selachii discards were attributed to the groundfish fishing sector (trawl, line and gillnet fisheries). Overall, bottom gillnets and lines tended to discard more frequently (35-37%) as opposed to trawl. Whereas, for weights, bottom trawl and line were associated with

higher weight than gillnets. For batoids, discards were largely attributed to the groundfish sectors as well as the bivalve dredge (scallop spp.) and shrimp trawl sectors. On average, groundfish trawl fisheries discarded 40% more frequently batoids than the other sectors. Differences existed among species, for example, winter, little and white skates — a rare species, were strongly associated with bivalve dredge. For weights, both bivalve dredges and groundfish trawls were discarding higher amount of skate species.

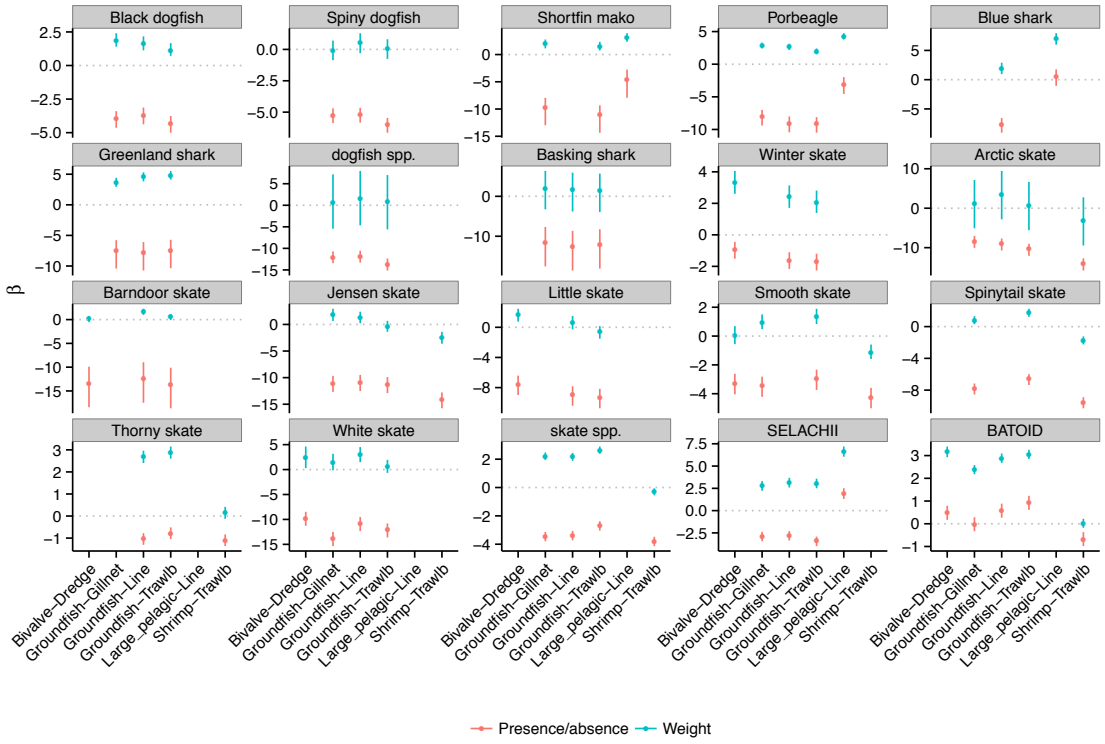


Figure 4.2: Summary effect and credible intervals for each fleet included in each species' model. Larger β signifies greater discards (occurrence or weight) for this species.

4.4.1.3 Depth effect

Figure 4.3 shows the effect of bathymetry for each species' models. The presence of a certain elasmobranch species discard was very well described by the preferred depth range of the latter. For example, black dogfish and Jensens skate are deep-water species mostly found on the continental slope (*Compagno et al.*, 2005). Both species have wide depth range up to $\sim$ 2500m, whereas spiny dogfish is a shelf species that is generally found in shallower waters than 200m (*Compagno et al.*, 2005). For other species, higher population concentrations are known to occur at mid-depth, for example smooth skates, which can

be found at depths of 25–1436m but prefer 70–480m depth range (*Kulka et al.*, 2006). For species that inhabit the water column (semi-pelagic, pelagic) as oppose to demersal species, the bathymetry relationships observed explain the depth-range targeted by the fleet sectors rather than the preferred depth of the species. For example, blue sharks are pelagic species that are found from the surface to about 350 m depth (*Compagno et al.*, 2005). Discarding of this species occurred mainly in the pelagic longline fisheries (77% of discards occurrence) which targets swordfish and tuna in the epipelagic zone on and off the continental shelf (see Chapter 3). Hence, a strong association with a fishery that was active offshore (as opposed to deep) explained the higher frequency of discards associated with deeper waters. The relationship between bathymetry and discard weight is not as clear, but for the majority of the species, deeper waters tended to be associated with higher weight of discards when bycatch occurred.

4.4.1.4 Temporal trends

For several species, the frequency of discards showed a clear increase over time, this is the case for shortfin mako, porbeagle, barndoor skate, and little skate, whereas, smaller albeit still apparent increase also exist for winter, Arctic, and Jensens skate, particularly since the mid-2000s. Decreasing frequency of discards was noticeable for spiny dogfish and for thorny skate from 1996-2002 and again since 2005. Similarly, basking shark discards decreased overall in occurrence since 2004 until 2012. Frequency of discards for all other species have remained relatively stable although with large annual and seasonal fluctuations (Fig. 4.4).

For the discarded weight, trends appeared to have remained more stable for the great majority of the species (Fig. 4.5). Nonetheless, porbeagle shark, barndoor and Jensens skate showed increasing trends, suggesting that when bycatch occurred, discards tended to be larger i.e, either more fish and/or larger individuals over time. This trend was the opposite for thorny skate until mid-2000s and for shortfin mako, prior to 2004, i.e. fewer fish or smaller-size fish were captured. No temporal trends were found for dogfish spp. and basking shark. Both models provided a lesser fit to the data compare to other species and may have failed to capture the presence of temporal trends.

4.4.1.5 Spatial patterns

Figures 4.6 and 4.7 show the summary effect of the spatial field included in the occurrence and weight models and identify areas of greater risk of 1) frequency of discard occurring

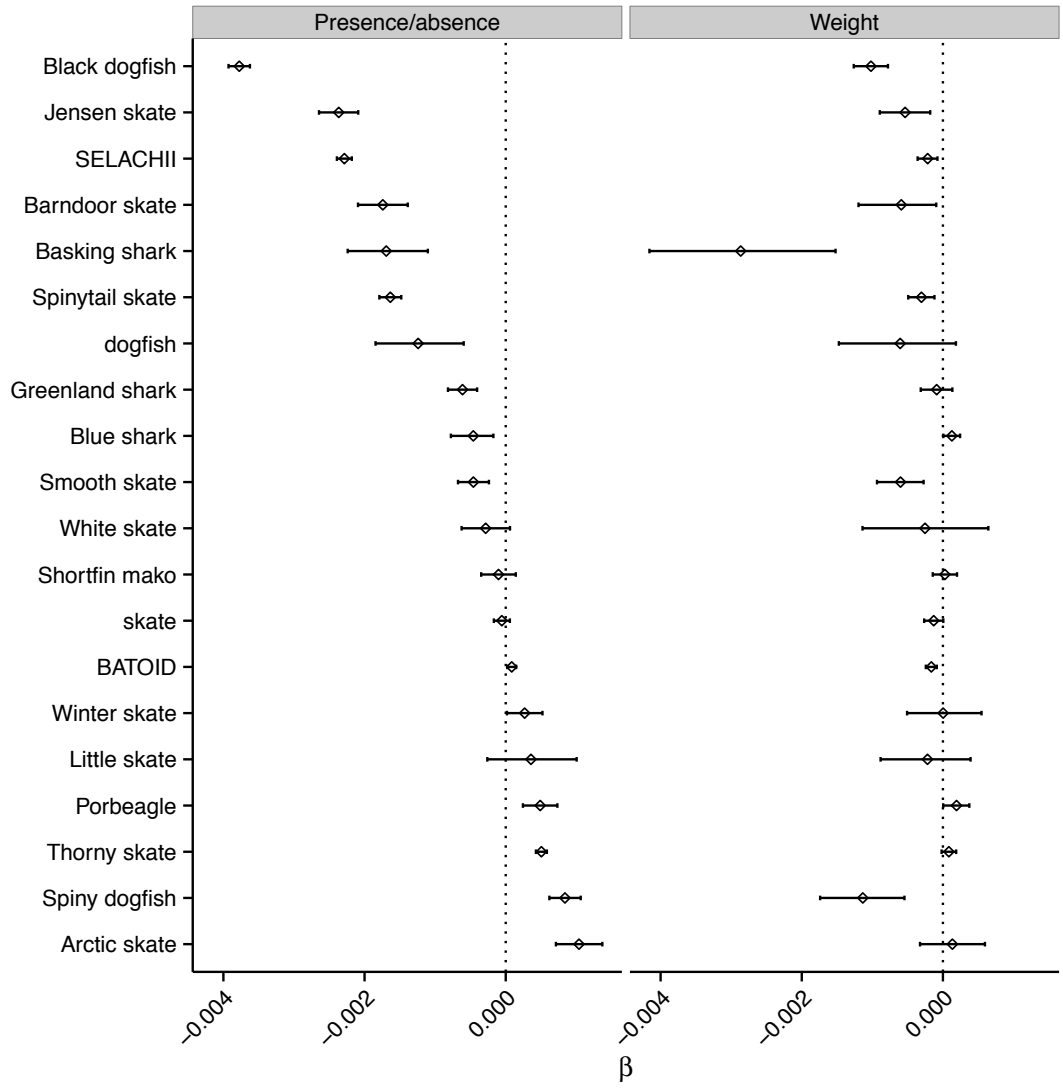


Figure 4.3: Summary effect and credible intervals of bathymetry. Negative β signifies greater discard occurrence in deeper waters.

and 2) weight of discards after accounting for fishing effort (number of fishing units), bathymetry, temporal and fleet effect. These maps are reasonable proxy of the species distribution and biomass concentration. Overall, areas associated with higher selachii discards were the Gulf of Maine, Bay of Fundy, Laurentian Channel, eastern Davis strait, and southwest Baffin bay (Selachii map in Fig. 4.6). For batoid, areas of high discarding probabilities were concentrated in the Gulf of St Lawrence, the tail of the Grand Banks and Baffin Bay. Larger amount of discards were concentrated in the Gulf of Maine, Hamilton Bank (north of Newfoundland), and Northwest of Baffin Bay for all sharks combined

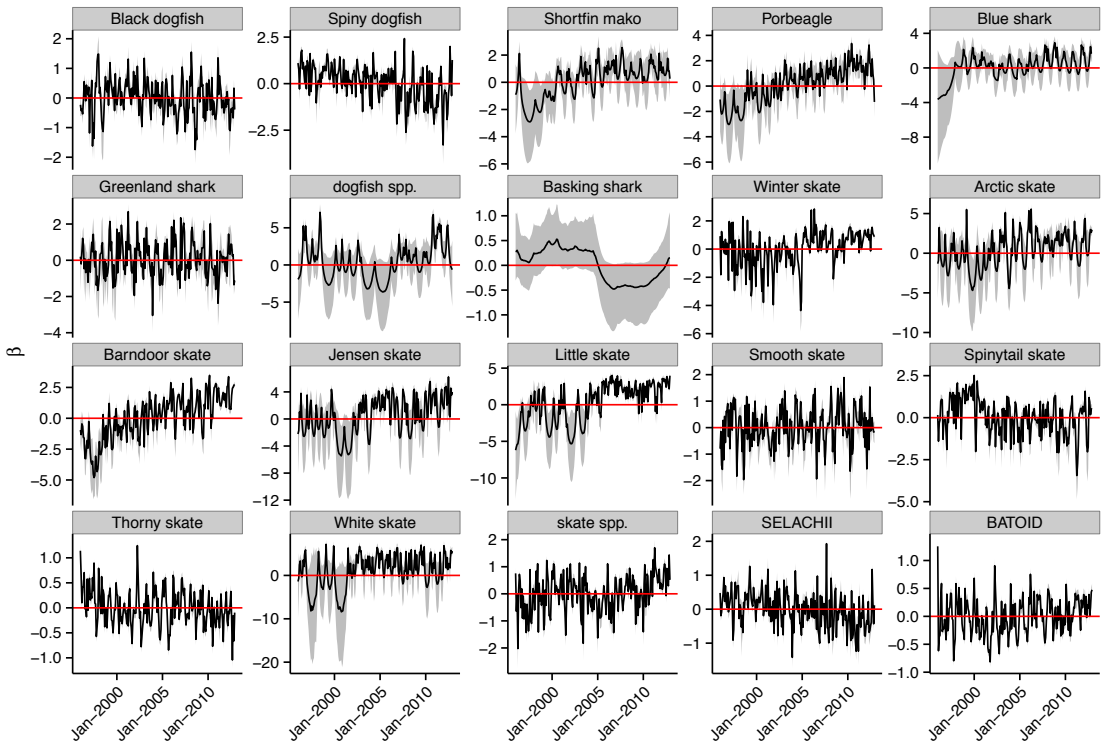


Figure 4.4: Temporal trends of elasmobranch discards in Canadian fisheries, 1996-2010. The figure shows the mean effect of time with the 95% credible intervals (grey ribbons).

whereas for skate species, these areas were Georges Bank, the tail of the Grand Bank, and all along northern Labrador up to south of Baffin Bay. Differences were species-species.

4.4.2 Predictions

Prediction performance of each model was evaluated based on their predictability for the observer data. Performance varied among species, but in general, the occurrence models overpredicted the frequency of discards by an order of $< 1-2\%$ for skate species and $< 1-8\%$ for selachii species. Fisheries associated with fewer discards were usually associated with greater errors due to the limited information available. The gamma models for weight were associated with greater relative errors and in general predictions were more often biased low. In general relative errors varied between $> 1-30\%$. Again fisheries associated with less amount of discards were associated with greater relative errors.

Results presented in this section are the final summary statistics (mean or median and 95% credible intervals) on the total fishing effort (commercial fisheries data). Total estimates of discards by fleet sector for both selachii and batoid from 1996-2012 are

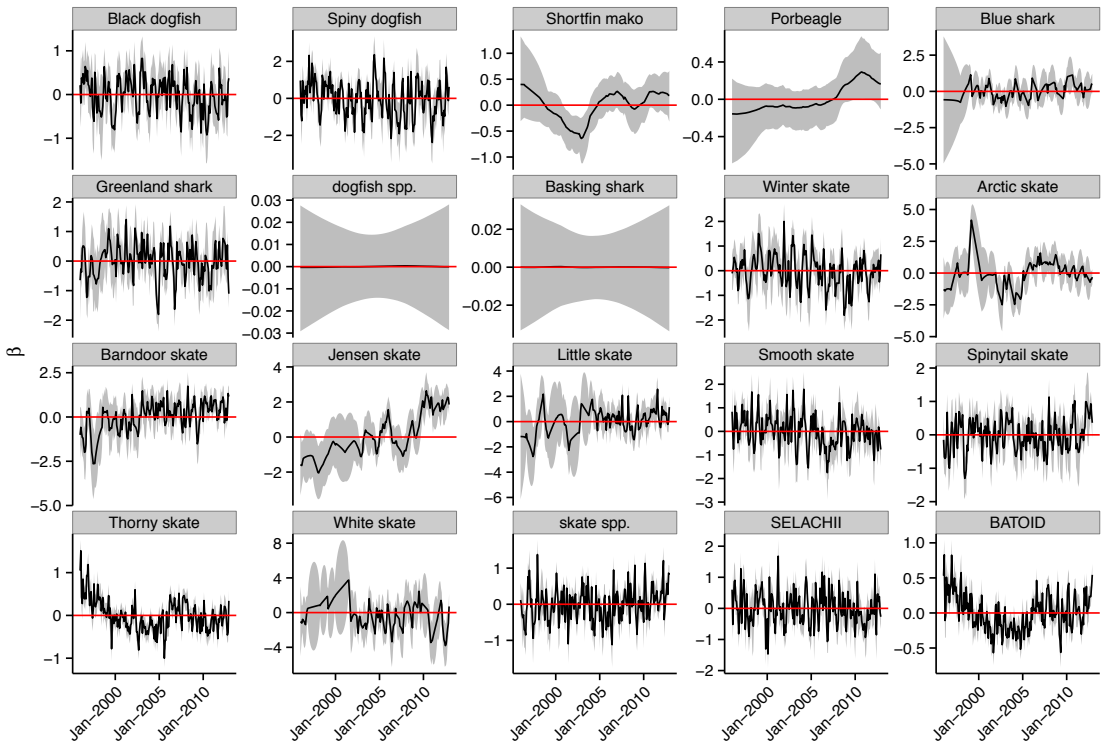


Figure 4.5: Temporal trends of elasmobranch discards weight in Canadian fisheries, 1996-2010. The figure shows the mean effect of time with the 95% credible intervals (grey ribbons).

presented in Fig. 4.8.

4.4.2.1 Temporal trends by fleet sectors

In 2012, annual total discards were estimated at 3250 mt (2722-3849, 95% credible intervals) of selachii and 1772 mt (1642-1911) of batoids. However, there has been large variations over time. Discards peaked for batoids in 2008, where total estimates were 2814 mt (2617-3010) whereas for selachii, discards were highest in 1998 with 4541 mt (3666-5616). For selachii, large pelagic longline fishery was associated with the largest discarded weight followed by the groundfish trawls sector. Total discards of selachii have decreased on average 10-15% in groundfish gillnet and bottom longline fisheries since 1996. For batoids, bivalve dredges and groundfish trawls contributed most of the discards (about 85%).

Figure 4.9 presents results by species. Species composition tended to be very much associated with the fishery sector. For example, over 90% of the discarded weight associated with bivalve dredges was comprised of winter and little skates whereas large pelagic

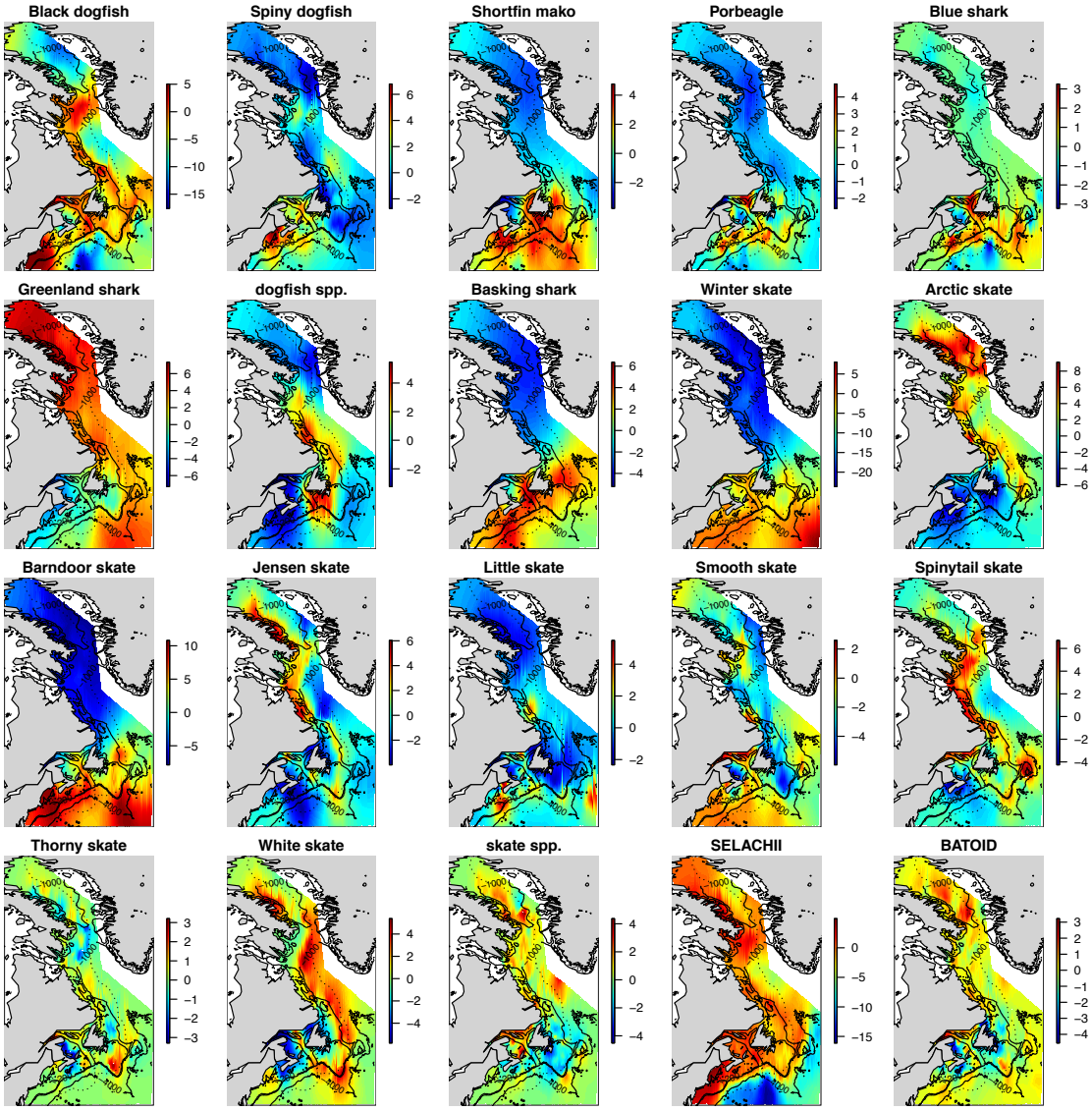


Figure 4.6: Maps of the summary spatial effect of each species' occurrence model. Red region shows areas with higher risks of discards. Note that the scale differs among species.

longline discarded almost exclusively shortfin mako, porbeagle, and blue sharks. The bulk of the groundfish bottom trawls sector discards was composed of spiny dogfish and thorny skate and to a lesser extend, Greenland shark, smooth and spinytail skate. Groundfish longliners and gilnetters tended to discard a majority of deeper water species such as black dogfish, Jensens, and spinytail skate as well as Arctic skate.

Higher discards levels occurred during the 4th quarter of the year (Sept-Dec) for most fleet sectors (fig. 4.10).

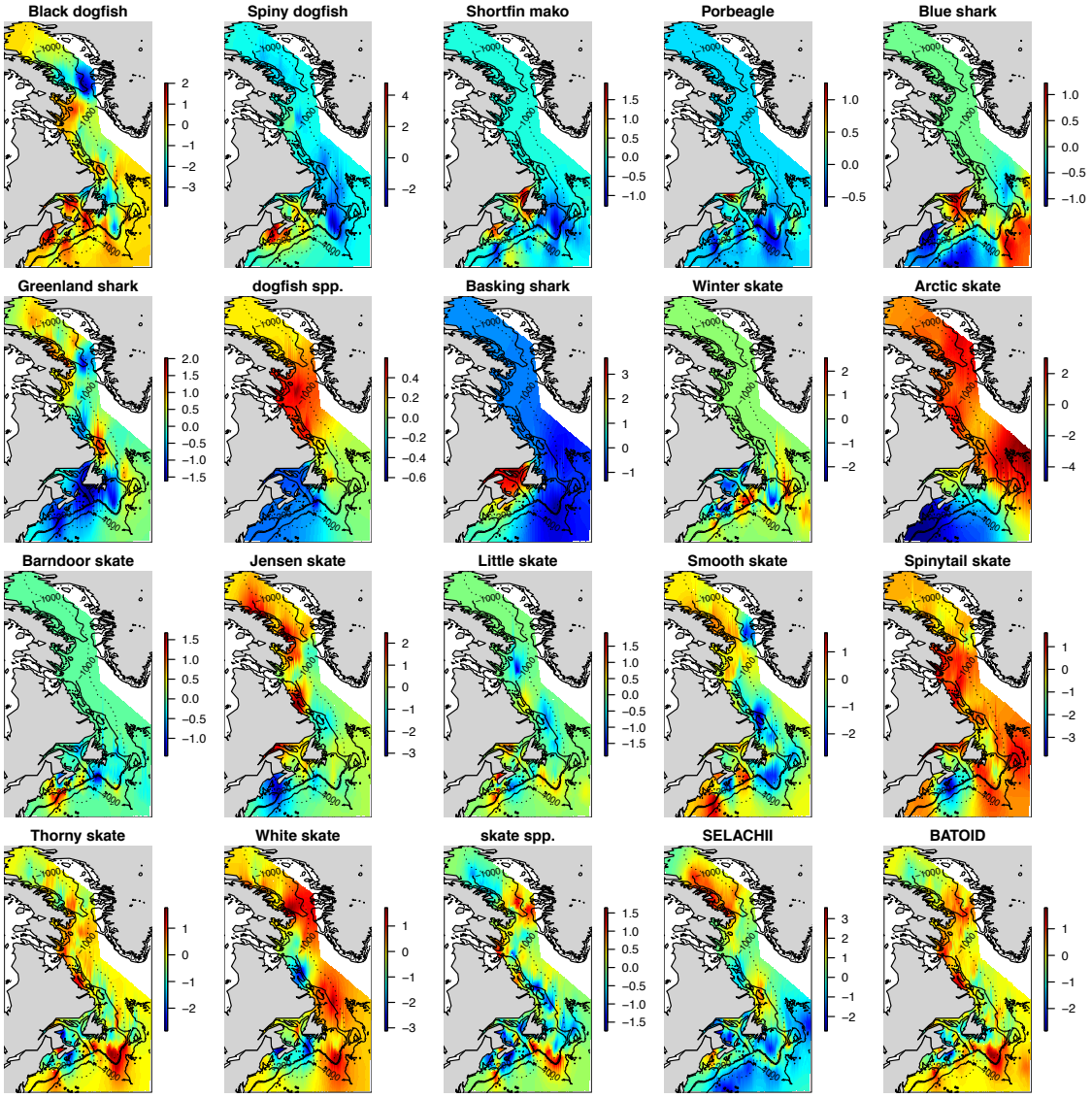


Figure 4.7: Maps of the summary spatial effect of each species' weight model. Red regions indicate areas with higher discard weights. Note that the scales differ among species. The basking shark model fitted more poorly; results are to be interpreted with caution.

4.4.2.2 Spatial patterns

The bulk of elasmobranch capture occurred in the Scotia Fundy area (fig. 4.11). Figure 4.12 and 4.13 shows maps of this region broken down by species which were prevalent in the area. Higher level of discards were localized in a few key areas on the Canadian portion of George Bank for winter, Barndoor, and thorny skates, while the Bay of Fundy was associated with winter, little, smooth, and to a lesser extend thorny skate. Spiny dogfish were discarded in the Jordan Basin along the Canadian EEZ while Brown Bank, Emerald

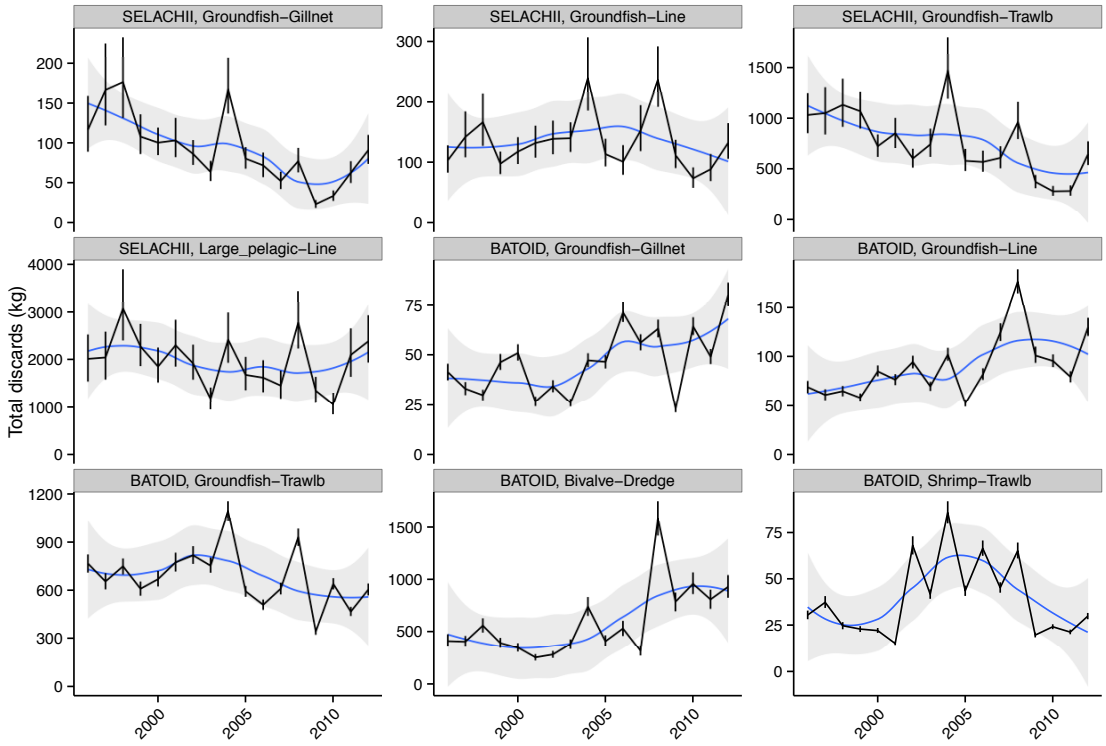


Figure 4.8: Mean and 95% credible intervals (black error bars) of total discards in metric tons (mt) for selachii and batoid per fleet sectors are plotted for the years 1996-2012. A locally weighted smoothing regression was added to visualize trend over time.

Bank and Basin were important areas of larger shark discards.

Discards hotspots for deep-water species are presented in fig. 4.14. For these species, Davis Strait was a prominent area of discard for Greenland shark, black dogfish, and spinytail skate. The Laurentian Channel and off east Newfoundland were others areas where black dogfish are recurrently discarded. Arctic skates are primarily discarded in the southeast portion of Baffin Bay whereas Jensens skates appear to be discarded in greater amount on the southwest of Baffin Bay as well as the tail of the Grand Bank. This region was also associated with higher discards of white skates.

4.5 Discussion

4.5.1 Key areas of cumulative impacts in Scotia-Fundy

These results present the most comprehensive analysis of the distribution of elasmobranch discards in Canadian Northwest Atlantic waters and Arctic adjacent seas. Our analysis

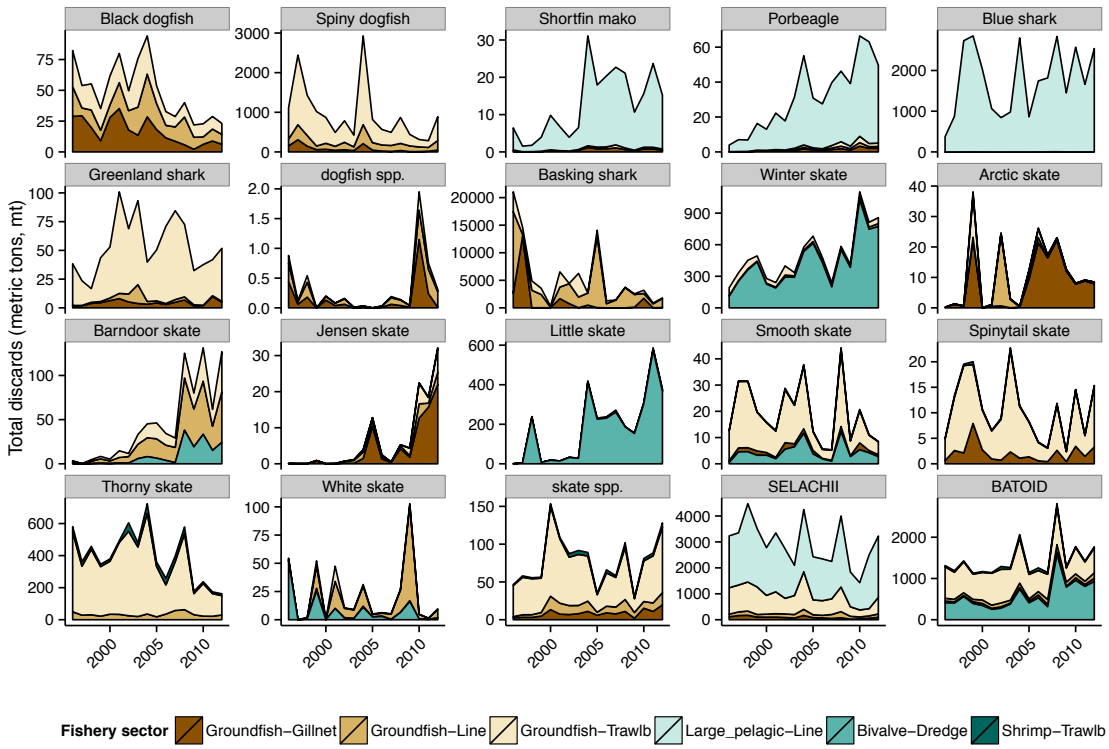


Figure 4.9: Median estimates of total discards per fleet sector for the main elasmobranch species captured in Canadian waters for the year 1996-2012.

revealed that the highest cumulative impacts of fisheries discards on elasmobranch species is localized in a few key areas to the south on the Scotian Shelf in the Bay of Fundy and Gulf of Maine. Elasmobranch discards in these regions were primarily composed of spiny dogfish, blue shark, winter, little, and thorny skates and to a lesser extend porbeagle shark, smooth and barndoor skate. These discards hotspots were driven by both species density and fishing intensity.

In the Northwest Atlantic, elasmobranch abundance and diversity increase from north to south (*Kyne et al.*, 2012). Research surveys of the Scotian Shelf and U.S. waters are consistent with our findings that demersal elasmobranchs are more diverse and abundant in southern Canadian waters (*Simon et al.*, 2003; *Simpson et al.*, 2012a; *Kulka et al.*, 2002). The frequency of discards have clearly increased for both barndoor and porbeagle sharks, which is consistent with the observation that both populations are showing signs of recovery after long periods of overexploitation (*Campana et al.*, 2012; *Coutré and Gedamke*, 2013). In fact, density-dependent changes are now noticeable in barndoor skate,

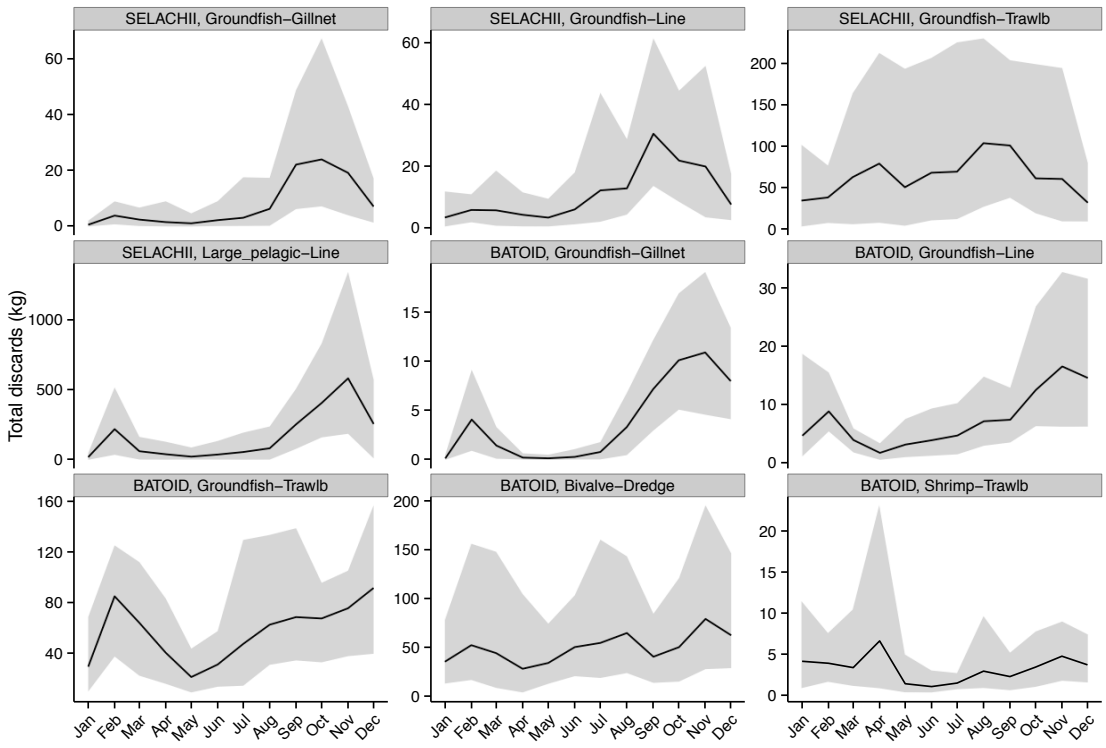


Figure 4.10: Mean and 95% credible intervals (grey ribbon) monthly estimates of total discards for selachii and batoids for 1996-2012.

both growth rate and age at maturity have increased with increasing biomass (*Coutré and Gedamke*, 2013). However, barndoor skate were found historically in very shallow waters but overfishing has limited their distribution to deeper refuge areas (*Casey and Myers*, 1998). Discards of this species remained associated with deeper waters, suggesting that, despite recovery, the species has yet to expand its range again into shallower areas, at least in Canadian waters. Levels of discards were in the same order of magnitude previously estimated for both species, about $\sim 50-200$mt for Barndoor skate (*COSEWIC*, 2010) and $\sim$ 60mt for porbeagle sharks (*Campana et al.*, 2011a).

Shortfin mako sharks are highly migratory species that are at their northern distribution within Canadian waters (*Campana et al.*, 2005). Estimates of population trends remain highly uncertain (*ICCAT*, 2012). Higher levels of discards observed in Canadian fisheries may be a sign of distribution shifts in response to warming oceans (*Cheung et al.*, 2013; *Pinsky et al.*, 2013) or of a change in population trends, or fishing practices that increasingly target tunas in warmer waters (*Paul and Neilson*, 2009). Misidentification of porbeagle

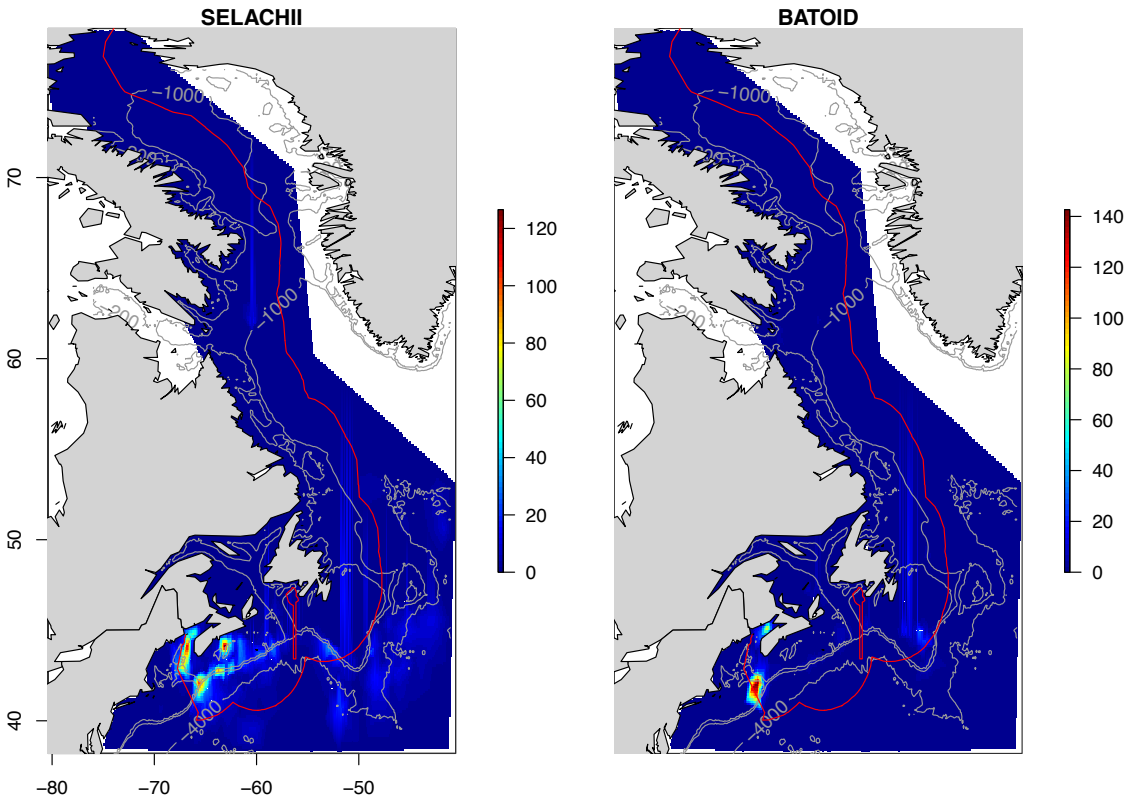


Figure 4.11: Maps showing the total annual mean estimates of discards for selachii and batoids from 1996-2012. The solid red line shows the Canadian Exclusive Economic Zone (EEZ).

sharks is also a possible source of error. Porbeagle and shortfin mako sharks as well as blue sharks are mostly taken by the swordfish and tunas pelagic longline fleet. Detailed spatial modeling of total bycatch in this fishery is presented in chapter 3.

Winter and little skate are sympatric and overlap in most of their range and consequently, are susceptible to similar fishing gear in similar geographical areas (*Simon et al.*, 2003). The majority of their discards is attributed to scallop dredges, an association that is also observed in U.S. waters (*Sosebee and Terceiro*, 2000). However, it is interesting to note that Georges Bank is a discard hotspot for winter but not for little skate. Although geographically overlapping, diet studies suggest that they have different ecological niches (*McEachran et al.*, 1976). Differences in discard patterns may be indicative of differences in habitat use in the region. In U.S. waters, both species showed increased biomass since 2008 (*New England Fishery Management Council*, 2012). Similar trends were observed in discard frequency in Canadian waters.

Approximately 50% of the winter skate's global range occurs in Canadian waters which

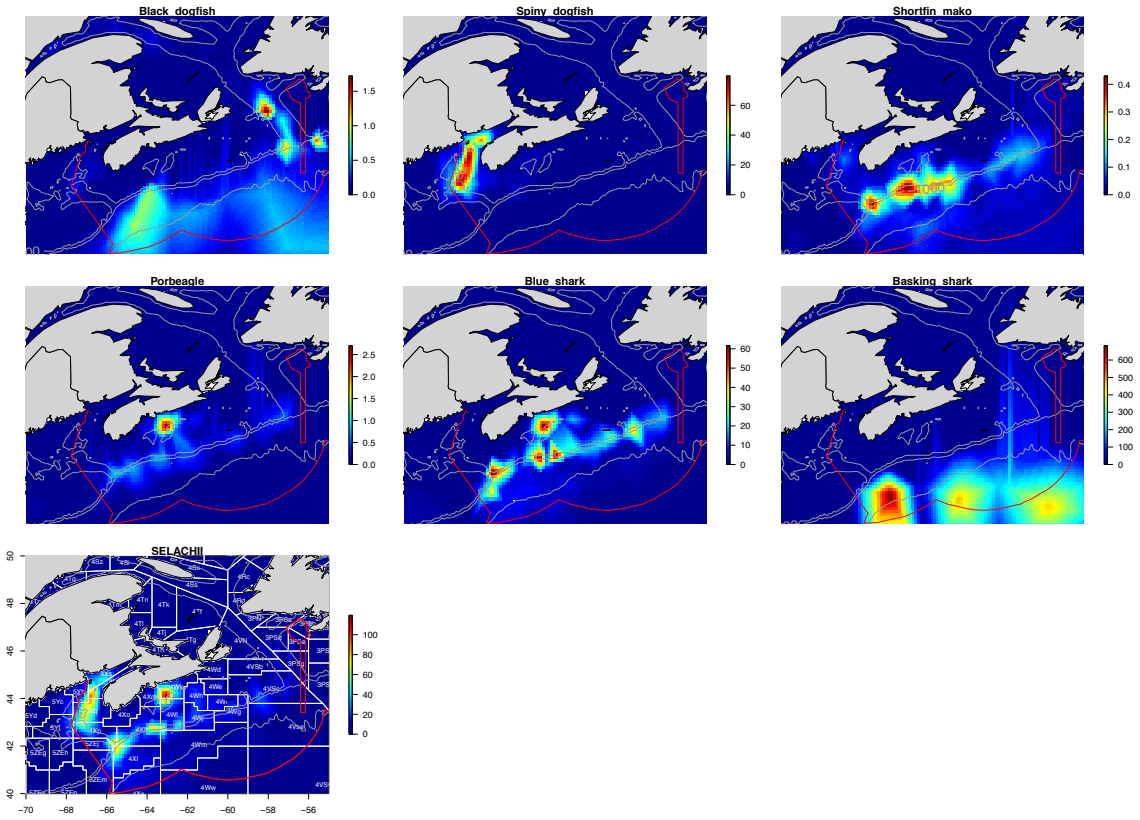


Figure 4.12: Maps showing the total annual median estimates of discards per species from 1996-2012. The solid red line shows the Canadian Exclusive Economic Zone (EEZ). White divisions are the Northwest Atlantic Fisheries Organization (NAFO) unit areas.

are known to harbour four distinct populations (*COSEWIC*, 2005). The southern Gulf of St-Lawrence population is on the brink of extinction (due to historical fishing) and recent findings suggest that this population may in fact represent a different species based on size-at-maturity and maximum size (*Kelly and Hanson*, 2013). Large winter skate discards were estimated for Georges Bank-Western Scotian Shelf-Bay of Fundy population (NAFO divisions 4X5Ze), which has been assessed as Special Concern by COSEWIC in 2005 (*COSEWIC*, 2005). The population is concentrated in the Bay of Fundy and Browns Bank and research surveys have not been able to detect any population trends since 1970s (*COSEWIC*, 2005). Bycatch estimates of winter skate had previously not been quantified for this region, whereas other estimates on the Scotian shelf dated back to 2000 in Div. 4VsW, where directed fishing occurred from 1994-2006 (*Simon and Frank*, 2000). Discard estimates were less than 200 mt (*Simon and Frank*, 2000). Our results suggest that discards have increased in recent years to nearly 900 mt annually in the Scotia-Fundy

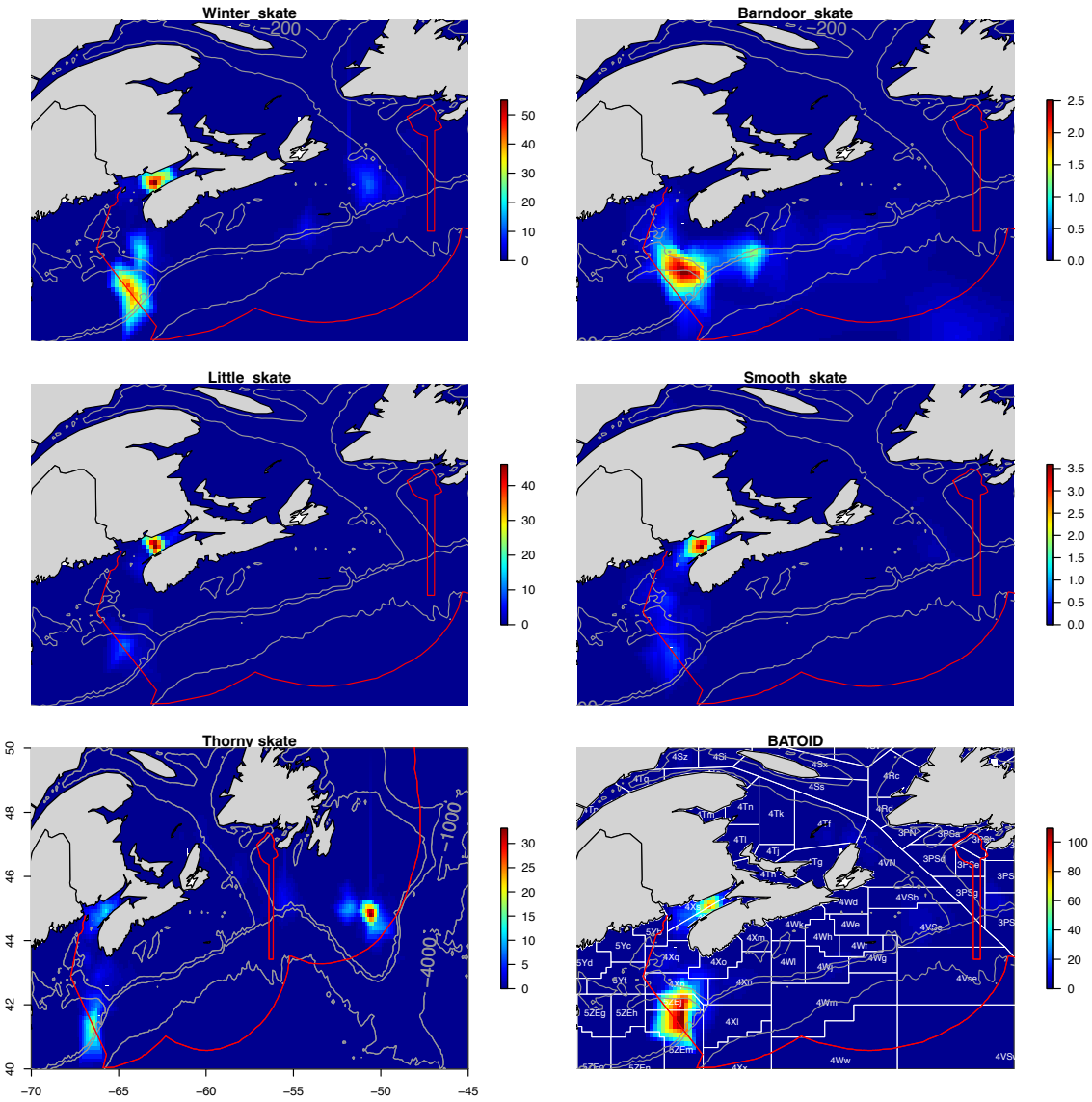


Figure 4.13: Maps showing the total annual median estimates of discards per species from 1996-2012. The solid red line shows the Canadian Exclusive Economic Zone (EEZ). White divisions are the Northwest Atlantic Fisheries Organization (NAFO) unit areas.

region. Post-release mortality of winter skate in scallop dredge is small, at least in the southern Gulf of St-Lawrence where research has been conducted (*Benoît et al.*, 2010). Survival rates may differ from one region to another depending on fishing practices and gear specifications. Post-release survival is unknown for other groundfish fisheries, but it is suspected to be greater for fixed-gear ($\sim 40-50\%$) and intermediate for trawlers (*Benoît*, 2006). The impact of this level of discards is unknown, but population size is estimated to be around 1.7 million individuals, making the largest of the Canadian populations. It is

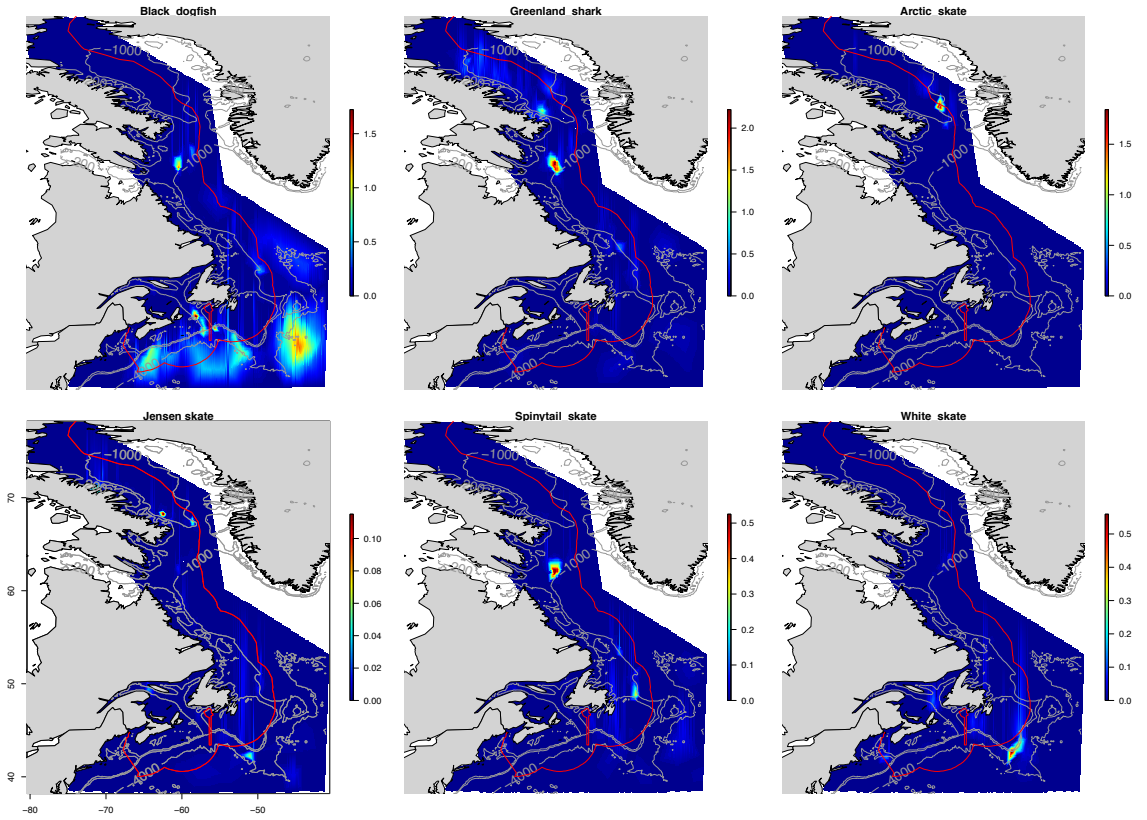


Figure 4.14: Maps showing the total annual median estimates of discards per deep-water species from 1996-2012. The solid red line shows the Canadian Exclusive Economic Zone (EEZ).

also likely mixing with the U.S. population on George Bank (*COSEWIC*, 2005).

Smooth skate is endemic to the Northwest Atlantic with 80% of its distribution in Canada (*Kulka et al.*, 2006). The species appeared to have a complex population structure with five distinct concentrations along the Scotian shelf up to the Labrador shelf (*Kulka et al.*, 2006). COSEWIC considered four designable units of which 2 remain data deficient (i.e., off Labrador — Hopedale Channel and in Newfoundland — Nose of the Grand Bank population) (*COSEWIC*, 2012b). High discards are affecting the Scotian Shelf/Laurentian Channel/Southwest Grand Banks population, which is the largest population with an estimated 5,704,000 individuals (more than 20x larger than the other populations) (*Kulka et al.*, 2006). Rough estimates of discards were available for smooth skate and reported an annual average of 50-200 mt on the Scotian Shelf (*Kulka et al.*, 2006). Our results are concordant and suggest that discards level have most likely been under 50mt since 1996, but appear to be highly concentrated in the upper Bay of Fundy.

Thorny skate are found throughout the North Atlantic and about 30-40% of their global range is in Canadian waters (*COSEWIC*, 2012a). They make up a large percentage of the Canadian demersal fish complex and are the most common skate species, which explain their high occurrence in the observer dataset. The population is treated as one homogeneous unit in Canadian waters. Our findings support scientific surveys that thorny skates are at their highest concentration on the Grand Banks (*Simpson et al.*, 2012b). Not surprisingly, the main discard hotspot is also found in this area whereas secondary areas are located on Georges Bank as well as the upper Bay of Fundy. Trends in the frequency of discards are showing a decline up to early 2000s with an upward trend until around 2007 and downward trend in recent years. These patterns are identical to the ones estimated from the Scotian Shelf/Bay of Fundy scientific surveys which suggest that the population has reached a historical low (95% decline) with some fluctuation in recent years (*COSEWIC*, 2012a). Discards levels have declined in recent years and are small compared to the average 6,500 mt directly harvested in Newfoundland waters (*Simpson et al.*, 2012b).

In the Bay of Fundy and southwest Nova Scotia, spiny dogfish were subject to a directed fishery during the summer months. The fishery has been mostly inactive since 2008 when landings dropped from 2500mt to 5mt due to declining markets (*DFO*, 2014c). Approximately half the population resides in Canadian waters during the summer months (*Campana et al.*, 2009a). The species has a long history of exploitation in U.S. waters and has suffered severe decline in the 1990s, however the stock has been rebuilt in recent years and the fishery was even granted a Marine Stewardship Council Certification in 2012 (*Worm et al.*, 2014). It is unclear why this recovery is not apparent when analyzing discard frequency in Canadian waters, quite the opposite, a downward trend is more apparent, suggesting that fishermen are catching this species less frequently. Estimates of discards in Canadian waters were much higher in 1996 but have since appeared to decline to less than 500 mt annually and are concentrated in the Gulf of Maine (Jordan Basin) along the Canadian EEZ.

Basking sharks are rarely recorded in the observer dataset (71 records), but they make up a noticeable part of the observed elasmobranch weight - due to their large body-size. The majority of the observations occurred south of Newfoundland. Because they are sometimes mistaken with Greenland sharks (*Campana et al.*, 2008), we decided to ignore observation north of 52N but this is open to discussion since satellite tagging have observed the species

north of 51N (*Gore et al.*, 2008). The Bay of Fundy is a known area of aggregation during the summer months (*Campana et al.*, 2008). The highest discard concentration appeared to be offshore the northeast channel, located between Georges Bank and Browns Bank; this is the major channel that links the Gulf of Maine with the open Northwest Atlantic. Basking sharks are known to concentrate at shelf-break fronts (*Sims et al.*, 2003) which may explain higher levels of discards at this location.

4.5.2 Deep water and northern elasmobranch species

Deep-water elasmobranch species comprised nearly half of the world's estimated elasmobranch species, however the great majority of them remains data deficient (*Kyne and Simpfendorfer*, 2010; *Dulvy et al.*, 2014). On average, these species grow slower, mature later and have longer longevity (*Garcia et al.*, 2008). Rates of population increase are among the lowest observed to date and are on average less than half those estimated for other cartilaginous fish (*Rigby and Simpfendorfer*, 2013). Consequently, deep-water may be driven to extinction under less than half the fishing pressure compared with the average shelf species (*Garcia et al.*, 2008) and vulnerability increases with depth (*Rigby and Simpfendorfer*, 2013). Once depleted, recovery of deep-water species is expected to take centuries (*Simpfendorfer and Kyne*, 2009).

Of the 18 most common species observed in Canadian fisheries, 6 would be classified as deep-water species: black dogfish, Greenland shark, white skate, Arctic skate, Jensens skate and spinytail skate. Some of these species are occasionally found in research trawls in Newfoundland and Labrador waters particularly since 1995, when the maximum depths surveyed increased. But overall, very limited information is available on their life history, diet, and distribution in the Northwest Atlantic and Arctic regions. However, more scientific attention has recently been given to Greenland sharks (e.g., *Campana et al.*, 2013; *MacNeil et al.*, 2012; *Hussey et al.*, 2014) and Arctic skate (*Peklova et al.*, 2014) because of their interactions with newly expanding Arctic fisheries.

Figures 4.6 and 4.7 quantify for the first time the spatial distribution of these species in the Northwest Atlantic waters and Arctic adjacent seas. The Hopedale Channel on the Labrador Shelf appears to be an area of higher concentration of both Arctic and Jensens skates. This area is also populated by a smooth skate population, which is data deficient. Our results support that Arctic skate are an important groundfish in Baffin Bay (*Treble*, 2002). The species is known to dominate deeper regions (below 1000 m) (*Jorgensen et al.*,

2005), however, our results suggest that frequency of discards increases with shallower waters. This contradictory pattern may be indicative of the dynamic of the fisheries rather than the ecology of the species, or might be explained by a mix of both. Our results are also influenced by southern fisheries included in the Arctic skate model. However, as opposed to the research vessel surveys which were conducted primarily during the months of September and October (*Treble*, 2002), our data combined information throughout the year and may suggest that Arctic skates utilize shallower waters some times of the year.

Discards frequency appeared to have increased for Arctic and Jensens skates since mid-2000s (Fig. 4.4). The later also show an increasing trend in discarded weight, suggesting that not only fishermen capture more frequently this fish, catches are also larger in terms of total weight. It is unclear what is causing these trends; they may suggest that 1) in the case of Jensens skate, population has increased, or 2) that the fleet might be fishing new grounds in deeper waters and is now encountering more frequently these species, or 3) that trends are confounded by misidentification and/or lack of species-specific reporting, among others. Arctic and Jensens skates are known to have overlapping distribution and misidentification is not uncommon. Arctic skate are the most prevalent skate species in polar region (Baffin bay) (*Treble*, 2002) whereas Jensens are found at lower latitudes. Overall, this is the pattern emerging from the observer dataset, but caution is warranted when interpreting these results. These fish are mostly associated with groundfish gillnet and bottom longlines, primarily targeting Greenland halibut in deeper waters on continental slopes. Fishing for Greenland halibut in David Strait has remained stable since 1994, however north, in Baffin Bay, fishing effort has increased since 2001 and new fishing grounds were explored (*DFO*, 2005) (see chpt. 2), which may explain the apparent increasing trends in discards. White skates occur in much deeper waters along the slopes off the Laurentian Channel and Newfoundland and Labrador to Baffin Bay. Spinytail skate is one of the largest species of skates recorded from the Atlantic. The species' distribution was previously thought to be broad ranging from Georges Bank, the Scotian Shelf, to Davis Strait (Fig.61, p.67 *Simon et al.*, 2009). Figure 4.6 reveals a narrower pattern concentrated on the Flemish Cap, along the Labrador shelf to Davis Strait. Moreover, spinytail and white skates may also be easily confused due to their large size and pointed snout (*Simon et al.*, 2009).

Although deep-water elasmobranchs are captured in much lower quantities than shallow-water ones, these species appear to represent about 3-5% of the total elasmobranch discards

in Canadian fisheries and have averaged about 150 mt/year. However, these estimates remains uncertain. Large quantities of skates remain unidentified (or clearly misidentified) in the observer dataset. Higher frequency of these unidentified records are located on the tail of the Grand Bank, along the Labrador shelf, and in Davis Strait and Baffin Bay (fig. 4.6), where deep-water species are more frequently captured. Total discard estimates of these unidentified skate species represents another 100 mt and may likely include many deep-water species. Round skate (*Raja fyllae*) and deep-water skate (*Rajella bathyphila*) were observed in scientific surveys in Div. 0AB (*Treble*, 2002), but were rarely recorded in the observer dataset i.e., 71 occurrences recorded for round skate and none for deep-water skate. High discards of black dogfish, Greenland shark and spinytail skate are also located in Davis Strait. Highest concentrations of black dogfish are located in the Laurentian Channel and suggests that this northern hotspot is primarily driven by fishing effort. Similarly, highest density of Greenland sharks were located in Baffin Bay whereas spinytail skates appeared to be abundant in the region and suggest that higher discard levels are also driven by species abundance. The impact of discards on these populations are unknown. However, the perceived idea that most of these species are effectively sheltered from overexploitation since they have limited interaction with fisheries may need to be revisited.

4.5.3 Geostatistical models for at-sea observer data

High variability of discards, limited information, and the inherent low frequency of some bycatch species make analysis of discards challenging. This is particularly true when one wants to look at the cumulative impacts across fisheries. For large-scale analysis, the main method employed is usually a simple scaling-up approach, where a mean observed proportion of discarded catch to target catch is estimated and then multiply by the total fishing effort (usually total landings) by fisheries and stratified by season and/or year, depending on data quality and resolution (e.g., *Wigley*, 2010; *Gavaris et al.*, 2010). If no catch were observed at a certain time/area, then it is not possible to infer any bycatch rates, hence it requires to aggregate the data into larger sample size, for example months into quarters or larger fishery areas (e.g., NAFO divisions). Consequently, areas with higher bycatch/discard levels can only be approximately identified. To address these issues, we used elaborated geostatistical models that include Gaussian Random Field (GRF) and time-serie. Geostatistical models are gaining attention in ecology for being statistically

more efficient and better utilizing the available data (*Michalsen et al.*, 2013; *Shelton et al.*, 2014). Using nearby information in both time and space, the current models automatically estimate densities in un-sampled areas, hence it is possible to infer a probability of discards across space (tiles in this case) and over time. Note however that we did not have to aggregate fishing sets per tiles, this was only done for minimizing computational time. However, different meshes and tessellations were tested during analysis and did not affect the overall results.

An intricate web of factors — environmental, biological and socio-economic, may determine discards in fishery (e.g., *Feekings et al.*, 2012; *Rochet and Trenkel*, 2005; *Matsuno et al.*, 2012). Depth is often a good determinant of species composition and discard amounts in demersal fisheries (e.g., *Sanchez et al.*, 2004; *Machias et al.*, 2001). For many species, this was indeed the case; depth was found to be critical to infer frequency of discards. Depending on the biology and/or dynamic of the interactions with fisheries, this relationship explained depth preferences for demersal species or offshore fishery-selectivity for pelagic species. These new tools allowed us to also easily extract distribution and biomass maps. Such information is traditionally only estimated form scientific research surveys. In fact, geostatistic models have improved the accuracy of abundances indices compared to conventional "design-base" fish sampling for 28 groundfish species off the U.S. West Coast (*Michalsen et al.*, 2013). More complex models with additional variables and a smaller spatial field, would yield more detailed results. These distribution maps have nonetheless been informative particularly for species that are either 1) not sampled by traditional research vessels e.g., pelagic fish or 2) fall outside the sampling areas e.g., northern species (although some research vessels were conducted in Baffin Bay and Davis Strait regions) and deep-water species.

Currently, the models combined all data collected over the time-serie to estimate the spatial effect. Consequently, we do make the assumption that spatial autocorrelation patterns have remained stable temporally and across fisheries. This most likely is not the case and local variation has probably been observed over time (for an example, see chpt. 3). Moreover, depending on the fishery sector, spatial patterns vary. However, considering the scope of the temporal and geographical area under study, the models capture the general and broad scale patterns. Whereas the smooth functions over time (year-month) do capture some of the temporal variations. We tested more elaborated models with spatial

effect replicated over time (see chpt. 2 for examples), but at the end, the models presented provided the best fit for the majority of the species.

Catchability is well-known to vary with fishing gear specifications and significant research efforts are dedicated to quantifying these variations (e.g., *Ordines et al.*, 2006; *Madsen*, 2007; *Huse et al.*, 2000). Understanding catchability is fundamental to the successful conservation and management of species. In this large-scale analysis, we coarsely included selectivity by fleet-sector, which was likely adequate for the scale of this research. However, a limitation of this approach is that by merging different fisheries, we masked sector-specific spatial and temporal trends. The models seem to capture well the overall patterns but limit fishery-specific information required by fisheries management. Moreover, fisheries with greater information on discards, which may of may not be the fleet-sector that received the largest amount of observer coverage, ultimately had greater influence on results. We did not find this problem omnipresent, but rather more obvious when some species were highly associated with one particular fleet, such as blue sharks in pelagic longline. The groundfish sector comprised the majority of the targeted fish (n=32) as opposed to other fishery sectors that included less than five species. Further stratification of this sector may be more efficient for some species, for example deep-water species, which tended to be highly associated with Greenland halibut fisheries. Our models are ultimately based on available data and rest on the assumption that observed fishing effort approximates a random sample of all activities, but this may not always be the case (*Rochet and Trenkel*, 2005; *Benoît and Allard*, 2009). Moreover, fishing activities undeniably changed over time, from gear-characteristics, fishing practices to spatio-temporal distribution of effort. The environment has also fluctuated, along with changes in fish biomass, population structure, and natural mortality. All of these factors may have contributed to changes in catchability of these fish, however their effects remains unquantified. Potential high level of misidentifications and a lack of species-specific reporting among skate species, particularly in northern latitudes, also limits our ability to accurately assess the status of some species.

Moreover, although we identified clear hotspots of higher discards in Canadian waters, other areas might need similar conservation and management attention. Lower levels of discards may threaten populations that are already in a critical state (e.g., winter skate in the southern gulf of St-Lawrence or smooth skate in the Funk Island Deep). This is

particularly true for skate species which tend to be relatively sedentary species at all life stages and hence have limited ability for migration and dispersal (*Dulvy et al.*, 2014). Moreover, 77% of the tail of the Grand Bank occurs in international waters, but appears to be an area of high concentration of deep-water species. Further research on the impact on foreign fleet should be assessed.

4.5.4 Concluding Remarks

These results present the most comprehensive analysis of the distribution of elasmobranch discards in the Northwest Atlantic waters and Arctic adjacent seas. The emphasis was particularly on deep-water species for which we still know very little about. Our results support that greater scientific and managerial attention is needed for these species particularly in northern latitudes. Waters off Labrador, Davis Strait and Baffin Bay are associated with larger concentration of these species, but are also under increasing commercial fishing pressure. The common belief that deep-water species are sheltered from exploitation needs to be revisited for Arctic skate, Jensens skate, and spinytail skate in Canadian waters.

Moreover, this study presented a methodological approach to analyze at-sea observer data using geostatistical models which include time-serie. We demonstrated that GRF included in these models can be proxy for species distribution and biomass concentration — which are information that are traditionally estimated from scientific research surveys. Hence, this approach is particularly powerful and interesting for species that are not well sampled by traditional demersal surveys, including pelagic and deep-water species.

Acknowledgements

We acknowledge our collegues at DFO, particularly Margaret Treble, Steven E. Campana, Sylvain Hurtubise, Peter Comeau, Todd Inkpen, and Carol Ann Peters. This research was supported by the Natural Sciences and Engineering Research Council of Canada (NSERC) through an Industrial Postgraduate Scholarship co-sponsored by WWF-Canada to A.C.G.

CHAPTER 5

THE EFFECT OF CIRCLE HOOKS ON SHARK CATCHABILITY AND AT-VESSEL MORTALITY RATES IN LONGLINE FISHERIES

5.1 Abstract

Fisheries bycatch is a main cause of population declines in several species of sharks and skates (elasmobranchs) around the world. Circle hooks have gained recent attention as a cost-effective bycatch mitigation tool in pelagic longline fisheries, particularly for marine turtles. Over the last few years, a growing number of studies have investigated the use of circle hooks and their effects on other species, including elasmobranchs. To elucidate the potential value of circle hook as a tool for shark conservation and management in pelagic longline fisheries, we conducted a quantitative review of all available studies to date. We compiled 15 published and 8 grey literature studies and where possible tested the effects of circle hooks on catchability and at-vessel mortality rates with random effects meta-analysis and analysis of covariance. Results indicate that as a tool to reduce at-vessel mortality for sharks, circle hooks tend to benefit over J-hooks and should thus be seen as one potential tool to help reduce at-vessel mortality of sharks. However, the high level of heterogeneity found between the studies highlights the need for shark-specific controlled experiments to provide more definitive results.

5.2 Introduction

Worldwide, unintended capture (bycatch) of threatened species is one of the most prominent issues facing the commercial fishing industry. There are particular concerns regarding bycatch of marine turtles, cetaceans, seabirds, and sharks, as these are particularly vulnerable to fishing mortality because of their life history, characterized by slow growth, late maturity, long life span, and low fecundity rates (*Musick*, 1999; *Lewison et al.*, 2004a). Substantial research efforts to reduce bycatch mortality have been devoted to marine turtles and seabirds (*FAO*, 2009c,b), but large knowledge gaps exist with regards to sharks. In recent years, the conservation and management of elasmobranchs has drawn increased attention as numerous species around the world have suffered large declines in abundance (e.g. *Dulvy et al.*, 2008; *Pikitch et al.*, 2008). One of the most challenging problems to the management of sharks globally is the high bycatch rate associated with longline fisheries (*Lewison et al.*, 2004a). Longlines are passive, non-selective gears that typically catch a wide range of species. Pelagic longline fisheries generally occur on the high seas and are multispecies fisheries primarily targeting tunas (*Thunnus* species), swordfish, *Xiphias gladius* and Mahi mahi, *Coryphaena hippurus* (*Watson and Kerstetter*, 2006; *Ward and Hindmarsh*, 2007). Benthic or bottom longline fisheries are generally conducted in coastal waters and target a variety of bony fish (e.g. *Serranidae* and *Lutjanidae*) as well as elasmobranchs (e.g *Coelho and Erzini*, 2008). Although practices vary and a combination of different hooks are used on commercial vessels, longliners targeting tuna typically use Japan tuna hooks, while vessels targeting swordfish and sharks more often use J hooks (*Watson and Kerstetter*, 2006). For their apparent conservation benefits for marine turtles and in recreational fisheries, circle hooks have been the subject of much attention over the last decade and there has been a growing movement to replace traditional J-style hooks (i.e. J hooks and Japan tuna hooks) with circle hooks. Consequently, several countries have adopted or are considering the use of circle hooks as a mean to reduce bycatch and increase post release survivorship, particularly for marine turtles.

For example, five Regional Fisheries Management Organizations (RFMOs) (Commission for the Conservation of Southern Bluefin Tuna, CCSBT, Inter-American Tropical Tuna Commission, IATTC, International Commission for the Conservation of Atlantic Tunas, ICCAT, Indian Ocean Tuna Commission, IOTC, Western and Central Pacific Fisheries Commission, WCPF) are encouraging their contracting parties and cooperating members

(CCMs) to undertake research trials of appropriate-size circle hooks in their commercial pelagic longline fisheries. Since January 2010, WCPFC was the first RFMO to include the use of large circle hooks with an offset that does not exceed 10 degrees as one available bycatch mitigation method required for implementation by all CCMs fishing for swordfish using shallow longline sets (*WCPFC*). Although less attention has been given to circle hook usage on demersal longlines, in the United States, commercial fisheries are required to use circle hooks to reduce marine turtle bycatch in the bottom longline Gulf of Mexico reef fish fishery (*Fed. Reg.*, 2011). Over the last few years, a growing number of studies have investigated the use of circle hooks and their effects on a range of species, including elasmobranchs. However, for sharks, managers and scientists are confronted with multiple studies of small sample sizes with either conflicting results or no statistical significance and no clear conclusions. The goal of this review is to synthesize existing results and provide overall conclusions on the value of circle hooks as a potential tool for shark conservation and management in longline fisheries.

5.3 Materials and Methods

5.3.1 Data Selection and Manipulation

To examine trends in circle hook effects on sharks, we conducted a systematic review of all empirical studies that compared catch rate (i.e., catchability) and at-vessel mortality rates (i.e., if a shark was alive or dead at the vessel during haulback of the gear) associated with circle hooks vs J-style hooks on both pelagic and demersal longline fisheries. Where applicable, we also gathered information on hooking locations. Relevant published and grey literature was located via electronic database searches and additional unpublished data collected by individuals currently active in this area of research. Following the methodology used by *Cooke and Suski* (2004) and *Serafy et al.* (2009), each set of species-specific results from individual studies was considered an independent study. Because of the paucity of data on sharks, if multiple circle and J-style hook sizes and offsets were compared in a given study, we pooled the data into a single hook category (i.e., circle or J-style). However, in most cases, in the original study, only one hook type was compared or results were already pooled into a single hook category. Hooks with a parallel point to the shank and no apparent curvature of the shaft were categorized as J-style hooks (*Serafy et al.*, 2009). For a detail guide on hook types used in pelagic longline fisheries, please

refer to *SPC* (2009).

5.3.2 Meta-Analysis

To better elucidate the overall differences between circle and J hooks, we completed a meta-analysis on pooled data of all shark species. Rays (*Dasyatidae* and *Mobulidae*) were excluded from these analyses owing to their different biology and ecology. However, for the most common elasmobranchs, including pelagic stingrays (*Pteroplatytrygon violacea*) we performed a meta-analysis on the pooled data at the species and family level. Contingency tables were developed using study as a categorical variable. Since these studies are likely to have numerous differences, between-study variability (heterogeneity) is believed to be present and for this reason, random effects meta-analyses using the DerSimonian and Laird method were employed (*DerSimonian and Laird*, 1986). An effect size was calculated and reported as pooled Odds Ratio (OR) with 95% confidence interval (CI). Studies were weighted according to inverse of variance of the outcomes of interest in individual studies. I2 percentage values were calculated to assess statistical heterogeneity (*Higgins et al.*, 2003). All analyses were conducted using metabin functions of the version 1.6-1 meta package of the R statistical programming language (*R Development Core Team*, 2008). Influential analysis using metainf functions of the same package was also performed to further evaluate the effect of omitting one study at a time on the pooled estimates. The number of sharks caught on circle and J hooks and the total number of hooks used in each category was used to calculate odd ratios of each study in the meta-analysis on catchability. Similarly, the number of dead sharks caught on circle and J hooks and the total numbers of sharks caught were used to calculate the odd ratio in the at-vessel mortality meta-analysis. Pooled results were tested against the null hypothesis that shark catch or at-vessel mortality is not different between hooks types. Using analysis of covariance (ANCOVA), the effects of variables influencing catchability and at-vessel mortality were further quantitatively examined. Models were coded using the SAS PROC GLM with bait type, taxonomic family, and study area as covariates, and hook type as a treatment. The use on different leader material (monofilament versus wire) could not be included in the analysis because information was too incomplete.

5.4 Results

We compiled 15 published and 8 grey literature studies as well as unpublished data from the National Oceanic and Atmospheric Administration (NOAA). *Kerstetter and Graves* (2006) study was treated as two separate studies in the meta-analysis because the original paper comprised two distinct datasets (spring and fall) that were not combined but analyzed independently in the original paper. Same wise, *Bolten et al.* (2005) phase 1, phase 2, and phase 4A of their experiment accounted for 3 individual studies in the meta-analysis. The vast majority of the studies were conducted in the Northwest Atlantic and Western Central Pacific (Figure 5.1, Table 5.1). No clear standards exist among hook classifications; studies employed a variety of hooks that differ in width, degree of offset, orientation of the point, length, gape, bite sizes, and material, and hook specifications were sometimes missing. In general, in the pooled dataset, over 60% of the J-style hooks were 8/0 and 9/0 size J hooks with some degree of offset (10°-20°) while 75% of the circle hooks were the larger sizes (i.e. 16/0 and 18/0) usually with zero to minimal degree of offset (5°-10°). Two reports (*Ingram et al.*, 2005; *Hale et al.*, 2011) were available to generate information on the effects of circle hooks on shark catches in bottom longline fisheries. All bottom longline studies were conducted in the U.S Gulf of Mexico and Northwest Atlantic Ocean.

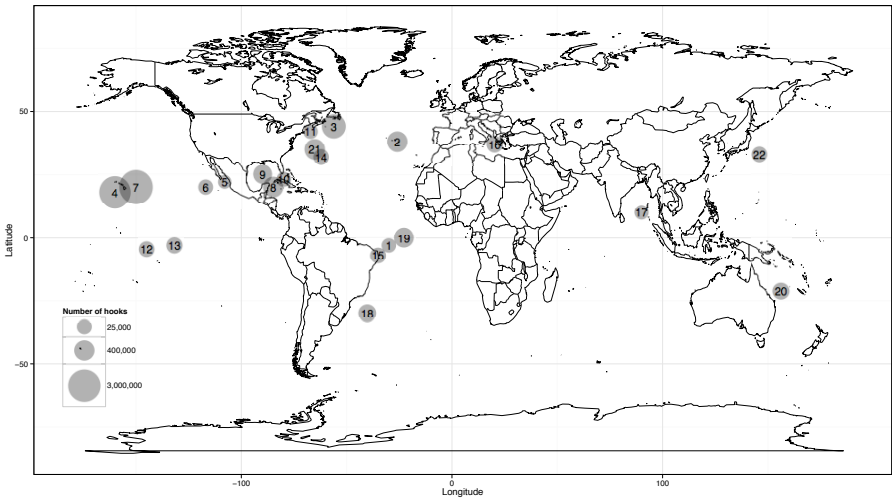


Figure 5.1: World map showing the location of all studies included in the analysis. The sizes of the points is proportional to the total number of hooks tested. Additional details are presented in table 5.1.

	Study	Hooks
1	Afonso and al. 2011	7,800
2	Bolten and al. 2005 (phase 1, 2, 4A)	416,199
3	Carruthers and al. 2009	949,999
4	Curran and Bigelow 2011	2,773,427
5	Galeana-Villasenor and al. 2008*	2,400
6	Galeana-Villasenor and al. 2009*	22,560
7	Gilman and al. 2007	3,433,422
8	Hale and al. 2011	400,000
9	Ingram and al. 2006	254,500
10	Kerstetter and Graves 2006 spring	16,560
11	Kerstetter and Graves 2006 fall	14,040
12	Kim and al. 2006	44,100
13	Kim and al. 2007	62,464
14	NOAA (unpublished)	>400,000
15	Pacheco and al. 2011	50,170
16	Piovano and al. 2010	86,116
17	Promjinda and al. 2008	6,227
18	Sales and al. 2010	145,828
19	Santos and al. 2012	305,352
20	Ward and al. 2009	95,150
21	Watson and al. 2005	427,312
22	Yokota and al. 2006	35,027

Table 5.1: List of studies and sample size (total number of hooks) comprised in this review. Asterisk (*) indicates studies that have not been included in meta-analysis because data required for the analysis were not accessible; however, summary of the results was included in the discussion.

5.4.1 Catchability

Results from these studies suggest that on pelagic longlines, most often, hook type does not have a statistically significant effect on shark catchability (*Yokota et al.*, 2006; *Kerstetter and Graves*, 2006; *Galeana-Villasenor et al.*, 2009, 2008; *Ward et al.*, 2009; *Pacheco et al.*, 2011; *Promjinda et al.*, 2008). However, higher catch rate on circle hooks has been reported multiple times for different species (*Bolten et al.*, 2005; *Watson et al.*, 2005; *Kim et al.*, 2007; *Ward et al.*, 2009; *Sales et al.*, 2010; *Afonso et al.*, 2011; *Pacheco et al.*, 2011) and, less often, lower shark catch on circle hooks have also been reported for specific species (*Kim et al.*, 2006; *Gilman et al.*, 2007; *Curran and Bigelow*, 2011). For sharks, the meta-analysis conducted on 18 studies supports the null hypothesis that no significant difference in catchability exists between hook types when all shark species are combined

(p=0.21) (Figure 5.2).

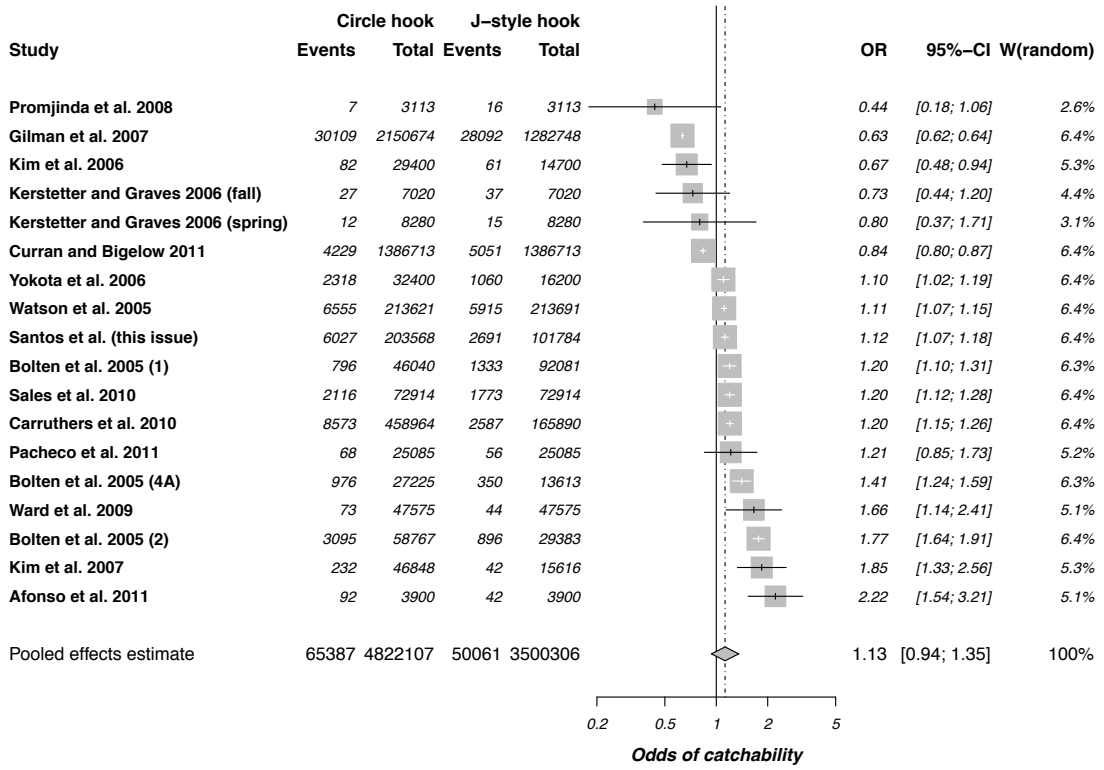


Figure 5.2: Meta-analysis on catchability showing the summary effect and each study effect size (odd ratio, OR) and 95% confidence interval (CI). OR > 1 means a higher shark catch was calculated on circle hooks versus J-style hooks. (Events) represent the total number of sharks caught on each hook category and (Total) represent the total number of hook used in each category. The area of each square is proportional to the study's weight in the meta-analysis (W). The dotted vertical line shows the pooled random effects estimate.

However, the influential analysis revealed that Gilman et al. (2007) had a significant effect on the pooled results and when removed from the analysis, the effect size results in a slight increase in shark catch on circle hooks (OR=1.2, CI=[1.07; 1.33], p=0.0016). When data were examined at the species-specific level, some minor differences emerged. For pelagic stingrays all studies report lower catchability on circle hooks (*Kerstetter and Graves*, 2006; *Piovano et al.*, 2010; *Promjinda et al.*, 2008; *Curran and Bigelow*, 2011; *Pacheco et al.*, 2011) with the meta-analysis supporting this trend, although not statistically significant (Table 5.2). Sufficient data were available to further evaluate circle hook effects on catchability for blue sharks, *Prionace glauca*, shortfin mako sharks, *Isurus oxyrinchus*, and crocodile sharks, *Pseudocarcharias kamoharai* and at the family level for

mackerel sharks, *Lamnidae*, thresher sharks, *Alopiidae*, and all remaining requiem sharks, *Carcharhinidae* (excluding blue sharks). No statistical differences in catchability between hook types were found for any of these analyses. For most meta-analyses I^2 percentage were extremely high, showing severe heterogeneity between studies (Table 5.2).

Category	Number of studies	OR	CI	I^2 (%)
All sharks combined	18	1.13	0.94-1.35	99.30
Prionace glauca	15	1.15	0.92-1.44	99.40
Pteroplatytrygon violacea	9	0.44	0.19-1.03	97.50
Isurus oxyrinchus	6	1.08	0.69-1.71	70.30
Pseudocarcharias kamoharai	4	2.07	0.93-4.64	61.10
Other Requiem, *Carcharhinidae*	8	1.13	0.72-1.77	68.80
Mackerel, *Lamnidae*	8	0.97	0.33-2.83	96.90
Thresher, *Alopiidae*	5	0.75	0.46-1.22	58.80

Table 5.2: Summary of the results of the meta-analysis on catchability showing the summary effect size (odd ratio, OR) and 95% confidence interval (CI). OR > 1 means a higher shark catch was calculated on circle hooks versus J-style hooks. I^2 describes the percentage of total variation across studies that are due to heterogeneity rather than chance. Values over 25%, 50%, and 75% are categorized has low, moderate, and high.

The ANCOVA model indicated that bait type, study area, and taxonomic family were significant covariates ($p < 0.05$) in the catchability of circle vs J hooks (Table 5.3).

In bottom longline fisheries, *Ingram et al.* (2005) found a significantly higher catch rate on circle hooks for all shark species combined and for 5 species individually in pairwise comparisons (11 species, total catch of 4469 individuals). However, an analysis from data in *Hale et al.* (2011) found significantly higher catch rate on J hooks for all sharks combined as well as by all individual species (8 species).

5.4.2 At-Vessel Mortality

Results on at-vessel mortality from pelagic longlines varies among studies with some reporting reduced at-vessel mortality with the use of circle hooks (*Carruthers et al.*, 2009; *Afonso et al.*, 2011) while others did not find any statistical differences between circle and J hooks (*Kerstetter and Graves*, 2006; *Yokota et al.*, 2006; *Pacheco et al.*, 2011; *Curran and Bigelow*, 2011). The meta-analysis on pelagic longlines on 8 studies supports a reduction of at-vessel mortality when using circle hooks vs J hooks for all shark species combined (p=0.0062) (Figure 5.3) and for blue shark ($p = 0.025$) (Table 5.4). The influential analysis did not identify any study with a significant effect on the pooled results. Severe levels of

	df	Type III SS	MS	F-Ratio	P
Bait type					
Model	5	1.90	0.38	1.87	0.10
Hook	1	0.01	0.01	0.04	0.85
Bait type	2	1.84	0.92	4.51	0.01
Hook*Bait type	2	0.07	0.03	0.16	0.85
Family					
Model	13	3.57	0.27	1.35	0.19
Hook	1	0.04	0.04	0.18	0.67
Family	6	3.32	0.55	2.72	0.02
Hook*Family	6	0.24	0.04	0.20	0.98
Area					
Model	7	3.42	0.49	2.51	0.02
Hook	1	0.27	0.27	1.39	0.24
Area	3	3.03	1.01	5.19	0.00
Hook*Area	3	0.39	0.13	0.67	0.57

Table 5.3: ANCOVA results for the effects of bait type, taxonomic family, and study area (covariate) and hook type on catchability.

heterogeneity were calculated for both analyses (Table 5.4). ANCOVA indicated that bait type was a significant covariate ($p < 0.05$) in the effect of at-vessel mortality on circle vs J hooks (Table 5.5). On bottom longlines, data were only available from *Hale et al.* (2011) and indicated no significant difference in at-vessel mortality rates between hook types for 15 species of sharks.

Category	Number of studies	OR	CI	I^2 (%)
All sharks combined	8	0.60**	0.42-0.86	92.70
Prionace glauca	7	0.65*	0.45-0.95	92.30

Table 5.4: Summary of the results of the meta-analysis on at-vessel mortality showing the summary effect size (odd ratio, OR) and 95% confidence interval (CI). OR > 1 means that a higher at-vessel mortality rate was calculated on circle hooks versus J-style hooks. I^2 describes the percentage of total variation across studies that are due to heterogeneity rather than chance. Values over 25%, 50%, and 75% are categorized has low, moderate, and high. Asterisk (**) indicates $p < 0.01$ (*) $0.05 > p > 0.01$. Studies included in "all sharks combined" analysis: refer to Figure 5.3. Studies included in *Prionace glauca* analysis — Afonso et al. 2011, Carruthers et al. 2009, Curran and Bigelow 2011, Kerstetter and Graves 2006 (fall), Pacheco et al. 2011, Yokota et al. 2006, NOAA (unpublished data).

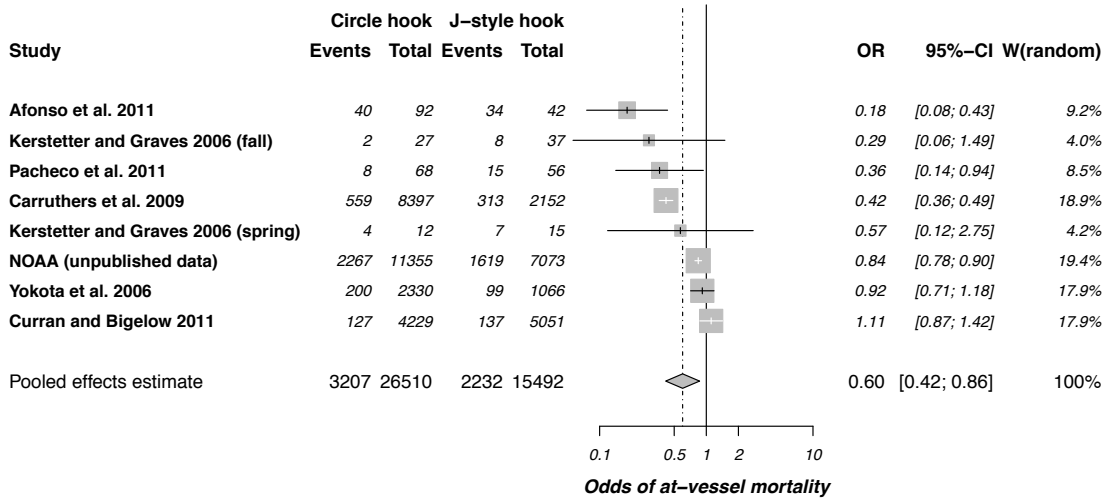


Figure 5.3: Meta-analysis on at-vessel mortality showing the summary effect and each study effect size (odd ratio, OR) and 95% confidence interval (CI). (Events) represent the total number of sharks identified as dead on each hook category and (Total) represent the total number of sharks caught in each category. The area of each square is proportional to the study's weight in the meta-analysis (W). The dotted vertical line shows the pooled random effects estimate.

	df	Type III SS	MS	F-Ratio	P
Model	3	2.16	0.72	6.47	0.00
Hook	1	1.36	1.36	12.18	0.00
Bait type	1	0.78	0.78	6.97	0.01
Hook*Bait type	1	0.19	0.19	1.70	0.21

Table 5.5: ANCOVA results for the effects of bait type (covariate) and hook type on at vessel mortality.

5.4.3 Hooking Location

Information on hooking location was only available from studies of pelagic longline gear. Because most studies focused on species other than sharks or because sharks were caught in insufficient quantities to allow meaningful comparisons, hooking location data are not readily reported and a meta-analysis could not be completed. Hooking locations were reported as percentages of external or internal (deep) hooking in a consistent fashion across the studies. Researchers generally defined deep-hooking events as those in which the hook was lodged beyond the jaw or mouth and not visible to the data recorder when the shark was brought to the side of the vessel. In general, authors found that a higher percentage of sharks get hooked externally (i.e. mouth or jaw) on circle hooks as opposed to J hooks,

which tend to lodge mostly internally (i.e. in the throat, oesophagus or gut) (*Watson et al.*, 2005; *Carruthers et al.*, 2009; *Afonso et al.*, 2011; *Pacheco et al.*, 2011). Two other studies did not find any statistical differences and indicated that sharks are hooked externally regardless of hook type (*Kerstetter and Graves*, 2006; *Ward et al.*, 2009). All studies reporting hooking location for pelagic stingrays found concordant results. Regardless of hook types, stingrays are most often hooked in the mouth or jaw (*Carruthers et al.*, 2009; *Piovano et al.*, 2010; *Promjinda et al.*, 2008).

5.5 Discussion

5.5.1 Catchability

Circle hooks can be a valuable conservation tool if their usage reduces catchability or mortality rate of bycatch species. Moreover, in order to be widely accepted by the industry, such gear modification must maintain or even increase fishing efficiency for target species (*Gilman et al.*, 2006b). Results from our review and meta-analysis suggest that, overall, on pelagic longlines, circle hooks do not affect shark catch rate. For all meta-analyses, moderate and severe heterogeneity was present, meaning that the differences between the results of the studies are not compatible with chance alone and that, as expected, other factors are affecting variances in catchability and contribute to the high inconsistency of findings. Wide ranges of study sample sizes, variety of hooks and other fishing practices were included in this meta-analysis and contributed to confound the summary results. Additionally, morphology and predation behaviour of sharks differ markedly among species, for example, common thresher sharks, *Alopias vulpinus* are typically hooked by their caudal fin as they utilize their elongate upper caudal lobe to immobilize their prey (*Nakano et al.*, 2003; *Aalbers et al.*, 2010). The meta-analysis on thresher shark did not show any statistical significant effect, however the odd ratio was the lowest after pelagic stingrays, suggesting that perhaps the shape of circle hooks (i.e., point perpendicular to the shank) curtail tail hooking. Likewise, the summary result for crocodile shark showed an opposite trend suggesting much higher catch rates associated with circle hooks. *Ward et al.* (2009) and *Pacheco et al.* (2011) reported large differences in catchability of crocodile sharks with circle hook, more than twice that of the J-style hook. Other studies have previously demonstrated that catch rates on circle hooks exceed J-style hooks for a number of teleost species, such as swordfish and yellowfin tuna, *Thunnus albacares* (e.g. *Falterman*

and Graves, 2002; *Kerstetter and Graves*, 2006; *Ward et al.*, 2009). In fact, circle hooks were designed to increase fish catchability (*Cooke and Suski*, 2004) and others studies have shown a similar pattern for some species of sharks (e.g. *Ward et al.*, 2009; *Sales et al.*, 2010; *Afonso et al.*, 2011; *Pacheco et al.*, 2011). Based on interviews conducted with pelagic longline fishers from eight different countries, their qualitative experience supports nonetheless that hook type may not have a large effect on shark catch rates (*Gilman et al.*, 2007). Many factors were not controlled for in the meta-analysis and consequently, our results should be interpreted with considerable caution and differences may exist among species. The lower catch rate on larger circle hooks observed for pelagic stingray is also likely explained by their morphology and feeding behavior. These species possess a small subterminal mouth and employ a different feeding pattern (i.e., sucking) than shark species. *Piovano et al.* (2010) showed that regardless of the hook type, larger size hook captured significantly lower number of stingrays per 1000 hooks, such that the gape size of the hook is likely the main constraint rather than the actual shape of the hook. Bait type, study area, and taxonomic family covariates examined in our analysis were all found to significantly effect catchability in pelagic longlines. Other studies support that bait type contributes greatly to shark catchability. Specifically, authors have found that the largest reduction, regardless of hook types is achieved when squid is replaced with fish (usually mackerel species) (*Watson et al.*, 2005; *Gilman et al.*, 2007; *Galeana-Villasenor et al.*, 2009). Interviews conducted with Italian and Japanese longline fishermen reaffirmed these results; many of them avoid using squid as bait in order to reduce shark interactions (*Gilman et al.*, 2008). This may be explained in part by the fact that squid baits have longer 'longevity' (i.e., remain longer on the hooks, are less likely to deteriorate or lose their attractant qualities over time, and hence have an ability to catch more fish (*Ward et al.*, 2004). It would be valuable to examine in further detail how the use of different bait affects soaking time, shark attractant qualities, and further catch rates of sharks. The study by (*Gilman et al.*, 2007) had a significant influence on the pooled result on the meta-analysis for all shark species combined. This study had the largest sample size (total number of hooks analyzed) and had one of the lowest OR (i.e., much lower shark catch was calculated on circle hooks versus J-style hooks). When omitted, a slight increase in shark catch on circle hooks is apparent. In fact, the significant drop in shark catches (36%) analyzed in (*Gilman et al.*, 2007) in the Hawaiian swordfish longline fishery following the regulations

to mitigate bycatch of marine turtles (i.e. the fishery was required to switch from using J hooks with squid baits to larger 18/0 circle hooks 10°offset with fish bait) was primarily attributed to the change of bait rather than the hook, although not statistically tested.

5.5.2 At-Vessel Mortality and Hooking Location

There is a clear association between hooking location and the severity of the injury; mouth-hooking usually induces a smaller injury to the fish than deep-hooking, and is associated with lower at-vessel mortality rate and post-release mortality (i.e., species released alive but subsequently dying from injuries or stress). For example, *Campana et al.* (2009b) observed that 96% of sharks that had swallowed the hook were either severely injured or dead, while 97% sharks that were hooked superficially (mouth or jaw) were released healthy (lively with no apparent trauma). Moreover, postmortem pathology studies have also indicated that deeply embedded hooks (i.e., oesophagus and gastric wall) in the blue shark caused chronic systemic disease *Borucinska et al.* (2001, 2002). In contrast to J hooks, circle hooks are expected to result in higher jaw or mouth hooking frequency because of their round shape, which is expected to rotate more readily inside a fishs mouth (*Cooke and Suski*, 2004). Our review suggests that the majority of the study indicate that shark species are more often mouth or jaw hooked (i.e., external) on circle hooks. The general belief is that circle hooks contribute to reduce deep-hooking and consequently reduce at-vessel mortality of sharks. Results from the meta-analysis supports that circle hooks help reduce at-vessel mortality for all shark species combined and blue shark. At-vessel mortality rates differ among species (*Carruthers et al.*, 2009). Because of data availability, only blue shark at-vessel mortality could be statistically analyzed. Blue sharks are known to be a hardy species compared to others, with many studies reporting survival rates of 70 to 95% (*Diaz and Serafy*, 2005; *Campana et al.*, 2009b). Consequently it is not surprising to see a slight decrease in the effect of circle hooks on at-vessel blue shark mortality compared to all shark species combined.

Based on this review and meta-analysis, the use of circle hooks does not affect catches of shark species (data combined) and contribute to reducing at-vessel mortality of shark species (data combined) and individually for the blue shark. Whilst this suggests a tendency for circle hooks to benefit shark conservation, these indicative advantages may not outweigh their negative effect on shark catch rates (i.e., increase catchability) for some species. Nevertheless, as a first step, we believe that, where experimental results support

the conservation benefits of using circle hooks and where live-release is legislated and monitored adequately, there is sufficient evidence to promote the use of circle hooks in commercial pelagic longline. As for demersal longline fisheries, too little information is available to advance any concluding remarks. As noted by previous authors, circle hooks are not a panacea for species conservation (e.g. *Cooke and Suski*, 2004; *Serafy et al.*, 2009). In fisheries where there is regulatory framework for sharks and a desire to further minimize shark bycatch and subsequent mortality, managers and scientist are urged to explore additional bycatch mitigation options, such as bait type, which could work in conjunction with the promotion of circle hooks. The high level of heterogeneity level found between the studies highlights the need for further controlled experiments designed specific to sharks over a range of treatments, such that all probable factors affecting catch rates, including hook type can be effectively modeled.

Acknowledgements

The authors wish to acknowledge funding from the National Science and Engineering Research Council of Canada and thank K. Andrews (National Marine Fisheries Service) for discussions relative to data analysis and G. Davenport (National Marine Fisheries Service), R. Davies (WWF International), and B. Worm (Dalhousie University) for their advice and comments on this manuscript.

CHAPTER 6

NO EFFECT FROM RARE-EARTH METAL DETERRENT ON SHARK BYCATCH IN A COMMERCIAL PELAGIC LONGLINE TRIAL

6.1 Abstract

The indiscriminate capture of non-target organisms (bycatch) in commercial fisheries undermines the sustainable development of marine resources. In the Northwest Atlantic, blue sharks (*Prionace glauca*) account for most of the bycatch in the Canadian pelagic longline swordfish fishery. Minimizing the capture of this species is of interest to conservationists as well as the fishing industry because the high incidence of shark bycatch negatively affects fishing operations through bait loss and increased handling time. Electropositive metals (e.g., lanthanide) oxidize in seawater and create electric fields, which can alter the swimming and feeding behaviours of several species of sharks. Although electropositive metals appear to have the potential to reduce shark bycatch in pelagic longline fisheries, there have not been any controlled trials reported from a commercial fishery. A total of 7 sets (6300 hooks) with 3 hook treatments (standard hooks, hooks with electropositive metals (neodymium/praseodymium), and hooks with lead weights) were deployed in 2011 on the Scotian Shelf in the Northwest Atlantic. The results of this study show that electropositive metals did not reduce the catch of blue sharks or other common shark bycatch species, and hence do not present a practical bycatch mitigation measure for the Canadian longline fishery.

6.2 Introduction

The incidental capture of sharks in fisheries worldwide has been implicated as one of the leading causes of observed shark population declines and represents an important challenge for their management and conservation (*Lewison et al.*, 2004a; *Gilman et al.*, 2007; *Dulvy et al.*, 2008; *Camhi et al.*, 2009). Pelagic longline fisheries are well known for their significant shark bycatch, which accounts for a large percentage of the total catch (*Gilman et al.*, 2007; *Mandelman et al.*, 2008). In the Northwest Atlantic, sharks often contribute $> 30\%$ of the Canadian pelagic longline fishery catch by weight (*Campana et al.*, 2009b). This fishery primarily targets swordfish (*Xiphias gladius*) and, more recently, albacore (*Thunnus alalunga*), bigeye (*T. obesus*), and yellowfin (*T. albacares*) tunas (*Paul and Neilson*, 2009). Fishing practices differ for tuna and swordfish-targeted sets (*Brazner and Mcmillan*, 2008; *He et al.*, 1997), with the latter accounting for most of the shark bycatch (*Campana et al.*, 2006). Blue sharks (*Prionace glauca*) account for more than 90% of the shark catch, with shortfin mako (*Isurus oxyrinchus*) and porbeagle sharks (*Lamna nasus*) contributing the majority of the remaining catch (*Campana et al.*, 2006).

High numbers of shark bycatch can result in gear damage, bait loss, lower target catch, and handling risks to the fishing crew (*Gilman et al.*, 2008). As such, reducing shark interactions is a priority for fishermen, and interest in the topic has further increased with the fishery assessment for Marine Stewardship Council (MSC) certification (*Carruthers and Neis*, 2011; *Carruthers et al.*, 2009, 2011). Many fishermen aim to avoid shark catch (*Carruthers and Neis*, 2011), although convincing solutions are not yet available. One possible option involves mitigation measures that take advantage of the electrosensory system in sharks (*Swimmer et al.*, 2008). Sharks employ a variety of sensory mechanisms to detect and localize prey. Their electroreceptors or ampullary organs can detect low frequency bioelectric fields (5-10 nV/cm) produced by prey at short range (*Murray*, 1960; *Kalmijn*, 1971; *Tricas*, 2001). The ampullae of Lorenzini are restricted to the head in sharks and consist of a network of hundreds of receptor cells located below the surface of the skin (*Collin and Whitehead*, 2004). Electropositive metals (e.g. lanthanide) oxidize in seawater and create electric fields that can be hundreds of times greater than the threshold of sensitivity for some elasmobranchs (*Kajiura et al.*, 2010). Experiments have indicated that the presence of these metals can alter the swimming and feeding behaviours of several species of sharks (*Rigg et al.*, 2009; *Brill et al.*, 2009; *OConnell et al.*, 2010; *Stoner and*

Kaimmer, 2008; *Wang et al.*, 2008). The mechanism of deterrence is not fully understood, but it is believed that these metals perturb the electrosensory system in sharks and cause the animals to exhibit avoidance behaviours (*Rice*, 2008). Because tunas and swordfish do not have electroreceptors, this method has the potential to reduce shark bycatch rates without affecting target catch rates. However, data from controlled trials under commercial pelagic longline fishing conditions are not yet available. In collaboration with the World Wildlife Fund (WWF)-Canada and the Canadian swordfish industry, we tested the null hypothesis that electropositive metals do not reduce shark bycatch or target (swordfish) catch in a commercial pelagic longline fishery.

6.3 Methods

Fishing operations were conducted aboard the commercial fishing vessel Addie n'Ainslie using longline gear typical for targeting swordfish. A total of 7 sets (70 trials) made up of 6300 hooks were deployed between September 27 and October 3, 2011. The exact fishing locations were selected based on the local knowledge of the Captain (Figure 6.1). The fishing gear included 16/0 10°offset circle hooks (Mustad 39966) attached to 8-m branchlines clipped to the mainline. The gear was set to fish in the upper 20 m (4.5-m drop lines) with 3 hooks fished between buoys (a 'basket'). Each section consisted of approximately 20 baskets (3 km in length). Gear was baited with Atlantic mackerel (*Scomber scombrus*), set in the evening at approximately 5:00 pm local time (9:00 pm UTM). The soak time averaged 7h and 05 min and ranged from 6h and 13 min to 8h and 18 min.

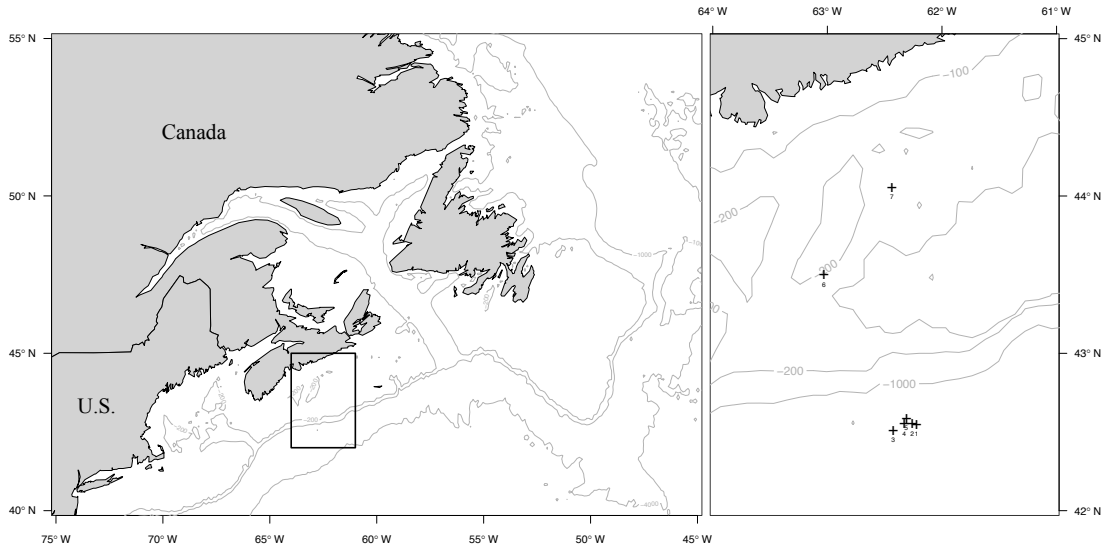


Figure 6.1: Map showing locations of experimental fishing on Scotian Shelf off Nova Scotia Canada.

Three different treatments were tested: standard hooks, hooks with electropositive metals, and hooks with lead weights as inert controls for the shape and weight of the deterrent metals. The electropositive metal pieces were an alloy of 76% neodymium and 23% praseodymium (Nd/Pr) from HEFA Rare Earth Canada Co., Ltd (Richmond, BC, Canada). The experimental design was developed in collaboration with fishermen to minimize interference with standard fishing practices. The lead weights and Nd/Pr alloys were mounted with cable ties on small tuna clips (Figure 6.2) to facilitate rapid clipping and unclipping on the gangion. The small tuna clips were attached approximately 20 cm above the hook such that the lead weight and Nd/Pr alloys were positioned just above the hooks. The Nd/Pr alloys were provided as half spherical weights that were approximately 55 g with a 5 mm hole for attachment (Figure 6.2). Because the Nd/Pr alloys react electrochemically with and dissolve in seawater, the metals were used for 2 sets and then replaced with new ones. Commercial lead weights (57 g tidal flat sinkers) with a comparable weight and shape served as procedural controls and were attached in a similar manner as the Nd/Pr alloys.

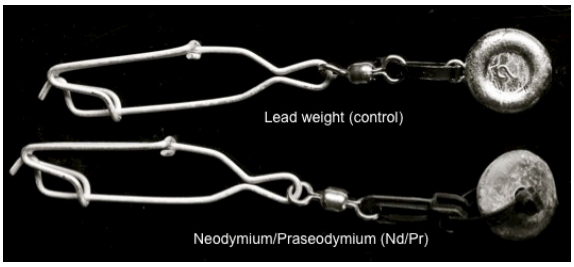


Figure 6.2: Nd/Pr alloy and lead attached to a small tuna clip using a cable tie (total length, approximately 17.5 cm). The diameters of the Nd/Pr and lead weights were approximately 3 cm.

Each longline set had a total of 900 hooks and consisted of 10 trials of the following 3 treatments: a standard hook treatment, Nd/Pr alloys, and a control lead weight. Each treatment was applied over blocks of 30 hooks each. Each trial consisted of an Nd/Pr treatment, followed by a control lead weight treatment, followed by a standard hook treatment such that a trial consisted of 90 hooks. Two to 4 standard baskets were set in addition to the experimental gear and fished during the experiment. The data from these hooks were excluded from analysis. An experienced scientific observer from Javitech Limited (Bedford, Nova Scotia) contracted by the Department of Fisheries and Oceans Canada monitored the experimental protocols and collected information on the number and estimated weight of all species caught for each 30-hook treatment.

The mean catch-per-unit effort (CPUE, number per 1000 hooks) was calculated to assess the relative catchability between treatments (all sets included). To account for possible dependencies between treatment subsections within sets, the count data were further analyzed by fitting generalized linear mixed models (GLMM) with a random-effect for 'set' and a log-link function (Poisson regression). Model adequacy was verified using Pearson residuals plotted against fitted values, and F tests were used to test for treatment differences. For blue sharks, all sharks combined, and swordfish, the GLMM predicts the mean catch as the number of individuals per set as a function of the treatment. All statistical analyses were performed using the function glmmPQL in the MASS package in R version 2.14.0 (*R Development Core Team*, 2008).

6.4 Results

Overall, 337 individuals from 7 species were captured, with an overall catch rate of 53 fish per 1000 hooks. Blue sharks, swordfish and shortfin mako sharks accounted for 97.3% of the total catch (Table 6.1). Shark catch rates (all species combined) varied between 33.3 and 43.8 per 1000 hooks, while swordfish catch rates varied between 10 and 22.9 per 1000 hooks (Figure 6.3). Blue shark catch varied greatly from 3 to 80 individuals per set with the last three sets accounting for over 80% of the catch.

For blue sharks and all sharks combined, no significant differences in the CPUE were observed between the treatments (Table 6.2). However, the swordfish catch was significantly reduced on the hooks treated with the control lead and Nd/Pr weights by 56% and 48%, respectively, compared to the standard hooks (Table 6.2).

Species	Standard n (CPUE)	Lead n (CPUE)	Nd/Pr n (CPUE)	N (CPUE)	Percent composition
Swordfish *Xiphias gladius*	48 (22.86)	21(10.00)	25(11.90)	94(14.92)	27.90
Blue shark *Prionace glauca*	69(32.86)	84(40.00)	65(30.95)	218(34.60)	64.70
Shortfin mako *Isurus oxyrinchus*	6(2.86)	8(3.81)	2(0.95)	16(2.54)	4.70
Porbeagle *Lamna nasus*	1(0.48)	0	3(1.43)	4(0.63)	1.20
Bluefin tuna *Thunnus thynnus*	1(0.48)	1(0.48)	0	2(0.32)	0.60
Albacore *Thunnus alalunga*	0	0	1(0.48)	1(0.16)	0.30
Anglerfish *Lophiiformes spp.*	0	0	1(0.48)	1(0.16)	0.30
Unidentified	1(0.48)	0	0	1(0.16)	0.30
Total	126	114	97	337	100.00

Table 6.1: Catch composition, mean catch-per-unit effort (CPUE, number per 1000 hooks), total catch (N), and percent composition for all species captured (total of 6300 hooks) with 3 different treatments (2100 hooks per treatment).

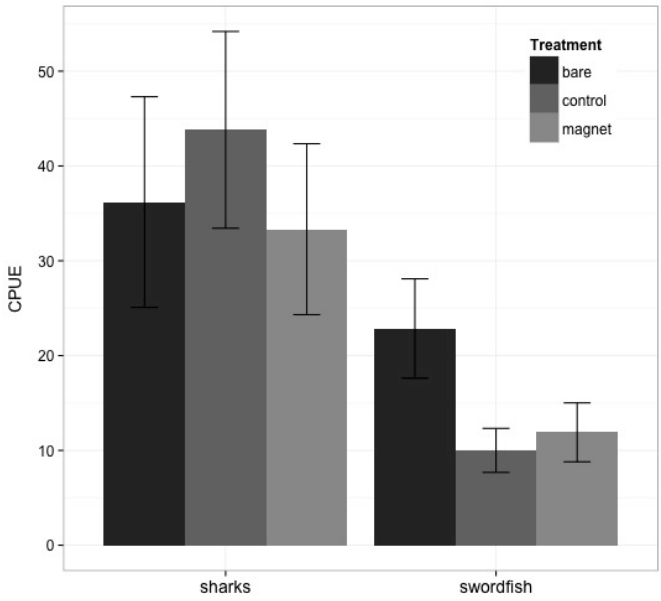


Figure 6.3: Mean catch-per-unit effort (CPUE; individuals per 1000 hooks) in the experimental longline trial (7 sets) for swordfish and all shark species combined. Error bars represent the 95% confidence intervals.

6.5 Discussion

This study suggests that electropositive metals do not have any significant deterrent effect on the most common shark bycatch species in a pelagic longline fishery. Over the last five years, an increasing number of studies have investigated the effects of electropositive metals as a shark deterrent (*Kaimmer and Stoner*, 2008; *Stoner and Kaimmer*, 2008; *Tallack and Mandelman*, 2008; *Wang et al.*, 2008; *Brill et al.*, 2009; *OConnell et al.*, 2010, 2011; *Robbins et al.*, 2011; *Hutchinson et al.*, 2012). Several controlled experiments (e.g., laboratory trials) have shown that these electrochemically active metals can have a significant deterrent effect on some shark species. Yet, few studies were able to verify these findings in commercial fishing operations. To date, fishing trials on longline gear typical of that used in the Pacific halibut fishery in the northwestern Gulf of Alaska showed that electropositive metals reduced spiny dogfish (*Squalus acanthias*) catch by 19% (*Kaimmer and Stoner*, 2008). This contrasts with a 70% reduction in dogfish catch that was reported in previous laboratory trials (*Stoner and Kaimmer*, 2008). In the Gulf of Maine, bottom longline and rod-and-reel trials did not find any effect on spiny dogfish catches (*Tallack*

	Value	Std. Error	DF	t-value	p-value
Blue sharks					
(Intercept)	1.79	0.49	12	3.67	0.00
Lead	0.20	0.18	12	1.08	0.30
Nd/Pr	-0.06	0.19	12	-0.31	0.76
All sharks combined					
(Intercept)	2.08	0.39	12	5.39	0.00
Lead	0.19	0.18	12	1.04	0.32
Nd/Pr	-0.08	0.20	12	-0.42	0.68
Swordfish					
(Intercept)	1.75	0.31	12	5.69	0.00
Lead	-0.83	0.25	12	-3.37	0.01
Nd/Pr	-0.65	0.23	12	-2.82	0.02

Table 6.2: Estimated parameter values, standard error, degrees of freedom, t-values, and p-values from the generalized linear mixed models.

and Mandelman, 2008) whereas a recent study which tested a combination of magnetic and electropositive metals (known as the SMART hook) reported a 28% reduction (*O'Connell et al.*, 2014). Other coastal bottom longline trials on the east coast of Oahu, Hawaii showed a significant reduction in juvenile scalloped hammerhead (*Sphyrna lewini*) catch, but did not detect any effect on other coastal shark species (*Hutchinson et al.*, 2012). Similar to our findings, results of pelagic longline trials off the coast of Southern California and Ecuador indicated that there were no differences in the catch rates of blue sharks, shortfin mako sharks, and other pelagic species among electropositive and control hooks (*Hutchinson et al.*, 2012).

Several factors can influence the deterrent effects of electropositive metals in the field, such as shark density, competition and hunger level (*Stoner and Kaimmer*, 2008; *Brill et al.*, 2009; *Robbins et al.*, 2011), presence of conspecifics (*Robbins et al.*, 2011), and differences in feeding ecology (*Rigg et al.*, 2009; *Stoner and Kaimmer*, 2008). In the present study, blue shark catch varied greatly among sets, with the majority of sharks (over 80%) captured during the last three fishing sets. Hypothetically, high local densities of blue sharks may have increased competition and aggressiveness, thereby limiting the effects of the electropositive metals in this study. In addition, studies on shark sensory physiology and brain structure have shown that pelagic species have significantly fewer electrosensory pores than coastal ones (*Kajiura et al.*, 2010). This could explain why electropositive metals are a more effective deterrent in some coastal regions, but are not as effective in the

pelagic environment (see detailed discussion in (*Hutchinson et al.*, 2012)).

A contributing factor in this experiment might have been the visual differences between the hooks with and without metals attached. Large pelagic teleosts, such as marlins, swordfish, and tuna, are predators that rely largely on their vision to catch prey. Increased visibility of the fishing gear (e.g. through multifilament lines) can reduce the catch of pelagic fish, including swordfish (*Stone and Box*, 2001). The swordfish catch was lower using both the Nd/Pr and lead control weights, which suggests that the observed effects may have been the result of the physical structure attached to the branchline rather than the electromagnetic properties of the metal. Although pelagic sharks are also visual predators, the visual cues of the metals did not decrease their catch rates. Previous authors (*Kaimmer and Stoner*, 2008; *O'Connell et al.*, 2014; *Hutchinson et al.*, 2012) agreed that the use of electropositive metals is currently impractical on a fast-past commercial fishing scale because of the cost and repetitive replacement of the deterrents. Following discussions with fishermen, the small tuna clip method was ultimately the only realistic approach for setting metals in a timely manner without altering fishing methods. This technique increases gear visibility, which may have caused the reduced swordfish catch in these trials. This unwanted side effect further impedes the use of these deterrents in a commercial pelagic longline fishery.

We caution that the sample size in this experiment (total n=21) limits the power of the statistical tests. Yet we note that differences in the catch rates of sharks were minimal between treatments, indicating that any possible effect of electropositive metals would likely remain small, even if sample size was increased and the results were statistically significant. Electropositive metals may still be an option for reducing shark bycatch in coastal environments for particular shark species (e.g., juvenile scallop hammerheads (*Hutchinson et al.*, 2012)), but additional commercial fishery trials are necessary to tests these methods. In our view it is important that experiments be conducted under realistic conditions that reflect typical fishing operations for which these deterrents may be used as bycatch mitigation tools.

Acknowledgements

We thank Captain Frankie and Gussie Reyno and the crew of the Addie nAinslie (Troy Gilkie, John Gilkie, Donald Gilkie, Danny Laing, Gordie Reyno, and Toby Nauglen) for

their help and enthusiasm with the project. Funding for this project was provided by WWF-Canada, the Government of Canada Habitat Stewardship Program for Species at Risk, and the Natural Sciences and Engineering Research Council. We also would like to thank Troy Atkinson, the staff at Javitech Limited, particularly David Spallin, the fishery observer who collected all of the data, Mike Eagles and Steven Campana at the Department of Fisheries and Oceans, and Jarrett Corke from WWF-Canada.

CHAPTER 7

CONCLUSION

7.1 Elasmobranch Bycatch and Discards in the Northwest Atlantic and Arctic Adjacent Seas

One of the primary goals of this dissertation was to identify and quantify bycatch risk: (i) for particular species of interest — Greenland shark (*Somniosus microcephalus*) (chapter 2); (ii) for fisheries with higher bycatch rate — Canadian Northwest Atlantic swordfish and tuna pelagic longlines (chpt. 3); and, (iii) to provide a comprehensive analysis of cumulative impacts for all elasmobranches across fisheries in Eastern Canadian fisheries since 1996 (chpt. 4).

Quantifying bycatch and identifying hotspots can help prioritize management, but may also provide new insights into the biology of these species. In some cases, at-sea observer data is the only information available; for instance, for species that are not captured by research vessels e.g., pelagic fish, or that are beyond the geographical area sampled e.g., deep-water species.

Overall, results demonstrate clear delineated areas of higher bycatch and discard risks despite high spatiotemporal variabilities in shark and skate catches in Canadian fisheries.

In Baffin Bay, bycatch risk of Greenland shark is up to 100 times higher in shallower areas near coastal fjords, as well as in the northeast part of the bay, a relatively new fishing ground (chpt. 2). Some fjords in the region, such as Scott Inlet and Sam Ford Trough, have been associated with repeated occurrences of juvenile Greenland sharks and suggest that these areas are of biological importance to the species (*Hussey et al.*, 2014). The large-scale analysis reveals that Greenland sharks are more prevalent in Baffin Bay, however Davis Strait is associated with higher discard rates, due to higher levels of fishing

(chpt. 4) and, consequently, should be a priority area for management. In particular, bottom trawl fisheries targeting Greenland turbot (*Reinhardtius hippoglossoides*) should be further investigated, as they contribute to the majority of the catch of this species (chpt. 4).

The Canadian Northwest pelagic longline fishery has a relatively small footprint, both in terms of geographical distribution and fishing effort (annual number of fishing sets), but nonetheless contributes to the bulk of discards of large sharks in Canadian waters — porbeagle sharks (*Lamna nasus*), blue sharks (*Prionace glauca*), and shortfin mako (*Isurus oxyrinchus*) (chpt. 4). The fishery captures these species on average seven times more often than any other fishery sector (chpt. 4). New estimates of total bycatch (in counts of animals) suggest that catches are 2x, 3x, and 5x larger for blue, porbeagle, and mako sharks, respectively, compared to previous estimates for this fishery (chpt. 3).

Discrepancies in estimates from chpt. 3 and 4 can be explained as: (i) chpt. 3 estimates total catch in the number of individuals whereas chpt. 4 estimates the total discarded weight, and as such does not include fish that were landed; (ii) weights are estimated by on-board observers per fishing set, and consequently are a coarser approximation than counts; and, (iii) chpt. 3 analysed the data with elaborated spatiotemporal models that allowed spatial autocorrelation to vary every year and included seas surface temperature for better predictions. These differences ultimately resulted in different estimates which are difficult to compare, but do overlap within estimated uncertainties. Higher bycatch levels in this fishery do raise some concerns, particularly for porbeagle sharks — an endangered species, which are managed under a strict recovery plan.

Both studies are concordant with the locations of hotspots in the pelagic longline fishery, the largest amount of bycatch and discards are aggregated in three key areas on the Scotian Shelf: Browns Bank, Emerald Bank and Emerald Basin (chpt. 3 and 4).

Cumulative impacts across fisheries on demersal elasmobranch species are concentrated in southern Canadian waters bordering the United States of America, particularly in the upper Bay of Fundy, Jordan Basin, Georges Bank, and Browns Bank. These areas are associated with spiny dogfish, winter, little, and thorny skates and to a lesser extend smooth and barndoor skates (although increasingly so in recent years). Several of these populations are increasing in American waters, notably spiny dogfish, winter, little, and barndoor skate and estimated Canadian populations, if available, suggest that cumulative discards are mostly affecting the largest of the Canadian populations (chpt. 4). In 2012, total discards

were estimated at 3250 mt (2722-3849, 95% credible intervals) of selachii and 1772 mt (1642-1911) of batoids. Current discard levels are not alarming, however close monitoring should be in order, particularly for winter skate in scallop dredge fisheries, for which discards have increased to 900 mt in recent years. Directed fisheries remain the main threat to thorny skate.

Chapter 4 was necessary to identify areas with important gaps that require urgent scientific and managerial attention. We currently poorly understand deep-water species. Our analysis suggests that a large concentration of these species are found at northern latitudes, and that Davis Strait is an area of higher discard levels. At-sea observer data is particularly dubious in these regions and limits greatly our ability to accurately assess the status of these species.

7.2 Bycatch Mitigation

Where bycatch mitigation techniques have been implemented, it is usually first to minimize impacts on other commercially important fish (e.g., cod, *Gadus morhua*) rather than elasmobranch species. However, mitigation techniques can and have been beneficial to cartilagenous fish. Some good examples in Canadian waters are the shrimp fisheries which use Nordmore Grate (since 1997) that sorts groundfish from shrimps, and consequently, discard levels of demersal sharks and skates are low (*DFO*, 2010). This pattern is apparent in the overall analysis; shrimp fisheries contribute very little to the overall discards of skate species (chpt. 4). Moreover, circle hooks have been widely adopted worldwide by pelagic longliners primarily for their conservation benefits to marine turtles. Our review clarifies and supports that this cost-effective tool reduces discard shark mortality (chpt. 5).

However, these mitigation methods do not address high catch rates, and thus, we further investigated the use of electropositive shark deterrent in the Canadian swordfish pelagic longline fishery. However, results were less than desirable; the method is costly and has no significant impact on shark catches (chpt. 6). Consequently, other management measures should be investigated in this higher risk fishery.

Candidate areas for spatial management were investigated in chpt. 3 and supports the idea that such methods would be viable to reduce catch rates and overall mortality of large sharks in longline fisheries. In the Canadian context, this solution appears to be particularly promising for porbeagle sharks, which are associated with shallower shelf waters and are

showing more consistent hotspot patterns over time (chpt. 3).

7.3 Geostatistical Models and New Statistical Developments for Bycatch Studies

Geostatistical tools are gaining popularity in fisheries sciences (*Michalsen et al.*, 2013; *Quiroz et al.*, 2014; *Muñoz et al.*, 2012). New statistical developments, such as INLA and SPDE, and their R interface (R-INLA) are making these models much more accessible to a greater scientific audience. Another important part of this research was to develop and apply, for the first time, these new statistical tools to the study of bycatch. Chapters 2, 3, and 4 provide examples of these powerful new tools, each of these chapters building upon the other to illustrate the flexibility of these methods with regards to bycatch studies. Chapter 2 presents the breath of truly spatiotemporal models that could be quickly and efficiently modeled using the INLA/SPDE approach (see Table 2.2). The study describes these methods and the advantages of using Gaussian Random Field (GRF) in bycatch studies. All R codes are now available to the scientific community on the R-INLA website (www.r-inla.org).

Chapter 3 uses low-observer coverage (less than 5%) to predict bycatch rates at each logbook fishing set (total fishing effort). This chapter further introduced a new approach to statistically quantify hotspots, namely the Excursion Functions, which provide a clear joint statistical interpretation of hotspots (*Bolin and Lindgren*, 2014). This new method provides a practical tool to inform decisions on location and extent of spatial management. Different levels of threats can hence, now be easily investigated (see Appendix B).

Chapters 2 and 3 are fitting models to count data and provide examples of highly zero-inflated likelihood for Greenland sharks and negative binomial likelihood for the large shark complex in pelagic longline fisheries. However, in many instances, counts data are not available from at-sea observer data, particularly for smaller-sized species, for example demersal fish, such as batoids and *squalus* species. Hence, chpt. 4 presents the methodological approach to analyze such data-types using geostatistical models which include a time-series (year-month). These models are an extension of the more commonly known Delta-GLMM models which are sometimes used to estimate abundance indices from scientific surveys (for an introduction, see *Michalsen et al.*, 2013). To facilitate the use of the INLA/SPDE approach, as well as maximize computational time, we aggregated

the data using Dirichlet Tessellations (for details, see chpt. 4 and figure C.1). This spatial partitioning method allowed us to reduce the size of the dataset — particularly useful when dealing with large amount of zeros and large datasets, while maintaining the ability to model space with GRF rather than stratified areas. Moreover, aggregation was done according to data availability (where discard occurred), hence maximizing the use of the information collected and increasing the ability to capture finer-scale spatial autocorrelation processes.

These studies show novel approaches to modeling space and time in bycatch studies that are very different than the current *status quo* (see the Introduction of chpt. 2). By using nearby time and space information, Bayesian geostatistical models automatically estimate densities in un-sampled areas, and also fully account for all sources of uncertainties. Thus, it is possible to infer a probability of bycatch or discards across space and over time at a very fine resolution — precision much needed by fisheries management.

Moreover, the summary statistics of the GRF can provide additional biological information that would otherwise not be available. For example, the spatial correlation range, which defines the distance beyond which the spatial dependence between observations is unsubstantial, may indicate hotspots subject to different unknown underlying processes e.g., oceanographical, biological, fisheries. This may indicate that these hotspots could be managed differently, by using spatially explicit bycatch management, for example, specific gear restrictions in different regions, hence offering additional management flexibility. Chapter 2 discusses such a scenario.

Chapter 4 demonstrates that GRF included in the encounters (presence/absence) discard model, and the catch rates of positive catches (weight of discards) are proxies for species distribution and biomass concentration — which are information that are traditionally estimated from scientific research surveys. This approach is particularly powerful and interesting for species that are not well-sampled by traditional demersal surveys, these include pelagic and deep-water species. Chapter 4 shows examples of such information and meanwhile provides the first quantitative assessment of the distribution and biomass of some of the deep-water elasmobranch species occurring in the Northwest Atlantic and Arctic adjacent seas, notably black dogfish (*Centroscyllium fabricii*), Arctic skate (*Amblyraja hyperborea*), Jensens (*Amblyraja jenseni*), spinytail skate (*Bathyraja spinicauda*), and white skate (*Bathyraja spinosissima*).

7.4 Next Steps

In summary, this dissertation demonstrates the high spatiotemporal variability and inherent complexity of bycatch. It further supports the use of Bayesian geostatistical models to better analyze bycatch and provide new, finer-scale spatial information for fisheries management. The dissertation focused on elasmobranch species in Eastern Canadian fisheries to help prioritizing future conservation efforts in the region. It also shows the importance of conducting both smaller-scale, fishery-specific research to identify sensitive areas of biological importance and management opportunities, as well as larger-scale studies across fisheries to prioritize future research and conservation efforts.

Future research would be beneficial to investigate the effect of fishing on deep-water elasmobranch populations. In order to do so, better taxonomic species identifications, particularly for skate species, is a critical first step that needs to be addressed in Canadian waters. Spatial closure appear to be a promising management option to minimize porbeagle catch in the swordfish and tuna pelagic longline fishery. However, future work should evaluate plausible impacts, such as effort displacement, effects on targeted and other bycatch species, as well as socio-economic impacts. Additional research on how geostatistical models could account for uncertainties related to non-random sampling would also be highly useful.

The new statistical models and extensions (Excursion Functions) presented and used in this dissertation are providing powerful means to analyze, predict, and manage fisheries bycatch, or more generally, wildlife and natural resources in a rapidely changing environment. It is my belief that such tools will become omnipresent in fisheries sciences.

BIBLIOGRAPHY

Aalbers, S. A., D. Bernal, and C. A. Sepulveda, The functional role of the caudal fin in the feeding ecology of the common thresher shark Alopias vulpinus, *Journal of Fish Biology*, *76*, 1863–1868, 2010.

Afonso, A. S., F. H. Hazin, F. Carvalho, J. C. Pacheco, H. Hazin, D. W. Kerstetter, D. Murie, and G. H. Burgess, Fishing gear modifications to reduce elasmobranch mortality in pelagic and bottom longline fisheries off Northeast Brazil, *Fisheries Research*, *108*, 336–343, 2011.

Alverson, D., M. Freeberg, J. Pope, and S. Murawski, A global assessment of fisheries bycatch and discards, *Tech. rep.*, FAO Fisheries Technical Paper. No. 339, Rome, FAO, 1994.

Amante, C., and B. W. Eakins, ETOPO1 1 Arc-Minute Global Relief Model: Procedures, Data Sources and Analysis, *Tech. rep.*, NOAA Technical Memorandum NESDIS NGDC-24, 19 pp, March 2009., 2014.

Barlow, P. F., and J. Berkson, Evaluating methods for estimating rare events with zero-heavy data: a simulation model estimating sea turtle bycatch in the pelagic longline fishery, *Fishery Bulletin*, *110*, 344–360, 2012.

Baum, J. K., and B. Worm, Cascading top-down effects of changing oceanic predator abundances, *The Journal of animal ecology*, *78*, 699–714, 2009.

Baum, J. K., R. A. Myers, D. G. Kehler, B. Worm, S. J. Harley, and P. A. Doherty, Collapse and conservation of shark populations in the Northwest Atlantic, *Science*, *299*, 389–392, 2003.

Benoît, H., Estimated discards of winter skate (Leucoraja ocellata) in the southern Gulf of St. Lawrence, 1971-2004, 2006.

Benoît, H. P., and J. Allard, Can the data from at-sea observer surveys be used to make general inferences about catch composition and discards?, *Canadian Journal of Fisheries and Aquatic Sciences*, *66*, 2025–2039, 2009.

Benoît, H. P., D. P. Swain, M. Niles, S. LeBlanc, and L.-A. Davidson, Incidental catch amounts and potential post-release survival of winter skate (Leucoraja ocellata) captured in the scallop dredge fishery in the southern Gulf of St. Lawrence (2006-2008), 2010.

Bessell, P. R., L. Matthews, A. Smith-Palmer, O. Rotariu, N. J. C. Strachan, K. J. Forbes, J. M. Cowden, S. W. J. Reid, and G. T. Innocent, Geographic determinants of reported human Campylobacter infections in Scotland., *BMC public health*, *10*, 423, 2010.

Best, N., S. Richardson, and A. Thomson, A comparison of Bayesian spatial models for disease mapping, *Statistical Methods in Medical Research*, *14*, 35–59, 2005.

Bjorge, A., M. Skern-Mauritzen, and M. C. Rossman, Estimated bycatch of harbour porpoise (Phocoena phocoena) in two coastal gillnet fisheries in Norway, 20062008. Mitigation and implications for conservation, *Biological Conservation*, *161*, 164–173, 2013.

Bolin, D., and F. Lindgren, Excursion and contour uncertainty regions for latent Gaussian models, *Journal of the Royal Statistical Society: Series B (Statistical Methodology)*, 2014.

Bolten, A., H. Martins, E. Isidro, M. Santos, R. Ferreira, E. Bettencourt, A. Giga, A. Cruz, and K. Bjorndal, Experiment to Evaluate Gear Modication on Rates of Sea Turtle Bycatch in the Longline Fishery in the AzoresPhase 4. Final project report submitted to the US National Marine Fisheries Service, *Tech. rep.*, Archie Carr Center for Sea Turtle (ACCSTR), University of Florida, Gainesville, Florida, USA, 2005.

Borucinska, J., J. Martin, and G. Skomal, Peritonitis and pericarditis associated with gastric perforation by a retained fishing hook in a blue shark, *Journal of Aquatic Animal Health*, *13*, 347354, 2001.

Borucinska, J., N. Kohler, L. Natanson, and G. Skomal, Pathology associated with retained fishing hooks in blue sharks, Prionace glauca (L.), with implications for their conservation, *Journal of Fish Diseases*, *25*, 515–521, 2002.

Brander, K., Disappearance of common skate Raia batis from Irish Sea, *Nature*, *250*, 48–49, 1981.

Brazner, J., and J. Mcmillan, Loggerhead turtle (Caretta caretta) bycatch in Canadian pelagic longline fisheries: Relative importance in the western North Atlantic and opportunities for mitigation, *Fisheries Research*, *91*, 310–324, 2008.

Brill, R., P. Bushnell, L. Smith, C. Speaks, R. Sundaram, E. Stroud, and J. Wang, The repulsive and feeding-deterrent effects of electropositive metals on juvenile sandbar sharks (Carcharhinus plumbeus), *Fisheries Bulletin*, *107*, 298–307, 2009.

Brodziak, J., and W. A. Walsh, Model selection and multimodel inference for standardizing catch rates of bycatch species : a case study of oceanic whitetip shark in the Hawaii-based longline fishery, *Canadian Journal of Fisheries and Aquatic Sciences*, *1740*, 1723–1740, 2013.

Cambiè, G., N. Sánchez-Carnero, T. Mingozzi, R. Muiño, and J. Freire, Identifying and mapping local bycatch hotspots of loggerhead sea turtles using a GIS-based method: implications for conservation, *Marine Biology*, *160*, 653–665, 2012.

Cameletti, M., F. Lindgren, D. Simpson, and H. v. Rue, Spatio-temporal modeling of particulate matter concentration through the SPDE approach, *AStA Advances in Statistical Analysis*, 2012.

Camhi, M. D., S. V. Fordham, and S. L. Fowler, Domestic and international management for pelagic sharks, in *Sharks of the open ocean: biology, fisheries and conservation*, edited by M. D. Camhi, E. K. Pikitch, and E. A. Babcock, pp. 418–444, Blackwell Publishing, Oxford, UK, 2008.

Camhi, M. D., S. V. Valenti, S. V. Fordham, S. L. Fowler, and C. Gibson, The Conservation Status of Pelagic Sharks and Rays: Report of the IUCN Shark Specialist Group Pelagic Shark Red List Workshop. Biological Conservation. Vol. 4. Newbury, UK, April 2009. Newbury, UK.ks and Rays: Report of the IUCN Shark Specialist Group Pel, *Tech. Rep. 3*, IUCN Species Survival Commission Shark Specialist Group, Newbury, UK, 2009.

Campana, S., A. Fisk, and A. Klimley, Movements of Arctic and northwest Atlantic Greenland shark (Somniosus microcephalus) monitored with archival satellite pop-up tags suggest long-range migrations, *Deep Sea Res Part II*, 2013.

Campana, S., J. Gibson, J. Brazner, L. Marks, W. Joyce, J.-F. Gosselin, R. D. Kenney, P. Shelton, M. Simpson, and J. Lawson, Status of Basking sharks in Atlantic Canada, *Tech. rep.*, Canadian Science Advisory Secretariat, 2008.

Campana, S. E., and W. Joyce, Temperature and depth associations of porbeagle shark (Lamna nasus) in the northwest Atlantic, *Fisheries Oceanography*, *13*, 52–64, 2004.

Campana, S. E., L. Marks, and W. Joyce, The biology and fishery of shortfin mako sharks (Isurus oxyrinchus) in Atlantic Canadian waters, *Fisheries Research*, *73*, 341–352, 2005.

Campana, S. E., L. Marks, W. Joyce, and N. E. Kohler, Effects of recreational and commercial fishing on blue sharks (Prionace glauca) in Atlantic Canada , with inferences on the North Atlantic population, *Canadian Journal of Fisheries and Aquatic Sciences*, *63*, 670–682, 2006.

Campana, S. E., W. Joyce, and D. W. Kulka, Growth and reproduction of spiny dogfish off the eastern coast of Canada, including inferences on stock structure, in *Biology and management of dogfish sharks*, edited by V. F. Gallucci, G. A. McFarlane, and G. G. Bargmann, pp. 195–208, American Fisheries Society, Bethesda, Maryland, 2009a.

Campana, S. E., W. Joyce, and M. J. Manning, Bycatch and discard mortality in commercially caught blue sharks Prionace glauca assessed using archival satellite pop-up tags, *Marine Ecology Progress Series*, *387*, 241–253, 2009b.

Campana, S. E., J. Brading, and W. Joyce, Estimation of pelagic shark bycatch and associated mortality in Canadian Atlantic fisheries, *Tech. rep.*, CSAS. Canadian Science Advisory Secretariat Research Document 2011/067, 2011a.

Campana, S. E., A. Dorey, M. Fowler, W. Joyce, Z. Wang, D. Wright, and I. Yashayaev, Migration pathways, behavioural thermoregulation and overwintering grounds of blue sharks in the Northwest Atlantic., *PloS one*, *6*, 1–13, 2011b.

Campana, S. E., A. J. F. Gibson, M. Fowler, A. Dorey, and W. Joyce, Population dynamics of Northwest Atlantic porbeagle (Lamna nasus), with an assessment of status and projections for recovery, *Tech. rep.*, DFO Can. Sci. Advis. Sec. Res. Doc. 2012/096. iv + 84 p., 2012.

Carruthers, E. H., and B. Neis, Bycatch mitigation in context: Using qualitative interview data to improve assessment and mitigation in a data-rich fishery, *Biological Conservation*, *144*, 2289–2299, 2011.

Carruthers, E. H., D. C. Schneider, and J. D. Neilson, Estimating the odds of survival and identifying mitigation opportunities for common bycatch in pelagic longline fisheries, *Biological Conservation*, *142*, 2620–2630, 2009.

Carruthers, E. H., J. D. Neilson, and S. C. Smith, Overlooked bycatch mitigation opportunities in pelagic longline fisheries: Soak time and temperature effects on swordfish (Xiphias gladius) and blue shark (Prionace glauca) catch, *Fisheries Research*, *108*, 112–120, 2011.

Casey, J. M., and R. A. Myers, Near extinction of a large, widely distributed fish, *281*, 690–693, 1998.

Catchpole, T., and S. Cadrin, Bycatch and discards: from improved knowledge to mitigation programmes, *ICES Journal of Marine Science*, *71*, 1216–1218, 2014.

Cheung, W. W. L., R. Watson, and D. Pauly, Signature of ocean warming in global fisheries catch, *Nature*, *497*, 365–8, 2013.

Christiansen, J. r. S., C. W. Mecklenburg, and O. V. Karamushko, Arctic marine fishes and their fisheries in light of global change., *Global change biology*, *20*, 352–9, 2014.

Ciannelli, L., P. Fauchald, K. Chan, V. Agostini, and G. Dingsor, Spatial fisheries ecology: Recent progress and future prospects, *Journal of Marine Systems*, *71*, 223–236, 2008.

Clark, J. S., S. R. Carpenter, M. Barber, S. Collins, A. Dobson, J. a. Foley, D. M. Lodge, M. Pascual, R. Pielke, W. Pizer, C. Pringle, W. V. Reid, K. a. Rose, O. Sala, W. H. Schlesinger, D. H. Wall, and D. Wear, Ecological forecasts: an emerging imperative, *Science*, *293*, 657–60, 2001.

Clarke, S., E. Milner-Gulland, and T. Bjorndal, Perspective: Social, Economic and Regulatory Drivers of the Shark Fin Trade, *Marine Resource Econimics*, *22*, 305–327, 2007.

Clarke, S. C., S. J. Harley, S. D. Hoyle, and J. S. Rice, Population trends in Pacific Oceanic sharks and the utility of regulations on shark finning., *Conservation Biology*, *27*, 197–209, 2013.

Clarke, S. C., M. K. McAllister, E. J. Milner-Gulland, G. P. Kirkwood, C. G. J. Michielsens, D. J. Agnew, E. K. Pikitch, H. Nakano, M. S. Shivji, and E. Letters, Global estimates

of shark catches using trade records from commercial markets., *Ecology letters*, *9*, 1115–26, 2006.

Coelho, R., and K. Erzini, Effects of fishing methods on deep water shark species caught as by-catch off southern Portugal, *Hydrobiologia*, *606*, 187–193, 2008.

Collin, S., and D. Whitehead, The functional roles of passive electroreception in non-electric fishes, *Animal Biology*, *54*, 1–25, 2004.

Compagno, L., M. Dando, and S. Fowler, *A field guide to the sharks of the world*, Harper Collins, London, UK, 2005.

Cooke, S., and C. Suski, Are circle hooks and effective tool for conserving marine and freshwater recreational catch-and-release fisheries?, *Aquat. Conserv.: Mar. Freshw. Ecosyst.*, *14*, 299–326, 2004.

Corke, J., *Identification guide to sharks, skates, rays and chimaeras of Atlantic Canada*, WWF-Canada, 2012.

Cortés, E., F. Arocha, L. Beerkircher, F. Carvalho, A. A. Domingo, M. Heupel, H. Holtzhausen, M. N. Santos, M. Ribera, and C. Simpfendorfer, Ecological risk assessment of pelagic sharks caught in Atlantic pelagic longline fisheries, *Aquatic Living Resources*, *23*, 25–34, 2009.

Cosandey-Godin, A., and B. Worm, Keeping the lead: How to strengthen shark conservation and management policies in Canada, *Marine Policy*, *34*, 995–1001, 2010.

Cosandey-Godin, A., J. K. Carlson, and V. Burgener, The effect of circle hooks on shark catchability and at-vessel mortality rates in longlines fisheries, *Bulletin of Marine Science*, *88*, 469–483, 2012.

Cosandey-Godin, A., T. Wimmer, J. H. Wang, and B. Worm, No effect from rare-earth metal deterrent on shark bycatch in a commercial pelagic longline trial, *Fisheries Research*, *143*, 131–135, 2013.

Cosandey-Godin, A., E. Teixeira Krainski, B. Worm, and J. Mills Flemming, Applying Bayesian spatio-temporal models to fisheries bycatch in the Canadian Arctic, *Canadian Journal of Fisheries and Aquatic Sciences*, 2014.

COSEWIC, COSEWIC assessment and status report on the winter skate Leucoraja ocellata in Canada, *Tech. rep.*, Committee on the Status of Endangered Wildlife in Canada., Ottawa, 2005.

COSEWIC, COSEWIC assessment and status report on the Barndoor Skate Dipturus laevis in Canada, *Tech. rep.*, Committee on the Status of Endangered Wildlife in Canada, Ottawa, 2010.

COSEWIC, COSEWIC assessment and status report on the Thorny Skate Amblyraja radiata in Canada, *Tech. rep.*, Committee on the Status of Endangered Wildlife in Canada, Ottawa, 2012a.

COSEWIC, COSEWIC assessment and status report on the Smooth Skate Malacoraja senta in Canada, *Tech. rep.*, Committee on the Status of Endangered Wildlife in Canada, Ottawa, 2012b.

Coutré, K., and T. Gedamke, Indication of density-dependent changes in growth and maturity of the Barndoor skate on Georges bank, *Marine and Coastal . . .*, *5*, 260–269, 2013.

Cressie, N., C. a. Calder, J. S. Clark, J. M. Ver Hoef, and C. K. Wikle, Accounting for uncertainty in ecological analysis: the strengths and limitations of hierarchical statistical modeling., *Ecological applications : a publication of the Ecological Society of America*, *19*, 553–70, 2009.

Croxall, J. P., The role of science and advocacy in the conservation of Southern Ocean albatrosses at sea, *Bird Conservation International*, *18*, 2008.

Curran, D., and K. Bigelow, Effects of circle hooks on pelagic catches in the Hawaii-based tuna longline fishery, *Fisheries Research*, *109*, 265–275, 2011.

Davies, R., S. Cripps, A. Nickson, and G. Porter, Defining and estimating global marine fisheries bycatch, *Marine Policy*, *33*, 661–672, 2009.

Davis, B., D. L. VanderZwaag, A. Cosandey-Godin, N. E. Hussey, S. T. Kessel, and B. Worm, The Conservation of the Greenland Shark (Somniosus microcephalus): Setting Scientific, Law, and Policy Coordinates for Avoiding a Species at Risk, *Journal of International Wildlife Law & Policy*, *16*, 1:30, 2013.

DerSimonian, R., and N. Laird, Meta-analysis in clinical trials., *Controlled clinical trials*, *7*, 177–88, 1986.

DFO, Fishery Management Plan Greenland Halibut. NAFO Subarea 0 2006-2008, 2005.

DFO, Development of a Closed Area in NAFO 0A to protect Narwhal Over-Wintering Grounds, including Deep-sea Corals., 2007a.

DFO, National plan of action for the conservation and management of sharks, *Tech. rep.*, 2007b.

DFO, Northern Shrimp (SFAs) 0-7 and the Flemish Cap, *Tech. rep.*, 2010.

DFO, Canadian Atlantic Swordfish and Other Tunas Integrated Fishery Management Plan Atlantic Canada, 2013.

DFO, Offshore Ecologically and Biologically Significant Areas in the Scotian Shelf Bioregion, *Tech. Rep. August*, DFO Can. Sci. A dvis. Sec. Sci. Advis. Rep. 2014/041, 2014a.

DFO, Integrated Fishery Management Plan Greenland Halibut (Reinhardtius hippoglossoides) Northwest Atlantic Fisheries Organization Subarea 0 Effective 2014, 2014b.

DFO, Assessment of Spiny dogfish in the Northwest Atlantic, 2014c.

Diaz, G. A., and J. E. Serafy, Longline-caught blue shark (Prionace glauca): factors affecting the numbers available for live release, *Fisheries Bulletin*, *103*, 720–724, 2005.

Diggle, P., and P. J. Ribeiro, *Model-based Geostatistics*, springer ed., New York, 2007.

Douvere, F., The importance of marine spatial planning in advancing ecosystem-based sea use management, *Marine Policy*, *32*, 762–771, 2008.

Dulvy, N. K., J. D. Metcalfe, J. Glanville, M. G. Pawson, and J. D. Reynolds, Fishery stability, local extinctions, and shifts in community structure in skates, *Conservation Biology*, *14*, 283–293, 2000.

Dulvy, N. K., Y. Sadovy, and J. D. Reynolds, Extinction vulnerability in marine populations, *Fish and Fisheries*, *4*, 25–64, 2003.

Dulvy, N. K., J. K. Baum, S. Clarke, L. J. V. Compagno, E. Cortes, A. Domingo, S. Fordham, S. L. Fowler, F. P. Malcom, C. Gibson, J. Martinez, J. A. Musick, A. Soldo, J. D. Stevens, and S. Valenti, You can swim but you cant hide : the global status and conservation of oceanic pelagic sharks and rays, *Aquatic Conserv: Mar. Freshw. Ecosyst.*, *482*, 459–482, 2008.

Dulvy, N. K., S. L. Fowler, J. a. Musick, R. D. Cavanagh, P. M. Kyne, L. R. Harrison, J. K. Carlson, L. N. Davidson, S. V. Fordham, M. P. Francis, C. M. Pollock, C. a. Simpfendorfer, G. H. Burgess, K. E. Carpenter, L. J. Compagno, D. a. Ebert, C. Gibson, M. R. Heupel, S. R. Livingstone, J. C. Sanciangco, J. D. Stevens, S. Valenti, and W. T. White, Extinction risk and conservation of the world's sharks and rays, *eLife*, *3*, 1–25, 2014.

Dunn, D. C., A. M. Boustany, and P. N. Halpin, Spatio-temporal management of fisheries to reduce by-catch and increase fishing selectivity, *Fish and Fisheries*, *12*, 110–119, 2011.

Falterman, B., and J. E. Graves, A preliminary comparison of the relative mortality and hooking efficiency of circle and straight shank ("J") hooks used in the pelagic longline industry, *Americain Fisheries Society Symposium*, *30*, 80–87, 2002.

FAO, International plan of action for the conservation and management of sharks, 1999.

FAO, *Report of the FAO Expert Consultation on the Implementation of the FAO International Plan of Action for the Conservation and Management of Sharks. Rome, 68 December 2005.*, vol. 795, FAO Fisheries Report. No. 795. Rome, FAO., 2006.

FAO, Report of the Technical Workshop on the Status, Limitations and Opportunities for Improving the Monitoring of Shark Fisheries and Trade. Rome, 36 November 2008, *Tech. Rep. 897*, FAO Fisheries and Aquaculture Report. No. 897, Rome, Italy, 2009a.

FAO, Fishing operations. 2. Best practices to reduce incidental catch of seabirds in capture fisheries., in *FAO Technical Guidelines for Responsible Fisheries*, chap. No. 1, Sip, p. 49, Rome, FAO, 2009b.

FAO, Guidelines to reduce sea turtle mortality in fishing operations, *Tech. rep.*, FAO. Fisheries Department, Rome, 2009c.

FAO, *International Guidelines on Bycatch Management and Reduction of Discards*, Food and Agriculture Organization of the United Nations., Rome, 2011.

FAO, *The State of World Fisheries and Aquaculture 2014*, Rome, 2014.

Fed. Reg., Fisheries of the Caribbean, Gulf, & South Atlantic., *Tech. rep.*, 2011.

Feekings, J., V. Bartolino, N. Madsen, and T. Catchpole, Fishery discards: factors affecting their variability within a demersal trawl fishery, *PloS one*, *7*, e36409, 2012.

Ferretti, F., R. A. Myers, F. Serena, and H. K. Lotze, Loss of large predatory sharks from the Mediterranean Sea, *Conservation Biology*, *22*, 952–964, 2008.

Ferretti, F., B. Worm, G. L. Britten, M. R. Heithaus, and H. K. Lotze, Patterns and ecosystem consequences of shark declines in the ocean, *Ecology letters*, 2010.

Findlay, J. D., C. M. Cross, and A. G. Bodsworth, Marlin fisheries management in Australia, *Marine and Freshwater Research*, *54*, 535–543, 2003.

Frank, K., and J. Simon, An evaluation of the Emerald/Western Bank juvenile haddock closed area, *Tech. rep.*, Canadian Stock Assessment Secretariat Research Document 98/53, 1998.

Frisk, M. G., T. J. Miller, and M. J. Fogarty, Estimation and analysis of biological parameters in elasmobranch fishes: a comparative life history study, *Canadian Journal of Fisheries and Aquatic Sciences*, *58*, 969–981, 2001.

Galeana-Villasenor, I., F. Galvan-Magana, and R. Gomez-Aguilar, Influence of hook type and fishing depth on longline catches of sharks and other pelagic species in the northwest Mexican Pacific, *Revista de Biología Marina y Oceanografía*, *43*, 99–110, 2008.

Galeana-Villasenor, I., F. Galvan-Magana, and H. Santana-Hernandez, Fishing by hooks in longliners from the Mexican Pacific Ocean: effects in the catch rate and weight of sharks and other species, *Revista de Biología Marina y Oceanografía*, *44*, 163–172, 2009.

Garcia, S. M., A. Zerbi, C. Aliaume, T. Do Chi, and G. Lasserre, *The ecosystem approach to fisheries. Issues, terminology, principles, institutional foundations, implementation and outlook*, fao fisher ed., 2003.

Garcia, V. B., L. O. Lucifora, and R. A. Myers, The importance of habitat and life history to extinction risk in sharks, skates, rays and chimaeras, *Proceedings. Biological sciences / The Royal Society*, *275*, 83–9, 2008.

Gardner, B., P. Sullivan, S. Epperly, and S. Morreale, Hierarchical modeling of bycatch rates of sea turtles in the western North Atlantic, *Endangered Species Research*, *5*, 279–289, 2008.

Gavaris, S., K. J. Clark, A. R. Hanke, C. F. Purchase, and J. Gale, Overview of Discards from Canadian Commercial Fisheries in NAFO Divisions 4V , 4W , 4X , 5Y and 5Z for 2002-2006 Canadian Technical Report of Fisheries and Aquatic Sciences 2873, *Tech. rep.*, 2010.

Gilman, E., P. Dalzell, and S. Martin, Fleet communication to abate fisheries bycatch, *Marine Policy*, *30*, 360–366, 2006a.

Gilman, E., E. Zollett, S. Beverly, H. Nakano, K. Davis, D. Shiode, P. Dalzell, and I. Kinan, Reducing sea turtle by-catch in pelagic longline fisheries, *Fisheries Research*, pp. 1–22, 2006b.

Gilman, E., D. Kobayashi, T. Swenarton, N. Brothers, P. Dalzell, and I. Kinan-kelly, Reducing sea turtle interactions in the Hawaii-based longline swordfish fishery, *Biological Conservation*, *139*, 19–28, 2007.

Gilman, E., S. Clarke, N. Brothers, J. Alfaroshigueto, J. Mandelman, J. Mangel, S. Petersen, S. Piovano, N. Thomson, and P. Dalzell, Shark interactions in pelagic longline fisheries, *Marine Policy*, *32*, 1–18, 2008.

Gore, M., D. Rowat, J. Hall, F. R. Gell, and R. F. Ormond, Transatlantic migration and deep mid-ocean diving by basking shark., *Biology letters*, *4*, 395–8, 2008.

Grantham, H. S., S. Petersen, and H. P. Possingham, Reducing bycatch in the South African pelagic longline fishery: the utility of different approaches to fisheries closures, *Endangered Species Research*, *5*, 291–299, 2008.

Haas, S. E., M. B. Hooten, D. M. Rizzo, and R. K. Meentemeyer, Forest species diversity reduces disease risk in a generalist plant pathogen invasion., *Ecology letters*, *14*, 1108–16, 2011.

Hale, L. F., S. J. B. Gulak, and A. M. Napier, Characterization of the shark bottom longline fishery: 2010, 2011.

Hall, M. a., D. L. Alverson, and K. I. Metuzals, By-Catch: Problems and Solutions, *Marine Pollution Bulletin*, *41*, 204–219, 2000.

Halliday, R. G., and A. Pinhorn, The delimitation of fishing areas in the Northwest Atlantic, *Journal of Northwest Atlantic Fishery Science*, *10*, 1–51, 1962.

Halpern, B. S., S. Walbridge, K. a. Selkoe, C. V. Kappel, F. Micheli, C. D'Agrosa, J. F. Bruno, K. S. Casey, C. Ebert, H. E. Fox, R. Fujita, D. Heinemann, H. S. Lenihan, E. M. P. Madin, M. T. Perry, E. R. Selig, M. Spalding, R. Steneck, and R. Watson, A global map of human impact on marine ecosystems., *Science*, *319*, 948–52, 2008.

Harden, L. A., and A. S. Willard, Using spatial and behavioral data to evaluate the seasonal bycatch risk of diamondback terrapins Malaclemys terrapin in crab pots, *Marine Ecology Progress Series*, *467*, 207–217, 2012.

Harley, S. J., and R. a. Myers, Hierarchical Bayesian models of length-specific catchability of research trawl surveys, *Canadian Journal of Fisheries and Aquatic Sciences*, *58*, 1569–1584, 2001.

He, X., K. A. Bigelow, and C. H. Boggs, Cluster analysis of longline sets and fishing strategies within the Hawaii-based fishery, *Fish. Res.*, *31*, 147–158, 1997.

Held, L., B. Schrodle, and H. v. Rue, Posterior and Cross-validatory Predictive Checks: A Comparison of MCMC and INLA, in *Statistical Modelling and Regression Structures*, pp. 111–131, 2010.

Higgins, J. P. T., S. G. Thompson, J. J. Deeks, and D. G. Altman, Measuring inconsistency in meta-analyses., *BMJ (Clinical research ed.)*, *327*, 557–60, 2003.

Hobday, A. J., and K. Hartmann, Near real-time spatial management based on habitat predictions for a longline bycatch species, *Fisheries Management and Ecology*, *13*, 365–380, 2006.

Hobday, A. J., J. R. Hartog, T. Timmiss, and J. Fielding, Dynamic spatial zoning to manage southern bluefin tuna (Thunnus maccoyii) capture in a multi-species longline fishery, *Fisheries Oceanographyher*, *19*, 243–253, 2010.

Hoenig, J. M., and S. H. Gruber, Life-history patterns in the elasmobranchs: Implications for fisheries management, in *Elasmobranchs as living resources: Advances in the biology, ecology, systematics, and the status of the fisheries*, edited by H. L. Pratt Jr, S. H. Gruber, and T. Taniuchi, NOAA Technical Report 90, 1990.

Holand, A. M., I. Steinsland, S. Martino, and H. Jensen, Animal models and integrated nested Laplace approximations, *G3*, *3*, 1241–51, 2013.

Huang, H.-W., and Y.-M. Yeh, Impact of Taiwanese distant water longline fisheries on the Pacific seabirds: finding hotspots on the high seas, *Animal Conservation*, *14*, 562–574, 2011.

Huse, I., S. Lokkeborg, and V. Soldal, Relative selectivity in trawl, longline and gillnet fisheries for cod and haddock, *ICES Journal of Marine Science*, *57*, 1271–1282, 2000.

Hussey, N. E., A. Cosandey-Godin, R. P. Walter, K. J. Hedges, M. VanGerwen-Toyne, A. N. Barkley, S. T. Kessel, and A. T. Fisk, Juvenile Greenland sharks Somniosus microcephalus (Bloch & Schneider, 1801) in the Canadian Arctic, *Polar Biology*, 2014.

Hutchings, J. a., and D. J. Fraser, The nature of fisheries- and farming-induced evolution., *Molecular ecology*, *17*, 294–313, 2008.

Hutchings, J. A., R. A. Myers, V. B. Garcia, L. O. Lucifora, and A. Kuparinen, Life-history correlates of extinction risk and recovery potential, *Ecological Applications*, *22*, 1061–1067, 2012.

Hutchinson, M., J. H. Wang, Y. Swimmer, K. Holland, S. Kohin, H. Dewar, J. Wraith, R. Vetter, C. Heberer, and J. Martinez, The effects of a lanthanide metal alloy on shark catch rates, *Fisheries Research*, *131-133*, 45–51, 2012.

ICCAT, Report of the 2008 shark stock assessments meeting, *Collect. Vol. Sci. Pap. ICCAT*, *64*, 1343–1491, 2009.

ICCAT, 2010 Shortfin mako stock assessment and ecological risk assessment meeting, *Tech. rep.*, Olhao, Portugal, 2012.

Illian, J. B., S. Martino, S. H. Sø rbye, J. B. Gallego-Fernández, M. Zunzunegui, M. P. Esquivias, and J. M. J. Travis, Fitting complex ecological point process models with integrated nested Laplace approximation, *Methods in Ecology and Evolution*, pp. n/a–n/a, 2013.

Ingram, W., T. Henwood, M. Grace, L. Jones, W. Driggers, and K. Mitchell, Catch rates, distribution, and size composition of large coastal sharks collected during NOAA Fisheries bottom longline surveys from the U.S. Gulf of Mexico and U.S. Atlantic Ocean, *Tech. rep.*, National Marine Fisheries Service, Southeast Fisheries Science Center, Pascagoula, Mississippi SEDAR 11 LCS05/06-DW-27, 2005.

Jackson, J. B. C., M. X. Kirby, W. H. Berger, K. A. Bjorndal, L. W. Botsford, B. J. Bourque, R. H. Bradbury, R. Cooke, J. Erlandson, J. A. Estes, T. P. Hughes, S. Kidwell, C. B. Lange, H. S. Lenihan, M. Pandolfi, C. H. Peterson, R. S. Steneck, M. J. Tegner, R. R. Warner, and J. M. Pandolfi, Historical overfishing and the recent collapse of coastal ecosystems, *Science*, *293*, 629–638, 2001.

Jannot, J. E., and D. S. Holland, Identifying ecological and fishing drivers of bycatch in a U.S. groundfish fishery, *Ecological Applications*, *23*, 1645–1658, 2013.

Jensen, O. P., S. Ortega-garcia, S. J. D. Martell, R. N. M. Ahrens, M. L. Domeier, C. J. Walters, and J. F. Kitchell, Local management of a highly migratory species : The effects of long-line closures and recreational catch-and-release for Baja California striped marlin fisheries, *Progress in Oceanography*, *86*, 176–186, 2010.

Jiménez, S., A. Domingo, and A. Brazeiro, Seabird bycatch in the Southwest Atlantic: interaction with the Uruguayan pelagic longline fishery, *Polar Biology*, *32*, 187–196, 2009.

Johnson, D. S., J. M. London, and C. E. Kuhn, Bayesian Inference for Animal Space Use and Other Movement Metrics, *Journal of Agricultural, Biological, and Environmental Statistics*, *16*, 357–370, 2011.

Jorgensen, O., C. Hvingel, P. Moller, and M. Treble, Identification and mapping of bottom fish assemblages in Davis Strait and southern, *Canadian Journal of Fisheries and Aquatic Sciences*, *62*, 1833–1852, 2005.

Kaimmer, S., and A. W. Stoner, Field investigation of rare-earth metal as a deterrent to spiny dogfish in the Pacific halibut fishery, *Fisheries Research*, *94*, 43–47, 2008.

Kajiura, S. M., A. D. Cornett, and K. E. Yopak, Sensory adaptations to the environment: electroreceptors as a case study, in *Sharks and their relatives II Biodiversity, adaptive physiology, and conservation*, edited by J. C. Carrier, J. A. Musick, and M. R. Heithaus, pp. 393–429, Taylor & Francis Group, LLC, 2010.

Kalmijn, A. J., Electric sense of sharks and rays, *Journal of Experimental Biology*, *55*, 371–383, 1971.

Keefe, C. E. O., S. X. Cadrin, and K. D. E. Stokesbury, Evaluating effectiveness of time/area closures, quotas/caps, and fleet communications to reduce fisheries bycatch, *ICES Journal of Marine Science*, *71*, 1286–1297, 2014.

Kelleher, K., Discards in the worlds marine fisheries. An update., 2005.

Kelly, J. T., and J. M. Hanson, Maturity, size at age and predator-prey relationships of winter skate Leucoraja ocellata in the southern Gulf of St Lawrence: Potentially an undescribed endemic facing extirpation, *Journal of Fish Biology*, *82*, 959–978, 2013.

Kerstetter, D., and J. Graves, Effects of circle versus J-style hooks on target and non-target species in a pelagic longline fishery, *Fisheries Research*, *80*, 239–250, 2006.

Kim, S.-S., D.-Y. Moon, D.-H. An, and J.-R. Koh, Comparison of circle hook and J hook catch rate for target and bycatch species taken in the Korean tuna longline fishery, *Tech. Rep. 4*, Paper presented at the second regular session scientific committee. Western and Central Pacific Fisheries Commission. WCPFC-SC2-2006/EB WP-12, 2006.

Kim, S.-S., D.-Y. Moon, D.-H. An, and J.-R. Koh, Comparison of circle hook and J hook catch rate for target and bycatch species taken in the Korean tuna longline fishery during 2005-2006, Paper presented at the Third Regular Session of the Scientific Committee, 13-24 August 2007 Honolulu, United States. Western and Central Pacific Fisheries Commission WCPFC-SC3-EB SWG/WP-11, 2007.

Kirby, D., and P. Ward, Standards for the effective management of fisheries bycatch, *Marine Policy*, *44*, 419–426, 2014.

Kulka, D., and D. Waldron, The Atlantic observer programs a discussion of sampling from commercial catches at sea, in *Sampling commercial catches of marine fish and invertebrate*, edited by W. Doubleday and D. Rivard, pp. pp. 255–262, Can. Spec. Publ. Fish. Aquat. Sci. No. 66., 1983.

Kulka, D., D. Swain, M. Simpson, C. Miri, J. Simon, J. Gauthier, R. McPhie, J. Sulikowski, and L. Hamilton, Distribution, abundance, and life history of Malacoraja senta (Smooth Skate) in Canadian Atlantic waters with reference to its global distribution, *Tech. rep.*, Canadian Science Advisory Secretariat. Research Document 2006/093, 2006.

Kulka, D. W., K. Frank, and J. Simon, Barndoor skate in the Northwest Atlantic off Canada: Distribution in relation to temperature and depth based on commercial fisheries data, 2002.

Kyne, P., and C. Simpfendorfer, Deepwater chondrichthyans, in *Sharks and Their Relatives II*, Marine Biology, pp. 37–113, CRC Press, 2010.

Kyne, P. M., J. K. Carlson, D. A. Ebert, S. V. Fordham, J. J. Bizzarro, R. T. Graham, D. W. Kulka, E. E. Tewes, and L. R. Harrison, The conservation status of North American, Central American, and Caribbean chondrichthyans, *Tech. rep.*, Vancouver, Canada, 2012.

Law, R., Fishing, selection, and phenotypic evolution, *ICES Journal of Marine Science*, *57*, 659–668, 2000.

Lewison, R. L., L. B. Crowder, A. J. Read, and S. A. Freeman, Understanding impacts of fisheries bycatch on marine megafauna, *Trends in Ecology and Evolution*, *19*, 598–604, 2004a.

Lewison, R. L., S. A. Freeman, and L. B. Crowder, Quantifying the effects of fisheries on threatened species: the impact of pelagic longlines on loggerhead and leatherback sea turtles, *Ecology Letters*, *7*, 221–231, 2004b.

Lewison, R. L., L. B. Crowder, B. P. Wallace, J. E. Moore, T. Cox, R. Zydelis, S. McDonald, A. DiMatteo, D. C. Dunn, C. Y. Kot, R. Bjorkland, S. Kelez, C. Soykan, K. R. Stewart, M. Sims, A. Boustany, A. J. Read, P. Halpin, W. J. Nichols, and C. Safina, Global patterns of marine mammal, seabird, and sea turtle bycatch reveal taxa-specific and cumulative megafauna hotspots., *Proceedings of the National Academy of Sciences of the United States of America*, *111*, 5271–6, 2014.

Li, Y., P. Brown, H. v. Rue, M. Al-Maini, and P. Fortin, Spatial modelling of lupus incidence over 40 years, *Journal of the Royal Statistical Society: Series C (Applied Statistics)*, pp. 99–115, 2012.

Lindgren, F., Continuous Domain Spatial Models in R-INLA, pp. 1–8, 2013.

Lindgren, F., and H. v. Rue, Bayesian Spatial and Spatio-temporal Modelling with R-INLA, *Journal of Statistical Software*, p. In press, 2013.

Lindgren, F., H. v. Rue, and J. Lindstrom, An explicit link between Gaussian fields and Gaussian Markov random fields: the stochastic partial differential equation approach, *Journal of the Royal Statistical Society: Series B (Statistical Methodology)*, *73*, 423–498, 2011.

Lotze, H. K., and B. Worm, Historical baselines for large marine animals., *Trends in ecology & evolution*, *24*, 254–62, 2009.

Lotze, H. K., M. Coll, and J. a. Dunne, Historical Changes in Marine Resources, Food-web Structure and Ecosystem Functioning in the Adriatic Sea, Mediterranean, *Ecosystems*, *14*, 198–222, 2010.

Machias, A., V. Vassilopoulou, D. Vatsos, P. Bekas, A. Kallianiotis, C. Papaconstantinou, and N. Tsimenides, Bottom trawl discards in the northeastern Mediterranean Sea, *Fisheries Research*, *53*, 181–195, 2001.

MacNeil, M., B. McMeans, N. Hussey, P. Vecsei, J. Svavarsson, K. M. Kovacs, C. Lydersen, M. Treble, G. B. Skomal, M. Ramsey, and A. Fisk, Biology of the Greenland shark Somniosus microcephalus, *Journal of fish biology*, *80*, 991–1018, 2012.

Madsen, N., Selectivity of fishing gears used in the Baltic Sea cod fishery, *Reviews in Fish Biology and Fisheries*, *17*, 517–544, 2007.

Mandelman, J. W., P. W. Cooper, T. B. Werner, and K. M. Lagueux, Shark bycatch and depredation in the US Atlantic pelagic longline fishery, *Reviews in Fish Biology and Fisheries*, *18*, 427–442, 2008.

Manire, C., R. Hueter, E. Hull, and R. Spieler, Serological Changes Associated with Gill-Net Capture and Restraint in Three Species of Sharks, *Transactions of the American Fisheries Society*, *130*, 1038–1048, 2001.

Martin, T. G., B. a. Wintle, J. R. Rhodes, P. M. Kuhnert, S. a. Field, S. J. Low-Choy, A. J. Tyre, and H. P. Possingham, Zero tolerance ecology: improving ecological inference by modelling the source of zero observations., *Ecology letters*, *8*, 1235–46, 2005.

Martins, T. G., D. Simpson, F. Lindgren, and H. v. Rue, Bayesian computing with INLA: New features, *Computational Statistics & Data Analysis*, *67*, 68–83, 2013.

Matsuno, K., A. Yamaguchi, and I. Imai, Mediterranean fishery discards: review of the existing knowledge, *ICES Journal of Marine Science*, *69*, 1205–1217, 2012.

Maunder, M., and A. Punt, Standardizing catch and effort data: a review of recent approaches, *Fisheries Research*, *70*, 141–159, 2004.

McClellan, C. M., A. J. Read, B. A. Price, W. M. Cluse, and M. H. Godfrey, Using telemetry to mitigate the bycatch of long-lived marine vertebrates., *Ecological Applications*, *19*, 1660–71, 2009.

McEachran, J., D. Boesch, and J. Musick, Food division within two Sympatric species-pairs of skate (Pisces: Rajidae), *Marine Biology*, *35*, 301–317, 1976.

Megalofonou, P., D. Damalas, M. Deflorio, and G. De Metrio, Modeling environmental, spatial, temporal, and operational effects on blue shark by-catches in the Mediterranean long-line fishery, *Journal of Applied Ichthyology*, *25*, 47–55, 2009.

Michalsen, K., J. Burgos, D. Righton, P. Steingrund, and T. Jakobsen, Geostatistical delta-generalized linear mixed models improve precision for estimated abundance indices forWest Coast groundfishe, *ICES Journal of Marine Science*, 2013.

Minami, M., C. E. Lennert-Cody, W. Gao, and M. Roman-Verdesoto, Modeling shark by-catch : The zero-inflated negative binomial regression model with smoothing, *Fisheries Research*, *84*, 210–221, 2007.

Moore, J. E., and A. J. Read, A Bayesian uncertainty analysis of cetacean demography and bycatch mortality using age-at-death data., *Ecological Applications*, *18*, 1914–31, 2008.

Moore, J. E., K. a. Curtis, R. L. Lewison, P. W. Dillingham, J. M. Cope, S. V. Fordham, S. S. Heppell, S. a. Pardo, C. a. Simpfendorfer, G. N. Tuck, and S. Zhou, Evaluating sustainability of fisheries bycatch mortality for marine megafauna: a review of conservation reference points for data-limited populations, *Environmental Conservation*, *40*, 329–344, 2013.

Morato, T., R. Watson, T. Pitcher, and D. Pauly, Fishing down the deep, *Fish and Fisheries*, *7*, 24–34, 2006.

Muñoz, F., M. G. Pennino, D. Conesa, A. López-Quílez, and J. M. Bellido, Estimation and prediction of the spatial occurrence of fish species using Bayesian latent Gaussian models, *Stochastic Environmental Research and Risk Assessment*, *27*, 1171–1180, 2012.

Murray, K. T., and C. D. Orphanides, Estimating the risk of loggerhead turtle Caretta caretta bycatch in the US mid-Atlantic using fishery-independent and -dependent data, *Marine Ecology Progress Series*, *477*, 259–270, 2013.

Murray, K. T., A. J. Read, and A. R. Solow, The use of area/time closures to reduce bycatches of harbour porpoises: lesson from the Gulf of Maine sink gillnet fishery, *Journal of Cetacean Research and Management*, *2*, 135–141, 2001.

Murray, R., Electrical sensitivity of the ampullae of Lorenzini, *Nature*, *187*, 957, 1960.

Musenge, E., T. F. Chirwa, K. Kahn, and P. Vounatsou, Bayesian analysis of zero inflated spatiotemporal HIV/TB child mortality data through the INLA and SPDE approaches: Applied to data observed between 1992 and 2010 in rural North East South Africa, *International Journal of Applied Earth Observation and Geoinformation*, pp. 1–13, 2012.

Musick, J., G. Burgess, G. Cailliet, M. Camhi, and S. Fordham, Management of sharks and their relatives (Elasmobranchii), *Fisheries*, *25*, 9–13, 2000.

Musick, J. A., Life in the slow lane: ecology and conservation of long-lived marine animals, *American Fisheries Society Symposium*, *23*, 1–10, 1999.

Myers, R. a., and B. Worm, Rapid worldwide depletion of predatory fish communities., *Nature*, *423*, 280–3, 2003.

Nakano, H., H. Matsunaga, H. Okamoto, and M. Okazaki, Acoustic tracking of bigeye thresher shark Alopias superciliosus in the eastern Pacific Ocean, *Marine Ecology Progress Series*, *265*, 255–261, 2003.

New England Fishery Management Council, 2012-2013 Northeast Skate Complex Specifications Environmental Assessment Regulatory Impact Review and Initial Regulatory Flexibility Analysis Northeast Skate Complex, *Tech. rep.*, 2012.

O'Connell, C. P., P. He, J. Joyce, E. M. Stroud, and P. H. Rice, Effects of the SMART (Selective Magnetic and Repellent-Treated) hook on spiny dogfish catch in a longline experiment in the Gulf of Maine, *Ocean & Coastal Management*, *97*, 38–43, 2014.

Okabe, A., B. Boots, and K. Sugihara, *Spatial Tessellations: Concepts and Applications of Voronoi Diagrams, 2nd ed*, new york: ed., 2000.

Ordines, F., E. Massuti, B. Guijarro, and R. Mas, Square mesh codend in a multi-species trawl fishery of the western Mediterranean: effects on catch composition, yield, size selectivity and discards, *Aquatic Living Resources*, *19*, 329–338, 2006.

Orphanides, C. D., Protected Species Bycatch Estimating Approaches: Estimating Harbor Porpoise Bycatch in U.S. Northwestern Atlantic Gillnet Fisheries, *Journal of Northwest Atlantic Fishery Science*, *42*, 55–76, 2010.

Ortiz, M., and F. Arocha, Alternative error distribution models for standardization of catch rates of non-target species from a pelagic longline fishery: billfish species in the Venezuelan tuna longline fishery, *Fisheries Research*, *70*, 275–297, 2004.

OConnell, C. P., D. C. Abel, P. H. Rice, E. M. Stroud, and N. C. Simuro, Responses of the southern stingray (Dasyatis americana) and the nurse shark (Ginglymostoma cirratum) to permanent magnets, *Marine and Freshwater Behaviour and Physiology*, *43*, 63–73, 2010.

OConnell, C. P., D. C. Abel, S. H. Gruber, E. M. Stroud, and P. H. Rice, Response of juvenile lemon sharks, Negaprion brevirostris, to a magnetic barrier simulating a beach net, *Ocean & Coastal Management*, *54*, 225–230, 2011.

Pacheco, J., D. Kerstetter, F. Hazin, H. Hazin, R. Segundo, J. Graves, F. Carvalho, and P. Travassos, A comparison of circle hook and J hook performance in a western equatorial Atlantic Ocean pelagic longline fishery, *Fisheries Research*, *107*, 39–45, 2011.

Paul, S. D., and J. D. Neilson, An exploration of targeting variables in the Canadian swordfish longline CPUE, *Coll. Vol. Sci. Pap.*, *65*, 124–134, 2009.

Pauly, D., Fishing Down Marine Food Webs, *Science*, *279*, 860–863, 1998.

Pauly, D., V. Christensen, S. Guénette, T. J. Pitcher, U. R. Sumaila, C. J. Walters, R. Watson, and D. Zeller, Towards sustainability in world fisheries, *Nature*, *418*, 689–695, 2002.

Pauly, D., R. Watson, and J. Alder, Global trends in world fisheries: impacts on marine ecosystems and food security., *Philosophical transactions of the Royal Society of London. Series B, Biological sciences*, *360*, 5–12, 2005.

Peklova, I., N. Hussey, K. Hedges, M. Treble, and A. Fisk, Movement, depth and temperature preferences of an important bycatch species, Arctic skate Amblyraja hyperborea, in Cumberland Sound, Canadian Arctic, *Endangered Species Research*, *23*, 229–240, 2014.

Petersen, S. l., M. b. Honig, P. G. Ryan, L. G. Underhill, and L. J. V. Compagno, Pelagic shark bycatch in the tuna- and swordfish-directed longline fishery off southern Africa, *African Journal of Marine Science*, *31*, 215–225, 2009.

Pezzack, D. S., C. M. Frail, P. Lawton, D. A. Robichaud, and M. B. Strong, Update on stock status of American Lobster, Homarus americanus, Lobster Fishing Area 34, *Tech. rep.*, Canadian Science Advisory Secretariat Research Document 2001/156, 2001.

Pikitch, E. K., M. D. Camhi, and E. A. Babcock, Introduction to sharks of the open ocean, in *Sharks of the open ocean: biology, fisheries and conservation*, edited by M. D. Camhi, E. K. Pikitch, and E. A. Babcock, pp. 1–13, Blackwell Publishing, Oxford, UK, 2008.

Pikitch, E. K., C. Santora, E. A. Babcock, A. Bakun, R. Bonfil, D. O. Conover, P. Dayton, P. Doukakis, D. Fluharty, B. Heneman, E. D. Houde, and J. Link, Ecosystem-based fishery management, *Science*, *305*, 346–347, 2004.

Pinsky, M. L., B. Worm, M. J. Fogarty, J. L. Sarmiento, and S. A. Levin, Marine taxa track local climate velocities, *Science*, *341*, 1239–1242, 2013.

Piovano, S., S. Clò, and C. Giacoma, Reducing longline bycatch : The larger the hook , the fewer the stingrays, *Biological Conservation*, *143*, 261–264, 2010.

Promjinda, S., S. Siriraksophon, N. Darumas, and P. Chaidee, Efficiency of the Circle Hook in Comparison with J-Hook in Longline Fishery, *The Ecosystem-Based Fishery Management in the Bay of Bengal*, 2008.

Quiroz, Z. C., M. O. Prates, and H. v. Rue, A Bayesian approach to estimate the biomass of anchovies off the coast of Perú, *Biometrics*, 2014.

R Core Team, *R: A Language and Environment for Statistical Computing*, R Foundation for Statistical Computing, Vienna, Austria, 2013.

R Development Core Team, *R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna*, 2008.

Rice, P., A shocking discovery: How electropositive metals (EPMs) work and their effects on elasmobranchs, in *Shark deterrent and incidental capture workshop, 1011 April 2008*, edited by Y. Swimmer, J. Wang, and L. McNaughton, pp. 21–25, US Department of Commerce, NOAA Technical Memorandum NOAA-TM-NMFS-PIFSC-16, 2008.

Rigby, C., and C. a. Simpfendorfer, Patterns in life history traits of deep-water chondrichthyans, *Deep-Sea Research Part II: Topical Studies in Oceanography*, pp. 1–11, 2013.

Rigg, D. P., S. C. Peverell, M. Hearndon, J. E. Seymour, B. Peverell, D. P. R. A, S. C. P. B, M. H. C, and J. E. S. A, Do elasmobranch reactions to magnetic fields in water show promise for bycatch mitigation ?, *Marine And Freshwater Research*, *60*, 942–948, 2009.

Rijnsdorp, A. D., H. M. J. V. Overzee, and J. J. Poos, Ecological and economic trade-offs in the management of mixed fisheries : a case study of spawning closures in flatfish fisheries, *Marine Ecology Progress Series*, *447*, 179–194, 2012.

Rivot, E., E. Prévost, A. Cuzol, J.-l. Baglinière, and E. Parent, Hierarchical Bayesian modelling with habitat and time covariates for estimating riverine fish population size by successive removal method, *133*, 117–133, 2008.

Robbins, W., V. Peddemors, and S. Kennelly, Assessment of permanent magnets and electropositive metals to reduce the line-based capture of Galapagos sharks, Carcharhinus galapagensis, *Fisheries Research*, *109*, 100–106, 2011.

Robbins, W. D., M. Hisano, S. R. Connolly, and J. H. Choat, Ongoing collapse of coral-reef shark populations., *Current Biology*, *16*, 2314–2319, 2006.

Rochet, M.-j., and V. M. Trenkel, Factors for the variability of discards: assumptions and field evidence, *Canadian Journal of Fisheries and Aquatic Sciences*, *62*, 224–235, 2005.

Roe, J. H., S. J. Morreale, F. V. Paladino, G. L. Shillinger, S. R. Benson, A. E. Scott, H. Bailey, P. S. Tomillo, S. J. Bograd, T. Eguchi, P. H. Dutton, J. A. Seminoff, B. A. Block, and J. R. Spotila, Predicting bycatch hotspots for endangered leatherback turtles on longlines in the Pacific Ocean, *Proceedings of the Royal Society B: Biological Sciences*, *281*, 2014.

Rue, H., and L. Held, *Gaussian Markov Random Fields: Theory and Applications*, Boca Raton, Chapman and HallCRC, 2005.

Rue, H., S. Martino, and N. Chopin, Approximate Bayesian inference for latent Gaussian models by using integrated nested Laplace approximations, *Journal of the Royal Statistical Society: Series B (Statistical Methodology)*, *71*, 319–392, 2009.

Rue, H. v., and S. Martino, Approximate Bayesian inference for hierarchical Gaussian Markov random field models, *Journal of Statistical Planning and Inference*, *137*, 3177–3192, 2007.

Sales, G., B. B. Giffoni, F. N. Fiedler, V. G. Azevedo, J. E. Kotas, Y. Swimmer, and L. Bugoni, Circle hook effectiveness for the mitigation of sea turtle bycatch and capture of target species in a Brazilian pelagic longline fishery, *Aquatic Conservation: Marine and Freshwater Ecosystems*, *20*, 428–436, 2010.

Sanchez, P., M. Demestre, and P. Martín, Characterisation of the discards generated by bottom trawling in the northwestern Mediterranean, *Fisheries Research*, *67*, 71–80, 2004.

Serafy, J. E., D. W. Kerstetter, and P. H. Rice, Can circle hook use benefit billfishes?, *Fish and Fisheries*, *10*, 132–142, 2009.

Shelton, A. O., J. T. Thorson, E. J. Ward, and B. E. Feist, Spatial, semi-parametric models improve estimates of species abundance and distribution, *Canadian Journal of Fisheries and Aquatic Sciences*, *71*, 1655–1666, 2014.

Simon, J., and K. Frank, Assessment of the winter skate fishery in division 4VsW, *Tech. rep.*, Canadian Stock Assessment Secretariat. Research Document 2000/140., 2000.

Simon, J. E., L. E. Harris, and J. T. L., Distribution and abundance of winter skate, Leucoraja ocellatta, in the Canadian Atlantic, *Tech. rep.*, Canadian Science Advisory Secretariat. Research Document 2003/028., 2003.

Simon, J. E., A. Cook, S. Rowe, and M. Simpson, Pre-COSEWIC review of barndoor skate Dipturus laevis in the Canadian Atlantic, 2009.

Simpfendorfer, C. a., and P. M. Kyne, Limited potential to recover from overfishing raises concerns for deep-sea sharks, rays and chimaeras, *Environmental Conservation*, *36*, 97, 2009.

Simpson, M., L. Mello, C. Miri, M. Treble, and T. Siferd, Distribution, abundance, and life history of smooth skate (Malacoraja senta Garman 1885) in Northwest Atlantic waters, *Tech. rep.*, DFO Can. Sci. Advis. Sec. Res. Doc. 2011/116. iv + 40 p, 2012a.

Simpson, M., C. Miri, and C. Busby, Assessment of Thorny skate (Amblyraja radiata Donovan, 1808) in NAFO Divisions 3LNO and Subdivision 3Ps, *Tech. rep.*, DFO Can. Sci. Advis. Sec. Res. Doc. 2012/139. iv + 38 p, 2012b.

Sims, D. W., E. J. Southall, A. J. Richardson, P. C. Reid, and J. D. Metcalfe, Seasonal movements and behaviour of basking sharks from archival tagging : no evidence of winter hibernation, *Marine Ecology Progress Series*, *248*, 187–196, 2003.

Sims, M., T. Cox, and R. Lewison, Modeling spatial patterns in fisheries bycatch: improving bycatch maps to aid fisheries management., *Ecological Applications*, *18*, 649–61, 2008.

Smith, S. E., D. W. Au, and C. Show, Intrinsic rebound potentials of 26 species of Pacific sharks, *Marine And Freshwater Research*, *49*, 663–678, 1998.

Sosebee, K., and M. Terceiro, Assessment of the United States Northeast Region Skate Complex for 2000, *Tech. rep.*, NAFO SCR Doc. 00/15, 2000.

SPC, Longline terminal gear Identification guide, 2009.

Spiegelhalter, D. J., N. G. Best, B. P. Carlin, and A. Van Der Linde, Bayesian measures of model complexity and fit, *Journal of the Royal Statistical Society: Series B (Statistical Methodology)*, *64*, 583–639, 2002.

Stevens, J., R. Bonfil, N. K. Dulvy, and P. Walker, The effects of fishing on sharks, rays, and chimaeras (chondrichthyans), and the implications for marine ecosystems, *ICES Journal of Marine Science*, *57*, 476–494, 2000.

Stone, H. H., and P. O. Box, A comparison of catches of swordfish, Xiphias gladius, and other pelagic species from Canadian longline gear configured with alternating monofilament and multifilament nylon gangions, *Fisheries Bulletin*, *99*, 210–216, 2001.

Stoner, A. W., and S. M. Kaimmer, Reducing elasmobranch bycatch: Laboratory investigation of rare earth metal and magnetic deterrents with spiny dogfish and Pacific halibut, *Fisheries Research*, *92*, 162–168, 2008.

Swimmer, Y., J. H. Wang, and L. Mcnaughton, Shark Deterrent and Incidental Capture Workshop April 10-11, 2008, *Tech. Rep. November*, U.S. Dep. Commer. NOAA Tech. Memo., NOAA-TM-NMFS-PIFSC-16, 2008.

Tallack, S. M. L., and J. W. Mandelman, Do rare-earth metals deter spiny dogfish? A feasibility study on the use of electropositive "mischmetal" to reduce the bycatch of Squalus acanthias by hook gear in the Gulf of Maine, *ICES Journal of Marine Science*, *66*, 315–322, 2008.

Techera, E. J., and N. Klein, Fragmented governance: Reconciling legal strategies for shark conservation and management, *Marine Policy*, *35*, 73–78, 2011.

Thorpe, T., and D. Frierson, Bycatch mitigation assessment for sharks caught in coastal anchored gillnets, *Fisheries Research*, *98*, 102–112, 2009.

Thrush, S. F., and P. K. Dayton, Disturbance to marine benthic habitats by trawling and dredging: Implications for marine biodiversity, *Annual Review of Ecology and Systematics*, *33*, 449–473, 2002.

Tobler, W. R., A Computer Movie Simulating Urban Growth in the Detroit Region, *Economic Geography*, *2*, 234–240, 1970.

Trebilco, R., R. Gales, E. Lawrence, R. Alderman, G. Robertson, and G. B. Baker, Characterizing seabird bycatch in the eastern Australian tuna and billfish pelagic longline fishery in relation to temporal, spatial and biological influences, *Aquatic Conservation: Marine and Freshwater Ecosystems*, *20*, 531–542, 2010.

Treble, M. a., Analysis of data from the 2004 trawl surveys in NAFO division OA, *Tech. rep.*, NAFO SCR Doc. 02/47, 2002.

Tricas, T., The neuroscience of the elasmobranch electrosensoryworld: why peripheral morphology shapes behavior, *Environ. Biol. Fish.*, *60*, 77–92, 2001.

Turner, S. J., S. F. Thrush, J. E. Hewitt, V. J. Cummings, and G. Funnell, Fishing impacts and the degradation or loss of habitat structure, *Fisheries Management and Ecology*, *6*, 401–420, 1999.

Vannuccini, S., Shark utilization, marketing and trade, *Tech. rep.*, FAO Fisheries Technical Paper 398, 1999.

Venables, W., and C. Dichmont, GLMs, GAMs and GLMMs: an overview of theory for applications in fisheries research, *Fisheries Research*, *70*, 319–337, 2004.

Viana, M., A. L. Jackson, N. Graham, and A. C. Parnell, Disentangling spatio-temporal processes in a hierarchical system: a case study in fisheries discards, *Ecography*, pp. no–no, 2012.

Wade, P. R., Bayesian Methods in Conservation Biology, *Conservation Biology*, *14*, 1308–1316, 2000.

Wang, J., L. McNaughton, and Y. Swimmer, Galapagos and sandbar shark aversion to electropositive metal (Pr-Nd alloy), in *Shark Deterrent and Incidental Capture Workshop, 1011 April 2008*, edited by Y. Swimmer, J. H. Wang, and L. McNaughton, pp. pp. 31–35, US Department of Commerce, NOAA Technical Memorandum NOAA-TM-NMFS-PIFSC-16. 72 pp.er, J. H. Wang, and L. McNaughton. US Department of Commerce, NOAA Technical Memorandum NOAA-TM-NMFS-PIFSC-16. 72 pp., 2008.

Ward, P., and S. Hindmarsh, An overview of historical changes in the fishing gear and practices of pelagic longliners, with particular reference to Japans Pacific fleet, *Reviews in Fish Biology and Fisheries*, *17*, 501–516, 2007.

Ward, P., R. A. Myers, and W. Blanchard, A Fish lost at sea : the effect of soak time on pelagic longline catches, *Fisheries Bulletin*, *102*, 179–195, 2004.

Ward, P., S. Epe, D. Kreutz, E. Lawrence, C. Robins, and A. Sands, The effects of circle hooks on bycatch and target catches in Australia s pelagic longline fishery, *Fisheries Research*, *97*, 253–262, 2009.

Watson, J. W., and D. W. Kerstetter, Pelagic longline fishing gear: A brief history and review of research efforts to improve selectivity, *Marine Technology Society Journal*, *40*, 6–11, 2006.

Watson, J. W., S. P. Epperly, A. K. Shah, and D. G. Foster, Fishing methods to reduce sea turtle mortality associated with pelagic longlines, pp. 965–981, 2005.

Waugh, S. M., G. B. Baker, R. Gales, and J. P. Croxall, CCAMLR process of risk assessment to minimise the effects of longline fishing mortality on seabirds, *Marine Policy*, *32*, 442–454, 2008.

WCPFC, Conservation and management of sea turtles, in *Fifth regular session 8-12 December 2008*, Busan, Korea.

Wigley, S. E., Methods to Determine Discards from Observed Trips, 2010.

Wikle, C. K., Hierarchical Models in Environmental Science, *International Statistical Review*, *71*, 181–199, 2007.

Wilking, H., M. Höhle, E. Velasco, M. Suckau, and T. Eckmanns, Ecological analysis of social risk factors for Rotavirus infections in Berlin, Germany, 2007-2009., *International journal of health geographics*, *11*, 37, 2012.

Winter, A., Y. Jiao, and J. A. Browder, Modeling Low Rates of Seabird Bycatch in the US Atlantic Long line Fishery, *Waterbirds*, *34*, 289–303, 2011.

Wintle, B. a., M. a. McCARTHY, C. T. Volinsky, and R. P. Kavanagh, The use of Bayesian model averaging to better represent uncertainty in ecological models, *Conservation Biology*, *17*, 1579–1590, 2003.

Worm, B., H. K. Lotze, and R. A. Myers, Predator diversity hotspots in the blue ocean., *Proceedings of the National Academy of Sciences of the United States of America*, *100*, 9884–8, 2003.

Worm, B., B. Davis, L. Kettemer, C. Ward-Paige, D. Chapman, M. Heithaus, S. Kessel, and S. Gruber, Global catches, exploitation rates, and rebuilding options for sharks, *Marine Policy*, *40*, 194–204, 2013.

Worm, B., A. Cosandey-Godin, and B. Davis, Fisheries management and regulations, in *Sharks: Conservation, governance, and management*, edited by E. J. Techera and N. Klein, taylor and ed., 2014.

Worm, B., E. B. Barbier, N. Beaumont, J. E. Duffy, C. Folke, B. S. Halpern, J. B. C. Jackson, H. K. Lotze, F. Micheli, S. R. Palumbi, E. Sala, K. a. Selkoe, J. J. Stachowicz, and R. Watson, Impacts of biodiversity loss on ocean ecosystem services., *Science*, *314*, 787–90, 2006.

Worm, B., R. Hilborn, J. K. Baum, T. A. Branch, J. S. Collie, C. Costello, M. J. Fogarty, E. A. Fulton, J. A. Hutchings, S. Jennings, O. P. Jensen, H. K. Lotze, P. M. Mace, T. R. McClanahan, C. Minto, S. R. Palumbi, A. M. Parma, D. Ricard, A. A. Rosenberg, R. Watson, and D. Zeller, Rebuilding global fisheries., *Science*, *325*, 578–585, 2009.

Yeh, Y.-M., H.-W. Huang, K. S. Dietrich, and E. Melvin, Estimates of seabird incidental catch by pelagic longline fisheries in the South Atlantic Ocean, *Animal Conservation*, *16*, 141–152, 2013.

Yokota, K., M. Kiyota, and H. Minami, Shark catch in a pelagic longline fishery: Comparison of circle and tuna hooks, *Fisheries Research*, *81*, 337–341, 2006.

Yu, R., and P. Leung, A Bayesian hierarchical model for modeling white shrimp (Litopenaeus vannamei) growth in a commercial shrimp farm, *Aquaculture*, *306*, 205–210, 2010.

APPENDIX A

BAYESIAN HIERARCHICAL MODEL

A.1 Model Structure

In the Generalized Linear Model (GLM) framework, we define a link function that maps the mean of the response to the linear predictor. For example, under the assumption that the response variable has a negative binomial (NB) distribution, and that the bycatch at location $\mathbf{s}$ at time t is equal to k with dispersion parameter n, then we can write $P[Y(\mathbf{s},t) = k]$ as

$$NB(k,n,\theta(\mathbf{s},t)) = \frac{\Gamma(k+n)}{\Gamma(n)\Gamma(k+1)}\theta(\mathbf{s},t)(1-\theta(\mathbf{s},t))). \tag{A.1}$$

The mean of the above distribution, $\mu(\mathbf{s},t) = n\frac{1-\theta(\mathbf{s},t)}{\theta(\mathbf{s},t)}$, is linked to the linear predictor $\eta(\mathbf{s},t)$ by $\mu(\mathbf{s},t) = \exp^{\eta(\mathbf{s},t)}$ where

$$\eta(\mathbf{s},t) = X(\mathbf{s},t)\beta + f(\mathbf{s},t). \tag{A.2}$$

Note that in the above expression $X(\mathbf{s},t)$ is the design matrix with p fixed covariates, $\beta = (\beta_0,\ldots,\beta_p)$ is the regression coefficients vector, and $f(\mathbf{s},t)$ introduces the spatio-temporal structure via random effects by considering a Gaussian random field (that is spatially and temporally correlated) as per *Cameletti et al.* (2012). To proceed, we consider $f(\mathbf{s},t)$ to be a realization of a continuously indexed spatial process (random field) changing in time denoted by

$$\{f(\mathbf{s},t) : (\mathbf{s},t) \in D \subseteq \Re^2 \times \Re\}. \tag{A.3}$$

These realizations are used to make inference about the process and to predict it at desired locations. Usually, we deal with a Gaussian field (GF) that is completely specified by its

mean and spatio-temporal covariance function

$$Cov(f(\mathbf{s},t), f(\mathbf{s}',t')) = \sigma^2 Cov((\mathbf{s},t),(\mathbf{s}',t')), \tag{A.4}$$

defined for each $(\mathbf{s},t)$ and $(\mathbf{s}',t')$ in $\Re^2 \times \Re$. Unfortunately implementation of a GF suffers from the so-called *big n problem* because it requires the factorization of the covariance matrix in order to compute the likelihood, see *Diggle and Ribeiro* (2007), this arises particularly with large datasets in space and time. One solution is to represent a GF as a *discretely* indexed random process, i.e. a Gaussian Markov Random Field (GMRF, see *Rue and Held* (2005)). This proposal is based on the work of Lindgren et al. (2011) where an explicit link between GFs and GMRFs is proven through the use of Stochastic Partial Differential Equations (SPDE). This is the approach taken here.

By assuming a zero-inflated negative binomial distribution rather than a standard negative binomial distribution, we are allowing for additional probability of $Y(\mathbf{s},t) = 0$. For example, under the Type 2 formulation we have that

$$P[Y(\mathbf{s},t) = k] = p(\mathbf{s},t) I_{k=0} + (1 - p(\mathbf{s},t)) \times NB(k,n,\theta(\mathbf{s},t)) \tag{A.5}$$

where $p(\mathbf{s},t)$ is the extra probability added to the negative binomial for $k = 0$ (a similar relationship exists between the Poisson and zero-inflated Poisson distribution). The extra probability of zero, $p(\mathbf{s},t)$ depends on the linear predictor, $\eta(\mathbf{s},t)$, i.e., the extra probability depends on the covariates and on the spatio-temporal random effect. That is,

$$p(\mathbf{s},t) = 1 - [\exp^{\eta(\mathbf{s},t)} / (1 + \exp^{\eta(\mathbf{s},t)})]^{\alpha}. \tag{A.6}$$

This means that the probability of zero bycatch at location $\mathbf{s}$ at time t is inversely proportional to the linear predictor $\eta(\mathbf{s},t)$ which makes sense because under the negative binomial distribution, for example, we have $\mu(\mathbf{s},t) = \exp^{\eta(\mathbf{s},t)}$, i.e. $\eta(\mathbf{s},t)$ is proportional to the expected number of sharks that were recorded as bycatch. We also have that $\pi(\mathbf{s},t) = \exp^{\eta(\mathbf{s},t)} / (1 + \exp^{\eta(\mathbf{s},t)})$ is proportional to $\eta(\mathbf{s},t)$ and, writing this in one equation, we arrive at

$$P[Y(\mathbf{s},t) = k] = (1 - \pi(\mathbf{s},t)^{\alpha})_{k=0} + (\pi(\mathbf{s},t)^{\alpha}) \times NB(k,n,\theta(\mathbf{s},t)). \tag{A.7}$$

The parameter α can be thought of as a hyperparameter that controls both the impact of the linear predictor on the extra probability at zero as well as that for the positive counts. When $\alpha = 0$ we have $\pi(\mathbf{s},t)^{\alpha} = 1$ and simply $P[Y(\mathbf{s},t) = k] = NB(k,n,\theta(\mathbf{s},t))$, namely, no *extra* probability at zero. Because $\pi(\mathbf{s},t)^{\alpha} \in (0,1)$ and $\alpha > 0$ if α increases, $\pi(\mathbf{s},t)$ also increases and the extra probability of zero decreases.

A.2 Priors

We used the default priors and hyperparameters currently implemented in R-INLA. For further information please refer to the R-INLA documentation available on the website. Note that this is an active area of research for the R-INLA team. For the overdispersion parameter *n* (size), represented as $\log(n)$, the prior is a *loggamma(1,1)*. For the α parameter on the type 2 zero-inflated model, we assume N(0.693, 1) for $\log(\alpha)$.

Every components of the latent field $\theta = \{\beta_0, \beta, f\}$ have priors. For the smooth function of bathymetry, we assumed a random walk of order one, which is defined in terms of a Gaussian distribution N(0, $\tau_1 R_1$), where R_1 is the (fixed and know) structure matrix and τ_1 is the precision parameter. To model the spatial correlation, we assume a spatial Matérn correlation using the SPDE approach, with parameters $\theta_1 = \log(\tau)$ and $\theta_2 = \log(\kappa)$ and priors defined in *Lindgren and Rue* (2013).

The SPDE framework allows to easily link different GRF in time and this way, develop dynamic model where your spatial field evolve according to time-varying processes, for example an autoregressive process. For these dynamic models i.e. m3a, m3b, m5a, and m5b, we also had an additional correlation parameter ρ for time, $\log(\frac{1+\rho}{1-\rho})$, with a Gaussian prior.

$\log(n)$	loggamma(1,1)
$\log(\alpha)$	N(0.693, 1)
β_0	N(0, ∞)
$\beta_j, j = 1,2,3$	N(0, 1/0.001)
τ_1	Gamma(1,0.00005)
θ_1	N(0, 10)
θ_2	N(0, 1)
$\log(\frac{1+\rho}{1-\rho})$	N(0,0.15)

Table A.1: Summary of the default priors used in the model tested.

APPENDIX B

EXCURSION FUNCTIONS MAPS

The interpretation of these maps is as follow: hotspots regions where the function is above, say, 0.95 is the largest region such that with probability 0.95 at least X times the average number of sharks will be caught at all locations in the region for all years 2003-2013. Note that the red line on these maps indicates the 200 nautical miles Canadian Exclusive Economic Zone and isobaths of the region were plotted for reference.

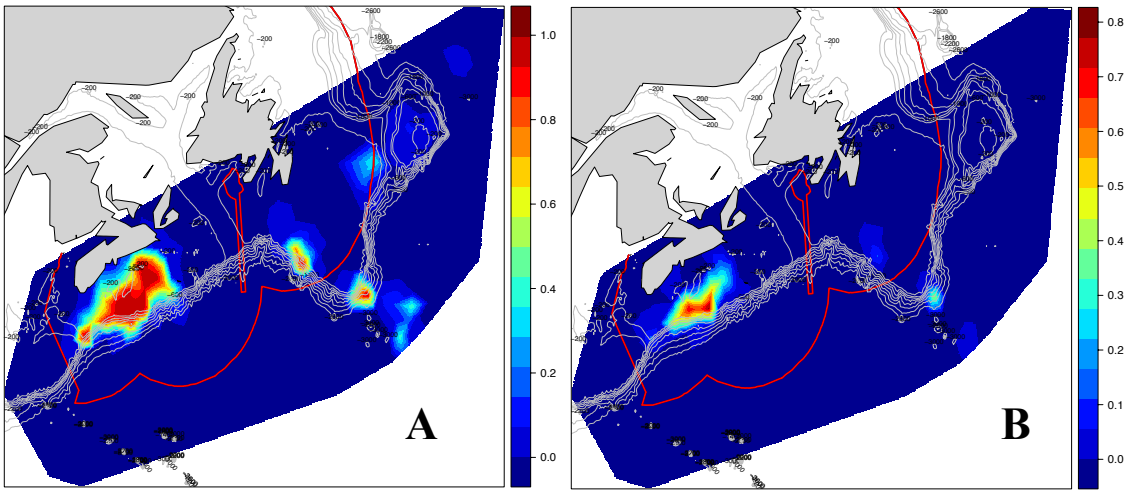


Figure B.1: Excursions functions for blue shark (*Prionace glauca*) in the Canadian pelagic longline fishery, 2003-2013. A) For catching 2 times the average number of sharks per fishing set (66 sharks/set) B) For catching 5 times the average number of sharks per fishing set (165 sharks/set)

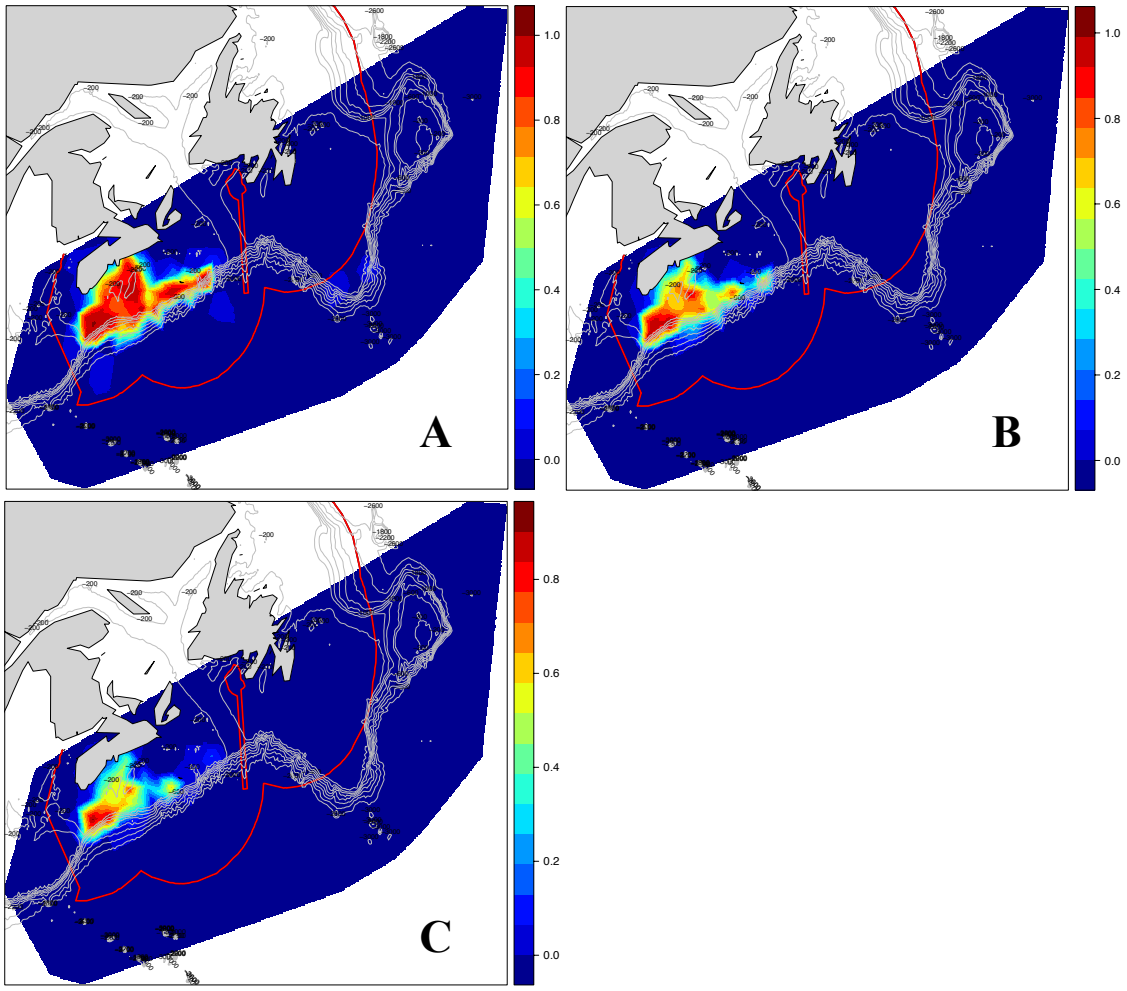


Figure B.2: Excursions functions for porbeagle shark (*Lamna nasus*) in the Canadian pelagic longline fishery, 2003-2013. A) For catching 2 times the average number of sharks per fishing set (4 sharks/set) B) For catching 5 times the average number of sharks per fishing set (10 sharks/set) C) For catching 10 times the average number of sharks per fishing set (20 sharks/set).

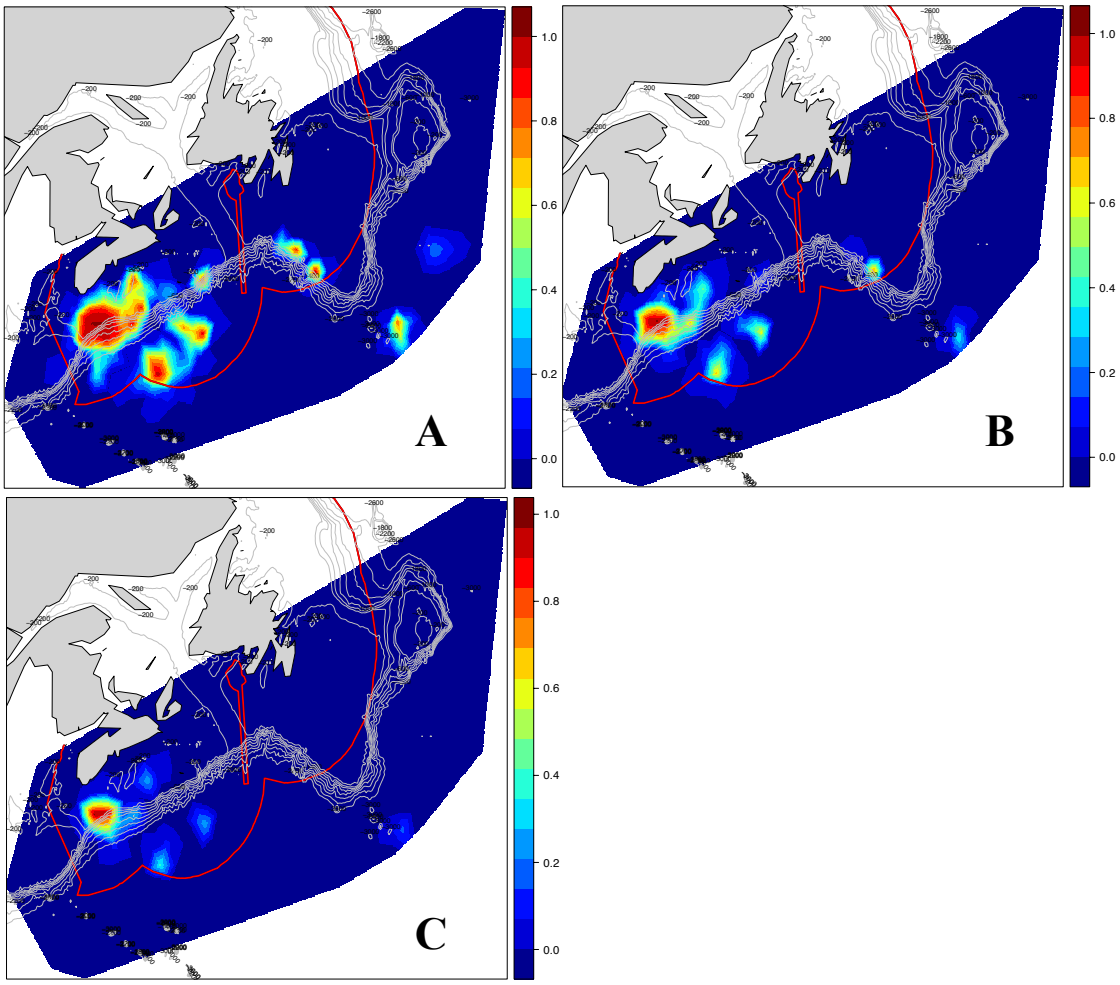


Figure B.3: Excursions functions for shortfin mako (*Isurus oxyrinchus*) in the Canadian pelagic longline fishery, 2003-2013. A) For catching 2 times the average number of sharks per fishing set (4 sharks/set) B) For catching 5 times the average number of sharks per fishing set (10 sharks/set) C) For catching 10 times the average number of sharks per fishing set (20 sharks/set).

APPENDIX C

DATA SPECIFICATIONS

Table C.1: List of fishing gear included under in each of the 11 categories. Note that some gear are repeated (small caps). These are the original coding which differ from DFO regions.

Gear	Category
PERIWINKLE DIVERS	Diver
CLAM DREDGE	Dredge
SCALLOP DREDGE	Dredge
DREDGE (BOAT)	Dredge
GILLNET	Gillnet
Gillnet (set)	Gillnet
SET GILLNETS	Gillnet
HARPOONS	Harpoon
HANDLINE	Hook
Hand line (baited)	Hook
HANDLINES (INC.POLELINES,JIG-L	Hook
AUTOMATIC JIGGER	Jigger
Manual jigger	Jigger
TROLL LINES	Jigger
LONGLINE	Line
Longline (drift)	Line
Longline (fixed)	Line
DRIFT LINES (DRIFTING LONGLINE	Line

Continued on next page

Continued from previous page

Gear	Category
LONGLINE (TYPE NOT SPECIFIED)	Line
SET LINES (BOTTOM OR NEAR BOT.	Line
PURSE SEINE	Seine
Purse seine	Seine
BAR SEINE	Trap
CAPELIN TRAP	Trap
POTS	Trap
TRAP	Trap
COVERED POTS (LOB.,CRAB.ETC),F	Trap
UNCOVERED POUND NETS (EG.CODTR	Trap
BEACH SEINE	Trap
TRAPS (TYPE NOT SPECIFIED)	Trap
DANISH SEINE	Trawlb
NORWEGIAN SEINE	Trawlb
OTTER TRAWL (STERN)	Trawlb
SCOTTISH SEINE	Trawlb
SHRIMP TRAWL	Trawlb
TRIPLE TRAWL	Trawlb
TWIN TRAWL	Trawlb
Bottom otter trawl (side)	Trawlb
Bottom otter trawl (stern)	Trawlb
Bottom pair trawl	Trawlb
Danish seine	Trawlb
Scottish seine	Trawlb
Shrimp trawl (side) with a grid	Trawlb
Shrimp trawl (stern) with a grid	Trawlb
Twin trawl	Trawlb
BOTTOM OTTER TRAWL (SIDE)	Trawlb
BOTTOM OTTER TRAWL (STERN)	Trawlb
OTTER SHRIMP TRIPLE TRAWL	Trawlb

Continued on next page

Continued from previous page

Gear	Category
OTTER SHRIMP TWIN TRAWL	Trawlb
OTTER TWIN TRAWL	Trawlb
MIDWATER TRAWL	Trawlm
MIDWATER PAIR	Trawlm
Midwater trawl (stern)	Trawlm
MIDWATER TRAWL (CHARTERS)	Trawlm
MIDWATER TRAWL (STERN)	Trawlm

Table C.2: List of species included under each of the 7 broader categories.

Species sought	**Latin name**	**Category**
COCKLE	Cardiidae spp.	Bivalve
SURFCLAM (SIMPSON)	Mactromeris polynyma	Bivalve
SCALLOP	Pectinidae spp.	Bivalve
QUAHOG	Mercenaria mercenaria	Bivalve
CRAB(CANC.IRR.)	Cancer irroratus	Crab_Lobster
CRAB(SNOW,QUEEN)	Chionoecetes bairdi	Crab_Lobster
TOAD CRAB(HYAS.SP)	Hyas spp.	Crab_Lobster
LOBSTER	Homarus americanus	Crab_Lobster
SPINY CRAB	Acantholithodes hispidus	Crab_Lobster
DEEPSEA RED CRAB(CHAC.QUIN.)	Chaceon quinquedens	Crab_Lobster
JONAH CRAB(CANC.BOR.)	Cancer borealis	Crab_Lobster
AMERICAN PLAICE	Hippoglossoides platessoides	Groundfish
MONKFISH	Lophius spp.	Groundfish
ARGENTINE(ATL)	Argentina silus	Groundfish
COD(ATL)	Gadus morhua	Groundfish
WINTER FLOUNDER	Pseudopleuronectes americanus	Groundfish
HALIBUT(GREENLAND)	Reinhardtius hippoglossoides	Groundfish
HAGFISH(ATL)	Myxine glutinosa	Groundfish

Continued on next page

Table C.2 – *Continued from previous page*

Species sought	**Latin name**	**Category**
WHITE HAKE	Urophycis tenuis	Groundfish
HALIBUT(ATL)	Hippoglossus hippoglossus	Groundfish
LUMPFISH	Cyclopteridae spp.	Groundfish
POLLOCK	Pollachius spp.	Groundfish
REDFISH	Sebastes fasciatus	Groundfish
SKATE(NS)	Rajidae spp.	Groundfish
WITCH FLOUNDER	Glyptocephalus cynoglossus	Groundfish
YELLOWTAIL FLOUNDER	Pleuronectes ferruginea	Groundfish
SANDLANCE	Ammodytidae spp.	Groundfish
COD(ARC)	Arctogadus glacialis	Groundfish
BLACK DOGFISH	Centroscyllium fabricii	Groundfish
LOOKDOWN	Selene vomer	Groundfish
HEADLIGHTFISH	Diaphus effulgens	Groundfish
SMOOTH FLOUNDER	Pleuronectes putnami	Groundfish
SPINY DOGFISH	Squalus acanthias	Groundfish
SUMMER FLOUNDER	Paralichthys dentatus	Groundfish
CUNNER	Tautogolabrus adspersus	Groundfish
CUSK	Brosme brosme	Groundfish
HADDOCK	Melanogrammus aeglefinus	Groundfish

Continued on next page

Table C.2 – *Continued from previous page*

Species sought	**Latin name**	**Category**
LONGHORN SCULPIN	Myoxocephalus octodecimspinosus	Groundfish
ROCK GRENADIER(ROUNDNOSE)	Coryphaenoides rupestris	Groundfish
SILVER HAKE	Merluccius bilinearis	Groundfish
RED HAKE	Urophycis chuss	Groundfish
THORNY SKATE	Amblyraja radiata	Groundfish
WINTER SKATE	Leucoraja ocellata	Groundfish
DOGFISH	Squalidae spp.	Groundfish
SEACU	Holothuroidea spp.	Invert
SQUID	Teuthida spp.	Invert
WHELK	Buccinidae spp.	Invert
SEA URCHINS	Echinoidea spp.	Invert
BILLFISH	Istiophoridae spp.	Large_pelagic
SWORDFISH	Xiphias gladius	Large_pelagic
BLUEFIN TUNA	Thunnus thynnus	Large_pelagic
BIGEYE TUNA	Thunnus obesus	Large_pelagic
PORBEAGLE	Lamna nasus	Large_pelagic
ALBACORE TUNA	Thunnus alalunga	Large_pelagic
YELLOWFIN TUNA	Thunnus albacares	Large_pelagic
PAND.BOR.	Pandalus borealis	Shrimp

Continued on next page

Table C.2 – *Continued from previous page*

Species sought	**Latin name**	**Category**
PAND.MON.	Pandalus montagui	Shrimp
CAPELIN	Mallotus villosus	Small_pelagic
HERRING(ATL)	Clupea harengus	Small_pelagic
MACKEREL(ATL)	Scomber scombrus	Small_pelagic

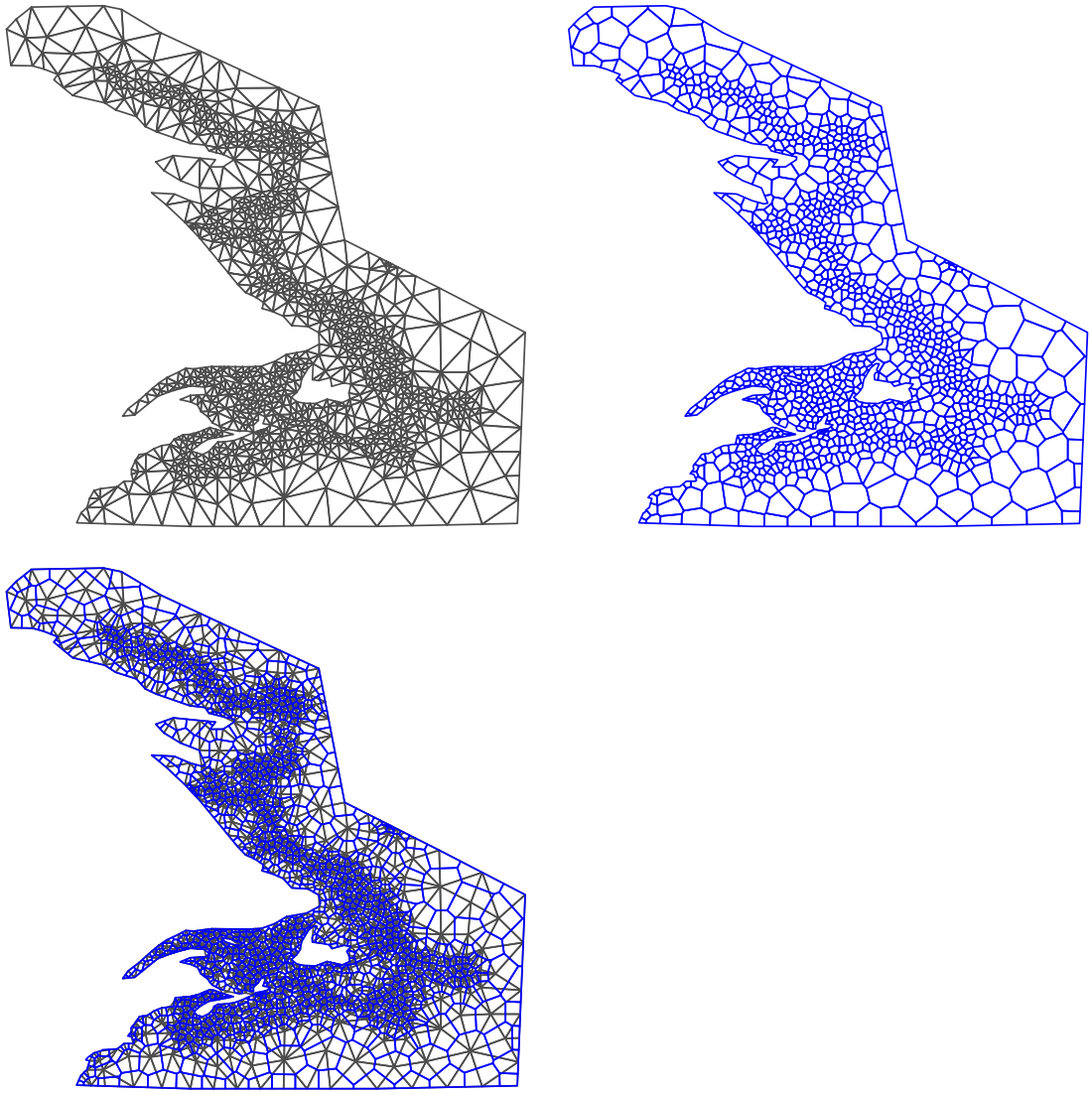


Figure C.1: Example of the mesh and Dirichlet tessellation used in the Selachii models.

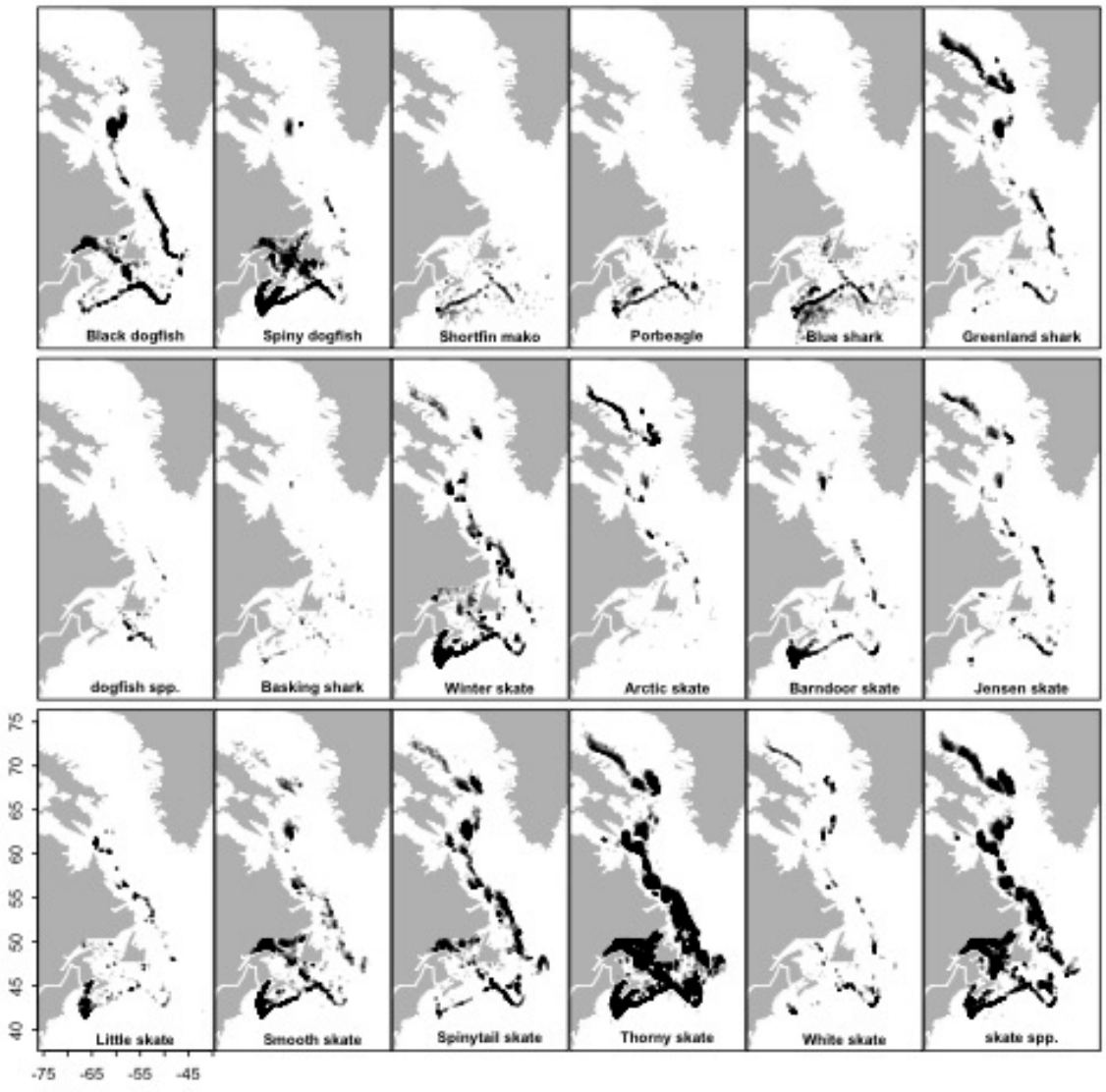


Figure C.2: Maps of the Northwest Atlantic showing the location of observed discards of elasmobranch species in Canadian fisheries between 1996-2012 (raw data). Darker locations shows areas with overlapping observations.