DTU Library

Bycatch of seabirds in Danish gillnet fisheries - assessing scale and testing mitigation

Glemarec, Gildas

Publication date:
2020

Document Version
Publisher's PDF, also known as Version of record

Bycatch of seabirds in Danish gillnet fisheries - assessing scale and testing mitigation

Ph.D. thesis by Gildas Glemarec

December 2016 – March 2020

National Institute of Aquatic Resources (DTU Aqua), Technical University of Denmark

Section for ecosystem-based marine management, Kgs. Lyngby, Denmark

PREFACE

The present thesis was submitted in partial fulfilment of the requirements for obtaining a Doctor of Philosophy (Ph.D.) degree. The thesis consists of a synopsis and three supporting papers. When submitted, one paper was published, one was submitted and one was a manuscript.

The work took place in the Technical University of Denmark at the National Institute of Aquatic Resources (DTU Aqua) in the Section for Ecosystem Based Marine Management in Lyngby (Denmark), from December 2016 to March 2020.

Academic supervisors

Senior research scientist Finn Larsen
Section for Ecosystem Based Marine Management, National Institute of Aquatic Resources
Technical University of Denmark

Research scientist Lotte Kindt-Larsen
Section for Ecosystem Based Marine Management, National Institute of Aquatic Resources
Technical University of Denmark

Evaluation committee

Senior research scientist Ludvid Ahm Krag
Section for Ecosystem Based Marine Management, National Institute of Aquatic Resources
Technical University of Denmark

Dr. Simon Northridge
Sea Mammal Research Unit
University of St Andrews
United Kingdom

Research scientist Dominik Marchowski
Ornithological Station
Museum and Institute of Zoology, Polish Academy of Sciences, Gdańsk
Poland

ACKNOWLEDGEMENTS

I would like to express my sincere gratitude to my supervisors, Finn Larsen and Lotte Kindt-Larsen for their support, for their enthusiasm, and, last but not least, for their trust.

The financial support from the European Maritime and Fisheries Fund and the Danish Fisheries Agency (grant number 33113-I-16-040), which made this work possible, is gratefully acknowledged.

This whole Ph.D. would not have been possible without the dedication of the electronic monitoring (EM) analysts who watched so many hours of videos, and somehow always managed to keep their spirits high. Thank you guys, to each one of you. Special thanks to Tejs Nielsen and Mathias Søgaard for helping so much with the data collection.

Thank you to all the fishers who participated in the EM data collection, and especially to Jacob for helping us with the bycatch mitigation work.

I also would to thank:

Anne-Mette Kroner for the help she provided both with EM data analyses and with fieldwork and, in general, for being one of the best colleague anybody could hope for,

Thomas Noack for advising me during the writing of this thesis,

My section leader Anna Rindorf,

My colleagues from my section Ecosystem Based Marine Management section,

My Ph.D. fellows at DTU Aqua, especially the ones with whom I have been sharing an office during all these years,

The Department of Bioscience at Aarhus University in Risø, particularly Morten Frederiksen,

The members of assessment committee.

My most grateful thanks to my wife and partner in life Louise, whose love has made me a better person. Undoubtedly, I would not have made it without you.

Three and a half years. That is the time it took to get from the novice Ph.D. student I was when I started to this point in time, where, I hope, I have gained some experience. I consider myself lucky to have met some great people along the way, some of whom I probably forgot to include in this short list. My apologies if you should have been in.

Lyngby, March 2020

Gildas Glemarec

TABLE OF CONTENTS

Summary (English)

Over the last decades, a large number of seabird populations have faced rapid declines due to a variety of threats, including deadly interactions with fisheries. In particular, incidental captures in gillnets are responsible for the drowning of hundreds of thousands of marine birds annually. In the European Union (EU), the existing biodiversity legislation (Birds and Habitats Directives) theoretically imposes a comprehensive monitoring of incidental capture of seabirds to guarantee the long-term conservation of the populations. Moreover, the EU is engaged in an ecosystem approach to fisheries management to ensure the sustainability of fisheries in the Union. As such, the effects of fishing practices on the different components of the ecosystems – including seabirds – need to be monitored and mitigated if necessary. However, there is a general scarcity of reliable fisheries-dependent data on bird bycatch from most fisheries. Especially, limited information from small-scale gillnet fisheries, where bycatch rates are known to be high, strongly impairs our capacity to estimate bycatch mortality at population level. In recent years, remote electronic monitoring systems (EM) with closed-circuit television cameras (CCTV) have demonstrated their ability to monitor marine mammals bycatch in Danish gillnet fisheries. This thesis illustrates the current developments of the Danish EM bycatch monitoring programme, with a focus on bycatch of seabirds. Long time-series of high-quality seabird bycatch data from commercial gillnet vessels were collected with EM and used to determine the spatio-temporal variability of incidental captures of seabirds in gillnets and identify the main environmental and operational drivers of bycatch (Paper 1; Paper 2). Furthermore, this information was used to estimate fleet-wide bycatch mortality in a Danish gillnet fishery (Paper 2). Finally, technical mitigation measures using experimental seabird bycatch reduction devices were tested in real commercial fishing conditions (Paper 3).

In Paper 1, nine years of EM data from three Danish commercial gillnetters operating in the Western Baltic Sea were used to identify and quantify seabird bycatch. Most of the 700 drowned birds registered in this study (90%) belonged to three species only, common eider (*Somateria mollissima*), great cormorant (*Phalacrocorax carbo*) and common guillemot (*Uria aalge*). Recording seabird bycatch at haul level allowed identifying clear species-specific spatial and temporal bycatch variability patterns. Additionally, the records of the entire yearly fishing activity of the vessels demonstrated the overwhelming contribution of mass bycatch events to total bycatch mortality.

In Paper 2, fleet-wide seabird bycatch mortality was estimated in gillnet fisheries operating in Inner Danish waters. A simple scaling up of the bycatch rates from the vessels monitored with EM provided highly variable seabird mortality estimates with wide confidence intervals, notably because of few, very influential mass bycatch events. A model was developed to predict seabird bycatch from fine-scale environmental and operational factors. Additional fine-scale fishing effort data, obtained from the statistical treatment of AIS data from the commercial fleet, were fed to the model to calculate refined mean bycatch rate estimates in the area. These conservative bycatch rate estimates were scaled up to fleet level, revealing fleet-wide bycatch mortality.

In Paper 3, bycatch reduction devices (BRDs) were tested in a commercial gillnet fishery with a known seabird bycatch problem. The experimental BRDs consisted of flashing white LED lights attached to the net panels and of 3 kHz acoustic pingers also fixed on the nets. Despite potentially encouraging results for the LED at reducing the bycatch rates of pelagic-diving seabirds, the low number of birds captured during the experiment did not allow to draw definitive conclusions. Pingers on the other hand showed no signs of reducing bycatch of seabirds.

Résumé (Dansk)

I løbet af de sidste årtier har mange bestande af havfugle oplevet hurtige nedgange på grund af en række trusler, herunder interaktioner med fiskeriet. Specielt er garnfiskeriet ansvarligt for over hundrede tusind bifangster af havfugle årligt. I Den Europæiske Union (EU) har den eksisterende biodiversitetslovgivning (Fugle- og Habitatdirektiverne) teoretisk set indført en omfattende overvågning af bifangster af havfugle for på lang sigt at kunne sikre en gunstig bevaringsstatus for havfuglepopulationerne. Derudover har EU vedtaget en økosystemsbaseret tilgang til fiskeriforvaltning for at sikre bæredygtigheden af de forskellige fiskerier i EU. Her skal fiskerieffekterne på de forskellige komponenter af økosystemet overvåges og afhjælpes hvis nødvendigt, herunder bifangster af havfugle. Der er dog en generel mangel på pålidelige data vedrørende bifangster af havfugle i mange fiskerier. Især er pålidelige data fra garnfiskeriet meget begrænsede. Bifangstraterne er ellers kendt for at være høje i netop disse fiskerier, men manglen på pålidelige data vanskeliggør mulighederne for at kunne beregne effekterne af bifangsterne på populationsniveau. I de seneste år har elektroniske moniteringsystemer (EM) demonstreret at kunne overvåge bifangst af havpattedyr og havfugle i dansk garnfiskeri.

Denne afhandling belyser den aktuelle udvikling af det danske overvågningsprogram for bifangster med fokus på bifangst af havfugle. Lange tidsserier af højkvalitets bifangstdata for havfugle fra kommercielle garnfartøjer er blevet indsamlet med EM og brugt til at bestemme den rumlige og tidsmæssige variation i bifangster af havfugle i garnfiskeriet, og til at identificere de vigtigste miljømæssige og operationelle faktorer, der er bestemmende for bifangsten (Artikel 1; Artikel 2). Desuden er disse oplysninger brugt til at estimere totale bifangster af havfugle i dansk garnfiskeri (Artikel 2). Endelig blev tekniske afværgningeforanstaltninger til reduktion af havfuglebifangst i kommercielle garnfiskerier afprøvet (Artikel 3).

I Artikel 1 blev ni års EM-data fra tre danske kommercielle garnfartøjer, der opererer i den vestlige Østersø, brugt til at identificere og kvantificere bifangst af havfugle. De fleste af de 700 bifangne havfugle, der blev registreret i dette studie, tilhørte 90% kun tre arter, edderfugl (Somateria mollissima), skarv (Phalacrocorax carbo) og lomvie (Uria aalge). Registreringen af bifangsterne blev udført på trækniveau, hvilket gjorde det muligt at identificere klare artsspecifikke, rumlige og temporale bifangstvariationer. Derudover demonstrerede den totale monitering, at et stort antal massebifangster bidrog til den totale bifangst. I Artikel 2 blev den totale bifangst af havfugle estimeret for indre danske farvande. En simpel ekstrapolering af bifangstfangstraterne fra de EM-overvågede fartøjer gav meget varierende estimater af bifangne havfugle med store usikkerheder, navnlig på grund af få, meget betydende massebifangster. Der blev udviklet en model til at forudsige bifangst af havfugle udfra miljø- og operationelle faktorer. Yderligere data om finskala fiskeriindsats, baseret på AIS-data fra den kommercielle flåde, blev anvendt i modellen for at beregne gennemsnitlige bifangstrater i de forskellige områder. Disse konservative estimater for bifangstrater blev ekstrapoleret til flådeniveau, hvilket gjorde det muligt at beregne bifangsten af havfugle i hele flåden.

I Artikel 3 blev 2 typer af afværgningeforanstaltninger testet i det kommercielle garnfiskeri. De eksperimentelle test bestod i henholdsvis at teste blinkende hvide LED-lys og i at teste 3 kHz akustiske pingere. På trods af potentielt positive resultater ved brug af LED-lys til reduktion af bifangstraten for pelagisk dykkende havfugle, tillader det lave antal fugle, der blev fanget under forsøget, ikke at drage endelige konklusioner. Pingerne viste på den anden side ingen tegn på at kunne reducere bifangsten af havfugle.

List of Ph.D. papers

Paper I

Assessing seabird bycatch in gillnet fisheries using electronic monitoring

Status: published in Biological Conservation

Glemarec, G., Kindt-Larsen, L., Lundgaard, L. S., & Larsen, F. (2020). Assessing seabird bycatch in gillnet fisheries using electronic monitoring. Biological Conservation, 243, 108461.

Paper II

Estimating seabird bycatch in a small-scale commercial gillnet fisheries

Gildas Glemarec, Morten Frederiksen, Lotte Kindt-Larsen, Finn Larsen

Status: submitted to Marine Ecology Progress Series

Paper III

Lights reduce seabird bycatch in a Western Baltic Sea demersal gillnet fisheries, but pingers do not

Gildas Glemarec, Anne-Mette Kroner, Lotte Kindt-Larsen, Finn Larsen

Status: submitted manuscript

Other noteworthy work during the Ph.D.

Published work

ICES, 2018. Report of the Joint OSPAR/HELCOM/ICES Working Group on Marine Birds (JWGBIRD), 1–5 October 2018, Ostende, Belgium. ICES CM 2017/ACOM:24. 79pp.

Christensen-Dalsgaard, S., Anker-Nilssen, T., Crawford, R., Bond, A., Sigurðsson, G.M., **Glemarec, G.**, Hansen, E.S., Kadin, M., Kindt-Larsen, L., Mallory, M., & Merkel, F.R. (2019). What's the catch with lumpsuckers? A North Atlantic study of seabird bycatch in lumpsucker gillnet fisheries. *Biological Conservation, 240*, 108278.

Working group meetings

Joint OSPAR/HELCOM/ICES Working Group on Marine Birds (JWGBIRD). *1 – 5th October 2018, Ostende, Belgium.*

Working Group on Commercial Catches (WGCATCH). *4 – 8th November 2019; Gdańsk, Poland*

LIST OF ABBREVIATIONS

ABR. Auditory Brainstem Response

AIS. Automatic Identification System

BPUE. Bycatch per Unit Effort

BRD. Bycatch Reduction Device

CCTV. Closed-Circuit Television

CFP. European Union Common Fisheries Policy

CPUE. Catch per Unit Effort

DCF. Data Collection Framework

EAF. Ecosystem-Approach to Fisheries management

EC. European Commission

EEC. European Economic Community

EEZ. Exclusive Economic Zone

EU. European Union

(R)EM. (Remote) Electronic Monitoring

EU-PoA. Action Plan for reducing incidental catches of seabirds in fishing gears

GES. Good Environmental Status

HELCOM. Helsinki Commission for Baltic marine protection

ICES. International Council for the Exploration of the Sea

IUCN. International Union for Conservation of Nature

MPA. Marine Protected Area

MSFD. Marine Strategy Framework Directive

OSPAR. OSlo-PARis Convention

PET. Protected, Endangered and Threatened species

VMS. Vessel Monitoring System

GLOSSARY

Benthic-feeding seabirds are marine birds who feed predominantly on prey living on the seafloor (e.g. some bivalves, crabs…), or inside of it (some worms, razor clams…). This group comprises a number of seaducks commonly seen in Danish waters like the common eider *Somateria mollissima*, the velvet scoter *Melanitta fusca* or the common scoter *Melanitta nigra*.

Biologgers are, in the largest acceptation of the term, electronic devices that researchers attach on animals to track their movements and/or collect and store ecological information from the animal's environment (temperature, depth, hours of daylight, or any other relevant data).

Bycatch (or by-catch) refers to the unintentional capture in fishing gears of commercial or non-commercial species. Some bycatch species may be desirable for fishers, e.g. turbot *Scophthalmus maximus* and plaice *Pleuronectes platessa* fisheries, whereas others are unwanted, in particular marine megafauna, including seabirds.

Fishery-dependent data refer to fisheries data (e.g. catch composition including bycatch, catch weight, fishing activity, fishing effort) collected on fishing vessels or on shore, either directly by fishers, or by fisheries observers and electronic monitoring.

Pelagic-feeding seabirds are marine birds who forage predominantly, but not necessarily exclusively, in the water column, typically feeding on mobile prey like fish. In Denmark, this group includes the great cormorant *Phalacrocorax carbo*, the common guillemot *Uria aalge* or the razorbill *Alca torda*.

Small-scale fisheries (as opposed to large-scale fisheries), also called artisanal fisheries (as opposed to industrial fisheries) are not clearly defined, but a general agreement is that they are constituted of fishing vessels below 12 metres overall length, which use non-towed gears and operate near shore with short-lasting fishing trips. This definition however needs to be supplemented with other potentially important elements like a generally higher job number in small-scale compared to large-scale fisheries, and thus its importance for local communities.

(Commercial fish) stock is defined as an identifiable portion of a species meta-population, isolated geographically and/or genetically from other or sub-populations of the same species.

Vulnerable taxa refer in this thesis to all the non-target, air-breathing marine animals captured unwillingly in fishing gears. This includes in particular seabirds, marine mammals (cetaceans and pinnipeds) and marine reptiles (sea turtles).

List of Species

List of all the species mentioned in the text, including scientific names, English vernacular and Danish vernacular names.

Scientific names	English vernacular	Dansk navn
Seabirds		
Somateria mollissima	Common eider	Ederfugl
Uria aalge	Common guillemot	Lomvie
Phalacrocorax carbo	Great cormorant	Skarv
Phalacrocorax aristotelis	European shag	Topskarv
Alca torda	Razorbill	Alk
Melanitta nigra	Common scoter	Sortand
Melanitta fuscus	Velvet scoter	Fløjlsand
Cerorhinca monocerata	Rhinoceros Auklet	Næsehornsalk
Morus bassanus	Northern gannet	Sule
Ardenna gravis	Great shearwater	Storskråpe
Clangula hyemalis	Long-tailed duck	Havlit
Leucocarbo bougainvillii	Guanay cormorant	Peruskarv
Sula variegata	Peruvian booby	Perusule
Spheniscus humboldti	Humboldt penguin	Humboldt pingvin
Ardenna creatopus	Pink-footed shearwater	Chileskråpe
Procellaria aequinoctialis	White-chinned petrel	Hvidhaget Skråpe
Aythya marila	Greater scaup	Bjergand

Scientific names	English vernacular	Dansk navn
Bony fish		
Gadus morhua	Cod	Torsk
Pleuronectes platessa	European plaice	Rødspætte
Cyclopterus lumpus	Lumpsucker	Stenbider
Scophthalmus maximus	Turbot	Pighvar
Clupea harengus	Atlantic herring	Sild
Osmerus eperlanus	Smelt	Smelt
Oncorhynchus keta	Sockeye salmon	Stillehavslaks
Marine mammals		
Phocoena phocoena	Harbour porpoise	Marsvin
Lipotes vexillifer	Baiji	Baiji
Phocoena sinus	Vaquita	Vaquita

1. GENERAL INTRODUCTION

The incidental capture (bycatch) of birds in net fisheries, and how this phenomenon affects populations, have first been documented in Greenland almost half a century ago (Tull et al., 1972). Since then, a number of such examples have been reported worldwide, and the sustainability of fisheries for the conservation of marine megafauna has been questioned, not just for seabirds, but also for cetaceans, pinnipeds, sea turtles and elasmobranchs (Burgess et al., 2018; Komoroske and Lewison, 2015; Lewison et al., 2004, 2014; Moore et al., 2009; Northridge, 2009; Soykan et al., 2008; Žydelis, 2013). However, there is still a limited understanding of which factors influence seabird bycatch mortality in net fisheries, and large knowledge gaps remain on the best methods to mitigate seabird bycatch (ICES, 2018; Northridge et al., 2017).

Problematically, seabird bycatch data are only rarely recorded (if ever) in many fisheries. For a number of target species, detailed statistics are routinely collected by fisheries observers, independent of the fishing industry, which allow fisheries managers to take informed decisions. Yet, bycatches of seabird in fishing gears is often highly variable in time and space, and observers coverage is usually not sufficient to estimate fleet-wide mortality (Lewison, 2013). In the European Union (EU), despite the obligation for fishers to record all incidental catches of protected species, including seabirds (EU, 2017a), bycatch data are only rarely reported in official logbooks. Fisheries observers mostly concentrate their effort on large-scale fisheries, and essentially ignore small-scale coastal net fisheries, despite a very large contribution of the latter to seabird mortality (Bellebaum et al., 2013; Bradbury et al., 2017; Dias et al., 2019; ICES, 2018; Sonntag et al., 2012; Žydelis, 2013; Žydelis et al., 2009). Recent technological developments such as electronic monitoring (EM) with Closed-Circuit Television cameras (CCTV) can provide large quantities of fine-scale quality fisheries-dependent data to supplement fisheries observers data (Bartholomew et al., 2018; Kindt-Larsen et al., 2012a). In small-scale fisheries, where observer coverage is low, EM is a possible way forward to collect data and estimate the spatial and temporal variations in bycatch mortality at fleet level.

The initial goal of this Ph.D. was to **investigate whether EM was a suitable solution for collecting and assessing the magnitude of seabird bycatch** in Danish commercial small-scale gillnet fisheries (Paper 1). In Denmark, a unique dataset compiling EM with CCTV from coastal gillnetters going back to 2010 was available to estimate fine-scale fleet-wide fishing effort and bycatch per unit effort (BPUE) of vulnerable marine birds. Using the experience from the first project, the second phase of the Ph.D. focused on **estimating fleet-wide seabird bycatch mortality in gillnets** in the Danish Belt Seas, using (i) EM data from a sample of the fleet to model bycatch rates based on operational and ecological factors, (ii) available Automatic Identification System (AIS) data to model fine-scale variations of the fishing effort, and (iii) official logbooks and sales notes to scale up BPUE estimates to fleet level and calculate total mortality (Paper 2). Finally, methods to **mitigate seabird bycatch in gillnets, using sound and light**, were tested in real commercial fishing conditions (Paper 3).

1.1. SEABIRD BYCATCH: A GLOBAL PROBLEM

With 359 species belonging to 17 families, seabirds represent about 3.5% of all known bird species (Dias et al., 2019). Members of this diverse group are chiefly characterised by their dependence of the marine environment for at least some parts of the year (Croxall et al., 2012). Seabirds can interact with fisheries in different ways, i.e. competing for a shared resource (Cury et al., 2011), predating on fishing gears (Kumar et al., 2016), or feeding on offal and discards (Soriano-Redondo et al., 2016). Fishers may also use at-sea aggregations of seabirds to detect the large shoals of fish that the birds predate upon (Crawford and Shelton, 1978). However, this proximity with fishing vessels also exposes marine birds to additional threats. In fact, seabirds are among the most threatened group of birds, and are particularly threatened by incidental captures (bycatch) in fishing gears (Dias et al., 2019). In the literature, the word bycatch has a wide range of definitions. In general, it describes the unintentional catch of non-commercial and/or commercial species, including undersized fish, as well as the capture of protected, endangered and threatened species (PETs) (Alverson et al., 1994; Davies et al., 2009; FAO, 2011; Kelleher and Nations, 2005). For the sake of brevity, and unless stated otherwise, bycatch should be understood in this thesis specifically as the direct mortality resulting from the interactions between PET and fishing gears, with a specific focus on the bycatch of seabirds.

Bycatch is a recurring issue in commercial and recreational fisheries worldwide and a major threat to marine megafauna. Air-breathing animals including seabirds, sea turtles, cetaceans or pinnipeds, who need to resurface regularly, are particularly at risk when foraging in areas where fishing occurs (Northridge et al., 2017). Globally, the unintentional catch of reptiles, mammals and birds kills millions of animals each year (Anderson et al., 2011; Dias et al., 2019; Lewison et al., 2004; Žydelis, 2013), often affecting entire populations (Pardo et al., 2017) and participating in the (probable) extinction of some species, e.g. the baiji *Lipotes vexillifer* in China (Turvey et al., 2007) or the eminent loss of others, e.g. the vaquita *Phocoena sinus* in Mexico (Taylor et al., 2017).

Less than a third of all seabird species (100) are directly at risk of incidental capture in fishing gears, which corresponds to approximately 22% of the global seabird population (ca. 190 million individuals) (Dias et al., 2019). Nevertheless, important knowledge gaps exist regarding the actual scale of seabird bycatch mortality in a number of fisheries. Generally, large-scale fisheries undergo more thorough controls than small-scale fisheries, e.g. EC (2017). As a result, the impact of fishing on the marine environment, and in particular on seabirds, is better known in large- than in small-scale fisheries. Reliable data on seabird bycatch is not systematically available and varies widely between regions, fisheries, vessel size, gear types, and, within the same area, there can also be important disparities between seasons (Pott and Wiedenfeld, 2017). In many countries, fisheries observer programmes collect bycatch data routinely. However, the cost of sending observers on-board limits the number of vessels on which they can be deployed, leaving some fisheries largely unmonitored (Le Bot et al., 2018). Besides, the distribution of seabirds varies spatially and temporally, so in most cases, extrapolating the number of observed casualties to fleet level is challenging. Yet, long-time series datasets are essential to understand the conditions that lead to the bycatch of seabirds, and eventually to tackle it (Northridge et al., 2017). Nonetheless, the estimated average scope (i.e. the percentage of the population affected) and severity (i.e. rapidity at which a population declines because of bycatch) is similar for small- and large-scale fisheries (Dias et al., 2019). Almost all types of fishing gears have been documented to affect seabirds, but passive gears have the highest reported number of interactions (Dias et al., 2019; Pott and Wiedenfeld, 2017; Tasker, 2000). For instance, the death toll is estimated to be more than 160,000 animals per year in hook and line fisheries globally (Anderson et al., 2011), while at least 400,000 birds drown annually in gillnet fisheries alone (Žydelis, 2013), making the latter the most deadly gear type for seabirds worldwide.

1.2. FISHING WITH STATIC NETS

i. General description of net fisheries

Gillnet is commonly used to refer collectively to different types of nets. Here, fishing with gillnets encompasses fishing with "true" gillnets, trammel nets and entangling nets. Gillnets have been traditionally constructed with cotton or hemp fibres, but are mostly made of nylon since the 1950's (Gabriel and Brandt, 2005). Target species are captured by swimming into the net and can be retained in various ways, the main method of capture being gilling (the catch is retained by the gills; Figure 1). Depending on the size and shape of the catch, and the physical properties of the gear, the animals can also be wedged (the catch is retained by the body), jawed (the catch is retained by the jaw), or entangled. Gillnets are relatively size selective in that small fish (relative to the mesh size) will likely swim through the net, while larger individuals will tend to bounce off of it (He and Pol, 2010).

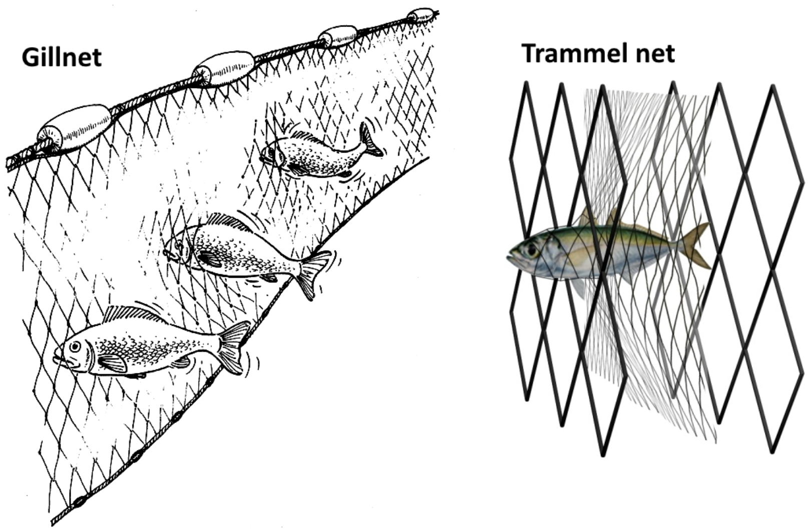

***Figure 1: Capture methods in gillnets** [left – modified from Rosman and Maugeri (1980)], and in trammel nets [right – from Seafish.org (2018)].*

A net fleet is composed of a series of net panels held together by a rope frame. The upper rope (a.k.a. floatline, cork line or headrope) is positively buoyant, while the bottom rope (a.k.a. leadline or footrope) is weighted (Figure 2a); therefore, with the stretch exerted on both sides, the panels are maintained opened (He and Pol, 2010). Strictly speaking, "true" gillnets (Figure 1, left) are made of one single layer of webbing, whereas trammel nets (Figure 1, right) are composed of one layer of thin-meshed net panel enchased between two layers of large-meshed nettings. Loosening the vertical tension of the panels, e.g. by using less floats on the headrope, will create a slack; such nets are usually defined as entangling nets.

The length of one net panel is typically less than one hundred meters, but a full net fleet is commonly made of several panels attached together (Figure 2). Depending on the fishers' needs and preferences, the total length of a net fleet may vary from a few hundred meters to more than ten kilometres. The type of floats and weights will control the position of the net in the water column (Figure 2); a set-net is anchored to the sea floor (Figure 2a), whereas a driftnet is not and hangs from the surface (Figure 2b). A bottom-set net is a type of net moored at both ends of the fleet, with the leadline ensuring a continuous contact with the sea floor (Figure 2d). Bottom-set nets typically target benthic (e.g. flatfish) and benthopelagic species (e.g. cod *Gadus morhua*). On the contrary, driftnets are designed to capture pelagic species near the surface (e.g. salmon, swordfish or tuna). All intermediate configurations can be utilised, such as sub-surface driftnets (Figure 2c), or off-bottom gillnets (Figure

2e). To maximize the catch of target species, fishers can adapt the design of their net by varying key characteristics among which panel height, mesh size, twine material and diameter, twine colour and hanging ratio.

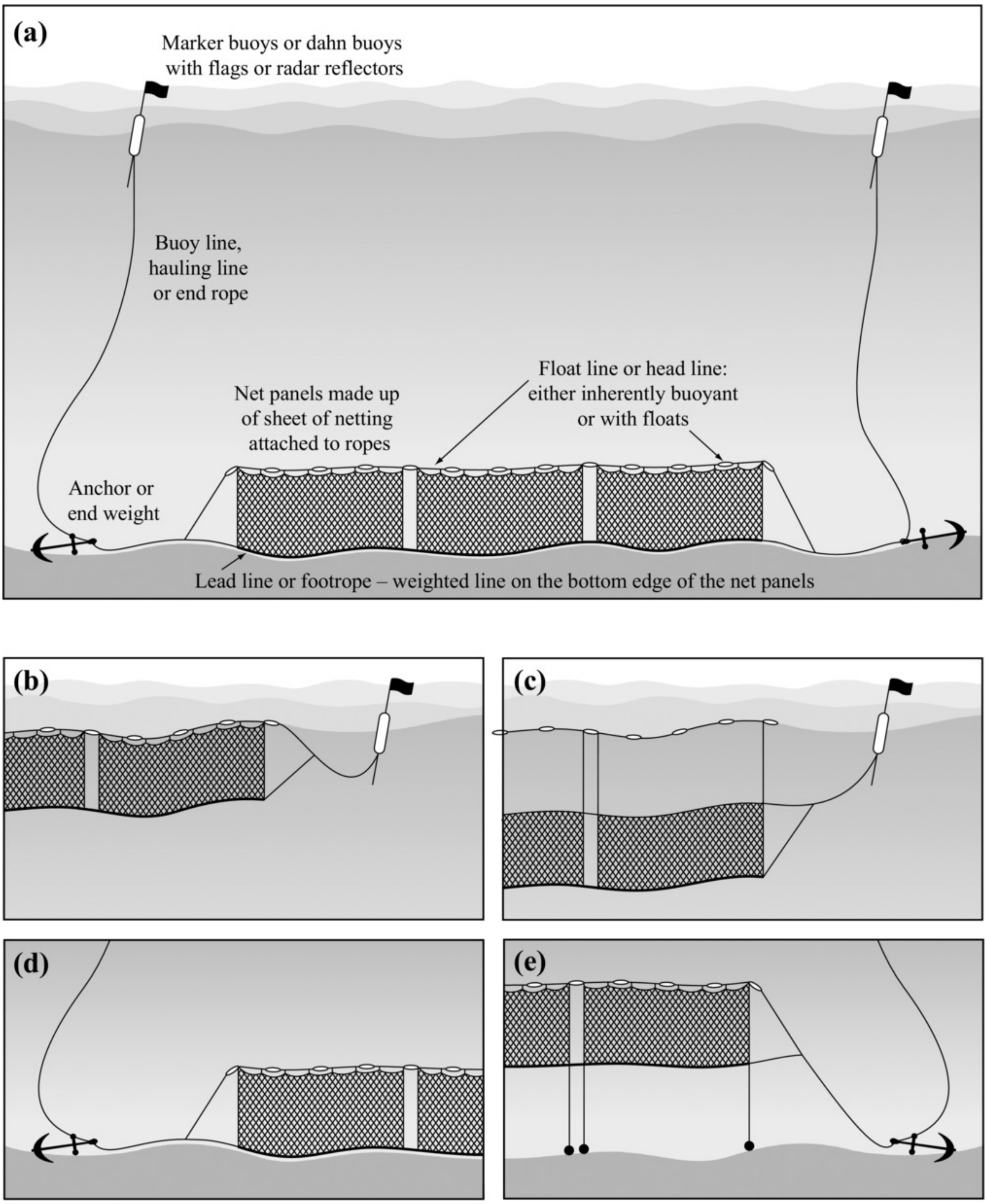

Figure 2: Schematic of the main types of gillnets. *(a) Typical bottom-set gillnet and its main components; (b) standard driftnet; (c) sub-surface driftnet; (d) standard bottom-set gillnet; and (e) off-bottom set-gillnet. Modified from Northridge et al. (2017).*

Depending on the target species, location, season, weather or currents, the nets can be soaked (i.e. left in the water to capture fish) from less than one hour (e.g. some wreck fisheries), up to several days (e.g. fisheries targeting lumpsucker *Cyclopterus lumpus* or turbot *Scophthalmus maximus*). Nets are often hauled manually on small vessels, or using a mechanical net hauler on larger commercial netters. Because each catch needs to be disentangled by hand, the work on board is particularly labour intensive. Consequently, hauls with a large number of catches have a significant impact on the overall handling time. This, in turn, increases the duration of the fishing trips and makes them less profitable (Bellido et al., 2011; Morandeau et al., 2014). Therefore, it is in the fishers' interest to capture only the target species for which the market value is maximal, and to avoid bycatch species of low or no market value, which includes seabirds (Savina et al., 2017). Furthermore, because handling time limits the overall fishing power, i.e. the number and/or the length of nets a vessel can operate during one fishing trip, gillnet vessels are often smaller compared to vessels using active gears. Consequently, overall boat length, gross tonnage or engine power are poor descriptors of the fishing power of a gillnet vessel. Nonetheless, larger vessels are more versatile, can store more nets, or nets of different mesh sizes, and can operate in rougher weather, which ensures more days at sea on average than what is possible on smaller vessels. In addition, some fishing opportunities are only opened to larger gillnet vessels, e.g. when fishing in deep waters, gillnetters need to use large and powerful hauler that cannot be accommodated on a small ship (Savina, 2018).

ii. The Danish bottom-set gillnet fleet

The number of gillnet vessels in Denmark has more than halved since the 1990's, but gillnets remain one of the major gear type in the country (Andersen et al., 2012). About 44% of all registered fishing vessels in Denmark declare using gillnet as a primary or secondary gear, making it the most common fishing gear nationally [data from 2010-2018, Fiskeridirektoratet (2020)]. The most important harbours in terms of landings are located in Jutland on the west coast of Denmark (Savina, 2018), but gillnets are utilised all around the country.

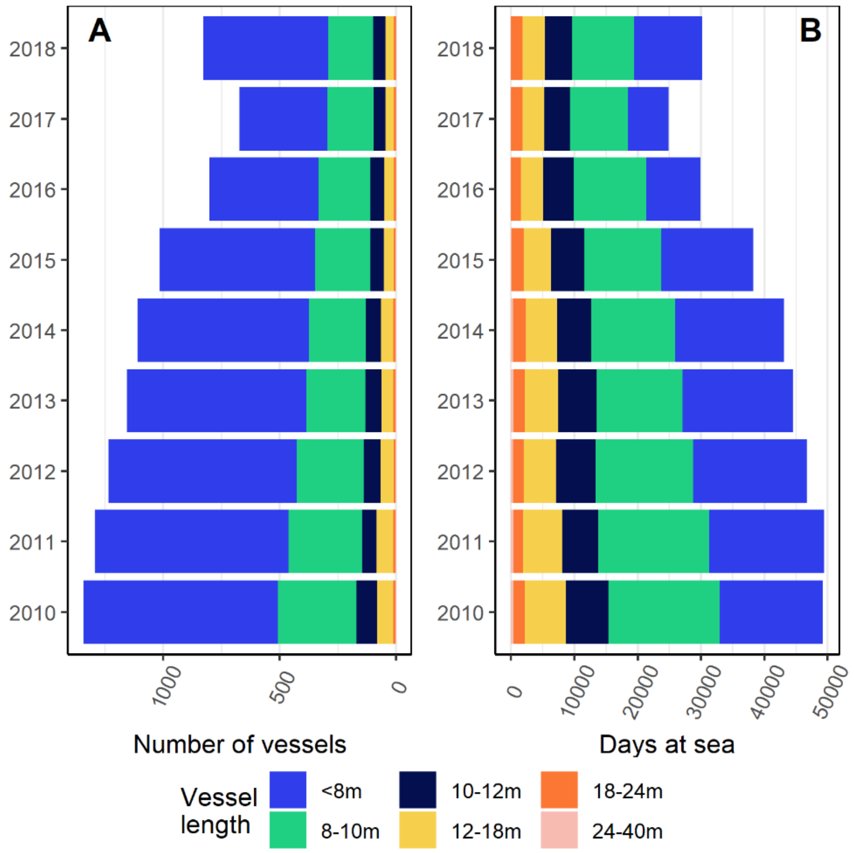

Figure 3: Summary of the Danish gillnet fleet. *A. Number of vessels per length class per year; B. Yearly fishing effort (in days at sea) per length class per year. The data presented here cover all Danish vessels having registered gillnet as their primary or secondary gear, and declared landings at least once in a calendar year.*

Although popular, fishing with gillnets contributes only marginally to national landings. Between 2014 and 2018, the part of the landings provided by the Danish gillnet fleet represented on average 1.46% (SD = 0.24) of the total landings in weight from all fishing gears combined, excluding industrial species (Logbook and Sales notes data from the Danish Fisheries Agency, processed by DTU Aqua). The vast majority of Danish gillnetters are below 8 m in overall length (Figure 3a). Small-scale vessels, which are not build for long and distant fishing trips, usually go out at sea for short durations, typically less than one day, yet they contribute to most of the total registered fishing effort (Figure 3b). On the contrary, larger vessels can sustain rougher weather conditions and carry more equipment on-board (e.g. different types of nets, longer net fleets, etc…), but also switch to other gears seasonally if required. Consequently, the mean yearly fishing effort per vessel (in days at sea) is typically higher for larger than for smaller length classes (Figure 4).

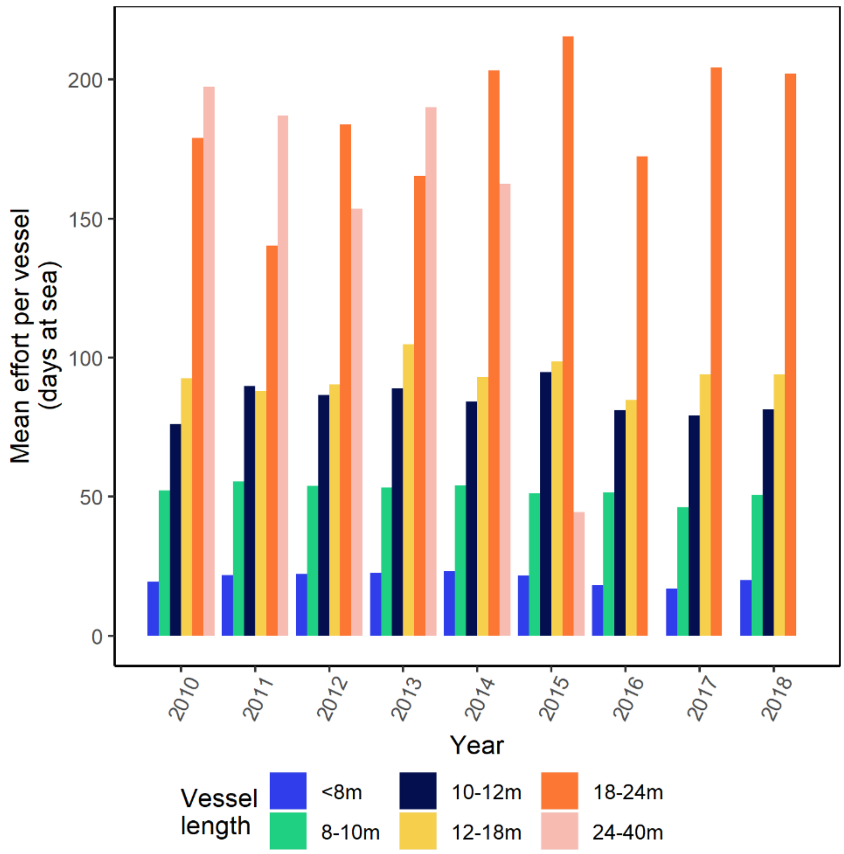

Figure 4: Mean fishing effort per vessel grouped per length class per year. *The data presented here cover all Danish vessels having registered gillnet as their primary or secondary gear, and declared landings at least once in a calendar year.*

Historically, fisheries-dependent data collection in Denmark has focused on active gears (Savina, 2018), partly because of the low contribution of gillnets to national landings, but also because overall gillnet fishing effort is spread over a large number of vessels. In recent years, fisheries observers, in charge of collecting scientific data from fisheries, covered annually only 0.1% of the total gillnet fishing effort [data from 2019, Anonymous (2019)]. Besides, *days at sea*, the effort metric reported in official logbooks and used to evaluate fleet-wide fishing effort, is relatively uninformative. For instance, small one-crewed gillnetters may set few short nets per fishing trip (or day at sea), while the largest commercial vessels could soak dozen of kilometres of nets in the same amount of time. Additionally, Danish gillnet fishers do not have to report the length and height of their nets, the number of fleets or the soak duration. Consequently, there is generally a scarcity of information from the Danish gillnet fleet.

1.3. The European Union legislative framework

Despite a consensus that healthy marine ecosystems are fundamental for the sustainability of the world's fisheries, fisheries management has traditionally focused on maximising the catch of single target species, ignoring the potential negative effects of fisheries activities on the marine environment (Bellido et al., 2011). In recent years however, the fishing sector has undergone a profound transformation with the introduction of ecosystem approaches to fisheries management, also referred to as ecosystem-based management (EBM) (Pikitch, 2004). Typically, EBM "acknowledges the complexity and interspecies relationship within ecological systems, but many also account for social and governance objectives, with the latter aspects broadening the range of definitions" (Long et al., 2015).

The European Union (EU) has taken commitments to safeguard marine ecosystems against unsustainable development, which is in line with international agreements such as the United Nations Convention on the Law of the Sea (UNCLOS), the United Nations Convention on Biological Diversity (CBD) and the Convention on the Conservation of Migratory Species of Wild Animals (CMS). However, in the EU, there has been a historical conflict between regulations pertaining to nature and biodiversity conservation – the Birds and Habitats Directives – and the ones related to fisheries management – the Common Fisheries Policy (CFP) (Appleby and Harrison, 2019; Owen and Chambers, 2004). With the introduction of the Marine Strategy Framework Directive (MSFD) in 2007, and later with the reform of the CFP in 2013, the EU has sought to make these different policies more coherent, and engaged in ecosystem approach to fisheries management. The CFP describes ecosystem-based management as "an integrated approach to managing fisheries within ecologically meaningful boundaries which seeks to manage the use of natural resources, taking account of fishing and other human activities, while preserving both the biological wealth and the biological processes necessary to safeguard the composition, structure and functioning of the habitats of the ecosystem affected, by taking into account the knowledge and uncertainties regarding biotic, abiotic and human components of ecosystems" (EU, 2013). Therefore, the potentially negative impacts of fishing activities on the marine ecosystems (including seabirds) must now be considered in the management of fisheries, and, if necessary, mitigated.

i. Council Directive 2009/147/EC on the conservation of wild birds (Birds Directive)

The conservation of wild birds is at the heart of the European Union legislation to protect and maintain biodiversity. The first Birds Directive (EEC, 1979) was adopted in 1979 and constituted one of the earliest attempt of environmental law at international level. Revised in 2009, the Birds Directive (EU, 2010) remains a cornerstone of the nature and biodiversity laws in the EU. Both individual species and habitats are given protection measures. As such, the deliberate killing of naturally occurring birds is prohibited in the EU, as is their intentional capture, trade or disturbance; some bird species are nonetheless exempted under, e.g. hunting, and listed in Annex II of the Directive. In terms of site protection, areas of international value should be protected and designated as Special Protection Areas (SPAs). Member States need to ensure sufficient habitat for all wild species, and ensure that effective monitoring programmes are in place to achieve the objectives of the Directive.

ii. Council Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora (Habitats Directive)

The Habitats Directive (EEC, 1992) is often considered as the Birds Directive's twin sister. Established in 1992, it aims at protecting both habitats (Annex I) and the wild species that live in these habitats (Annex II), using a network of Special Areas of Conservation (SCA). Some species benefit from strict protection measures in the Union, prohibiting killing, trade or capture (Annex IV of the Habitats Directive). Additionally, SCAs and SPAs form the Natura 2000 network, which aims at being a coherent network of protected areas favouring biodiversity at the continent scale. As is the case with the Birds Directive, monitoring programmes must be implemented to ensure that the measures to protect habitats and species are effective.

iii. Directive 2008/56/EC of the European Parliament and of the Council establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive)

The Marine Strategy Framework Directive (MSFD) applies an ecosystem approach to the management of the marine environment in the EU and aims at ensuring the sustainable use of EU marine waters (EC, 2008a). The Directive defines the Good Environmental Status (GES) of marine ecosystems with eleven *qualitative descriptors* (Annex I), each of them describing one specific aspect of the marine environment, e.g. biodiversity (Descriptor 1), or sustainability of fish stocks (Descriptor 3). The criteria and methodological standards to assess these descriptors proved difficult to interpret for most Member States. After identifying shortcomings in the initial implementation of the Directive, the Commission established a more flexible framework in 2017 (EU 2017a). Bycatch of protected species (including seabirds) is addressed under criterion D1C1, stating, "Member States shall establish the threshold values for the mortality rate from incidental by-catch per species, through regional or sub-regional cooperation".

iv. Regulation (EU) No 1380/2013 of the European Parliament and of the Council on the Common Fisheries Policy

A strong focus of the earliest versions of the CFP (1983, 1992 and 2002) was to ensure the sustainability of single fish stocks. Some of the criticism of the CFP back then pertained to its inherent incapacity to incentivise sustainable fishing practices (Daw and Gray, 2005; Sissenwine and Symes, 2007). The reform of the CFP in 2013 introduced an ecosystem-approach to fisheries management (EAF), which aimed at warranting the sustainability of European fisheries and at contributing to the objectives of the MSFD, i.e. the achievement of Good Environmental Status in all EU marine waters. To ensure compliance, Member States are required to collect fisheries-related data of biological, environmental and economic nature, in accordance with the Data Collection Framework, or DCF (EU, 2017a). This includes the collection of data on "incidental by-catch of all birds, mammals and reptiles and fish protected under Union legislation and international agreements (…) for all types of fisheries". The multi-annual Union programme for data collection (EU-MAP) coordinates the data collection at national and regional level, which ensures comparability and reduces costs.

In theory, these data should provide accurate information on spatial and temporal variations of bycatch events, but information on seabird bycatch (and other protected species) remains largely missing in most fisheries (Konrad et al., 2019). Taking note of these facts, and more generally that it was not going to achieve some of its objectives, the CFP was amended in 2019 with the technical measures regulation, which "lay[ed] down technical measures concerning (…) the interaction of fishing activities with marine ecosystems" (EU, 2019a).

v. Action Plan for reducing incidental catches of seabirds in fishing gears

The changes introduced with the revised CFP in 2013 concerning fisheries management, i.e. moving toward an ecosystem approach to fisheries management, required to develop a framework to assess the magnitude of seabird bycatch in European fisheries, but also to find ways of preventing and possibly eliminate incidental catches of vulnerable birds. Before the reform of the CFP was officially enacted, the EU "Action Plan for reducing incidental catches of seabirds in fishing gears" (EU-PoA) already pointed out a series of objectives to achieve these goals (EC, 2012). This included, among others, to determinate the caveats in the current management plans, to establish a solid framework for data collection of seabird bycatch in EU fisheries, and to come up with methods to minimise incidental captures in problematic fisheries, especially for the most threatened groups of birds. Therefore, Member States were asked to provide accurate data on seabird bycatch at national and regional level in order to tackle the problem of seabird bycatch and to comply with the MSFD obligation to achieve GES in all EU waters by 2020. As of today, and although the effectiveness of the EU-PoA has not fully been evaluated yet, it seems evident that serious deficiencies remain to achieve these goals, as many Member States failed to monitor seabird bycatch in their own fisheries and/or to address the problem by implementing effective mitigation measures (ICES, 2018).

2. MONITORING SEABIRD BYCATCH IN SMALL-SCALE FISHERIES

Seabird bycatch mortality in gillnets is a serious threat to several vulnerable populations (Section 1.1), but, probably because of the relatively low contribution of gillnet fisheries to total landings in most EU countries (Section 1.2.ii), this issue remains largely under-monitored, in particular in a number of small-scale fisheries (Section 1.3). Reliable data for fleet-wide fishing effort and bycatch per unit effort (BPUE) are nevertheless fundamental knowledge to acquire in order to adapt management decisions appropriately, so that both objectives of biodiversity conservation and economic sustainability of fisheries can be achieved (Le Bot et al., 2018). Below, different methods to collect fisheries-related data are described and their interest for establishing reliable seabird bycatch estimates is discussed.

2.1. SEABIRD BYCATCH MONITORING IN FISHERIES

i. Fisheries observers programmes

In the EU, data generated by fisheries observers constitute a cornerstone of the CFP (EU, 2013), and provides essential information on stocks caught by commercial and recreational fisheries in- and outside of the Union (EU, 2017a). Fisheries observer work generates high-quality data for scientific stock assessments and for evaluating the impact of fisheries on marine ecosystems. However, the cost of observers can be prohibitively high, in particular if a large proportion of the fishing fleet needs to be surveyed (Mangi et al., 2015). For practical reasons, observers generally cannot be randomly allocated on-board fishing vessels, which can potentially bias the data that they collect. Besides, observers follow sampling protocols designed primarily to collect data for commercial species; they often lack training with regard to bycatch of protected species, and they may not be able to fulfil the contradictory requirements of collecting biological fish samples and monitor potential incidental captures simultaneously (ICES, 2019). Moreover, the "observer effect", which is the changes in fishing practices that the presence of an observer aboard a fishing vessel may induce, can weaken the representativity of the observers-collected data. This is particularly true for monitoring illegal practices (e.g. discards) or captures of rare species (Babcock et al., 2003). Additionally, as observers cover on average less than 2% of all the fishing trips in the EU (Mangi et al., 2015), and because incidental catch rates of unwanted species (i.e., BPUE) are low in most fisheries, total bycatch estimates are likely imprecise using only this source of data (Amande et al., 2012).

In the EU, few monitoring programmes integrate seabird bycatch in their protocols, and they are for the most part carried out by fisheries observers in relation to the national and/or regional fisheries data collection programmes (DCF and EU-MAP) (ICES, 2019). The Joint OSPAR/HELCOM/ICES Working Group on Marine Birds (JWGBIRD) compiled a list of the existing monitoring programmes in the EU and Norway, in which seabird bycatches are recorded [Table 8.1 in the JWGBIRD Report 2018; ICES (2018)]. Gillnet fisheries are the most problematic with regards to seabird bycatch, but represent only 5 out of the 32 listed monitoring programmes, with a fleet coverage systematically below 1%. Alongside the difficulties to estimate fleet-wide seabird mortality based on such small samples, fishing effort data are also often simply lacking for small-scale gillnet fisheries (ICES, 2018). Although fisheries observer data are generally considered the most accurate way of collecting fisheries-related data

(Babcock et al., 2003), alternative methods have been developed to obtain bycatch and effort estimates, especially for fisheries in which observer coverage is low.

ii. Self-reporting

Self-reported data relates to fisheries-related information provided directly by skippers, e.g. from sales notes and logbooks. In the EU, logbooks are only mandatory for vessels >10 m (>8 m in the Baltic Sea), and e-logs (i.e. electronic logbooks) for vessels >12 m (EC, 2009; EU, 2011, 2015, 2016, 2019a). However, these documents are not an exhaustive documentation on target (landings + discards) and bycatch species, or on fishing effort. In particular, only catches of species kept on board and above 50 kg must be recorded (there are exceptions); that is, registrations of seabirds bycatches is not mandatory (again, there are exception, e.g. in the General Fisheries Commission for the Mediterranean area; EU 2015). As a result, data from national fisheries statistics compiling individual logbooks generally does not contain occurrences of seabird bycatch. In addition, for gillnets and entangling nets, fishing effort is to be reported in logbooks as the "number of times nets [are] shot during the day", whereas mentions of length and height of the net fleets are only optional (EU, 2011). In Denmark, for instance, fishing effort is reported as days at sea per ICES statistical rectangle and there is no obligation to mention the number of nets used during one fishing trip, the length of the net fleets, or the estimated soak time. In addition, for various reasons, fishers may misreport part of their catch or fishing effort (Hentati-Sundberg et al., 2014). Self-reported logbook data being generally not validated, this raises important reliability concerns (Mangi et al., 2015). Consequently, logbooks and sales notes data should not be the unique source of data for estimating seabird bycatch in gillnet fisheries.

Official data collection programmes can supplement fisheries observer data with self-sampled data from the fishing industry, in particular in fisheries where observer coverage is deemed not sufficient. In principle, these self-sampled data should be of high quality, as long as fishers compliance is maintained (e.g. Hoare et al. 2011). However, the captures of protected species may be underestimated on self-sampling vessels, if there is an incentive not to report them (Mangi et al., 2015). In Norway, an advanced self-sampling programme has been established since 2001 to obtain long time series of detailed information on commercial catches and fishing activity. A number of contracted commercial fishing vessels, referred to as the Norwegian Reference Fleet, collects and reports comprehensive fishing data for the account of the Norwegian Institute of Marine Research (Institute of Marine Research, 2020). These data include comprehensive records of the fishing activity (using e-logs), total target species catches and discards, as well as self-sampled biological data (e.g. length frequency measurements, otoliths, etc…). Incidental captures of seabirds are also registered, so that fleet-wide mortality estimates could be calculated in the whole coastal Norwegian gillnet fishery, an estimation spanning between 1580 and 11500 individuals per year (Bærum et al., 2019). Nevertheless, several authors argue that data from reference fleets may be biased, and should not be used as the exclusive source of data (Mangi et al., 2015; Roman et al., 2011).

iii. Indirect evidence of seabird bycatch mortality

In the absence of direct evidence of bycatch (e.g. from fisheries observers or logbook registrations), carcass collection from voluntary fishers or from stranding can provide information on the magnitude of incidental seabird captures in an area. Combining carcass collection from voluntary fishers, interviews and observer monitoring, a study estimated total seabird bycatch in gillnet fisheries in the German part of the Baltic Sea (Bellebaum et al., 2013). Because of the difficulties to obtain accurate estimates of the fishing effort from small-scale gillnetters in this area, the authors calculated BPUE as the yearly/monthly average number of birds per cooperating fisher, and then multiplied this BPUE estimate by the estimated total number of fisher per year/month in the area. The yearly number of seabirds bycatch in gillnets between November and May was estimated to be >17,000 individuals in the study area. These results were obtained assuming homogeneity between fishers regarding their individual contribution to total bycatch mortality in German coastal waters. However, it is probable that there is an important variability between fishers, and that specific fishing behaviour has a major influence on seabird captures. Nevertheless, this method relies heavily on the fishers will to collaborate, and as for self-reported data, this raises concerns about potential bias. Still, the authors expected participating fishers to underreport (rather than exaggerate) the number of dead seabirds they caught, so that their estimates could be considered conservative.

iv. Electronic monitoring with video

Electronic monitoring (EM) systems deployed on fishing vessels are an effective and cost-effective alternative to on-board fisheries observers (Helmond et al., 2020; Mangi et al., 2015; Plet-Hansen et al., 2019). Typical EM systems integrate signals from video cameras, position sensor (GPS) and gear sensors (pressure or drum rotation sensors) to record the entire fishing activity of a vessel. EM data is stored on local hard drives, which need to be collected regularly, or transmitted over-the-air to dedicated servers (Figure 5).

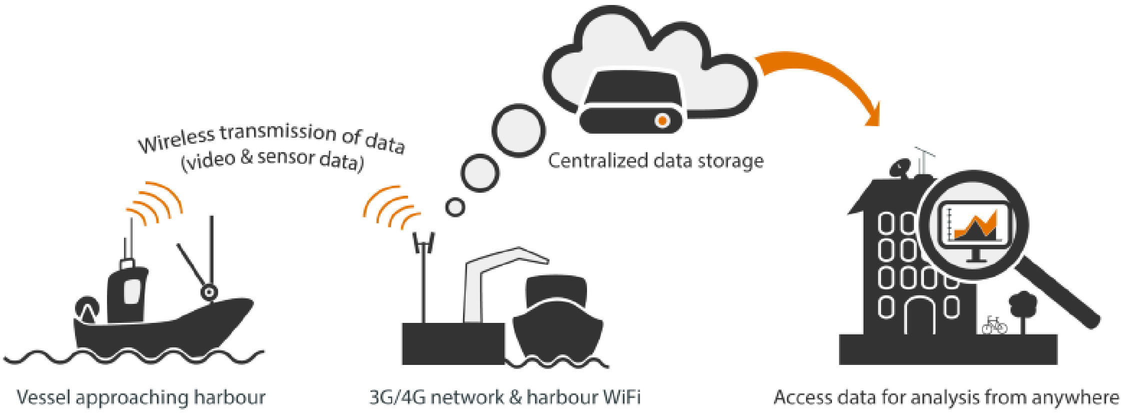

Figure 5: Schematic of electronic monitoring data collection, over-the-air transmission and storage on distant servers *(courtesy of Anchorlab K/S, Copenhagen).*

In the past 20 years, multiple trials in various fisheries worldwide have demonstrated the ability of EM to document target species catches, fishing behaviour and/or incidental captures of vulnerable species at a fine spatial and temporal scale (Helmond et al., 2020; Kindt-Larsen et al., 2012a, 2012b, 2011; McElderry et al., 2007; Needle et al., 2015; Piasente et al., 2012; Plet-Hansen et al., 2015; Ulrich et al., 2015). In some cases, EM has been evaluated as a substitute to fisheries observers (e.g. Evans and Molony, 2011), but it is generally admitted that EM and human observers data are complementary, and that one should not replace the other. Although the relatively high installation and running costs of EM can limit implementation, the budget necessary for monitoring a large portion of a fleet remains below what an equivalent observer programme would cost (Bartholomew et al., 2018; Dinsdale et al., 2013; Gilman et al., 2019; Kindt-Larsen et al., 2012a). Moreover, this cost is likely to decrease with future technological enhancements (Plet-Hansen et al., 2019). Nonetheless, EM systems have a relatively bad press among professionals in the fishing sector, who sometimes see the presence of cameras on-board as an intrusion in their privacy, a.k.a. the "Big Brother is watching you" effect (Helmond et al., 2020; Mangi et al., 2015). This is particularly true for fishers without prior experience with EM (Plet-Hansen et al., 2017). Generally, there is a lack of support of EM from fishers who believe there is a risk (real or imaginary) that video footage could be used against them by media – e.g. to document the incidental catches of protected or iconic species – or by authorities – e.g. to monitor vessels' compliance to fishing regulations like the EU landing obligation (Helmond et al., 2020; Michelin et al., 2018).

In Denmark, trials with EM on-board commercial fishing vessels started in 2008 to test whether EM could be used to estimate discards (Dalskov and Kindt-Larsen, 2009). This initial experience was a success and resulted in extending the trials to provide full documentation of fisheries (FDF) and implement a catch quota management system (CQMS) (Kindt-Larsen et al., 2011; Plet-Hansen et al., 2019; Ulrich et al., 2015). Among others, this second round of trials provided unique fine-scale fisheries data on bycatch of vulnerable species in Danish gillnet fisheries. Specifically, EM data from the Danish demersal gillnet fleet have been collected since 2010 to evaluate the bycatch of harbour porpoise *Phocoena phocoena*, and to establish areas of high risks of incidental captures in Danish waters (Kindt-Larsen et al., 2016, 2012a).

2.2. Case study: monitoring seabird bycatch using EM (Paper 1)

In the earliest stage of this Ph.D., the prospect of having access to several years of video monitoring data from Danish commercial gillnetters to assess seabird bycatch was both exhilarating and frightening. Because of i) a low observer coverage in Danish small-scale gillnet fisheries, and ii) a scarcity of information from official logbooks and sales notes, EM systems installed on-board coastal gillnetters constituted a unique source of fisheries-dependent data to assess seabird bycatch in Danish waters. Previously, the data collection process using EM had been developed with incidental captures of marine mammals in mind, and important modifications to the workflow needed to be implemented to adapt it to seabird bycatch. This first paper was thought as a proof-of-concept, aiming to demonstrate the interest of EM as an effective method to monitor seabird bycatch in a small-scale gillnet fishery, and to extract relevant information for future bycatch assessments at fleet level. Therefore, this study was the first part of a larger project aiming at estimating seabird mortality in gillnets in Danish gillnet fisheries.

The pilot study focused on EM data from three demersal gillnet vessels operating in coastal waters in the Sound (ICES Area 3b23). In this area, as in other coastal areas in the Western Baltic, bycatch in gillnets is a recognised threat for seabirds, particularly for wintering populations. Previous studies in the region focused highly on evaluating local or regional bycatch mortalities (Bellebaum et al., 2013; Degel et al., 2010; Žydelis, 2013; Žydelis et al., 2009), or on estimating the vulnerability of seabirds to bycatch in gillnets (Sonntag et al., 2012). Conversely, the primary goal of this study was to use long time-series of EM data collected from different commercial vessels to evaluate the fine-scale temporal and spatial variability of the fishing activity and of BPUE in the area (a type of data that is rarely available in small-scale fisheries). In most bycatch studies, fisheries-dependent data are either lacking precision in terms of spatial or temporal resolution, or both. Although our results would be based on a small sample of the fleet, the selected vessels were actively fishing during most of the year, with a relatively high mean yearly number of days at sea. Nevertheless, the sampled vessels participated in the EM programme voluntarily, and were thus likely not representative of the entire fleet. It is conceivable that participating fishers were more aware than average of the potential negative impact of their activity on populations of seabirds, and therefore were more attentive to limit their own contribution to bycatch. Consequently, bycatch rates from voluntary fishers were likely to be conservative.

By the time the paper was ready to be written, 9 years of data (2010-2018) had been collected and entirely analysed. The work was facilitated by the use of specialised software to review the EM data, which combined information related to speed (as time line) and position of individual vessels (as map), and of the corresponding videos from the different cameras installed on-board (Figure 6). We collected all the information related to the fishing operations (date, time and positions of the fishing fleets for each set and each haul), as well as relevant operational and environmental factors (soak duration, average soaking depth, net fleet length). Each haul was reviewed at varying playback speeds depending on the quality of the footage. Any bycatch of seabirds identified from the video footage was registered in the database. When possible, seabirds were identified at species level. For some species, it was possible to determine the sex and breeding status (Figure 7).

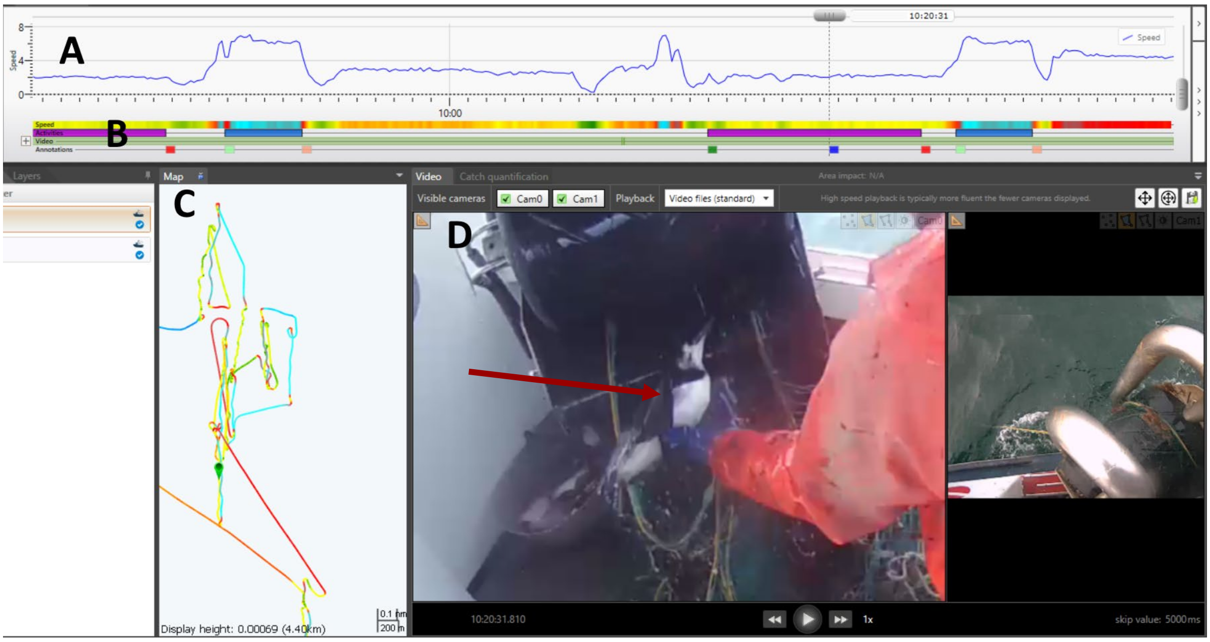

Figure 6: Snapshot of an electronic monitoring analyser sofware *(here: BlackBox Analyzer - Anchorlab K/S, Copenhagen). The on-screen information that could allow identifying the fishing vessel was removed from this picture. A. time line indicating the instant speed of the vessel; B. Annotations used for identifying fishing operations and bycatch events; C. Map with the trace of the fishing trip and the current position of the vessel; D. video footage (the arrow points to a common guillemot* Uria aalge *captured in the net).*

In general, very few individuals were not identifiable: out of 700 birds found as bycatch, only eight could not be identified (Paper 1). This abundance of fine-scale data allowed estimating species-specific seasonal bycatch rates in the study area, revealing that three species constituted 90% of all catches: the common eider *Somateria mollissima*, the great cormorant *Phalacrocorax carbo* and the common guillemot *Uria aalge* (Figure 7). Although bycatch was registered all along the year, most of it happened between September and April. In addition, there was some clear spatial clustering, indicating areas where the risk of species-specific bycatch was higher. For the species were there was a clear difference in plumage between sex (e.g. common eider) or between age classes (e.g. great cormorant), it was possible to calculate ratios showing the intra-specific differences in risk to bycatch.

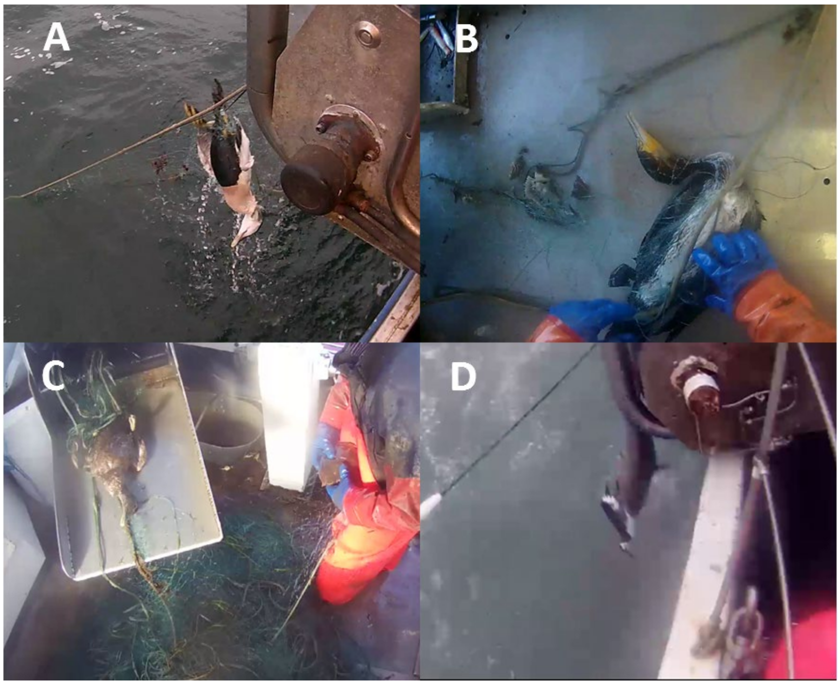

Figure 7: Video extracts from EM analysing software used to identify seabirds taken as bycatch*. From Glemarec et al. (2020).*

Another important finding of this first paper was that seabird bycatch events are not randomly distributed among hauls or fishing trips. This information is essential to consider in future EM programmes, as it implies that analysing only a fraction of the EM sampling effort could reduce the certainty of the bycatch estimates considerably. In the data that we analysed, 40% of the seabirds registered in gillnets had been captured in less than 0.2% of the hauls. The high variability in bycatch rates pertains to rare mass bycatch events where several dozens of birds can be caught in one single haul. Ignoring these hauls would produce biased estimates. It is now generally admitted that, if the objective of a EM programme is to assess rare events (like incidental catches of threatened species), a census of the video data should be analysed, as opposed to EM programmes aiming at monitoring catches/discards of commercial species, where a random selection of a percentage of the recorded videos is usually analysed (Helmond et al., 2020). Here, analysing only a fraction of the EM sampling effort would very likely underestimate total seabird bycatch in this fishery. Therefore, our results suggest that, in the optic of assessing seabird bycatch in a small-scale fishery, new EM programmes should give priority to analysing the entire fishing activity from few representative gillnet vessels, rather than spreading the sampling effort over a higher number of vessels whose fishing activity would be only partially analysed.

3. ESTIMATING SEABIRD BYCATCH IN GILLNET FISHERIES

Despite the hundreds of thousands of individuals drowning in nets each year globally, catching a seabird is for most small-scale fishers a relatively rare event: occasionally, one or maybe a few birds are captured – and then promptly thrown overboard. Even more rarely, mass bycatch events may occur, where dozens or even hundreds of birds are be caught in one haul. At fleet level, the sum of these bycatches can rapidly become significant for bird populations. Still, in small-scale fisheries where monitoring coverage is low, seabird bycatch estimates are often not available (Chapter 2), impairing the capacity of fisheries and conservation management bodies to take informed and adequate decisions.

3.1. SEABIRD BYCATCH HIGH-RISK AREAS

Incidental bycatch in gillnets results from a concomitant use of the same area by vulnerable seabirds (for foraging or resting) and fishers (for setting their gears). Mapping the overlap between the distributions of seabirds and gillnet fisheries allows identifying potential bycatch-problematic areas. The areas where the risk is highest can be highlighted using e.g. a grading or scaling system to represent the intensity of conflict.

This approach was used in Europe recently, in Germany (Sonntag et al., 2012) and in the United Kingdom (Bradbury et al., 2017). Both studies created an index representing the vulnerability to bycatch in gillnets for different seabird species, based on a combination of ecological and behavioural traits (e.g. diving behaviour, adult and/or juvenile survival rates…), as well as species conservation status. In parallel, the spatio-temporal variations in seabird at-sea distributions were calculated from historical data using aerial and ship-based surveys. Fishing effort was estimated in the UK study using only VMS (Vessel Monitoring System) records of the gillnetters operating in the study area. VMS data provides information notably on speed and position of individual vessels at regular intervals (usually once every 2 hours), which can be filtered to identify areas where fishing is likely to occur (Lee et al., 2010). However, in EU waters, only "large" vessels are required to use VMS (vessels >15 m before 2012, and vessels >12 m from 2012). In Germany, in order to account for the large number of small vessels in the study area, VMS data were combined with the observed presence/absence of fishing gears in the area (based on counts of gillnet flags from ship-based surveys). Finally, seabird bycatch risk was calculated seasonally as the product of bycatch vulnerability (per species or group of species) and fishing effort. Nevertheless, using solely VMS data to estimate gillnet fishing effort (as in the UK) will necessarily greatly underestimates total gillnet fishing effort, as small vessels are not legally bound to use VMS. For illustration, Figure 8 from Sonntag et al. (2012) shows that, in German EEZ, vast areas are simply not prospected by (larger) vessels carrying VMS on-board, in particular the areas nearest to shore. Moreover, the duration between two consecutive VMS polls (usually 2 hours) limits the capacity of identifying the fishing activity correctly in short-set fisheries (O'Farrell et al., 2017). In addition, although VMS data can in principle be linked back to logbooks and sales notes, it was not done in these studies, so that information on gear type and mesh size was "lost" in the process. As such, these studies assumed a bycatch risk that is directly proportional to the estimated intensity of the local fishing effort. However, it is likely that there is a differential risk of seabird bycatch depending on the type of gear (demersal gillnet, trammel net, driftnet, etc…) and the operational factors (net dimensions, mesh size, soak time, etc…), as is the case for other vulnerable taxa (Northridge et al., 2017).

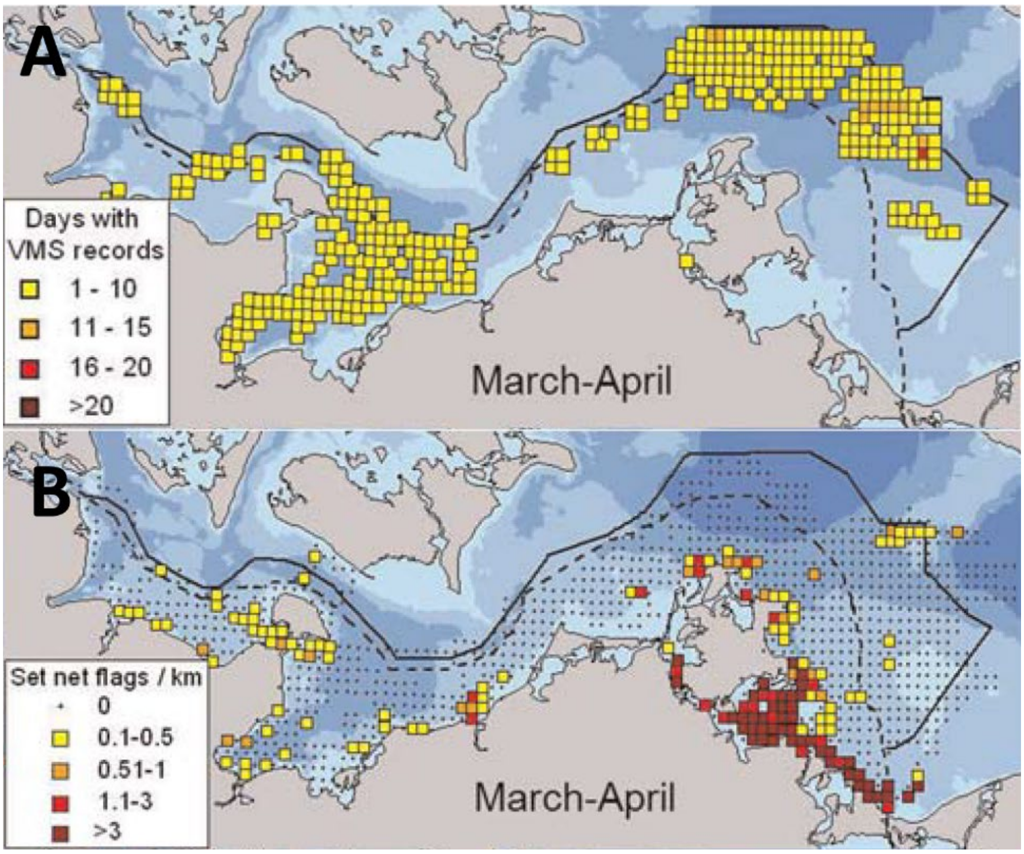

__Figure 8: Comparison of gillnet fishing effort in German coastal waters.__ A. using VMS records from German vessels (effort in days with VMS records per grid cell); B. using net flag counts from ship-based surveys (effort in number of flags observed per km of ship travel per grid cell). Comparing maps A and B shows that some areas with intense fishing effort are not covered using solely VMS records. Modified from Sonntag et al. (2012).

In recent years, the development of biologgers has considerably advanced the knowledge of marine megafauna distribution (Hays et al., 2016), particularly the complex relations that exist between bycatch-vulnerable seabirds and large-scale fisheries (Soriano-Redondo et al., 2016; Torres et al., 2013). Such maps are important decision-support tools for fisheries and wildlife managers (Hays et al., 2019; Oppel et al., 2018). For instance, Clay et al. (2019) mapped the spatial-temporal overlap between pelagic seabirds (e.g. petrels and albatrosses) and several large-scale fisheries operating in the southern hemisphere. They were able to identify clearly the areas where the risks of interaction were highest by overlapping biologging and VMS data. Similarly, in Denmark, bycatch risk was calculated in a coastal gillnet fishery for a small cetacean, the harbour porpoise, using satellite-tracking data to model the species densities and EM data to estimate the variations of fishing effort intensity and bycatch rates (Kindt-Larsen et al., 2016). A statistical model was utilised to estimate the areas where the risks of interaction were highest based on operational factors (e.g. soak time) and ecological factors (e.g. porpoise density).

3.2. FLEET-WIDE BYCATCH MORTALITY ESTIMATES

Fisheries management bodies assess the state and fate of important commercial stocks periodically in order to estimate the maximum level of harvesting that the targeted populations can sustain on the long-term (i.e. Maximal Sustainable Yield, MSY). A stock assessment combines information on one or several stocks, including "life history, fishery monitoring, and resource surveys for estimating stock size and harvest rate relative to sustainable reference points" (Cadrin and Dickey-Collas, 2015). Based on MSY, each metier is allocated a total amount of fish allowed to be extracted, stratified at individual vessel level; what is generally known as quota in the European Union. Stock assessments often rely upon complex statistical models, which aim at understanding the demographics of harvested populations given different fishing mortality scenario. Likewise, provided that enough quality data are available, similar methods could be applied to estimate the impact of seabird bycatch at population level in different geographic areas.

i. Bycatch per unit effort estimates

Estimating mean BPUE from a sample of the fishing fleet

Bycatch per unit effort represents a number of incidental captures per unit of effort. BPUE is a catch rate, an effective metric of fishing effort, which requires estimating both the nominal effort (e.g. number of hauls, number of days at sea, etc…) and the amount of bycatch in the corresponding strata (McCluskey and Lewison, 2008).

In the EU, (nominal) fishing effort must be reported as days at sea for all metiers, so BPUE is generally expressed as number of bycatch per days at sea (Table 4 in EU 2019a). Ideally, logbook data from recreational and commercial fishers would provide accurate and comprehensive information on interactions with seabirds for each individual vessel, as well as fine-scale spatial and temporal information on fishing effort. Using these records, one could simply sum up every drowned animals registered per metier/fleet to obtain fleet-wide estimates of seabird mortality in fishing gears. Unfortunately, such exhaustive datasets do not exist for EU small-scale fisheries. Therefore, bycatch rate estimates need to be obtained from a sample of the metier/fleet under scrutiny first (e.g. fisheries observers, electronic monitoring, etc…). Scientific data from DCF sea-sampling programmes, which are the most important providers of fishery-dependent data for most EU Member States, are often incomplete with regards to seabird bycatch. On the one hand, EU fisheries observers are not fully equipped and/or trained to record bycatch of PETs effectively (Table 1 in ICES 2019), and on the other hand, observer coverage in small-scale fisheries is very low in the EU. As a result, available seabird bycatch data in the EU is often not sufficient to extrapolate observed bycatch rates to fleet level.

Conversely, other countries have developed methods to estimate bycatch mortalities based on samples of their national fishing fleets. For instance, in a recent study, Christensen-Dalsgaard et al. (2019) assessed the levels of seabird bycatch in the different gillnet fisheries targeting lumpsucker in the Northern Atlantic. These fish are captured preferably with bottom-set gillnets set in coastal waters, often in areas used by wintering seabirds. Net fleets are generally large-meshed (>200 mm, to increase the chances of entangling the target species) and are soaked for several days. These characteristics make these fisheries very prone to capturing seabirds. Unlike most EU countries, Norway and Iceland have developed, besides their regular national monitoring scheme, dedicated monitoring programmes to collect seabird bycatch data in lumpsucker gillnet fisheries. In these countries, enough data were available to estimate bycatch rates (BPUE) and to extrapolate mean estimates to fleet level. Data from Denmark were also provided in this study, based on estimates from the EM gillnet fleet. However, because of the low spatial coverage of the Danish data, it was not possible to extrapolate confidently to fleet level. Another study in Norway analysed a decade of seabird bycatch data from the Norwegian coastal gillnet Reference Fleet (Bærum et al., 2019). This work identified the species that were the most affected by bycatch in gillnets, and the authors calculated mean yearly bycatch rates from their large sample. Applying these estimates to the rest of the fleet was possible using days at sea as a measure of the effort, as this is what Norwegian fishers are meant to report in official logbooks.

Likewise, in the first project presented in this Ph.D. (Section 2.2 and Paper 1), seabird bycatch rates were estimated in a small-scale Danish gillnet fishery from a sample of three vessels monitored with EM for several years. Extremely rare mass bycatch events (defined as fishing trips where more than six birds per trip were caught) represented in our dataset 14 fishing trips out of 2118, but summed up to more than 40% of the total amount of seabird captures. Therefore, using a fraction of the recorded fishing activity for estimating BPUE would likely underestimate bycatch rates immensely. These results suggested that for obtaining accurate fleet-wide seabird bycatch rate estimates, it was necessary to collect long-time series of the fishing activity from a sample of representative gillnet vessels. For an equivalent sampling effort, EM would be more cost-effective and possibly less biased than using fisheries observers alone (Helmond et al., 2020).

Predicting BPUE from statistical models

Statistical modelling has become increasingly popular in the past years to study ecological systems (e.g. Bolker et al., 2009; Zuur et al., 2009, 2007). The availability of powerful and affordable personal computers, as well as the popularity of accessible free and open-source resource like the statistical language R (R Core Team, 2020) have considerably facilitated scientific research in ecology.

Statistical models are useful tools to understand the underlying reasons of bycatch and unveil some of the complex relationships between fisheries and seabirds. Modelling approaches have mostly been used to study bycatch in large-scale and/or longline fisheries (e.g. Petersen et al. 2009, Dietrich et al. 2009, Yeh et al. 2013, Melvin et al. 2014). Comparatively, gillnet fisheries have received less attention, despite their large contribution to global seabird bycatch mortality. In gillnet fisheries, seabird bycatch events are usually so rare that long time series of fine-scale records of the fishing activity are often necessary to capture the variations of BPUE spatially and temporally. Still, although small samples may not perfectly reflect the overall fishing fleet, they can bring important information about the relationships linking BPUE to operational and environmental factors.

Bærum et al. (2019) is one of few examples of examining the variations of seabird BPUE in a gillnet fishery using a GLMM (Generalized Linear Mixed Model). This work is based on 10 years of monitoring data from the Norwegian Reference Fleet. The authors found that a mixture of temporal (year and month), spatial (Norwegian statistical fishing area), environmental (distance to coast, fishing depth) and operational (number of nets per fishing trip) fixed predictors, was the best at explaining the observed levels of bycatch (vessel name was used as a random intercept). Such a model using fine-scale data is extremely useful to inform which parameters favour seabird bycatch, but, unless the same metrics are collected at fleet level, this model cannot be used for prediction.

ii. Fleet-wide fishing effort estimates

Fisheries control in EU waters utilises data from logbooks (providing information on the fishing activity by region, “metier Level 6”, and on nominal fishing effort, “days at sea”, in particular), sales notes (providing information on landings) and VMS data (providing fine-scale spatial information on fishing effort). As seen in section 3.1, information from VMS data permits the identification of fishing grounds with precision using simple speed filtering methods, or more advanced statistical modelling techniques (Hintzen et al., 2012; Lee et al., 2010; Mills et al., 2007; Muench et al., 2018; Russo et al., 2014). However, VMS is only mandatory for vessels >12 m, limiting its relevance for small-scale fisheries. Gillnet fisheries in the European Union are characterised by a high proportion of small-scale vessels, often <10 m (EC, 2020). Vessels below this size are not required to fill in logbooks to report fishing activity (the limit is <8m in the Baltic Sea). This situation results in a poor knowledge of the spatial distribution of fishing effort for small-scale fisheries in general, and for small-scale gillnet fisheries in particular.

The growing popularity of the AIS constitutes a promising alternative to VMS for estimating fishing activity and effort, especially in coastal waters where AIS coverage is higher than offshore (Russo et al., 2016). Originally, AIS is a VHF transceiver-based vessel identification system that was developed as an anti-collision device (Fournier et al., 2018). AIS signals spread omni-directionally to be received by neighbouring vessels, shore-based receiving stations and, for some advanced models, by satellites (Natale et al., 2015). Despite not being compulsory for small vessels in the EU (AIS is mandatory for vessels >15 m since 2014), the system has been adopted by a growing number of commercial and recreational vessels of smaller size classes, especially in waters with busy shipping traffic, primarily for safety reasons. For instance, all shipping traffic entering or leaving the Baltic Sea must travel through one of two narrow channels, namely the Great Belt in the West and the Øresund in the East. In these parts of Danish waters, where shipping traffic is intense, AIS records are covering on average 27% of the nominal effort (in days at sea) registered in official logbooks (data spanning from 2014 to 2017, obtained from the Danish Maritime Authority and from the Danish Fisheries Agency, processed by DTU Aqua).

One of the advantages of AIS over VMS pertains to its much higher ping frequency. Where VMS usually transmits a position via satellite every 2 hours, AIS updates the ship's position every few minutes through VHF (commonly every 5 minutes in coastal waters). This higher temporal resolution can be used to distinguish between fishing and non-fishing activities in metiers with short-lasting fishing operations. Typically, gillnetters in Inner Danish waters use relatively short net fleets, and take between 20 minutes to less than an hour to haul their nets. As such, VMS frequency is easily susceptible to miss a potential haul, whereas AIS would be able to record a haul throughout. Nonetheless, unlike VMS, AIS was not designed with fisheries control in mind, and its usage for monitoring fisheries comes with some important caveats. First, AIS can easily be turned off, so that tracks from AIS data may not perfectly reflect a ship's activity. Additionally, AIS positions rely on the GPS installed on-board the vessel, and may transmit wrong data if the sensors are not correctly calibrated (in particular in relation to speed). Moreover, a comparative study between AIS and VMS in the Mediterranean Sea showed that AIS might be less adapted to offshore fisheries due the short range of VHF transmissions (Russo et al., 2016).

In spite of these limitations, AIS remains an interesting additional source of data in a fisheries management context. Especially for metiers limited to near-shore areas (e.g. small-scale coastal gillnet fisheries in Europe), AIS provides valuable information that can be used to improve fishing pattern detection at a fine-scale, to map the spatial and temporal variations of the fishing effort, or to identify fishing areas more accurately (Fournier et al., 2018). Similar methods can be used to analyse VMS and AIS data, from the simple speed rule to more sophisticated statistical and modelling approaches. However, the performance of the simpler speed rule methods is typically lower for differentiating fishing and non-fishing for passive gears than it is for active gears (Muench et al., 2018; Natale et al., 2015). Therefore, for AIS data stemming from gillnet fisheries, more refined algorithms need to be developed, and their accuracy should be assessed with care (de Souza et al., 2016).

3.3. Effects of bycatch on seabird populations

Evaluating the impact of anthropogenic activities on seabird populations (fisheries bycatch being only one of many threats) can be difficult, as this requires some knowledge on the mortality rates associated to these pressures and on the demographics of the considered populations. Generally, the sustainability of a population is ensured if long-term mortality (natural and human-related) does not exceed natality. Consequently, there is theoretically a tipping point, or threshold, above which mortality is too high for a population to ensure long-term viability. Ultimately, fisheries and wildlife managers are interested in knowing these thresholds, to decide whether fisheries constitute a threat to seabird populations. To that end, environmental indicators, defining these thresholds based on scientific evidence for each population in a geographical area, can provide a baseline to assess the sustainability of fishing activities.

The MSFD "qualitative descriptors" are such environmental indicators, which aim at ensuring that Good Environmental Status is achieved in EU waters; the first descriptor (D1) requires that "[b]iological diversity is maintained" (EC, 2008a). Yet the MSFD itself does not provide any specific target. The revised version of the MSFD (EU, 2017b) specifies that threshold values must be set at national or regional level for the different descriptors, including incidental bycatch of "birds, mammals, reptiles and non-commercially-exploited species of fish and cephalopods" (criterion D1C1). Regionally, few specific indicators exist for seabird bycatch mortality. Among them are HELCOM core indicators, used to assess the state of the Baltic Sea (HELCOM, 2018). One of these indicators, the *number of drowned mammals and waterbirds in fishing gear*, defines initial threshold values for some, but not all, seabird species in the region, owing to a relative scarcity of available data. OSPAR, which currently does not have an indicator dedicated to seabird bycatch, defines two other indicators for marine bird abundance and breeding success, which may reflect the response of seabirds to the various pressures they face (including bycatch).

Thresholds linked to environmental indicators are important to assess GES achievement. However, in an ecosystem-approach to fisheries as defined under the CFP, the goal should be to "minimise and, where possible, eliminate the incidental catches of seabirds" (EC, 2012). Therefore, it seems important to stress at this point that the (existing or upcoming) threshold values as defined under the MSFD, HELCOM or OSPAR biodiversity indicators are not meant as management goals. Instead, they *indicate* whether a Member State is achieving GES. In turn, thresholds should not be interpreted as acceptable bycatch rate, and differ in that respect from the concept of Maximal Sustainable Yield (MSY) used in commercial stock assessments. Unlike fish, seabirds have a slow reproductive rate (K-strategists), so that even low additional mortality rate can have dreadful consequences, in particular for small populations (Palialexis et al., 2019). Derogations to the Birds Directive (Article 9) allow the killing of a "small numbers" of birds. In the context of hunting, a small number was defined as "any taking of around 1% of the annual mortality for species which may be hunted" (EC, 2008b). Following this recommendation, experts' groups as JWGBIRD adopted a similar recommended threshold value of "1% of all mortality of the species" (ICES, 2018), while NGOs like BirdLife are more restrictive asking to limit bycatch mortality at "1% of the natural annual adult mortality of the species" (BirdLife International, 2019).

A number of statistical and modelling techniques have been developed over the years in conservation research to determine bycatch mortality thresholds for seabirds. Below, some of the most common methods to calculate the effect of a threat on a population are listed, and their interest to evaluate specifically the effects of bycatch on seabird populations is discussed. This section utilises elements of the 2018 JWGBIRD report (ICES, 2018), which I participated in writing.

i. Potential Biological Removal

The Potential Biological Removal (PBR) is a simple algorithm that calculates the maximal non-natural (human-induced) mortality that a population can sustain to maintain above (or to reach) the maximum net productivity level (Wade, 1998). Specifically, the method was designed initially to estimate the maximal level of human-caused mortality that a population of marine mammal could sustain in the long-term without falling below half its carrying capacity (a management requirement in the United States at the time). Below PBR, the population would maintain itself at (or increases toward) carrying capacity, while above PBR, the same population would decrease to extinction (in this sense, PBR is similar to MSY for fish stocks). Although simple, PBR is usually considered a conservative method to assess whether a population is threatened by anthropogenic pressures (Hall and Donovan, 2001).

The original PBR algorithm uses few parameters (i.e., the maximum intrinsic growth rate at small population size, R_{max}, the minimum population size estimate, N_{min}, and a user-defined recovery factor, f, accounting for uncertainty). A major advantage of PBR relates to its ability to estimate maximal mortality thresholds for data-poor populations. However, adapting PBR can be problematic for seabird populations, as R_{max} is often difficult to estimate. To determine whether an additional source of mortality can be a threat to seabird populations, Niel and Lebreton (2005) proposed to compare the level of additional mortality to the annual growth rate λ_{max} (such as $\lambda_{max} = 1 - R_{max}$). λ_{max} was approximated as a function of adult survival probability, s, and age at first reproduction, α. In turn, Dillingham and Fletcher (2008) combined PBR (Wade, 1998) to the approach used by Niel and Lebreton (2005) to model the maximal level of mortality a seabird population can theoretically sustain using only λ_{max}, s, α, N_{min} and f. Since then, PBR and its adaptations, have become among the most common methods to assess the maximal human-induced mortality levels that seabird populations can sustain (e.g. Dillingham and Fletcher, 2011; Richard et al., 2020; Richard and Abraham, 2013; Žydelis et al., 2009)

PBR is a useful tool to determine if a seabird population subject to anthropogenic pressures is over-harvested, or if a specific threat, e.g. bycatch, is the main driver of the current level of mortality in the studied population. The ICES Advisory Committee recommends to use PBR to assess whether a risk of unsustainable human-induced mortality exists for seabird populations, and, if the answer is positive, whether bycatch constitutes a "substantial portion of that mortality" for the studied populations (ICES, 2013). If there is no certainty about the impact of bycatch mortality on the population, more data should be collected. For instance, in the absence of detailed quantitative data on the demographics of the affected species, Žydelis et al. (2009) calculated the PBR of three species for which they had data, and concluded that bycatch mortality in gillnets was a threat for at least two of them (i.e. mortality due to bycatch was above PBR estimates).

However, PBR relies on restrictive implicit assumptions that may easily be overlooked for seabird populations, thus leading to incorrect conclusions (Green et al., 2016; O'Brien et al., 2017). Particularly, PBR assumes density dependence, which means that as annual mortality (from all sources) increases, annual growth rate should also increase in return to maintain the population at (or grow toward) carrying capacity. Instead, some populations may show a positive correlation between density and growth rate (known as the Allee effect), or may be acting as a sink, i.e. the growth of an individual population/colony is dependent on immigration from other populations/colonies (Frederiksen et al., 2005; O'Brien et al., 2017). In a paper studying the response of theoretical seabird populations (differing notably by their population density regulation) to PBR-informed removal levels, O'Brien et al. (2017) demonstrated unambiguously that PBR should not be used to determine the maximal level of mortality a seabird population can sustain, as PBR-informed thresholds may still lead to population declines. Likewise, Marchowski et al. (2020) estimated the vulnerability to fisheries bycatch of the greater scaup *Aythya marila* European flyway population using three alternative modelling methods, including PBR and a more sophisticated age-structured matrix population model (see next section). They concluded that a PBR-informed bycatch cap was likely to have a significant negative impact on the population, even for low recovery factor values.

The simplicity of PBR makes it a practical and informative metric to judge whether a current level of mortality constitutes a likely threat to a population. However, PBR is not suitable to define an acceptable bycatch limit or a threshold below which an additional source of mortality will not negatively affect a population. Therefore, PBR should be avoided in environmental impact assessments for seabirds (Green et al., 2016; ICES, 2018; Marchowski et al., 2020; Miller et al., 2019).

ii. Population Viability Analysis (PVA)

Provided that enough demographic parameters are available, stochastic population models can be used to predict the trajectory of a population under different harvesting scenario, without having to impose theoretical growth rates or carrying capacities. Population viability analyses (PVA) provide a recommended alternative to PBR to examine the potential impact of anthropogenic threats on populations (ICES, 2018). Instead of one single method, PVA consist of a variety of modelling techniques, which all incorporate quantitative predictors in order to forecast the likely outcomes of a population (Boyce, 1992). In the case of seabird populations, an age-structured population model known as a Leslie matrix is often preferred, where different probabilities can be set for each age class, e.g. mortality, fecundity or probability to be recruited as a breeder. This is important for long-lived animals like seabirds, as experienced breeding pairs might have much higher success than younger individuals at raising their chicks (Lewis et al., 2009), or because younger mature seabirds may be less likely to find a breeding pair than older ones (Genovart et al., 2016).

The flexibility of PVA allow testing different management situations, from fisheries closures to business-as-usual, or even increase in bycatch mortality. Although the need to parameterise age-structured models can be challenging, unknown demographic factors can be chosen based on expert knowledge, or using known parameters from related populations/species. This is particularly important for declining species currently subject to bycatch, as the population viability assessments using PBR may be highly deceptive. As an example, Marchowski et al. (2020) forecasted the trends of the greater scaup Northwest European population over the next 30 years using a PVA. Assuming two initial baselines (stable or declining), the authors applied different annual “harvest” levels to the population (corresponding to the bird mortality in gillnets). Using the PBR-informed bycatch limit as the yearly bycatch mortality led systematically to population decreases after 30 years of simulations. Moreover, the model predicted a decline of 36% of the greater scaup within the next 30 years if the current level of bycatch was maintained as it is today. In line with these results, ICES (2018) and BirdLife International (2019) both recommend to use PVA for assessing bycatch mortality.

iii. Other methods to evaluate bycatch mortality limits

In its Revised Management Procedure (RMP) for Baleen Whales, the International Whaling Commission Scientific Committee developed a simulation-based approach known as the Catch Limit Algorithm (CLA). CLA is used to calculate the maximal allowable commercial catches on a regional basis, for different baleen whale populations, while accounting for uncertainty in the population and harvesting estimates (International Whaling Commission, 2020a; Punt and Donovan, 2007). The population dynamics model component of the CLA relies only on time series of abundance and catches. In practice, CLA can also be used as a simulation tool to test the effect of individual threats on whale populations, e.g. bycatch (International Whaling Commission, 2020b). Nevertheless, as the RMP is primarily destined to manage commercial whaling, another modelling technique known as the Removal Limit Algorithm (RLA), inspired by the CLA, is currently under development (Hammond, P. S. et al., 2019). While CLA depends on on known catches, RLA relies on unknown sources of mortality in order to minimise the risk of population decrease. This last approach holds a lot of interest for evaluating the impact of seabird bycatch mortality at population level, given the high uncertainty in fishing effort and BPUE in many of the fisheries with a bycatch problem.

As for the models cited above, RLA and CLA are designed to determine a mortality limit (threshold) based on conservation objectives, e.g. that the population is maintained at, or is above, 50% of its estimated carrying capacity after a number of years/generations. These limits are accounting all sources of mortality, regardless of the cause (e.g. bycatch, hunting/poaching, oiling…). Therefore, much care should be taken using these thresholds as management targets (O'Brien et al., 2017).

3.4. CASE STUDY: ESTIMATE FLEET-WIDE BYCATCH MORTALITY (PAPER 2)

For the second study in this thesis, the initial goal was to estimate fleet-wide bycatch mortality from EM data and identify which parameters were the most important to explain seabird bycatch in Danish gillnet fisheries. Similarly, in a recent study, Bærum et al. (2019) analysed a decade of data from 43 gillnet vessels to understand the spatial and temporal variations of seabird bycatch in commercial gillnet fisheries. They created a statistical model showing that the number of seabird casualties in gillnets depended on temporal variables, fishing area, distance to coast, number of nets and fishing depth. Interestingly, including mesh size in the candidate models did not significantly improve the fit. The authors scaled up seabird bycatch estimates to fleet level by multiplying the mean observed bycatch rates per trip with the total number of fishing trips in the different investigated areas.

In Denmark, the EM dataset contained much fewer fishing vessels than the above study, but a comparable modelling approach could be applied to explore seabird bycatch variability in the areas where EM coverage was deemed sufficiently high. The data were subset to EM vessels operating in areas where numerous incidental catches of seabirds had been registered. At the time of starting the analysis, most bycatch had been recorded from vessels operating in the Belt Seas (ICES statistical areas IIIc22 and IIIb23), a marine region located between the Baltic Proper and the North Sea, enchased between mainland Denmark, Germany and Sweden (Figure 9). Assuming that the gillnetters sampled with EM were representative of the entire fishing fleet, fleet-wide bycatch mortality were estimated by multiplying the estimated mean BPUE observed on EM vessels with the number of days at sea registered in official logbooks, and the associated confidence intervals were obtained with bootstrap (Table 1). However, the presence of few mass bycatch events in the EM dataset led to wide confidence intervals (Paper 1). Collecting more data could have reduced the uncertainty around the mean estimates, but the cost of implementing new EM systems limited the number of vessels that could be monitored. Alternatively, data analyses from VMS and/or AIS data could bring valuable additional fine-scale information on fishing effort location (Muench et al., 2018; Natale et al., 2015). Nevertheless, in the EU, the level of adoption of on-board satellite-tracking systems is low for small fishing vessels, as these systems are mandatory only from vessel lengths of 12 m (VMS) or 15 m (AIS). Still, in the Belt Seas, a relatively high proportion of commercial gillnet vessels are equipped with AIS for safety reasons, to limit the risk of collisions (Figure 9), and all the vessels monitored with EM in this area used

AIS. Therefore, it was possible to predict individual vessels' fishing activity from AIS tracks, and to validate the fishing effort predictions using known fishing effort locations from EM data.

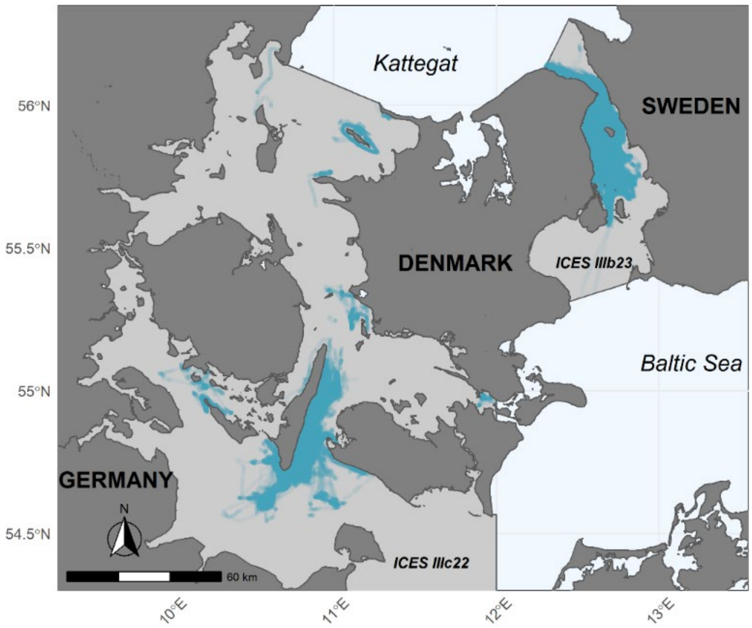

Figure 9: Map of the raw AIS polls (blue) from 48 Danish commercial gillnetters operating in the Belt Seas (ICES areas IIIc22 and IIIb23, in grey) between 2014 and 2018. *Surrounding countries and basins are indicated.*

Table 1: Estimated quarterly fleet-wide seabird bycatch mortality in gillnets and 95% bootstrapped confidence intervals (10,000 repetitions), in ICES area IIIc22 and IIIb23. *Mean seabird bycatch mortality estimates are the product of the mean bycatch rates from EM vessels with the mean number of days at sea registered in official logbooks. BPUE (birds per trip), fishing effort (days at sea) and vessel count are the mean quarterly estimates for the period 2014 to 2018.*

		Logbook data (fleet)		**EM data (6 vessels)**	**Bootstrapped bycatch estimates (fleet)**			
ICES Area	Quarter	Vessel count	Registered days at sea	Observed days at sea	Mean BPUE	Mean bycatch	Low	High
IIIc22	1	134.4	1972.8	56.2	0.26	505	203	1689
	2	153	2430.8	106.2	0.01	27	9	50
	3	134.2	1660	85.75	0.03	53	24	102
	4	141.6	1670.8	93	0.09	157	94	261
IIIb23	1	49.4	643.6	71.4	0.36	229	155	368
	2	45.2	680.6	64.2	0.11	68	15	287
	3	53.4	891.6	56.8	0.25	219	157	301
	4	63.2	1129.6	35.4	1.69	1819	1079	3332

The EM dataset contained the records of the fishing activity of six commercial gillnet vessels, three operating in the Øresund (IIIb23), and three in the Belts (IIIc22), between 2014 and 2018. A generalised linear mixed model (GLMM) was developed based on these data to identify the operational and ecological factors that explain the observed level of seabird bycatch in gillnets in the study area. The results of the model investigations showed that fishing effort (as net fleet length x soak time), depth and distance to shore dominated the prediction probabilities in the seabird bycatch model. Mesh size was weakly positively correlated to the response, but was not found to be significant. Time of the year was a major driver of bycatch, with important reductions of bycatch predictions between April and October.

Moreover, we developed a binomial model to predict whether the position of an AIS poll corresponded to a haul or not, based notably on the vessel speed, depth, time of the day and month, as well as the size and engine power of the vessel. The model predictions were validated using data from EM vessels, which all used AIS. The overall accuracy of the model was acceptable (0.81), and more than 96% of the haul positions were correctly identified, while 74% of the non-haul positions were correct. The predicted hauls were checked for inconsistencies in terms of net length and duration of the hauls. The problematic AIS polls were classified as non-hauls (Figure 10).

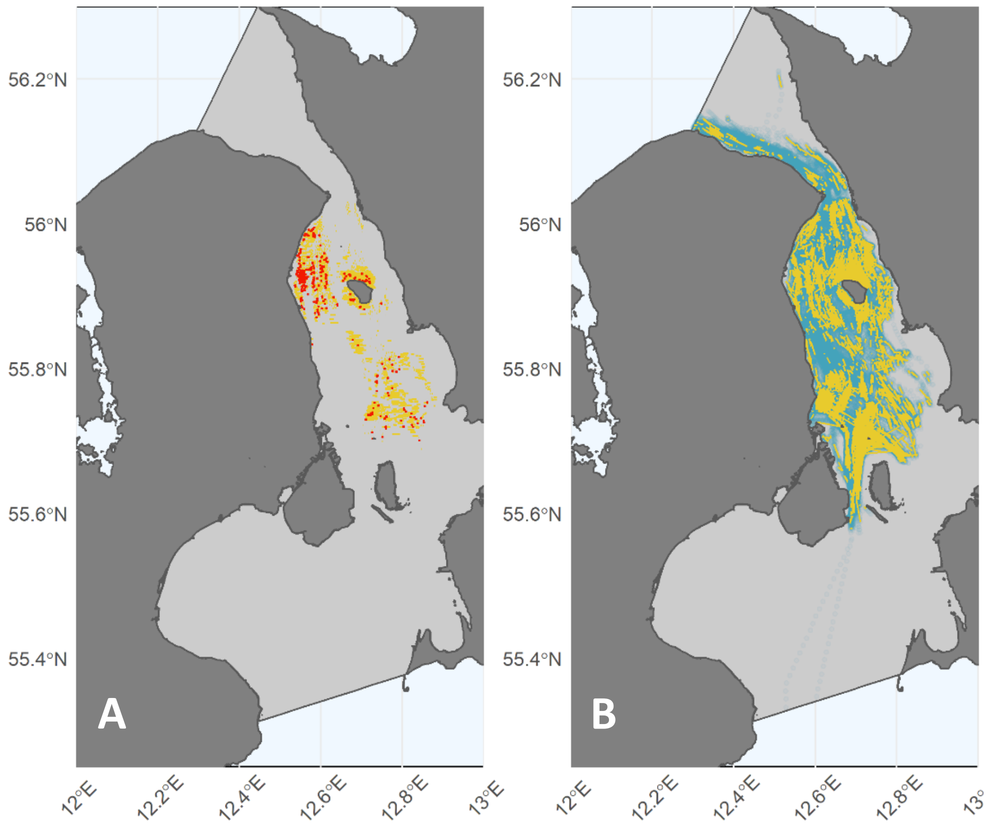

Figure 10: Comparison between A. observed hauls (EM data) and B. predicted hauls (AIS data) in ICES area IIIb23.
A. Observed hauls (yellow) and seabird bycatch positions (red) of three EM gillnet vessels in ICES IIIb23, and B. predicted haul positions (yellow) from the analysis of the raw AIS polls (blue) of 20 vessels using AIS in the same fleet. Aggregated data from 2014 to 2018.

The resulting dataset contained information on fishing activity at haul level of a relatively large fraction of the fleet in some areas (Figure 10). In turn, seabird captures were predicted for each haul using the bycatch model developed earlier, and BPUE, measured as bird bycatch per day at sea, was evaluated. Finally, stratified mean BPUE (by quarter and ICES area) were multiplied by the corresponding number of days at sea registered in logbooks in each strata to obtain fleet-wide bycatch mortality estimates (Table 2). The resulting mean mortality estimates obtained in this way were smaller than if using observed BPUE from EM vessels, because of the inability of the bycatch model to predict very rare and localised large bycatch events. For instance, the highest number of seabird captures predicted from the model was six birds caught during one trip (or day at sea), whereas in reality we observed on occasions up to 57 seabirds captured in one single trip on EM vessels. Besides, as the contribution of mass bycatch events is ignored in Table 2, the width of the bootstrap confidence intervals is substantially reduced compared to extrapolations from observed bycatch on EM vessels. Nevertheless, assuming that the AIS vessels are sufficiently representative of the entire fleet, the number of seabird casualties calculated from the predicted fishing activity of the AIS vessels can be considered conservative estimates of seabird bycatch mortality in commercial gillnets in the study area.

Table 2: Estimated quarterly fleet-wide seabird bycatch mortality in gillnets and 95% bootstrapped confidence intervals (10,000 repetitions) in ICES areas IIIc22 and IIIb23 using mean BPUE from AIS vessels. *Mean seabird bycatch mortality estimates are the product of the mean bycatch rates from AIS vessels with the mean number of days at sea registered in official logbooks. BPUE (birds per trip), fishing effort (days at sea) and vessel count are the mean quarterly estimates for the period 2014 to 2018.*

		AIS data (48 vessels)	**Bootstrapped bycatch estimates (fleet)**			
ICES Area	Quarter	Registered days at sea	Predicted BPUE	Mean bycatch	Low	High
IIIc22	1	196,8	0.23	434	371	533
	2	120,4	0.01	60	41	107
	3	181	0.04	89	77	105
	4	159,8	0.24	276	211	376
IIIb23	1	110,2	0.32	218	193	248
	2	87,4	0.02	18	14	23
	3	132,8	0.08	94	78	112
	4	174	0.66	1078	976	1186

The approach described above ignores any potential species-specific vulnerability to incidental captures in gillnets. Evidently, assessing the effect of bycatch at population level requires calculating individual species mortality in gillnets. Table 3 presents the quarterly bycatch estimates for the common eider, the most frequent bycaught species on EM vessels in the study area. The annual bycatch at fleet level was estimated between 1107 and 1455 individuals (95% bootstrapped confidence intervals, 10,000 repetitions). Once again, these estimates are conservative, as potential mass bycatch events are not predicted well in the model.

Table 3: Estimated quarterly fleet-wide common eider bycatch mortality in gillnets and 95% bootstrapped confidence intervals (10,000 repetitions) in ICES areas IIIc22 and IIIb23 using mean BPUE from AIS vessels. *Mean common eider bycatch mortality estimates are the product of the mean bycatch rates from AIS vessels with the mean number of days at sea registered in official logbooks. BPUE (common eider per trip) is the mean quarterly estimates for the period 2014 to 2018.*

		Common eider bootstrapped bycatch estimates (fleet)			
ICES Area	Quarter	Mean common eider bycatch per trip	Mean bycatch	Low	high
IIIc22	1	0.20	389	349	434
	2	0.01	31	23	42
	3	0.02	30	28	33
	4	0.10	174	139	219
IIIb23	1	0.22	138	125	156
	2	0.01	7	6	10
	3	0.04	33	29	37
	4	0.41	464	409	525

An extension of this work will be to determine if the levels of bycatch calculated for the common eider in the study area are unsustainable. Despite differences between flyway populations, the common eider is generally decreasing in the EU, and it qualifies as endangered following the IUCN classification. Current population estimates span between 449,000 and 640,000 mature individuals (BirdLife International, 2015). Assuming that all common eiders were mature upon capture, this would mean that 0.17 to 0.32% of the population in the EU is captured annually in gillnets in the Belt Seas. Although below the 1% recommendation from ICES (2018), these levels of bycatch must be added to the other sources of anthropogenic mortality, e.g. hunting, wind mills, or ship strikes, and might constitute a threat for the different sub-populations. Besides, video analyses from EM vessels show that a large proportion of the common eiders taken as bycatch are mature males (see Table 3 in Paper 1). This sex-biased vulnerability to bycatch might affect the demographics of the populations (Gianuca et al., 2017). More work is thus required to estimate the effects of bycatch on the common eider for the flyway populations present in Danish waters.

The goal of this analysis was to estimate fleet-wide seabird bycatch mortality in a Danish gillnet fishery. Data from electronic monitoring was utilised to establish mean BPUE in the area, and extrapolated to fleet level. These bycatch rates extrapolated to the fleet resulted in large bycatch mortality estimates with wide uncertainties, resulting from few large bycatch events. In addition, we developed a model to predict seabird bycatch per haul from EM data. We also used the AIS records of a portion of the gillnet fleet to infer the fishing activity of these vessels and applied the bycatch model to this dataset. Extrapolating the resulting mean BPUE to the entire fleet resulted in narrower uncertainty estimates, but was unable to account for the rare mass bycatch events. We finally calculated the estimated bycatch of common eider in gillnets in the study area using the same approach. These estimates will contribute to a future assessment on the impact of bycatch in gillnet on the populations of common eider in the region, using e.g. a population viability analysis.

4. Reducing seabird bycatch in gillnet fisheries

4.1. The seabird perception of the environment

Not all seabirds dive to capture their prey, some simply forage at or directly below the water surface (e.g. seagulls). Diving birds on the other hand are amphibious animals, who must submerge themselves to search for food. Such animals have developed specific sensory adaptations, so that they can perceive their environment both in-air and underwater, and identify their prey. Diving seabirds belong to a number of families and differ widely in their foraging behaviour, depending on the type of prey that they favour (e.g. sessile bivalves for many seaducks, evasive prey for most of the other groups). However, even for the birds who feed on evasive prey items (e.g. fish), dive characteristics vary whether the targets are detected from the surface before submersion (e.g. gannets, albatross, petrels or shearwaters), or while underwater (e.g. cormorants, auks, loons or penguins) (Martin, 2017; Martin and Crawford, 2015). Nevertheless, vision is likely the primary sense for most predatory seabirds.

i. Vision

Although vision may be playing a major role in the detection of prey for many of the species mentioned above, most seabirds probably have relatively poor visual acuity while diving. First, water and air have very different refraction indices, so that an eye is primarily adapted to one medium or the other. Therefore, switching from one to the other will result in a blurring the image perceived on the retina; a bird whose eyes are adapted to vision in-air will be short-sighted underwater (Figure 11). Moreover, light levels decrease rapidly with depth, while differential absorption in water tends to filter out the longer wavelengths (towards red in the light spectrum down to blue).

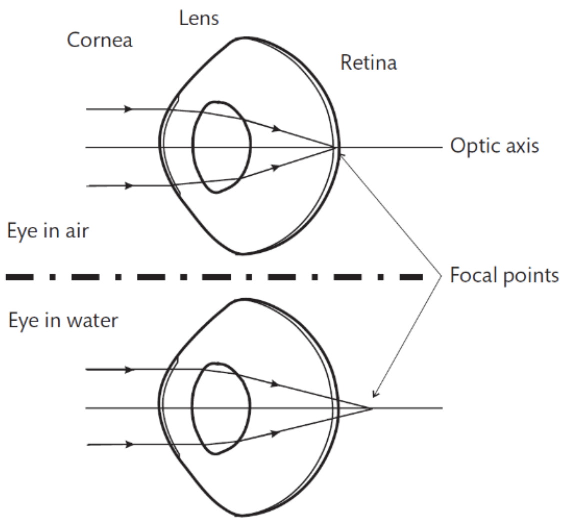

Figure 11: Image focus on the retina in-air, and behind the retina underwater. *An eye perceives a sharp image when the light entering through the cornea focuses on the retina, i.e. the focal point is situated directly on the retina. A bird whose eye is adapted to vision in-air, sees a blurred image when underwater, because of the difference in refraction indices between air and water that moves the focal point behind the retina. From Martin (2017).*

Structural adaptations of the eye have been observed in several species in response to these environmental conditions. For instance, to compensate for the loss of "sharpness" on the retina, penguins have a relatively flat cornea, which allows them to see reasonably well underwater, whereas they are short-sighted while on land (Martin et al., 1984; Sivak and Millodot, 1977). Great cormorants *P. carbo* on the other hand seem to be able to compensate the loss of power of the cornea when submerged by correcting the shape of the eye lens (Katzir and Howland, 2003). However, visual tests demonstrate that the visual capabilities of cormorants remain limited underwater (Martin et al., 2008; White et al., 2008, 2007). In fact, great cormorants, European shags *Phalacrocorax aristotelis* and possibly other species with low underwater visual acuity use a foraging technique with which they trigger an escape response in inconspicuous prey; the predatory bird only distinguishes a "moving blur" that is swiftly captured (White et al., 2007).

Nevertheless, some birds commonly dive to depths where light levels are so low that they can be considered nocturnal foragers (Martin, 2017). For example, auks go down to more than 100 m in search for food items and some penguins to 300 m. Once again, specific adaptations have been described, including the presence of very wide pupils to accommodate the low light levels. These adaptations come with the drawback that, if a bird is exposed to unnatural high light levels while foraging at depth (e.g. lights put on a net to make it more visible), the pupils will shut rapidly and render the animal blind, and potentially more exposed to capture. Consequently, Martin and Crawford (2015) advice against using light at depth as a bycatch mitigation method.

ii. Touch

Most seaducks prey upon slow-moving or non-motile organisms attached to the substratum, which they detect using their sense of touch, sometimes combined with taste cues (Martin, 2017; Martin et al., 2007). Although, some birds possess facial bristle feather, similar to vibrissae in mammals, which they use as tactile receptors (Brown, 2008; Cunningham, 2010), tactile detection of prey items underwater by seabirds is primarily done with their sensitive bill. In addition, it cannot be excluded that birds preying upon evasive prey also use tactile cues occasionally. Common guillemots *U. aalge*, for instance, can forage at such low light levels that they are literally swimming blind at depth (Regular et al., 2011). Thus, they depend on random encounters with prey, which are detected either with vision, or after swimming into them (Martin, 2017).

iii. Audition

Hearing is a major sense for birds, notably for intra-specific communication (songs, calls, etc…), and has been the object of extensive research for decades (Dooling, 1992). Auditory capacities of diving seabirds have been specifically explored through behavioural studies and direct physiological measurements (using auditory brainstem responses, ABR) for a number of species (Crowell, 2016; Maxwell et al., 2017, 2016), showing that maximal in-air auditory capabilities range between 1.7 and 3.0 kHz for most species (Crowell et al., 2015). Yet, few studies have demonstrated the ability of seabirds to hear underwater, with the exception of the great cormorant (Hansen et al., 2017; Johansen et al., 2016). Although there is no undisputable evidence that great cormorants (and generally seabirds), are able to hear directionally while diving, i.e. pin point the direction of a sound (Martin and Crawford, 2015), based on the results from psychophysics tests on the great cormorant (Figure 12), this hypothesis cannot be excluded (Hansen et al., 2017). A recent publication compared the ABR of fledging great cormorants in-air and underwater, and concluded that underwater hearing of this bird were at least as good in both media; associated with the particular morphological features of the outer and middle ear of these birds, these results suggest that great cormorants are in fact adapted to amphibious hearing (Larsen et al., 2020).

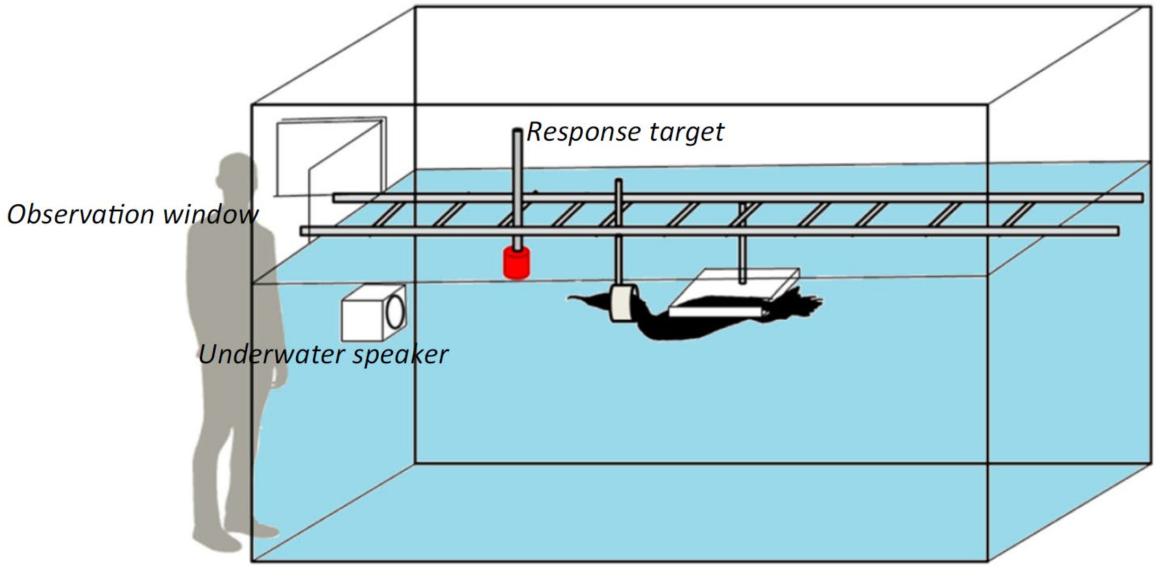

*Figure 12: **Example of a psychophysics experimental setup, to test underwater hearing abilities of a seabird** (here, a great cormorant* P. carbo*). The bird is trained to swim to the test station and its hearing capabilities are tested using a GO/NOGO paradigm. That is, if a tone is heard, the bird must tap the response target with the beak (GO); it will then receive a fish reward. If no sound is heard, the bird must stay put until the trial is over (NOGO); it can then receive a fish reward. From Hansen et al. (2017).*

iv. Olfaction (chemoreception)

Seabird do not seem to be able to use olfaction while diving, either for orientation or for detecting potential prey (Martin, 2017). Nevertheless, most procellariforms have a developed sense of smell, which they use to detect aggregations of prey from long distances during prospection flights (Nevitt, 2000; Nevitt et al., 2008), or to find their way back to the nesting colonies (Padget et al., 2017).

4.2. Preventing seabird bycatch

As described previously, bycatch in gillnets results from the failure for a seabird to detect the fishing gear in due time to avoid it, ending up in the animal's entanglement and eventual drowning. In a recent literature review, Northridge et al. (2017) identified 16 factors that influence seabird bycatch in gillnets, and divided them in 4 categories: **environmental** (wind or weather, water depth), **operational** (location, time of day, time of year or season), **technical** (mesh size, net height, depth set, twine colour, twine type, lead line) and **biological** (vision, acoustic, other behaviour). Understanding these mechanisms, i.e. the underlying causes of unintentional bycatch of seabirds in gillnets, is fundamental to conceptualise and design appropriate mitigation measures. Bycatch reductions can be achieved by (1) **reducing the overall gillnet fishing effort**, thereby limiting the number of potential encounters between wild birds and fishing gears; or (2) **reducing the bycatch per unit effort** (BPUE). Ultimately, adequate reduction in bycatch depends on what is considered "low enough" in specific situations. A critically endangered species will need radical solutions to eliminate any additional fishing mortality, whereas for less vulnerable species, less costly measures that ensure the stability of the populations may suffice. In both cases, following the principles of ecosystem approaches to fisheries management, careful assessment of the collateral effects that potential changes in fishing practices have on the whole marine ecosystem are necessary, e.g. using appropriate metrics to rank the impact of each measure (Chuenpagdee et al., 2003).

i. Reducing the fishing effort

Fisheries closures

Incidental catches in gillnets are a consequence of the co-occurrence of target species and vulnerable bycatch species; time/area closures aim at reducing the level of spatial and/or temporal overlap between target and bycatch species. Such closures can be limited in space, time (time of year, time of day), or permanent (e.g. Marine Protected Area).

The long-lasting closure of the large-scale salmon and cod gillnet fishery in the Eastern Canada in 1992 offers an explicit example of how closing a fishery can indirectly benefit the demographics of entire seabird populations (Regular et al., 2013). The authors compared the local breeding populations of auks (3 species) and Northern gannet *Morus bassanus*, all susceptible to bycatch in gillnets, before and after the moratorium was enacted. Less than 20 years after banning gillnets from the area, the populations of all four vulnerable bird species increased by one order of magnitude. Generally, for vulnerable seabird populations who aggregate in high numbers at particular times of the year, fisheries closures may be the best mitigation approach (Žydelis, 2013). For instance, observed bycatch rates of breeding populations of auks in Newfoundland were much higher near the colonies as further away from them, as most individuals were only flying relatively short distances to their foraging grounds (Benjamins et al., 2008; Piatt and Nettleship, 1987). Setting up protected areas around the breeding colonies could thus considerably limit the interactions between these seabirds and fisheries, and result in a reduction in bycatch rates. Likewise, timely and seasonal closures of important resting and feeding grounds outside the breeding season may be used to prevent bycatch of migrating seabirds. For instance, in the North Sea, 60% of the annual observed bycatch was reported from December to March (Erdmann, 2006). Similarly, bycatch rates estimated in the German Baltic coastal gillnet fisheries were higher in the winter fishing season (November to April) than in the summer (Bellebaum et al., 2013; Mendel et al., 2008). In Denmark, similar patterns were recorded in the Sound and in the Belt Sea (papers 1 and 2). More species could benefit from short localised fisheries closures, e.g. during the moulting period – a critical time of the year when birds cannot fly – when some species aggregate in high numbers (e.g. Figure 13). Additionally, fishing closures may also be restricted to specific times of the day when bycatch risk is the highest. For instance, in the Fraser sockeye/pink salmon driftnet fisheries (State of Washington, USA), fishing restrictions apply during nighttime to protect seabirds specifically (Washington Department of Fish and Wildlife, 2019).

Adequately enforced fisheries closures are without doubt the most straightforward way to tackle the problem of bycatch: clearly, no fishing will not generate any bycatch. However, attentive management is essential as the closure of one area could displace the problem to another, and wipe out the benefits from the closed areas. Besides, prohibiting fishing, even temporarily, is likely to provoke frictions with the fishers for whom a closure could reduce the profit they generate from their activity (O'Keefe et al., 2014; Suuronen et al., 2010). Real-time spatial management could offer a satisfying solution to preserve vulnerable seabird populations from bycatch together with maintaining economic viability for the fishers. With the support of modern communication systems, bycatch data can be collected from fishing vessels, transmitted to the authorities and analysed rapidly. Based on the observed bycatch rates, a decision to close a specific area can then be enacted within hours (Little et al., 2015).

Figure 13: August 2019, near Hirtshals (Denmark). *A few of the thousands of common scoters (*Melanitta nigra*) gathered at a short distance from the shore.*

In the European Union, the Natura 2000 coordinated network of protected areas was designed specifically to ensure the preservation of biodiversity at the scale of the continent, and to reduce habitat fragmentation, which is largely viewed as a major threat to the conservation of vulnerable species. Special Areas of Conservation (SACs) and Special Protection Areas (SPAs) need to be designated by each EU Member States to protect habitats and species of Community interest, and can include specific regulations to limit the impact of anthropogenic activities. For instance, fishing restrictions may apply in a marine protected area to protect wildlife and conserve biodiversity.

Bycatch caps

Quota allocation is generally associated to the management of commercial stocks (e.g. Total Allowable Catches in the EU). Fishing quota, i.e. the maximal quantity of a species that is allowed to be extracted, can be applied at individual vessel level or fleet-wide to manage commercial stocks or bycatch species (Alverson et al., 1994). Based on estimations of the states of the populations, the maximum sustainable yield can be determined for each stock as well as the corresponding fishing mortality. In the EU, species-specific maximal threshold values have to be defined regionally for the mortality rate of non-target species from incidental bycatch (EU, 2017b). For vulnerable seabirds, maximal sustainable mortality rates can be calculated in a variety of ways depending on prior existing knowledge of the population demographics. Nevertheless, there are limitations in the current methods to calculate these thresholds, which need to be carefully considered (Section 3.3). The Joint ICES/OSPAR/HELCOM Working Group on Marine Birds (JWGBIRD) advices that, in an objective of bycatch elimination, a maximal value of 1% of the species estimated population could be adopted, which includes mortalities from all anthropogenic activity (including fishing) (ICES, 2018). This threshold corresponds to the definition of “small numbers” in the Birds Directive (EC, 2008b).

Alternative gears

In times/areas where gillnets pose a threat to seabirds, regulatory bodies can impose the use of alternative bird-safe fishing gears. New gears should ideally suppress, or at the very least reduce, seabird bycatch, but also maintain, or even increase, fishers’ income. In fact, this approach is more likely to work if fishers are involved in the process of gear testing, and trained with the use of the alternative gear. Additionally, financial compensations, e.g. to cover the cost of acquiring new equipment, may help overcome the initial distrust of the fishing communities (Field et al., 2019). The gears mentioned below are few examples of alternatives to gillnets in order to reduce seabird bycatch.

Pots and fish traps

Baited pots offer a functional alternative to gillnets and are already in use in a number of countries including Denmark. Pot designs are highly customisable and relatively cheap to build and operate. They can achieve excellent target species selectivity (e.g. using adequate mesh sizes) and are able to prevent the bycatch of unwanted species, including diving birds, by adjusting the size of the entrance (e.g. using an exclusion grid). Besides, unlike most fishing techniques, fish trapped in pots remains alive, so fishers can expect a higher market value for their catch (Koschinski and Strempel, 2012).

Trials conducted in the Baltic Sea testing fish traps and fyke nets as an alternative to gillnets showed encouraging results in terms of seabird bycatch reductions (Vetemaa and Ložys, 2009). However, all these trials, including the ones with longlines, raised concerns regarding the full implementation of these alternative gears in terms of cost for the fishers (acquisition of new equipment and running cost), possible reduced selectivity of target species and potential higher bycatch of other protected species, e.g. seals.

Finally, large fish traps such as pound nets and fyke nets can potentially replace gillnets in shallow areas, and are for the most part safe in relation to bycatch of protected species (however, pound nets equipped with a fyke net aft end can potentially trap seabirds underwater). Nevertheless, such large static structures require important maintenance, and are more labour intensive than gillnets. Besides, heavy algal blooms in the summer and ice in the winter, often limit the duration of the fishing season with these gears. Pontoon traps, which consist of a large fish chamber mounted on a pontoon that can be filled with compressed air, were originally designed to reduce seal depredation in the Swedish salmon fishery. The whole trap can be lifted up to recover the catch and put back into place by adjusting the buoyancy of the pontoon. As with baited pots, the trap entrance can be equipped with a grid to avoid unwanted bird bycatch. Work is currently underway in Denmark and Sweden to assess the selectivity of pontoon traps and their viability for fishers.

Hooks and lines

Hook fisheries, especially longlines, have a dreadful reputation in terms of bycatch of seabirds, and have been reported to kill at least 160,000 birds killed annually in the world, threatening entire populations (Anderson et al., 2011). These extreme numbers mostly pertain to albatrosses, petrels and shearwaters; where the distribution of these pelagic seabirds overlaps with the one of longline fisheries, the death toll can be dramatic if no adequate mitigation measure are undertaken (Melvin et al., 2014; Sullivan et al., 2018). Nevertheless, longlining (or jigging) could be a suitable option in areas where bycatch of seaducks in gillnets constitutes most of the captures of seabirds, as is the case in the Baltic Sea. For instance, in the Eastern Baltic Sea, in trials with demersal longlines designed to catch cods conducted between 2005 and 2009, no bycatch of seabirds were reported (Vetemaa and Ložys, 2009). Problematically, longlines (and jigs) may effectively solve the problem of seaduck bycatch, but could then become a threat to other groups of seabirds, particularly the ones feeding on discard and offal. Surface-feeding piscivorous birds may be attracted to the hooked baits when the lines are near the surface (e.g. seagulls and terns), risking injuries or death. Additionally, other diving piscivorous species may also feed on the baits at depth (e.g. auks or cormorants). Nevertheless, multiple mitigation solutions specifically designed to limit the risk of interactions with seabirds have been developed over the years. A number of pelagic and demersal longlines fisheries worldwide have adopted such measures, and reduced or eliminated the problem of seabird bycatch (Sullivan et al., 2018).

ii. Reducing seabird bycatch per unit effort

Opaque portions of the net as visual deterrent

Early successful attempts to seabird bycatch reductions in net fisheries were conducted in the Puget Sound driftnet salmon fishery at the end of the last century (Melvin et al., 1999). The authors proposed to modify the upper part of the nets in order to make them more visible to the seabirds floating at the surface. The authors controlled the effectiveness of their method by comparing against an identical control the number of the alcids commonly captured in this fishery (here, common guillemot and rhinoceros auklet *Cerorhinca monocerata*), and by comparing the catch rates of the target species (here, sockeye salmon *O. keta*). They tested two versions of a modified driftnet (Figure 14), one with the upper 20 meshes modified to be highly visible (upper 10% of the net), and the other one with the upper 50 meshes (upper 25% of the net). The 20-mesh panels reduced BPUE of common guillemot significantly, but not BPUE of rhinoceros auklet, and maintained CPUE of salmon. The 50-mesh version of the panels significantly reduced BPUE of both seabird species, but also reduced CPUE of salmon. Based on these results, the use of the 20-mesh driftnet panels was adopted later in the fishery management plan, together with other measures to protect seabirds from bycatch (Washington Department of Fish and Wildlife, 2019).

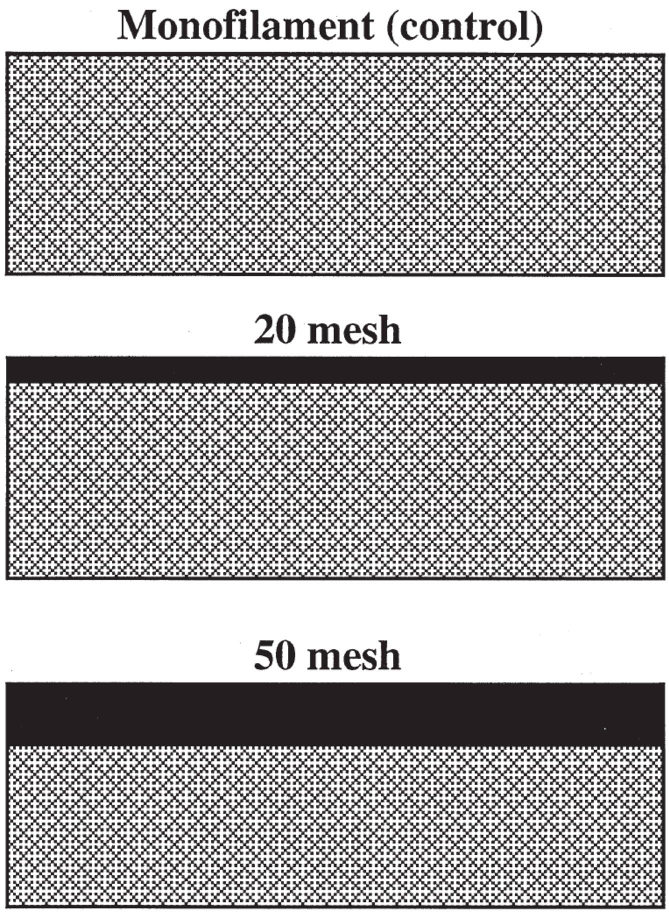

Figure 14: Schematic of the experimental driftnets used in the Puget Sound sockeye salmon fishery (USA), as viewed underwater at 90°. *Modified from Melvin et al.(1999).*

"Reflective" thread material

Barium sulphate impregnated gillnets were designed originally to be more "visible" to odontocetes (toothed whales), thanks to a greater acoustic reflectivity of the twines compared to traditional monofilament nets. Odontocetes use echolocation to perceive their environment; they emit sound waves, which reflect on their surroundings, and listen to the echo to detect the objects ahead of them. In a Canadian fishery, gillnets on which a barium sulphate treatment had been applied were showed to reduce significantly BPUE of harbour porpoise, an odontocete, but also great shearwater *Ardenna gravis*, a diving seabird (Trippel et al., 2003). However, the impregnation treatment coloured the panels dark blue, and the authors hypothesised that the resulting increased visibility of the nets could explain the observed bird bycatch reduction. Nonetheless, a research conducted in the North Sea using a similar trial setup (with nets treated with high-density iron oxide) came to different conclusions. In these trials, harbour porpoise bycatch rates were reduced in the treated nets, but so were catches of target species (here, cod). The authors argued that the observed (by)catch reductions were likely the result of the increased stiffness of the treated nets due to the chemical treatment, rather than the reflectivity or even the colour (Larsen et al., 2007).

Contrast panels

In a paper focusing on the sensory abilities of bycatch-prone animals in gillnet fisheries, Martin & Crawford (2015) proposed using high-contrast warning panels attached onto the nets in order to reduce deadly interactions with vulnerable species (Figure 15). These low-cost contrast panels made of alternate white and black bands would be visually detectable to most vulnerable taxa (i.e. marine birds, reptiles and mammals) and trigger in them an escape response. On the contrary, it is unlikely that the catch rates of target fish species would be reduced using such a bycatch reduction device (BRD). Based on these considerations, Field et al. (2019) tested 0.6 x 0.6 m contrast panels in the Lithuanian and Polish coastal bottom-set gillnet fisheries as a means to reduce the bycatch of seabirds. The use of these panels showed no significant difference in BPUE of seabirds between experimental nets and controls. It might be that the contrast panels covered too little an area of the net. Compared to the driftnet salmon fishery in the Puget Sound, where at least 10% of the nets were made of opaque material (Melvin et al., 1999), the contrast panels tested in the Baltic Sea covered 1% to 8% of the net fleet. What is more, bycatches of long-tailed ducks *C. hyemalis* increased significantly in the nets equipped with contrast panels. The authors propose that long-tailed ducks could be attracted to the warning panels in the winter, as the panels present a black and white contrast similar to the wintering ducks' plumage.

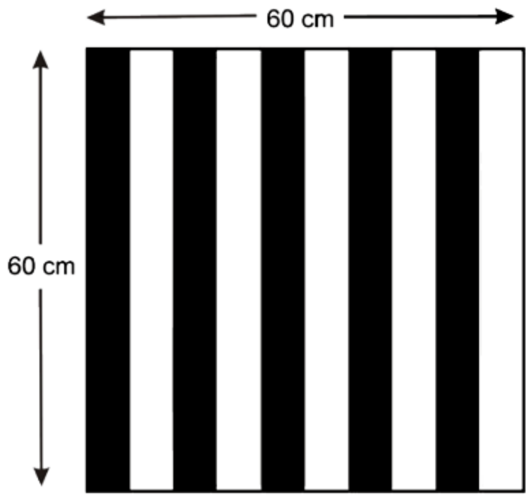

Figure 15: Example of a contrast panel to reduce capture rates of bycatch-vulnerable taxa in gillnet fisheries. *Modified from Martin & Crawford (2015).*

Net illumination

Extensive research has been conducted to reduce bycatch rates of vulnerable taxa using LED lights to illuminate the nets in net fisheries (Bielli et al., 2020; Mangel et al., 2018; Ortiz et al., 2016; Wang et al., 2013, 2010). In particular, illuminating bottom-set gillnets with constant green LED lights reduced the bycatch of Guanay cormorants *L. bougainvillii* by 85.1 % in experimental gillnets compared to controls, without affecting the catch of target species (Mangel et al., 2018). However, the authors also reported the capture of four Peruvian boobies *S. variegata* in the illuminated nets, and none in the controls, which suggest that different groups of birds may react differently to the same visual cues; some species may find the visual cues aversive, while others may be attracted to the lights. During their field experiments, Bielli et al. (2020), who utilised another type of green LED light, captured three diving seabird species in the experimental driftnet sets: the Humboldt penguin *S. humboldti*, the pink-footed shearwater *A. creatopus* and the white-chinned petrel *P. aequinoctialis*, as well as three unidentified birds. The authors reported 84% reduction in BPUE in the illuminated driftnets compared to the controls (no bird was taken in the control or experimental bottom-set gillnets).

In the Baltic Sea, diving piscivorous and benthivorous seabird species are particularly at risk of entanglement in gillnets (Žydelis et al. 2009, Sonntag et al. 2012, Žydelis 2013, Bellebaum et al. 2013, ICES 2018, Paper 1; Paper 2). Following the encouraging results obtained in South America with LED lights, experiments using similar BRDs were thus conducted in the Baltic basin. However, these LED trials generated contrasting results in terms of seabird bycatch reductions. Field et al. (2019) tested two kinds of LED lights in three different gillnet fisheries: (1) constant green LED lights in the Polish bottom-set gillnet fishery for cod; (2) constant green LED lights in the Polish bottom-set gillnet fishery for Atlantic herring *Clupea harengus*; (3) flashing white LED lights in the Lithuanian bottom-set gillnet fishery for smelt. The BRDs were fixed at 10 meters intervals on the headrope (Figure 16). The utilisation of constant green lights showed no significant difference in seabird BPUE between experimental sets and controls, neither in the cod nor in the herring gillnet fisheries. Conversely, in the Lithuanian waters, the experimental sets with flashing white LED lights resulted in significantly higher BPUE of seabirds (81% of whom were long-tailed ducks). Catch per unit effort of target species remained unchanged between experimental sets and controls regardless of the type of LED.

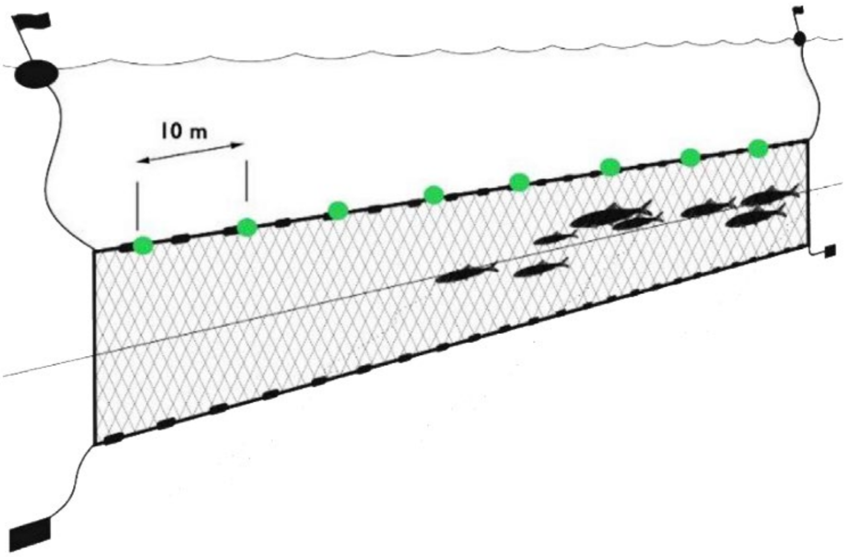

Figure 16: Schematic of the experimental bottom-set gillnets used in Field et al. (2019). *Green: bycatch reduction device (either a constant green, or a flashing white LED light), attached on the headrope at 10 meters interval. Controls were identical bottom-set gillnets with no light. Modified from Field et al. (2019).*

Acoustic deterrents

Since their debut in the late 1990's, acoustic devices have been used successfully as bycatch deterrents for echolocating cetaceans in many fisheries, in particular in order to reduce the bycatch of harbour porpoise (Kraus *et al.*, 1997; Northridge *et al.*, 2017). Soon after Kraus and colleagues' publication in Nature, Melvin and his group tested pingers in the driftnet salmon fishery in the Puget Sound, USA (Melvin et al., 1999). The devices, with a nominal output frequency of 1.5 kHz (± 1 kHz), were attached to the cork line (i.e. the floating headrope of a driftnet) every 50 m. The pingers were designed to emit an identical 300 ms signal at 120 dB every 4 s. The nets equipped with pingers captured significantly less common guillemots than the controls, whereas BPUE of the rhinoceros auklet remained unchanged. The acoustic BRDs did not significantly affect the CPUE of the target species. Since then however, no study has demonstrated an effect of pingers on seabird bycatch in gillnet fisheries.

Surface scarers

BirdLife and its partners are currently developing a visual device to scare seabirds away from static gears, using a "super-stimulus" to induce an aversive reaction in wild birds (Inglis, 1980). On land, showing a moving eye-shape pattern on a large television screen (suggesting that the "eyes" are moving toward the watcher; Figure 17) could effectively repel birds of prey and corvidae from an area (Hausberger et al., 2018).

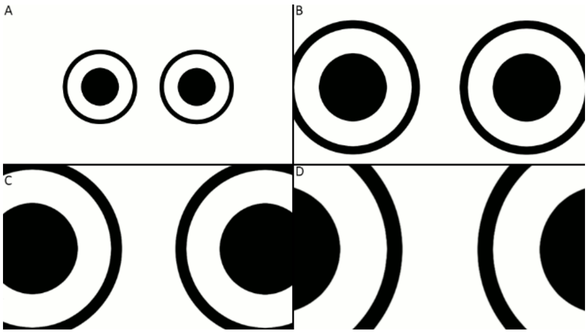

Figure 17: Snapshots of a sequence figuring an eye-shaped pattern "looming" toward the observer. *The full sequence (A, B, C, then D) is composed of more intermediate images, and its repetition induces a super-stimulus, which is supposed to repel birds from the screened area. Modified from Hausberger et al. (2018).*

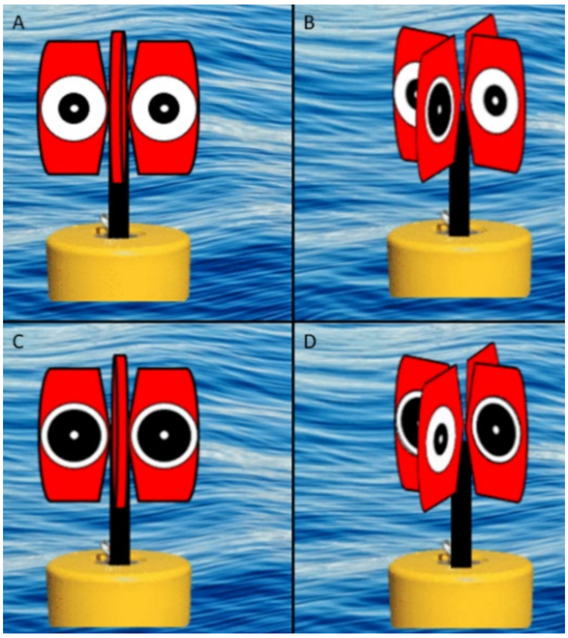

Figure 18: Principle of a floating windmill seabird scarer for static fishing gears. *The windmill turns with the wind, alternatively showing small and big eye shapes (A, B, C, then D), inducing an avoidance reaction in seabirds. Courtesy of Yann Rouxel, Royal Society for the Protection of Birds (RSPB).*

Contrary to other visual or acoustic bird scarers commonly used on land, which lose efficacy as birds become habituated to the stimuli (Inglis, 1980; Stevens et al., 2007), the "looming eye-shape" method did not seem to induce habituation in the targeted groups of birds. A similar approach to prevent seabirds from approaching is currently being tested at sea using low-tech floating windmills (Figure 18). The project is currently testing prototype fixed buoys to assess their efficacy as bird scarers. If the pilot trials are successful, this method will be adapted to net fisheries, by attaching the buoys at regular intervals on the head rope of bottom-set gillnets, or directly on the cork line of a driftnet (Yann Rouxel, Royal Society for the Protection of Birds (RSPB), personal communication).

4.3. Case study: seabird bycatch mitigation in a small-scale gillnet fishery in Denmark (Paper 3)

In the winter 2018-2019, two potential seabird BRDs were trialled in a real commercial fishing situation on a Danish gillnetter operating in the coastal gillnet fishery for cod and flatfish in the Sound (Western Baltic). We tested a flashing white LED light (visual deterrent; Figure 19a) and a 3 kHz pinger (acoustic deterrent; Figure 19b).

The lights were the exact same model than the one used by Field et al. (2019) in the coastal Lithuanian gillnet fishery. In 2017-2018, preliminary results from these authors indicated that the vast majority of bycatches in gillnets in Lithuania were long-tailed ducks, a benthivorous seaduck (Žydelis and Ruškytė, 2005). On the contrary, analyses from EM data showed that the species being most susceptible to bycatch in Danish coastal waters during the winter are the common eider (a benthivorous seaduck), and two pelagic-feeding species, the great cormorant and the common guillemot (Paper 1). Therefore, running experimental trials using LED lights in Danish waters was an opportunity to test the possible differential response to the same visual signal in two groups of seabirds.

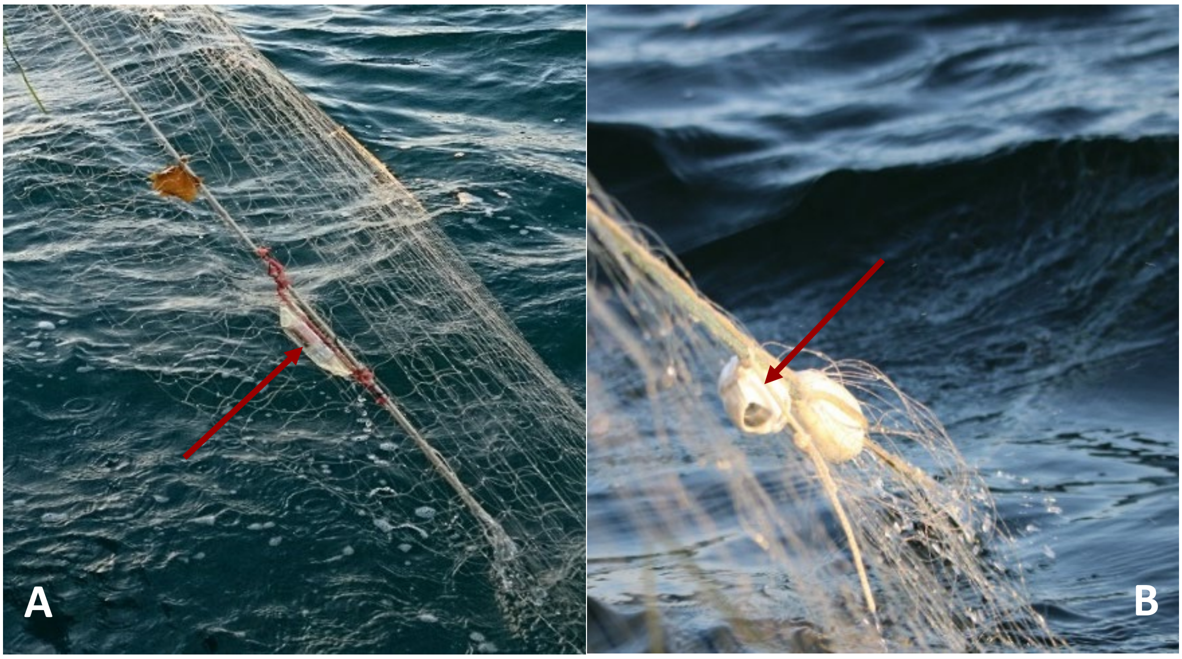

Figure 19: Bycatch reduction devices used in the trials conducted in a commercial bottom-set gillnet fishery in the Western Baltic. *A. Flashing white LED light;* ***B.*** *3 kHz pinger attached on the headrope (next to a float).*

In parallel, the results obtained in the Puget Sound two decades ago suggest that at least some seabirds are deterred from fishing gears when using acoustic alarms (Melvin et al., 1999). In that study, the pingers were tuned to operate at a nominal frequency of 1.5 kHz, based on general knowledge of hearing of birds and of salmon gathered in the literature. Since then, however, although important progress has been made to understand their in-air and underwater auditory capabilities, seabirds remain largely understudied compared to land birds (Crowell, 2016). Recently, in-air and underwater hearing sensitivities of the great cormorant were investigated using ABR and psychophysical responses (Hansen et al., 2017; Johansen et al., 2016; Larsen et al., 2020; Maxwell et al., 2016). Psychophysical experiments conducted on an adult male great cormorant revealed that his underwater hearing peak sensitivity was situated at approximately 2 kHz with a threshold of 71 dB re 1 µPa (Hansen et al., 2017), while ABR measurements on a sample of 8 great cormorants found an average peak at 1 kHz with a threshold of 85 dB re 1 µPa (Larsen et al., 2020). For comparison, the psychophysical-threshold of the long-tailed duck is above 90-95 dB re 1 µPa (Crowell, 2014, cited in Hansen *et al.*, 2017). We could not find equivalent measurements of the underwater hearing sensitivity for other bird species, but in-air ABRs of 10 seabirds indicated that the best hearing frequencies were in the range of 1.7 kHz to 3 kHz (Crowell et al., 2015). Although, none of these studies resolved the question of the ability for seabirds to hear directionally while diving, the recent results from Larsen et al. (2020) show possible anatomical adaptations to underwater hearing in the great cormorant.

As described in Paper 3, the experimental trials in the Western Baltic suggested that flashing white LED lights could reduce the bycatch rates of pelagic-feeding seabirds (i.e., in this area, great cormorant, common guillemot and razorbill), but not benthic feeding seaducks (here, common eider, common scoter and velvet scoter). However, these results are based on very small bycatch samples, and should be considered inconclusive until more data can be collected. Using the same analysis with the data from the 3 kHz pingers trials indicated that these BRDs failed at reducing bycatch rates for both groups of seabirds. Catch rates of target species remained unchanged for both treatments, except for a small, yet significant, increase of the CPUE of plaice when using LED lights. Nevertheless, as stated above, these results are based on few seabird bycatch records (respectively, 6 and 3 pelagic-feeding seabirds were captured in the nets equipped with LED lights, and in the controls). Although the fieldwork took place from the end of fall up until early spring – when seabird bycatch is maximal in Danish coastal waters (Paper 1) – less bycatch events were recorded in the during the trials compared to what was observed during the same period in the previous years. This raises the question of the representativity of these data. Yet, these results seem to be in line with what has been described in the Peruvian gillnet fisheries (Bielli et al., 2020; Mangel et al., 2018), and could partly explain the absence of bycatch reduction registered in Lithuania. That is, pelagic-feeding seabirds, who are using visual cues to detect prey in the water column or near the sea floor, would be startled by repeating flashes of intense light, while seaducks, who forage on organisms living on/in the sea floor would not be so dependent on vision to detect their prey, and would thus ignore the lights. More data must be collected to confirm (or infirm) this hypothesis.

In January 2020, we started a new collaboration with a commercial gillnetter targeting lumpsucker in Kattegat (Western Baltic) to test LED lights in this fishery. Lumpsuckers are targeted mostly for their valuable row, across the whole distribution range of the species, during the cold winter months when the females migrate to shallow waters to spawn. The lumpsucker gillnet fisheries are very problematic with regard to seabird bycatch (Christensen-Dalsgaard et al., 2019), and preliminary analyses of EM data collected on this same vessel in 2019 confirmed that BPUE of pelagic-feeding seabirds (in particular of common guillemot) could be very high. The results from this new field experiment should be able to reveal with more certainty whether net illumination can effectively reduce seabird bycatch.

5. CONCLUDING REMARKS AND FUTURE PERSPECTIVES

5.1. SUMMARY OF THE FINDINGS

Since 2010, the fishing activity from a total of 18 Danish commercial small-scale vessels has been monitored and analysed using electronic monitoring (EM) with closed-circuit television (CCTV) cameras. The analysis of EM data was able to cast a light on a number of parameters usually not accessible from traditional fisheries observer data. Fine-scale temporal and spatial estimates of seabird bycatch rates in gillnets were found to be highly variable in time and in space, resulting partly from the presence of very few, but very influential mass bycatch events. These outliers, though rare from a fisher's viewpoint, could considerably increase fleet-wide bycatch mortality estimates. Fine-scale gillnet fishing effort, e.g. position and number of hauls, depth at soaking location, could be inferred from satellite-tracking data of commercial vessels (AIS). These data were used to calculate conservative estimates of seabird bycatch mortality at fleet level. The last part of this thesis focused on testing seabird bycatch mitigation devices, namely sound (3 kHz pingers) and lights (flashing white LED). A controlled sampling design showed that white LED lights attached to the net might reduce incidental captures of some diving seabirds. However, the low number of registered bycatches could not allow to reach a definitive conclusion and more data need to be collected to confirm these findings.

5.2. FUTURE PERSPECTIVES

The scarcity of data on incidental bycatch of seabirds in gillnet fisheries weakens our ability to calculate reliable mortality estimates in European waters. Long-term monitoring programmes as the one developed in Denmark to survey Danish commercial gillnetters have brought important insights on the magnitude of incidental captures of seabirds in the Western Baltic. Seabird mortality in small-scale gillnet fisheries is probably largely underestimated in many regions because of a lack of sufficient monitoring. Notwithstanding the necessity to increase the quality of seabird bycatch data collected by fisheries observers, electronic monitoring with CCTV cameras on a sample of the fishing fleet appears as an efficient method to obtain additional reliable fisheries-dependent data. This is coherent with the necessity to improve monitoring schemes enacted by the European Union (EU, 2019b) and recently highlighted by ICES (ICES, 2020), by focusing not only on fish stocks, but also on all compartments of the marine ecosystems, including seabirds.

If these data are not collected in the future, the evidence of important bycatch mortality rates in many important fishing areas, in particular around Denmark, as well as the potential detrimental impact of additional mortality on seabird populations, may persuade fisheries managers to decide on drastic measures, following the precautionary principle. In the Baltic and North Sea, fisheries closures (spatially and/or temporarily) are the only measures that are assured to reduce – and indeed suppress – seabird bycatch in gillnet fisheries. Reducing seabird bycatch will probably require a mixture of effective technical mitigation solutions, to reduce bycatch rates, together with adequate fisheries management solutions, including closures, to reduce fishing effort. However, the socio-economic consequences of closures will likely be unacceptable for local fishing communities, and more research should be done to maintain small-scale gillnetting in our waters without impairing the future state of seabird populations. Close collaboration between fishers, authorities and scientists is thus crucial to measure the potential bycatch reductions, the effects on the marine ecosystems and the economic consequences on fishers of implementing such measures.

In the light of the above, additional work with the use of lights as a mitigation device is required to confirm – or infirm – their efficacy at reducing bycatch rates of pelagic-feeding seabirds. In addition, the possible adaptation to underwater hearing of the great cormorant (Hansen et al., 2017; Larsen et al., 2020) may lead to new ways of using sounds to deter seabirds from gillnets. Other mitigation devices are being tested in the Baltic Sea and could be tested in Denmark in the future. For instance, aerial scarers (Figure 17 and Figure 18) could be trialled on some of the numerous pound nets and fyke nets present along the Danish shore. The current prototypes are relatively heavy and bulky, therefore not necessarily practical for small-scale gillnet vessels to deploy. On the other hand, fixed at-sea fishing structures with protruding poles like fykes and pound nets tend to attract coastal seabirds, e.g. great cormorant and several species of gulls, who use them alternatively for resting and feeding on trapped fish. A simple experiment would consist of testing whether surface scarers induce a reaction and potentially a displacement of these birds, in particular of the great cormorant, which is one of the species the most prone to bycatch in gillnets in Danish waters.

Moreover, an obvious extension of the work presented in this thesis is to establish whether the current levels of bycatch in Danish gillnet fisheries constitute a threat for vulnerable seabird populations. Several methods could be employed, as the ones listed in section 3.3. Note that Potential Biological Removal (PBR) should only be used as a rough indication of a potential bycatch problem for a given species, as recommended in ICES (2013). More elaborated population models need to be employed to predict the future trends of the populations under bycatch pressure. For instance, the Baltic and Wadden Sea flyway population of common eiders *S. mollissima*, which is the most frequent bycaught seabird in gillnets in Danish waters, is currently declining and trends indicate that this decline will continue in the near future (Tjørnløv et al., 2019). Refining common eider bycatch estimates at fleet level would be valuable to evaluate the total mortality of this species and help wildlife managers to maintain the population at sustainable levels. Besides, thanks to the close collaborations established with fishers within the EM programme, we started collecting the carcasses of bycaught seabirds from several fishing vessels since 2017. Currently, more than 200 dead birds, mostly common eiders, are individually identified and conserved in freezers. Information on date and location of capture are available for each animal, as well as the characteristics of the nets in which the birds were caught, i.e. soak time, mesh size, netting colour, presence of other birds in the same net fleet, etc… A project to analyse the stomach content of the birds is presently underway, but genetic analyses should follow to link these birds back to their population of origin. With this information, we hope to be able to assess the effects of bycatch in gillnets for individual (flyway) populations.

REFERENCES

Alverson, D.L., Freeberg, M.H., Murawski, S.A., Pope, J.G., 1994. A Global Assessment of Fisheries Bycatch and Discards, FAO. ed, FAO Fisheries Technical Paper. Food & Agriculture Organisation, Rome.

Amande, M.J., Chassot, E., Chavance, P., Murua, H., de Molina, A.D., Bez, N., 2012. Precision in bycatch estimates: the case of tuna purse-seine fisheries in the Indian Ocean. ICES J. Mar. Sci. 69, 1501–1510. https://doi.org/10/f37v74

Andersen, B.S., Ulrich, C., Eigaard, O.R., Christensen, A.-S., 2012. Short-term choice behaviour in a mixed fishery: investigating métier selection in the Danish gillnet fishery. ICES J. Mar. Sci. 69, 131–143. https://doi.org/10/bf9cm9

Anderson, O.R.J., Small, C.J., Croxall, J.P., Dunn, E.K., Sullivan, B.J., Yates, O., Black, A., 2011. Global seabird bycatch in longline fisheries. Endanger. Species Res. 14, 91–106. https://doi.org/10/d5dgwb

Anonymous, 2019. Annual report from Denmark on the implementation of Council Regulation (EC) No 812/2004. Unpublished report to the EC. DTU Aqua.

Appleby, T., Harrison, J., 2019. Taking the Pulse of Environmental and Fisheries Law: The Common Fisheries Policy, the Habitats Directive, and Brexit. J. Environ. Law 31, 443–464. https://doi.org/10/ggkz84

Babcock, E.A., Pikitch, E.K., Hudson, C.G., 2003. How much observer coverage is enough to adequately estimate bycatch? Pew Institute of Ocean Science and Oceana.

Bærum, K.M., Anker-Nilssen, T., Christensen-Dalsgaard, S., Fangel, K., Williams, T., Vølstad, J.H., 2019. Spatial and temporal variations in seabird bycatch: Incidental bycatch in the Norwegian coastal gillnet-fishery. PLoS ONE 14. https://doi.org/10/ggkz9g

Bartholomew, D.C., Mangel, J.C., Alfaro-Shigueto, J., Pingo, S., Jimenez, A., Godley, B.J., 2018. Remote electronic monitoring as a potential alternative to on-board observers in small-scale fisheries. Biol. Conserv. 219, 35–45. https://doi.org/10/gc8phz

Bellebaum, J., Schirmeister, B., Sonntag, N., Garthe, S., 2013. Decreasing but still high: bycatch of seabirds in gillnet fisheries along the German Baltic coast. Aquat. Conserv. Mar. Freshw. Ecosyst. 23, 210–221. https://doi.org/10/f4wfmn

Bellido, J.M., Santos, M.B., Pennino, M.G., Valeiras, X., Pierce, G.J., 2011. Fishery discards and bycatch: solutions for an ecosystem approach to fisheries management? Hydrobiologia 670, 317–333. https://doi.org/10/cnt5qz

Benjamins, S., Kulka, D.W., Lawson, J., 2008. Incidental catch of seabirds in Newfoundland and Labrador gillnet fisheries, 2001–2003. Endanger. Species Res. 5, 149–160. https://doi.org/10/brpj4d

Bielli, A., Alfaro-Shigueto, J., Doherty, P.D., Godley, B.J., Ortiz, C., Pasara, A., Wang, J.H., Mangel, J.C., 2020. An illuminating idea to reduce bycatch in the Peruvian small-scale gillnet fishery. Biol. Conserv. 241, 108277. https://doi.org/10/ggkz79

BirdLife International, 2019. BirdLife position D1criteria_02092019_FINAL.pdf [WWW Document]. URL https://portal.helcom.fi/meetings/Incidental%20bycatch%20WS%201-2019-647/MeetingDocuments/BirdLife%20position%20D1criteria_02092019_FINAL.pdf (accessed 3.4.20).

BirdLife International, 2015. IUCN Red List of Threatened Species: Common Eider [WWW Document]. IUCN Red List Threat. Species. URL https://www.iucnredlist.org/en (accessed 2.27.20).

Bolker, B.M., Brooks, M.E., Clark, C.J., Geange, S.W., Poulsen, J.R., Stevens, M.H.H., White, J.-S.S., 2009. Generalized linear mixed models: a practical guide for ecology and evolution. Trends Ecol. Evol. 24, 127–135. https://doi.org/10/fwwbzr

Boyce, M.S., 1992. Population viability analysis. Annu. Rev. Ecol. Syst. 23, 481–497. https://doi.org/10/bqp2nz

Bradbury, G., Shackshaft, M., Scott-Hayward, L., Rexstad, E., Miller, D., Edwards, D., 2017. Risk assessment of seabird bycatch in UK waters (No. MB0126). Wildfowl & Wetlands Trust (Consulting), United Kingdom.

Brown, S., 2008. Bird uses "whiskers" like a cat. Nature news.2008.674. https://doi.org/10/djxsns

Burgess, M.G., McDermott, G.R., Owashi, B., Peavey Reeves, L.E., Clavelle, T., Ovando, D., Wallace, B.P., Lewison, R.L., Gaines, S.D., Costello, C., 2018. Protecting marine mammals, turtles, and birds by rebuilding global fisheries. Science 359, 1255–1258. https://doi.org/10/gc8mrz

Cadrin, S.X., Dickey-Collas, M., 2015. Stock assessment methods for sustainable fisheries. ICES J. Mar. Sci. 72, 1–6. https://doi.org/10/ggmxjn

Christensen-Dalsgaard, S., Anker-Nilssen, T., Crawford, R., Bond, A., Sigurðsson, G.M., Glemarec, G., Hansen, E.S., Kadin, M., Kindt-Larsen, L., Mallory, M., Merkel, F.R., Petersen, A., Provencher, J., Bærum, K.M., 2019. What's the catch with lumpsuckers? A North Atlantic study of seabird bycatch in lumpsucker gillnet fisheries. Biol. Conserv. 240, 108278. https://doi.org/10/ggk9gb

Chuenpagdee, R., Morgan, L.E., Maxwell, S.M., Norse, E.A., Pauly, D., 2003. Shifting gears: assessing collateral impacts of fishing methods in US waters. Front. Ecol. Environ. 1, 517–524. https://doi.org/10/bmztc4

Clay, T.A., Small, C., Tuck, G.N., Pardo, D., Carneiro, A.P.B., Wood, A.G., Croxall, J.P., Crossin, G.T., Phillips, R.A., 2019. A comprehensive large-scale assessment of fisheries bycatch risk to threatened seabird populations. J. Appl. Ecol. 56, 1882–1893. https://doi.org/10/ggmpg8

Crawford, R.J.M., Shelton, P.A., 1978. Pelagic fish and seabird interrelationships off the coasts of South West and South Africa. Biol. Conserv. 14, 85–109. https://doi.org/10/bhbnpf

Crowell, S.C., 2016. Measuring In-Air and Underwater Hearing in Seabirds, in: The Effects of Noise on Aquatic Life II. Springer, pp. 1155–1160.

Crowell, S.E., Wells-Berlin, A.M., Carr, C.E., Olsen, G.H., Therrien, R.E., Yannuzzi, S.E., Ketten, D.R., 2015. A comparison of auditory brainstem responses across diving bird species. J. Comp. Physiol. A 201, 803–815. https://doi.org/10.1007/s00359-015-1024-5

Croxall, J.P., Butchart, S.H.M., Lascelles, B., Stattersfield, A.J., Sullivan, B., Symes, A., Taylor, P., 2012. Seabird conservation status, threats and priority actions: a global assessment. Bird Conserv. Int. 22, 1–34. https://doi.org/10/ggk25d

Cunningham, S.J., 2010. Tactile senses and foraging in birds, with emphasis on kiwi: a thesis presented in partial fulfilment of the requirements for the degree of Doctor of Philosophy in Ecology at Massey University, Manawatu, New Zealand (PhD Thesis). MasseyUniversity, Manawatu, New Zealand.

Cury, P.M., Boyd, I.L., Bonhommeau, S., Anker-Nilssen, T., Crawford, R.J.M., Furness, R.W., Mills, J.A., Murphy, E.J., Österblom, H., Paleczny, M., Piatt, J.F., Roux, J.-P., Shannon, L., Sydeman, W.J., 2011. Global Seabird Response to Forage Fish Depletion—One-Third for the Birds. Science 334, 1703–1706. https://doi.org/10/fzk8sp

Dalskov, J., Kindt-Larsen, L., 2009. Final report of fully documented fishery (No. 204–2009), DTU Aqua. DTU Aqua, National Institute of Aquatic Resources, Charlottenlund.

Davies, R.W.D., Cripps, S.J., Nickson, A., Porter, G., 2009. Defining and estimating global marine fisheries bycatch. Mar. Policy 33, 661–672. https://doi.org/10/cz2wzc

Daw, T., Gray, T., 2005. Fisheries science and sustainability in international policy: a study of failure in the European Union's Common Fisheries Policy. Mar. Policy 29, 189–197. https://doi.org/10/dmvssr

de Souza, E.N., Boerder, K., Matwin, S., Worm, B., 2016. Improving fishing pattern detection from satellite AIS using data mining and machine learning. PloS One 11. https://doi.org/10/gbqxjd

Degel, H., Petersen, I.K., Holm, T.E., Kahlert, J., 2010. Fugle som bifangst i garnfiskeriet. Estimat af utilsigtet bifangst af havfugle igarnfiskeriet i området omkring Ærø (No. DTU Aqua-rapport nr. 227-2010). DTU Aqua. Institut for Akvatiske Ressourcer, Charlottenlund.
Dias, M.P., Martin, R., Pearmain, E.J., Burfield, I.J., Small, C., Phillips, R.A., Yates, O., Lascelles, B., Borboroglu, P.G., Croxall, J.P., 2019. Threats to seabirds: A global assessment. Biol. Conserv. 237, 525–537. https://doi.org/10/ggkz85
Dietrich, K.S., Parrish, J.K., Melvin, E.F., 2009. Understanding and addressing seabird bycatch in Alaska demersal longline fisheries. Biol. Conserv. 142, 2642–2656. https://doi.org/10/chvwnv
Dillingham, P.W., Fletcher, D., 2011. Potential biological removal of albatrosses and petrels with minimal demographic information. Biol. Conserv. 144, 1885–1894. https://doi.org/10/fwk47k
Dillingham, P.W., Fletcher, D., 2008. Estimating the ability of birds to sustain additional human-caused mortalities using a simple decision rule and allometric relationships. Biol. Conserv. 141, 1783–1792. https://doi.org/10/c2r8bp
Dinsdale, R., Catarino, R., Needle, C.L., 2013. Comparing the costs of onboard observers and Remote Electronic Monitoring (REM): a Scottish Case Study, in: 7th International Fisheries Observer & Monitoring Conference, Vina Del Mar, Chile. pp. 8–12.
Dooling, R.J., 1992. Hearing in birds, in: The Evolutionary Biology of Hearing. Springer, pp. 545–559.
EC, 2020. Database - Eurostat [WWW Document]. URL https://ec.europa.eu/eurostat/data/database (accessed 3.11.20).
EC, 2017. Implementation and evaluation of Regulation (EC) 1224/2009 establishing a Union control system for ensuring compliance with the rules of the common fisheries policy as required under Article 118 REFIT Evaluation of the impact of the fisheries regulation {SWD(2017) 134 final} (REPORT FROM THE COMMISSION TO THE EUROPEAN PARLIAMENT AND THE COUNCIL). European Commission, Brussels.
EC, 2012. Action Plan for reducing incidental catches of seabirds in fishing gears {SWD(2012) 369 final} {SWD(2012) 370 final}. https://doi.org/10.1163/2210-7975_HRD-4679-0058
EC, 2009. Council Regulation (EC) No 1224/2009 of 20 November 2009 establishing a Community control system for ensuring compliance with the rules of the common fisheries policy, amending Regulations (EC) No 847/96, (EC) No 2371/2002, (EC) No 811/2004, (EC) No 768/2005, (EC) No 2115/2005, (EC) No 2166/2005, (EC) No 388/2006, (EC) No 509/2007, (EC) No 676/2007, (EC) No 1098/2007, (EC) No 1300/2008, (EC) No 1342/2008 and repealing Regulations (EEC) No 2847/93, (EC) No 1627/94 and (EC) No 1966/2006. Off. J. Eur. Union 52, 1–50.
EC, 2008a. Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive) (Text with EEA relevance), 164.
EC, 2008b. Guidance document on hunting under Council Directive 79/409/EEC on the conservation of wild birds “The Birds Directive".
EEC, 1992. Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora, 206.
EEC, 1979. Council Directive 79/409/EEC of 2 April 1979 on the conservation of wild birds, 103.
Erdmann, F., 2006. Untersuchungen von Vogelbeifängen in der deutschen Küstenfischerei der Ostsee durch eine Umfrage zur Stellnetzfischerei. Räumliches Zeitliches Muster Verluste Von See-Wasservögeln Durch Küstenfisch. Mecklenbg.-Vorpommern Möglichkeiten Zu Deren Minderung Greifswald Neu Broderstorf 5–40.
EU, 2019a. Regulation (EU) 2019/1241 of the European Parliament and of the Council of 20 June 2019 on the conservation of fisheries resources and the protection of marine ecosystems through technical measures, amending Council Regulations (EC) No 1967/2006, (EC) No 1224/2009 and Regulations (EU) No 1380/2013, (EU) 2016/1139, (EU) 2018/973, (EU) 2019/472 and (EU) 2019/1022 of the European Parliament and of the Council, and repealing Council Regulations

(EC) No 894/97, (EC) No 850/98, (EC) No 2549/2000, (EC) No 254/2002, (EC) No 812/2004 and (EC) No 2187/2005, OJ L.

EU, 2019b. Commission Delegated Decision (EU) 2019/910 of 13 March 2019 establishing the multiannual Union programme for the collection and management of biological, environmental, technical and socioeconomic data in the fisheries and aquaculture sectors, OJ L.

EU, 2017a. Regulation (EU) 2017/1004 of the European Parliament and of the Council of 17 May 2017 on the establishment of a Union framework for the collection, management and use of data in the fisheries sector and support for scientific advice regarding the common fisheries policy and repealing Council Regulation (EC) No 199/2008, OJ L.

EU, 2017b. Commission Decision (EU) 2017/848 of 17 May 2017 laying down criteria and methodological standards on good environmental status of marine waters and specifications and standardised methods for monitoring and assessment, and repealing Decision 2010/477/EU (Text with EEA relevance.), OJ L.

EU, 2016. Regulation (EU) 2016/1139 of the European Parliament and of the Council of 6 July 2016 establishing a multiannual plan for the stocks of cod, herring and sprat in the Baltic Sea and the fisheries exploiting those stocks, amending Council Regulation (EC) No 2187/2005 and repealing Council Regulation (EC) No 1098/2007, OJ L.

EU, 2015. Commission Implementing Regulation (EU) 2015/1962 of 28 October 2015 amending Implementing Regulation (EU) No 404/2011 laying down detailed rules for the implementation of Council Regulation (EC) No 1224/2009 establishing a Community control system for ensuring compliance with the rules of the common fisheries policy, OJ L.

EU, 2013. Regulation (EU) No 1380/2013 of the European Parliament and of the Council of 11 December 2013 on the Common Fisheries Policy, amending Council Regulations (EC) No 1954/2003 and (EC) No 1224/2009 and repealing Council Regulations (EC) No 2371/2002 and (EC) No 639/2004 and Council Decision 2004/585/EC, OJ L.

EU, 2011. Commission Implementing Regulation (EU) No 404/2011 of 8 April 2011 laying down detailed rules for the implementation of Council Regulation (EC) No 1224/2009 establishing a Community control system for ensuring compliance with the rules of the Common Fisheries Policy, OJ L.

EU, 2010. Directive 2009/147/EC of the European Parliament and of the Council of 30 November 2009 on the conservation of wild birds, 020.

Evans, R., Molony, B., 2011. Pilot evaluation of the efficacy of electronic monitoring on a demersal gillnet vessel as an alternative to human observers (Fisheries Research Report No. 221). Department of Fisheries, Western Australia, North Beach, Western Australia.

FAO (Ed.), 2011. International guidelines on bycatch management and reduction of discards =: Directives internationales sur la gestion des prises accessoires et la réduction des rejets en mer = Directrices internacionales para la ordenación de las capturas incidentales y la reducción de los descartes. Food and Agriculture Organization of the United Nations, Rome.

Field, R., Crawford, R., Enever, R., Linkowski, T., Martin, G., Morkūnas, J., Morkūnė, R., Rouxel, Y., Oppel, S., 2019. High contrast panels and lights do not reduce bird bycatch in Baltic Sea gillnet fisheries. Glob. Ecol. Conserv. 18, e00602. https://doi.org/10/ggkz8q

Fiskeridirektoratet, 2020. Fiskeridirektoratet [WWW Document]. Dan. Fish. Agency - Fish. Stat. Vessels. URL https://dwp.fiskeristyrelsen.dk/fartoejstabel/fartoejsrapport__front_matter (accessed 2.21.20).

Fournier, M., Casey Hilliard, R., Rezaee, S., Pelot, R., 2018. Past, present, and future of the satellite-based automatic identification system: areas of applications (2004–2016). WMU J. Marit. Aff. 17, 311–345. https://doi.org/10/ggm6hj

Frederiksen, M., Wright, P.J., Harris, M.P., Mavor, R.A., Heubeck, M., Wanless, S., 2005. Regional patterns of kittiwake Rissa tridactyla breeding success are related to variability in sandeel recruitment. Mar. Ecol. Prog. Ser. 300, 201–211. https://doi.org/10/b4nst3

Gabriel, O., Brandt, A. von (Eds.), 2005. Fish catching methods of the world, 4th ed. ed. Blackwell Pub, Oxford, UK ; Ames, Iowa.

Genovart, M., Arcos, J.M., Álvarez, D., McMinn, M., Meier, R., Wynn, R.B., Guilford, T., Oro, D., 2016. Demography of the critically endangered Balearic shearwater: the impact of fisheries and time to extinction. J. Appl. Ecol. 53, 1158–1168. https://doi.org/10/f8wpz8

Gianuca, D., Phillips, R.A., Townley, S., Votier, S.C., 2017. Global patterns of sex- and age-specific variation in seabird bycatch. Biol. Conserv. 205, 60–76. https://doi.org/10/f9n6mm

Gilman, E., Legorburu, G., Fedoruk, A., Heberer, C., Zimring, M., Barkai, A., 2019. Increasing the functionalities and accuracy of fisheries electronic monitoring systems. Aquat. Conserv. Mar. Freshw. Ecosyst. 29, 901–926. https://doi.org/10/ggk2f8

Glemarec, G., Kindt-Larsen, L., Lundgaard, L.S., Larsen, F., 2020. Assessing seabird bycatch in gillnet fisheries using electronic monitoring. Biol. Conserv. 243, 108461. https://doi.org/10/ggmrwr

Green, R.E., Langston, R.H.W., McCluskie, A., Sutherland, R., Wilson, J.D., 2016. Lack of sound science in assessing wind farm impacts on seabirds. J. Appl. Ecol. 53, 1635–1641. https://doi.org/10/ggnjg3

Hall, M.A., Donovan, G.P., 2001. Environmentalists, Fishermen, Cetaceans and Fish: Is There a Balance and Can Science Help to Find it?, in: Evans, P.G.H., Raga, J.A. (Eds.), Marine Mammals: Biology and Conservation. Springer US, Boston, MA, pp. 491–521. https://doi.org/10.1007/978-1-4615-0529-7_14

Hammond, P. S., Paradinas, I., Smout, S.C., 2019. Development of a Removals Limit Algorithm (RLA) to set limits to anthropogenic mortality of small cetaceans to meet specified conservation objectives, with an example implementation for bycatch of harbour porpoise in the North Sea (No. 628), JNCC Report. JNCC, Peterborough.

Hansen, K.A., Maxwell, A., Siebert, U., Larsen, O.N., Wahlberg, M., 2017. Great cormorants (Phalacrocorax carbo) can detect auditory cues while diving. Sci. Nat. 104, 45. https://doi.org/10/ggkz8s

Hausberger, M., Boigné, A., Lesimple, C., Belin, L., Henry, L., 2018. Wide-eyed glare scares raptors: From laboratory evidence to applied management. PloS One 13. https://doi.org/10/ggk9fr

Hays, G.C., Bailey, H., Bograd, S.J., Bowen, W.D., Campagna, C., Carmichael, R.H., Casale, P., Chiaradia, A., Costa, D.P., Cuevas, E., Nico de Bruyn, P.J., Dias, M.P., Duarte, C.M., Dunn, D.C., Dutton, P.H., Esteban, N., Friedlaender, A., Goetz, K.T., Godley, B.J., Halpin, P.N., Hamann, M., Hammerschlag, N., Harcourt, R., Harrison, A.-L., Hazen, E.L., Heupel, M.R., Hoyt, E., Humphries, N.E., Kot, C.Y., Lea, J.S.E., Marsh, H., Maxwell, S.M., McMahon, C.R., Notarbartolo di Sciara, G., Palacios, D.M., Phillips, R.A., Righton, D., Schofield, G., Seminoff, J.A., Simpfendorfer, C.A., Sims, D.W., Takahashi, A., Tetley, M.J., Thums, M., Trathan, P.N., Villegas-Amtmann, S., Wells, R.S., Whiting, S.D., Wildermann, N.E., Sequeira, A.M.M., 2019. Translating Marine Animal Tracking Data into Conservation Policy and Management. Trends Ecol. Evol. 34, 459–473. https://doi.org/10/gf8s6g

Hays, G.C., Ferreira, L.C., Sequeira, A.M.M., Meekan, M.G., Duarte, C.M., Bailey, H., Bailleul, F., Bowen, W.D., Caley, M.J., Costa, D.P., Eguíluz, V.M., Fossette, S., Friedlaender, A.S., Gales, N., Gleiss, A.C., Gunn, J., Harcourt, R., Hazen, E.L., Heithaus, M.R., Heupel, M., Holland, K., Horning, M., Jonsen, I., Kooyman, G.L., Lowe, C.G., Madsen, P.T., Marsh, H., Phillips, R.A., Righton, D., Ropert-Coudert, Y., Sato, K., Shaffer, S.A., Simpfendorfer, C.A., Sims, D.W., Skomal, G., Takahashi, A., Trathan, P.N., Wikelski, M., Womble, J.N., Thums, M., 2016. Key Questions in Marine Megafauna Movement Ecology. Trends Ecol. Evol. 31, 463–475. https://doi.org/10/f8p3pz

He, P., Pol, M., 2010. Fish Behavior near Gillnets: Capture Processes, and Influencing Factors, in: He, P. (Ed.), Behavior of Marine Fishes. Wiley-Blackwell, Oxford, UK, pp. 183–203. https://doi.org/10.1002/9780813810966.ch8

HELCOM, 2018. State of the Baltic Sea–Second HELCOM holistic assessment 2011-2016, in: Baltic Sea Environment Proceedings. p. 155.

Helmond, A.T.M. van, Mortensen, L.O., Plet-Hansen, K.S., Ulrich, C., Needle, C.L., Oesterwind, D., Kindt-Larsen, L., Catchpole, T., Mangi, S., Zimmermann, C., Olesen, H.J., Bailey, N., Bergsson, H., Dalskov, J., Elson, J., Hosken, M., Peterson, L., McElderry, H., Ruiz, J., Pierre, J.P., Dykstra, C., Poos, J.J., 2020. Electronic monitoring in fisheries: Lessons from global experiences and future opportunities. Fish Fish. 21, 162–189. https://doi.org/10/ggk3qd

Hentati-Sundberg, J., Hjelm, J., Österblom, H., 2014. Does fisheries management incentivize non-compliance? Estimated misreporting in the Swedish Baltic Sea pelagic fishery based on commercial fishing effort. ICES J. Mar. Sci. 71, 1846–1853. https://doi.org/10/f22f4s

Hintzen, N.T., Bastardie, F., Beare, D., Piet, G.J., Ulrich, C., Deporte, N., Egekvist, J., Degel, H., 2012. VMStools: Open-source software for the processing, analysis and visualisation of fisheries logbook and VMS data. Fish. Res. 115–116, 31–43. https://doi.org/10/cbqsjk

Hoare, D., Graham, N., Schön, P.-J., 2011. The Irish Sea data-enhancement project: comparison of self-sampling and national data-collection programmes—results and experiences. ICES J. Mar. Sci. 68, 1778–1784. https://doi.org/10/br39b7

ICES, 2020. Report of the Working Group on Commercial Catches. ICES, 4-8 Nov 2019, Gdańsk , Poland.

ICES, 2019. Joint WGBYC-WGCATCH Workshop on sampling of bycatch and PET species (WKPETSAMP) (No. CM2018/EOSG:35). ICES, 24–26 April 2018, SLU Aqua, Lysekil, Sweden.

ICES, 2018. Report of the Joint OSPAR/HELCOM/ICES Working Group on Marine Birds (JWGBIRD) (No. ICES CM 2017/ACOM:24). ICES, 1–5 October 2018, Ostende, Belgium.

ICES, 2013. Introduction, Overviews and Special Requests, in: Report of the ICES Advisory Committee 2013, ICES Advice. ICES, Copenhagen, p. 348.

Inglis, I., 1980. Visual bird scarers: an ethological approach. Bird Probl. Agric. 121–143.

Institute of Marine Research, 2020. Referanseflåten [WWW Document]. Havforskningsinstituttet. URL https://www.hi.no/hi/tokt/referanseflaten-1 (accessed 2.18.20).

International Whaling Commission, 2020a. The Revised Management Procedure (RMP) for Baleen Whales [WWW Document]. URL https://iwc.int/rmpbw (accessed 3.8.20).

International Whaling Commission, 2020b. Revised Management Procedure (RMP) [WWW Document]. URL https://iwc.int/rmp (accessed 3.8.20).

Johansen, S., Larsen, O.N., Christensen-Dalsgaard, J., Seidelin, L., Huulvej, T., Jensen, K., Lunneryd, S.-G., Boström, M., Wahlberg, M., 2016. In-air and underwater hearing in the great cormorant (Phalacrocorax carbo sinensis), in: The Effects of Noise on Aquatic Life II. Springer, pp. 505–512.

Katzir, G., Howland, H.C., 2003. Corneal power and underwater accommodation in great cormorants (Phalacrocorax carbo sinensis). J. Exp. Biol. 206, 833–841. https://doi.org/10/dxh2sq

Kelleher, K., Nations, F. and A.O. of the U., 2005. Discards in the World's Marine Fisheries: An Update. Food & Agriculture Org.

Kindt-Larsen, L., Berg, C.W., Tougaard, J., Sørensen, T.K., Geitner, K., Northridge, S., Sveegaard, S., Larsen, F., 2016. Identification of high-risk areas for harbour porpoise Phocoena phocoena bycatch using remote electronic monitoring and satellite telemetry data. Mar. Ecol. Prog. Ser. 555, 261–271. https://doi.org/10/f849c3

Kindt-Larsen, L., Dalskov, J., Stage, B., Larsen, F., 2012a. Observing incidental harbour porpoise Phocoena phocoena bycatch by remote electronic monitoring. Endanger. Species Res. 19, 75–83. https://doi.org/10/ggk2cp

Kindt-Larsen, L., Kirkegaard, E., Dalskov, J., 2011. Fully documented fishery: a tool to support a catch quota management system. ICES J. Mar. Sci. 68, 1606–1610. https://doi.org/10/bm2pkd

Kindt-Larsen, L., Larsen, F., Stage, B., Dalskov, J., 2012b. Fully Documented Fishery onboard gillnet vessels> 15 m.

Komoroske, L.M., Lewison, R.L., 2015. Addressing fisheries bycatch in a changing world. Front. Mar. Sci. 2. https://doi.org/10/gftmrd

Konrad, C., Sala, A., Doerner, H., European Commission, Joint Research Centre, 2019. Review of the implementation of the EU regulation on the incidental catches of cetaceans (STECF-19-07).

Koschinski, S., Strempel, R., 2012. Strategies for the prevention of bycatch of seabirds and marine mammals in Baltic Sea fisheries. Rep. Prep. Ger. NGOs NABU GSM GRD Based Inf. Compil. Regard Ger. Fish. 69.

Kumar, K.V.A., Pravin, P., Meenakumari, B., 2016. Bait, Bait Loss, and Depredation in Pelagic Longline Fisheries–A Review. Rev. Fish. Sci. Aquac. 24, 295–304. https://doi.org/10/ggk25g

Larsen, F., Eigaard, O.R., Tougaard, J., 2007. Reduction of harbour porpoise (Phocoena phocoena) bycatch by iron-oxide gillnets. Fish. Res. 85, 270–278. https://doi.org/10/chrzrf

Larsen, O.N., Wahlberg, M., Christensen-Dalsgaard, J., 2020. Amphibious hearing in a diving bird, the great cormorant (Phalacrocorax carbo sinensis). J. Exp. Biol. https://doi.org/10/ggngkp

Le Bot, T., Lescroël, A., Grémillet, D., 2018. A toolkit to study seabird–fishery interactions. ICES J. Mar. Sci. 75, 1513–1525. https://doi.org/10/gfjj45

Lee, J., South, A.B., Jennings, S., 2010. Developing reliable, repeatable, and accessible methods to provide high-resolution estimates of fishing-effort distributions from vessel monitoring system (VMS) data. ICES J. Mar. Sci. 67, 1260–1271. https://doi.org/10/cmmqqh

Lewis, S., Elston, D.A., Daunt, F., Cheney, B., Thompson, P.M., 2009. Effects of extrinsic and intrinsic factors on breeding success in a long lived seabird. Oikos 118, 521–528. https://doi.org/10/bgghbw

Lewison, R., Crowder, L., Read, A., Freeman, S., 2004. Understanding impacts of fisheries bycatch on marine megafauna. Trends Ecol. Evol. 19, 598–604. https://doi.org/10/d6tf8c

Lewison, R.L., 2013. Finding the missing pieces: working to solve the fisheries bycatch puzzle. Anim. Conserv. 16, 153–154. https://doi.org/10/ggk84b

Lewison, R.L., Crowder, L.B., Wallace, B.P., Moore, J.E., Cox, T., Zydelis, R., McDonald, S., DiMatteo, A., Dunn, D.C., Kot, C.Y., Bjorkland, R., Kelez, S., Soykan, C., Stewart, K.R., Sims, M., Boustany, A., Read, A.J., Halpin, P., Nichols, W.J., Safina, C., 2014. Global patterns of marine mammal, seabird, and sea turtle bycatch reveal taxa-specific and cumulative megafauna hotspots. Proc. Natl. Acad. Sci. 111, 5271–5276. https://doi.org/10.1073/pnas.1318960111

Little, A.S., Needle, C.L., Hilborn, R., Holland, D.S., Marshall, C.T., 2015. Real-time spatial management approaches to reduce bycatch and discards: experiences from Europe and the United States. Fish Fish. 16, 576–602. https://doi.org/10/ggk3pb

Long, R.D., Charles, A., Stephenson, R.L., 2015. Key principles of marine ecosystem-based management. Mar. Policy 57, 53–60. https://doi.org/10/f7g95s

Mangel, J.C., Wang, J., Alfaro-Shigueto, J., Pingo, S., Jimenez, A., Carvalho, F., Swimmer, Y., Godley, B.J., 2018. Illuminating gillnets to save seabirds and the potential for multi-taxa bycatch mitigation. R. Soc. Open Sci. 5, 180254.

Mangi, S.C., Dolder, P.J., Catchpole, T.L., Rodmell, D., de Rozarieux, N., 2015. Approaches to fully documented fisheries: practical issues and stakeholder perceptions. Fish Fish. 16, 426–452. https://doi.org/10/ggkz8b

Marchowski, D., Jankowiak, Ł., Ławicki, Ł., Wysocki, D., Chylarecki, P., 2020. Fishery bycatch is among the most important threats to the European population of Greater Scaup *Aythya marila*. Bird Conserv. Int. 1–18. https://doi.org/10/ggkz8p

Martin, G., 2017. The sensory ecology of birds, First edition. ed, Oxford avian biology series. Oxford University Press, Oxford.

Martin, G.R., Crawford, R., 2015. Reducing bycatch in gillnets: A sensory ecology perspective. Glob. Ecol. Conserv. 3, 28–50. https://doi.org/10.1016/j.gecco.2014.11.004

Martin, G.R., Jarrett, N., Williams, M., 2007. Visual fields in Blue Ducks Hymenolaimus malacorhynchos and Pink-eared Ducks Malacorhynchus membranaceus: visual and tactile foraging. Ibis 149, 112–120. https://doi.org/10/bf4s3v

Martin, G.R., White, C.R., Butler, P.J., 2008. Vision and the foraging technique of Great Cormorants Phalacrocorax carbo: pursuit or close-quarter foraging? Ibis 150, 485–494.

Martin, G.R., Young, S.R., Land, M.F., 1984. The eye of the humboldt penguin, Spheniscus humboldti: visual fields and schematic optics. Proc. R. Soc. Lond. B Biol. Sci. 223, 197–222. https://doi.org/10/dszb45

Maxwell, A., Hansen, K.A., Larsen, O.N., Christensen-Dalsgaard, J., Wahlberg, M., Siebert, U., 2016. Testing auditory sensitivity in the great cormorant (Phalacrocorax carbo sinensis): Psychophysics vs. Auditory brainstem response. Presented at the Fourth International Conference on the Effects of Noise on Aquatic Life, Dublin, Ireland, p. 050001. https://doi.org/10/ggkz83

Maxwell, A., Hansen, K.A., Ortiz, S.T., Larsen, O.N., Siebert, U., Wahlberg, M., 2017. In-air hearing of the great cormorant (*Phalacrocorax carbo*). Biol. Open 6, 496–502. https://doi.org/10/ggkz8r

McCluskey, S.M., Lewison, R.L., 2008. Quantifying fishing effort: a synthesis of current methods and their applications. Fish Fish. 9, 188–200. https://doi.org/10/cqx936

McElderry, H., McCullough, D., Schrader, J., Illingworth, J., 2007. Pilot study to test the effectiveness of electronic monitoring in Canterbury fisheries. DOC Res. Dev. Ser. 264.

Melvin, E.F., Guy, T.J., Read, L.B., 2014. Best practice seabird bycatch mitigation for pelagic longline fisheries targeting tuna and related species. Fish. Res. 149, 5–18. https://doi.org/10/ggkz8d

Melvin, E.F., Parrish, J.K., Conquest, L.L., 1999. Novel Tools to Reduce Seabird Bycatch in Coastal Gillnet Fisheries. Conserv. Biol. 13, 1386–1397. https://doi.org/10/d26x7j

Mendel, B., Sonntag, N., Wahl, J., Schwemmer, P., Dries, H., Guse, N., Müller, S., Garthe, S., 2008. Profiles of seabirds and waterbirds of the German North and Baltic Seas, Naturschutz und biologische Vielfalt. Bundesamt für Naturschutz, Bonn-Bad Godesberg.

Michelin, M., Elliott, M., Bucher, M., Zimring, M., Sweeney, M., 2018. Catalyzing the Growth of Electronic Monitoring in Fisheries, Building Greater Transparency and Accountability at Sea, Opportunities, Barriers, and Recommendations for Scaling the Technology. The Nature Conservancy & California Environmental Associates.

Miller, J.A.O., Furness, R.W., Trinder, M., Matthiopoulos, J., 2019. The sensitivity of seabird populations to density-dependence, environmental stochasticity and anthropogenic mortality. J. Appl. Ecol. 56, 2118–2130. https://doi.org/10.1111/1365-2664.13448

Mills, C.M., Townsend, S.E., Jennings, S., Eastwood, P.D., Houghton, C.A., 2007. Estimating high resolution trawl fishing effort from satellite-based vessel monitoring system data. ICES J. Mar. Sci. 64, 248–255. https://doi.org/10/bq5fjz

Moore, J.E., Wallace, B.P., Lewison, R.L., Žydelis, R., Cox, T.M., Crowder, L.B., 2009. A review of marine mammal, sea turtle and seabird bycatch in USA fisheries and the role of policy in shaping management. Mar. Policy 33, 435–451. https://doi.org/10/fszqdc

Morandeau, G., Macher, C., Sanchez, F., Bru, N., Fauconnet, L., Caill-Milly, N., 2014. Why do fishermen discard? Distribution and quantification of the causes of discards in the Southern Bay of Biscay passive gear fisheries. Mar. Policy 48, 30–38. https://doi.org/10/ggkz86

Muench, A., DePiper, G.S., Demarest, C., 2018. On the precision of predicting fishing location using data from the vessel monitoring system (VMS). Can. J. Fish. Aquat. Sci. 75, 1036–1047. https://doi.org/10/ggkz8v

Natale, F., Gibin, M., Alessandrini, A., Vespe, M., Paulrud, A., 2015. Mapping fishing effort through AIS data. PloS One 10. https://doi.org/10/f7sfgs

Needle, C.L., Dinsdale, R., Buch, T.B., Catarino, R.M., Drewery, J., Butler, N., 2015. Scottish science applications of remote electronic monitoring. ICES J. Mar. Sci. 72, 1214–1229. https://doi.org/10/f7f2pb

Nevitt, G.A., 2000. Olfactory foraging by Antarctic procellariiform seabirds: life at high Reynolds numbers. Biol. Bull. 198, 245–253. https://doi.org/10/b59t2z

Nevitt, G.A., Losekoot, M., Weimerskirch, H., 2008. Evidence for olfactory search in wandering albatross, Diomedea exulans. Proc. Natl. Acad. Sci. 105, 4576–4581. https://doi.org/10/d5dh4p

Niel, C., Lebreton, J., 2005. Using demographic invariants to detect overharvested bird populations from incomplete data. Conserv. Biol. 19, 826–835. https://doi.org/10/dmqsfc

Northridge, S., 2009. Bycatch, in: Perrin, W.F., Würsig, B., Thewissen, J.G.M. (Eds.), Encyclopedia of Marine Mammals (Second Edition). Academic Press, London, pp. 167–169. https://doi.org/10.1016/B978-0-12-373553-9.00044-4

Northridge, S., Coram, A., Kingston, A., Crawford, R., 2017. Disentangling the causes of protected-species bycatch in gillnet fisheries: Causes of Bycatch. Conserv. Biol. 31, 686–695. https://doi.org/10/ggkz8k

O'Brien, S.H., Cook, A.S.C.P., Robinson, R.A., 2017. Implicit assumptions underlying simple harvest models of marine bird populations can mislead environmental management decisions. J. Environ. Manage. 201, 163–171. https://doi.org/10/gbs2tr

O'Farrell, S., Sanchirico, J.N., Chollett, I., Cockrell, M., Murawski, S.A., Watson, J.T., Haynie, A., Strelcheck, A., Perruso, L., 2017. Improving detection of short-duration fishing behaviour in vessel tracks by feature engineering of training data. ICES J. Mar. Sci. 74, 1428–1436. https://doi.org/10/ggm2k6

O'Keefe, C.E., Cadrin, S.X., Stokesbury, K.D.E., 2014. Evaluating effectiveness of time/area closures, quotas/caps, and fleet communications to reduce fisheries bycatch. ICES J. Mar. Sci. 71, 1286–1297. https://doi.org/10/ggkz8m

Oppel, S., Bolton, M., Carneiro, A.P.B., Dias, M.P., Green, J.A., Masello, J.F., Phillips, R.A., Owen, E., Quillfeldt, P., Beard, A., Bertrand, S., Blackburn, J., Boersma, P.D., Borges, A., Broderick, A.C., Catry, P., Cleasby, I., Clingham, E., Creuwels, J., Crofts, S., Cuthbert, R.J., Dallmeijer, H., Davies, D., Davies, R., Dilley, B.J., Dinis, H.A., Dossa, J., Dunn, M.J., Efe, M.A., Fayet, A.L., Figueiredo, L., Frederico, A.P., Gjerdrum, C., Godley, B.J., Granadeiro, J.P., Guilford, T., Hamer, K.C., Hazin, C., Hedd, A., Henry, L., Hernández-Montero, M., Hinke, J., Kokubun, N., Leat, E., Tranquilla, L.M., Metzger, B., Militão, T., Montrond, G., Mullié, W., Padget, O., Pearmain, E.J., Pollet, I.L., Pütz, K., Quintana, F., Ratcliffe, N., Ronconi, R.A., Ryan, P.G., Saldanha, S., Shoji, A., Sim, J., Small, C., Soanes, L., Takahashi, A., Trathan, P., Trivelpiece, W., Veen, J., Wakefield, E., Weber, N., Weber, S., Zango, L., Daunt, F., Ito, M., Harris, M.P., Newell, M.A., Wanless, S., González-Solís, J., Croxall, J., 2018. Spatial scales of marine conservation management for breeding seabirds. Mar. Policy 98, 37–46. https://doi.org/10/ggkz82

Ortiz, N., Mangel, J., Wang, J., Alfaro-Shigueto, J., Pingo, S., Jimenez, A., Suarez, T., Swimmer, Y., Carvalho, F., Godley, B., 2016. Reducing green turtle bycatch in small-scale fisheries using illuminated gillnets: the cost of saving a sea turtle. Mar. Ecol. Prog. Ser. 545, 251–259. https://doi.org/10.3354/meps11610

Owen, D., Chambers, F., 2004. Interaction between the EU Common Fisheries Policy and the Habitats and Birds directives: briefing. Institute for European Environmental Policy, London.

Padget, O., Dell'Ariccia, G., Gagliardo, A., González-Solís, J., Guilford, T., 2017. Anosmia impairs homing orientation but not foraging behaviour in free-ranging shearwaters. Sci. Rep. 7, 1–12. https://doi.org/10/gbvp83

Palialexis, A., Connor, D., Gonzalvo, J., Damalas, D., Mitchel, I., Micu, D., Korpinen, S., Somma, F., Rees, A.F., European Commission, Joint Research Centre, 2019. Indicators for status assessment of species, relevant to MSFD biodiversity descriptor.

Pardo, D., Forcada, J., Wood, A.G., Tuck, G.N., Ireland, L., Pradel, R., Croxall, J.P., Phillips, R.A., 2017. Additive effects of climate and fisheries drive ongoing declines in multiple albatross species. Proc. Natl. Acad. Sci. 114, E10829–E10837. https://doi.org/10/gcrfd4

Petersen, S.L., Honig, M.B., Ryan, P.G., Underhill, L.G., 2009. Seabird bycatch in the pelagic longline fishery off southern Africa. Afr. J. Mar. Sci. 31, 191–204. https://doi.org/10/bgd655

Piasente, M., Stanley, B., Timmiss, T., McElderry, H., Pria, M., Dyas, M., 2012. Electronic onboard monitoring pilot project for the Eastern Tuna and Billfish Fishery. Aust. Fish. Manag. Auth. Canberra.

Piatt, J.F., Nettleship, D.N., 1987. Incidental catch of marine birds and mammals in fishing nets off Newfoundland, Canada. Mar. Pollut. Bull. 18, 344–349. https://doi.org/10/fd7bpf

Pikitch, E.K., 2004. ECOLOGY: Ecosystem-Based Fishery Management. Science 305, 346–347. https://doi.org/10/d9c5mj

Plet-Hansen, K.S., Bergsson, H., Ulrich, C., 2019. More data for the money: Improvements in design and cost efficiency of electronic monitoring in the Danish cod catch quota management trial. Fish. Res. 215, 114–122. https://doi.org/10/ggk2d4

Plet-Hansen, K.S., Eliasen, S.Q., Mortensen, L.O., Bergsson, H., Olesen, H.J., Ulrich, C., 2017. Remote electronic monitoring and the landing obligation–some insights into fishers' and fishery inspectors' opinions. Mar. Policy 76, 98–106. https://doi.org/10/f9qvj8

Plet-Hansen, K.S., Ulrich, C., Olesen, H.J., Mortensen, L.O., Bergsson, H., 2015. Fully documented fisheries-is remote electronic monitoring the future tool in fisheries control?, in: ICES Annual Science Conference 2015.

Pott, C., Wiedenfeld, D.A., 2017. Information gaps limit our understanding of seabird bycatch in global fisheries. Biol. Conserv. 210, 192–204. https://doi.org/10/gbqqht

Punt, A.E., Donovan, G.P., 2007. Developing management procedures that are robust to uncertainty: lessons from the International Whaling Commission. ICES J. Mar. Sci. 64, 603–612. https://doi.org/10/dz78xg

R Core Team, 2020. R: A language and environment for statistical computing.

Regular, P., Montevecchi, W., Hedd, A., Robertson, G., Wilhelm, S., 2013. Canadian fishery closures provide a large-scale test of the impact of gillnet bycatch on seabird populations. Biol. Lett. 9, 20130088. https://doi.org/10/ggkz8g

Regular, P.M., Hedd, A., Montevecchi, W.A., 2011. Fishing in the Dark: A Pursuit-Diving Seabird Modifies Foraging Behaviour in Response to Nocturnal Light Levels. PLoS ONE 6, e26763. https://doi.org/10.1371/journal.pone.0026763

Richard, Y., Abraham, E., Berkenbusch, K., 2020. Assessment of the risk of commercial fisheriesto New Zealand seabirds, 2006–07 to 2016–17 (No. 237), New Zealand Aquatic Environment and Biodiversity Report. Fisheries New Zealand.

Richard, Y., Abraham, E.R., 2013. Application for Potential Biological Removal Methods to Seabird Populations. Ministry for Primary Industries.

Roman, S., Jacobson, N., Cadrin, S.X., 2011. Assessing the Reliability of Fisher Self-Sampling Programs. North Am. J. Fish. Manag. 31, 165–175. https://doi.org/10/fqsg95

Rosman, I., Maugeri, S., 1980. Fishing with bottom gillnets, FAO training series. Food and Agriculture Organization of the United Nations ; UNIPUB [distributor], Rome : New York.

Russo, T., D'Andrea, L., Parisi, A., Cataudella, S., 2014. VMSbase: An R-Package for VMS and Logbook Data Management and Analysis in Fisheries Ecology. PLoS ONE 9. https://doi.org/10/f6cwb6

Russo, T., D'andrea, L., Parisi, A., Martinelli, M., Belardinelli, A., Boccoli, F., Cignini, I., Tordoni, M., Cataudella, S., 2016. Assessing the fishing footprint using data integrated from different tracking devices: Issues and opportunities. Ecol. Indic. 69, 818–827. https://doi.org/10/f9dfms

Savina, E., 2018. Gear technical contributions to an ecosystem approach in the Danish bottom set nets fisheries (PhD Thesis). DTU Aqua, Hirtshals.

Savina, E., Krag, L.A., Frandsen, R.P., Madsen, N., 2017. Effect of fisher's soak tactic on catch pattern in the Danish gillnet plaice fishery. Fish. Res. 196, 56–65. https://doi.org/10/ggk7cg

Seafish.org, 2018. Trammel Nets - Gears Database [WWW Document]. Seafish.org. URL https://www.seafish.org/geardb/gear/trammel-nets/index.html (accessed 2.20.20).

Sissenwine, M., Symes, D., 2007. Reflections on the common fisheries policy. Rep. Gen. Dir. Fish. Marit. Aff. Eur. Comm. 75.

Sivak, J., Millodot, M., 1977. Optical performance of the penguin eye in air and water. J. Comp. Physiol. 119, 241–247. https://doi.org/10/fjmd3b

Sonntag, N., Schwemmer, H., Fock, H.O., Bellebaum, J., Garthe, S., 2012. Seabirds, set-nets, and conservation management: assessment of conflict potential and vulnerability of birds to bycatch in gillnets. ICES J. Mar. Sci. 69, 578–589. https://doi.org/10/ggkz8z

Soriano-Redondo, A., Cortés, V., Reyes-González, J.M., Guallar, S., Bécares, J., Rodríguez, B., Arcos, J.M., González-Solís, J., 2016. Relative abundance and distribution of fisheries influence risk of seabird bycatch. Sci. Rep. 6, 1–8. https://doi.org/10/f9dmm9

Soykan, C.U., Moore, J.E., Zydelis, R., Crowder, L.B., Safina, C., Lewison, R.L., 2008. Why study bycatch? An introduction to the Theme Section on fisheries bycatch. Endanger. Species Res. 5, 91–102. https://doi.org/10/dc4drn

Stevens, M., Hopkins, E., Hinde, W., Adcock, A., Connolly, Y., Troscianko, T., Cuthill, I.C., 2007. Field experiments on the effectiveness of 'eyespots' as predator deterrents. Anim. Behav. 74, 1215–1227. https://doi.org/10.1016/j.anbehav.2007.01.031

Sullivan, B., Kibel, B., Kibel, P., Yates, O., Potts, J., Ingham, B., Domingo, A., Gianuca, D., Jiménez, S., Lebepe, B., 2018. At-sea trialling of the Hookpod: a 'one-stop'mitigation solution for seabird bycatch in pelagic longline fisheries. Anim. Conserv. 21, 159–167. https://doi.org/10/gdfh7f

Suuronen, P., Jounela, P., Tschernij, V., 2010. Fishermen responses on marine protected areas in the Baltic cod fishery. Mar. Policy 34, 237–243. https://doi.org/10/ckv2w9

Tasker, M., 2000. The impacts of fishing on marine birds. ICES J. Mar. Sci. 57, 531–547. https://doi.org/10/cmz9nw

Taylor, B.L., Rojas-Bracho, L., Moore, J., Jaramillo-Legorreta, A., Hoef, J.M.V., Cardenas-Hinojosa, G., Nieto-Garcia, E., Barlow, J., Gerrodette, T., Tregenza, N., Thomas, L., Hammond, P.S., 2017. Extinction is Imminent for Mexico's Endemic Porpoise Unless Fishery Bycatch is Eliminated. Conserv. Lett. 10, 588–595. https://doi.org/10/ggk25x

Tjørnløv, R.S., Pradel, R., Choquet, R., Christensen, T.K., Frederiksen, M., 2019. Consequences of past and present harvest management in a declining flyway population of common eiders Somateria mollissima. Ecol. Evol. 9, 12515–12530. https://doi.org/10/ggn4cb

Torres, L.G., Sagar, P.M., Thompson, D.R., Phillips, R.A., 2013. Scaling down the analysis of seabird-fishery interactions. Mar. Ecol. Prog. Ser. 473, 275–289. https://doi.org/10/f4mkr7

Trippel, E.A., Holy, N.L., Palka, D.L., Shepherd, T.D., Melvin, G.D., Terhune, J.M., 2003. Nylon Barium Sulphate Gillnet Reduces Porpoise and Seabird Mortality. Mar. Mammal Sci. 19, 240–243. https://doi.org/10/dt27kz

Tull, C.E., Germain, P., May, A.W., 1972. Mortality of Thick-billed Murres in the West Greenland Salmon Fishery. Nature 237, 42–44. https://doi.org/10/dpvbs7

Turvey, S.T., Pitman, R.L., Taylor, B.L., Barlow, J., Akamatsu, T., Barrett, L.A., Zhao, X., Reeves, R.R., Stewart, B.S., Wang, K., Wei, Z., Zhang, X., Pusser, L. t, Richlen, M., Brandon, J.R., Wang, D., 2007. First human-caused extinction of a cetacean species? Biol. Lett. 3, 537–540. https://doi.org/10/b3s7xv

Ulrich, C., Olesen, H.J., Bergsson, H., Egekvist, J., H\aakansson, K.B., Dalskov, J., Kindt-Larsen, L., Storr-Paulsen, M., 2015. Discarding of cod in the Danish Fully Documented Fisheries trials. ICES J. Mar. Sci. 72, 1848–1860. https://doi.org/10/f7nkjg

Vetemaa, M., Ložys, L., 2009. Action D1 – Use of by-catch safe fishing gear in pilot project areas (LIFE Nature project No. LIFE 05 NAT/LV/000100), Marine Protected Areas in the Eastern Baltic Sea.

Wade, P.R., 1998. Calculating Limits to the Allowable Human-Caused Mortality of Cetaceans and Pinnipeds. Mar. Mammal Sci. 14, 1–37. https://doi.org/10/dxjhws

Wang, J., Barkan, J., Fisler, S., Godinez-Reyes, C., Swimmer, Y., 2013. Developing ultraviolet illumination of gillnets as a method to reduce sea turtle bycatch. Biol. Lett. 9, 20130383. https://doi.org/10/gb9fv7

Wang, J., Fisler, S., Swimmer, Y., 2010. Developing visual deterrents to reduce sea turtle bycatch in gill net fisheries. Mar. Ecol. Prog. Ser. 408, 241–250. https://doi.org/10.3354/meps08577

Washington Department of Fish and Wildlife, 2019. Puget Sound Commercial Salmon Regulations.

White, C.R., Butler, P.J., Grémillet, D., Martin, G.R., 2008. Behavioural strategies of cormorants (Phalacrocoracidae) foraging under challenging light conditions. Ibis 150, 231–239. https://doi.org/10/bvhcwb

White, C.R., Day, N., Butler, P.J., Martin, G.R., 2007. Vision and foraging in cormorants: more like herons than hawks? PLoS One 2.

Yeh, Y.-M., Huang, H.-W., Dietrich, K.S., Melvin, E., 2013. Estimates of seabird incidental catch by pelagic longline fisheries in the South Atlantic Ocean. Anim. Conserv. 16, 141–152. https://doi.org/10/f4wwx6

Zuur, A., Ieno, E.N., Smith, G.M., 2007. Analyzing ecological data. Springer.

Zuur, A., Ieno, E.N., Walker, N., Saveliev, A.A., Smith, G.M., 2009. Mixed effects models and extensions in ecology with R. Springer Science & Business Media.

Žydelis, R., 2013. The incidental catch of seabirds in gillnet fisheries: A global review. Biol. Conserv.

Žydelis, R., Bellebaum, J., Österblom, H., Vetemaa, M., Schirmeister, B., Stipniece, A., Dagys, M., van Eerden, M., Garthe, S., 2009. Bycatch in gillnet fisheries – An overlooked threat to waterbird populations. Biol. Conserv. 142, 1269–1281. https://doi.org/10/fwdfxp

Žydelis, R., Ruškytė, D., 2005. Winter foraging of long-tailed ducks (Clangula hyemalis) exploiting different benthic communities in the Baltic Sea. Wilson J. Ornithol. 117, 133–141.

PAPER 1

ASSESSING SEABIRD BYCATCH IN GILLNET FISHERIES USING ELECTRONIC MONITORING

Glemarec, G., Kindt-Larsen, L., Lundgaard, L. S., & Larsen, F.

Biological Conservation, 2020, **243**, 108461.

Abstract

The unintentional capture (bycatch) of seabirds in gillnet fisheries kills hundreds of thousands of individuals annually and is thought to threaten the conservation of entire populations. However, data from commercial fisheries is often lacking to confirm these suspicions. In Denmark, sparse or incomplete catch data from small-scale gillnetters prevent managers from gaining a comprehensive overview of the importance of seabird bycatch in coastal waters. In this study, electronic monitoring (EM) with video is used to identify and quantify seabird bycatch in a Danish coastal gillnet fishery. Three gillnetters were monitored over a period of 9 years, resulting in 2118 fishing trips and 10,964 hauls; 700 birds from six families were identified. Three species composed >90% of the incidental captures, the common eider (*Somateria mollissima*), the great cormorant (*Phalacrocorax carbo*) and the common guillemot (*Uria aalge*), respectively qualifying regionally as endangered, least concerned and near threatened. There was a clear species-specific spatial and seasonal variability in bycatch per unit effort (BPUE) estimates, highlighting areas of high risk of seabird bycatch. Approximately 40% of all bycatch events were observed in 0.2% of the hauls, suggesting that the full fishing activity should be analysed to obtain accurate seabird bycatch estimates.

Assessing seabird bycatch in gillnet fisheries using electronic monitoring

Gildas Glemarec, Lotte Kindt-Larsen, Louise Scherffenberg Lundgaard, Finn Larsen

Abstract

The unintentional capture (bycatch) of seabirds in gillnet fisheries kills hundreds of thousands of individuals annually and is thought to threaten the conservation of entire populations. However, data from commercial fisheries is often lacking to confirm these suspicions. In Denmark, sparse or incomplete catch data from small-scale gillnetters prevent managers from gaining a comprehensive overview of the importance of seabird bycatch in coastal waters. In this study, electronic monitoring (EM) with video is used to identify and quantify seabird bycatch in a Danish coastal gillnet fishery. Three gillnetters were monitored over a period of 9 years, resulting in 2118 fishing trips and 10,964 hauls; 700 birds from six families were identified. Three species composed >90% of the incidental captures, the common eider (*Somateria mollissima*), the great cormorant (*Phalacrocorax carbo*) and the common guillemot (*Uria aalge*), respectively qualifying regionally as endangered, least concerned and near threatened. There was a clear species-specific spatial and seasonal variability in bycatch per unit effort (BPUE) estimates, highlighting areas of high risk of seabird bycatch.
Approximately 40% of all bycatch events were observed in 0.2% of the hauls, suggesting that the full fishing activity should be analysed to obtain accurate seabird bycatch estimates.

Keywords: Electronic monitoring, Seabirds, Bycatch, Gillnets, Fisheries interaction

Running head: Electronic monitoring of seabird bycatch

1. Introduction

Unintentional captures in fishing gears (or bycatch) are a major cause of mortality for air-breathing marine animals like seabirds, sea turtles, or cetaceans (Tasker et al., 2000; Lewison et al., 2014; Northridge et al., 2017). In particular, entanglement in gillnets is responsible for the drowning of hundreds of thousands of seabirds each year (Žydelis et al., 2013), and has been identified as a major threat for some vulnerable populations (Žydelis et al., 2009; Croxall et al., 2012; Dias et al., 2019). In the European Union, despite the commitments of the Member States to protect and conserve avifauna (EU, 2009), and although a strong legislative body is in place to guarantee no or minimal bycatch of sensitive species (through the Common Fisheries Policy and the Marine Strategy Framework Directive notably), seabird bycatch remains an unresolved problem. In 2012, the European Commission established an “Action Plan for reducing incidental catches of seabirds in fishing gears” (EC, 2012), calling upon Member States to estimate the impact of their national fisheries on seabirds, and to come up with effective methods to reduce or suppress incidental catches. However, in most European countries, bycatch data collection relies upon non-dedicated programmes conducted under the Data Collection Framework (DCF). Only few dedicated bycatch sampling programmes exist, usually limited in time and space (ICES, 2018), with the noteworthy exception of the long-term protected species bycatch monitoring programme (PSBMP) in the United Kingdom.

Gillnets, a common type of fishing gear consisting of vertical walls of nettings invisible to fish, are considered the most deadly gear for seabirds (Dias et al., 2019). Bycatch of seabirds in gillnet fisheries alone is estimated to kill ca. 400,000 birds globally each year, with at least 76,000 in the Baltic Sea (Žydelis et al., 2013). Detailed registrations of incidental catches over extended periods are crucial to understand what influences seabird bycatch in time and space, and ultimately how to remedy it (Northridge et al., 2017). Accordingly, recording bycatch of protected species is now a requirement for all EU fisheries (EC, 2016). Yet, reliable data is frequently limited for small-scale and artisanal fisheries (Pott and Wiedenfeld, 2017; ICES, 2018). Worldwide, long-term data series on bycatch remain the exception, and in most regions incidental catches of birds are only sporadically monitored, e.g. through independent on-board observer programmes (Le Bot et al., 2018).

Gillnet fishing fleets often consist of numerous small-scale vessels. In high-waged countries with like Denmark, a regular on-board observer monitoring scheme can rapidly become prohibitively expensive (Kindt-Larsen et al., 2011). Danish commercial gillnetters are mostly vessels below 15 m in length, and the national on-board observer programme supervised by the National Institute of Aquatic Resources (DTU Aqua) covers only about 0.1% of the whole fleet (Anonymous 2019). In these conditions, catches of seabirds, rare by nature, are likely to remain undetected. In Denmark, bird bycatches must be reported in fishing logbooks (EC, 2016). However, in small-scale fisheries where no constraining enforcement or verification protocol is in place, self-reported bycatch raises concerns of reliability (Mangi et al., 2015). Indirect observations, e.g. carcass collections and interviews with fishers, can complete the overall picture locally, but also commonly lead to underreporting (see e.g. Bellebaum et al., 2013). Therefore, the characteristics and the magnitude of seabird bycatch in the Danish gillnet fishery are essentially unknown.

Deploying electronic monitoring (EM) systems on fishing vessels offers an alternative to on-board observers, while reducing overall costs (Mangi et al., 2015; Plet-Hansen et al., 2019). EM systems consist of a set of closed-circuit television (CCTV) cameras, gear and position sensors (GPS), and a dedicated computer permanently installed on-board a fishing vessel. The fishing activity is recorded and stored, either locally on a hard drive, or on a dedicated storage server. These data are then readily accessible for researchers to analyse the characteristics of the fishing activity, including distribution of the fishing effort and catch composition. DTU Aqua first started using EM systems in 2008 on commercial trawlers, seiners and gillnetters around Denmark, originally as a means to evaluate discards in these fisheries (Dalskov and Kindt-Larsen, 2009). Rapidly, EM became a tool to assess the effects of the implementation of the European catch quota management system (Kindt-Larsen et al., 2011; 2012a; Plet-Hansen et al., 2019), and later still, to study incidental catches of marine mammals in the Danish gillnet fisheries (Kindt-Larsen et al., 2012b; 2016). Unlike on-board observers, EM systems are able to follow a fishing vessel all-year long and can potentially record every catch provided that no technical failure occurs and that the system is not tampered with. Using EM, discreet and rare fishing events, such as seabird bycatch, can thus be registered and accounted for.

In the present study, three commercial gillnet vessels operating on the East coast of Denmark were equipped with an electronic monitoring system with CCTV. We recorded the entire fishing activity, including the bycatch of seabirds. Using data spanning from 2010 to 2018, we examined the fine-scale spatio-temporal variations of the fishing activity, and we estimated the variations of seabird bycatch rates in the study area. Based on these observations, we discuss the ability of EM technologies with CCTV to provide precise information on incidental catches of seabirds in a small-scale gillnet fishery.

Such data are valuable both in a fisheries management and in a conservation context, as this issue is often largely ignored in small-scale fisheries. Specifically, we show that EM technologies with CCTV can be used to record incidental catches of seabirds accurately in a commercial gillnet fishery. Then, we describe the seasonal variations of bycatch rates per species in the study area. Finally, we identify the benefits and weaknesses of using EM for collecting seabird bycatch data.

2. Material and methods

2.1. Study area and sampled fishing vessels

The data collection was conducted using EM systems on-board three anonymised Danish commercial gillnetters. The periods during which EM was active differed between vessels: vessel 1 was sampled from May 2010 to June 2016, vessel 2 from March 2016 to December 2018, and vessel 3 from May 2010 to April 2014 and from March 2016 to December 2018. The vessels operated in the Sound (Figure 1), an important fishing ground for small-scale gillnetters below 15 m, situated on the Eastern coast of Denmark. Skippers went out for daily coastal trips of a few hours, targeting mostly cod (*Gadus morhua)* and European plaice (*Pleuronectes platessa)* year long, with seasonal shift to lumpsucker (*Cyclopterus lumpus)* between end of January and end of April. Other valuable species included Atlantic salmon (*Salmo salar)*, turbot (*Scophthalmus maximus)* and Atlantic mackerel (*Scomber scombrus)*.

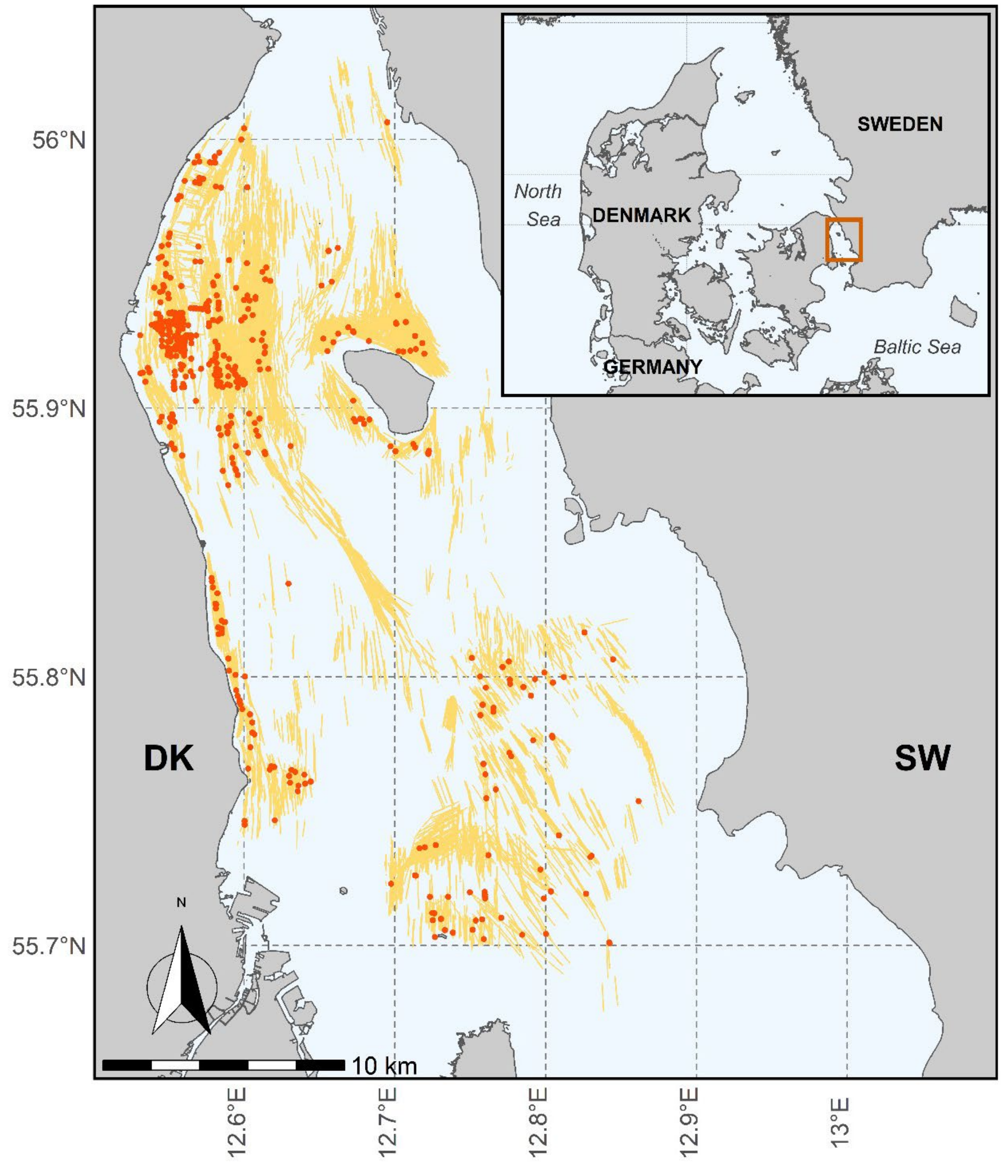

Figure 1: Study area, positions of the hauls (in yellow), and positions of seabird bycatch events (in orange) observed on three gillnetters using electronic monitoring for the period 2010-2018.

The sampled vessels' overall length varied between 9.63 m and 11.05 m, for a gross tonnage (GT) of 5.8 GT to 10.7 GT and an engine power of 74 kW to 107 kW. For these small-scale gillnetters, the duration of a fishing trip never exceeded 9 h. All three vessels used traditional monofilament bottom-set gillnets – or rarely trammel nets – with mesh sizes between 120 mm to 250 mm. Vessels typically set 5 net fleets per fishing trip (median: 5, mean: 5.2, sd: 1.9), with a net length spanning between 200 m and 5500 m (median: 731 m, mean: 790 m, sd: 324 m). Most fishing trips consisted of hauling the net fleets set the previous day, but the soak time could be considerably longer when the target species was lumpsucker (median soak time: 13.8 h, mean: 40.8 h, sd: 43.7 h). Fishing depth varied considerably between vessels and within vessels along the year. Vessel 1 generally set net fleets in deeper waters (median: 15.0 m, mean: 16.9 m, sd: 10.3 m), than the two others (respectively, median: 7 m and 14 m, mean: 8.0 m and 13.0 m, sd: 3.4 m and 3.8 m). On occasions, other gears (e.g. pots, fyke nets) were used and the corresponding trips were excluded. One vessel was one-man crewed (vessel 2), while the two others were operating with either one or two crewmembers on deck.

2.2. EM systems

Two different EM systems were used to monitor the fishing activity. Originally, two vessels were equipped with EM Observe, a solution developed by Archipelago Marine Research Ltd, Canada (http://www.archipelago.ca). Later, these systems were replaced with Black Box Video, developed by Anchorlab, Denmark (http://www.anchorlab.dk/). The third vessel was also equipped with this system. Both hardware solutions worked on the same general principle: a control box installed in the wheelhouse, associated with 2 to 4 waterproof rugged closed-circuit television (CCTV) cameras recording the activity on deck from different angles, and linked to a position sensor (GPS receiver). A monitor plugged onto the control box allowed checking the camera recordings and the system status. The videos were stored on-board on high-capacity hard drives. For the EM Observe system, once the storage capacity was below 25%, a DTU Aqua staff manually replaced the hard drive with a new one in the control box. The full hard drives were retrieved or sent by mail to DTU Aqua. For the Black Box Video system, data were uploaded to a dedicated server automatically every time the vessel was in an area covered with Wi-Fi or GSM/3G/4G mobile network (e.g. the harbour).

On each vessel, one camera was oriented to observe the net breaking the water surface during the hauling phase, while another camera was placed above the sorting table to monitor catch composition and discard. On one vessel, two additional cameras were installed to record the activity on the deck (Figure 2). All cameras were fixed on existing structures whenever possible, but sometimes the addition of a mounting rack was required to obtain the desired field of view.

The quality of the digital recordings varied considerably between the two EM systems (Table 1). In particular, because it was necessary to replace the full hard drives manually on EM Observe, the number of frames per second (fps) and the resolution of the picture were reduced in order to extend the recording time. Conversely, Black Box Video was transmitting data directly over the air, so internal storage capacity was not an issue, and picture quality was thus increased to the maximum. In these conditions, Black Box Video was generally much better at picking up small details in the picture than EM Observe was.

Figure 2: Examples of bird bycatch in Black Box Analyzer [A: common eider (*Somateria mollissima*) adult male; B: great cormorant (*Phalacrocorax carbo*) immature; C: common eider adult female], and in EMI Observe [D: common guillemot (*Uria aalge*)].

Table 1: Comparison between the electronic monitoring systems

	EM Observe		Black Box Video		
Company	Archipelago Marine Research Ltd		Anchor Lab		
Vessel	vessel 1	vessel 3	vessel 1	vessel 2	vessel 3
Monitoring period	May 2010 to November 2013	March 2011 to April 2014	December 2013 to September 2016	March 2016 to December 2018	March 2016 to December 2018
Temporal resolution	1 GPS position recorded every 10 seconds		1 GPS position recorded every 10 seconds		
Lenses	2.6 to 8 mm	2.6 to 8 mm	2.6 to 8 mm	2.6 to 8 mm	2.6 to 8 mm
Frames per second (overview)	2 fps	2 fps	5 fps	5 fps	5 fps
Frames per second (haul & catch)	6 to 9 fps	6 to 9 fps	5 fps	5 fps	5 fps
Camera resolution	640 x 480	640 x 480	1280 x 800 to 1360 x 768	1360 x 768	1360 x 768
Dedicated software for data analysis	EM Interpret (Europe release, Version 11.3.11189)		Black Box Analyzer (Version 4.0.3.0)		

2.3. Video analysis: identifying fishing activity and bycatch events

Analyses of the temporal and spatial characteristics of the fishing trips were done using an electronic monitoring analyser software (EM analyser). Each EM system came with its own dedicated EM analyser (Table 1). Simply put, the recordings made on-board the fishing vessels were stored in a database that associated time, GPS positions and videos. For the end-user, EM analysers presented one or several fishing trips at a time for each vessel, displaying alongside a map with the GPS trace, a timeline indicating the instantaneous vessel speed, and the video recordings from the different cameras.

Analysing video monitoring data is without doubt tedious, and auditing the fishing activity as a fulltime job was considered likely to end up lowering the overall quality of the data. Therefore, for this task, students were hired on 12 h per week contracts, and taught how to identify the fishing activity and detect seabird bycatch events. Four weeks of initial training with an experienced auditor were necessary before a new recruit could work independently. In addition, a random 10% check was put into place to verify the quality of the analysed data. In case important differences were found, a senior staff would watch the corresponding footages with the auditor to understand where the errors originated and what to do to remedy them.

Relevant information was inserted manually as notes in the EM analyser software, and consisted of supplementary rows added to the database. These notes included temporal (date and time), spatial (longitude, latitude) and categorical data, and were used to identify the start and end positions of the sets (i.e. the deployment of the net) and of the hauls (i.e. the retrieval of the net), as well as the occurrence of seabird bycatch. The fishing activity (i.e. steaming, hauling or setting nets) was generally detectable using only the speed, position and course of the vessel. Conversely, detecting seabird bycatch events required watching the videos of each individual haul at no more than 3 to 5 times the normal speed, depending on the quality of the recordings. The resulting database was extracted as a spreadsheet for later use in a statistical software.

Some important parameters were not directly available from the EM analyser, e.g. fishing depth and distance to shore. A GIS software was used to obtain these variables respectively by overlaying the vessels' GPS trace with a high-definition bathymetric map provided by DTU Aqua and using an adhoc "distance to feature" function.

2.4. Video analysis: bird identification

Each bycatch was identified at the lowest possible level using all characteristic features visible on the videos, i.e. general shape and size, colour(s) of the plumage, beak and feet shape and colour, or any other distinctive clue. When possible, sex and other information related to age (e.g. adult, juvenile, first or second winter, breeding or non-breeding, eclipse plumage for male ducks, etc…) were also recorded. EM analyser software provided the possibility to play the recorded footage frame by frame, zoom onto distinctive anatomical features of the animals (plumage, beak, feet…), use different camera angles and replay the sequences as many times as necessary. Being able to review the key characteristics guaranteed that each individual was identified with the highest degree of certainty (Figure 2). Nevertheless, ambient luminosity, weather, cleanliness of the camera lenses or sun glares could strongly affect the overall readability of the picture and thus the identification process. Likewise, fishers would sometimes block the view of the camera, e.g. when disentangling a bird with their back turned toward the lens. In most cases, at least a few frames were exploitable to identify an animal, but at times, very bad video quality made the identification impossible. Such birds were marked as not identified.

2.5. Fishing effort and seabird bycatch rate estimates

Fishing effort was calculated at a fine scale (in kilometer.hour) as the product of total net fleet length and soak time (i.e. the duration of submersion of gillnet fleets). First, the net fleet length was measured as the distance in a straight line between the positions of the start and of the end of a haul; these were defined as the moments (year, month, day, hour, minute, second) where the beginning of the first panel and the end of last panel of the fleet, respectively, broke the water surface. Next, each set and each haul was assigned a unique time value corresponding to the difference between their respective start and end times. Finally, soak time was approximated as the duration between the averaged time of a set and the averaged time of the matching haul.

Mean yearly bycatch rate estimates and associated confidence intervals were obtained using 100,000 bootstrap iterations. Seabird bycatch per unit effort (BPUE) was calculated using two alternative metrics: number of birds captured per fishing trip (*bpt*) and number of birds captured per kilometer.hour (*bkh*). The former metric is a widely used estimate, useful for comparing BPUE across regions; the latter gave access to a measure of bycatch rates at haul level, using the product of length and soak time of the submerged net fleets.

2.6. Fishing logbooks

To verify the completeness of the EM data, official logbooks were collected from the three sampled vessels for the period 2010-2018. Danish fishers are legally bound to fill in these logbooks, which must include information for each individual trip: departure/arrival date and time, type of fishing gear and mesh size, as well as total catch in weight by species by ICES (International Council for the Exploration of the Sea) rectangle. Danish logbooks make no mention of fishing effort in terms of number of nets, soak time or net length. The fishing trips recorded in the EM database were matched to the ones in the logbooks to verify how many trips that actually occurred were missed, i.e. not recorded with the EM systems.

3. Results

3.1. Details of the observed seabird bycatch

Although a European requirement (EC, 2016), official fishing logbooks did not mention seabird bycatch for any of the sampled gillnetters throughout the time of the study. Instead, the video analysis of the EM data allowed the detection of 700 birds from six different families, most of them identifiable at species level (Table 2 and Figure 3). Only eight animals (1.1%) could not be identified; yet, although the species could not be determined, the crew's behaviour clearly indicated that these were indeed birds. A fisher disentangling a bird exhibits a different behaviour than if the catch is a fish. That is, the handling of a dead bird usually takes more time than that of a fish, and bird bycatch are normally stored apart from the boxes containing fish, if not directly discarded overboard. The yearlong sampling scheme gave an insight into the species-specific seasonal variations in bird bycatch in the commercial gillnet fishery taking place in the Sound. Table 2 presents the bycatch records per season and the associated bycatch per unit effort (BPUE) estimates. Anecdotally, a dozen seagulls and great cormorants (*Phalacrocorax carbo)* were entangled while trying to predate on discards; all the affected animals were swiftly freed and released alive by the fishers and did not seem to suffer any injuries. These events were not recorded as bycatch.

Table 2: Seasonal variations of the number of birds taken as bycatch in gillnets, grouped by family and species; the corresponding bycatch per unit effort (expressed as the number of birds per kilometer.hour) is indicated in the parentheses. The identification is given at the lowest possible level (species, genus, family). Data were recorded on three electronically monitored Danish commercial gillnetters in the Sound for the period 2010-2018 (spring = March-May; summer = June-August; fall = September-November; winter = December-February).

Family	**Species**	***% total bycatch***	**Spring**	**Summer**	**Fall**	**Winter**	**YEAR**
Anatidae	Common eider *Somateria mollissima*	***58.4***	n = 106 (0.000606)	n = 14 (0.000289)	n = 236 (0.054200)	n = 53 (0.007150)	**n = 409 (0.001758)**
	Scoter *Melanitta spp.*	***3.1***	n = 2 (0.000007)	-	n = 18 (0.000383)	n = 2 (0.000006)	**n = 22 (0.000099)**
	Not identified	***0.4***	n = 2 (0.000008)	-	n = 1 (0.000026)	-	**n = 3 (0.000009)**
Phalacrocoracidae	Great cormorant *Phalacrocorax carbo*	***19.6***	n = 2 (0.000008)	n = 15 (0.000417)	n = 84 (0.002272)	n = 36 (0.009180)	**n = 137 (0.009040)**
Alcidae	Common guillemot *Uria aalge*	***12.4***	n = 1 (0.000003)	-	n = 39 (0.001335)	n = 47 (0.001954)	**n = 87 (0.000823)**
	Razorbill *Alca torda*	***2.3***	-	n = 1 (0.000024)	n = 8 (0.000136)	n = 7 (0.000096)	**n = 16 (0.000064)**
	Not identified	***1.0***	n = 4 (0.000013)	-	n = 3 (0.000077)	-	**n = 7 (0.000023)**
Laridae	Gull *Larus spp.*	***0.4***	n = 1 (0.000002)	n = 1 (0.000014)	n = 1 (0.000011)	-	**n = 3 (0.000007)**
Gavidae	Loon *Gavia spp.*	***0.6***	n = 1 (0.000005)	-	n = 3 (0.000073)	-	**n = 4 (0.000019)**
Podicipedidae	Great crested grebe *Podiceps cristatus*	***0.4***	-	-	-	n = 3 (0.000047)	**n = 3 (0.000012)**
	Red-necked grebe *Podiceps grisegena*	***0.1***	-	n = 1 (0.000031)	-	-	**n = 1 (0.000008)**
Unidentified bird		***1.1***	n = 1 (0.000002)	n = 1 (0.000007)	n = 2 (0.000033)	n = 4 (0.000093)	**n = 8 (0.000034)**
All birds		**100%**	**n = 120 (0.000653)**	**n = 33 (0.000782)**	**n = 395 (0.009430)**	**n = 152 (0.003142)**	**n = 700 (0.003300)**

Generally, bird bycatch, when reported (e.g. using official logbooks), is not associated to a specific position, but is instead mentioned as number of birds per fishing trip per statistical area (e.g. ICES statistical rectangle level). Therefore, having access to the exact coordinates of every incidental 251 catch was a major benefit of using electronic monitoring (Figure 3).

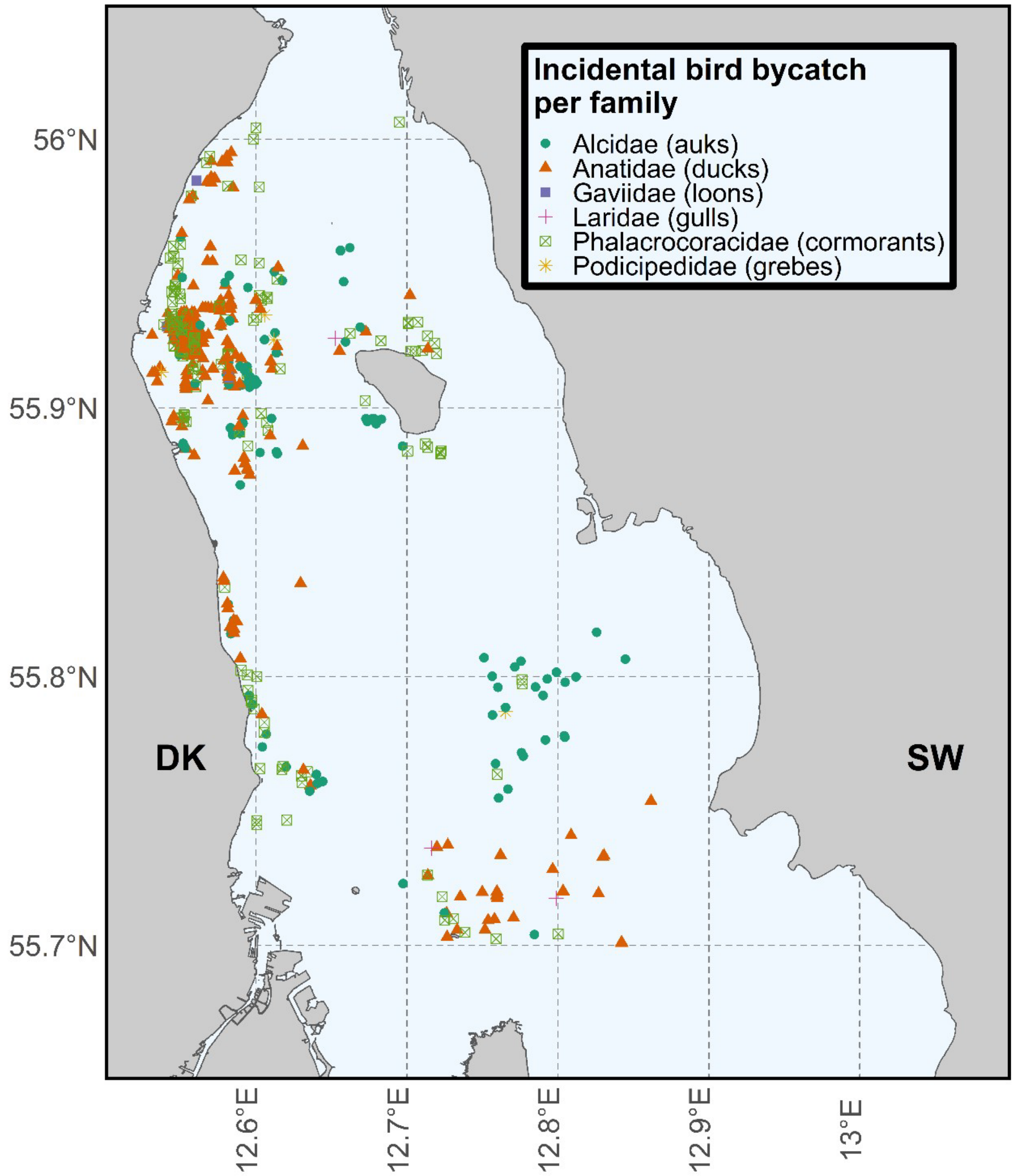

Figure 3: Positions of the incidental catch of seabirds in gillnet recorded using EM and grouped by families.

Moreover, electronic monitoring provided information on the structure of the populations of seabirds captured in gillnets. Some groups of seabirds, e.g. ducks (Anatidae), display characteristic dimorphism between sexes, or between juveniles, immatures and adults. In particular, it was possible to identify the sex and maturity (adult or juvenile) of the common eiders (*Somateria mollissima)* taken as bycatch, and to distinguish juvenile (or immature) great cormorants from breeding and non-breeding adults (Table 3). Male common eiders represented 69.7% of the catches and females 23.0%, the rest being juveniles (3.4%) and unidentifiable individuals (3.9%). Bycatch of great cormorant was dominated by juvenile and immature birds (56.2%), while adult birds made up less than a third of the yearly average bycatch. However, due to a system failure, 21 individuals (i.e. about 15%) could not be classified. The reason for this is that some of the oldest video data, where these bycatches were recorded, were lost; species identification had been done prior to the data loss, but not the aging of the birds.

Table 3: Bycatch composition for the two bird species the most frequently captured in gillnets in the Sound, the common eider and the great cormorant, per season (spring = March-May; summer = June-August; fall = September-November; winter = December-February) for the period 2010-2018. The number of observations per group is indicated in the parentheses.

Species	Status	Spring	Summer	Fall	Winter	**Yearly average**
Common eider	Female	16.0% (n=17)	28.6% (n=4)	25.0% (n=59)	26.4% (n=14)	**23.0%**
	Male	75.5% (n=80)	42.9% (n=6)	69.5% (n=164)	66.0% (n=35)	**69.7%**
	Juvenile (undetermined sex)	4.7% (n=5)	0.0% (n=0)	2.5% (n=6)	5.7% (n=3)	**3.4%**
	Unidentified	3.8% (n=4)	28.6% (n=4)	3.0% (n=7)	1.9% (n=1)	**3.9%**
Great cormorant	Adult (breeding and non-breeding)	50.0% (n=1)	6.7% (n=1)	33.3% (n=28)	25.0% (n=9)	**28.5%**
	Juvenile and immature	0.0% (n=0)	66.7% (n=10)	50.0% (n=42)	69.4% (n=25)	**56.2%**
	Unidentified	50.0% (n=1)	26.7% (n=4)	16.7% (n=14)	5.6% (n=2)	**15.3%**

3.2. Fishing effort

Through the study periods, official logbooks recorded 2748 fishing trips in total, while sensor data from EM systems recorded 2118 trips, consisting of 10964 hauls (Table 4). Fishing trips registered in official logbooks and recorded using electronic monitoring could in general be linked together, although some gaps were found (Table 4 and Table 5). Unrecorded trips in the EM system resulted from occasional technical issues, e.g. GPS sensor defects or power failure in the wheelhouse. These failures often required to send a technician on-board the fishing vessel to fix the problem, and could sometimes last for extended periods (e.g. vessel 2). A number of trips (78) that were recorded with electronic monitoring were not mentioned in the official logbooks. On one vessel, a closer look at the logbook data showed that, sometimes, the skipper aggregated two consecutive trips into one or that some fishing trips were simply not registered at all. Therefore, and although the logbooks are normally assumed to provide an exact measure of the fishing effort, a small uncertainty exists concerning the real number of fishing trips per vessel over the whole study period. Furthermore, the mean monthly fishing effort varied considerably along the year between and within vessels (Table 5). For example, in case of adverse weather conditions or strong currents that could damage their nets, skippers normally choose to stay in harbour.

Table 4: Comparison between the numbers of trips registered in logbooks and recorded with the EM systems. The years indicate the periods where electronic monitoring was active for each vessel. The total number of hauls recorded and analysed using electronic monitoring are indicated per vessel, as well as the corresponding number of hauls per trip (±1 standard deviation).

VESSEL	**Total number of fishing trips** (from logbooks)	**Number of fishing trips recorded with EM** *(% covered)*	**Number of hauls recorded with EM** *(mean number of hauls per trip ± sd)*
Vessel 1 (2010-2016)	1344	1197 *(89%)*	6798 *(3.70 ± 2.17)*
Vessel 2 (2016-2018)	436	196 *(45%)*	532 *(2.10 ± 1.14)*
Vessel 3 (2010-2014 and 20162018)	968	725 *(75%)*	3635 *(3.16 ± 1.66)*
TOTAL	2748	2118 *(77%)*	10965 *(3.44 ± 2.02)*

Table 5: Comparison of the mean number of fishing trips per month per vessel recorded using EM (in bold) and registered in logbooks (in italics in parentheses). The years indicate the periods where electronic monitoring was active for each vessel.

Vessel	Jan	Feb	Mar	Apr	May	June	July	Aug	Sep	Oct	Nov	Dec
Vessel 1 (2010-2016)	**22.3** *(23.2)*	**15.5** *(16.2)*	**21.5** *(21.2)*	**9.0** *(7.8)*	**17.0** *(21.0)*	**20.5** *(20.0)*	**14.6** *(15.7)*	**13.3** *(18.0)*	**15.0** *(20.2)*	**19.8** *(23.5)*	**23.0** *(23.5)*	**19.7** *(18.7)*
Vessel 2 (2016-2018)	**5.0** *(10.0)*	**8.0** *(9.5)*	**7.3** *(14.0)*	**5.3** *(14.0)*	**3.5** *(13.7)*	**7.5** *(11.0)*	**6.0** *(11.0)*	**4.0** *(16.3)*	**10.3** *(14.3)*	**7.3** *(10.7)*	**10.0** *(18.0)*	**5.0** *(9.3)*
Vessel 3 (2010-2014; 2016-2018)	**6.0** *(7.8)*	**14.3** *(13.2)*	**17.8** *(19.8)*	**7.8** *(10.3)*	**20.2** *(23.2)*	**19.2** *(23.3)*	**20.8** *(23.0)*	**19.2** *(19.2)*	**10** *(12.7)*	**0** *(4.0)*	**4.0** *(19.8)*	**9.7** *(16.3)*

3.3. Bycatch rate estimates

Bycatch of birds occurred in 13.3% of the fishing trips recorded with EM and in 3.5% of the hauls. A few mass bycatch events may have had an overly strong influence on the mean bycatch rate estimates (Table 2). Especially, estimated bycatch per unit effort (BPUE) rates observed in November were higher than for any other months, notably because of several extreme bycatch events recorded 302 in 2014. In fact, in 95% of the trips where bycatch was observed, no more than six birds per trip were captured, while in the remaining 5%, up to 57 birds per trip were taken as bycatch. These 5% represented only 14 out of 2118 fishing trips, but accounted for 40% of the total incidental catch of seabirds observed during the study.

In order to visualise the influence of rare mass bycatch events on mean bycatch rate estimates, BPUE was calculated both with the full dataset (Figure 4), and after having excluded the 14 fishing trips where more than six birds had been captured (Figure 5).

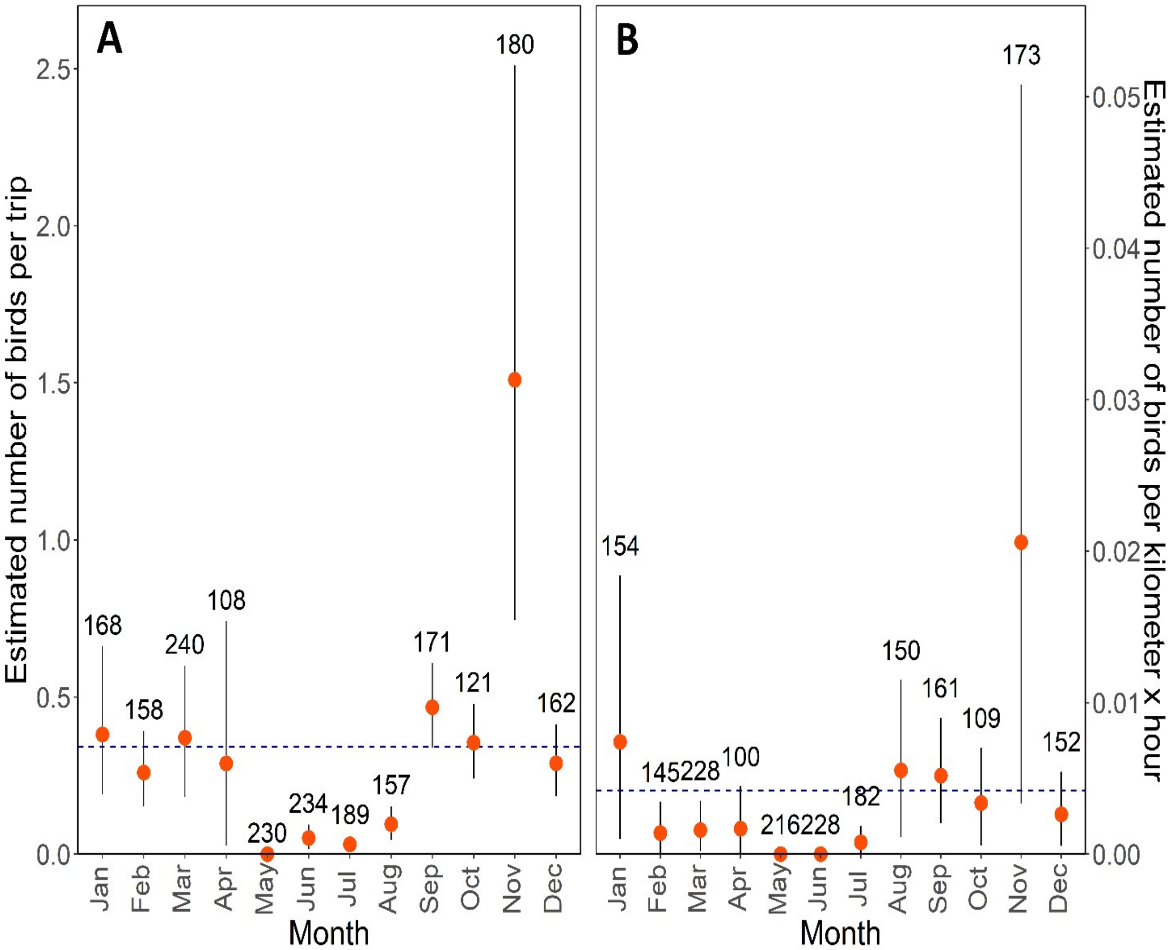

Figure 4: Monthly estimated bycatch per unit effort (BPUE). A. Total number of incidental bird bycatch per fishing trip; B. Total number of incidental bird bycatch per kilometer.hour. Orange dot: mean BPUE estimates; plain black bars: 95% confidence intervals; dashed blue bar: average yearly bycatch rate. The total number of fishing trips for each month is indicated on top of the CI bars. The estimates are based on 100,000 bootstrap repetitions.

Using the full dataset, mean BPUE was estimated at 0.00418 bird per kilometer.hour (95% confidence interval: 0.00075 to 0.00966; 100,000 bootstraps), or 0.34 bird per trip (95% confidence interval: 0.18 to 0.56; 100,000 bootstraps), with important variation between months (Figure 4). The reduced dataset containing only the fishing trips where no more than 6 birds had been captured led to an estimated yearly average of 0.0026 bird per kilometer.hour (95% confidence interval: 0.0006 to 0.0052; 100,000 bootstraps), and 0.21 bird per trip (95% confidence interval: 0.14 to 0.30; 100,000 bootstraps) (Figure 5).

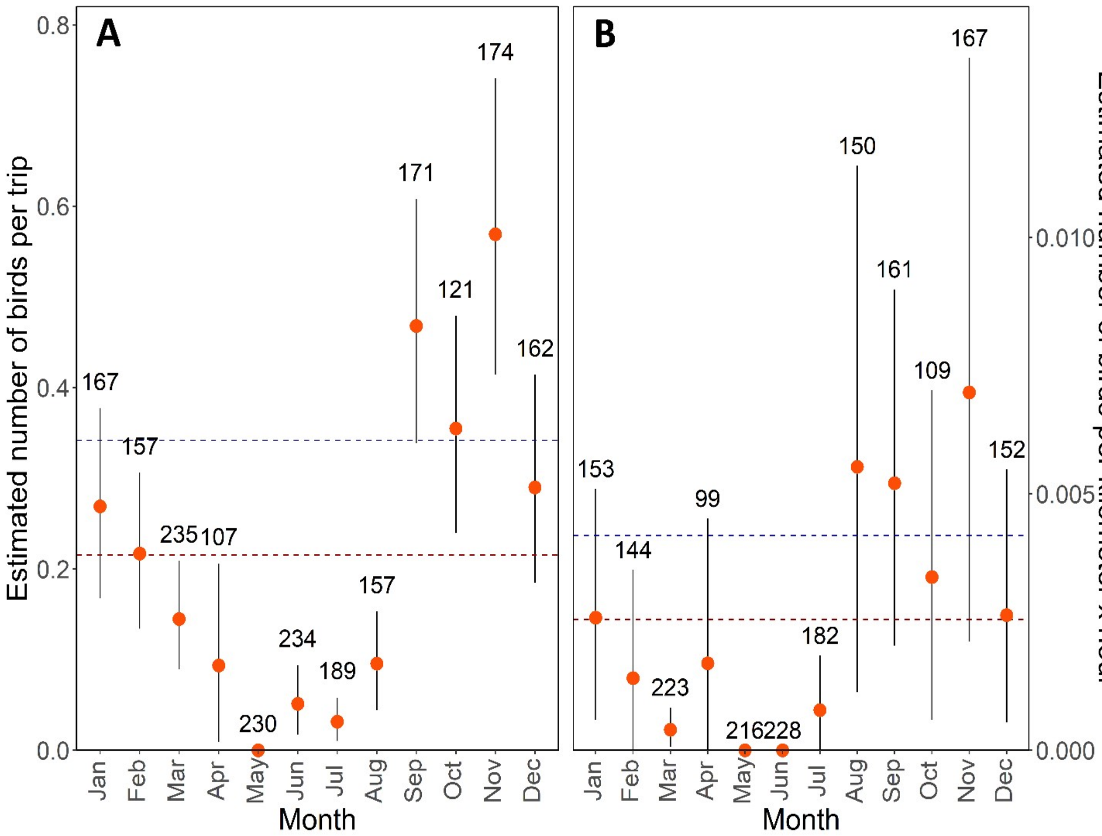

Figure 5: Monthly estimated bycatch per unit effort (BPUE), after excluding the 14 most extreme bycatch events, corresponding to the 5% upper quantile: A. Total number of incidental bird bycatch per fishing trip; B. Total number of incidental bird bycatch per kilometer.hour. Orange dot: mean BPUE estimates; plain black bars: 95% confidence intervals; dashed red bar: average yearly bycatch rate excluding the 14 fishing trips where more than six birds were captured (i.e. 95% of the trips with bycatch); dashed blue bar: average yearly bycatch rate including the whole dataset. The total number of fishing trips for each month is indicated on top of the CI bars. The estimates are based on 100,000 bootstrap repetitions.

In the full dataset, the influence of mass bycatch events on BPUE estimates was clear. November in particular stood out as the month where birds were the most at risk of drowning in fishing nets. Overall, fall and winter were the seasons for which the rate of incidental catch was the highest, when many seabirds use the Sound as a feeding and resting ground. The period between the end of spring and summer showed in comparison very few occurrences of bycatch. Seasonal variations of BPUE also revealed a spatial component (Figure 6), with important local differences in mean bycatch rates between seasons, indicating potential seasonal bycatch hotspots.

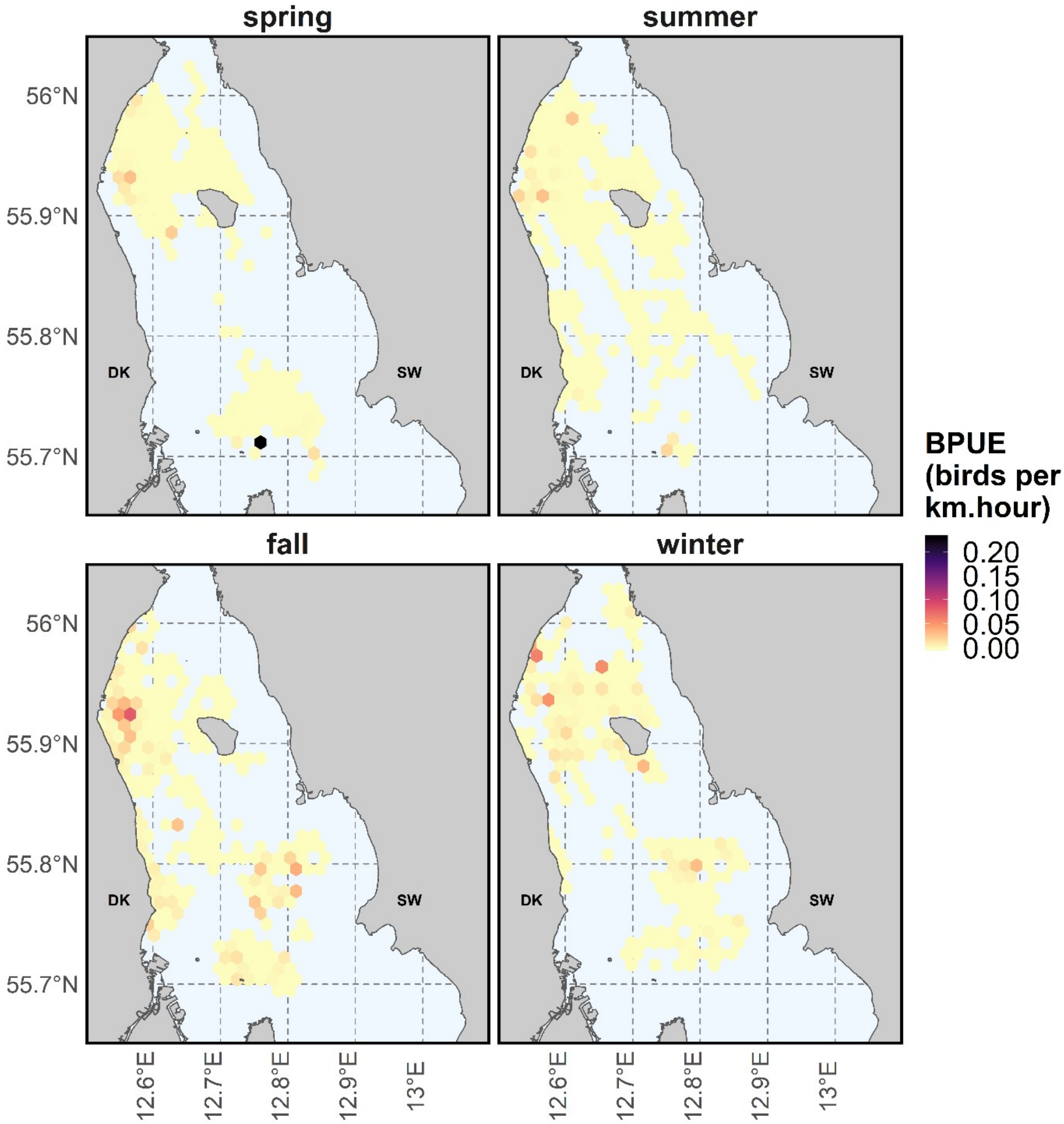

Figure 6: Seasonal variations in mean estimated bycatch per unit effort (in number of bird per kilometer.hour) in the Sound for the period 2010-2018 (spring = March-May; summer = June-August; fall = September-November; winter = December-February).

4. Discussion

There is a relative lack of knowledge on the overall impact of fishing on seabird populations in Europe (EC, 2012). In this context, the present paper demonstrates the feasibility of using electronic monitoring (EM) systems to collect long fine-scale time-series of incidental catch of seabirds in commercial fisheries. In this study, whereas no bycatch was reported in official logbooks, EM systems were able to detect incidental catches of seabirds. Bycatch could be identified at species level, with only 1.1% of the 700 specimen remaining unidentified (Table 2). Combined with the exact position of the fishing gears and the precise duration of soak time, EM allowed measuring fishing effort at high-resolution (spatial and temporal). The systems offered the possibility to calculate estimates of bird bycatch per unit effort (BPUE) at haul level (Figure 4), thus uncovering potential areas of high risk of bycatch (Figure 6). In addition, for some bird species with plumage dissimilarities between sex (e.g. common eider), or between juveniles and adults (e.g. great cormorant), it was possible to establish ratios revealing differences in bycatch risks between groups of individuals of the same species (Table 3).

The quality of bycatch data collected using CCTV monitoring is highly dependent on the quality of the videos on the one hand, and on the other hand, on the ability of the video analysts to detect bycatch. The EM systems, in particular the cameras, were upgraded in 2014. At first, the main motivation for this change was to provide a more convenient workflow for the analysts, and to reduce the risk of data loss by going from manual data acquisition (postal delivery of the hard drives) to automatized over-the-air data acquisition. In the first part of the study, to reduce the number of postal shipments to a minimum, the Archipelago EM Observe systems were set up to maximize the recording time by limiting the video quality in terms of fps and resolution. After the change to the Black Box Video system, internal storage capacity was not an issue anymore, so the video quality was increased accordingly: higher resolution, higher number of frames per seconds, better contrasts (Table 1). Nevertheless, even with the best cameras installed on-board, the readability of the recorded footage still degrades quickly if the lenses are covered with smudge, water droplets or salt crusts. It was therefore essential to contact skippers regularly to re-address the importance of frequently wiping clean the cameras and to use simple prevention measures such as applying rain repellent on the lenses.

Another recurring concern is to maintain the capacity of the human analysts to correctly and consistently identify the fishing activity (anchor sets, retrieving of the nets, bycatch). Sometimes, unanticipated events may occur, and a critical eye is necessary to understand and judge the situation correctly. This is particularly true for gillnet fisheries when estimating soak time from EM recordings. For instance, after a storm, a net fleet might have been broken apart and scattered. Associating the correct set, and thus the correct soak duration, to each net fragment requires high focus. Generally, hours on end of video analysing leads to mental fatigue for the analysts. Therefore, video auditors were asked to work no more than 6 h daily with a pause every 2 h in order to maintain the required level of concentration. Another incentive to keep standards high is to operate random quality checks on already audited data. Moreover, feedback from the EM analysts is essential. Regular meetings with the whole team to discuss possible methodological improvement, data flaws and to plan future work clearly improved the quality of the bycatch data collection over time.

Besides human operators, computer-assisted image recognition, artificial intelligence and deep-learning algorithms were initially considered to facilitate and speed up the analysing process. These fields of research are progressing rapidly (e.g. Chen et al., 2014; Niemi and Tanttu, 2018; Hong et al., 2019). Still, at the time the study started, no algorithm could perform better than a trained human being does, at least not with data from small-scale vessels. Standardising video footage might help accelerate the development of efficient image-recognition software. However, EM systems are always customised configurations, which are adapted to specific vessel's characteristics, e.g. in terms of camera placement, arrangement of deck and fishing procedures. Obtaining standardised images for all small-scale vessels is therefore unrealistic. Nonetheless, there is no doubt that in a near future, these technologies will be mature enough to be implemented in operational electronic monitoring systems.

Even if the analysts assess the videos from the fishing vessels with the greatest care, there is always a risk of missing an inconspicuous bird. Ideally, vessels participating in such a study should at least register the number of bycatch per fishing trip – and if possible the species – as is already a requirement in the European Union (EC, 2016). Nevertheless, the accuracy of fisher-reported data is questionable (Mangi et al., 2015). Skippers may make mistakes filling in logbooks (Kindt-Larsen et al., 2011). Regarding bycatch of protected species, some authors report a systematic lack of congruence between EM data (and/or on-board observers data) and logbook data (Macbeth et al., 2018; Emery et al., 2019). Fishers may also sometimes simply miss a bycatch, e.g. if a bird falls from the net before being hauled up on board. Therefore, EM analysts should treat logbook data with a grain of salt, and they should not only audit the days where fishers registered bycatch.

Bearing in mind that the quality of the data collected with EM was not always optimal and that some fishing trips were not recorded in the first place (Table 4), the bird bycatch per unit effort (BPUE) estimations presented in this study should be considered as conservative. Yet, the overall temporal trend was clear. Estimated BPUE was one order of magnitude higher in fall and winter than in spring and summer (Table 2), leading to more bird casualties in this period (547 versus 153, respectively). This was expected, as the Sound is a major wintering area for many migratory birds (Skov, 2011). In terms of proportion, three species made up to more than 90% of the total observed bycatch: the common eider *S. mollissima* (58.4%), the great cormorant *P. carbo* (19.6%) and the common guillemot *Uria aalge* (12.4%). Except for seagulls (0.4%), all the birds found drowned in gillnets were diving species. These findings confirm that diving seabirds are generally more vulnerable to bycatch in gillnets than are surface feeding seabirds (Žydelis et al., 2009; 2013). This contrasts with a recent study from Norway, which found that the largest proportion of bird bycatch in the Norwegian Reference Fleet coastal gillnet fishery was a surface-feeding seabird, the Northern fulmar *Fulmarus glacialis* (Bærum et al., 2019). Moreover, the distribution of bycatch in the Sound showed important disparities between species and a possible clustering for some (Figure 3). Common eider bycatch was registered mostly in shallow waters, whereas pursuit divers such as common guillemots were typically observed farther offshore. This is in line with what is known of the feeding strategies of those species. Common eiders feed principally on molluscs and forage on the seabed. In the Sound, their favourite prey item, the blue mussel *Mytilus edulis*, is abundant and grows in large aggregations (*aka* mussel banks). Problematically for these birds, fishers tend to set their nets in and around mussel banks where they expect to find the largest cods. On the contrary, common guillemots can dive farther down to catch the fish they feed on, and they were accordingly often captured in nets set in deeper waters. Conversely, incidental catch of great cormorants, also a pursuit diver, did not seem to be associated with depth or distance to shore. Individuals taken as bycatch may have been specialised in foraging in nets, which would put them at higher risks of entanglement (Bregnballe and Frederiksen, 2006). Furthermore, a differential risk of drowning was observed within two species: the common eider and the great cormorant (Table 3). Common eider vulnerability to bycatch was clearly sex-biased. Males represented almost 70% of the total catch, but this proportion reflects the male bias in the Baltic population (Ramula et al., 2018). Great cormorant bycatch appeared age-biased (56% juveniles and immature birds). Bregnballe and Frederiksen (2006) hypothesised that young and less experienced individuals are more at risk of interacting with soaked fishing gears and drown.

These few examples emphasize that deadly seabird-fishery interactions cannot be summarised as a simple overlap between fishing effort distribution and seabird distribution. Complex and species-specific relationships exist between birds and fisheries, and depend on many factors including behavioural, operational, environmental or meteorological (Torres et al., 2013; Clay et al., 2019). In the Sound, in the absence of detailed maps of the fishing effort and of the seabird distribution, long term EM monitoring of coastal gillnetters provides insightful data, which helps understanding underlying bird-fisheries interactions. This knowledge is essential to improve and advance both the management of coastal fisheries and the conservation of marine avifauna (Northridge et al., 2017; Le Bot et al., 2018). Besides, understanding the possible impact of fisheries bycatch at population levels requires further investigation. Two of the most affected species in this study, the common eider and the common guillemot, regionally qualify as near threatened on the IUCN Red List, while the great cormorant is considered least concerned (IUCN 2019). Moreover, because of the large decline observed since the 1990's, the HELCOM Red List categorises the common eider wintering population as endangered (Kontula and Haldin, 2013). Additionally, fishing effort is not randomly distributed, since skippers normally set their nets in areas where they expect to maximize the catch of their target species. Therefore, bycatch numbers are only relevant in relation to fishing intensity. The literature often reports bycatch rates in gillnet fisheries as the mean number of animals captured per trip, or as the mean number of animals captured per net. Here, EM was utilized to access fine-scale effort data over long periods and to identify fishing grounds and bycatch hotspots precisely. Furthermore, incidental catches are rare events, and authors studying seabird-fisheries interactions often work with datasets containing sporadic bird bycatch. Such data are typically overdispersed (relative to the Poisson assumption), with a high proportion of zeros (i.e. no bycatch in a haul/trip) and localised large counts due to the gregarious behaviour of some species (Sims et al., 2008). As suggested by Bærum et al. (2019), these unpredictable extreme events could considerably bias mean BPUE estimates and lead to exaggeratedly high predictions if used to feed a statistical model. In the present study, 14 fishing trips (corresponding to the 5% upper quantile) captured more than six birds per trip. To visualize the influence of mass bycatch, BPUE estimates were presented with and without these extreme events (Figure 4 and Figure 5). Moreover, BPUE estimates were also reported both as the number of bird captured per fishing trip (*bpt*) – as is the case in numerous publications on seabird bycatch in gillnet fisheries (Le Bot et al., 2018) – and as the number of birds per kilometer.hour (*bkh*). The latter estimates BPUE at haul level by incorporating explanatory operational factors (soak duration and net length). Regardless of the metric, the comparison of Figures 4 and 5 showed that mass bycatch events clearly affect the mean estimator of BPUE: after excluding the 5% upper quantile, mean yearly estimated bycatch rates dropped by almost a third, and months where extreme bycatch had been observed (especially November) appeared much less peculiar. Therefore, it may be necessary to remove these outliers when building a predictive statistical model to allow the model to converge or at least to calculate reliable estimates. However, with 40% of all observed bycatch recorded in mass bycatch events, ignoring these will necessarily result in over-optimistic results.

A straightforward solution to overcome the problem of accuracy of bycatch rate estimates is to increase the monitoring effort. However, the cost associated with EM (both implementation and running cost) is often pointed out as a weakness (van Helmond et al., 2019). Consequently, it is tempting to choose to analyse only a randomly selected fraction of the fishing activity. Problematically, bird bycatch events are rare and not randomly distributed. In this study, 40% of the casualties were recorded in less than 0.2% of the hauls. As a result, examining a sample of the complete dataset would likely result in inaccurate estimates. Still, compared to alternatives like human observers, EM is generally less biased (no observer effect) and more cost-effective (Michelin et al., 2018). Additionally, self-reporting of bycatch for the vessels equipped with EM could help reduce the number of hauls to analyse and the cost associated with it, as long as quality control procedures are in place. In turn, a dedicated EM programme should aim at evaluating bycatch rates accurately on few representative vessels instead of spreading the monitoring effort on a large portion of the fleet whose activity will be only partially analysed.

Quality EM data requires the full cooperation of the participating fishing vessels. Crewmembers need to comprehend the necessity to keep a clear and unobstructed view for the CCTV cameras, and not withdraw information by switching off the monitoring system. A close collaboration between the fishing industry and scientists, as well as strong incentives (e.g. in the form of additional quotas or days at sea), is necessary to overcome the initial distrust that the fishing community might have toward EM systems (Mangi et al., 2015). Ideally, a monitoring programme should randomly select the fishing vessels to survey. It was not the case here. The sampled vessels were all volunteers, and consequently, they cannot be considered representative of the overall fishing fleet. Besides, regular contacts with the skippers involved in the project may have made them aware of seabird bycatch issues. In turn, they may have avoided areas where they believed the risk of incidental catch was high, thus making the estimated BPUE for the sampled gillnetters lower than for the rest of the fleet. However, on small-scale vessels such as the ones in the Sound, catching many birds increases handling time enormously and reduces profitability. Therefore, fishers tend to minimise unwanted catches, avoiding fishing grounds where the possibility of capturing many seabirds is high, even if this means relocating their nets to areas potentially less attractive in terms of catches of target species (Savina, 2018).

In summary, i) bycatch rate estimates were based on a fraction of the total fishing effort of the sampled vessels (Table 4), ii) mass bycatch events were excluded to obtain more reliable mean BPUE estimates (Figure 4 and Figure 5), and iii) participating fishing vessels may have been more attentive to seabird bycatch than average. Consequently, the seabird bycatch rates presented here ought to be considered conservative estimates. Nevertheless, determining such a baseline is essential to unfold the overall impact of gillnets on the seabird populations of the western Baltic Sea.

Establishing long-term electronic monitoring programmes in small-scale gillnet fisheries can provide unique information on incidental captures of seabirds and on the factors associated with bycatch, including fishing effort. Collecting such data is essential in fisheries with a suspected bird bycatch problem. For instance, the lumpsucker gillnet fisheries in the North Atlantic, characterized by long soak times, extensive net length and the use of large meshes, have been reported to capture large numbers of seabirds (Christensen-Dalsgaard et al., 2019). In these fisheries, EM with CCTV could, together with on-board observers, be the most efficient way to collect seabird bycatch data, essential both for fisheries managers to ensure the sustainability of artisanal coastal fisheries and for conservation scientists to tackle seabird populations decline.

Conclusions

Electronic monitoring with CCTV appears to be a reliable solution for monitoring the bycatch of seabirds in coastal gillnet fisheries, where vessels are usually too small to accommodate an on-board observer. Video monitoring data is accurate enough to identify individuals at species level and for some species to age and sex them. The high precision of the bycatch rates estimates, both spatially and temporally, allows the determination of areas of high risks of bird bycatch, although mean BPUE are arguably underestimated due to the nature of the sampling scheme. More in-depth analysis of the EM data collected for this study will allow determining which operational and non-operational factors influence seabird bycatch in gillnets, which in turn will permit estimating the overall number of bird casualties at fleet level.

Declaration of competing interest

None.

Acknowledgements

This work was made possible through project funding from the European Maritime and Fisheries Fund and the Danish Fisheries Agency (grant number 33113-I-16-040), which is gratefully acknowledged. Moreover, the authors wish to thank all three vessels' crewmembers without whom this study would not have been feasible, as well as all the DTU analysts who participated in the auditing of the video monitoring data. Finally, the authors express their gratitude to Anchorlab and Archipelago Marine Research Ltd for their technical support.

References

Anonymous, 2019. Annual report from Denmark on the implementation of Council Regulation (EC) No 812/2004.

Bærum, K.M., Anker-Nilssen, T., Christensen-Dalsgaard, S., Fangel, K., Williams, T. and Vølstad, J.H., 2019. Spatial and temporal variations in seabird bycatch: Incidental bycatch in the Norwegian coastal gillnet-fishery. PloS one, 14(3), p.e0212786.

Bellebaum, J., Schirmeister, B., Sonntag, N. and Garthe, S., 2013. Decreasing but still high: bycatch of seabirds in gillnet fisheries along the German Baltic coast. Aquatic Conservation: Marine and Freshwater Ecosystems, 23(2), pp.210-221.

Bregnballe, T. and Frederiksen, M., 2006. Net-entrapment of great cormorants Phalacrocorax carbo sinensis in relation to individual age and population size. Wildlife Biology, 12(2), pp.143-151.

Chen, G., Han, T.X., He, Z., Kays, R. and Forrester, T., 2014, October. Deep convolutional neural network based species recognition for wild animal monitoring. In 2014 IEEE International Conference on Image Processing (ICIP) (pp. 858-862). IEEE.

Christensen-Dalsgaard S., Anker-Nilssen T., Crawford R., Bond A., Sigurðsson G.M., Glemarec G.,
EHansen E.S., Kadin M., Kindt-Larsen L., Mallory M., Merkel F.R., Petersen A, Provencher J., Bærum K.M., 2019. What's the catch with lumpsuckers? A North Atlantic study of seabird bycatch in lumpsucker gillnet fisheries, Biological Conservation, 240,

Clay, T.A., Small, C., Tuck, G.N., Pardo, D., Carneiro, A.P., Wood, A.G., Croxall, J.P., Crossin, G.T. and Phillips, R.A., 2019. A comprehensive large-scale assessment of fisheries bycatch risk to threatened seabird populations. Journal of Applied Ecology, 56, pp.1882-1893.

Croxall, J.P., Butchart, S.H., Lascelles, B.E.N., Stattersfield, A.J., Sullivan, B.E.N., Symes, A. and Taylor, P.H.I.L., 2012. Seabird conservation status, threats and priority actions: a global assessment. Bird Conservation International, 22(1), pp.1-34.

Dalskov, J., & Kindt-Larsen, L., 2009. Final report of Fully Documented Fishery. Charlottenlund: National Institute of Aquatic Resources, Technical University of Denmark. DTU Aqua-rapport, No.
204-09

Dias, M.P., Martin, R., Pearmain, E.J., Burfield, I.J., Small, C., Phillips, R.A., Yates, O., Lascelles, B., Borboroglu, P.G. and Croxall, J.P., 2019. Threats to seabirds: a global assessment. Biological Conservation.

Emery, T.J., Noriega, R., Williams, A.J. and Larcombe, J., 2019. Measuring congruence between electronic monitoring and logbook data in Australian Commonwealth longline and gillnet fisheries. Ocean & coastal management, 168, pp.307-321.

European Commission, 2012. Communication from the Commission to the European Parliament and the Council: Action Plan for reducing incidental catches of seabirds in fishing gears. European Commission, Brussels.

European Commission, 2016. Commission Implementing Decision (EU) 2016/1251 of 12 July 2016 adopting a multiannual Union programme for the collection, management and use of data in the fisheries and aquaculture sectors for the period 2017-2019 (notified under document C(2016) 4329).

European Union, 2009. Directive 2009/147/EC of the European Parliament and of the Council of 30 November 2009 on the conservation of wild birds. Official Journal of the European Communities, 20, pp.7-25.

Hong, S.J., Han, Y., Kim, S.Y., Lee, A.Y. and Kim, G., 2019. Application of Deep-Learning Methods to Bird Detection Using Unmanned Aerial Vehicle Imagery. Sensors, 19(7), p.1651.

ICES, 2018. Report from the Working Group on Bycatch of Protected Species (WGBYC), 1–4 May 2018, Reykjavik, Iceland. ICES CM 2018/ACOM:25. 128 pp.

IUCN, 2019. The IUCN Red List of Threatened Species. Version 2019-2. http://www.iucnredlist.org. (downloaded on 01 November 2019.).

Kindt-Larsen, L., Kirkegaard, E. and Dalskov, J., 2011. Fully documented fishery: a tool to support a catch quota management system. ICES Journal of Marine Science, 68(8), pp.1606-1610.

Kindt-Larsen, L., Larsen, F., Stage, B. and Dalskov, J., 2012a. Final report. Fully documented fishery onboard gillnet vessels > 15 m. Charlottenlund: DTU Aqua. Institut for Akvatiske Ressourcer.

Kindt-Larsen, L., Dalskov, J., Stage, B. and Larsen, F., 2012b. Observing incidental harbour porpoise Phocoena phocoena bycatch by remote electronic monitoring. Endangered Species Research, 19(1), pp.75-83.

Kindt-Larsen, L., Berg, C.W., Tougaard, J., Sørensen, T.K., Geitner, K., Northridge, S., Sveegaard, S. and Larsen, F., 2016. Identification of high-risk areas for harbour porpoise Phocoena

phocoena bycatch using remote electronic monitoring and satellite telemetry data. Marine Ecology Progress Series, 555, pp.261-271.

Kontula, T. and Haldin, J., 2013. HELCOM Red List of Baltic Sea species in danger of becoming extinct.

Le Bot, T., Lescroël, A., Grémillet, D. and Handling editor: Stephen Votier, 2018. A toolkit to study seabird–fishery interactions. ICES Journal of Marine Science, 75(5), pp.1513-1525.

Lewison, R.L., Crowder, L.B., Wallace, B.P., Moore, J.E., Cox, T., Zydelis, R., McDonald, S., DiMatteo, A., Dunn, D.C., Kot, C.Y. and Bjorkland, R., 2014. Global patterns of marine mammal, seabird, and sea turtle bycatch reveal taxa-specific and cumulative megafauna hotspots. *Proceedings of the National Academy of Sciences*, *111*(14), pp.5271-5276.

Macbeth, W.G., Butcher, P.A., Collins, D., McGrath, S.P., Provost, S.C., Bowling, A.C., Geraghty, P.T.
and Peddemors, V.M., 2018. Improving reliability of species identification and logbook catch reporting by commercial fishers in an Australian demersal shark longline fishery. Fisheries Management and Ecology, 25(3), pp.186-202.

Mangi, S.C., Dolder, P.J., Catchpole, T.L., Rodmell, D. and de Rozarieux, N., 2015. Approaches to fully documented fisheries: practical issues and stakeholder perceptions. Fish and Fisheries, 16(3), pp.426-452.

Michelin, M., Elliott, M., Bucher, M., Zimring, M. and Sweeney, M., 2018. Catalyzing the growth of electronic monitoring in fisheries. The Nature Conservancy and California Environmental Associates.

Niemi, J. and Tanttu, J., 2018. Deep Learning Case Study for Automatic Bird Identification. Applied Sciences, 8(11), p.2089.

Northridge, S., Coram, A., Kingston, A. and Crawford, R., 2017. Disentangling the causes of protected-species bycatch in gillnet fisheries. Conservation Biology, 31(3), pp.686-695.

Plet-Hansen, K.S., Bergsson, H. and Ulrich, C., 2019. More data for the money: Improvements in design and cost efficiency of electronic monitoring in the Danish cod catch quota management trial. Fisheries Research, 215, pp.114-122.

Pott, C. and Wiedenfeld, D.A., 2017. Information gaps limit our understanding of seabird bycatch in global fisheries. Biological Conservation, 210, pp.192-204.

Ramula, S., Öst, M., Lindén, A., Karell, P. and Kilpi, M., 2018. Increased male bias in eider ducks can be explained by sex-specific survival of prime-age breeders. PloS one, 13(4), p.e0195415.

Savina, E., 2018. Gear technical contributions to an ecosystem approach in the Danish bottom set nets fisheries (PhD thesis). DTU Aqua.

Sims, M., Cox, T. and Lewison, R., 2008. Modeling spatial patterns in fisheries bycatch: improving bycatch maps to aid fisheries management. Ecological applications, 18(3), pp.649-661.

Skov, H., 2011. Waterbird populations and pressures in the Baltic Sea (Vol. 550). Nordic Council of Ministers.

Tasker, M.L., Camphuysen, C.J., Cooper, J., Garthe, S., Montevecchi, W.A. and Blaber, S.J., 2000. The impacts of fishing on marine birds. ICES journal of Marine Science, 57(3), pp.531-547.

Torres, L.G., Sagar, P.M., Thompson, D.R. and Phillips, R.A., 2013. Scaling down the analysis of seabird-fishery interactions. Marine Ecology Progress Series, 473, pp.275-289.

van Helmond, A.T.M, Mortensen, L.O., Plet-Hansen, K.S., Ulrich C., Needle C.L., Oesterwind D., KindtLarsen L., Catchpole T., Mangi S., Zimmermann C., Olesen H.J., Bailey N., Bergsson H., Dalskov J., Elson J., Hosken M., Peterson L., McElderry H., Ruiz J., Pierre J.P., Dykstra C., Poos J.J., 2019. Electronic monitoring in fisheries: Lessons from global experiences and future opportunities. Fish and Fisheries. https://doi.org/10.1111/faf.12425

Żydelis, R., Bellebaum, J., Österblom, H., Vetemaa, M., Schirmeister, B., Stipniece, A., Dagys, M., van Eerden, M. and Garthe, S., 2009. Bycatch in gillnet fisheries–An overlooked threat to waterbird populations. Biological Conservation, 142(7), pp.1269-1281. https://doi.org/10.1016/j.biocon.2009.02.025

Żydelis, R., Small, C. and French, G., 2013. The incidental catch of seabirds in gillnet fisheries: A global review. Biological Conservation, 162, pp.76-88. https://doi.org/10.1016/j.biocon.2013.04.002

PAPER 2

ESTIMATING SEABIRD BYCATCH IN DANISH COMMERCIAL GILLNET FISHERIES

Glemarec, G., Frederiksen, M., Kindt-Larsen, L., & Larsen, F.

Manuscript

Abstract

Bycatch of seabirds suffers from a general lack of monitoring in small-scale fisheries in Europe. In the Western Baltic Sea, Danish coastal waters are vital for millions of migrating, breeding or wintering birds, but also constitute essential fishing grounds for hundreds of commercial gillnetters. These vessels, usually small, record fishing effort in logbooks at a coarse scale and, despite suspicions of high seabird bycatch rates, hardly ever report incidental catches. Here, using a 4-step method, fleet-wide seabird bycatch was evaluated in the Danish commercial gillnet fishery of the Belt Seas, with data spanning from 2014 to 2018. First, a bycatch model was developed using fine-scale electronic monitoring data from commercial gillnetters. Second, the fine-scale distribution of gillnet fishing effort was modelled using Automatic Identification System (AIS) data from the commercial fleet. Third, fine-scale bycatch mortality rates were predicted from these data. Fourth, mean seabird bycatch mortality estimates were scaled up to fleet level. Bycatch mortality in the Danish commercial gillnet fleet in the area was estimated to 2265 seabirds annually (95%CI: 1961-2689), raising concerns of sustainability of this fisheries regarding the conservation of some seabird populations.

Abstract

Bycatch of seabirds suffers from a general lack of monitoring in small-scale fisheries in Europe. In the Western Baltic Sea, Danish coastal waters are vital for millions of migrating, breeding or wintering birds, but also constitute essential fishing grounds for hundreds of commercial gillnetters. These vessels, usually small, record fishing effort in logbooks at a coarse scale and, despite suspicions of high seabird bycatch rates, hardly ever report incidental catches. Here, using a 4-step method, fleet-wide seabird bycatch was evaluated in the Danish commercial gillnet fishery of the Belt Seas, with data spanning from 2014 to 2018. First, a bycatch model was developed using fine-scale electronic monitoring data from commercial gillnetters. Second, the fine-scale distribution of gillnet fishing effort was modelled using Automatic Identification System (AIS) data from the commercial fleet. Third, fine-scale bycatch mortality rates were predicted from these data. Fourth, mean seabird bycatch mortality estimates were scaled up to fleet level. Bycatch mortality in the Danish commercial gillnet fleet in the area was estimated to 2265 seabirds annually (95%CI: 1961-2689), raising concerns of sustainability of this fisheries regarding the conservation of some seabird populations.

Keywords: Seabirds, Bycatch, Gillnets, Automatic Identification System (AIS), Electronic monitoring

Running head: Estimate seabird bycatch in gillnets

1. Introduction

Incidental catches in fishing gears (bycatch) are one of the main threat for seabirds worldwide and can lead to population decrease (Croxall et al., 2012; Dias et al., 2019; Tasker, 2000). Set nets (mostly gillnets) are particularly deadly for seabirds, killing globally more than 400,000 individuals each year (Žydelis, 2013). In the European Union (EU), gillnets are the most popular type of fishing gears in number of vessels, typically used by smaller size classes (<12m) (EC, 2020). Yet, despite the requirements to provide estimates on incidental catches of protected species in all fisheries, notably to comply with the requirement of the Marine Strategy Framework Directive to achieve good environmental status (EC, 2008), the magnitude of seabird bycatch in gillnets and the impact of bycatch mortality on seabird populations, are still poorly known in EU waters. Although EU regulations require systematic reporting of seabird bycatch (EU, 2019, 2017), assessing incidental catches of birds in gillnets is made difficult by the relatively low fisheries observer effort in small-scale fisheries (Pott and Wiedenfeld, 2017).

In Denmark, gillnets are widespread fishing gears among commercial and recreational fishers alike. In particular, the Belt Seas (ICES areas IIIc22 and IIIb23), which consist of three narrow straits enclosed between Germany, Denmark and Sweden, are very important fishing grounds for hundreds of small-scale gillnet vessels from all surrounding countries (Figure 1). These relatively shallow waters are also a major migratory route for Palearctic birds and a key region for wintering seabirds. Previous studies pointed out the menace that gillnets represent for seabirds in this area (Bellebaum et al., 2013; Degel et al., 2010; Glemarec et al., 2020; Sonntag et al., 2012; Žydelis et al., 2009).

Historically, fleet-wide total catches in commercial fisheries have been estimated using fishing vessels' logbook and sales notes, complemented with data from official fisheries observers. In small-scale fisheries however, on-board observer coverage is scarce, which results in coarse fishing effort and bycatch estimates resolution. For instance, in the European Union (EU), logbook reports are typically reported at a spatial resolution of 30x30 nautical miles, i.e. the size of an ICES statistical rectangle. In the early 2000's, the introduction of autonomous satellite-tracking systems like AIS or VMS (respectively, Automatic Information System, and Vessel Monitoring System) considerably advanced fisheries research, by reducing temporal resolution from a daily to an hourly period. Using VMS/AIS polls, methods were developed for mapping the fine-scale variations of fishing effort at individual vessel- or at fleet-level (Gloaguen et al., 2015; Le Guyader et al., 2017; Muench et al., 2018; Natale et al., 2015; Russo et al., 2014). In the EU, these systems are only mandatory for large-scale fishing vessels (>15 m for AIS, and >12 m for VMS) (EU, 2013, 2011), limiting their adoption in small-scale fisheries. AIS has nevertheless become increasingly popular with smaller commercial fishing vessels as a means of preventing collision in coastal areas where shipping traffic is intense.

In parallel, in the two last decades, electronic monitoring systems (EM) with closed-circuit television cameras (CCTV) have been trialled in a number of EU fisheries for various purposes (Helmond et al., 2020). These trials included monitoring fishing effort and commercial catches (Mortensen et al., 2017; Ulrich et al., 2015), evaluating compliance and possible change in behaviour following new EU fishing regulations implementation (Kindt-Larsen et al., 2011; Van Helmond et al., 2016), or estimating the magnitude of bycatch of non-target species in small-scale gillnet fisheries (Glemarec et al., 2020; Kindt-Larsen et al., 2016, 2012). EM is considered a reliable and cost-effective method for obtaining long time-series of fine-scale fisheries-dependent data, in particular in metiers where on-board observer coverage is low (Helmond et al., 2020; James et al., 2019; Kindt-Larsen et al., 2011; Plet-Hansen et al., 2019).

In this study, we estimated the variations in time and space of seabird bycatch mortality in fishing nets in the small-scale Danish commercial gillnet fisheries operating in the Belts (ICES area IIIc22), and the Sound (ICES area IIIb23). Six commercial gillnetters were sampled using EM with CCTV between 2014 and 2018, and their entire fishing activity, including bycatch of seabirds was recorded. During the same period, AIS data from all the gillnet vessels registered in the area were collected and analysed to obtain fine-scale estimates of the commercial fishing effort in the area. We explored the spatial and temporal variability of seabird bycatch rates in relation to operational and ecological factors. Combining logbooks sales notes and AIS data, we used this knowledge to estimate seabird bycatch mortality at fleet-level in the Danish commercial gillnet fleet in ICES areas IIIc22 and IIIb23.

2. Material and Methods

2.1. Data

This research focused on ICES areas IIIc22 and IIIb23, jointly referred to as the Belt Seas (Figure 1), for the period 2014 to 2018. Four sources of data were used (Table 1): (i) long-term EM data of a sample of six Danish commercial gillnet vessels registered in the study area, (ii) raw AIS polls from 48 Danish commercial gillnet vessels registered in the study area, (iii) corresponding logbooks, and (iv) sales notes of 435 active Danish gillnetters registered in the area during the study period. A vessel was considered active if it had registered at least one day at sea in a calendar year.

Table 1: Summary of the raw data used in the study. Mean aggregated year/quarter data for the period 2014-2018.

		EM data			AIS data		Logbook data	
ICES Area	Quarter	Vessel count	Observed days at sea	Seabird bycatch	Vessel count	Registered days at sea	Vessel count	Registered days at sea
IIIc22	1	2.2	56.2	16.2	11.8	196.8	134.4	1972.8
	2	2.8	106.2	17.4	9	120.4	153	2430.8
	3	2.25	85.75	17.5	10.6	181	134.2	1660
	4	2	93	13.5	10.6	159.8	141.6	1670.8
IIIb23	1	2	71.4	34	6.4	110.2	49.4	643.6
	2	2	64.2	8.4	4.2	87.4	45.2	680.6
	3	1.6	56.8	19.6	6.6	132.8	53.4	891.6
	4	1.2	35.4	58	9.6	174	63.2	1129.6

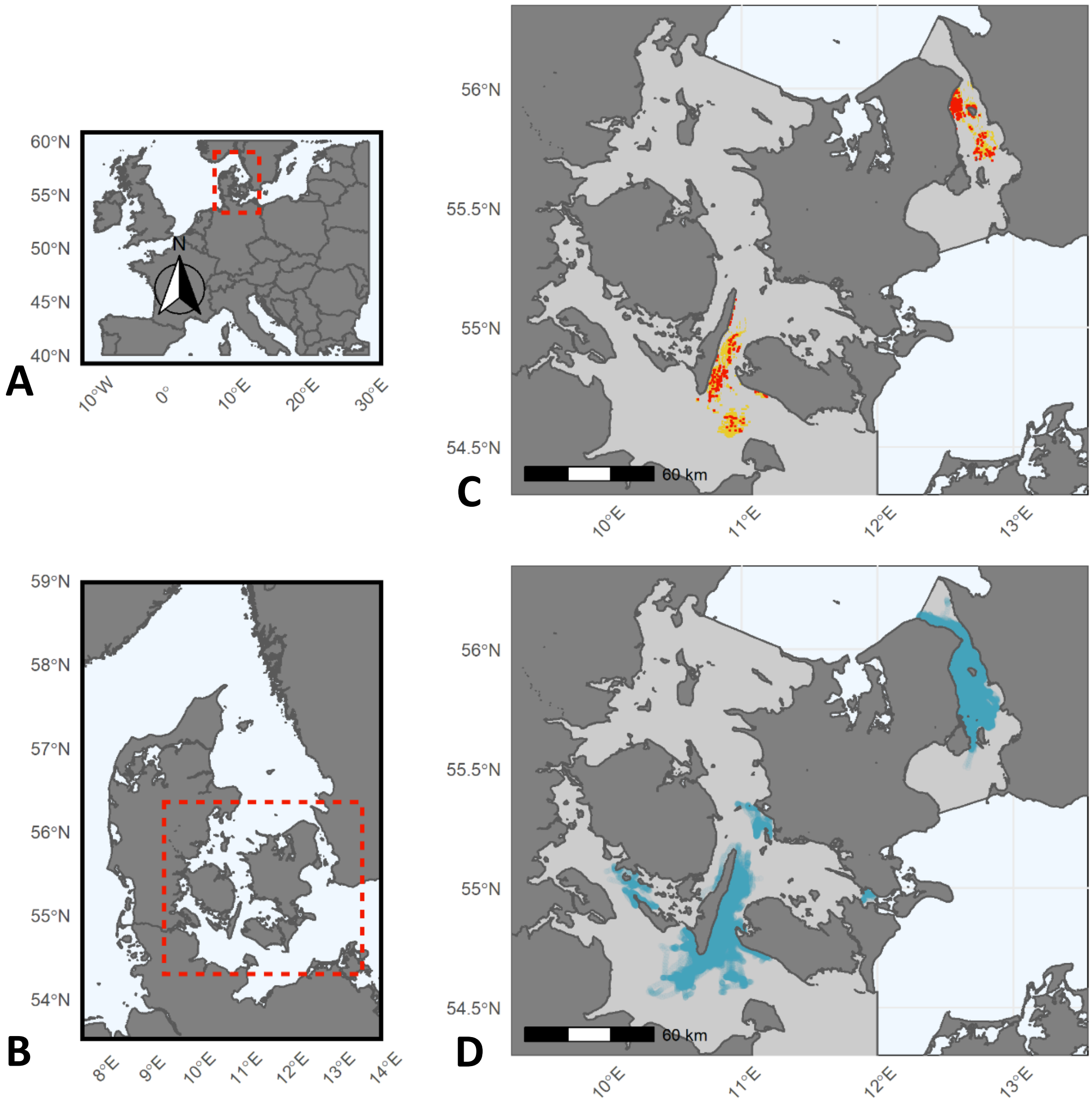

Figure 1: Comparison between the electronic monitoring sampling and the distribution of available AIS data of Danish commercial gillnetters in ICES statistical areas IIIc22 and IIIb23 (in light grey). Data aggregated from 2014 to 2018. Denmark is located within Europe (A), and the study area is positioned within Denmark (B). C. exact haul positions (yellow) and bird bycatch positions (red). D. Positions of each AIS poll (blue).

Seabird bycatch data in commercial gillnetters were collected using EM with CCTV (detailed methodology in Glemarec et al. (2020)). In total, six vessels were monitored between 2014 and 2018, during which 892 seabirds were identified as bycatch (Table 1). Fishing net locations were defined as the middle points between the GPS coordinates of start and stop positions of each individual haul. Operational factors (mesh size, soak time, net length, number of nets per fishing trip), ecological factors (depth, distance to shore, date), and catch-related information (main target species, number of birds captured per haul) completed the dataset. Additionally, each bird taken in the net was identified to the lowest taxon possible, down to species level.

Raw AIS data received by shore-based stations were provided by the Danish Maritime Authority and used in accordance with the conditions for the use of Danish public data. Each AIS poll, transmitted at a fixed ping interval (typically 5 minutes), contained information on vessel name (using the unique identifier MMSI: Maritime Mobile Service Identity), associated to instant records of the geographical position (longitude/latitude), speed, and course, as well as additional maritime safety related information (e.g. type of vessel, destination, estimated time of arrival). Using the EU fleet register (EC, 2020), the raw AIS dataset was restricted to fishing vessels whose primary or secondary gear were set nets (gillnet or trammel net), and whose harbour was situated in the study area. A few data points on land or outside the boundaries of the study area were excluded. Data exploration revealed few strong outliers in the speed variable for three of the fishing vessels (instantaneous speed >45 knots), which were removed. Self-added information on depth, distance to shore and average speed complemented the dataset. Average speed was calculated as the distance divided by the duration between two consecutive AIS messages, and usually differed from instantaneous speed. In addition, messages where depth was null were discarded, as the corresponding points were almost all concentrated in and around harbours, where no fishing normally occurs. The resulting dataset contained more than 318,000 AIS messages, with information on 48 Danish commercial gillnet vessels, henceforth called AIS dataset.

Official logbook and sales notes data from every Danish commercial gillnetter in ICES IIIb23 and IIIc22 (obtained from the Danish Fisheries Agency and processed by DTU Aqua) provided the nominal weight of each landed species per fishing trip, associated to date, metier, fishing gear and mesh size. A fishing trip was defined as one registered day at sea, under the assumption that, in this small-scale coastal gillnet fishery, daily catches are landed daily.

2.2. Method

The main steps of the analysis consisted of: (i) identifying the operational and ecological factors that explain the observed level of seabird bycatch in gillnets using a statistical model (EM dataset); (ii) predicting the fine-scale distribution of the fishing activity of the fleet (AIS dataset); (iii) applying the bycatch model to the fine-scale effort data to obtain mean bycatch rate estimates, and (iv) scaling up these bycatch rates to fleet level to estimate fleet-wide seabird bycatch mortality (Figure 3). Data management and statistical analysis were conducted in R (R Core Team, 2020).

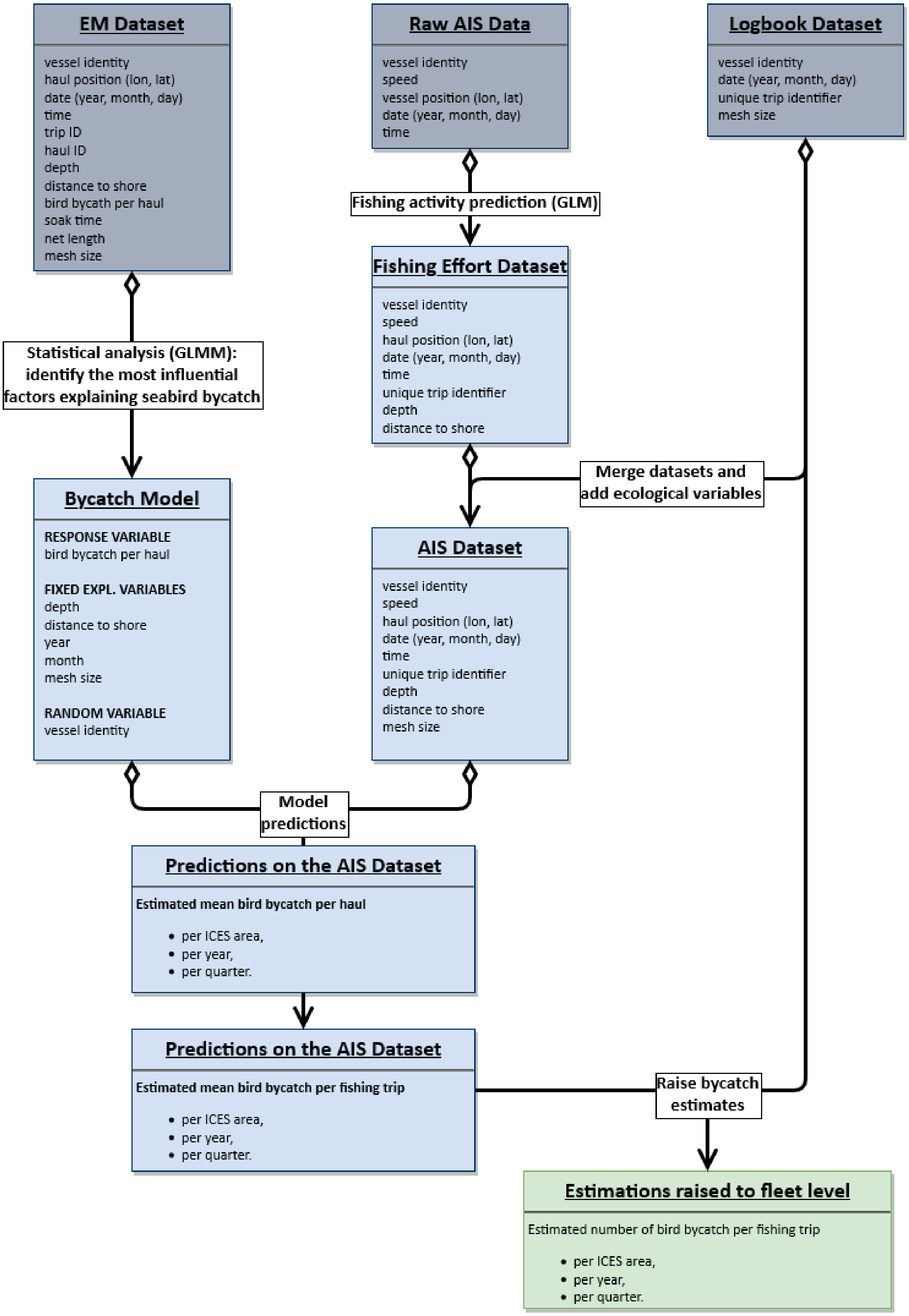

Figure 20: Simplified workflow representing the method used to calculate fleet-wide seabird bycatch estimates in commercial gillnets in Denmark.

2.2.1. Seabird bycatch model

The goal of the statistical analysis was to explore the variations of seabird bycatch in gillnets and, ultimately, to use the model predictions to raise bird bycatch estimates to fleet level. Since the study design was clearly hierarchical, a mixed modelling approach was preferred (Zuur et al., 2009). Candidate generalised linear mixed models (GLMM) were fitted with the R package spaMM (Rousset and Ferdy, 2014) and compared using the conditional Akaike Information Criterion cAIC (Vaida and Blanchard, 2005). Models with the lowest cAIC score were considered the best at fitting the data, however, models with a difference in cAIC smaller than 6 units were deemed equally good (Harrison et al., 2018). The response variable seabird bycatch, defined as the number of birds captured per haul, was a count and, as such, Poisson and negative-binomial response distributions were initially considered. Following the protocol in Zuur et al. (2009), a beyond optimal model, including all relevant fixed variables and interactions, was created as a starting point. Competing models were compared using their cAIC scores. The random effect structure was selected first by varying the random variables only (using restricted maximum likelihood to integrate over the fixed effect parameters). Once the best random effect structure was found, the optimal fixed effect structure was selected by varying the fixed variables only (using maximum likelihood). The following explanatory variables were used for the fixed structure: the product of soak time and net length (fishing effort), number of nets set per fishing trip (NN), mesh size (mesh), fishing depth (depth), distance to the nearest coastline (d2shore), their interaction (depth * d2shore), month (m), and year (y). To account for the experimental design, the random component structure was fit as random-intercept only model (i.e. no random slope), and included the unique vessel identifier (id) and the parameter year (y).

The goodness-of-fit of the best-fitting model was ensured using simulation-based tests in the R package DHARMa (Hartig, 2019). Because of the high number of zeros in the response, dispersion was a potential issue in these models. Therefore, overdispersion, and the special case of zero-inflation, as well as heteroscedasticity were tested for each candidate model. In addition, to control for possible misspecification, the residuals were plotted against each predictor in and out of the model. Finally, the models were tested for temporal and spatial autocorrelation (respectively Durbin-Watson, and Moran's I tests).

2.2.2. Fishing activity detection in the AIS dataset

Various methods have been developed to differentiate fishing from non-fishing events from the satellite tracks of individual vessels, by decomposing speed profiles to infer vessels' activity (Le Guyader et al., 2017; Natale et al., 2015), by using probabilistic modelling approaches (Muench et al., 2018), or by applying data mining techniques (Gloaguen et al., 2015). Here, the speed density profiles in the AIS dataset were examined to detect fishing activity and usually showed a clear tri-modal distribution of the speed for most vessels. In towed gear fisheries, a vessel's activity can be retrieved by decomposing the multimodal distributions of speed densities; fishing behaviour corresponds to relatively low to medium speeds, while high speeds relate to steaming (Bastardie et al., 2010). Based on this principle, mixture models using semi-automatic hierarchical clustering are able to classify data points in terms of fishing behaviour (Le Guyader et al., 2017). However, the accuracy of such a procedure is typically lower for passive gears, and in particular lower for gillnets, than for active gears (Natale et al., 2015). Instead, statistical models can attain much higher accuracy for fisheries with complicated patterns such as gillnet fisheries (Muench et al., 2018).

In the Danish AIS dataset, the probability that an AIS poll corresponded to a fishing event (i.e. the retrieval of a fishing net, or a haul) was modelled as a generalised linear model (GLM) with a binomial outcome. The explanatory variables included nine metrics. Five speed metrics were derived directly from the AIS data, namely speed, its square, the minimum average speed between two consecutive polls, its square, and a binary variable speed range. The variable speed range took a value of 1 when the speed of the current AIS poll did not diverge by more than 0.25 knots from the mean speed of the lag, current and lead AIS polls (and zero otherwise). Additional metrics included ecological (depth) and operational metrics (month, time of the day and the interaction between gross tonnage and vessel length). Models using the same structure, but different link functions were tested and the model generating the lowest AIC score was selected (Crawley, 2007). The model quality and goodness-of-fit were evaluated by examining the distribution of the deviance residuals, the variance of the residuals and the Cook's distance values (Thomas, 2017). Model validation was done on a subset of the AIS data including only the vessels monitored with EM, for which the exact GPS coordinates of each haul were known.

The accuracy of the predictions depended on the ability of the model to identify accurately the AIS polls corresponding to a haul (true positive) and to classify the AIS polls corresponding to a non-fishing activity correctly (true negative). The optimal cut-off threshold value was obtained using a receiver operating characteristics (ROC) analysis. A ROC curve plots the true positive rates of the predictions (sensitivity) against the false positive rates of the predictions (1 – specificity) at various threshold values. The Youden's J statistic (Youden, 1950) is the value of the threshold that maximizes the true positive rate and minimises the false negative rate. The implicit goal of this analysis was to achieve a true positive detection rate of at least 80%, without raising the false positive detection rate above 30%. Mean and confidence intervals of the different statistics were estimated using the bootstrap method. Specifically, the data was split between a training and a test dataset; 50% of the AIS polls were randomly drawn to fit the GLM, and the other half was kept for prediction; this procedure was repeated 10,000 times.

Finally, the overall accuracy of the predictive model using the Youden's J statistic as the cut-off threshold value was evaluated using a K-fold cross validation approach. The method is very similar to the one described above, and consisted of splitting the dataset randomly into two equal parts: a training set for fitting the model and a test set for validating the predictions. To control for sampling bias, the procedure was repeated 20 times, a number judged sufficiently big based on similar previous studies (Muench et al., 2018; O'Farrell et al., 2017). In turn, an individual haul was defined as a sequence of at least two consecutive AIS polls identified as fishing events within the same fishing trip. Finally, these predicted hauls were controlled for misspecifications, notably in terms of total length and total duration of hauling operation. Specifically, in the study area, net fleets are normally several hundred meters long and a hauling phase takes usually between 20 and 30 minutes and, on rare occasions, up to 3 hours. Practically, hauls whose length was shorter than 100 meters or longer than 4000 meters were excluded, as well as hauls whose duration was below 5 minutes or above 3.1 hours, as these could have been faulty predictions.

2.2.3. Spatio-temporal variations in seabird bycatch and extrapolation to fleet level

Seabird bycatch estimates in the part of the gillnet fleet carrying AIS, measured as the expected number of birds captured per haul, were calculated by applying the bycatch model to the fine-scale fishing effort distribution from the AIS dataset (function predict in R). The resulting estimates were summed per fishing trip to obtain the predicted bycatch of seabirds per day at sea. The mean stratified bycatch per unit effort estimates (number of birds per quarter per ICES area) and the 95% confidence interval were calculated using the bootstrap method (10,000 repetitions). Finally, fleet-wide bycatch estimates and the associated 95% confidence intervals were calculated by multiplying the stratified mean bycatch per unit effort with the number of fishing trips registered in official fishing logbooks.

3. Results

3.1. Seabird bycatch model

The model selection favoured the parsimonious model showed in Table 2. The variable depth was the main driver of the seabird bycatch model. Moreover, distance to shore and fishing effort were important contributors to the response. However, according to the model, mesh size was only marginally influencing seabird bycatch. Finally, the temporal variable month, negatively correlated to the response, was particularly influential between April and October, when the risk of bycatch is the lowest.

Table 4: Results of the general and parsimonious GLMM estimations, showing the effect of the parameters on seabird bycatch per haul.

	Full model			Parsimonious model		
Parameters	Coefficient	Standard error	t-value	Coefficient	Standard error	t-value
Intercept	-2.508571	0.553868	-4.52919	-3.2385109	0.4798390	-6.7492000
Fishing effort	0.118747	0.053062	2.23788	0.1219849	0.0544790	2.2391000
Number of soaked nets	-0.026322	0.043513	-0.60493	—	—	—
Mesh size	-0.002704	0.002432	-1.1122	0.0003884	0.0024160	0.1608000
Depth	1.447006	0.163598	8.84487	1.2065876	0.1065430	11.3249000
Distance to shore	-0.889394	0.171027	-5.20031	-0.7126801	0.1430840	-4.9808000
Depth x Distance to shore	0.282561	0.169377	-0.28435	—	—	—
February	-0.095922	0.337338	-0.86923	-0.2664503	0.3400390	-0.7836000
March	-0.297099	0.341795	-3.64238	-0.4399462	0.3398140	-1.2947000
April	-1.347329	0.369903	-5.16354	-1.4177630	0.3689300	-3.8429000
May	-3.892466	0.753837	-4.11424	-3.8180406	0.7574960	-5.0403000
June	-4.272013	1.038348	-4.39382	-4.1596607	1.0402450	-3.9987000
July	-1.824618	0.415269	-4.60959	-1.6745435	0.4147210	-4.0378000
August	-1.809714	0.392598	-3.85495	-1.6566969	0.3975440	-4.1673000
September	-1.460100	0.378760	-5.36135	-1.2975839	0.3744270	-3.4655000
October	-2.305587	0.430039	-1.0334	-2.1892029	0.4255760	-5.1441000
November	-0.323861	0.313395	-2.47769	-0.1171584	0.3107030	-0.3771000
December	-0.854217	0.344764	-0.08727	-0.7842002	0.3424750	-2.2898000
Year 2015	-0.019857	0.227524	-0.7205	—	—	—
Year 2016	-0.163576	0.227030	-0.27562	—	—	—
Year 2017	-0.062289	0.225995	-2.80649	—	—	—
Year 2018	-0.738455	0.263124	1.66823	—	—	—
No. observations	11 564			11 564		
cAIC	2884.847			2878.559		
logLik	-1419.424			-1424.41		

3.2. Fishing effort model

The algorithm used for predicting the fishing activity from the AIS polls performed very well. On the reduced sample, containing only the vessels monitored with EM for which the detailed fishing activity was known, the binomial GLM predicted the activity (*fishing* or *not fishing*) with an overall accuracy of 81% (AUC = 0.85; Kappa = 0.61), using a Youden's *J* statistic of 0.24 (95% CI: 0.20 - 0.25; K = 20). The true positive rate of the model reached 96% and generated a false negative rate of 26%.

3.3. Seabird bycatch estimates

Mean quarterly bycatch rate estimates were stratified per ICES area (Table 3). The estimated annual seabird bycatch rate for the whole study area was 0.24 seabirds per fishing trip (95%CI: 0.21-0.27; 10,000 bootstraps). Based on these estimated rates, the expected annual number of bird bycatch in the entire fleet reached 2265 seabirds (95%CI: 1961-2689; 10,000 bootstraps). Estimated bycatch numbers varied importantly both between and within years, and between ICES area IIIc22 and IIIb23.

Table 5: Mean fleet-wide quarterly seabird bycatch estimates based on the predicted fishing effort of Danish commercial using AIS (aggregated data from 2014 to 2018).

		AIS data	**Bycatch estimates (fleet)**		
ICES Area	Quarter	Mean number of seabird per day at sea	Mean seabird bycatch	Low 95%	High 95%
IIIc22	1	0.22	434	371	533
	2	0.02	60	41	107
	3	0.05	89	77	105
	4	0.17	276	211	376
IIIb23	1	0.34	218	193	248
	2	0.03	18	14	23
	3	0.11	94	78	112
	4	0.95	1078	976	1186

4. Discussion

Effective fisheries management requires a precise knowledge of the spatial and temporal distribution of the fishing effort and how this relates to the catch of both target and non-target species. To that end, this paper estimated fleet-wide temporal and spatial variations of seabird bycatch in gillnets in the Danish commercial fishery of the Belt Seas (ICES area IIIc22 and IIIb23), for the period 2014-2018. Four complementary datasets were combined: the fine-scale records of the fishing activity from six commercial gillnetters surveyed with electronic monitoring (EM), the collection of all the AIS polls emitted by commercial gillnet vessels active in the study area, the logbooks and the sales notes from the entire fleet. Although spatially limited, these results are important for both resource managers and conservation scientists, as they show that commercial gillnetters capture a substantial number of seabirds in Inner Danish waters. In the fishery examined here, total seabird bycatch averaged 2265 seabirds (95%CI: 1961-2689; 10,000 bootstraps) between 2014 and 2018. Yearly mean bycatch rates approximated 0.24 birds per fishing trip (95%CI: 00.21-0.27; 10,000 bootstraps), which is three times higher than what was reported recently from the nearby Norwegian coastal gillnet fishery (Bærum et al., 2019). These estimates however, are in line with most of the other bycatch studies in the Baltic Sea, reporting high bycatch rates in coastal gillnet fisheries for the whole region (Žydelis, 2013).

Using EM data, incidental catches of birds were modelled as a stochastic (random) process, i.e. the response variable was dependant only on the operational and ecological factors characteristic of the fishery. Although bycatch also arguably depends on the seabirds distribution in the area where fishing occurs (Clay et al., 2019), assuming complete randomness nevertheless allowed identifying the key factors influencing bird bycatch in gillnets in the study area. Except for mesh size, all the parameters in the parsimonious model contributed importantly to the predictions. Environmental parameters were among the most important contributors to bycatch. Depth was positively correlated to the response, as opposed to distance to shore. This is hardly surprising as most seabirds captured in the study area are seaducks, most of which common eiders (Glemarec et al., 2020). These coastal birds often forage on mussel banks in relatively shallow waters, yet keeping at a distance from the land. Expectedly, increasing fishing effort, corresponding to longer nets soaked for extended durations, resulted in higher seabird bycatch. Additionally, according to the model, larger meshes, which are designed to entangle rather than gill target species, such as the ones used to capture turbot Scophthalmus maximus or lumpsucker Cyclopterus lumpus, increased seabird bycatch. However, mesh size was only marginally contributing to the response in the models. In the EM dataset used to parametrise the model, the fishers used only few different mesh sizes during their fishing operations. Therefore, the contribution of mesh size to bycatch could have been blurred by the low variability of mesh sizes registered with EM. Additionally, the low t-value associated to the parameter mesh suggests its lack of significance for the model. Nevertheless, larger meshes are usually associated with a higher bycatch rate. For instance, mean seabird bycatch rates in the lumpsucker gillnet fisheries in Norway and in Denmark are estimated at about 0.40 individuals per fishing trip (Christensen-Dalsgaard et al., 2019), i.e. more than 60% higher than the average bycatch rates estimated in this study. Finally, the coefficient estimates associated to the variable month were much lower between April and October, compared to the coldest months of the year between November and March.

In the absence of a standardised data collection scheme to obtain bycatch and fine-scale effort data for the whole Danish small-scale gillnet fisheries, this study utilised data sources that arguably may not be fully representative of the entire gillnet fleet. Any study attempting to draw conclusions from a sample of the population assumes representativity (Kruskal and Mosteller, 1979); here the representativity of the electronically monitored vessels with respect to bycatch rates of the fishing fleet and the representativity of the vessels carrying AIS with respect to the spatio-temporal variations of fishing effort. Regarding the AIS dataset, vessels with AIS represented on average 8.9% of the fleet in number of vessels, and accounted for approximately the same amount of fishing effort, i.e. 10.5% of the nominal fishing effort (measured as days at sea). The AIS data, although covering a large part of the exploited fishing grounds in the study area, could not fully reflect the fishing intensity in some locations. For example, in a study assessing the magnitude of seabird bycatch in the small gillnet fleet fishing around the island of Ærø, located in ICES IIIc22, the authors reported that most of the fishing effort was concentrated directly south of the island (Degel et al., 2010). Yet, the records of the AIS polls from commercial fishers between 2014 and 2018 showed no ping in that area. Often, skippers operating in areas where shipping traffic is dense choose to use AIS for safety reasons, whereas others fishers working in less crowded waters, e.g. around Ærø island, might not need such equipment. Ultimately, it was assumed that the operational characteristics of the vessels using AIS were similar to those of the vessels without, and therefore were applicable to the entire fleet. Likewise, the representativity of the EM data could be discussed. The installation of

EM systems on-board gillnet vessels was done on a voluntary basis, so it cannot be excluded that the selected skippers were more aware of – and possibly more cautious about – the risk of bird bycatch than average. Besides, the presence of cameras on-board could have influenced fishing behaviour (Mangi et al., 2015), leading to biased bycatch estimates. However, admitting this was the case, the risk of bird bycatch calculated from the EM vessels would have been under-, rather than over-estimated. Accordingly, the bycatch rate estimates presented here are very likely conservative.

With this in mind, raising bycatch predictions to fleet level showed clear temporal trends over the whole study area (Table 3). These estimates were largely consistent with the patterns observed in previous studies in the region (Bellebaum et al., 2013; Sonntag et al., 2012; Žydelis, 2013). Estimated total bycatch numbers were generally relatively low in quarters 2 and 3 (from 1st April to 30th September), averaging 261 birds annually for the whole fleet. Conversely, yearly incidental catches were more than 7 times higher in quarters 4 and 1 (from 1st October to 31st March), summing up to 2006 bycatch on average.

In the Baltic, the populations of red-necked grebe (Podiceps grisegena), as well as the wintering populations of common eider (Somateria mollissima), velvet scoter (Melanitta fusca) and common scoter (Melanitta nigra) are all classified as regionally endangered and bycatch in gillnets is one major concern for these populations possible decline (Kontula and Haldin, 2013). A recurring question in seabird bycatch studies is to establish a link between additional fisheries-induced mortality and a potential negative effect on population dynamics (Le Bot et al., 2018; Tasker, 2000). In this study, no attempt was made to estimate the effect of bycatch on seabird populations, as too many elements were missing from the datasets. For example, knowing the provenance of the deceased birds, i.e. which flyway population they belong to, is fundamental to estimate the impact of bycatch mortality at population level. However, establishing trends over long periods is also essential to understand how fisheries can influence the demographics of seabird populations. In this regard, the use of EM in small-scale fisheries has shown a great potential to access both fishing effort and incidental catches of protected species. Problematically, EM implementation in the European Union is slowed down by the internal competition between neighbouring Member States (Plet-Hansen et al., 2017). Alternative methods exist, including on-board observers and fisher-reported data, but they also raise concerns of reliability (Mangi et al., 2015). Additionally, collecting satellite-tracking data from commercial should be encouraged as it allows accessing the fine-scale variations of the fishing activity of a large number of vessels, and sometimes the whole fleet. In large-scale fisheries, VMS data has already brought fisheries scientists crucial insight on the fine-scale location of the fishing grounds, allowing them to measure the impact of these fisheries on the ecosystem, including bycatch of seabirds (e.g. Soriano-Redondo et al. (2016)). Nevertheless, in small-scale coastal gillnet fisheries, the fishing activity, which usually consists of a repetition of short hauls, can be hard to capture using VMS whose ping frequency is typically 2 hours. As such, AIS is more adapted for small-scale fisheries (one ping every 5 minutes on average). Nonetheless, as AIS can be tampered with or turned off, these data should to be treated with caution and thoroughly checked before use.

Still, as small-scale gillnet fisheries are a known threat to seabird populations in the region, continuing collecting bycatch data in these fisheries, predicting spatial and temporal variations of fishing effort and raising the numbers of dead birds to fleet level, as was done here, can bring indispensable material to fisheries managers and seabird biologists alike in order to assess the potential impact of bycatch on population abundance. For many years now in Northern Europe, important fish stocks have been monitored, and their population dynamics modelled, allowing the fishing industry to thrive while maintaining the resource at sustainable levels. Likewise, as pointed out by Le Bot et al. (2018), the collection of reliable incidental bycatch data over extended periods is a key to understanding seabird-fisheries interactions and is urgently required to advance conservation goals and achieve the long-term sustainability of EU fisheries.

Declarations of interest

None.

References

Bærum, K.M., Anker-Nilssen, T., Christensen-Dalsgaard, S., Fangel, K., Williams, T., Vølstad, J.H., 2019. Spatial and temporal variations in seabird bycatch: Incidental bycatch in the Norwegian coastal gillnet-fishery. PLoS ONE 14. https://doi.org/10/ggkz9g

Bastardie, F., Nielsen, J.R., Ulrich, C., Egekvist, J., Degel, H., 2010. Detailed mapping of fishing effort and landings by coupling fishing logbooks with satellite-recorded vessel geo-location. Fish. Res. 106, 41–53. https://doi.org/10/c3s744

Bellebaum, J., Schirmeister, B., Sonntag, N., Garthe, S., 2013. Decreasing but still high: bycatch of seabirds in gillnet fisheries along the German Baltic coast. Aquat. Conserv. Mar. Freshw. Ecosyst. 23, 210–221. https://doi.org/10/f4wfmn

Christensen-Dalsgaard, S., Anker-Nilssen, T., Crawford, R., Bond, A., Sigurðsson, G.M., Glemarec, G., Hansen, E.S., Kadin, M., Kindt-Larsen, L., Mallory, M., Merkel, F.R., Petersen, A., Provencher, J., Bærum, K.M., 2019. What's the catch with lumpsuckers? A North Atlantic study of seabird bycatch in lumpsucker gillnet fisheries. Biol. Conserv. 240, 108278. https://doi.org/10/ggk9gb

Clay, T.A., Small, C., Tuck, G.N., Pardo, D., Carneiro, A.P.B., Wood, A.G., Croxall, J.P., Crossin, G.T., Phillips, R.A., 2019. A comprehensive large-scale assessment of fisheries bycatch risk to threatened seabird populations. J. Appl. Ecol. 56, 1882–1893. https://doi.org/10/ggmpg8

Crawley, M.J., 2007. Generalized Linear Models, in: The R Book. John Wiley & Sons, Ltd, pp. 511–526. https://doi.org/10.1002/9780470515075.ch13

Croxall, J.P., Butchart, S.H.M., Lascelles, B., Stattersfield, A.J., Sullivan, B., Symes, A., Taylor, P., 2012. Seabird conservation status, threats and priority actions: a global assessment. Bird Conserv. Int. 22, 1–34. https://doi.org/10/ggk25d

Degel, H., Petersen, I.K., Holm, T.E., Kahlert, J., 2010. Fugle som bifangst i garnfiskeriet. Estimat af utilsigtet bifangst af havfugle igarnfiskeriet i området omkring Ærø (No. DTU Aqua-rapport nr. 227-2010). DTU Aqua. Institut for Akvatiske Ressourcer, Charlottenlund.

Dias, M.P., Martin, R., Pearmain, E.J., Burfield, I.J., Small, C., Phillips, R.A., Yates, O., Lascelles, B., Borboroglu, P.G., Croxall, J.P., 2019. Threats to seabirds: A global assessment. Biol. Conserv. 237, 525–537. https://doi.org/10.1016/j.biocon.2019.06.033

EC, 2020. Database - Eurostat [WWW Document]. URL https://ec.europa.eu/eurostat/data/database (accessed 3.11.20).

EC, 2008. Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive) (Text with EEA relevance), 164.

EU, 2019. Commission Delegated Decision (EU) 2019/910 of 13 March 2019 establishing the multiannual Union programme for the collection and management of biological, environmental, technical and socioeconomic data in the fisheries and aquaculture sectors, OJ L.

EU, 2017. Regulation (EU) 2017/1004 of the European Parliament and of the Council of 17 May 2017 on the establishment of a Union framework for the collection, management and use of data in the fisheries sector and support for scientific advice regarding the common fisheries policy and repealing Council Regulation (EC) No 199/2008, OJ L.

EU, 2013. Regulation (EU) No 1380/2013 of the European Parliament and of the Council of 11 December 2013 on the Common Fisheries Policy, amending Council Regulations (EC) No 1954/2003 and (EC) No 1224/2009 and repealing Council Regulations (EC) No 2371/2002 and (EC) No 639/2004 and Council Decision 2004/585/EC, OJ L.

EU, 2011. Commission Directive 2011/15/EU of 23 February 2011 amending Directive 2002/59/EC of the European Parliament and of the Council establishing a Community vessel traffic monitoring and information system. Text with EEA relevance, 049.

Glemarec, G., Kindt-Larsen, L., Lundgaard, L.S., Larsen, F., 2020. Assessing seabird bycatch in gillnet fisheries using electronic monitoring. Biol. Conserv. 243, 108461. https://doi.org/10/ggmrwr

Gloaguen, P., Mahévas, S., Rivot, E., Woillez, M., Guitton, J., Vermard, Y., Etienne, M.-P., 2015. An autoregressive model to describe fishing vessel movement and activity. Environmetrics 26, 17–28. https://doi.org/10/f6v9vs

Harrison, X.A., Donaldson, L., Correa-Cano, M.E., Evans, J., Fisher, D.N., Goodwin, C.E., Robinson, B.S., Hodgson, D.J., Inger, R., 2018. A brief introduction to mixed effects modelling and multi-model inference in ecology. PeerJ 6, e4794. https://doi.org/10/gdh936

Hartig, F., 2019. DHARMa: residual diagnostics for hierarchical (multi-level/mixed) regression models. R Package Version 024.

Helmond, A.T.M. van, Mortensen, L.O., Plet-Hansen, K.S., Ulrich, C., Needle, C.L., Oesterwind, D., Kindt-Larsen, L., Catchpole, T., Mangi, S., Zimmermann, C., Olesen, H.J., Bailey, N., Bergsson, H., Dalskov, J., Elson, J., Hosken, M., Peterson, L., McElderry, H., Ruiz, J., Pierre, J.P., Dykstra, C., Poos, J.J., 2020. Electronic monitoring in fisheries: Lessons from global experiences and future opportunities. Fish Fish. 21, 162–189. https://doi.org/10/ggk3qd

James, K.M., Campbell, N., Viðarsson, J.R., Vilas, C., Plet-Hansen, K.S., Borges, L., González, Ó., van Helmond, A.T.M., Pérez-Martín, R.I., Antelo, L.T., Pérez-Bouzada, J., Ulrich, C., 2019. Tools and Technologies for the Monitoring, Control and Surveillance of Unwanted Catches, in: Uhlmann, S.S., Ulrich, C., Kennelly, S.J. (Eds.), The European Landing Obligation: Reducing Discards in Complex, Multi-Species and Multi-Jurisdictional Fisheries. Springer International Publishing, Cham, pp. 363–382. https://doi.org/10.1007/978-3-030-03308-8_18

Kindt-Larsen, L., Berg, C.W., Tougaard, J., Sørensen, T.K., Geitner, K., Northridge, S., Sveegaard, S., Larsen, F., 2016. Identification of high-risk areas for harbour porpoise Phocoena phocoena bycatch using remote electronic monitoring and satellite telemetry data. Mar. Ecol. Prog. Ser. 555, 261–271. https://doi.org/10/f849c3

Kindt-Larsen, L., Dalskov, J., Stage, B., Larsen, F., 2012. Observing incidental harbour porpoise Phocoena phocoena bycatch by remote electronic monitoring. Endanger. Species Res. 19, 75–83. https://doi.org/10/ggk2cp

Kindt-Larsen, L., Kirkegaard, E., Dalskov, J., 2011. Fully documented fishery: a tool to support a catch quota management system. ICES J. Mar. Sci. 68, 1606–1610. https://doi.org/10/bm2pkd

Kontula, T., Haldin, J., 2013. HELCOM Red List of Baltic Sea species in danger of becoming extinct (No. 140), Baltic Marine Environment Protection Commission. Baltic Marine Environment Protection Commission – Helsinki Commissi.

Kruskal, W., Mosteller, F., 1979. Representative sampling, II: Scientific literature, excluding statistics. Int. Stat. Rev. Int. Stat. 111–127. https://doi.org/10/b86fbt

Le Bot, T., Lescroël, A., Grémillet, D., 2018. A toolkit to study seabird–fishery interactions. ICES J. Mar. Sci. 75, 1513–1525. https://doi.org/10/gfjj45

Le Guyader, D., Ray, C., Gourmelon, F., Brosset, D., 2017. Defining high-resolution dredge fishing grounds with Automatic Identification System (AIS) data. Aquat. Living Resour. 30, 39. https://doi.org/10/gch5hg

Mangi, S.C., Dolder, P.J., Catchpole, T.L., Rodmell, D., de Rozarieux, N., 2015. Approaches to fully documented fisheries: practical issues and stakeholder perceptions. Fish Fish. 16, 426–452. https://doi.org/10/ggkz8b

Mortensen, L.O., Ulrich, C., Olesen, H.J., Bergsson, H., Berg, C.W., Tzamouranis, N., Dalskov, J., 2017. Effectiveness of fully documented fisheries to estimate discards in a participatory research scheme. Fish. Res. 187, 150–157. https://doi.org/10/ggnq25

Muench, A., DePiper, G.S., Demarest, C., 2018. On the precision of predicting fishing location using data from the vessel monitoring system (VMS). Can. J. Fish. Aquat. Sci. 75, 1036–1047. https://doi.org/10/ggkz8v

Natale, F., Gibin, M., Alessandrini, A., Vespe, M., Paulrud, A., 2015. Mapping fishing effort through AIS data. PloS One 10. https://doi.org/10/f7sfgs

O'Farrell, S., Sanchirico, J.N., Chollett, I., Cockrell, M., Murawski, S.A., Watson, J.T., Haynie, A., Strelcheck, A., Perruso, L., 2017. Improving detection of short-duration fishing behaviour in vessel tracks by feature engineering of training data. ICES J. Mar. Sci. 74, 1428–1436. https://doi.org/10/ggm2k6

Plet-Hansen, K.S., Bergsson, H., Ulrich, C., 2019. More data for the money: Improvements in design and cost efficiency of electronic monitoring in the Danish cod catch quota management trial. Fish. Res. 215, 114–122. https://doi.org/10/ggk2d4

Plet-Hansen, K.S., Eliasen, S.Q., Mortensen, L.O., Bergsson, H., Olesen, H.J., Ulrich, C., 2017. Remote electronic monitoring and the landing obligation–some insights into fishers' and fishery inspectors' opinions. Mar. Policy 76, 98–106. https://doi.org/10/f9qvj8

Pott, C., Wiedenfeld, D.A., 2017. Information gaps limit our understanding of seabird bycatch in global fisheries. Biol. Conserv. 210, 192–204. https://doi.org/10/gbqqht

R Core Team, 2020. R: A language and environment for statistical computing.

Rousset, F., Ferdy, J., 2014. Testing environmental and genetic effects in the presence of spatial autocorrelation. Ecography 37, 781–790. https://doi.org/10/ggnr5k

Russo, T., D'Andrea, L., Parisi, A., Cataudella, S., 2014. VMSbase: An R-Package for VMS and Logbook Data Management and Analysis in Fisheries Ecology. PLoS ONE 9. https://doi.org/10/f6cwb6

Sonntag, N., Schwemmer, H., Fock, H.O., Bellebaum, J., Garthe, S., 2012. Seabirds, set-nets, and conservation management: assessment of conflict potential and vulnerability of birds to bycatch in gillnets. ICES J. Mar. Sci. 69, 578–589. https://doi.org/10/ggkz8z

Soriano-Redondo, A., Cortés, V., Reyes-González, J.M., Guallar, S., Bécares, J., Rodríguez, B., Arcos, J.M., González-Solís, J., 2016. Relative abundance and distribution of fisheries influence risk of seabird bycatch. Sci. Rep. 6, 1–8. https://doi.org/10/f9dmm9

Tasker, M., 2000. The impacts of fishing on marine birds. ICES J. Mar. Sci. 57, 531–547. https://doi.org/10/cmz9nw

Thomas, R.J., 2017. Data analysis with R statistical software: a guidebook for scientists. Eco-explore.

Ulrich, C., Olesen, H.J., Bergsson, H., Egekvist, J., H\aakansson, K.B., Dalskov, J., Kindt-Larsen, L., Storr-Paulsen, M., 2015. Discarding of cod in the Danish Fully Documented Fisheries trials. ICES J. Mar. Sci. 72, 1848–1860. https://doi.org/10/f7nkjg

Vaida, F., Blanchard, S., 2005. Conditional Akaike information for mixed-effects models. Biometrika 92, 351–370. https://doi.org/10/cstbq8

Van Helmond, A., Chen, C., Trapman, B., Kraan, M., Poos, J., 2016. Changes in fishing behaviour of two fleets under fully documented catch quota management: Same rules, different outcomes. Mar. Policy 67, 118–129. https://doi.org/10/f8htf8

Youden, W.J., 1950. Index for rating diagnostic tests. Cancer 3, 32–35. https://doi.org/10/bjw536

Zuur, A., Ieno, E.N., Walker, N., Saveliev, A.A., Smith, G.M., 2009. Mixed effects models and extensions in ecology with R. Springer Science & Business Media.

Žydelis, R., 2013. The incidental catch of seabirds in gillnet fisheries: A global review. Biol. Conserv.

Žydelis, R., Bellebaum, J., Österblom, H., Vetemaa, M., Schirmeister, B., Stipniece, A., Dagys, M., van Eerden, M., Garthe, S., 2009. Bycatch in gillnet fisheries – An overlooked threat to waterbird populations. Biol. Conserv. 142, 1269–1281. https://doi.org/10/fwdfxp

PAPER 3

LIGHTS REDUCE SEABIRD BYCATCH IN A WESTERN BALTIC SEA DEMERSAL GILLNET FISHERIES, BUT PINGERS DO NOT

Gildas Glemarec, Anne-Mette Kroner, Lotte Kindt-Larsen, Finn Larsen

Manuscript

Abstract

Bycatch of non-target species is a major concern for the sustainability of many fisheries worldwide. In particular, a vast number of seabirds are caught in small-scale gillnet fisheries, which contributes to the global decline of some populations. However, only few bycatch reduction devices (BRDs) have shown encouraging results so far and, until now, no one-size-fit-all solution has emerged. In the Baltic Sea, diving birds, including seaducks, auks and cormorants, are particularly prone to capture in gillnets. In this paper, we report the results of two experimental bycatch reduction devices (BRDs) in the Western Baltic, using i) flashing white LED lights, and ii) acoustic alarms. Seabird bycatch per unit effort was recorded in experimental gillnets with BRD and compared to otherwise identical controls. Net lights showed promising results, reducing the bycatch of pelagic-foraging seabirds, but not of benthic-foraging seaducks. Acoustic pingers did not reduce seabird bycatch rates significantly. For both BRDs, catch rates of target fish were unaffected, and a small increase in catch per unit effort of European plaice *Pleuronectes platessa* was recorded in the nets equipped with lights. These findings are discussed in relation with the recent developments of seabird mitigation techniques in net fisheries.

Lights reduce seabird bycatch in a western Baltic Sea demersal gillnet fisheries, but pingers do not

Gildas Glemarec, Anne-Mette Kroner, Lotte Kindt-Larsen, Finn Larsen

Running page head: Seabird bycatch reductions using lights

ABSTRACT

Bycatch of non-target species is a major concern for the sustainability of many fisheries worldwide. In particular, a vast number of seabirds are caught in small-scale gillnet fisheries, which contributes to the global decline of some populations. However, only few bycatch reduction devices (BRDs) have shown encouraging results so far and, until now, no one-size-fit-all solution has emerged. In the Baltic Sea, diving birds, including seaducks, auks and cormorants, are particularly prone to capture in gillnets. In this paper, we report the results of two experimental bycatch reduction devices (BRDs) in the Western Baltic, using i) flashing white LED lights, and ii) acoustic alarms. Seabird bycatch per unit effort was recorded in experimental gillnets with BRD and compared to otherwise identical controls. Net lights showed promising results, reducing the bycatch of pelagic-foraging seabirds, but not of benthic-foraging seaducks. Acoustic pingers did not reduce seabird bycatch rates significantly. For both BRDs, catch rates of target fish were unaffected, and a small increase in catch per unit effort of European plaice Pleuronectes platessa was recorded in the nets equipped with lights. These findings are discussed in relation with the recent developments of seabird mitigation techniques in net fisheries.

Keywords: Seabird; Bycatch mitigation; Gillnet fisheries; Small-scale

INTRODUCTION

Gillnet fishing is responsible for the capture of numerous non-target species worldwide, including elasmobranchs, mammals, chelonians and seabirds (Lewison et al., 2014). Seabirds are globally declining (Dias et al., 2019), and gillnet mortality contributes to the decrease of some populations (Žydelis et al., 2013). All seabirds, including seaducks, share the common characteristics of spending a significant part of their existence at sea and of foraging for food principally in the water. Such a diversified group demonstrates a great variety of behaviour, so the effectiveness of bycatch deterring devices (BRDs) likely varies from one species to another (Mangel et al., 2018; Northridge et al., 2017).

The challenge of reducing bycatch in gillnet fisheries thus consists of making the nets conspicuous to seabirds, while remaining undetectable to the target fish species.

As for other amphibious animals, seabirds have evolved specific sensory adaptations to perceive their environment both in-air and underwater. For pelagic-foraging species (e.g. cormorants, auks, loons or penguins), vision seems to play a major role in the capture of prey (Martin, 2017; Martin et al., 2008; Martin and Crawford, 2015; White et al., 2007). Yet, the underwater visual acuity of the great cormorant Phalacrocorax carbo, the marine predator with the highest yield level measured to date (Gremillet et al., 2004), is comparable to that of a human being (Fay, 1992; Martin et al., 2008). Tactile rather than visual cues could be more important for benthic-foraging species, in particular for the seaducks who feed predominantly on sessile invertebrates (Martin, 2017). Hearing is generally a well-developed sense in the avian world, and many terrestrial and marine bird species use sound to communicate (Crowell, 2016; Dooling, 1992). However, the underwater hearing sensitivity of seabirds is less studied. Behavioural studies and direct physiological measurements (auditory brainstem response, ABR) demonstrated functional in-air and underwater auditory abilities for the great cormorant, with a best hearing frequency situated at 2 kHz in both media (Hansen et al., 2017;

Maxwell et al., 2016). Furthermore, in-air ABR measurements of 10 diving birds, showed a peak in hearing sensitivity between 1.7 kHz for the northern gannet Morus bassanus and 3 kHz for the lesser scaup Aythya affinis (Crowell et al., 2015). Nevertheless, hearing capacities are likely lower underwater than in-air for most seabirds. Besides, the ability for seabirds to hear directionally while diving and to locate the source of a sound underwater remains uncertain (Martin, 2017; Martin and Crawford, 2015).

In the past two decades, BRDs using visual or acoustic alerts have been developed to prevent seabird incidental catches in net fisheries. However, few of these mitigation technologies have been reported to reduce bycatch of seabirds while maintaining catch rates of commercial fish (Løkkeborg, 2011). In the Puget Sound salmon driftnet fishery (USA), the modification of the upper part of the nets to make them highly visible for seabirds floating at the surface, or the use of acoustic alarms (1.5 kHz pinger at 120 dB attached on the cork line every 50 m), both proved effective at reducing seabird bycatch rates (Melvin et al., 1999). Each method maintained catch rates of the target species, the sockeye salmon Oncorhynchus keta, and resulted in a large decrease in the bycatch rates of the common guillemot Uria aalge, but not of the rhinoceros auklet Cerorhinca monocerata. Furthermore, experiments conducted in the small-scale Peruvian gillnet fishery showed that using constant green lights on gillnets reduced the bycatch of a pelagic-foraging seabird, the Guanay cormorant Leucocarbo bougainvilliorum, without affecting the catch of target species (Mangel et al., 2018). In parallel, Martin and Crawford proposed to utilise high-contrast panels spaced at regular intervals on the nets to warn the birds diving in the vicinity (Martin and Crawford, 2015). This method was trialled in the Eastern Baltic Sea, in Lithuania and Poland, but failed at reducing overall seabird bycatch, and even increased the bycatch rates of a benthivorous seabird, the long-tailed ducks Clangula hyemalis (Field et al., 2019). The same authors also experimented with illuminating nets with LED lights. However, neither the steady green nor the flashing white LED lights that they tested were able to decrease the bycatch rates of the most commonly affected bird species. White lights increased significantly the bycatch of long-tailed duck, indicating that this species may have been attracted to the lights. These examples illustrate a major problem for seabird bycatch mitigation, which is that there is no universal solution to tackle this worldwide problem. Generally, trials using similar mitigation devices may be effective in one particular fishery but not in another, depending on fishery-specific operational factors (soak time, net length, mesh size), ecological factors (water transparency, depth) and on species-specific reactions to particular stimuli (visual, acoustic or other).

In this manuscript, we report the results of testing the effectiveness of i) flashing white LED lights and ii) acoustic pingers to reduce the bycatch of seabirds in the Danish demersal gillnet fishery. Specifically, we used two sets of paired identical gillnets, and compared the catches of target species and seabirds in experimental (with BRD) and control nets.

MATERIAL AND METHODS

The trials were conducted in the Sound, the strait separating Denmark from Sweden in the Western Baltic, where the risk of seabird bycatch in gillnets is known to be high between late fall and the end of winter (Glemarec et al., 2020b). One commercial gillnet fisher participated in the study and carried an electronic monitoring (EM) system with cameras as described in (Glemarec et al., 2020b). This same vessel had been monitored with EM since 2016, which ensured that the fishing practices were not modified after starting the experiment. An observer was present on board in 50% of the fishing trips (20 out of 41) to ensure that the protocol was respected and that the BRDs were functioning correctly.

The experimental setup presented in this study is purposefully analogue to the work of (Field et al., 2019) in the Eastern Baltic Sea, so that the findings in both studies can be compared easily. However, unlike these authors, we collected data from a single commercial gillnetter fishing in one location during one season between November 2018 and March 2019. We tested two candidate BRDs, one using acoustic (3 kHz pinger), and the other visual cues (flashing white LED light).

The effectiveness of the BRDs was tested using a simple paired design. Seabird bycatch per unit effort (BPUE) and target species catch per unit effort (CPUE) were compared between a pair of demersal gillnet fleet, which differed only by the addition of the trialled BRD on one of them (one control vs. one experimental fleet for each treatment). Each net fleet was made of transparent nylon monofilament gillnets with a stretched diagonal mesh size of 120 mm, of approximately 500 m in length and 3.6 m height. This type of net is typical in the study area to target cod Gadus morhua and European plaice Pleuronectes platessa. The skipper was solely responsible for choosing the fishing locations, depending notably on the daily weather conditions, current strength and wave height. Each pair of net fleets (one control vs. one experimental set) was submerged at the same location (similar depth and distance to shore), as a straight line, with a minimum space of approximately 200 m between the end of one set and the beginning of its pair. Both sets were retrieved together to ensure similar soak times. The data collected for each haul consisted of the total fish catch (total weight per species) and the number of birds of each species.

Net lights

We tested flashing white LED lights purchased from Fishtek Marine (UK). These battery-powered devices are identical to the ones used by (Field et al., 2019) in the Lithuanian coastal gillnet fishery trials in 2017-2018. The net lights emitted a sequence of intense white flashes (luminous flux = 10 lumen; wavelength: 430 – 630 nm; maximal intensity at 480 nm). One complete flash sequence lasted approximately 10 s and consisted of repeating 52 ms flashes in decreasing intervals of 2 s down to 250 ms, followed by a pause of 5.5 s. We used industrial alkaline batteries to power the devices, which ensured a lifetime of 800 h according to the manufacturer's specifications. The power level of each light was controlled after each haul, and all the batteries were replaced after 30 days at sea. The net lights were spaced alternatively on the lead line and on the floatline every 10 m, i.e. the horizontal distance between two consecutive lights on the floatline (leadline) was 20 m. Enchased in their rubber carrier, each net light (with battery) weighed 37 g in the water, so no additional float was added on the floatline to compensate for the extra weight.

Pingers

The pingers tested in this study operated at 3 kHz frequency with a sound intensity of 145 dB (Future Oceans, Australia). On average, rough weather conditions in the North Sea (corresponding to sea state 6) produce a background noise at 3 kHz of 60 dB at 20 m (Richardson et al., 2013), resulting in a signal-to-noise ratio of 85 dB at 1 m from the source. Sound absorption in seawater in the study area approximates 0.18 dB.km-1 at 3 kHz (Richardson et al., 2013), and can thus be ignored within 100 m from a pinger. A sound intensity of 85 dB is more than 10 dB higher than the minimal underwater hearing sensitivity of the great cormorant Phalacrocorax carbo (Hansen et al., 2017), one of the birds the most commonly captured in gillnets in the study area (Glemarec et al., 2020b). Therefore, using the hearing abilities of P. carbo as a reference, the minimum spacing between pingers was above 100 m. However, in this trial, in order to maximise their potential deterring effect, the pingers were spaced regularly on the floatline approximately every 12.5 m, totalling 42 pingers on the experimental net fleet. Mounted in a rubber carrier, the in-water weight of each pinger was less than 38 g, so no additional float was added to the floatline to compensate for the added weight.

Finally, the devices were supplied with C-cell lithium-ion batteries, allowing them to work throughout the duration of the study.

Data analysis

Bycatch per unit effort (BPUE) was calculated for each trial as the number of birds captured per kilometres of net times 24 h of soak [seabird bycatches / (net length x soak time)]. We expected to observe a difference in the response to BRDs between pelagic-foraging seabirds (e.g. auks and cormorants) – who rely heavily on vision to capture their preys – and benthic-foraging seabirds (e.g. seaducks) – who use principally tactile cues to detect the invertebrates they feed upon (Martin, 2017). We tested the null hypothesis that there was no difference in seabird BPUE between experimental and control sets for all birds pooled together, for pelagic-foraging birds only and for benthic-foraging birds only, using a randomisation test (Manly, 2018). Specifically, we estimated the mean and the 95% confidence intervals (CI) around the mean of the difference in BPUE using a bootstrapping method. This technique consists of randomly resampling the data with replacement to estimate the statistic of interest (here, the mean value of BPUE), and repeat the procedure a large number of times (here, 100,000 times) to obtain accurate estimates of the mean value of that statistic. We rejected the null hypothesis if the 95% CI of the difference in mean BPUE between experimental and control sets did not overlap zero, thus concluding that the BRD had a significant effect on bycatch. We applied the same method to compare the difference in catch per unit effort (CPUE) of the main target species. All data preparation and statistical analyses were conducted in R (R Core Team, 2020).

RESULTS

Between November 2018 and February 2019, 38 pairs of net fleets (32.9 km x 24 h) were deployed to test the effectiveness of flashing white lights as a seabird deterrent, and 41 pairs of nets (37.7 km x 24 h) were deployed to test 3 kHz pingers. This resulted in the capture of 27 birds (12 in the experimental sets and 15 in the controls) from six species in the net light trial, and 22 birds (10 in experimental sets and 12 in controls) from three species in the pinger trial (Table 1). All the birds recovered in the nets were dead upon collection. The common eider Somateria mollissima was the most affected bird with 30 bycaught individuals in total.

Table 1: Seabird bycatch in gillnets observed during the mitigation trials in the Danish Sound between November 2018 and February 2019.

	Flashing White Net Lights		**3 kHz Pingers**	
	Experiment	Control	Experiment	Control
Total seabird bycatch	**12**	**15**	**10**	**12**
Common eider (*Somateria mollissima*)	5	7	8	9
Common scoter (*Melanitta nigra*)	3	0	0	0
Velvet scoter (*Melanitta fusca*)	1	2	0	0
Great cormorant (*Phalacrocorax carbo*)	2	1	0	2
Common guillemot (*Uria aalge*)	1	4	2	1
Razorbill (*Alca torda*)	0	1	0	0

Change in BPUE in the net light trial

We found no significant difference in BPUE between the experimental and the control sets for all seabirds (bootstrapped mean difference: -0.13 birds per km x 24 h; 95% CI: -0.95 – 0.77 birds per km x 24 h; Fig 1). The presence of flashing white LED lights on the nets did not affect BPUE of benthic-foraging seabirds (bootstrapped mean difference: -0.10 birds per km x 24 h; 95% CI: -0.51 – 0.98 birds per km x 24 h; Fig 1). Nevertheless, we measured a small decrease in BPUE for pelagic-foraging seabirds in the nets equipped with lights (bootstrapped mean difference: -0.23 birds per km x 24 h; 95% CI: -0.62 – -0.02 birds per km x 24 h; Fig 1). The total weight of fish catch was highly variable, and the difference in mean total CPUE between the experimental and the control sets was not significant (bootstrapped mean difference: 1.1 kg of fish per km x 24 h; 95% CI: -7.4 – 5.9 kg of fish per km x 24 h; Fig 2 and Table 2). However, CPUE of plaice was significantly higher in the nets equipped with lights (bootstrapped mean difference: 0.95 kg of plaice per km x 24 h; 95% CI: 0.21 – 1.7 kg of plaice per km x 24 h; Table 2).

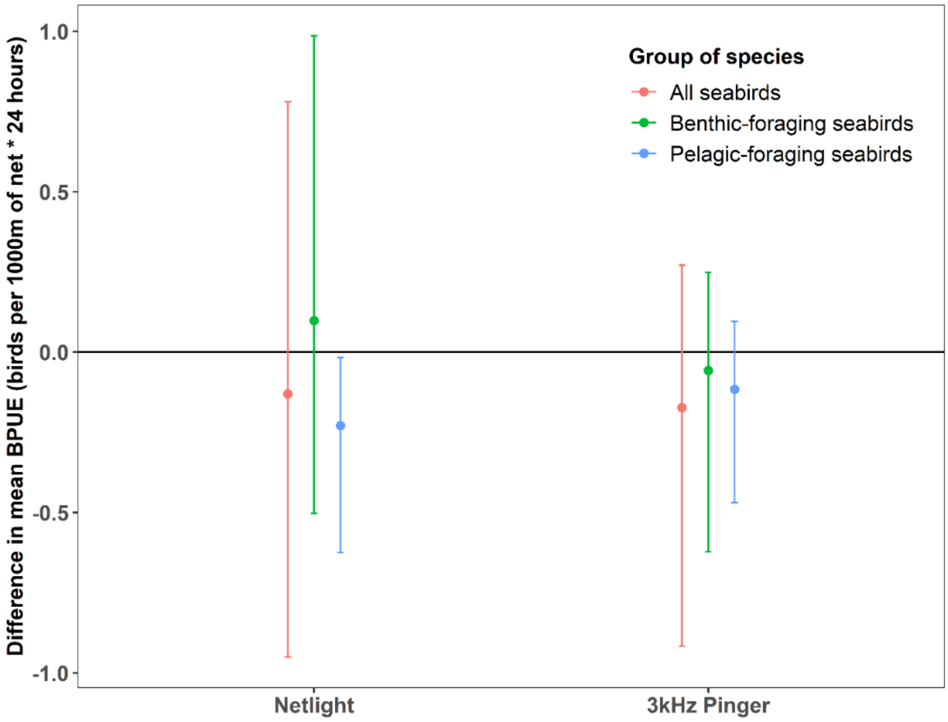

Fig 1. Difference in bycatch per unit effort (BPUE) of birds between paired experimental vs. control gillnet sets during the trial in the Danish Sound between November 2018 and February 2019. Mean bird BPUE (coloured dot) and 95% confidence intervals (error bars) were estimated using 100,000 bootstrap repetitions. Seabirds were pooled according to their foraging behaviour.

Table 2: Mean target species catch per unit effort (in kg of fish per km x 24 h) and standard error of the mean (s.e.) during the bycatch mitigation trials in the Danish Sound between November 2018 and February 2019.

	Flashing White Net Lights		**3 kHz Pingers**	
	Experiment	Control	Experiment	Control
Total CPUE	**44.6 ± 4.6**	**43.5 ± 5.0**	**52.0 ± 7.3**	**49.8 ± 6.7**
CPUE Cod (*Gadus morhua*)	38.6 ± 4.5	38.3 ± 5.0	44.2 ± 7.4	42.2 ± 6.8
CPUE European plaice (*Pleuronectes platessa*)	6.0 ± 0.7	5.1 ± 0.6	7.3 ± 1.2	7.2 ± 1.0
CPUE Atlantic salmon (*Salmo salar*)	0	0.2 ± 0.1	0.5 ± 0.3	0.4 ± 0.3

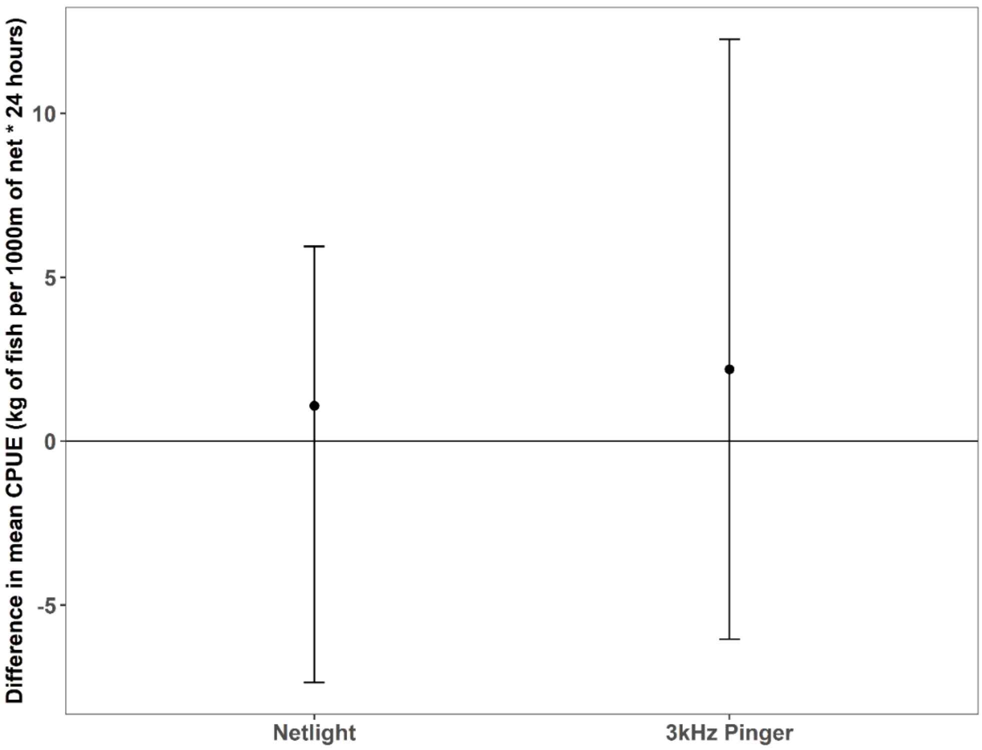

Fig 2: Difference in catch per unit effort (CPUE) of target species (fish) between paired experimental vs. control gillnet sets during the trials in the Danish Sound between November 2018 and February 2019. Mean CPUE of target species (black dot) and 95% confidence intervals (error bars) were estimated using 100,000 bootstrap repetitions.

Change in BPUE in the pinger trial

Using 3 kHz pingers on the nets did not have a significant effect on total seabird BPUE between experimental sets and controls (bootstrapped mean difference: -0.17 birds per km x 24 h; 95% CI: -0.91 – 0.27 birds per km x 24 h; Fig 1). Likewise, BPUE did not significantly differ for benthic-foraging seabirds only (bootstrapped mean difference: -0.06 birds per km x 24 h; 95% CI: -0.62 – 0.25 birds per km x 24 h; Fig 1), or for pelagic-foraging seabirds only (bootstrapped mean difference: -0.12 birds per km x 24 h; 95% CI: -0.47 – 0.10 birds per km x 24 h; Fig 1). As was the case in the net light trial, the total catch of fish in the pinger trial was not significantly different between experimental and control sets (bootstrapped mean difference: 2.2 kg of fish per km x 24 h; 95% CI: -6.1 – 12.2 kg of fish per km x 24 h; Fig 2 and Table 2).

DISCUSSION

The results of this study conducted in a bottom-set gillnet fishery in the Western Baltic Sea suggest that flashing white LED lights reduce incidental captures of pelagic-foraging seabirds (here, great cormorant Phalacrocorax carbo, common guillemot Uria aalge and razorbill Alca torda), but not of benthic-foraging seaducks (here, common eider Somateria mollissima, common scoter Melanitta nigra and velvet scoter Melanitta fusca). Conversely, in that fishery, acoustic pingers with an output signal centred at 3 kHz were not effective at reducing bycatch of either pelagic- or benthic-foraging seabirds. Catch rates of target fish species were unchanged whilst using bycatch reduction devices (BRDs), except for a small increase in mean catch per unit effort (CPUE) of plaice Pleuronectes platessa in the nets equipped with lights.

The flashing white LED lights used in Denmark (Western Baltic) had been previously tested in the Lithuanian coastal set net fishery (Eastern Baltic). The authors of this study concluded that using flashing white lights did not reduce seabird bycatch (Field et al., 2019). They even observed a significant increase of BPUE of long-tailed duck Clangula hyemalis in the experimental nets, suggesting that this benthic-foraging seabird was attracted to the net lights. In Denmark, only common eiders, common scoters and velvet scoters were recovered from the nets, and long-tailed ducks were totally absent from the records (Table 1). Contrasting with what was recorded in Lithuania, the bycatch rates of these benthic- foraging seaducks were not significantly different between the experimental nets equipped with lights and the controls in Denmark (Fig 1). The dissimilar response to the same treatment between both sides of the Baltic Sea suggests that different species of seaducks respond differently to flashing white lights, going from probable indifference (common eider and scoters) to potential attraction (long-tailed ducks). Moreover, results from the Peruvian set gillnet fishery of Constante showed that using constant green LED lights to illuminate gillnets reduced the bycatch of a pelagic-foraging bird, the Guanay cormorant Leucocarbo bougainvilliorum, by more than 85% (Mangel et al., 2018). Additionally, driftnets in the sockeye salmon fishery in the Puget Sound (USA), modified to make the upper section highly visible from the surface, reduced the bycatch of common guillemot considerably (Melvin et al., 1999). In the light of the above, increasing the visibility of gillnets seems to be an effective solution to reduce the bycatch of visual predators as pelagic-foraging seabirds, but not of other diving birds like benthic-foraging seaducks. However, reviewing the sensory adaptations of amphibious seabirds, (Martin and Crawford, 2015) advised against using lights on nets as BRD. They argued that seabird eyes need time to adapt to the low light conditions found at foraging depths. Exposed to intense light levels at depth, the pupils would be forced to shut rapidly, which in turn would result in temporary visual impairment for the birds. Even so, based on our results in the Danish gillnet fishery, we conjecture that birds like cormorants and auks may in fact be startled by repeating flash sequences. It is plausible that net lights could generate discomfort for these diving birds, reducing their overall predatory performance. In such a situation, the animals would likely be deterred from the source of visual disturbance, and swim away toward areas with more homogeneous light conditions.

High concentrations of particles in the water column directly affect water transparency, thereby reducing the potential of using light as BRD. The Baltic Sea is a semi-enclosed body of water, in which eutrophication has been a recurring problem for decades (HELCOM, 2018a). Generally, water turbidity is higher in the Baltic Proper than in the Western Baltic. For instance, in the Lithuanian coastal waters, where the net lights showed no reduction in seabird bycatch (Field et al., 2019), water clarity remains ordinarily below 3.8 m in the summer, while it averages 8.26 m in the Sound in the same period (HELCOM, 2018b). The relatively high clarity of Danish waters may have contributed to the lower bycatch rates of cormorants and auks observed in the nets equipped with lights in the Sound. These pelagic-foraging seabirds can perceive the flashes of light at a higher distance in more transparent waters, thus giving them more time to react before coming in contact with the nets. In contrast, benthic-foraging seaducks like common eiders and scoters typically search the sediment for food. Particle resuspension in the water column resulting from this behaviour can locally increase turbidity, making the nets and the net lights less visible for the birds. Benthic foragers may therefore be unable to detect the lights at a distance sufficient to elicit an evasive response.

Testing acoustic pingers was motivated by the relative lack of published work on the use of sound as a potential deterrent for seabirds. With one notable exception (Melvin et al., 1999), pingers have not been demonstrated to reduce bird bycatch in fishing nets. Still, in their study, the authors could not attribute the successful bycatch reductions to a direct avoidance behaviour. Many seabirds are certainly capable of hearing 3 kHz pingers underwater (Hansen et al., 2017), but their ability to pinpoint the source of a sound during a dive has not been elucidated (Martin, 2017; Martin and Crawford, 2015). In the trial we conducted, pinger effectiveness was not demonstrated, and bycatch rates were similar in both controls and experimental sets. These results suggest that seabirds either cannot locate the nets with pingers, or – if they can – that the sound they perceive does not trigger an aversive behaviour.

Mitigation device development needs to integrate fishers' needs for a practical and economical solution that does not affect the catches of target species. The methods that we tested in this study to reduce seabird bycatch did not decrease CPUE of fish. On the contrary, a small increase in CPUE of plaice, a fish of important commercial value in Denmark, was observed in the net light experimental sets (Fig 2 and Table 2). This could potentially make commercial fishers more eager to use a light-based mitigation device in the future. However, the utilisation of BRDs comes with drawbacks that may be difficult to overcome for some fishers. Beside the initial cost of such equipment, it is necessary to verify the battery level of each device at regular intervals and to change the battery when need be, which creates an additional running cost. Reducing the impact of commercial fisheries on ecosystems while maintaining economic value for the fishing sector is a priority in fisheries research (Aranda et al., 2019), so ensuring higher catch rates would certainly encourage the adoption of BRDs in commercial gillnet fisheries.

Here, we described two potential technologies to deter seabirds from gillnets in coastal fisheries. Acoustic pingers proved inefficient at reducing seabird bycatch, but flashing white lights fixed on the nets at regular intervals showed potential to reduce the capture rates of pelagic-foraging seabirds such as auks and cormorants. Although encouraging, our results differed from a previously published study using the same net lights, which found a net increase in bycatch of long-tailed ducks. This suggests that BRDs efficiency depends on species-specific responses to the stimuli, and future developments need to consider potential local particularisms. Finally, our study emphasises the need for more work in this field, notably to compare the relative effectiveness of similar BRDs in different fisheries on a variety of species prone to bycatch.

ACKNOWLEDGEMENTS

The authors express their sincere gratitude to the fisher who participated in the data collection, and without whom this study would not have been feasible. Moreover, the authors wish to thank the analysts who participated in the auditing of the video monitoring data, as well as the Ph.D. student Isabella Kratzer for helping setting up the mitigation devices on the net fleets. Finally, Anchorlab is also gratefully acknowledged for its technical support.

REFERENCES

Aranda, M., Ulrich, C., Le Gallic, B., Borges, L., Metz, S., Prellezo, R., Santurtún, M., 2019. Research for PECH Committee—EU fisheries policy–Latest developments and future challenges. European Parliament, Policy Department for Structural and Cohesion Policies, Brussels.

Crowell, S.C., 2016. Measuring In-Air and Underwater Hearing in Seabirds, in: The Effects of Noise on Aquatic Life II. Springer, pp. 1155–1160.

Crowell, S.E., Wells-Berlin, A.M., Carr, C.E., Olsen, G.H., Therrien, R.E., Yannuzzi, S.E., Ketten, D.R., 2015. A comparison of auditory brainstem responses across diving bird species. J Comp Physiol A 201, 803–815. https://doi.org/10.1007/s00359-015-1024-5

Dias, M.P., Martin, R., Pearmain, E.J., Burfield, I.J., Small, C., Phillips, R.A., Yates, O., Lascelles, B., Borboroglu, P.G., Croxall, J.P., 2019. Threats to seabirds: A global assessment. Biological Conservation 237, 525–537. https://doi.org/10.1016/j.biocon.2019.06.033

Dooling, R.J., 1992. Hearing in birds, in: The Evolutionary Biology of Hearing. Springer, pp. 545–559.

Fay, R.R., 1992. Structure and function in sound discrimination among vertebrates, in: The Evolutionary Biology of Hearing. Springer, pp. 229–263.

Field, R., Crawford, R., Enever, R., Linkowski, T., Martin, G., Morkūnas, J., Morkūnė, R., Rouxel, Y., Oppel, S., 2019. High contrast panels and lights do not reduce bird bycatch in Baltic Sea gillnet fisheries. Global Ecology and Conservation 18, e00602. https://doi.org/10.1016/j.gecco.2019.e00602

Glemarec, G., Kindt-Larsen, L., Scherffenberg Lundgaard, L., Larsen, F., 2020. Assessing seabird bycatch in gillnet fisheries using Electronic Monitoring. Biological Conservation 243. https://doi.org/10.1016/j.biocon.2020.108461

Gremillet, D., Kuntz, G., Delbart, F., Mellet, M., Kato, A., Robin, J.-P., Chaillon, P.-E., Gendner, J.-P., Lorentsen, S.-H., Maho, Y.L., 2004. Linking the foraging performance of a marine predator to local prey abundance. Funct Ecology 18, 793–801. https://doi.org/10.1111/j.0269-8463.2004.00914.x

Hansen, K.A., Maxwell, A., Siebert, U., Larsen, O.N., Wahlberg, M., 2017. Great cormorants (Phalacrocorax carbo) can detect auditory cues while diving. Sci Nat 104, 45. https://doi.org/10.1007/s00114-017-1467-3

HELCOM, 2018a. State of the Baltic Sea–Second HELCOM holistic assessment 2011-2016, in: Baltic Sea Environment Proceedings. p. 155.

HELCOM, 2018b. Water clarity HELCOM core indicator 2018 [WWW Document]. URL https://www.helcom.fi/wp-content/uploads/2019/08/Water-clarity-HELCOM-core-indicator-2018.pdf (accessed 2.17.20).

Lewison, R.L., Crowder, L.B., Wallace, B.P., Moore, J.E., Cox, T., Zydelis, R., McDonald, S., DiMatteo, A., Dunn, D.C., Kot, C.Y., Bjorkland, R., Kelez, S., Soykan, C., Stewart, K.R., Sims, M., Boustany, A., Read, A.J., Halpin, P., Nichols, W.J., Safina, C., 2014. Global patterns of marine mammal, seabird, and sea turtle bycatch reveal taxa-specific and cumulative megafauna hotspots. PNAS 111, 5271–5276. https://doi.org/10.1073/pnas.1318960111

Løkkeborg, S., 2011. Best practices to mitigate seabird bycatch in longline, trawl and gillnet fisheries—efficiency and practical applicability. Mar. Ecol. Prog. Ser. 435, 285–303. https://doi.org/10.3354/meps09227

Mangel, J.C., Wang, J., Alfaro-Shigueto, J., Pingo, S., Jimenez, A., Carvalho, F., Swimmer, Y., Godley, B.J., 2018. Illuminating gillnets to save seabirds and the potential for multi-taxa bycatch mitigation. Royal Society open science 5, 180254.

Manly, B.F., 2018. Randomization, bootstrap and Monte Carlo methods in biology. chapman and hall/CRC.

Martin, G., 2017. The sensory ecology of birds, First edition. ed, Oxford avian biology series. Oxford University Press, Oxford.

Martin, G.R., Crawford, R., 2015. Reducing bycatch in gillnets: A sensory ecology perspective. Global Ecology and Conservation 3, 28–50. https://doi.org/10.1016/j.gecco.2014.11.004

Martin, G.R., White, C.R., Butler, P.J., 2008. Vision and the foraging technique of Great Cormorants Phalacrocorax carbo: pursuit or close-quarter foraging? Ibis 150, 485–494.

Maxwell, A., Hansen, K.A., Larsen, O.N., Christensen-Dalsgaard, J., Wahlberg, M., Siebert, U., 2016. Testing auditory sensitivity in the great cormorant (Phalacrocorax carbo sinensis): Psychophysics vs. Auditory brainstem response, in: Proceedings of Meetings on Acoustics 4ENAL. Acoustical Society of America, p. 050001.

Melvin, E.F., Parrish, J.K., Conquest, L.L., 1999. Novel Tools to Reduce Seabird Bycatch in Coastal Gillnet Fisheries. Conservation Biology 13, 1386–1397. https://doi.org/10.1046/j.1523-1739.1999.98426.x

Northridge, S., Coram, A., Kingston, A., Crawford, R., 2017. Disentangling the causes of protected-species bycatch in gillnet fisheries: Causes of Bycatch. Conservation Biology 31, 686–695. https://doi.org/10.1111/cobi.12741

R Core Team, 2019. R: A language and environment for statistical computing.

Richardson, W.J., Greene Jr, C.R., Malme, C.I., Thomson, D.H., 2013. Marine mammals and noise. Academic press.

White, C.R., Day, N., Butler, P.J., Martin, G.R., 2007. Vision and foraging in cormorants: more like herons than hawks? PLoS One 2.

Žydelis, R., Small, C., French, G., 2013. The incidental catch of seabirds in gillnet fisheries: A global review. Biological Conservation 162, 76–88.