JNCC Report 726

Regional Seabird Bycatch Hotspot Analysis

Simon Northridge, Al Kingston and Alex Coram

July 2023

ISSN 0963 8091

JNCC's report series serves as a record of the work undertaken or commissioned by JNCC. The series also helps us to share, and promote the use of, our work and to develop future collaborations.

For further information on JNCC's report series please contact:
Joint Nature Conservation Committee
Quay House
2 East Station Road
Fletton Quays
Peterborough PE2 8YY
https://jncc.gov.uk/

Communications@jncc.gov.uk

This report was produced for JNCC under an external contract, by Scottish Oceans Institute, University of St Andrews for/under contract number C19-0241-1342.

This report was produced by JNCC for the Department for Environment Food and Rural Affairs (Defra) under Project Code ME6024. Copies of the report are available through both the Defra and JNCC websites.

This report should be cited as:
Northridge, S.P., Kingston, A.R. & Coram, A.J. 2023. Regional seabird bycatch hotspot analysis. *JNCC Report No. 726.* JNCC, Peterborough, ISSN 0963-8091.
[https://hub.jncc.gov.uk/assets/a00403c7-6f56-4f38-8b57-b2ab1859c564]

Acknowledgments:
Earlier versions of the report were reviewed by Matt Parsons and Lise Ruffino (JNCC) and the Seabird Bycatch Technical Advisory Group, which includes representatives from the Statutory Nature Conservation Bodies (SNCBs), RSPB, CEFAS and Seafish.

Evidence Quality Assurance:
This report is compliant with JNCC's Evidence Quality Assurance Policy
https://jncc.gov.uk/about-jncc/corporate-information/evidence-quality-assurance/

Summary

This study was initiated as a part of the UK marine bird bycatch plan of action.

The objective is to identify areas and fisheries around the UK that might inform options for regional pilot schemes to undertake seabird bycatch mitigation trials, and to identify a rationale for selecting candidate areas.

A brief overview of historical records of seabird bycatch in UK waters is provided, in which a wide variety of species are found to have been recorded for more than a century in many UK fisheries, including especially gillnets, pound nets and longline fisheries.

The UK Bycatch Monitoring Programme (BMP) has monitored over 21000 fishing operations since 1997 and during this time has recorded the bycatch of 585 individual birds.

Data from the BMP are summarised and compared with data from the at-sea commercial catch sampling programme (CSP) of England and Wales. The dedicated programme (BMP) has yielded much higher observed bycatch rates.

The BMP has focused on longlining (North and West of Scotland), gillnetting (mainly in the English North Sea and in the Celtic Sea), and midwater trawls (mainly in the Channel). The CSP has focused much more on demersal trawling in the North Sea and western waters.

Among longline fisheries, bycatch rates of seabirds is highest along the shelf edge to the North and west of Scotland, and mainly involves fulmars. Among observed gillnet hauls, the highest bycatch rates appear off the Northeast of England, off southeast Ireland, along parts of the South coast of England and off Shetland; the main species involved are guillemots, cormorants and razorbills. Among observed midwater trawling operations, most bycatches have been observed in the western Channel and involve guillemots.

Fulmar bycatch in the longline fishery is largely confined to a dedicated fleet that operates in a defined area north and west of Scotland. Mitigation trials are ongoing in this fishery.

Guillemot and cormorant bycatch in gillnets appear to be most frequent in gillnets set in shallow water (approximately 20 m) in winter months, especially in the areas mentioned above. Although smaller meshed nets at first sight appear to be more liable to bycatch birds than larger meshed nets, this is most likely because some types of smaller meshed nets are associated with winter fishing close to shore.

The key issues in developing any regional mitigation trails will be firstly to identify potential mitigation measures, then to find skippers who are willing and able to assist in carrying out trials, and finally to develop a suitably robust procedure for analysis and interpretation of results.

Contents

1 Background

In July 2018, the UK Department for the Environment, Food and Rural Affairs (Defra) asked the Joint Nature Conservation Committee (JNCC) to develop a UK marine bird bycatch Plan of Action (PoA) to: "Deliver a coherent approach to understand and where necessary reduce marine bird bycatch in UK fisheries, through engagement and dialogue with all interested parties and the implementation of subsequent recommendations". The Plan of Action has now been superseded by the Bycatch Mitigation Initiative.

Stemming from that request and subsequent developments in the PoA, a first assessment of broad-scale seabird bycatch mortality by some sectors of the UK-registered fishing fleet in UK and adjacent waters was undertaken using data collected under the UK Bycatch Monitoring Programme (BMP) and official fishing effort statistics (Northridge *et al.* 2020). The assessment revealed that several thousand seabirds may die annually because of being accidentally caught in the fishing gears considered in the study. The analysis also suggested that there might be some regional higher-risk areas ("hotspots"), for example of fulmar mortality in longlines to the north and west of Scotland, and mortality of a variety of species in static nets particularly around the Southwest and Northeast of England.

As the PoA continues to develop, there is a need to refine our understanding of the location and nature of any regional seabird bycatch hotspots to help focus concerted efforts between industry, regulators, and scientists to help reduce seabird bycatch where this is deemed necessary and to improve our general understanding of how, when and where bycatch occurs most frequently.

As a next step in this process Defra asked the JNCC to develop a second project with the main objectives of:

1. To provide more detailed analysis of the BMP dataset and other relevant data sources on bycatch rates, bycatch risk (i.e. Bradbury *et al.* 2017) and where available, relevant fishing effort data, to identify potential hotspots of seabird bycatch in UK waters for the primary purpose of informing options for regional pilot areas for undertaking targeted bycatch mitigation trials. Such an analysis also has the potential to highlight areas where increased monitoring and research efforts might be justified.

2. Propose and provide a rationale for possible candidate areas for pilot studies based on the above analyses and other relevant and available information.

Subsequently, in November 2019 the JNCC commissioned the Scottish Oceans Institute (SOI), University of St Andrews to carry out the analysis, and this report describes the analytical methodology, the results obtained and the rationale behind some proposed candidate areas for initial pilot studies based on data collated from several sources: the BMP, the English and Welsh commercial catch sampling programme (EW-CSP), the Bradbury *et al.* report (2017) and official fishing effort statistics maintained by the Marine Management Organisation (MO).

2 An overview of existing information related to hot spots of seabird bycatch in the UK

There have been few previous studies of seabird bycatch in UK waters, and no complete overview in the mainstream literature. Žydelis *et al.* (2009) address the issue of seabird bycatch in gillnets in the North and Baltic Seas, but the only UK reference in their review is that of Murray *et al.* (1994), which relates to guillemot and razorbill bycatch in salmon bag nets, which are staked shore nets with a fish pound. Melville (1973) reported catches of seabirds including guillemots, razorbills, shags, eider, black guillemots and kittiwake in fixed salmon nets on the Antrim coast, and also refers to reports of shags, guillemots, black guillemots, red-throated diver, gannet, tern and a fulmar, all caught in Scottish fixed salmon nets. Bibby (1971) also reported on seabird bycatches (guillemots) in Irish and Scottish salmon nets. Staff of the Sea Mammal Research Unit (SOI) have spent many months studying Scottish bag and stake net salmon fisheries over several decades, because these fisheries had recognised problems with seal depredation. Entanglements of guillemots, razorbills and some black guillemots were quite frequent in this fishery, though records have not been fully collated (SMRU unpublished data). The fishery was closed in 2016 to conserve salmon stocks, initially for three years. After an assessment of subsequent research and of the status of salmon stocks for 2019, the Scottish Government decided that the closure should remain in place.

Historically, there are no doubt many such references, but these have not been comprehensively collated, though Dunn (1994) covers some. For example, Cobb (1976) refers to observations by W.C. Mackintosh, the first director of the University of St Andrews Gatty Marine Laboratory (now the SOI) of about 4000 birds being killed during the summer fishing season in longline and driftnet fisheries by boats from St Andrews; these were mainly of guillemots, though divers *Gavia spp*, scoters *Melanitta spp*, scaup and razorbills were also taken.

Several other unpublished reports have highlighted specific seabird bycatch issues in several coastal fisheries. Robins (1991) provided a global overview of seabird bycatch in gillnet fisheries, but emphasised records from the UK, and focused particularly on the small-meshed winter set gillnet fishery for bass in and around St Ives Bay in Cornwall (targeted net fisheries for bass are not currently permitted under bass stock management measures). Bird bycatch was first noticed in this fishery in the early 1980s. During the winter of 1989/1990 clifftop observations of the fishery were made over sixteen four-day sampling periods, resulting in a bycatch estimate of about 520 birds, mainly guillemots and razorbills, during that whole fishing season. However, 536 dead birds were counted in this fishery on a single day the previous year, in February 1988. Bird mortality numbers seemed to be related, unsurprisingly, to the number of birds present in the Bay on any one day. Dunn (1994) provides further details and some figures for later years as well.

Dunn (1994) also notes “recent monitoring has also indicated that there are no serious bird bycatch problems in Scotland (Murray 1993) or Wales (Thomas 1992), or at least none that are not already being addressed”. However, he noted that an inshore cod fishery along the Yorkshire coast took more guillemots than the salmon nets at the time and threatened eider ducks. More recently, and after the decline in coastal cod gillnet fishing through the late 1990s and into the 21st century, and a ban on gillnetting close to seabird breeding cliffs at Flamborough Head, bycatch in salmon fixed nets (“J-nets”) has become more of an issue (Quayle 2015). Very high levels of seabird bycatch were reported in 2008 and since then a voluntary code of conduct has been implemented to minimise seabird bycatch in the salmon / sea trout fishery around Flamborough Head.

Seabird bycatch is also reported anecdotally, and quantitatively from observer programmes, along the Sussex coast and elsewhere on the South coast of England – mostly in bass drift nets, and was described in Coram *et al.* (2015).

Longline fishing has been a major focus of seabird bycatch studies throughout the world, but since the 1970s there had been little longline fishing by UK vessels, until a recent expansion by UK registered vessels fishing mainly for hake in offshore waters around the British Isles. French and Spanish vessels also operate in these fisheries. During the 1990s and 2000s there was some focus on bird bycatch in Norwegian and Faroese longline fisheries, mainly for cod, operating North of Shetland and in the Norwegian Sea (Løkkeborg 1998, 2011; Tasker *et al.* 2000; Dunn & Steel 2001). Tasker *et al.* (2000) reported bycatch rates of fulmars (95% of birds caught in this fishery were fulmars) ranging from 0.02 birds per 1000 hooks to 1.75 birds per 1000 hooks.

3 At sea sampling coverage and recorded seabird bycatch

In the UK, two long term at-sea fishery dependent monitoring programmes have been conducted since about the mid 1990's.

The BMP, which began collecting data in 1996, is managed by a team at the University of St Andrews and has the primary aim of quantifying non-commercial protected or vulnerable species bycatch in various fisheries to meet several international monitoring obligations including EU Council Regulation 812/2004 (recently repealed), Article 12 of the Habitats Directive (92/43/EEC), the Common Fisheries Policy Technical Conservation Regulation (2019/1241), the Agreement on the Conservation of Small Cetaceans in the Baltic, North East Atlantic, Irish and North Seas (ASCOBANS), the International Convention on the Regulation of Whaling (ICRW) and the Oslo and Paris Conventions (OSPAR).

The commercial catch sampling programmes (abbreviated here to: CSPs) began in the early 2000's and are managed largely by the UK national government fisheries science agencies (the Centre for Environment, Fisheries and Aquaculture Science (Cefas), Marine Scotland Science (MSS) and the Agri-Food and Biosciences Institute of Northern Ireland (AFBINI)). An at-sea observer programme is also managed by the Scottish Fishermen's Federation (SFF), jointly operated with the MSS scheme. The at-sea element of the CSPs have historically had the primary purpose of quantifying commercial species discard rates to meet obligations under the Data Collection Framework (DCF) of the Common Fisheries Policy, however data relating to non-commercial species bycatch - including protected species such as seabirds - may also have been collected. The latest version of the DCF (EU 2017/1004) has a more formal requirement for data on protected species bycatch to be recorded within the CSPs, but the revised Regulation contained no specific guidance on how data collection activities might be altered to achieve that. Consequently, a lot of work has recently been undertaken by ICES through various workshops (e.g. WKPETSAMP) and closer collaboration between relevant ICES expert working groups (WGBYC and WGCATCH) to provide guidance on this question. As a result, some new elements to on-deck sampling protocols within the CSPs are due to be implemented from 2020. These alterations to sampling procedures should improve the reliability of those data for inclusion in bycatch assessments going forward.

Both the BMP and CSPs use experienced sea-going fisheries observers to collect data at sea, but because the underlying purpose of each programme type differs, basic sampling designs (i.e. what fisheries are sampled) and sampling protocols (i.e. how data are collected) within each programme are generally optimised to achieve their respective goals.

The BMP currently has a subcontract with Cefas, part of which requires Cefas to provide annual updates of all CSP sampling conducted in England and Wales (EW) and supply records of protected species bycatch observed under the DCF or any other at-sea fisheries dependent sampling efforts. Although there are existing questions about the reliability of the CSP data in relation to seabird bycatch due to sampling protocols being largely designed to provide information on commercial species, it does provide an additional data source to the BMP and may provide insights into bycatch occurrences in gear types or areas not routinely sampled under the BMP. Therefore, in this section we have collated all the historical at-sea data from the EW-CSP up to 2018 and present them in an equivalent way to data collected under the BMP. These are tabulated in Table 1 and presented as maps in Figures 2 to 5 (Figure 1 provides a legend for the associated maps). This provides an initial representation of spatial sampling coverage and the locations of recorded seabird bycatch by species within each programme, as a first analytical step in identifying possible candidate areas for regional pilot studies.

Table 1. Sampling levels, seabird bycatch observations, bycatch rates and BMP:CSP rate ratio from data collected under the BMP and EW-CSP by broad gear type and ICES Subarea. (Symbols: x= factor difference, + = higher in BMP (factor not calculable), - = lower in BMP (factor not calculable), N/A = rates not calculable from either programme due to zero bycatch observations.

	ICES Subarea	BMP No hauls	BMP Seabirds	BMP Rate/haul	CSP No hauls	CSP Seabirds	CSP Rate/haul	BMP:CSP ratio
Nets	4	6629	130	0.0196	663	1	0.0015	x 13.0
	5	0	0	0	8	29	3.6250*	-
	6	587	0	0	8	13	1.6250*	-
	7	11936	235	0.0197	3758	13	0.0035	x 5.7
	8	53	0	0	0	0	0	N/A
Longlines	4	32	56	1.7500	28	0	0	+
	6	44	124	2.8182	82	9	0.1098	x 25.7
	7	30	8	0.2667	23	0	0	+
Midwater trawl	2	11	0	0	0	0	0	N/A
	4	256	1	0.0039	18	0	0	+
	6	258	0	0	0	0	0	N/A
	7	1783	31	0.0174	44	0	0	+
	8	19	0	0	29	0	0	N/A
Demersal trawl	2	0	0	0	4	0	0	N/A
	4	5	0	0	5276	2	0.0004	-
	6	0	0	0	17	0	0	N/A
	7	115	0	0	16298	5	0.0003	-
	14	0	0	0	13	0	0	N/A
All		21758	585	0.0269	26269	72	0.0027	x 9.8

*Data from sampling in the deep-water shark gillnet fishery which no longer operates.

From Table 1 it is evident that most sampling within the BMP has focussed on static net and midwater trawl fisheries, because these gear types are generally considered to be of highest risk in relation to marine mammal bycatch which was the initial focus of the BMP. Conversely, sampling within the EW-CSP has historically been heavily weighted towards demersal trawl fisheries, because these gears are typically associated with relatively high commercial fish species discard rates, but a reasonable amount of sampling has also been conducted in some net fisheries. Longline sampling has occurred within both programmes, mainly historically from the EW-CSP (during the 2000's) and more recently within the BMP (since 2010).

Calculated bycatch rates are generally higher in the BMP data. In specific metiers where a reasonable amount of sampling has occurred in both the BMP and EW-CSP (nets in Subareas 4 and 7, longlines in Subarea 6) the rates range from about six to 25 times higher in the BMP dataset. In several other metiers (area and gear type combination) we cannot make a proper comparison of rates because sampling in one or other programme has been very limited, and seabird bycatches have only been observed in one of the sampling programmes. In some of these metiers (longlines in Subareas 4 and 7, midwater trawls in Subareas 4 and 7) observed rates are higher within the BMP dataset, but in others (nets in Subareas 5 and 6 and demersal trawls in Subareas 4 and 7) rates are higher within the EW-CSP.

Overall, in the full datasets the unstratified seabird bycatch rates observed in the BMP are an order of magnitude higher than the rates observed in the EW-CSP. This is to be expected given the primary aim of the BMP is to quantify protected species bycatch rates whereas the CSPs are designed to quantify commercial fish species discard rates, so data collection protocols within the CSPs are not generally optimal for recording less frequent protected species bycatch.

The spatial sampling coverage for the main gear types sampled within each programme (nets, midwater trawls (BMP only), demersal trawls (CSP only) and longlines (sampling levels quite low but included as this is relevant to seabird bycatch)) are shown in Figures 2 and 5 and are presented as "heat" type maps by broad gear type and programme type. We used QGIS to plot all individual haul positions and then the "heat" is built up where the radii (in this case set at 10 km) around each haul position intersect. The purpose of this is to provide a general but accurate impression of relative spatial sampling intensity. The same intensity scale has been used across all the maps and details are provided in the legend in Figure 1.

In a partner map the locations of all recorded seabird bycatches by species, gear and sampling programme are also displayed. The "heat" and species legends have not been overlaid on each of the individual heat maps so as not to obscure the underlying data.

Figure 1. Sampling intensity legend and bird species key used throughout Section 3.

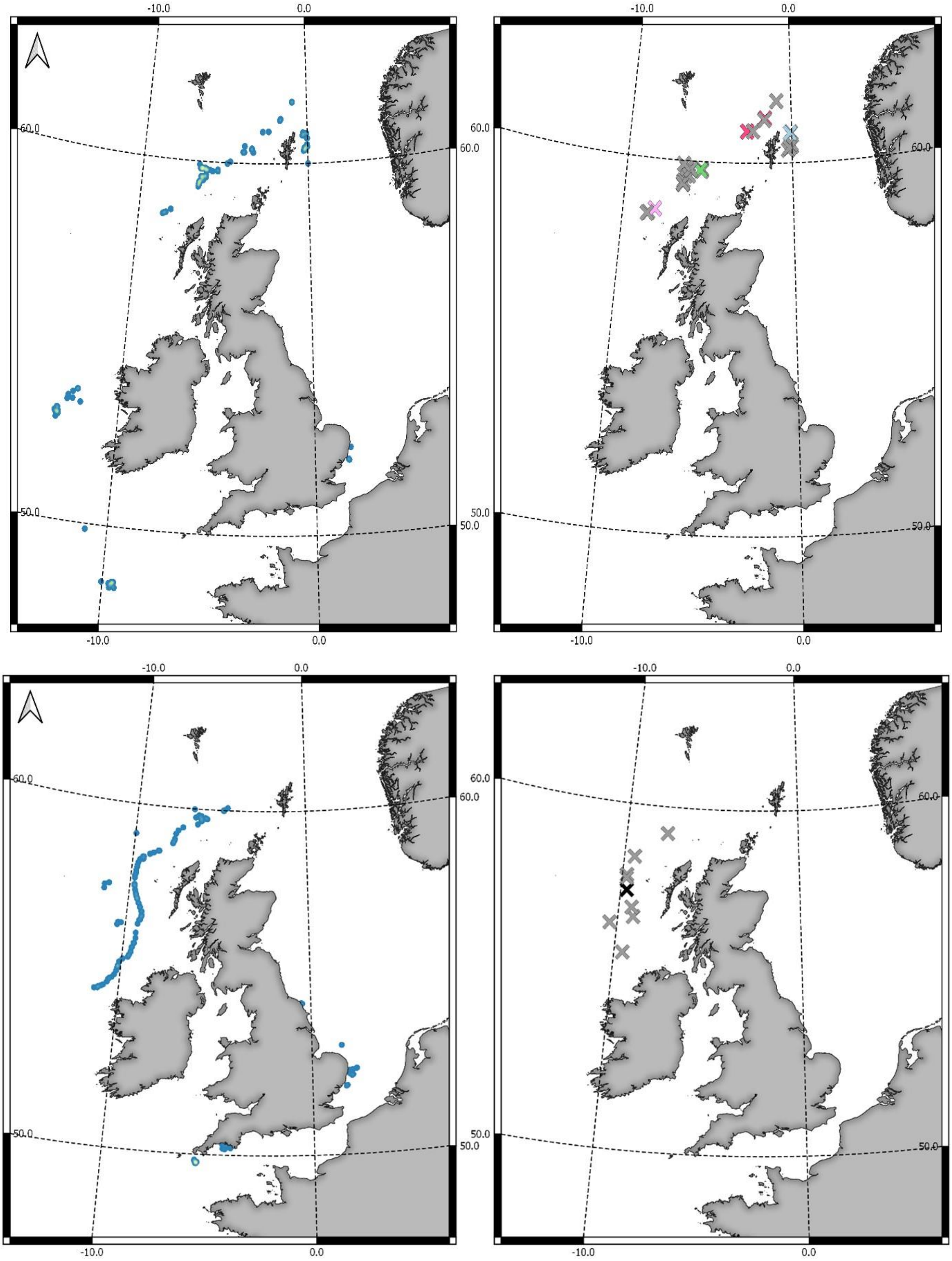

Figure 2. Longline sampling (left) and recorded seabird bycatch (right) from the BMP (top) and the EW-CSP (bottom).

Fairly low-level longline sampling has been conducted under both programmes to date and most of the sampling has occurred in offshore fisheries that operate mainly along the continental shelf edge to the west and north of Scotland. Longline sampling under the EW-CSP mainly occurred in the mid-2000's west of Scotland, whereas sampling under the BMP began in 2010 and is ongoing, so may eventually provide a more accurate reflection of recent patterns of fishing effort and bycatch within this fishery. All vessels operating in the offshore longline fishery are required to carry VMS systems and these data could be used to visualise the full extent of the fishery but have not been considered in this report. Seabird bycatch in the offshore longline fishery appears to be dominated by northern fulmar and bycatch rates are higher in the northern part (ICES Divisions 4a and 6a) of the fishery's range.

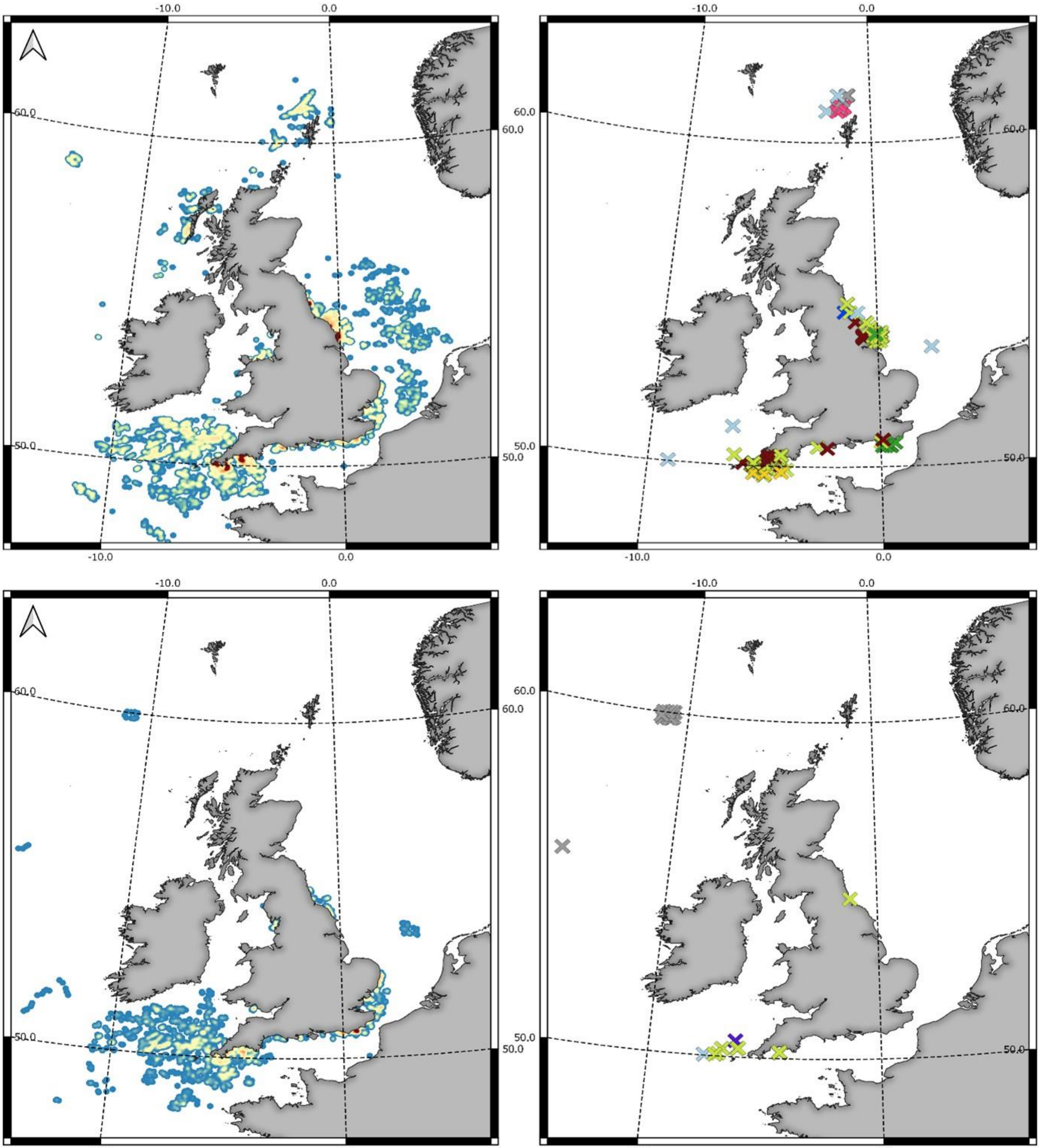

Figure 3. Static net sampling (left) and recorded seabird bycatch (right) from the BMP (top) and the EW-CSP (bottom).

A significant amount of sampling of net fisheries has occurred within both programmes though the overall number of sampled hauls is higher, and the geographical spread of sampling is wider, within the BMP. Seabird bycatch has been recorded more frequently and in general more species are represented within the BMP dataset. From the maps in Figure 3 it is evident that seabird bycatch in net fisheries is mainly concentrated in relatively coastal areas with apparently regional concentrations in the Northeast, Southeast and Southwest of England. The main exception to this is several fulmar bycatches recorded under the EW-CSP from a few hauls in a gillnet fishery targeting deep-water sharks off the Northwest of Scotland (ICES Subareas 5 and 6). This specific fishery was phased out through the 2000's as deep-water shark quotas gradually reduced. All deep-water sharks are now classed as prohibited species and cannot be landed, so this fishery no longer operates.

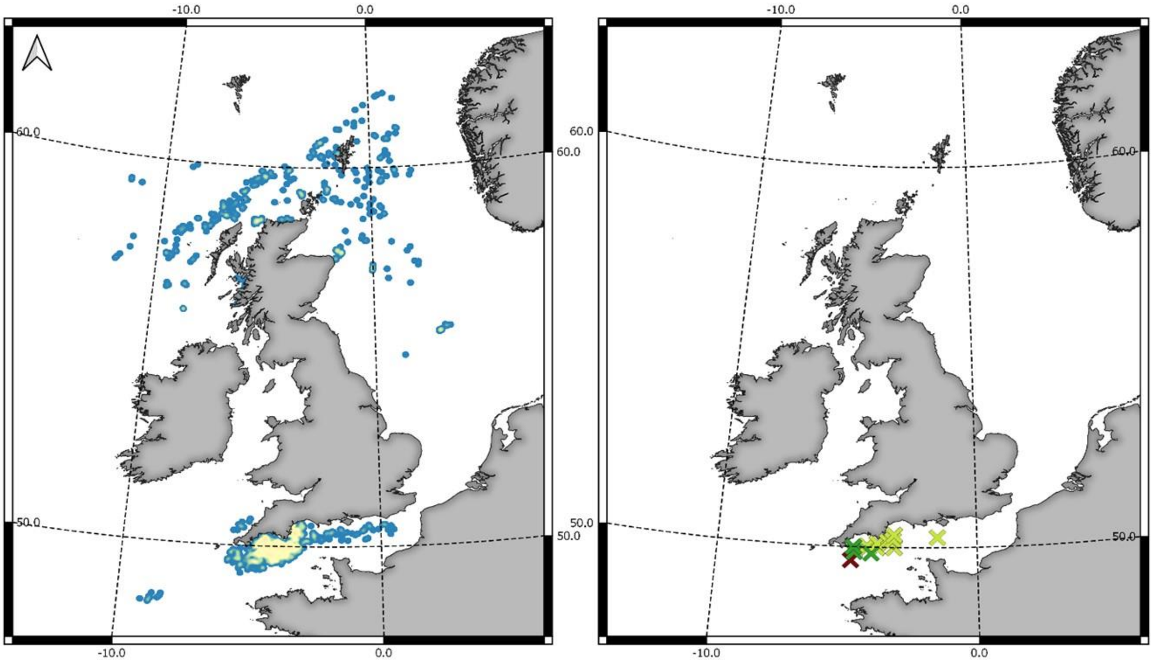

Figure 4. Midwater trawl sampling (left) and recorded seabird bycatch (right) from the BMP only.

Midwater trawl sampling within the BMP has focussed on fisheries targeting bass (fishery closed in 2015) and sprat in the English Channel (Divisions 7de) and various fisheries targeting mackerel, herring, sprat, blue whiting, boarfish and sandeel mainly in the North Sea (Subarea 4) and Northwest of Scotland (Subarea 6). Seabird bycatch has only been recorded in the English Channel fisheries. Further details of this are provided in (Northridge *et al.* 2020). A limited number of midwater trawl hauls (90) have been sampled under the EW-CSP in ICES Subareas 4, 7 and 8 and no seabird bycatch was recorded so these are not presented here.

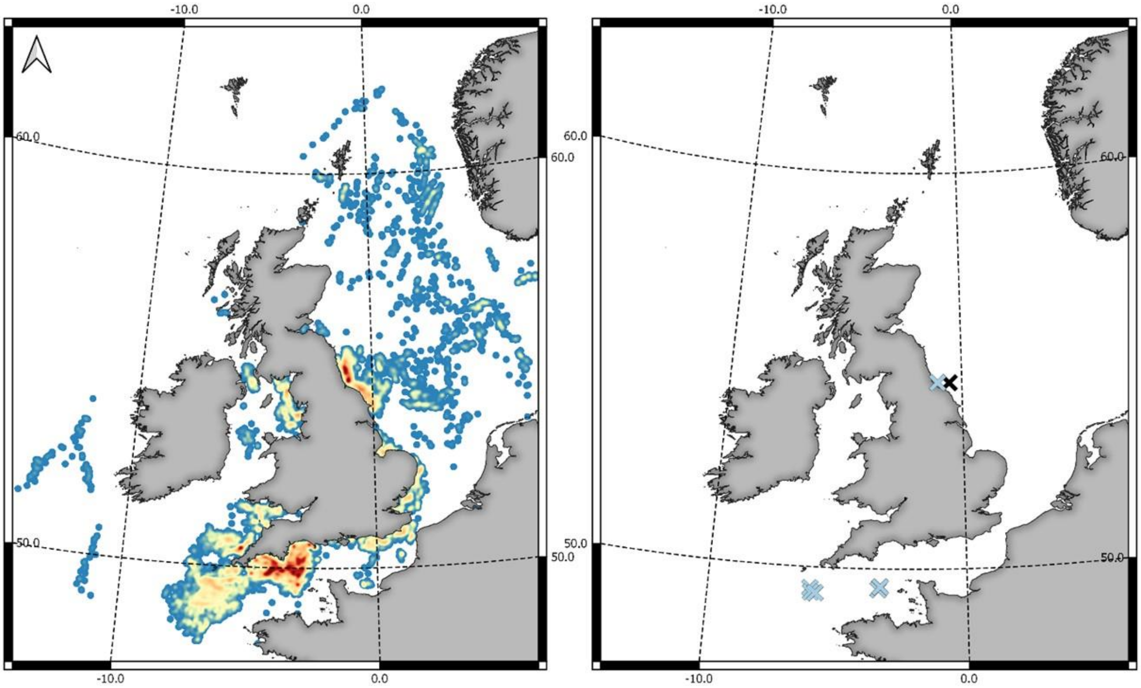

Figure 5. Demersal trawl sampling (left) and recorded seabird bycatch (right) from the EW-CSP.

A significant amount of demersal trawl sampling has been conducted under the EW-CSP, and this broad gear category makes up most of the at-sea data collection within that programme. This category includes beam trawls, scallop dredges and a variety of otter trawl gears. Most of the sampling effort is concentrated around the Southwest peninsula, with smaller concentrations in the Northeast and Southeast of England and in the Irish Sea. Some seabird bycatch has been recorded in demersal trawls, mostly northern gannets. Bycatch rates appear to be low, but they should be viewed in the context of data collection protocols within the CSPs potentially leading to underestimates of seabird bycatch. A limited number of demersal trawl hauls (120) have been sampled opportunistically under the BMP, mainly in ICES Subarea 7, with no recorded seabird bycatch so they are not presented here.

The data represented here from the EW-CSP have not been used to estimate bycatch levels or to identify hot spots, beyond what is covered in the maps and text above. This is because, as Table 1 makes clear, seabird bycatch generally appears to be under-reported in the EW-CSP data. In part this may be because it was not until recently that on-board sampling protocols within the CSPs were adapted to include seabird bycatch explicitly, but it is also because on board sampling for fish lengths and discards may conflict with an observer's ability to see and record seabird bycatch. In theory it may be feasible to sift through the data records from the EW-CSP and either exclude or down-weight trips or hauls where the potential for monitoring seabird bycatch may have been compromised but this would be a potentially very time-consuming exercise. Consequently, for the remainder of this report we use only data collected from the BMP to use bycatch rates as a means of identifying potential bycatch hotspots.

4 Areas, seasons and fisheries with highest seabird bycatch rates

The main purpose of this report is to use the data described above to find areas of high seabird bycatch, loosely referred to as hotspots. It is worth considering what this means in some more detail.

Identifying areas of highest 'risk' for seabird bycatch around the UK has been attempted previously by Bradbury *et al.* (2017), who categorised fisheries by their expected risk to different bird species, based on expert opinion and qualitative records, and depending on where in the water column those fisheries operate or where bycatch is considered most likely to occur within the fishing operation. From resulting tabulated 'entrapment risks', together with each species' conservation status, demographic parameters, and behavioural characteristics, 'sensitivity indices' were constructed for each UK seabird species for each of three categories of fishing gear (surface, midwater and demersal). These sensitivity indices were then applied to maps of seabird density to generate vulnerability maps. Vulnerability maps were the combined with fishing effort for over 15 m vessels to provide composite maps of population impact risk due to bycatch for all seabirds combined for each of three gear categories and for summer and winter periods (Figures 5 to 7, p38 to 40 in Bradbury *et al.* 2017).

The seabird bycatch risk maps presented in Bradbury *et al.* (2017) are quite difficult to interpret. They are produced from a combination of VMS (Vessel Monitoring System) fishing effort data, seabird density distribution data and expected sensitivity of specific birds to general gear types as described above. The risk that is being considered is therefore the risk *to the population*, rather than the risk *to any individual bird*, or the risk of a *bycatch event occurring during any specific fishing operation*. As our objective here is to identify regional pilot areas for undertaking targeted bycatch mitigation trials, we are most interested in areas with *the highest probability of a bycatch event occurring* during a fishing operation as these will provide most power to quantify the effectiveness of any trialled mitigation measure. The population level impacts are not important in this context. The probability of a bycatch event occurring is related to species specific vulnerability ('entrapment risk' + 'response to fishing activity' in Bradbury *et al*) and bird population density and is *independent of fishing effort density*. It can therefore be inferred directly from bird density distributions and measures of vulnerability to the gear concerned if those parameters can be quantified. But more directly this risk can be quantified from observed bycatch rates where sufficient observations have been made. The level of fishing effort then is only relevant if we need to consider the population level effect of bycatch: a high bycatch rate could occur in a low intensity fishery and have little overall impact on a seabird population, while a lower bycatch rate, if associated with a large and wide-ranging fishery, may have a more significant population level impact.

Given that our primary aim here is to highlight the areas with highest bycatch rates (i.e. those areas where there is an elevated probability of bycatch events occurring - which is most useful for planning mitigation trials) - then the risk maps presented by Bradbury *et al* (2017) are not particularly helpful in our context. The Bradbury *et al.* (2017) risk maps incorporate fishing effort data in the calculation of "risk" meaning areas with higher fishing effort will be considered as having relatively higher risk – whereas the highest **bycatch rates** (number of birds per unit of fishing effort) could occur in areas with little fishing effort but high bird density. If our aim had been to identify areas where highest overall annual bycatch events occur (highest number of seabirds per year) then it would be appropriate to combine areas of high bycatch rate (individuals per fishing operation) with fishing effort data to delineate high 'population risk' zones where workable mitigation measures could be focused. We reconsider this approach in relation to the Bradbury et al report in Appendix 2.

Accepting then that the primary purpose of this report is to identify areas with fisheries that are most likely to have a high probability of bycatch to help identify fisheries and areas where mitigation trials may most effectively be tested, we do not need to consider overall fleet effort intensity, except to ensure there is some threshold level of fishing effort with which mitigation measures could be tested. Instead, we need to focus on the areas where we have observed the highest bycatch rates. These may be considered some synthesis of vulnerability (or in fisheries terms ‘catchability’) and seabird density, though it is important to note that the relationship between seabird density and bycatch rate can by no means be assumed to be a linear one.

For longline fisheries, this situation is relatively clear, and is described in Sections 3 and 4 above. A map of bycatch records (Figure 6) shows where the highest bycatch rates have been observed, which coincides largely with a relatively large area where fulmar density is highest, to the north and west of Scotland (Bradbury *et al.* 2017, Figures 56 & 57 pp 112-113), which also happens to be where the fishery mainly operates. In section 3, where we have plotted the locations of observed hauls and observed bycatch, fulmar bycatch has been recorded throughout much of the area in which the fishery operates to the West and North of Scotland, notwithstanding the fact that fishing effort moves around from year to year within that general area.

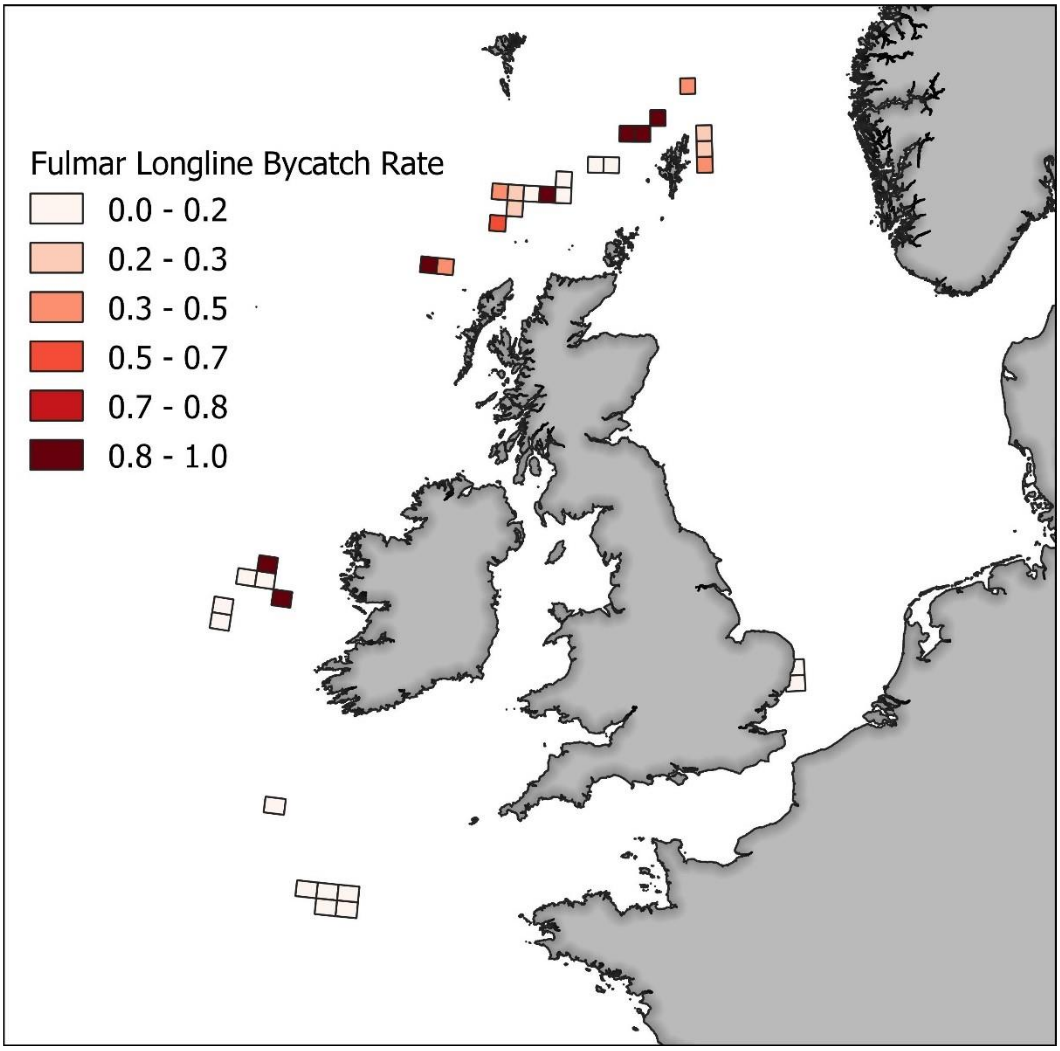

Figure 6. Map of observed fulmar bycatch rates (number per haul) in longline fishery: 14 sampled trips 2010-2018.

Furthermore, the UK offshore longline fishery is fairly homogenous, with one main target species and apparently similar modes of operation between all boats that we have sampled. Variation in fulmar bycatch rates is very high between trips (varying from 0 to 1.4) and we do not yet know if there are seasonal or vessel level effects, nor do we know whether there are any spatial gradients of risk within the general area. The mean observed rate of 0.23 birds per 1000 hooks is about midway between the minimum and maximum rates reported in Tasker *et al.* (2000).

Seabird bycatch in the much smaller inshore longline fishery has not yet been fully assessed, due to very low sampling effort, so we cannot include these data in any sensible analysis.

Pelagic trawl fisheries are also hard to decipher with respect to seabird bycatch as all recorded seabird bycatch from this gear type come from small vessels fishing in the English Channel. There is only one regular UK midwater trawl fishery now in the English Channel (which occurs in autumn/winter targeting sprat), and only a few bycatch observations from that specific fishery. We cannot deduce any finer scale contours of bycatch rates than we have from simple descriptive statistics above (Figure 4).

Within static net fisheries, there is more scope for assessing where and when the highest bycatch rates might occur. However, despite there being over 18000 observed hauls, fewer than 200 of these were bycatch positive, so there are limited data on which to base any detailed analysis to identify very specific areas. However, we can try to determine broad scale features that are associated with higher bycatch probabilities.

To this end, we can view the observed bycatch of seabirds in static nets (see Figure 7). The highest rates appear off the Northeast of England, off southeast Ireland, along parts of the South coast of England and off Shetland. These patterns obviously represent past observations and some of the fisheries involved may no longer exist or be greatly reduced in scale. Cod gillnetting, for example, which was once prevalent along the coast of northeast England, has declined considerably since the 1990s, and bass netting around much of the Welsh and southern English coasts is now considerably reduced due to bass conservation measures. This map also only represents where sampling has occurred irrespective of where fishing fleet effort may be most prevalent. It is therefore probably more useful to be able to identify specific factors that are associated with higher bycatch rates so that we can try to *predict* where, under current fishing conditions, high bycatch rates might be expected.

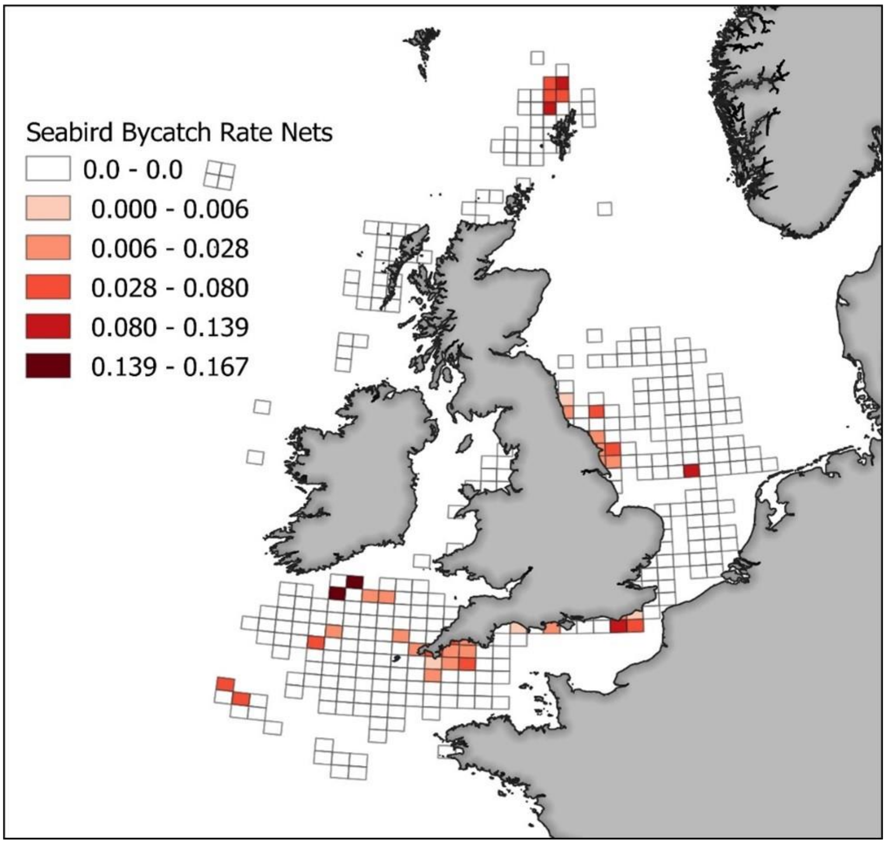

Figure 7. Observed bycatch rates of seabirds in static nets: birds per haul.

Some preliminary exploratory analysis of the data suggests that bycatch rates of guillemots and cormorants (the most numerous species bycaught in this gear-type) are highest in shallower waters (see Figure 8).

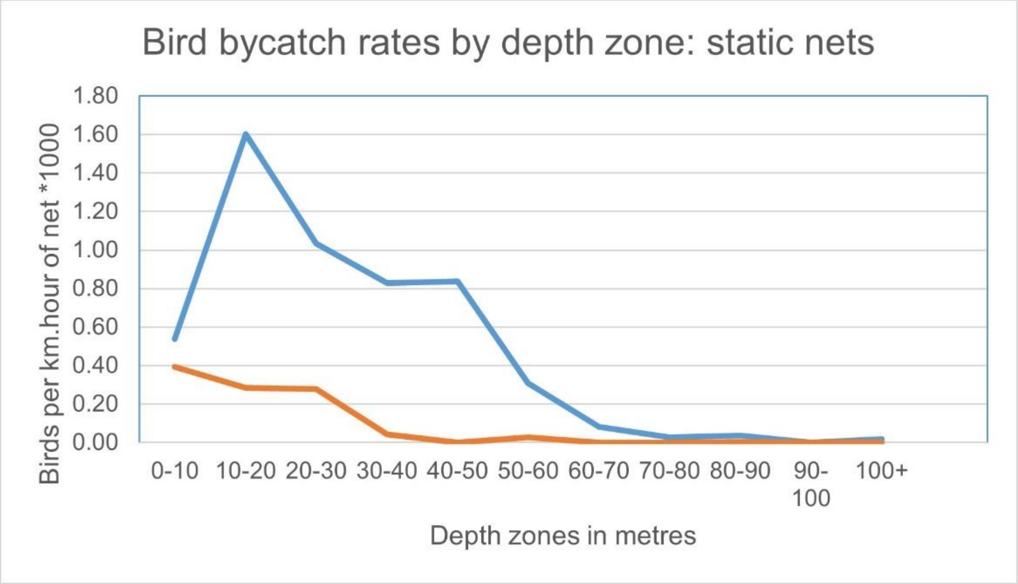

Figure 8. Bycatch per unit effort of guillemots (blue) and cormorants (orange) by water depth zone.

There also appears to be a clear relationship between the bycatch rates of both cormorants and guillemots by season, with much higher rates observed across the winter months (Figure 9).

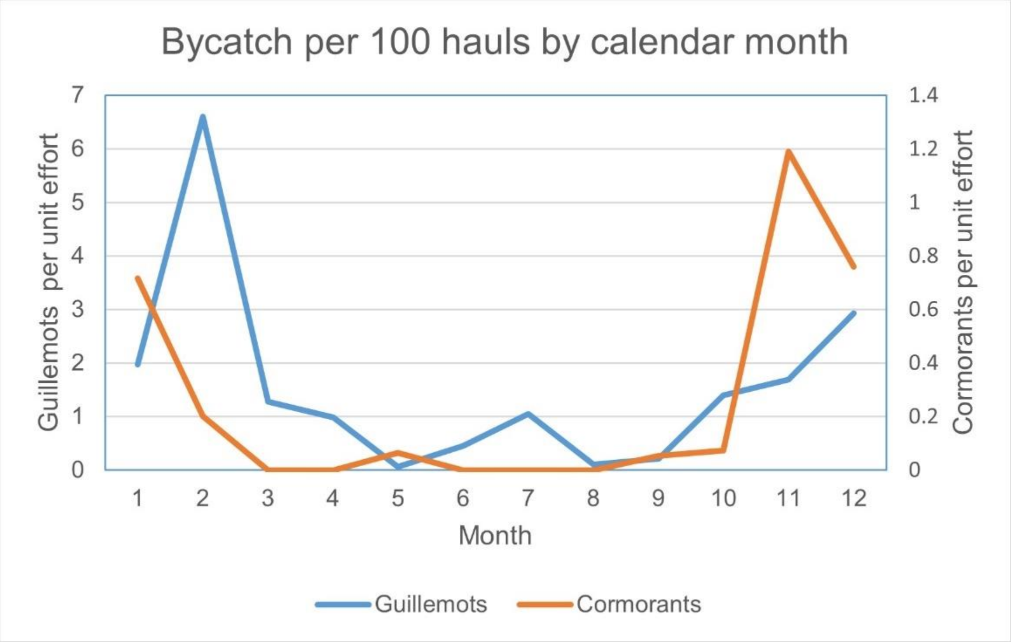

Figure 9. Bycatch rates by month – guillemots (blue) and cormorants (orange) in static nets.

Certain target species of fish, including salmon, mackerel, and bass, initially seemed to be associated with higher bird bycatch rates than others within the BMP data. But as there are dozens of recorded target species, any analysis by target fish species would lack statistical power and we would not be able to use it to robustly identify regions or areas of high expected bycatch rates. Instead, we have explored the data using a simpler fishery specific metric: mesh size. In Table 2, below, we have tabulated observed seabird bycatch rates by mesh size category.

We split mesh sizes into three relatively equal but essentially arbitrary categories: small = 0 to 100 mm, medium = 101 to 200 mm and large = greater than 200 mm. Small-meshed nets appear to have much higher bycatch rates of guillemots and cormorants than medium and large meshed nets.

Table 2. Numbers of birds recorded by species, and two bycatch rates: by mesh size category.

		Guillemot	**Razorbill**	**Cormorant**	**All Birds**
Mesh cat	**Numbers of birds observed**				
Large		45	0	9	73
Medium		168	11	20	226
Small		50	1	8	59
	Observed effort and bycatch per unit effort (net km.hour x10^3)				
Large	1305811	0.03	0	0.01	0.06
Medium	325781	0.52	0.03	0.06	0.69
Small	20014	2.5	0.05	0.4	2.95

	Observed hauls and bycatch per haul (x100)				
Large	7384	0.61	0	0.12	0.99
Medium	8799	1.91	0.13	0.23	2.57
Small	2135	2.34	0.05	0.37	2.76

These simple descriptive analyses suggest that smaller meshed nets fished during the winter months (October through March) in shallow water (less than 60 m) are associated with the highest bycatch rates of both cormorants and guillemots. However, it is also possible that these factors are confounding one another; for example, if small-meshed nets were mainly used in winter months or in shallower waters. We have therefore conducted some preliminary statistical modelling using the BMP dataset to address this question. We used a Generalised Additive Modelling approach to explore how the bycatch rate (as a binomial variable with respect to haul) might be affected by water depth, season, and mesh size category (Appendix 1).

We explored several alternative models for simple presence/absence by haul for cormorants and guillemots (data for razorbills were too limited). We found that the best fitting model (P value model selection) for cormorants was one that included net soak time (increasing risk of bycatch), water depth (decreasing bycatch with water depth), and month (more cormorants in winter). However, bycatch was very low for this species in the observed hauls. For guillemots the best model consisted of a smooth of depth with more bycatch in shallower water, and an interaction of month and mesh category. Model descriptions and output are presented in Appendix 1. Essentially the modelling confirmed highly significant seasonal and depth effects. There was evidence that in contrast to the simple descriptive statistics presented in Table 3, small-meshed nets have a lower probability of bycatch than medium and large meshed nets, once depth effects are accounted for. Therefore, small-meshed nets tend to be fished in shallow waters, but it is water depth that is most clearly associated with elevated bycatch rates.

5 Suggested areas and fisheries to trial bycatch mitigation

Our initial analysis above suggests that based on the (limited) information we currently have, fisheries with the highest observed rates of seabird bycatch, and the ones that therefore might be most suitable for bycatch mitigation trials, are the longline fishery to the Northwest of Scotland (see Figure 6) and static net fisheries in winter months and in shallow water (peaking at about 20 m).

The location of the longline fishery is well known and has been mapped by Bradbury *et al.* 2017 (Figure 132, p.188) who state that "longline fishing effort is concentrated along the shelf break north and west of Scotland in summer and winter"; sampled locations are shown in Figure 2.

Static net fisheries are more difficult to describe because they are more varied, being both seasonal and target-species specific, meaning there are potentially dozens of specific metiers. Landings data should provide us with a means to identify target fisheries, areas, and seasons for bycatch mitigation trials in gillnet fisheries. While landings data do not explicitly identify static net metiers, or identify the depth zones fished, there is often information available on mesh size used. Although our analysis above does not conclude that mesh size is an important factor in determining seabird (guillemot and cormorant) bycatch rates in net fisheries, season and water depth are both apparently significant variables, and winter fisheries in shallow waters appear to be disproportionately associated with smaller meshed nets.

In order to identify areas that might be suitable for bycatch mitigation trials or more targeted seabird bycatch monitoring, we have plotted (Figure 10) the amount of recorded fishing effort (days at sea) by vessels under 12 m in length in one recent pre-covid year (2017) using static nets in winter (October through March) in all ICES rectangles through which the British coastline passes (as a rough proxy for shallow water effort).

Most vessels under 12 m fish predominantly within the 12 nm territorial limit, which in most ICES rectangles will be shallower than water outside territorial waters. Therefore, we can assume that these effort data provide the best representation of shallow water winter static net fishing that are available from official fishing effort statistics. Because we have little confidence that mesh size on its own is driving seabird bycatch rates (see Appendix 2), we have mapped fishing effort for all mesh sizes in Figure 10.

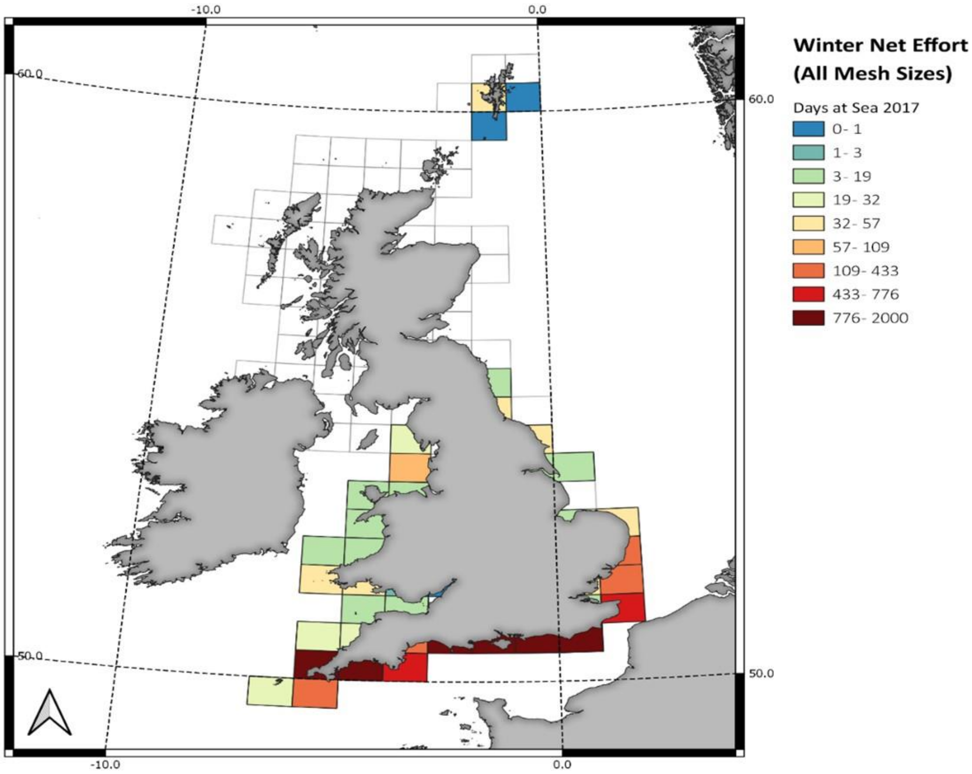

Figure 10. Distribution of winter static net fishing effort for 2017 in inshore waters and by vessels of 12 m or less, all mesh sizes.

This fishing effort map can then be compared with heatmaps of observed bycatch for the two species that are of most significance within our gillnet observations: guillemots and cormorants. Observed bycatch rates of guillemots and cormorants are mapped out below in Figures 11 and 12, together with an indication of the extent of our observations, here using a 20 km radius around our observed hauls and a quartic kernel function (QGIS kernel density plugin). The heat maps provide a way of visualising where we have observed highest bycatch rates in nets around the coast of the UK. Bycatch rates are expressed as the number of birds per observed haul. The maps are useful for highlighting areas where bycatch mitigation trials might be focused, but do not necessarily reflect where total mortality is highest because they do not include information about total fleet effort, and our sampling may not be fully representative of wider fishing effort levels. Furthermore, some areas like the Yorkshire coast, where gillnetting was quite common, now has little netting effort (see Figure 10).

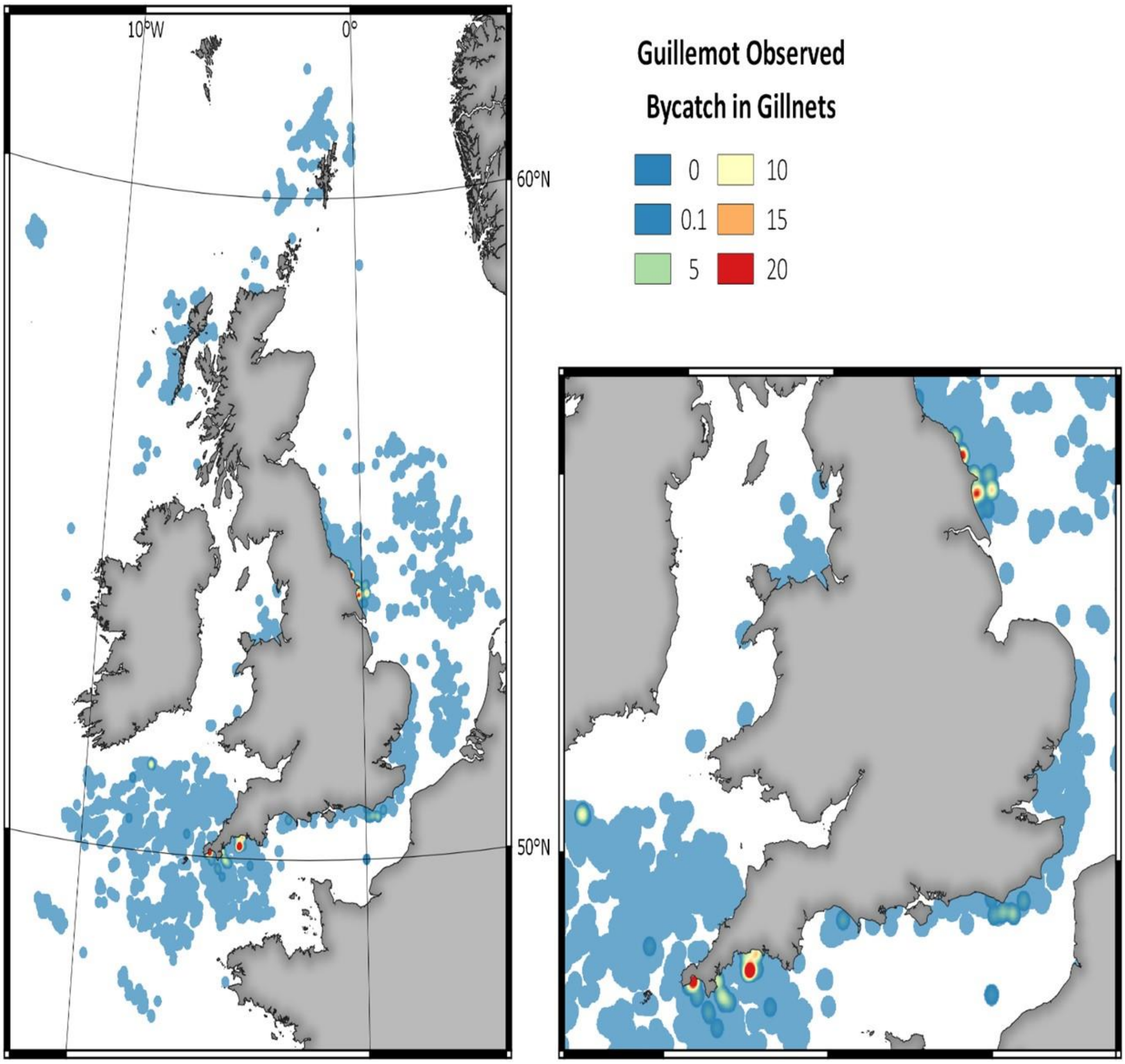

Figure 11. Guillemot bycatches in static nets heatmap. Map on right shows “zoomed in” view of the data showed in left-hand map.

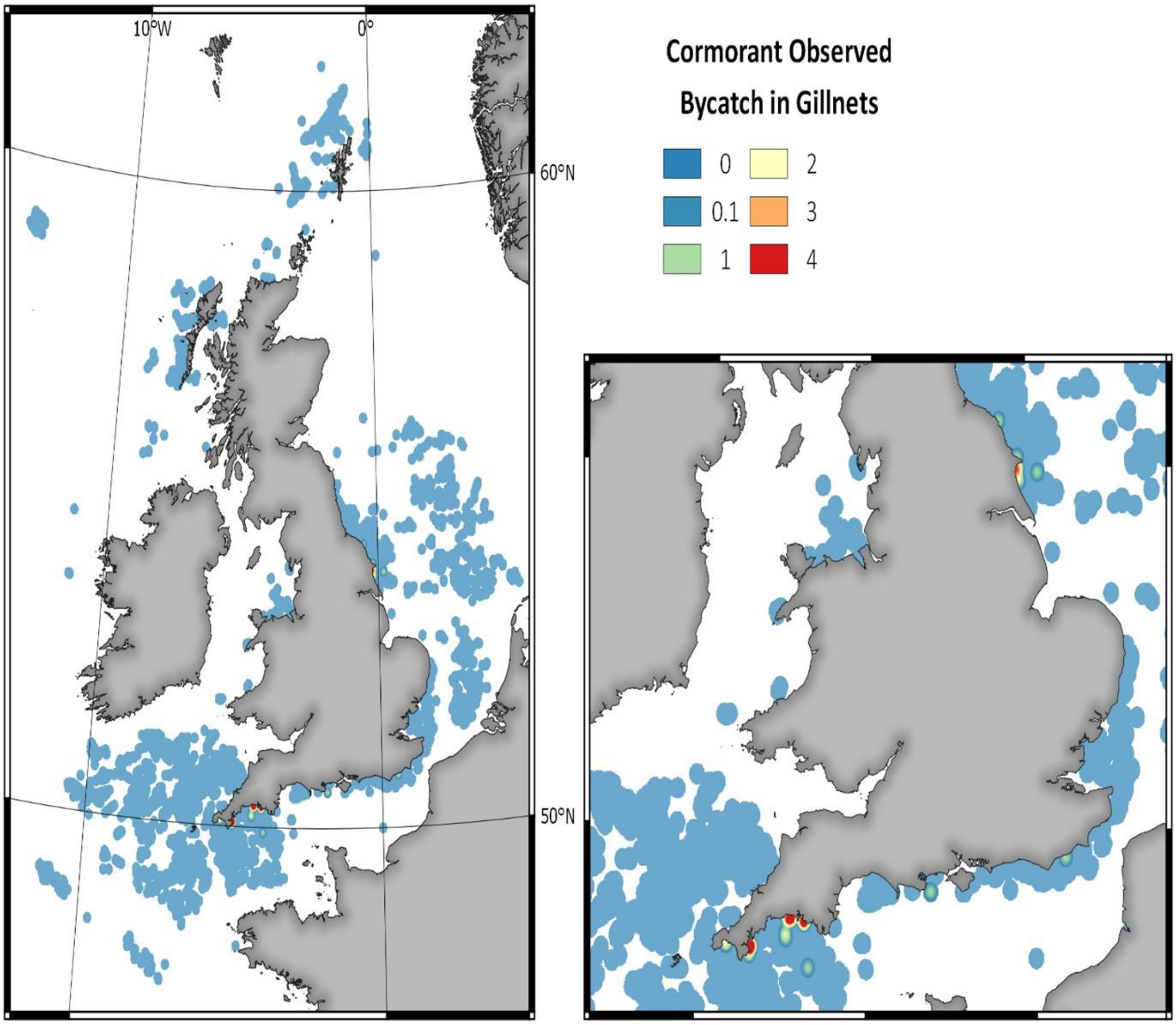

Figure 12. Cormorant bycatches in static nets heatmap. Map on right shows "zoomed in" view of the data showed in left-hand map.

There are some important differences to highlight between Figure 10 and Figures 11 and 12. Notably, one of the areas with the highest recorded less than 12 m net fishing *effort*, the Eastern English Channel (Fig. 10), does not appear to have as high observed guillemot or cormorant *bycatch rates* as areas further west in the English Channel (i.e. on the south coasts of Devon and Cornwall) or off the Yorkshire coast. The overlap of relatively high levels of winter gillnet fishing effort along the Cornish coast together with relatively high observed bird bycatch rates could suggest that this region might be an area where both bycatch events are relatively frequent *and* total mortality is relatively high.

Comparing our observed bycatch rate maps with the (winter) distribution map of guillemots in Bradbury *et al.* (2017, Figure 117, p173) suggests that the area of high bycatch we have observed along the south Cornwall coast is also associated with relatively high guillemot densities in winter. But there are several other high seabird density areas that we have sampled that appear not to have such high bycatch rates. These include the south Devon coast, the central eastern Channel, and the North Cornwall coast. Furthermore, while summer guillemot densities (Bradbury *et al.* 2017, Figure 116, p172) are lower than in winter in all these areas off southern England, they are a lot higher along the Yorkshire coast where we have also observed higher bycatch rates, but our observed rates off the Yorkshire coast are twice as high in the winter as during the summer. This suggests again that higher bird densities do not necessarily correlate with higher bycatch rates.

Nevertheless, we should caution that just because we have not observed high seabird bycatch rates in areas of high bird density where it might be expected, does not mean that there are not pockets of high bycatch that we have not yet observed. This could be due to limited observer coverage. It could therefore be argued that all the highlighted areas of high winter fishing effort in Figure 10 that overlap with areas of high winter guillemot density might deserve more focused monitoring to better understand this.

6 Conclusion

Figure 6 shows the area with highest observed fulmar bycatch in UK longline fisheries. Figure 10 shows those areas with highest gillnet fishing effort in winter months and in shallow waters. Some of these areas are predicted to have the highest bycatch rates (birds per haul) based on analysis of our observer data (Figures 11 & 12). These areas are also therefore probably suitable candidates for mitigation trials to test the efficacy of different mitigation measures and may also provide a useful starting point for the development and encouragement of industry to use voluntary codes of conduct that might help reduce seabird bycatch overall.

Conducting effective and conclusive seabird bycatch mitigation trials will take significant planning and a lot more effort than simply producing maps from existing data. The key issues will be firstly to identify potential mitigation measures, then to find skippers who are willing and able to assist in carrying out trials, and finally to develop a suitably robust procedure for analysis and interpretation of results. To underpin that, a long-term focus and associated funding will be required to find satisfactory and implementable solutions to address this issue.

References

Bibby, C. 1971. Auks drowned in fish nets. *Seabird Reports*, **2**, 48–49.

Bradbury, G., Shackshaft, M., Scott-Hayward, L., Rexstad, E., Miller, D. & Edwards, D. 2017. Risk assessment of seabird bycatch in UK waters. *Defra Project MB0126*. 272 pp.

Cobb, J.L.S. 1976. Seabird mortality. *Bird Study*, **23**, 299–300.

Coram, A., Kingston, A. & Northridge, S. 2015. Seabird bycatch in static nets along the south coast of England – a preliminary assessment and recommendations for further work. 1–50pp.

Dunn, E. 1994. Interactions between fisheries and marine birds. 44 pp.

Dunn, E. & Steel, C. 2001. The impact of longline fishing on seabirds in the north-east Atlantic: recommendations for reducing mortality. *RSPB, NOF, JNCC & Birdlife International.*

Løkkeborg, S. 1998. Seabird by-catch and bait loss in long-lining using different setting methods. *ICES Journal of Marine Science*, **55**, 145–149.

Løkkeborg, S. 2011. Best practices to mitigate seabird bycatch in longline, trawl and gillnet fisheries-efficiency and practical applicability. *Marine Ecology Progress Series*, **435**, 285–303.

Melville, D. 1973. Birds in Salmon Nets. *Seabird Reports*, **3**, 47–50.

Murray, S., Wanless, S. & Harris, M.P. 1994. The Effects of Fixed Salmon Salmo-Salar Nets on Guillemot Uria-Aalge and Razorbill Alca-Torda in Northeast Scotland in 1992. *Biological Conservation*, **70**, 251–256.

Northridge, S., Kingston, A. & Coram, A. 2020. Preliminary Estimate of Seabird Bycatch by UK Vessels in UK and Adjacent Waters. (In Prep). 37 pp.

Quayle, H. 2015. Filey Bay: Safe Seas for Seabirds December 2015. 17 pp. Available from: https://www.rspb.org.uk/globalassets/downloads/documents/campaigning-for-nature/case-studies/rspb-filey-bay-safe-seas-for-seabirds-2015-report.pdf

Robins, M. 1991. Synthetic gill nets and seabirds. 68 pp.

Tasker, M.L., Camphuysen, C.J.K., Cooper, J., Garthe, S., Montevecchi, W.A., & Blaber, S.J.M. 2000. The impacts of fishing on marine birds. *ICES Journal of Marine Science*, **57**, 531–547.

Žydelis, R., Bellebaum, J., Österblom, H., Vetemaa, M., Schirmeister, B., Stipniece, A., Dagys, M., *et al.* 2009. Bycatch in gillnet fisheries – An overlooked threat to waterbird populations. *Biological Conservation*, **142**, 1269-1281. Available from: https://www.sciencedirect.com/science/article/abs/pii/S0006320709001001?via%3Dihub [Accessed 28 March 2012].

Acknowledgements

We are grateful to numerous reviewers who have provided comments and suggestions over several iterations of this study, notably to Matt Parsons of JNCC who has provided detailed and thoughtful feedback throughout the gestation of the report. None of this work would have been possible without the support and funding that we have received from Defra and Marine Scotland to collect data on the bycatch of protected species in UK fisheries for over 20 years. We could not have collected these data without the dedicated work of our observer team, or without the co-operation and assistance from the skippers and owners of over 300 vessels that we have sampled since 1996. Finally, we appreciate very much the work of Charles Paxton who has revised a previous version of Appendix 1 – a statistical analysis of observed seabird bycatch rates.

Appendix 1: GAM Analysis of bycatch data (Charles Paxton)

Model outputs are presented below from a logistic GAM analysis of Guillemot and Cormorant bycatch (presence or absence per haul, n = 18178) in the gillnet fisheries, using water depth, month of the year, net type, sea state, and mesh size category (S/M/L), trip and ICES rectangle as potential explanatory variables. Trip number and vessel identity were considered both as fixed effects and random effects, but no effect was found. Water depth was considered in a tensor smooth with month and as a thin plate spline changing with mesh size category. The other variables were considered as thin plate splines with up to 10 degrees of freedom except month that was initially considered as a cyclic smooth interacting with mesh size category.

Some input variables were correlated with each other so the results should be interpreted with caution if considering ultimate mechanisms of causation. Model selection was backward from a model with all the variables listed above with a $P<0.05$ inclusion criteria.

Cormorant bycatch

Only 0.2% of considered hauls (i.e. those with full set of predictor variables) had cormorant bycatch. The final model selected consisted of soak time, water depth and month. The two former variables being linear in effect (on the scale of the link function). Soak time had the biggest effect in raising the overall probability of bycatch (Figure A3 vs others). However, there was little data at prolonged immersion times. The total explained deviance was 19.7%.

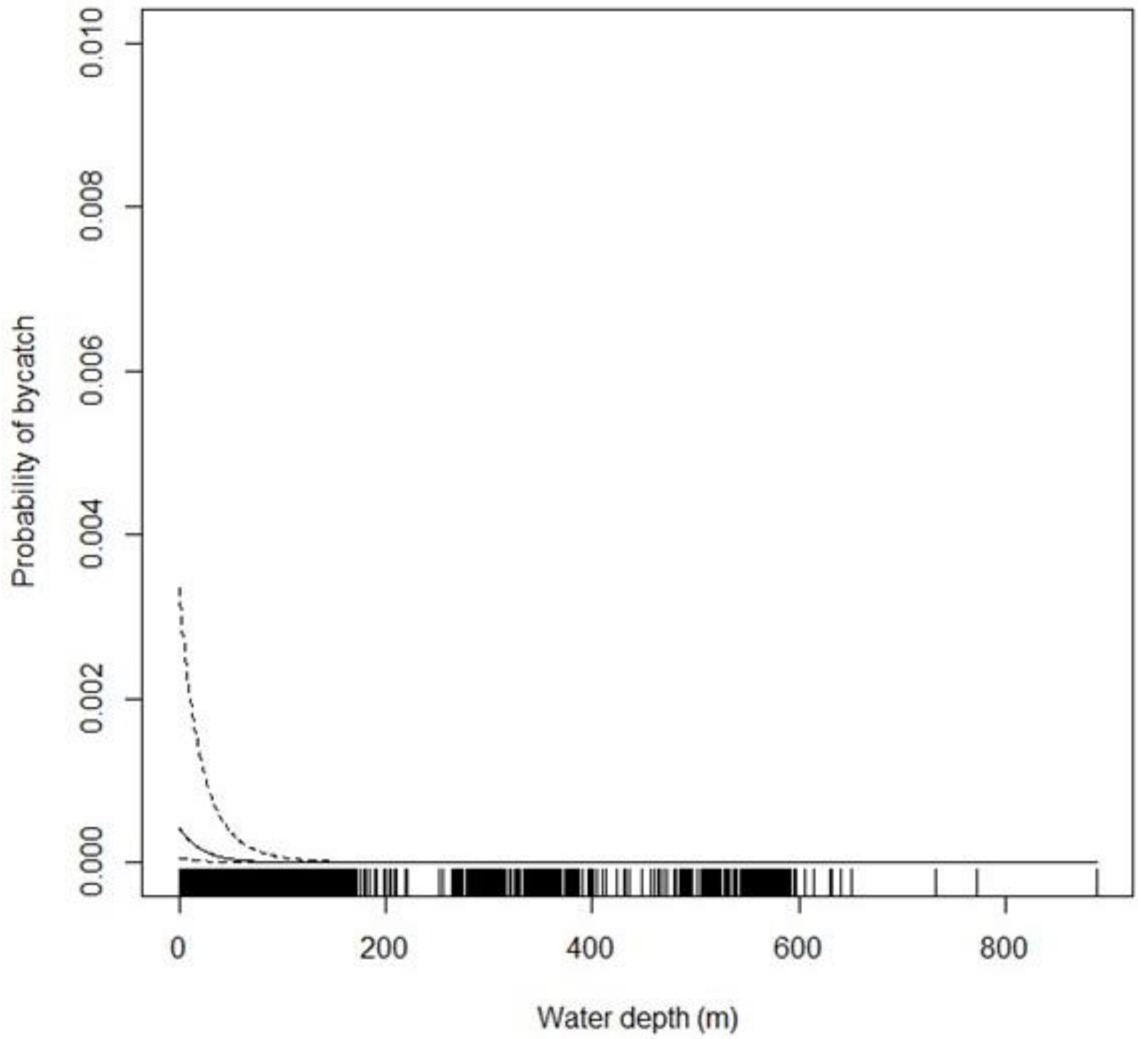

Figure A1. Effect of water depth on cormorant bycatch. Solid line: predicted probability of capture assuming deployment in June, with a mean sea state and soak time. Dashed lines: 95% confidence interval boundaries. Rugplot shows intensity of sampling by depth.

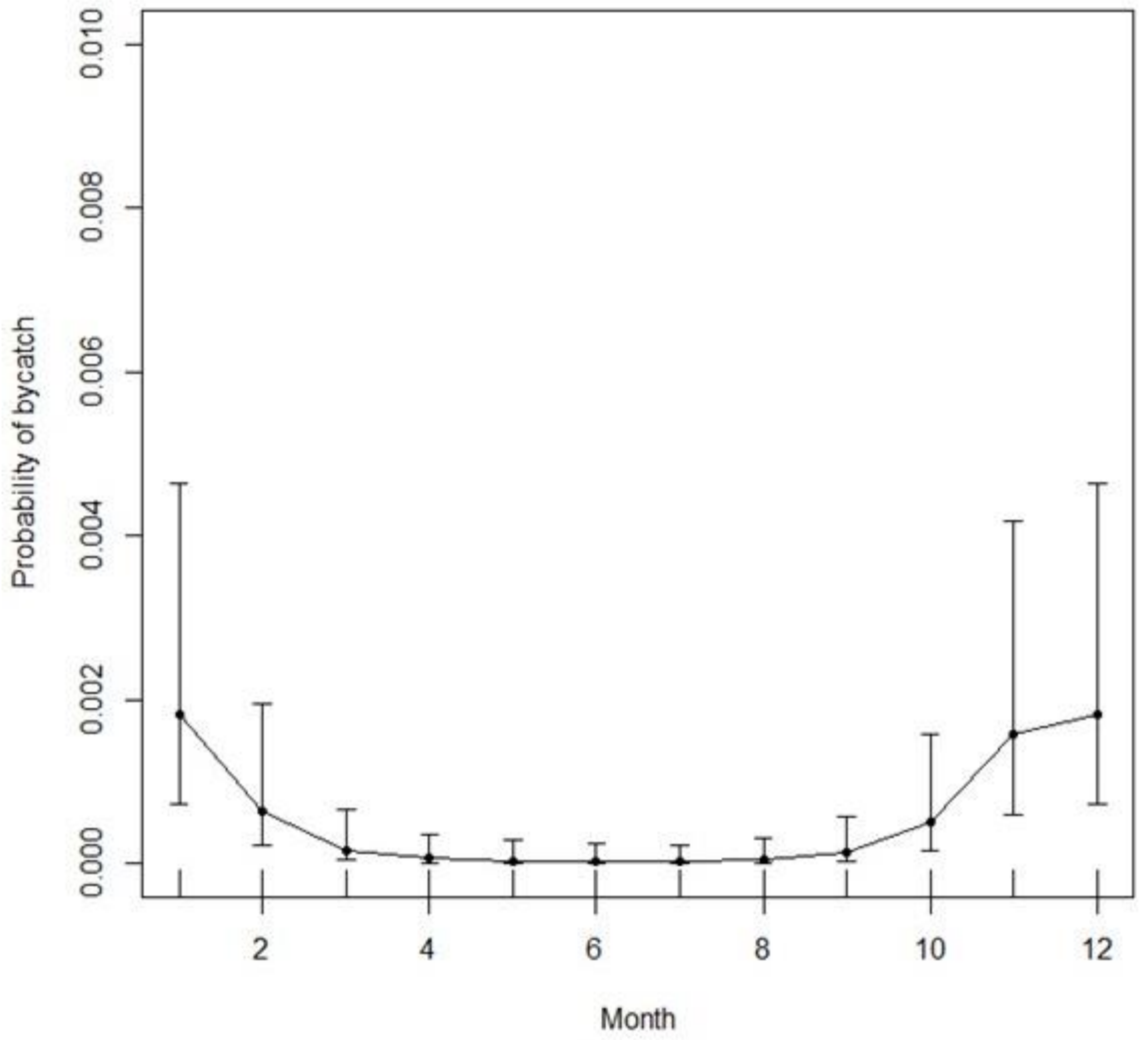

Figure A2. The effect of month on cormorant bycatch. Points: predicted probability of capture, with a mean sea state, water depth and soak time. Vertical lines: 95% confidence intervals.

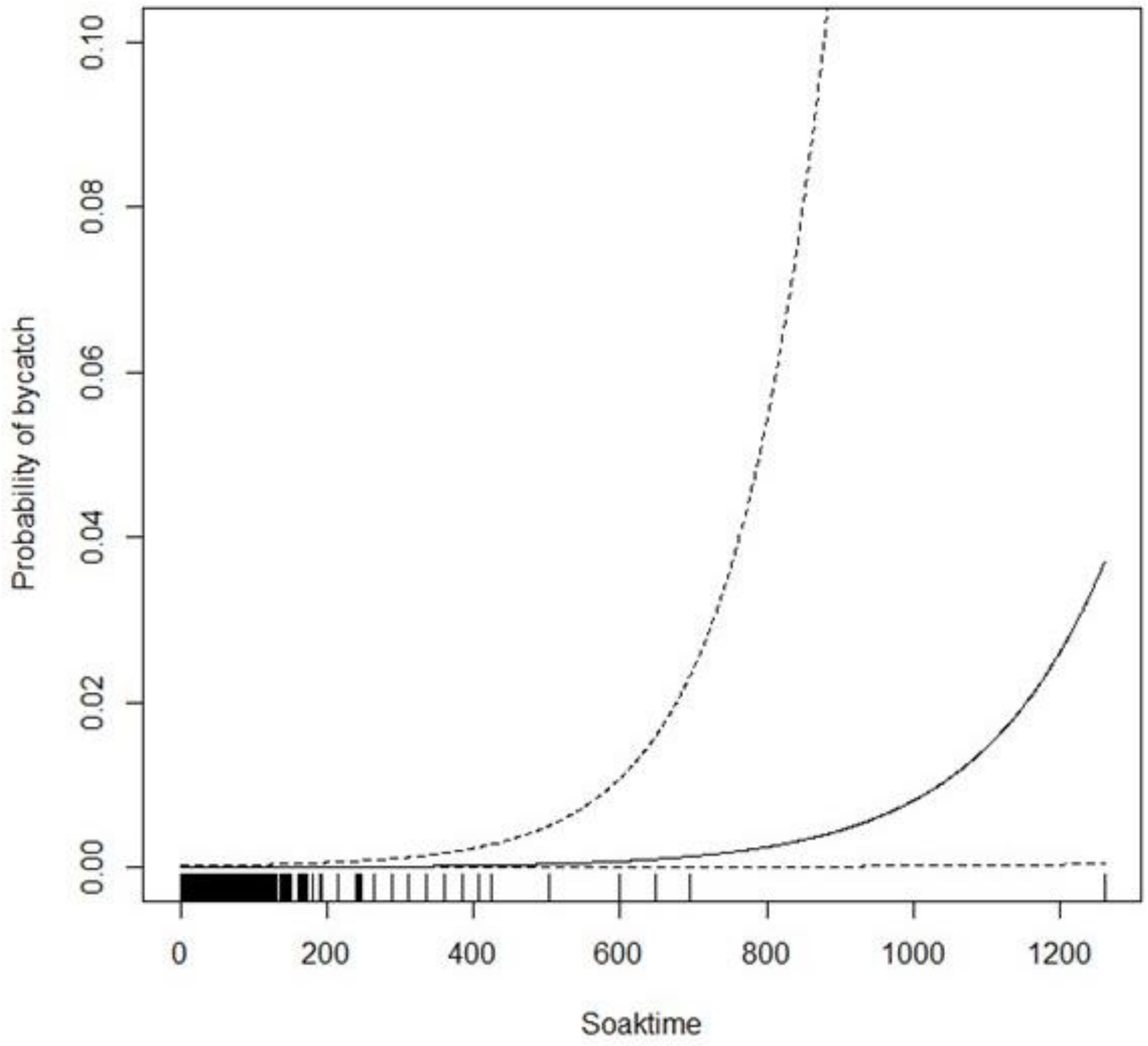

Figure A3. Effect of net soak time on cormorant bycatch. Solid line: predicted probability of capture assuming deployment in June, with a mean sea state and water depth. Dashed lines: 95% confidence interval boundaries.

Guillemot bycatch

Only 0.8% of hauls had guillemot bycatch. The final model selected consisted of a 2D smooth of water depth and month (Figure A4) and an interaction of month by mesh size category (Figure A5). The total explained deviance was just 16.3%.

Here the effect of mesh size (Figure A5) contradicts what a simple analysis catch by mesh size indicates (Table 2). The reason for this is that smaller meshes are employed at shallower depths, so the mesh category effect differs from the simple analysis when depth is included as an effect as well.

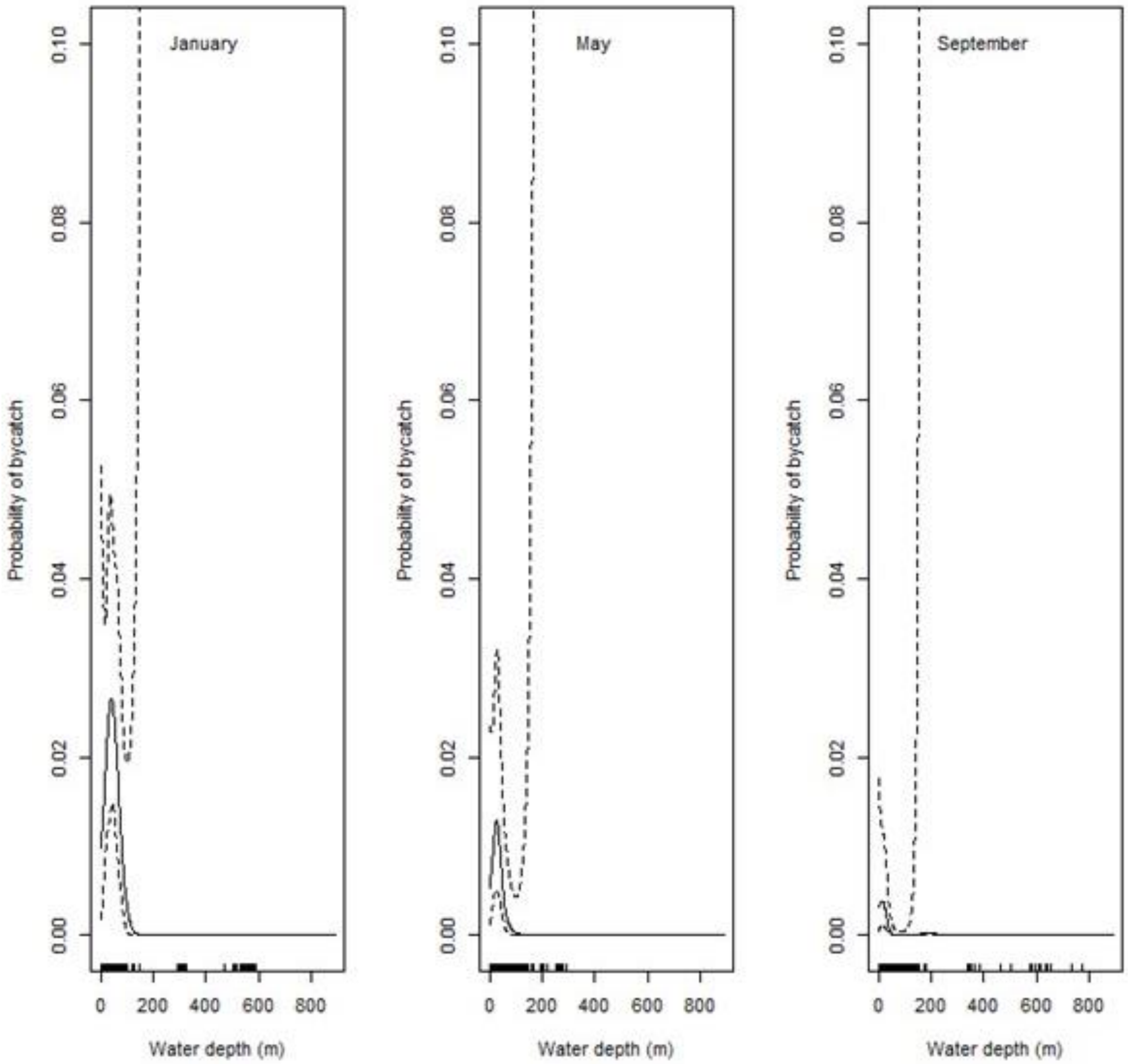

Figure A4. Effect of water depth on guillemot bycatch by month assuming a medium mesh size. Solid line: predicted probability of capture. Dashed lines: 95% confidence interval boundaries.

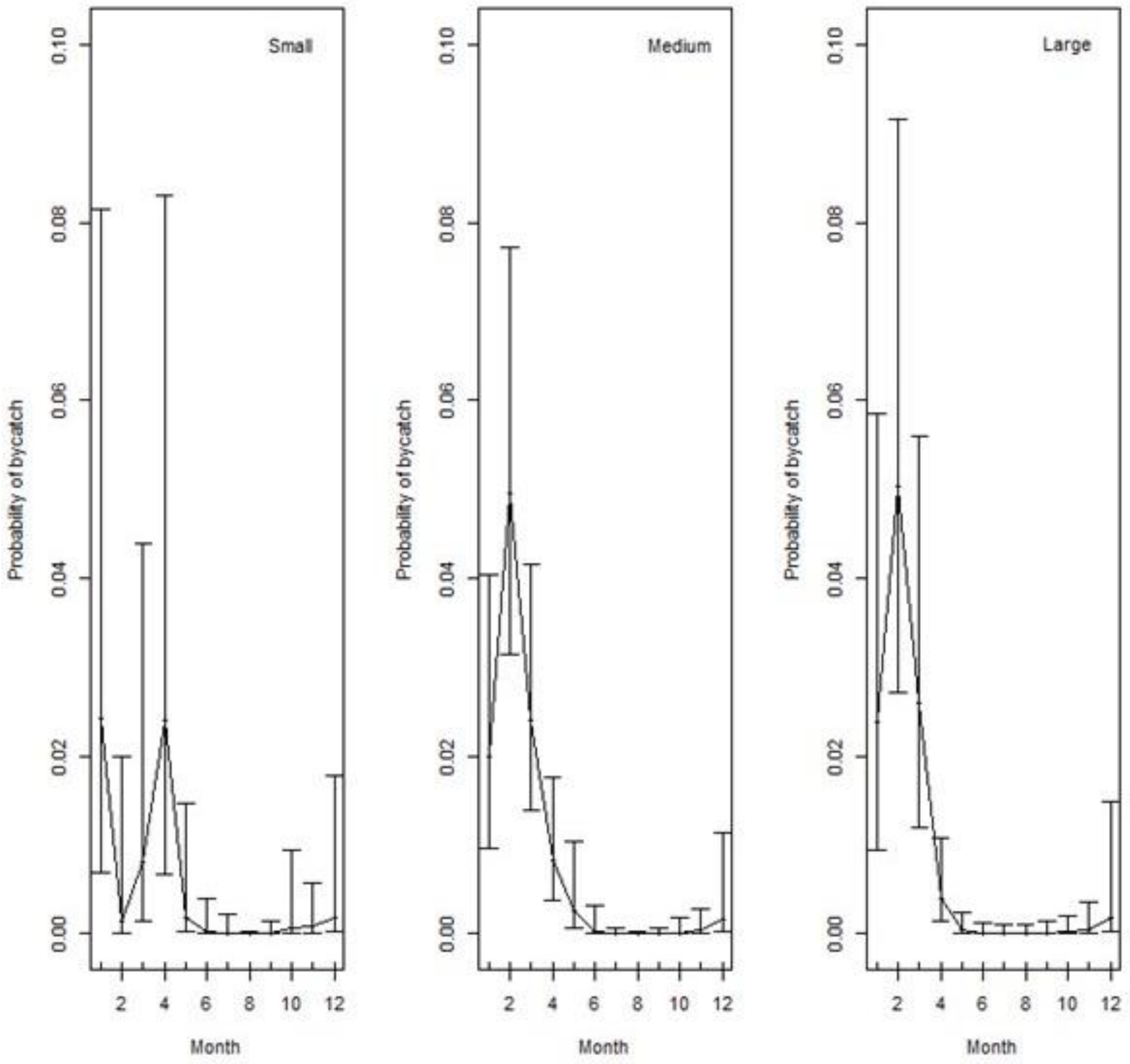

Figure A5. Effect of guillemot bycatch by month with from left to right small, medium and large mesh nets. Points: predicted probably of bycatch, vertical lines 95% confidence intervals.

Appendix 2: Bycatch risk assessment: a comparison of methods

As part of the present study, we were asked to provide further analysis of the BMP data and accounts of bycatch risk (as shown in Bradbury *et al.* 2017) and fishing effort to identify “hotspots” of bycatch in UK waters for the purpose of informing options for regional pilot areas for targeted bycatch mitigation and monitoring. The work of Bradbury *et al.* (2017) combines a wide variety of metrics into graphic representations of spatial risk to seabird populations from fishery bycatch. These maps are not easy to understand, combining as they do aspects of bird biology (demography, rarity, behaviour, habitat preference, assumed catchability) with seasonal population density maps and seasonal maps of fishing effort by over 15 m vessels in three broad fishing gear categories. A lot of assumptions and expert opinions are synthesised into maps that show where one might expect greatest impacts on seabird populations, with the stated aim of indicating where research (including the placement of observers) should be focused. Making comparisons with our data is very difficult, not least because the Bradbury *et al*. report provides very finely grained overviews of risk, whereas our data are very coarse grained (and patchy) in comparison.

Within the Bradbury et al report, fishing gears or methods have been aggregated into three depth categories: surface, pelagic (midwater) and benthic (demersal). Of course, all fishing gears spend some time at the surface where all birds are vulnerable, and that vulnerability may be heightened by specific foraging strategies relating to the fishery activity. This is recognised by the authors and is addressed by reviewing the literature and seeking expert opinion on the susceptibility or entrapment risk of each species to each fishery category. To account for multiple entrapment risks for any specific gear type, gears were allocated to more than one category. Thus, demersal longline fishing, which is well known to catch surface feeding fulmars, was classified for this reason as a ‘surface gear’ and a ‘pelagic gear’ and a ‘benthic gear’ to take account of this. In fact, all the gear types described are included in both surface and either benthic or pelagic categories (Bradbury *et al.* 2017, Table 8 p.26). While this approach works well for a broad overview of entrapment risk, it also conceals a great deal of more nuanced risk associated with specific gears used in specific areas which observational data can address, with the result that the entrapment risk in a particular area associated with a particular gear type as estimated in the Bradbury et al analysis may be over or under-emphasised. It also makes difficult any comparisons between spatially fine-grained data aggregated across gear types and more coarsely grained observational data that are focused on detailed differences between gear types.

In terms of the description of fishing effort, and as Bradbury *et al.* (2016) acknowledge, while fishing effort distribution identified by VMS provides accurate positional data, VMS is only carried by larger vessels (those over 15 m in the report). While this is useful for certain sectors like the offshore longline fishery, it is practically useless for gillnet vessels as more than 95% of boats that use nets are under 15 m and 96% of reported gillnet trips recorded in 2017 in official UK statistics were made by vessels of 12 m or less. The majority of our data in the BMP relate to gillnet fishing, but the risk posed by such fisheries is hardly represented in the Bradbury et al report which seriously constrains our ability to make any useful comparisons.

The fact that the data we use in this report are collected at very different spatial scales from those used by Bradbury *et al.* (2017) also compounds the problems with making comparisons. Our data do not provide complete coverage of the fishing areas around Britain, they are generally patchy, and the fishing effort data that we have available is only reported at the ICES rectangle level (30 minutes latitude by one degree longitude or approximately 30 x 30 nm at UK latitudes). In comparison with the interpolated maps provided by Bradbury *et al.* (2017), these are quite crude representations.

It would be possible to use calculated bycatch rates, overlain with seabird density maps and maps of fishing effort, to predict where total bycatch is likely to be highest, but this is a complicated task, and one for which the issue of granularity would cause problems. For example, at a fine scale, places where we observed bycatch in one year are not necessarily where fishing effort will be deployed in subsequent years. Similarly, fishing effort in subsequent years may be in areas that have not been sampled. Consequently, bycatch rates are typically calculated over relatively large areas – such as ICES Divisions, and not normally at smaller scales such as ICES rectangles or the even smaller 3x3 km areas used for seabird density maps in Bradbury *et al.* (2017). The use of hot spots in the present report can therefore be criticised from this perspective as the visualised data may be taken to imply a generalised risk in an area which is based on a limited sampling window.

We can explore how well our observations match some of the predictions from Bradbury *et al*. who also provide maps of all seabird 'vulnerability' – a combination of log species density and expected 'susceptibility' to bycatch. Crudely speaking, we might expect areas of high seabird 'vulnerability' to coincide with areas where we have observed high bycatch rates. To check whether our data agreed with the general predictions of Bradbury *et al.* we have plotted our observed bycatch rates in each of three different categories of fishery (surface = longline, pelagic = midwater trawl, and benthic = demersal gear) against all seabird species vulnerability (as defined by Bradbury *et al.* 2017) to each of their gear depth categories. (Note that our gear types do not map neatly onto the Bradbury *et al.* (2017) gear depth categories: all three types for example are included in the 'surface' gear category within the Bradbury *et al.* (2017) maps).

Bradbury *et al. (2017)* define vulnerability as follows:

$$\boldsymbol{Bycatch\ vulnerability} = \Sigma\ ((\mathbf{ln}(\boldsymbol{density}_{species} + \mathbf{1})) \times \boldsymbol{SSI}_{species})\ \text{(Equation 2, p. 24)}$$

The SSI – species sensitivity index – is based on a combination of conservation factors (the % of biogeographic population of each species that is in the UK, and the UK threat status), demographic/ecological factors (adult survival rate and habitat specialisation) and behavioural factors (entrapment risk and response to fishing activity). Entrapment risk is gear-category specific. All factors were scored 1 through 5 and combined by summing the first four and multiplying by the product of the last two in the list above. SSI values range from 6 to 96 and are dimensionless scaling factors which, when applied to species specific densities (birds per km^2), provide an index of vulnerability for each of the three gear depth categories. While vulnerability can be expressed for any individual seabird species, within the report it is presented as a composite for *all species* as described in the equation above.

To compare our observations with the seabird population risk maps in Bradbury *et al.* for each individual haul (approximately 21,000) that we have observed, we have used QGIS to identify the associated vulnerability score at that location from the electronic coverages associated with the Bradbury *et al.* (2017) report. All our observed *longline* hauls were associated with the seabird - *surface gear* vulnerability coverages ('surf'); while all our *midwater trawl* observations were linked to seabird *pelagic gear* vulnerability values, and each *static net* observation was linked to a *benthic gear* vulnerability value.

We then grouped all our observations into evenly spaced 'vulnerability' bins, based on the vulnerability score that Bradbury *et al.* (2017) predicted for that location, and then calculated mean bycatch rates (birds per haul or fishing operation) for each of the vulnerability bins. Vulnerability scores range from 0 to 1600 and we have grouped them into 17 bins of 100 units. Clearly if the maps of seabird vulnerability scores reflect areas of actual high bycatch risk, we might expect that our observations in high vulnerability score bins (i.e. in areas with high vulnerability) would have high seabird bycatch associated with them, and conversely locations with low vulnerability scores would be associated with infrequent bycatch events. Note that we used any-and-all observed seabird species bycatch in this initial test, comparing these with 'all seabird' vulnerability scores in Bradbury *et al.* (2017). We were unable to unpack the species and gear components of these vulnerability maps from the report or from the data that we had access to.

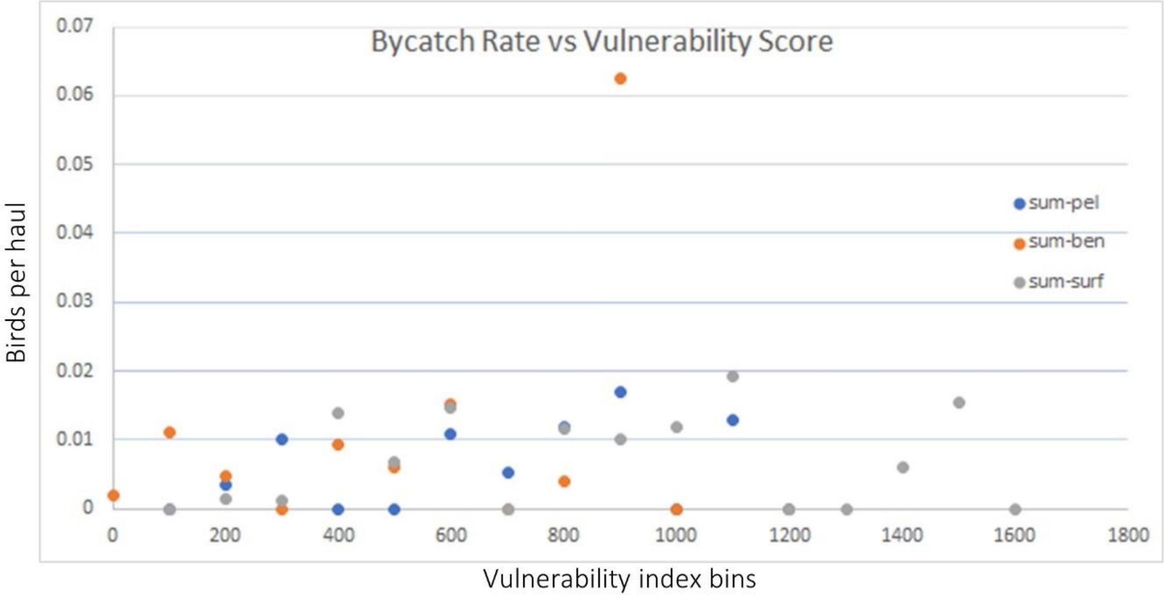

Figure A6. Comparison of vulnerability data to observed bycatch rates (birds per haul).

As shown in Figure A6, there is no clear relationship between where seabird vulnerability scores are highest in the Bradbury *et al.* (2017) maps and where we have recorded higher bycatch rates. This is perhaps not surprising – as several gear types and several seabird species have been mixed in the vulnerability indices, while our data too are for all seabirds in each of three gear types.

A somewhat clearer picture emerges if we plot just the observed longline fulmar bycatch against *all seabird vulnerability* to *surface gears*.

Figure A7 shows a stronger relationship between 'all-seabird vulnerability' and recorded fulmar longline bycatch. Nevertheless, in those areas with highest seabird vulnerability scores (greater than 450) we have not observed any longline bycatch (because we have made no observations there). This is because those areas are in the North Sea and vulnerability scores there are driven by the densities of bird species other than fulmars. This figure simply tells us that in areas with lots of seabirds (including fulmars), fulmar bycatch tends to be higher.

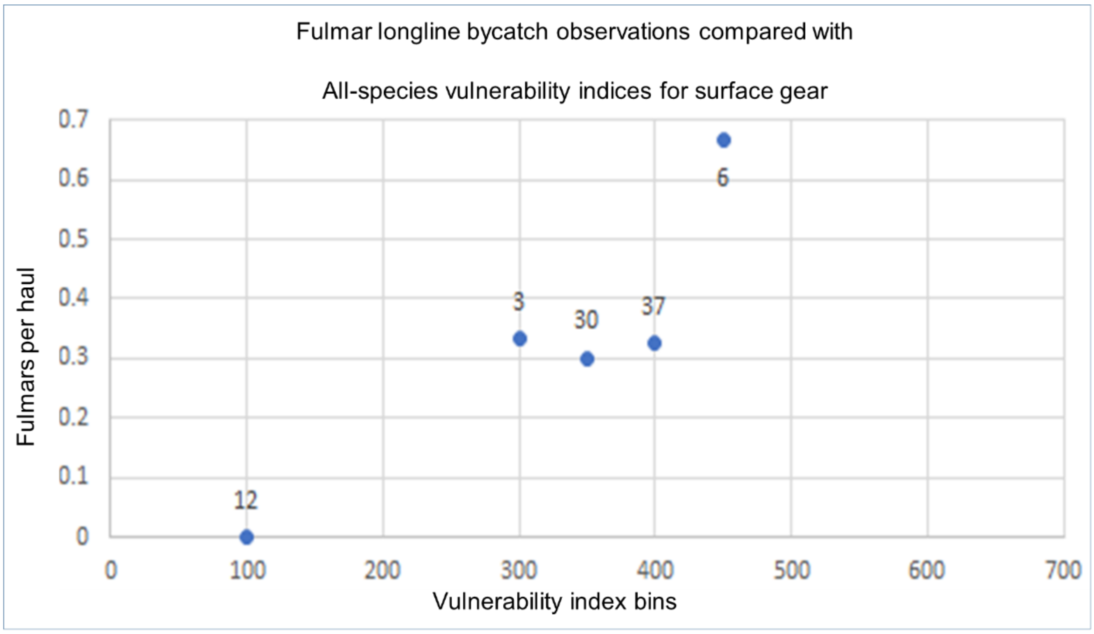

Figure A7. All species vulnerability to surface gears against observed fulmar bycatch rates.

By closer focus on specific gears and a particular species, the relationship between observed and predicted bycatch is improved. To investigate further we dropped the vulnerability scores generated by Bradbury *et al.* (2017) and focused solely on seabird density (i.e. making no further assumptions about susceptibility to generate any sort of vulnerability score). To this end, we were provided with additional bird density data by JNCC on which the vulnerability scores used by Bradbury *et al.* (2017) were based, although these were not published.

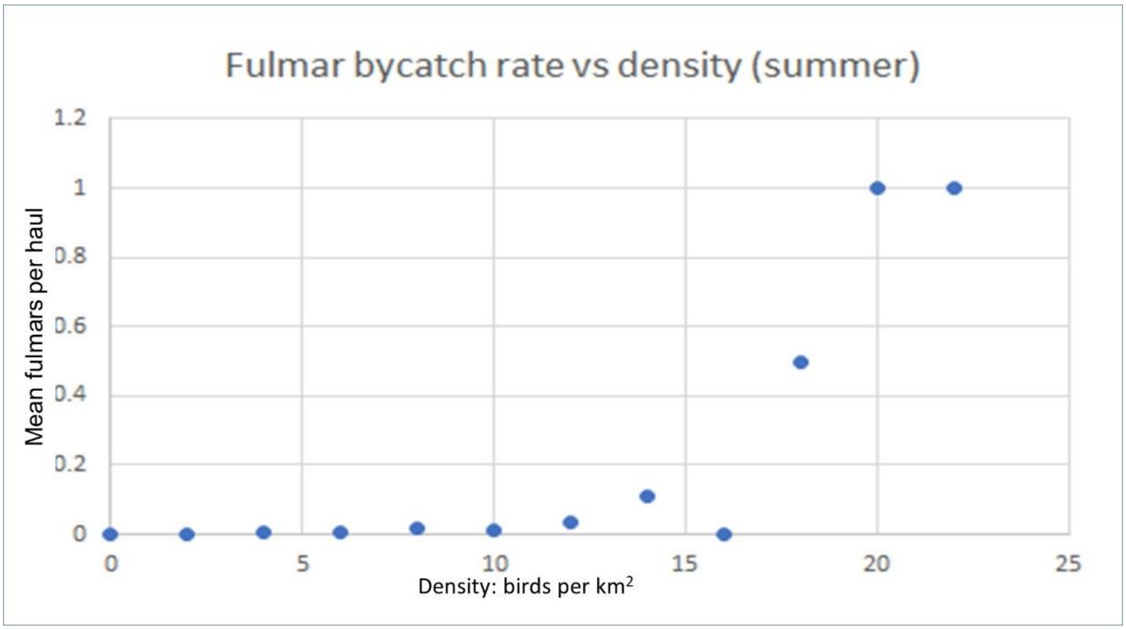

Figure A8. Fulmar density plotted against observed bycatch rates per haul.

Figure A8 shows that fulmar density in summer is indeed a reasonable predictor of fulmar longline bycatch rate. Note however, that using either fulmar vulnerability or fulmar density surfaces to *predict* areas of highest *overall* (total) bycatch could only be done if one knows where fishing effort will be distributed. Comparing the top and bottom lefthand panels in Figure 2 (data collected since 2015 versus data from the 2000s, respectively) suggests that the longline fishery has changed its distribution over time, assuming that sampling effort coverage (as shown in Figure 2) within each time period was representative of the wider fishery effort in the same period. Thus, it is unclear that population risk maps produced from vulnerability surfaces combined with past fishing effort distribution will predict where the highest levels of population risk will be in the future. It is also noteworthy that the relationship between fulmar density and longline bycatch is not linear, so that density maps are likely to be most useful in determining relative rather than absolute levels of bycatch risk.

The general conclusion from our comparison of our observed bycatch rate data with the predicted vulnerability and risk maps in Bradbury *et al.* (2017) is that there is a poor degree of agreement between the two datasets. However, the Bradbury *et al* report is focused on providing a series of overviews of population level impacts, which is why so much effort is devoted within that report to defining the Species Susceptibility Indices. In looking for areas of high bycatch rate (numbers of birds expected per unit of fishing effort), a simpler index is provided by using maps of bird density, with the proviso that there is not a linear relationship between bird density and bycatch rate.

Much more work could be done exploring this issue; the relationship between expected bird density and expected bycatch rate could be examined in more detail and at a range of spatial scales, but this is a task that extends well beyond the limited scope of this report.