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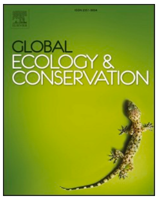

Bycatch of northern fulmar (*Fulmarus glacialis*) in Norwegian longline fisheries: Assessing spatiotemporal variations in scale and risk to improve management

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ABSTRACT

Seabirds are vulnerable to bycatch in longline fisheries but for most species the impacts are largely unknown. To address this knowledge gap, studies can estimate bycatch directly using observations or calculate the theoretical risk of bycatch using overlap indexes. Here we quantify the scale and risk of bycatch of northern fulmar (*Fulmarus glacialis*) in the Norwegian offshore longline fishery using a ten-year time series of bycatch observations from a reference fleet programme, and large-scale datasets of fishing activity and northern fulmar distribution. We estimated an average of 0.01 (95 % CI: 0.008–0.03) northern fulmars bycaught per 1000 hooks, which results in a highly varying estimated annual bycatch of between 51 and 16242 (95 % CI) northern fulmars per year, with the largest hotspot in the Norwegian Sea during June-August. We compared these estimates with overlap indexes calculated for northern fulmars and the same fishing activity. This pinpointed the highest risk of bycatch within the breeding season, where fishing activity increased in the waters around the largest cluster of breeding colonies in the northeast Atlantic. Strong correlations between estimated bycatch and calculated overlap indexes validate overlap indexes as an indirect evaluation of risk and strengthen evidence for management decisions based on the spatial and temporal trends identified in our analyses.

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1. Introduction

Seabirds are vulnerable to bycatch in commercial fisheries and the impact has long been linked to population declines (Tasker et al., 2000; Lewison and Crowder, 2003; Gilman et al., 2005; Anderson et al., 2011; Dias et al., 2019). Furthermore, the list of fisheries where seabird bycatch is of concern is growing as monitoring programmes are expanded to record incidental bycatches (Bærum et al., 2019; Christensen-Dalsgaard et al., 2022; Glemarec et al., 2020; Watkins et al., 2008). Commercial longline fisheries can result in particularly high seabird bycatch rates, with a global estimate of between 160,000 and 320,000 seabirds killed annually as a direct consequence of longline operations (Anderson et al., 2011). Considering that some seabird species are overrepresented, e.g. opportunistic scavengers such as petrels, albatrosses and fulmars (Zhou et al., 2019), and that there seems to be clear hotspots for bycatch (Clay et al., 2019), it is not surprising that some seabird populations are negatively impacted (Ramírez et al., 2024). Consequently, reliable estimates of numbers of birds killed in specific fisheries, and especially how these estimates vary in time and space, are required to inform adaptive management frameworks. A mean bycatch estimate across a fishing fleet might, for instance, be of little applied value when developing effective mitigation actions, particularly if bycatch mostly occurs in a small area and short period. Management actions based on the assumption that all fishing operations have similar rates of seabird bycatch might also be controversial and difficult to implement among fishers if the mitigation does not reflect their own experience of bycatch.

One important influence of estimated bycatch rates is the expected distribution of bycatch events. Seabird bycatch typically follows a pattern of most fishing events with zero or very few bycaught seabirds and then some events with very high bycatches observed (Bærum et al., 2019; Christensen-Dalsgaard et al., 2019; Glemarec et al., 2020; Parsa et al., 2020). Seabird bycatch is thus not only challenging to accurately forecast at the level of a fishing event, but it also requires a large monitoring effort to observe a representative sample of bycatch situations (Fangel et al., 2015). Furthermore, there is often limited information available to disentangle the cause of the observed bycatch distribution, which can be both natural (e.g. variation in overlap between bird distribution and fisheries) and vessel-dependent. For example, bycatch rates are expected to vary between vessels (Pennington and Helle, 2011), but a lack of information on the many potential reasons behind this variation (e.g., use of bycatch reduction devices, willingness to participate) means that variations across vessels must often be captured as a random effect (Bærum et al., 2019; Dietrich et al., 2009; Gilman et al., 2014).

Northern fulmar (*Fulmarus glacialis*; hereafter 'fulmar') is one of the seabird species most frequently taken as bycatch in the northern hemisphere (Anderson et al., 2011; Bærum et al., 2019; Fangel et al., 2017; Hedd et al., 2016; Ramírez et al., 2024). Fulmars are circumpolar boreo-arctic surface-feeding seabirds with an extensive offshore foraging range during their entire life cycle, even when attending their breeding colonies (Edwards et al., 2013). As generalist predators and scavengers feeding on the sea surface, fulmars often aggregate in large numbers around fishing vessels (Camphuysen and Garthe, 1997), making them vulnerable to bycatch. Being long-lived with a low reproductive rate, fulmar populations are especially sensitive to increases in adult mortality through bycatch (Croxall, 1987; Tasker et al., 2000), with a significant bias towards bycatch of adult male fulmars (Beck et al., 2021; Fangel et al., 2017). There is therefore an urgent need to understand both the magnitude of bycatch of fulmars and spatiotemporal variations to inform management.

This study aims to quantify the scale and risk of incidental bycatch of fulmar in the Norwegian offshore longline fishery, with a focus on spatial and temporal variations. We address this aim with the following three objectives:

1) Estimate total bycatch of fulmar in the Norwegian offshore fishery by extrapolating a time series spanning ten years (2012–2021) of complete catch records by three participating fishing vessels
2) Generate overlap index maps for the fishery by combining large-scale datasets on fishing effort and predicted fulmar distribution previously estimated by Fauchald et al. (2021).
3) Compare how estimated bycatch and overlap indexes vary in time and space to explore how different metrics best inform the management of bycatch.

Overlap indexes are often used to indirectly predict risk of negative impacts in the absence of direct observations (Le Bot et al., 2018). However, direct observations cannot be completely reliable given that they contain immeasurable biases such as misreporting. The rich data sources used in this study therefore offer a rare opportunity to compare and validate both methods. Congruence between methods will provide support for overlap indexes as a management tool where bycatch estimates are unavailable (e.g. distant water fisheries) and discrepancies will offer insight into the potential bycatch drivers to inform future research.

2. Materials and methods

2.1. *Case study fishery and data*

The Norwegian offshore longline fishery is defined here as Norwegian vessels greater than 28 m overall length using longlines to target demersal fishes in the Northeast Atlantic. The fishery operates year-round following seasonal distribution trends and quota availability of target stocks including gadoid species (cod, *Gadus morhua*; haddock, *Melanogrammus aeglefinus*; ling, *Molva molva*; and saithe, *Pollachius virens*), wolffish (Genus: Anarhichas), redfish (Genus: Sebastes) and Greenland halibut (*Reinhardtius hippoglossoides*).

2.1.1. *Norwegian Reference Fleet*

The Norwegian Reference Fleet is a group of active fishing vessels that are paid by the Norwegian Institute of Marine Research (IMR) to collect data on their catches and fishing activity, including bycatches (Clegg and Williams, 2020). Vessels apply for four-year contracts through an open tender process that aims to select typical vessels in priority fisheries. An expert panel filters applications for eligibility and representativeness, after which the contract is randomly awarded. Fishers are trained in sampling protocols and species identification (including seabirds). This training, coupled with the more coastal ranges of other gull species, means there is a negligible risk of misidentification in this study of offshore fisheries. IMR technicians are responsible for quality assuring data and maintain regular contact with vessels, including visits, to check adherence to protocols. This also offers an opportunity to informally chat with crew to get an honest insight into their perspectives of the science and management of their fisheries. An agreement between fishers, scientists, and Norwegian authorities ensures data collected for the Norwegian Reference Fleet programme is not used for control or enforcement purposes. This agreement has not been compromised in the 24-year history of the programme, creating a trustful incentive for fishers to report honestly.

Offshore longline vessels in the Norwegian Reference Fleet sample seabird bycatch using a single-stage cluster sampling design (Lohr, 2021), recording total number of seabirds bycaught for each fishing day. Ten Norwegian Reference Fleet vessels were active in the offshore longline fishery over the entire study period (2012–2021) out of a total of 50 vessels in the fishery. This study uses data from three of these vessels that voluntarily provided seabird bycatch records (Table 1). In addition to fulmar bycatch, seven other seabird species were bycaught at negligible rates (during four fishing days) so were excluded (Supplementary Materials; Table A1).

2.1.2. *Electronic Reporting System (ERS)*

All vessels in the Norwegian offshore longline fishery are required to submit a daily summary report of fishing activity to the Norwegian Directorate of Fisheries using the electronic reporting system (ERS). Alongside catch composition, each report includes fishing effort (in this case number of hooks), and a single location that best describes the geographical distribution of fishing activity that day. ERS data for all fishing activity in the Norwegian offshore longline fishery were obtained under the Norwegian licence for public data (NLOD) made available by the Norwegian Directorate of Fisheries (Fig. 1). We highlight here that it is mandatory to record all bycatch of seabirds in ERS reports (ERS-forskriften, 2009, § 12). However, no ERS reports included bycatch of seabirds in the dataset.

2.1.3. *Fulmar distribution (SEATRACK)*

Location information was obtained from global location sensor (GLS) loggers deployed by the SEATRACK project, an international research programme that aims at mapping the non-breeding distribution of seabirds breeding in the North Atlantic (https://seapop.no/en/seatrack/). Loggers were deployed on 312 adult breeding fulmars from eleven colonies spanning Norway (including Svalbard), the United Kingdom, the Faroe Islands and Iceland (Fauchald et al., 2021). Locations were estimated using procedures described in Bråthen et al. (2021) and sample biases corrected by procedures described in Fauchald et al. (2019). Location data were combined with environmental data and population count data to model the monthly distribution of fulmars from all breeding colonies in the Northeast Atlantic (Fig. 2, Fauchald et al., 2021) with a 0.25° x 0.25° cell resolution.

2.2. *Statistical analyses*

All analyses were done using R statistical computing software (v4.3.1; R Core Team, 2023). Models were fitted using the mgcv (Wood, 2011) package, and residual diagnostics ran using the DHARMa package (Hartig, 2022). Spatial data were handled and visualised using the sf (Pebesma, 2018), ncdf4 (Pierce, 2019), and raster (Hijmans, 2023) packages.

2.2.1. *Estimated total fulmar bycatch*

We estimated the total fulmar bycatch in the Norwegian offshore longline fishery by extrapolating observed fulmar bycatches by the Norwegian Reference Fleet. This analysis was limited to fishing activity in the Barents Sea, Norwegian Sea, and North Sea (Fig. 1). Other fishing activity was excluded because the Norwegian Reference Fleet was not designed to be representative of these fisheries

Table 1
Summary of total and observed fishing activity (number of vessels and fishing days) in the Norwegian offshore longline fishery. Note that vessels are active over many years, so total unique vessels is not the sum of all years.

Year	Vessels		Fishing days	
	Total	Observed	Total	Observed
2012	37	2	8309	386
2013	35	2	7096	434
2014	25	2	5000	374
2015	27	2	5833	451
2016	28	2	5706	483
2017	28	3	5743	526
2018	28	3	5988	730
2019	29	3	6142	706
2020	29	2	5603	330
2021	26	2	5821	456
TOTAL (unique)	50	3	61,241	4876

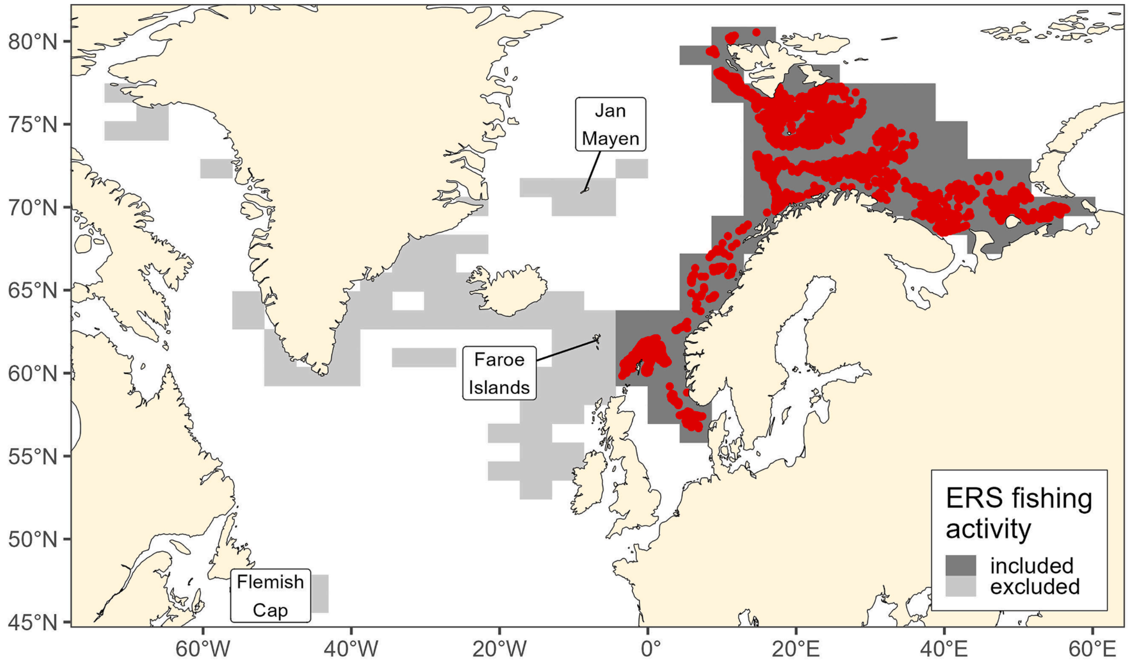

Fig. 1. Geographical extent of the Norwegian offshore longline fishery between 2012 and 2021. Fishing days by Norwegian Reference Fleet vessels used for bycatch modelling (red points) overlaid on total fishing activity based on data from the electronic reporting system (ERS; see Section 2.1.2). ERS reports are binned to a 1°x1° grid (See Section 2.2.2).

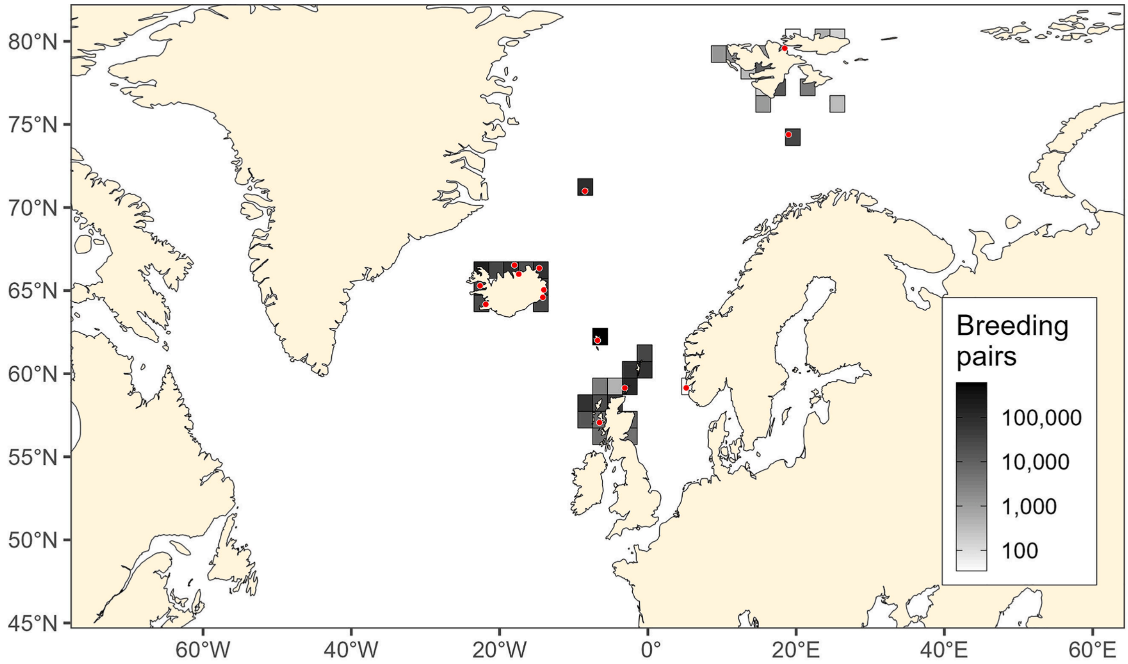

Fig. 2. Distribution of northern fulmar (*Fulmarus glacialis*) breeding pairs in the Northeast Atlantic used in SEATRACK models. Colonies tracked using global location sensor loggers are highlighted as red points. Population maps for colonies without any tracked birds borrowed information from the nearest tracked colony (Fauchald et al., 2021).

with different access rights for Norwegian vessels.

Numbers of fulmar bycaught per fishing day were estimated using generalised linear and additive mixed-effects models. We defined six candidate models (Table 2) of increasing complexity based on covariates that are recorded by all vessels through the ERS, allowing us to predict total bycatch in the fishery (Table 3). We included fishing effort (number of hooks) as an offset in all models to help comparison with other fisheries (Anderson, 2011). We also included vessels as a random intercept to account for repeated sampling by individual Norwegian Reference Fleet vessels. All models were fitted using a negative binomial response distribution to account for the patchiness of seabird distribution and bycatch events (Zar, 1999).

The temporal component was fixed as we assumed that bycatch was sufficiently described on the scale of months and years (models M2-M6; Table 2). Models M3-M5 aim to describe bycatches as a function of fulmar distribution, which could inform spatial management measures. We finally compared this to a model describing spatial variations as a two-dimensional nonlinear function (M6), which does identify the drivers behind the spatial variations and improve identification of hotpots.

Models were compared using Akaike information criterion differences (ΔAIC), such that the model with the best support from the data has $\Delta AIC = 0$ (Burnham and Anderson, 2002). Due to difficulties in interpreting mixed-effects models using conventional residuals, we used residual diagnostics generated from simulated ($n = 1000$) and scaled residuals. Patterns in residual variation were assessed using Q-Q plots and by plotting residuals against independent and dependent variables listed in Table 3 (Supplementary materials; Figure A1). Although there was a high risk of zero-inflation (93 % zero observations), we only considered a zero-inflation component if we detected residual patterns or if the distribution of expected number of zeros in the data did not sufficiently capture those observed (Zuur and Ieno, 2016).

Total fulmar bycatch in the offshore longline fishery was estimated using the final model (M6; Table 2) to predict bycatch for all fishing activity reported in the ERS in the same fishing areas (Figure A1). Uncertainty was estimated using a Monte Carlo approach (2000 replicates) applied to estimated model parameters. Fixed parameters were drawn from a multivariate normal distribution, and the random vessel parameter was drawn from a zero-centred normal distribution, then the 95 % confidence interval using the percentile method.

2.2.2. Overlap between fulmar distribution and fishing activity

We calculated a monthly relative overlap index using equations modified from Vanderlaan et al. (2008). Fulmar distribution data (Fauchald et al., 2021) and ERS reports of fishing days were aggregated to a 1 × 1° grid, which was deemed appropriate for longline fishing gear (White et al., 2019). For cell i for month t, the relative probability that a vessel was actively fishing was calculated as:

$$P_{rel}(fishing)_{i,t} = \frac{effort_{i,t}}{\sum_{i=1}^{n}\sum_{t=1}^{12} effort_{i,t}} \tag{1.2}$$

and similarly, the relative probability of fulmar presence in the respective cell was calculated as:

$$P_{rel}(fulmar)_{i,t} = \frac{density_{i,t}}{\sum_{i=1}^{n}\sum_{t=1}^{12} density_{i,t}} \tag{1.1}$$

Combining Eqs. (1.1) and (1.2), the relative monthly overlap index was then calculated as:

$$P_{rel}(overlap)_{i,t} = \frac{P_{rel}(fulmar)_{i,t} \times P_{rel}(fishing)_{i,t}}{\sum_{i=1}^{n}\sum_{t=1}^{12} P_{rel}(fulmar)_{i,t} \times P_{rel}(fishing)_{i,t}} \tag{1.3}$$

Overlap indexes sum to one across all cells and months, allowing for aggregation across different temporal and spatial scales, and are also expressed as percentages in the text to help readability.

2.2.3. Comparing estimated bycatch and overlap index

To explore the congruence between estimated bycatch and overlap indexes, we first converted estimated bycatch into a relative probability to allow for comparison with overlap indexes on the same scale:

$$P_{rel}(bycatch)_{i,t} = \frac{bycatch_{i,t}}{\sum_{i=1}^{n}\sum_{t=1}^{12} bycatch_{i,t}} \tag{1.4}$$

Then for each month we ran a Spearman's rank correlation test between overlap index and estimated bycatch. To explore finer scale spatiotemporal variations in differences between the two methods, we mapped the difference between the relative probability of bycatch and overlap, such that a positive (or negative) value indicates the relative probability of bycatch was larger (or smaller) than

Table 2
Candidate models. All models included 'vessel' as a random intercept. Final model (M6) highlighted in bold. ΔAIC: difference in Akaike information criterion relative to lowest model AIC value; df: degrees of freedom. See Table 3 for description of explanatory variables.

Model	Formula	ΔAIC	df
M1	~ offset(log(n_hooks))	190.1	2
M2	~ offset(log(n_hooks)) + year + month	114.0	22
M3	~ offset(log(n_hooks)) + year + month + dist_coast	113.3	23
M4	~ offset(log(n_hooks)) + year + month + dist_colony	114.3	23
M5	~ offset(log(n_hooks)) + year + month + dist_colony * colony_npairs	100.5	25
M6	**~ offset(log(n_hooks)) + year + month + s(long, lat)**	**0.0**	**47.5**

Table 3
Explanatory variables used in models listed in Table 2. * Variable standardised by dividing by the standard deviation.

Variable	Type	Description
fulmar	Integer	Number of fulmar individuals bycaught
n_hooks	Integer	Total number of hooks
year	Factor	Calendar year
month	Factor	Calendar month
distcoast	Continuous	Distance from coast, km
dist_colony*	Continuous	Distance from the nearest breeding colony, km
colony_npairs*	Integer	Number of breeding pairs in nearest colony
long	Continuous	Longitude, decimal degrees
lat	Continuous	Latitude, decimal degrees
vessel	Factor	Unique vessel identifier (call signal)

the respective overlap index.

3. Results

3.1. Estimated total fulmar bycatch

In the period 2012–2021, three offshore longline vessels in the Norwegian Reference Fleet reported 1304 fulmars taken as bycatch during 7 % of fishing days. Of these bycatch events, 42 % involved one individual, and 93 % involved ten or fewer individuals. Applying the best model (M6; Table 2) to all fishing activity in the Norwegian offshore longline fishery (using the ERS dataset), we estimated an average of 0.011 (0.0076–0.0250) fulmars were bycaught per 1000 hooks, or 0.43 (0.31–1.00) bycaught fulmar per fishing day.

Total bycatch of fulmar was highly variable in both space and time. Although total bycatch varied between 128 and 6237 (95 % CI: 51–16242) individuals per year, there were no distinct trends and the uncertainty was variable (Fig. 3A). Annually, the bycatch rate peaked in the summer months of June-August (Fig. 3B), when rates were over double that of winter months (October-March). A strong spatial hotspot in bycatch rates were identified in the Norwegian Sea (Fig. 3C). Spatial variation in bycatch rates in the Barents Sea were not as strong, but nevertheless were detectable, with a hotspot on the south Svalbard coast and the northernmost coastline of mainland Norway. The model estimated a small dispersion parameter (theta = 0.054), indicating that bycatch events are highly clustered. This explains why months and years with relatively high bycatch rates have exponentially larger uncertainty than years with smaller estimated bycatch. Variations between vessels were not significant, but we note the increased risk of imprecise variance estimation with fewer than five random levels (Harrison et al., 2018).

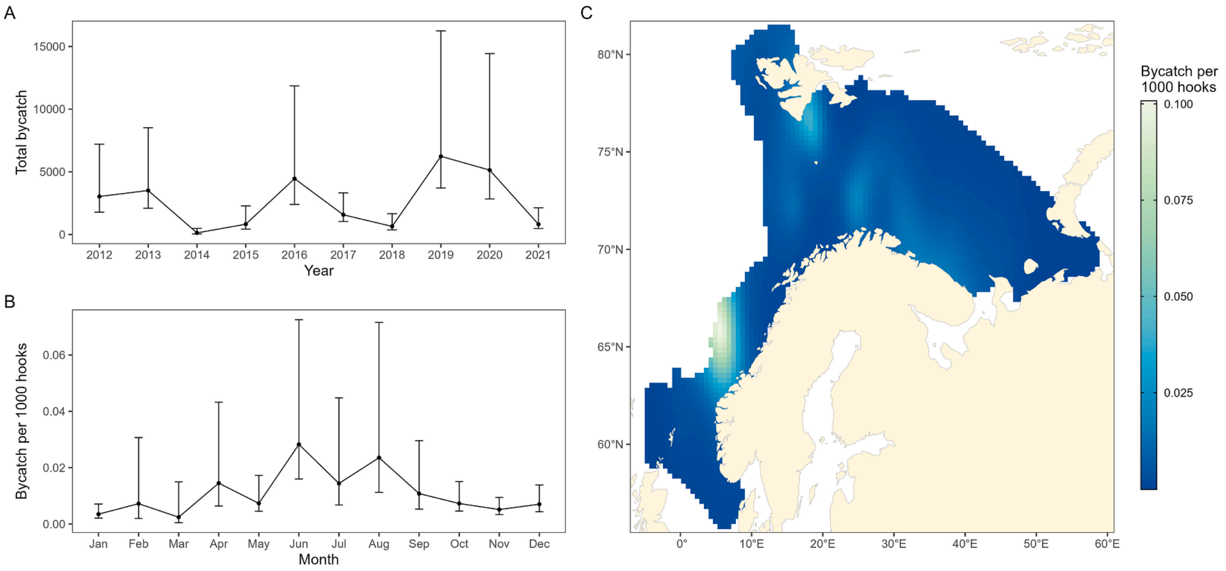

Fig. 3. Estimates (mean ± 95 % confidence interval) of fulmar bycatch rate in the Norwegian offshore longline fishery. (A) Total bycatch per year; (B) bycatch rate per month, (C) spatial variation in bycatch rate (average across all years and months).

3.2. Overlap between fulmar distribution and fishing activity

To aid interpretation, we present the overlap analysis aggregated based on the fulmar breeding season (April-August), and include comprehensive results per month in the Supplementary materials (Figure A2-Figure A4).

Most of the fishing activity occurred in the Barents Sea (including the Russian zone) and Norwegian Sea, but there was also fishing activity in foreign waters, specifically the United Kingdom, Iceland, and Greenland (Fig. 4A). In addition, there is highly localised fishing activity in the exploratory fishery around Jan Mayen (Fig. 1; Bogstad, 2023) and on the Flemish Cap (Fig. 1) where Norwegian vessels have limited access to the cod fishery based on historical rights.

Fulmar distribution was highly clustered around the Faroe Islands and northern United Kingdom (Fig. 4B), which is explained by the large number of colonies in those countries representing 52 % of all observed breeding pairs in the Northeast Atlantic study area

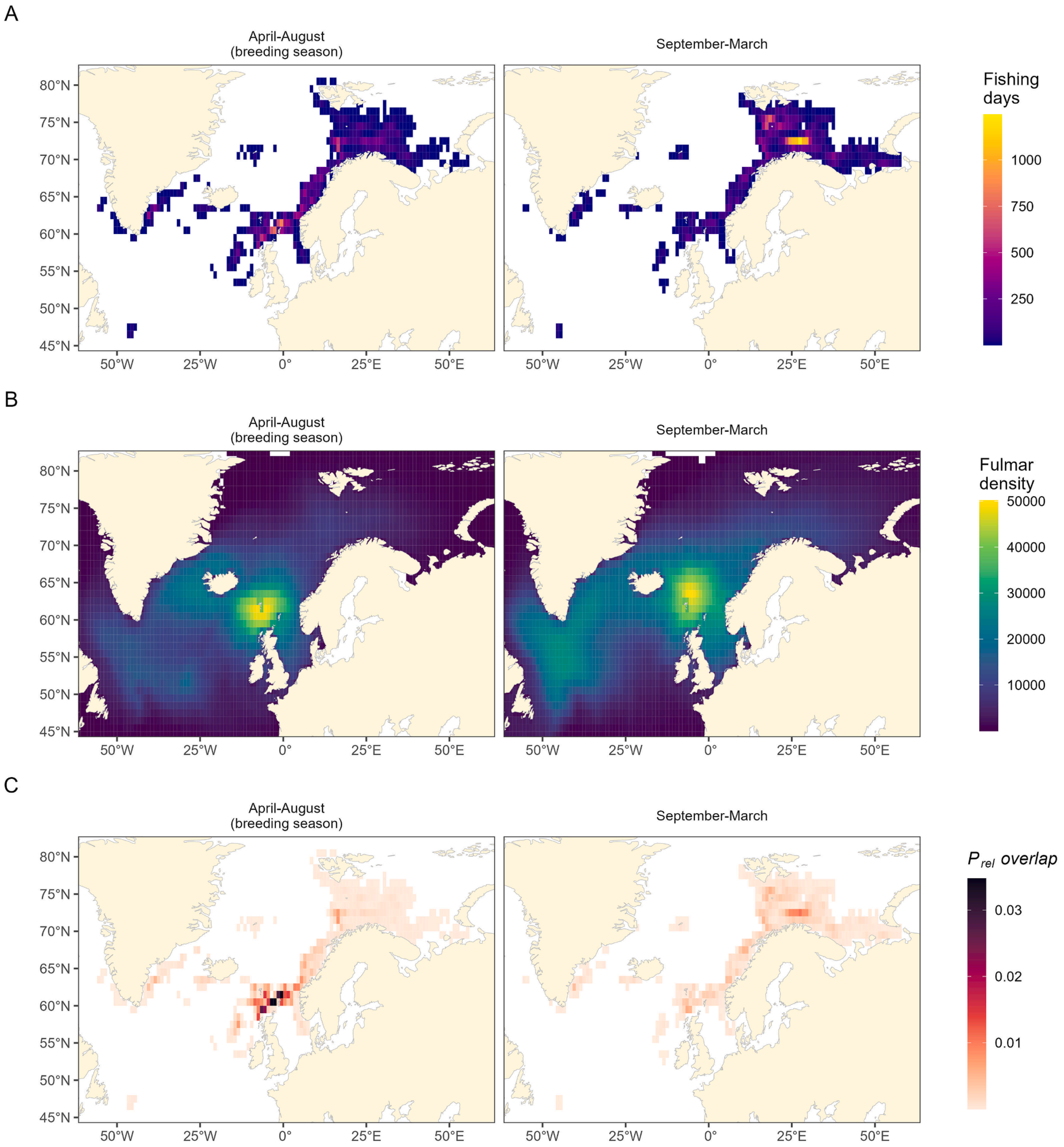

Fig. 4. Spatial distribution of (A) fishing activity, (B) fulmar density, and (C) overlap index (Eq. 1.3) in the Northeast Atlantic.

(Fig. 2; Fauchald et al., 2021). Outside of breeding season, fulmar densities increased in western areas of the Northeast Atlantic, the North Sea, and north into the Barents Sea.

The Norwegian offshore longline fishery overlapped with 14 % of the total fulmar distribution in the Northeast Atlantic (Fig. 4C). The relative probability of overlap was highest in the breeding season (62 % of total relative overlap) and confined almost exclusively north of the United Kingdom. Outside of the breeding season, the largest overlap was in the Barents Sea where most fishing activity occurred, despite the relatively low density of fulmars in that region. There was a notable overlap in easterly areas of the Barents Sea, which peaked in September-December when the Barents Sea fishery expanded out from the core fishery in the central Barents Sea.

3.3. *Comparing estimated bycatch and overlap index*

Overall, estimated total bycatch in the Norwegian offshore longline fishery was highly correlated with overlap indexes across months (Table 4), with June being the only deviation with a lower but nevertheless significant correlation coefficient of 0.53. Exploring spatial variations in differences between estimated bycatch and overlap indexes revealed deviations from this agreement between the two measures in the breeding season (Fig. 5). Estimated bycatch was higher than expected in the Norwegian Sea based on the overlap index, whilst the opposite was observed north of the United Kingdom where overlap index over-predicted the risk of bycatch. Outside of the breeding season, the measures were almost in perfect agreement, with only a weak tendency for the relative probability of bycatch to be larger than the overlap index.

4. Discussion

For over two decades, estimates of seabird bycatch in Norwegian offshore longline fisheries have been based on two studies deemed of ‘poor’ reliability (Anderson et al., 2011; Ramírez et al., 2024), estimating an upper range of 101 380 seabirds bycaught annually. Our results, based on direct observations, reduces this upper range by an order of magnitude. Whilst this comparison does not factor in a 20-year time difference between studies nor the large uncertainties, it importantly brings knowledge up to date and dramatically improves reliability by improving data collection and estimation methods. The average seabird bycatch rate in the Norwegian offshore longline fisheries (0.011 seabirds per 1000 hooks) is lower than respective estimates in the coastal longline fisheries for cod and haddock (0.064 seabirds per 1000 hooks; Fangel et al., 2015) and Greenland halibut (0.031–0.294 seabirds per 1000 hooks; Fangel et al., 2015, 2017). Even when accounting for the larger scale of the offshore fishery, total bycatches are still lower in comparison, given the additional bycatches of other species in coastal fisheries (Fangel et al., 2015).

Overall, there was a high correspondence between estimated total fulmar bycatch and the overlap between the distributions of fulmar density and fishing activity in the Norwegian offshore longline fishery. This agrees with previous overlap studies that have used light at night and foraging activity as indicators of “true” interactions (Dupuis et al., 2021, Darby et al., 2023), and also supports the understanding that increased time spent near vessels will result in increased bycatch risk (Dias et al., 2019; Anderson et al., 2022). Our findings showed that both the largest overlap between the Norwegian longline fishery and fulmars, and the highest risk of bycatch occurred during the breeding season. This is the first published evidence of such trends but is also supported by preliminary results from a pilot study in Faroese fisheries (Danielsen et al. in prep). Dupuis et al. (2021) and Darby et al. (2023) quantified nocturnal interactions with fishing vessels, using light sensors to detect the presence of a fishing vessel around tracked seabirds. However, due to methodological limitations related to the use of geolocators under different light conditions within diel and seasonal cycles, these studies were not able to quantify interactions during daytime or directly compare the number of interactions in summer versus winter months.

There were some interesting discrepancies between estimated bycatch and overlap index in both directions (Fig. 5), which were strongest in the breeding season. Regional differences in fishing practices could influence the likelihood of fulmars attending vessels. For instance, the longstanding discard ban in Norway (Gullestad et al., 2015) has led to increased utilisation of catches and retention of processing waste, which may reduce the availability of discards for scavenging birds. This could explain why fulmars were more likely

Table 4
Spearman’s rank correlation coefficient between overlap index and relative probability of bycatch. All correlation coefficients were highly significant ($p < 0.001$).

Month	Spearman’s rank correlation coefficient
January	0.75
February	0.68
March	0.65
April	0.71
May	0.64
June	0.53
July	0.74
August	0.79
September	0.72
October	0.74
November	0.81
December	0.76

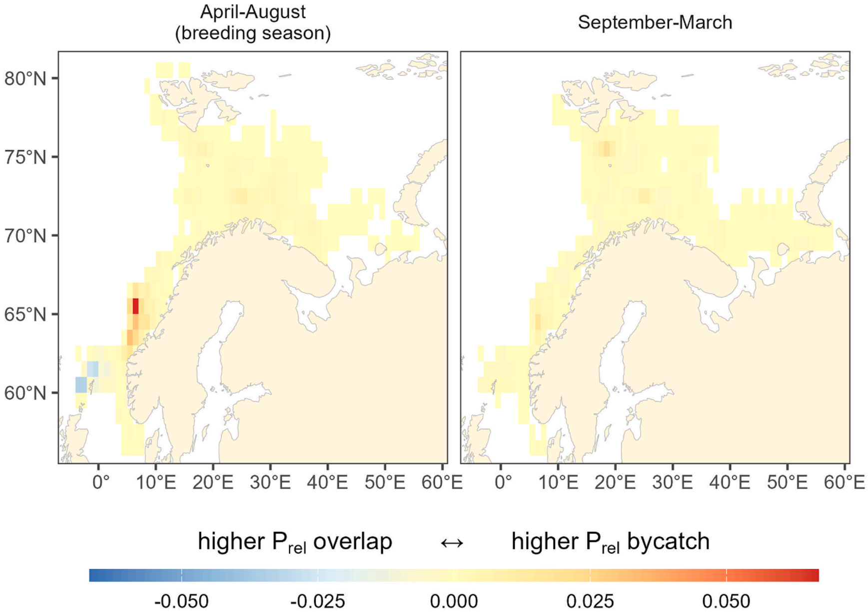

Fig. 5. Difference in scaled estimates of relative bycatch probability (Eq. 1.4) and overlap index (Eq. 1.3) in the Norwegian offshore longline fishery.

to interact with fishing vessels in UK waters (Darby et al., 2023). However, our results cannot address the drivers that could explain why the increased interaction was not reflected in an equally higher bycatch rate, given that a large overlap index could be a result of either high levels of fishing activity or fulmar distribution. Nevertheless, identifying strong spatial and temporal variations in bycatch rates can be useful for management advice such as the increased risk of bycatch during breeding season when fulmars remain closer to breeding colonies.

Fulmars are in decline throughout much of their Northeast Atlantic range. Recent counts show declines in the breeding populations in Ireland and UK (Burnell et al., 2023) and Iceland (Kolbeinsson et al., 2023), whilst breeding fulmars have almost disappeared from mainland Norway (Shimmings and Øien, 2015). However, the populations on Spitsbergen (Descamps & Strøm 2021) and Jan Mayen (Fayet et al., 2023) have remained stable. Fulmars are highly mobile, covering vast areas throughout their annual cycle (Edwards et al., 2013; Quinn et al., 2016; Thaxter et al., 2012). It is therefore important to consider all the potential impacted populations on a wide geographic range when assessing the effects of bycatch. In this study we benefited from the extensive dataset on fulmar distribution collected through SEATRACK, enabling us to quantify the overlap between fulmars breeding in the Northeast Atlantic and fishing activity on a large geographic scale. However, our study only includes information from the Norwegian offshore longline fishery. To fully understand the scale of fulmar bycatch throughout the Northeast Atlantic, there is therefore continued demand to quantify bycatch from other fisheries in the Northeast Atlantic that also impact fulmars (e.g., Anderson et al., 2011; Bærum et al., 2019; Kingston et al., 2023). To understand the cumulative bycatch pressures acting on fulmars in the North-Atlantic, it is consequently important to consider information across national jurisdictions and the whole annual flyway used by fulmars. The drivers of population declines are likely multifaceted and vary across time and space, with bycatch being only part of the explanation. However, simulations of bycatch in specific areas along the Canadian coast have also shown that bycatch in even a single fishery might be enough to cause population declines (Anderson et al., 2018). However, considerable uncertainty remains in bycatch estimates themselves, highlighting the need for more precise data and a better understanding of the drivers behind bycatch. This is essential before we can reliably assess population impacts of bycatch in the Northeast Atlantic.

The availability of daily logbook (ERS) data in offshore fisheries allowed for more complex models in this study but there are still many sources of variation in bycatch that are rarely accounted for. Ecological factors are useful for understanding the impact of bycatch on populations (Zhou et al., 2019), but are restricted by the lack of available knowledge at relevant scales as input for bycatch models. For example, averaged seabird densities might have insufficient spatio-temporal resolution to be meaningful for explaining bycatch events recorded at finer scales. Environmental factors such as weather (Gilman et al., 2005) may improve prediction accuracy but are less useful for management purposes and may increase uncertainty in models. Finally, human factors contain a wide range of fishing practices (e.g., offal discharge, deck lighting, bycatch reduction devices; Gilman et al., 2005) that influence the risk of bycatch. For the various factors listed, a compromise can be made by capturing variations in a limited yet well-described set of explanatory variables, such as the use of 2D smoothers to capture all spatial variation.

The reliability of self-reported data is often questioned (Steins et al., 2022), particularly for threatened, endangered, and protected species such as seabirds (Gray and Kennelly, 2018). Self-reported data are typically validated by comparing them with another trusted data source. Such a comparative data source is not currently available in Norwegian fisheries, but electronic monitoring has been suggested as a potential source (Moan et al., 2020). Conversations with fishers and IMR technicians suggest that seabird bycatch is under-reported (see also Christensen-Dalsgaard et al., 2019), but an open dialogue facilitates a better understanding of which species are vulnerable to misreporting and the motivations behind it. Industry-science collaborations benefit from trustful relationships (Kraan et al., 2013; Steins et al., 2022) which take time to build, particularly for the contentious topic of fisheries science and advice (Cvitanovic et al., 2021).

Concerns surrounding self-reported bycatches are especially relevant for extreme events (Christensen-Dalsgaard et al., 2019), which although rare can still contribute substantially to total bycatches (Glemarec et al., 2020). Electronic monitoring out-performs self-sampling in this regard as extreme events are observed at the same quality as typical bycatch events (Glemarec et al., 2020). Nevertheless, self-sampling creates ownership of scientific process that should not be overlooked. The Norwegian Reference Fleet has provided data for bycatch estimates of seabirds (Bærum et al., 2019; Fangel et al., 2015) and sea mammals (Moan et al., 2020) in coastal fisheries, allowing participating fishers to understand the entire scientific process. With this study being the first in the offshore longline fisheries, the process is still in early stages. This is reflected by the low number of vessels providing data for this study, which will impact precision (Pennington and Helle, 2011). As trust is built and more vessels contribute data, then spatial interpolation will be reduced and vessel-based effects will be better described, improving overall bias and precision. Nevertheless, the fact that seabird bycatches are entirely absent from mandatory reporting channels in Norway demonstrates the value of collaboration for scientific purposes. Whilst the current results should be recognised as underestimates because of the likely misreporting of some bycatch events, they should also be seen as a vital step towards improvement in quality of bycatch estimates and stakeholder inclusion.

5. Conclusion

Our study contributes to the large yet long-standing knowledge gap of seabird bycatch in longline fisheries in the Northeast Atlantic (Anderson et al., 2011; Ramírez et al., 2024), demonstrating the importance of more comprehensive and precise estimates. Furthermore, we have addressed concerns and the progressive development of self-sampling methods for reporting of threatened species (Gray and Kennelly, 2018). By linking direct estimates of bycatch with a theoretical measure of bycatch risk (overlap between fulmar density and fishing activity), we have validated potentially high-risk areas within a framework which is directly useful for management advice.

We identified that fulmars are at most risk of bycatch during the breeding season (April-August), with hotspots around the large collection of breeding colonies in the Faroe Islands and northern UK coast. We also found strong correlations between estimated bycatch and overlap indexes which provide validation for both methods, but particularly for overlap indexes which are wider ranging and easier to implement than direct observations. Nevertheless, without reliable bycatch rate estimates, population impacts of seabird bycatch (especially for far-ranging seabirds such as fulmars), or seabird population life history data, moving from bycatch risk to mitigation efforts might both be controversial and ineffective in specific cases. There are however some cases that warrant the fast track between risk analyses and mitigation, and this is particularly so for protected, endangered, and threatened species that are rare and would thus likely never obtain reliable bycatch estimates even with a high monitoring effort.

Ethics statement

Not applicable: This manuscript does not include human or animal research.

Declaration of Competing Interest

The authors declare no competing interests

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.gecco.2024.e03350.

Data Availability

Multiple data sources used with different permissions/access. We describe all availabilities in the acknowledgements. Norwegian Reference Fleet data cannot be shared publicly due to the sensitivity of the contents and the privacy of fishers involved in data collection. The data will be shared for research purposes on an individual basis by contacting the corresponding author. Data from the Electronic Reporting System is openly accessible through the Directorate of Fisheries (https://www.fiskeridir.no/Tall-og-analyse/AApne-data). For more information on SEATRACK data, please visit https://seatrack.net

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