ICES Journal of
Marine Science

International Council for the Exploration of the Sea
Conseil International pour l'Exploration de la Mer

ICES Journal of Marine Science (2017), 74(1), 332–342. doi:10.1093/icesjms/fsw149

Original Article

Incidental bycatch of northern fulmars in the small-vessel demersal longline fishery for Greenland halibut in coastal Norway 2012–2014

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Fangel, K., Bærum, K. M., Christensen-Dalsgaard, S., Aas, Ø., Anker-Nilssen, T. Incidental bycatch of northern fulmars in the small-vessel demersal longline fishery for Greenland halibut in coastal Norway 2012–2014. – ICES Journal of Marine Science, 74: 332–342.

Received 22 December 2015; revised 17 June 2016; accepted 27 July 2016; advance access publication 31 August 2016.

With seabird populations in rapid decline, understanding and reducing anthropogenic mortality factors is essential. One such factor is incidental bycatch in fisheries. Here we analyze bycatch in the small-vessel demersal longline fishery for Greenland halibut outside the coast of Northern Norway in 2012–2014, by means of self-reporting from fishers and independent observers. A sample of killed birds were analysed for sex, age, reproductive status and condition. Nearly all were northern fulmars. Estimated total bycatch for this fishery for the 3-year period was about 312 birds (SE $\approx$ 133) using a stratified estimator. Bycatch rate per 1000 hooks was estimated to approximately 0.031 (SE $\approx$ 0.012). Exploring per trip bycatch rates utilizing generalized linear mixed models, we found no convincing trends of environmental, spatial and temporal variables in explaining bycatch. However, trips using longlines with non-swivel hooks had a more than 100-fold larger bycatch rate (mean $\approx$ 0.760, SE $\approx$ 0.160) than those using swivel hooks (mean $\approx$ 0.008, SE $\approx$ 0.002). Further, trips with external observers had higher bycatch estimates (mean $\approx$ 0.75, *SE* $\approx$ 0.16) compared with trips where bycatch was registered by the fishers (mean $\approx$ 0.02, *SE* $<$ 0.01). Of the analysed birds, about two-thirds were adult birds and males dominated (71.1%). A majority were in good or moderate condition. The findings suggest that the incidental bycatch in the Greenland halibut fishery along the Norwegian coast is more limited than previous studies indicated, and that the use of swivel hooks can significantly reduce such bycatch. However, the impacts on the red-listed, diminishing population of fulmars breeding in mainland Norway should be assessed further and requires a method to assign killed birds to regions/colonies. Also, gaining a better understanding of what triggers events with extreme bycatch numbers is important to reduce the problem further and to improve bycatch modelling.

Keywords: coastal fisheries, mitigation, Norway, sampling method, swivel hooks.

Introduction

Incidental bycatch in commercial fisheries has received growing attention over the last couple of decades, and a large proportion of the seabird species that may be affected are in decline (ICES, 2013). Most of the focus has been drawn to the large numbers of albatrosses and petrels caught in demersal and pelagic longline fisheries in the southern hemisphere (e.g. Brothers *et al.*, 1999a; Cooper *et al.*, 2001; Anderson *et al.* 2011; Yeh *et al.*, 2013). However, other species and fisheries have received attention (e.g. Tasker *et al.*, 2000; Croxall, 2008; Žydelis *et al.*, 2013; Oliveira *et al.*, 2015; Fangel *et al.*, 2015). Incidental bycatch of seabirds has been shown to have potentially severe consequences for some species (see e.g. Tuck *et al.*, 2001; Lewison and Crowder, 2003). Globally, an estimated range of 160 000–320 000 seabirds are killed annually in longline fisheries (Anderson *et al.*, 2011), an estimate that portrays a large degree of uncertainty of bycatch numbers in many fisheries. This is especially true for the smaller longline vessels in the Northeast Atlantic Ocean where little or no data on seabird bycatch have been available for estimation of bycatch rates. Given the severity of many ongoing seabird declines in this region (e.g. Croxall *et al.*, 2012; Fauchald *et al.*, 2015), it is crucial to document seabird bycatch in these waters in more detail and gain a better understanding of the causes of the variability in bycatch numbers. Besides reducing estimate uncertainty, this is key information for identifying functional and effective mitigation measures that can be applied in the fisheries of most concern.

Mitigation measures can have significant effects in reducing incidental seabird bycatch in longline fisheries (e.g. Løkkeborg, 2003; Dietrich *et al.*, 2008). Possible mitigation measures include the use of bird-scaring lines (Løkkeborg, 2003), dyeing of bait (Cocking *et al.*, 2008), increased weighting for fast sinking of longlines (Robertson *et al.*, 2006; Dietrich *et al.*, 2008) and hook design (Li *et al.*, 2012). Other factors that may help explain bycatch rates are spatio-temporal circumstances such as time of year, distance from seabird colonies, light and weather conditions (including wind speed) and vessel type and activity (Brothers *et al.*, 1999b; Dietrich *et al.*, 2009; Jimenez *et al.*, 2014). The individual importance of the abovementioned variables may, however, depend on the specific fisheries, seabird species, habitats and locations in question, making generalizations complicated and not always relevant. Furthermore, sampling method may influence the results of bycatch studies (NMFS 2004), applicable for interview methods (Lien *et al.*, 1994) and data sampled by logbooks/catch reports or independent observer (Walsh *et al.*, 2002; Warner, 2004: Van Atten, 2007; Oliveira *et al.*, 2015).

In a study of three small-vessel fisheries outside Northern Norway in 2009, the longline fishery for Greenland halibut (*Reinhardtius hippoglossoides*) stood out as having a surprisingly high bycatch of seabirds, especially of northern fulmars (*Fulmarus glacialis*) (Fangel *et al.*, 2015). Overrepresentation of this species in seabird bycatch has also been found in other studies of longline fisheries in the Northeast Atlantic (Dunn and Steel, 2001; Løkkeborg and Robertson, 2002). The northern fulmar (hereafter fulmar) is a circumpolar boreo-arctic seabird. With between 2.4 and 4.4 million pairs coarsely estimated to breed in the Northeast Atlantic (BirdLife International, 2004; Mitchell *et al.*, 2004), the fulmar is one of the most widespread and abundant seabird species in the region. In terms of breeding numbers, this is, however, not the case in mainland Norway, where only an estimated 9.000 pairs bred in the early 2000s (Barrett *et al.*, 2006a). Due to a significant decrease over the next decade (e.g. Anker-Nilssen *et al.*, 2016), the now small mainland population (500–1000 pairs; Shimmings and Øien, 2015) is listed as endangered on the national red list (Kålås *et al.*, 2015). The fulmar is a surface-feeding seabird with an extensive offshore foraging range during its entire life cycle. Fulmars do not start to breed until 9 years old on an average (Ollason and Dunnet, 1978), so that the population includes large numbers of immature birds. In the Northeast Atlantic, there is a big floating population of fulmars (e.g. Barrett *et al.*, 2002, 2006b) originating from most breeding areas in the region.

Being generalist predators and scavengers feeding on the sea surface, fulmars are known to follow ships and often assemble in large numbers around fishing vessels (e.g. Camphuysen *et al.*, 1995). This interaction with fishing operations makes them vulnerable to being killed as bycatch. As with many other seabird species, fulmars are long-lived and have a low reproductive rate, making them especially sensitive to increases in adult mortality (e.g. Croxall, 1987; Tasker *et al.*, 2000). In addition to the actual bycatch rate, the population level effect of seabird bycatch will, therefore, also depend on the sex and age structure of the birds killed, which source populations they belonged to, and the status of those populations (Tasker *et al.*, 2000; Phillips *et al.*, 2010; Lewison *et al.*, 2012). To be able to assess these parameters, seabirds taken as bycatch must be collected and examined. To date there is no method to determine with reasonable accuracy the population origin for fulmars (but see van Franeker and Wattel, 1982; Burg *et al.*, 2003). Knowledge about bycatch rates, and sex and age distributions of fulmars killed, can, therefore, only point to the most likely order of size for possible impacts on any source population. As several of the fulmar populations in the Northeast Atlantic are now in decline (e.g. JNCC, 2015; Anker-Nilssen *et al.*, 2016), better knowledge about bycatch rates of fulmars in offshore fisheries, and the effects of possible measures to prevent it, may prove crucial. Given also the endangered status of the small remnant population of fulmars breeding on the Norwegian mainland coast (Kålås *et al.*, 2015), the small-vessel longline fishery for Greenland halibut off North Norway is of special interest in this context, since it has been documented to kill significant numbers of fulmars in some years (Fangel *et al.*, 2015).

With this background, the goal of this case study is to estimate incidental bycatch of fulmars in the Norwegian small-vessel longline fishery for Greenland halibut. The analyses are based on data from self-administered recordings made by the fishers and observations from independent on-board observers, and aim to (1) provide better estimates of fulmar bycatch in this fishery, (2) explore variations in this bycatch in a spatio-temporal landscape as a function of different mitigation efforts and (3) assess demographic traits of the fulmars killed. The findings are discussed in a management perspective.

Material and methods

The Greenland halibut fishery 2012–2014

Greenland halibut (hereafter halibut) is a circumpolar deep-sea flatfish that can reach 120 cm in length and a body mass of 20 kg. The north-eastern part of the population inhabits a wide geographical range along the deep continental slopes from the eastern coast of Canada to north of Spitsbergen. The halibut fishery in the Norwegian Economic Zone is strictly regulated. It includes an overall total quota, vessel-quotas for the targeted fishing, minimum length in catches, and restrictions to bycatch of other fish species. Targeted fishing for the halibut is legal in two fishing periods for vessels $<$ 28 m length over all (LOA), which mainly conduct their fishing as a coastal fishery along the continental slope from Vesterålen to Tromsøflaket and outside the county of Finnmark, in the Norwegian Sea and south-western Barents Sea (Figure 1).

In the years of this study, 2012–2014, the first period for targeted fishing started 26–28 (The two periods for targeted fishery opens at midnight the last Sunday in May and July.) May and lasted 2–3 weeks (holding 60–70% of the fishing quota for

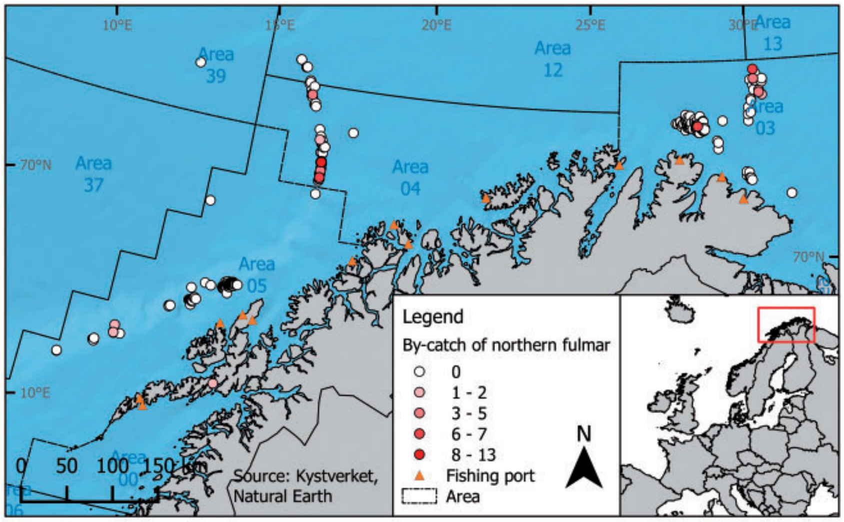

Figure 1. Positions of Greenland halibut fishing lines set in the years 2012–2014. White dots indicate trips with no registered bycatch of northern fulmars. Coloured dots indicate fishing trips with at least one fulmar registered as bycatch. Triangles indicate fishing harbours where the catches were delivered and registered, also encompassing harbours where the fishing vessels used in the study were recruited. Areas refer to specific Fishing areas as defined by Norwegian fishing authorities. Shading in the sea-area of the map represents rough depth estimates, where lighter tones indicates shallower water.

targeted fishing), the second period started 28–30 July and lasted 1–4 weeks until the quotas were reached. For the vessel-group up to 27.99 m LOA, the annual vessel quotas ranged between 12.5 and 17.5 tonnes depending on LOA. The annual total catch in the targeted halibut fishing in these years ranged between 4.5 and 5.8 thousand tonnes of halibut annually, within a total Norwegian annual quota of 8.8–10.0 thousand tonnes. Each year, between 186 and 220 vessels (mean length = 12.2 m LOA, SD = 2.4 m) participated in this coastal fishery.

The targeted fishery is normally practiced as 2–4 day outings (hereafter trips) from the vessel's home harbour. Because of the small size of the vessels, these fishers need to return to their harbour to deliver catch, rest and get supplies after only a few days. Usually, they fish most actively during the first 1–2 weeks of the first fishing period, within which many are able to reach their vessel quota. Much fewer vessels are thus active in the second fishing period. On an average, each vessel completed on median four trips annually (25–75% quartiles: 2–9) in the longline fishery for halibut in 2012–2014.

Study area

We aimed to cover the fishing in the major areas for the Norwegian halibut fishery as described above, namely that conducted along the continental shelf edge in the Norwegian Sea from Vesterålen up to the western part of Finnmark county, as well as that taking place in the shallower Barents Sea outside eastern Finnmark. Essentially, we aimed to investigate fishing in Fishing areas 05 (outside Vesterålen up to Tromsø), 04 (Tromsø—Nordkapp) and 03 (East of Nordkapp), as defined by Norwegian fishing authorities. The distance from the harbours to the fishing areas depended mainly on the distance to the nearest seafloor with suitable condition (depth) for halibut fishing, as considered by the fishers.

Data collection

Sampling design

We employed a multi-stage procedure to develop our sampling procedure. First, based on official landing statistics we identified a number of harbours important for the halibut fishery. We then selected seven harbours spread along the entire coastline of the study area, where a suitable local organizer could be hired to assist our sampling. These harbours were Båtsfjord and Berlevåg (area 03), Honningsvåg (areas 03 and 04), Torsvåg (area 04), Stø/Myre (area 05) and Ballstad (primarily fishing in area 04). In each harbour, vessels registered with a halibut quota were approached randomly and their key crew invited to participate in the study. Participation in the study was voluntary. Fishers from a total of 55 vessels (mean length 12.7 m LOA, SD = 2.8 m) were recruited and provided data spread across areas, years and fishing periods as shown in Table 1. Harbours where the catch were landed are shown in Figure 1. Overall, the study gathered data from 426 trips, which represent a large share (approx. one-fourth) of the overall fishing effort in this fishery.

We registered data on seabird bycatch and possible covariates by means of two approaches: (1) by self-administered registration conducted by the skipper/headman ($n = 389$ trips) and (2) by independent on-board observers ($n = 37$ trips). Both procedures used a detailed logbook for registering a range of parameters detailing the fishing activity and results, including location/position, fishing depth, weather, fish catch, catch of dead seabirds (i.e. bycatch), tackle/bait, hook type and number of hooks used. A sample of seabirds taken as unintentional bycatch (82% of all birds reported) were collected and brought to the harbour.

Examination of bycatch

Dead birds unintentionally taken as bycatch were collected and labelled with a tag identifying vessel and date of capture. The birds were stored frozen until they were packed and sent frozen to Norwegian Institute for Nature Research (NINA) in

Table 1. Description of variables used in the analysis of factors potentially effecting bycatch of northern fulmars in the Norwegian longline fishery for Greenland halibut.

Variable name	Description	Statistics (no. of vessels)
Fulmar bycatch	Count of fulmars caught as incidental bycatch on the specific trip. This only includes dead birds attached to the longline after hauling	Min = 0, mean = 0.24, max = 33
Hook type	Binomial indicator representing either a swivel hook or a hook without a swivel	*n*, swivel hook = 155 (22) *N*, hook without swivel = 271 (37)
Season	Binomial indicator representing the first (until and including June) or second (July and later) fishing period	*n*, spring = 309 (42) *n*, summer = 117 (19)
Wind	Continuous, indicating average wind speed in m s^{-1}	Min = 0, mean = 5.2, max = 12
Mitigation	Binomial indicator representing whether any mitigation measures to avoid incidental bycatch of birds was applied	*n*, yes = 368 (45) *n*, no = 53 (12)
Bait	Binomial indicator representing whether the bait was frozen or thawed	*n*, frozen = 100 (15) *n*, thawed = 279 (36)
Area	ID for the specific fishing area	*n*, area 3 = 154 (18) *n*, area 4 = 44 (13) *n*, area 5 = 228 (25)
Number of hooks	Continuous, total number of hooks set on the specific fishing trip	Min = 900, mean = 5259, max = 36000
Dataset	Binomial indicator representing whether the data was obtained from self-recordings made by the fishermen or from an independent on-board observer attending the specific trip	*n*, self registration = 389 (42) *n*, observations = 37 (14)
Year of capture	Year of each sample (i.e. fishing trip)	*n*, 2012 = 19 (7) *n*, 2013 = 201 (29) *n*, 2014 = 206 (28)
Vessel ID	Individual identification code assigned to each vessel. Used as a random effect in the model approach	Number of vessels = 55

The statistics are based on number of trips registered (in total 426 trips) for the respective variable-level, with the respective numbers of vessels represented within each variable level indicated in parentheses. Variable-specific statistics might not always sum up to total number of trips or vessels, as some trips might miss information on the respective variable and vessels might be represented in multiple levels of a variable (e.g. one vessel might have been fishing both with swivel hooks and non-swivel hooks).

Trondheim for further analysis. At the NINA laboratory, the birds were thawed and their species determined. All birds were then measured, dissected and sexed following the internationally standardized methods described by van Franeker (2004) and Camphuysen (2007). Based on the developmental stage of sexual organs and presence/size of bursa fabricius, each fulmar was aged as either juvenile, immature, subadult or adult (van Franeker and Meijboom, 2002). For analytical purposes, all juveniles, immatures and subadults were subsequently categorized as immatures for comparisons with adult birds. Body condition of fulmars was assessed using a subjective scoring system evaluating subcutaneous fat, internal fat and size of the bird's left breast muscle on a 0–3 scale as described by van Franeker (2004). The three values were then summed up to produce an overall index of condition ranging from 0 to 9, where 0 to 3 was categorized as 'poor', 4 to 6 as 'moderate' and 7 to 9 as 'good'. Where subcutaneous fat was present, the thickness of subcutaneous fat deposits was measured over the lower end of the breastbone to the nearest 0.1 mm. Each specimen was also assigned to one of four fulmar colour morphs: double light (LL), light (L), dark (D) or double dark (DD), colouration attributes that can point to latitude affiliation (van Franeker and Wattel, 1982; van Franeker and Luttik, 2008).

Analyses

Estimation of bycatch rates

Bycatch rate was estimated as a stratified mean bycatch per trip extrapolated to the total number of trips by all vessels. For trip k by vessel i in stratum h, let $X_{i,k,h}$ be the bycatch of seabirds in numbers. An estimator for mean bycatch per trip across all vessels in area h is then

$$\bar{x}_h = \frac{\sum_i \sum_k \hat{x}_{i,k,h}}{n_h} \quad (1.1)$$

where n_h is the total number of trips in area h. An estimator for the total bycatch in h is

$$\hat{X}_h = N_h \bar{x}_h \quad (1.2)$$

with variance

$$\mathrm{var}(\hat{X}_h) = N_h^2 \mathrm{var}(\bar{x}_h) \quad (1.3)$$

where N_h is the total number of trips in h by all vessels.

Seabird bycatch per unit effort (BPUE) was calculated as a stratified ratio-estimator, specifically as the mean ratio of bycatch in number of birds per 1000 hooks. For fishing operation j by vessel i in stratum h let $y_{i,j,h}$ be the hooks set and let $x_{i,j,h}$ be the bycatch of seabirds in numbers. An estimator for the mean bycatch ratio for vessel i in stratum h is then

$$\hat{R}_{i,h} = \frac{\sum_j x_{i,j,h}}{\sum_{j,h} y_{i,j,h}} \quad (1.4)$$

and an estimator for the mean bycatch ratio for all vessels in stratum h is

$$\hat{R}_h = \frac{\sum_i Y_{i,h} \times \hat{R}_{i,h}}{\sum_i Y_{i,h}}. \quad (1.5)$$

where $Y_{i,h}$ is the total catch of the target species for all fishing operations by vessel i in stratum h. The variance of (1.5) is estimated by bootstrapping. Since the sampling fraction of PSUs generally is negligible, we assumed sampling with replacement in the first stage (see e.g. Williams, 2000). Each bootstrap replicate was generated by first sampling the PSUs at random with replacement. The bycatch and catch samples within each PSU were then selected by simple random sampling.

The total bycatch of seabirds in stratum h and its variance is then estimated by

$$\hat{X}_h = \hat{R}_h Y_h \tag{1.6}$$

and

$$\mathrm{var}(\hat{X}_h) = Y_h^2 \mathrm{var}(\hat{R}_h), \tag{1.7}$$

respectively, where Y_h is the total catch taken by all vessels in stratum h obtained from the official landings statistics. An estimator of the total bycatch across strata is then

$$\hat{X} = \sum_h \hat{X}_h \tag{1.8}$$

with variance

$$\mathrm{var}(\hat{X}) = \sum_h \mathrm{var}(\hat{X}_h) \tag{1.9}$$

Upscaling

To estimate the total numbers of fulmars taken as bycatch in the halibut longline fishery in Norway for the study period, we extracted from the Norwegian fishery statistics the total number of trips where longline was used and halibut was registered in the catch. Further, we selected only catches registered within the legal time span for the halibut fishery, and where halibut (in kg) constituted $> 50\%$ of the total catch. Following this selection, most fishing trips selectively targeting halibut should be included, and amounted to 1912 trips over the 3 years. However, the probability of actually overestimating the numbers of trips that had halibut as their target species might be significant.

Modelling potential drivers of variations in bycatch of fulmars

Potential drivers of variations in fulmar bycatch were explored utilizing the generalized linear mixed model (GLMM) framework in the statistical software R (v. 3.1.2, R Development Core Team 2014). First, we included all variables of interest (Table 1) in a global model as additive fixed effects:

$$\begin{aligned} \textit{Fulmar bycatch} \sim\ & \textit{Hook type} + \textit{Season} + \textit{Year of capture} \\ & + \textit{Wind} + \textit{Mitigation} + \textit{Bait} + \textit{Dataset} + \\ & \textit{Area} + \textit{Number of hooks}. \end{aligned}$$

This model was also nested within the primary sampling unit, vessel. Specifically, random intercepts for *Vessel ID* were included in the model. A visual inspection of the response displayed many zero-events (i.e. trips with no bycatch), and indicated a relatively high proportion of multiple birds captured given a non-zero event. As the visual inspection gave no clear-cut answer to the appropriate response distribution, we included multiple candidate distributions to the global model including Poisson, negative binomial, zero-inflated Poisson and zero-inflated negative binomial distributions. The models were fitted using an AD-model builder platform through an R interface (glmmADMB package, Fournier *et al*., 2012; Skaug *et al*., 2014). The most appropriate distribution was selected based on *AICc*-values (weights calculated from the bbmle-package, Bolker and R Development Core Team, 2016), where the highest ranked model (i.e. with the lowest *AICc* value) was chosen for further analysis. We then constructed multiple models nested within our global model with the most appropriate probability distribution, to explore potential effects of the fixed variables. Each model represented different combinations, with inclusions or exclusions of the various fixed effects. In the candidate models, the variables included were interpreted as described in Table 1, expect for *Number of hooks* which was also assumed to be strictly proportional to fulmar bycatch (i.e. using log(*Number of hooks*) as an offset) in some models. Due to data limitations, we did not explore possible interaction effects between the fixed variables or included all possible combinations of variables. The final model used for evaluating effects on fulmar bycatch was chosen based on a comparison of two selection criteria, *AICc* and *BIC* (Kuha, 2004). These represent two different parsimony estimators that might be appropriate under slightly different model selection setting, depending on the process generating the data (see e.g. Aho *et al*., 2014). BIC is often considered more consistent compared with AIC as it picks the correct model whereas AIC picks a model more complex when increasing sample size (when sufficiently large) (Kass and Raftery, 1995). When choosing the most appropriate model for our system, we found it useful to use a combination of both selection methods as we had very little a prior knowledge of underlying processes generating the bycatch in this particular fishery. If the two model selection procedures were consistent in selecting the most supported model, we would have more confidence in that model.

Non-parametric testing of results from post-mortem examinations

To test for differences in distribution among age, sex and morph categories among the fulmars killed as bycatch, we applied simple Chi-square tests calculated as described by Zar (1984) and using the Yates correction for continuity (Yates, 1934) for all 2×2 and 2×1 contingency tables.

Results

Fulmar bycatch rates

In our study of the longline fishery for Greenland halibut outside North Norway in 2012–2014, 102 fulmars and two great black-backed gulls (*Larus marinus*) constituted the total bycatch of seabirds on the 426 fishing trips included in the analyses. Mean bycatch per trip was estimated to ≈ 0.24 birds ($SE \approx 0.09$). This was the naïve estimator, under the assumption of a simple random sample of trips from the entire fleet (i.e. ignoring possible variation between areas). Trying to account for the variation between areas, a stratified mean bycatch per trip was calculated to ≈ 0.16 birds ($SE \approx 0.07$). Estimated total bycatch of fulmars for all trips ($n = 1912$) in this fishery during 2012–2014 thus amounted to approximately 458 ($SE \approx 174$) birds utilizing the naïve estimator and 312 ($SE \approx 133$) birds using the stratified estimator. As a more fine-scale BPUE measure, the naïve estimator

Table 2. Differences in AICc and BIC values for the five most supported negative binomial GLMMs, from each model selection procedure.

Candidate model	ΔAICc
∼*Hook type* + *Dataset* + *Number of hooks (offset)*	0
∼*Hook type* + *Wind* + *Dataset* + *Number of hooks (offset)*	0.6
∼*Hook type* + *Dataset* + *Year of capture* + *Number of hooks (offset)*	1.2
∼*Hook type* + *Area* + *Dataset* + *Year of capture* + *Number of hooks (offset)*	5.8
∼*Hook type* + *Number of hooks (offset)*	5.9
	ΔBIC
∼*Hook type* + *Dataset* + *Number of hooks (offset)*	0
∼*Hook type* + *Number of hooks (offset)*	1.9
∼*Hook type* + *Wind* + *Dataset* + *Number of hooks (offset)*	4.6
∼*Hook type* + *Wind* + *Number of hooks (offset)*	5.9
∼ *1 (random intercept only)*	7.0

Only the fixed effects of the candidate models are shown. All models share common random effect: *Vessels ID*.

Table 3. Parameter estimates, including standard errors (*SE*), *z*-values and *p*-values for the most supported model from the model selection procedure.

Parameter	Estimate	*SE*	*z*-value	*Pr*(>|*z*|)
Intercept	−12.34	1.69	−7.30	<0.001
Non-swivel hooks	4.44	1.63	2.72	0.007
Self-registration	−3.66	1.25	−2.92	0.004

Table 4. Distribution of age and sex of northern fulmars caught in the Norwegian longline fishery for Greenland halibut.

	Age					
	Immature		Adult		All ages	
Sex	*N*	%	*N*	%	*N*	%
Female	7	29.2	17	70.8	24	28.2
Male	21	35.6	38	64.4	59	69.4
Unknown	1	50.0	1	50.0	2	2.4
Total	29	34.1	56	65.9	85	

The immature group includes also juveniles and subadult birds.

of bycatch rate per 1000 hooks was calculated to ≈ 0.045 ($SE \approx 0.017$), while the stratified bycatch rate per 1000 hooks was calculated to ≈ 0.031 ($SE \approx 0.012$). As there exist no records of the number of hooks set per trip in the Norwegian fishery statistics, we were not able to estimate the total bycatch of fulmars in this fishery based on the latter estimators.

Drivers of variations in fulmar bycatch

The most supported model, based on a cross-examination of *AICc* and *BIC* (Table 2), was a negative binomial model that included *Hook type*, *Dataset* and *Number of hooks* as an offset in the fixed effect structure (Table 3). The variance estimates for the random effect were ≈ 3.00 ($SD \approx 1.73$). Predictions for trips with non-swivel hooks revealed an almost 100-fold larger bycatch rate (mean = 0.76, $SE \approx 0.16$) compared with using swivel hooks (mean ≈ 0.008, $SE \approx 0.002$). There was also a very large effect of *Dataset* (i.e. type of observation) where trips with external observers had a more than 37 times higher predicted bycatch rate (mean ≈ 0.75, $SE \approx 0.16$) compared with trips where bycatch was registered by the fishers (mean ≈ 0.02, $SE < 0.01$). Surprisingly, we found no convincing effect of *Mitigation* (use of bird-scaring line). The model fit had a relatively low accuracy when compared with the bycatch data (r^2 of predicted bycatch vs. observed bycatch = 0.03). Accordingly, although our model picked up some trends for a few variables, much of the variation in our bycatch data appears to be random and remains unexplained by any of the measured variables. The model selection procedure based solely on AICc also revealed two other models competing for being the most supported (ΔAICc < 2), where one also included an effect of *Wind*, and the other *Year of capture* (see Table 2). We still put most emphasis on the model that both selection criteria agreed on as the most supported, also because the inclusion of either *Wind* or *Year of capture* to that model imposed very little change to the effect estimates for *Hook type* and *Dataset*, suggesting they would contribute little to the interpretation of our main results. Model averaged parameter estimates from the three most supported models from the AICc selection procedure are reported in Supplementary material (Table S1).

Sex, age and condition of fulmars killed

In total, 85 fulmars from the bycatch in 2012–2014 were examined post-mortem. About two-thirds of the birds were adults, and there was a strong sex bias with 71.1% of the sexed birds being males ($\chi^2 = 13.93$, $df = 1$, $p < 0.001$, Table 4). Most birds proved to be in good ($n = 50$, 60.2%) or moderate ($n = 31$, 37.3%) condition, and only 2 birds (2.4%) appeared to have low body reserves.

There was significant variation in both the spatial and the temporal distribution of the fulmar bycatch, with marked differences also between immatures and adults. Specifically, we found a temporal difference in the age distribution ($\chi^2 = 7.97$, $df = 1$, $p = 0.005$) with a predominance of adult birds (81.4%) in the catches in the second fishing period (July and August, $n = 43$), whereas catches in the first fishing period (May and June, $n = 42$) had an even distribution of the two age groups (50:50). On a spatial scale, immatures and adults were relatively evenly distributed in the catch in both the southern-most area (Area 05—Vesterålen, $n = 6$) and north-eastern area (03—Eastern Finnmark, $n = 11$), although sample sizes there were small, while adult birds dominated (72.3%) the catch in the more central area (04—Troms and West Finnmark, $n = 65$) ($\chi^2 = 12.06$, $df = 1$, $p < 0.001$).

The majority of the fulmars examined (92.6%) were of the light colour morphs (L and LL, Table 5), with no apparent sex- or age-related differences (tests of sums L + LL vs. sums D + DD, sex: $\chi^2 = 0.001$, $df = 1$, $p = 0.975$, age: $\chi^2 = 0.768$, $df = 1$, $p = 0.381$). The light morph adults were exclusively LL, in contrast to many of the light immatures ($\chi^2 = 15.46$, $df = 1$, $p < 0.001$). Interestingly, no double-dark birds were registered.

Discussion

In this study, we have estimated the incidental bycatch of northern fulmars in the Norwegian small-vessel longline fishery for Greenland halibut, a fishery that has received little focus in the literature, shown to have the potential for high rates of seabird

Table 5. Distribution of colour morphs (LL, double light; L, light; D, dark; DD, double dark) of northern fulmars caught in the Norwegian longline fishery for Greenland halibut in 2012–2014.

	Colour morph							
	LL		L		D		DD	
	N	%	*N*	%	*N*	%	*N*	%
Immature females	4	66.7	0	0	2	33.3	0	0
Immature males	9	56.3	6	37.5	1	12.5	0	0
Adult females	14	100	0	0	0	0	0	0
Adult males	30	93.8	0	0	2	6.3	0	0
Total	57	83.8	6	8.8	5	7.4	0	0

The immature group includes also juveniles and subadult birds.

bycatch (Fangel *et al.*, 2015). Our findings indicate that bycatch of fulmars in this fishery is currently relatively low, with an estimated total of between 180 and 630 birds taken in total over the years 2012–2014. Interestingly, we found that hook type affected bycatch rates, with swivel hooks having much lower bycatch rates than hooks without swivels. The exact causality behind this effect is not fully understood, but could for instance indicate that bait on a swivel hook behaves less predictably and is more difficult for the fulmars to catch, and/or that a swivel hook does not easily hook into the bird. It is also worth noticing that the swivel hook is heavier (~6 g) when compared with a similar sized non-swivel hook (~2 g), which may contribute to faster sinking rate of the line. In addition to the effect of hook type, bycatch estimates for trips with observers were considerably higher than for trips with no observer. The simplest interpretation is of course that bycatch of seabirds are underreported by the fishermen, which further implies that data from surveys without observers might seriously underestimate the true seabird bycatch rates. This has, for instance, been shown for bycatch of blue sharks in the Hawaii-based longline fishery (Walsh *et al.*, 2002) and for prohibited species catch within Alaskan fisheries (Warner, 2004). In contrast, Oliveira *et al.* (2015) reported bycatch estimates of seabirds were higher based on fishermen interviews compared with on-board observations. However, the distribution of fulmar bycatch per fishing trip shows an excess of zeros, and that, in a non-zero event, the probability of catching more than one bird is relatively high. These few yet sometimes extreme bycatch events are thus highly influential in controlling the bycatch estimates. As we have relatively few trips with observers compared with trips without observers, a few random events of unusually high bycatch on the observer trips could, therefore, have produced the observed effect. Hence, our restricted dataset does not allow us to conclude on the causality underling this difference. Nonetheless, the exclusion of the most extreme bycatch event (33 birds) among the observer trips still gave a significant effect of dataset (i.e. with/without observer). We can, however, conclude that the bycatch in general is difficult to predict from the spatio-temporal, environmental and other mitigation variables included in our analysis, which suggests that incidental bycatch of fulmars in the Greenland halibut fishery is more random than systematic. To fully address how, where and why such extreme bycatch events might occur, longer data series are needed and more variables should be explored. A possibly important factor not explored in this study is the effect of patchiness in fulmar distribution at sea, and how they accumulate around specific fishing vessels. More birds near the baited line might cause higher competition and bolder behaviour, and result in increased bycatch, especially in offshore areas where the more competitive gulls (Phillips *et al.*, 2010) are fewer.

The bycatch rates estimated in this study contradicts the higher bycatch rates documented in an initial study of the same fishery, through an access point survey interviewing fishermen about bycatch on their last fishing trip (Fangel *et al.*, 2015). However, the study of Fangel *et al.* (2015) was based on a much smaller sample size (19 trips) from a different year (2009), making direct comparisons less valid. Due to the seemingly high probability of capturing more than one bird in a non-zero bycatch event, only a few random non-zero bycatch events in a low sample of trips could produce an elevated bycatch estimate. When comparing the BPUE values, bycatch estimates have dropped substantially from an indicated 0.294 seabirds per 1000 hooks in the 2009 study (Fangel *et al.*, 2015), to 0.045 seabirds per 1000 hooks in the present, which as mentioned before, is based on a much larger data set. In a global review of bycatch in longline fisheries, Anderson *et al.* (2011) found that rates of BPUE in longline halibut fisheries ranged from 0.0071 seabirds per 1000 hooks in British Colombia, Canada, to 0.092 seabirds per 1000 hooks in the Kamchatka Region, Russia. Our new estimate for the Norwegian fishery is hence safely within the range of rates reported from these two other halibut fisheries.

Seabird distribution and numbers at sea vary considerably from year to year (e.g. Fauchald and Erikstad, 2002), which is likely to contribute to similar variations in bycatch rates. Even if our model results indicated little inter-annual variation in bycatch rates in 2012–2014, it is also important to point out that these years were all very poor breeding seasons for fulmars and other pelagic seabirds in north-western Norway (SEAPOP data portal, www.seapop.no), including Røst in the Lofoten area, the second largest fulmar colony on the Norwegian mainland. Breeding conditions for pelagic seabirds in 2009 were somewhat better and, even if fulmars still experienced a bad season (e.g. Anker-Nilssen, 2010), a higher colony attendance of fulmars in that summer as compared with the later years (T.A.-N., unpublished data) indicates that they had access to food that not only allowed them to stay longer in the colony area, but which may also have attracted birds from elsewhere. Another important difference between the studies worth noticing is a plausible difference in the ratio of swivel hooks, which were relatively new on the market in 2009 and gradually have come into use in the later years.

After many years of increase in European waters (e.g. Mitchell *et al.*, 2004), several populations of northern fulmar in the temperate part of the North Atlantic have declined significantly since the late 1990s, e.g. in UK and Norway (JNCC 2015; Anker-Nilssen *et al.*, 2016). Unfortunately, we know little about the status of the largest fulmar populations, most of which breed farther north and in general are less well monitored (e.g. Irons *et al.*, 2015). The drivers of the recent declines remain uncertain, but as these trends coincide with widespread problems for many other pelagic seabird species in the Nordic countries it is likely that climate-induced factors are involved, such as phenological mismatches between seabird breeding and prey availability (e.g. Durant *et al.*, 2005, Burthe *et al.*, 2012). Such effects may both be masked and amplified by fishery-induced variations in prey abundance (Frederiksen *et al.*, 2004, Bicknell *et al.*, 2013), making it even more complex to quantify the total population effects of bycatch on seabird survival rates.

Because of its high affinity to fishing vessels (e.g. Fisher, 1952), extreme foraging ranges in the breeding season (Weimerskirch *et al.*, 2001) and wide dispersal in winter (e.g. Hatch *et al.*, 2010), the sum of bycatch in a variety of fisheries might prove more important for the population dynamics of fulmars than for most other North Atlantic seabirds (Fisher, 1952). Assessing incidental bycatch is thus one important step on the way to uncover drivers behind the population declines, and certainly essential for identifying and implement effective mitigation measures to reduce seabird bycatch overall. In this context, it is still worth noting that the fulmar populations now in decline did increase substantially in the second half of the 20th century (e.g. Størkersen, 1994; Mitchell *et al.*, 2004; Lorentsen and Christensen-Dalsgaard, 2009), perhaps partly because they profited from discards and offal from the fishing fleet (e.g. Camphuysen and Garthe, 1997) which was then much larger (e.g. Kelleher, 2005). Thus, a recent reduction of this food source may have contributed to the population declines and will be further reduced when discarding of fish is soon to be banned in EU waters (as it successively was in Norway in the period 1987–2004). If the existing level of fulmar bycatch proves (or is likely) to be an important problem at the population level, it is also important to assess to what extent bycatch rates have increased in parallel with possible reductions in the overall food supply for fulmars or along with changes in fisheries practice towards a more automated offshore fishing.

Understanding what are the main drivers of survival and recruitment, as well as their relative importance, is key information for the management of any species. Assessing effects of bycatch on demographic traits of seabirds are, therefore, essential when trying to model its overall effects on the population level.

The effects of bycatch mortality on the population level strongly depend on the status of the source populations affected. The breeding population of fulmars on the Norwegian mainland is small and in decline, which makes it particularly vulnerable to increased mortality. In the present study, we were not able to determine the origin of the birds taken as bycatch. The fulmars sampled were, however, almost exclusively in the light colour morph, which strongly indicates they originated from populations in the temperate regions (van Franeker and Luttik, 2008). This was further supported by the lack of double-dark individuals, typically found in high-arctic colonies. Breeding fulmars have been estimated to have a max foraging range of 580 km (Thaxter *et al.*, 2012), with recent studies showing that they can travel up to 2700 km from the colony during the incubation period (Edwards *et al.*, 2013). In addition, failed breeders can undertake long trips of up to 1000 km away from the colony as the breeding season progresses (Edwards, 2015). The birds could thus originate from colonies located far away from the point of capture.

The significant age- and sex-specific bias in this study is comparable with that found for northern fulmars caught in the demersal longline fisheries in the Bering Sea (Phillips *et al.*, 2010). In both areas, there was a clear overweight of males and adults in the bycatch. This could suggest a sex-related variation in foraging locations, but dietary studies using fatty acids have shown that a difference in diet between males and females diminish during the incubation period (Owen *et al.*, 2013). As tracking studies so far have shown little evidence for sex-related differences in movements during the breeding season (Hatch *et al.*, 2010; Edwards, 2015), the male bias in the bycatch more likely suggests that males are more aggressive than females when competing for bait around the longliners.

In the last part of the breeding season (July and August) most of the birds captured were adults, compared with an even age distribution in May–June. This shift in age composition could be caused either by failed breeders from elsewhere dispersing into the area for feeding or by resident breeders staying longer in the study area when feeding conditions deteriorate than immatures, which are not constrained in their movements by having to return to the nest. As all three-study years were very poor breeding seasons for pelagic seabirds in north-western Norway, the latter alternative seems the most likely. Admittedly our sample size is low but, if the latter explanation is correct, an increased proportion of local fulmars in the bad years would imply that even low rates of bycatch could have significantly negative consequences for this already small and vulnerable population. A decreased food supply could perhaps also increase the remaining birds' attraction to fishing vessels and thereby their risk of bycatch, even if the body condition of most birds examined was seemingly fair.

We conclude that even if the bycatch of fulmar documented in this study was generally low, future studies are needed to address these issues in more detail, especially in the context of environmental changes predicted from climate change and what factors can trigger incidents of mass-bycatch.

Our study provided interesting findings in relation to the effect of mitigation measures. The much lower bycatch rate for swivel hooks is highly promising and needs confirmation by similar assessments for other longline fisheries. Bird-scaring lines are used extensively in the halibut fishery as in other longline fisheries, but in this case seemingly without any apparent effect in terms of reduced bycatch rates. However, this might be because the design and protocols for use are lacking and that scaring lines perhaps are only employed when many birds are present and bycatch has already occurred. Future research on optimal design and operational use of bird-scaring lines for small longline vessels is thus required.

Supplementary data

Supplementary material is available at the *ICESJMS* online version of the manuscript.

Acknowledgements

We are grateful to the fishermen that collected data and to our local organizers assisting the data collection in the field: B. Altman, Ø. Hansen, J. Andreassen, O. Telebond, L. Godvik, J.R. Knutsen and O. Berglund. We also want to thank members of the project's reference group: B. Veie-Rosvoll, M. Irgens (Norwegian Environment Agency), K. Nedreaas, J.H. Vølstad (Institute of Marine Research), M. Overvik (Directorate of Fisheries) and E. Lorentsen (The Norwegian Fishermens' Association), for valuable discussions and input. The editor and two anonymous reviewers have given helpful feedback on the manuscript. Hannah Harrison has corrected the language. We are thankful for these contributions, which have improved the article.

Funding

This research was funded by the Norwegian Environment Agency.

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Handling editor: Stephen Votier