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# Estimating protected species bycatch from limited observer coverage: A case study of seal bycatch in static net fisheries

Cian Luck <sup>a, b, \*</sup>, Mark Jessopp <sup>a, b</sup>, Oliver Tully <sup>c</sup>, Ronan Cosgrove <sup>d</sup>,  
Emer Rogan <sup>b</sup>, Michelle Cronin <sup>a</sup>

<sup>a</sup> MaREI Centre for Energy, Climate, and Marine, Environmental Research Institute, University College Cork, Ireland

<sup>b</sup> School of Biological, Earth, and Environmental Sciences, University College Cork, Ireland

<sup>c</sup> Marine Institute, Oranmore, Galway, Ireland

<sup>d</sup> Irish Sea Fisheries Board (BIM), New Docks, Galway, Ireland



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## ABSTRACT

Fisheries bycatch represents a major anthropogenic threat to marine megafauna worldwide. To identify populations at risk, it is essential to estimate the total number of individuals removed from a population as bycatch. However, estimating total bycatch remains challenging due to the often-limited scope of monitoring programmes. In this study, we aimed to maximise the value of limited bycatch data collected by scientific observers and self-reported by fishers to provide estimates of total seal bycatch for static net fisheries operating in Irish waters. We constructed a model of bycatch rate as a function of known predictors of seal bycatch, and used this to predict bycatch rates throughout the Irish Exclusive Economic Zone. Annual estimates of seal bycatch, from 2011 to 2016, ranged between 202 (90% CI: 2–433) and 349 (90% CI: 6–833) seals per annum. Estimated bycatch exceeded the precautionary threshold of Potential Biological Removal (PBR = 165–218;  $F_r = 0.5$ ) for the national grey seal population but was below less conservative threshold values (PBR = 330–437;  $F_r = 1.0$ ), with confidence intervals spanning both. Further research on the population structure of grey seals in the Northeast Atlantic is needed to set appropriate bycatch thresholds. Nonetheless, this study shows that by utilising predictive models to maximise the value of limited bycatch observer effort, we can produce informative estimates of protected species bycatch and highlight areas of high bycatch risk. We present this as a case study for maritime nations with comparatively limited bycatch data to fill key data gaps in protected species bycatch worldwide.

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## 1. Introduction

The incidental catch of non-targeted species, known as bycatch, is recognised as a major threat to marine species worldwide (Wallace et al., 2010; Anderson et al., 2011; Lewison et al., 2014; Dias et al., 2019). Air-breathing megafauna, including marine mammals, turtles and seabirds, are particularly vulnerable to this kind of mortality and have been recorded as bycatch in over 90 countries (e.g. Wallace et al., 2010; Anderson et al., 2011; Reeves et al., 2013; Lewison et al., 2014; Dias

\* Corresponding author. MaREI Centre for Energy, Climate, and Marine, Beaufort Building, Haulbowline Road, Ringaskiddy, County Cork, P43 C573, Ireland.

E-mail address: [cian.luck@ucc.ie](mailto:cian.luck@ucc.ie) (C. Luck).

et al., 2019). The majority of these bycatch events have involved static entangling nets and longline fisheries (Wallace et al., 2010; Žydelis et al., 2013; Lewison et al., 2014; Brownell et al., 2019).

For many endangered species, including the vaquita (*Phocoena sinus*; Taylor et al., 2017), Mediterranean monk seal (*Monachus monachus*; Karamanlidis et al., 2008), and several albatross species (Pardo et al., 2017), bycatch represents the dominant risk to population and species survival. To assess the conservation status of protected populations and enable effective mitigation for endangered species, we must first estimate the proportion of a population that is being removed by fisheries as bycatch (Soykan et al., 2008). Typically, this is achieved by first assessing the bycatch rate through on-board observations by dedicated scientific observers, then extrapolating observed bycatch levels to include the total fishing effort within a population's range or management area. Bycatch estimates can then be compared to threshold values to assess if the population can sustain the estimated level of mortality without failing to meet conservation or management targets (Wade, 1998; Lonergan, 2011; Curtis et al., 2015).

However, estimating the total number of individuals caught across an entire fishing fleet is challenging due to low observer coverage and a paucity of detailed data on the distribution of fishing effort. Scientific observers provide the best means of estimating bycatch rates at sea, however, the over-dispersed nature of bycatch in some fisheries often necessitates high observer effort to detect bycatch events (Babcock et al., 2003; Barlow and Berkson, 2012); low observer effort in a given area can, by chance, result in atypically high or low bycatch rates (Rogan and Mackey, 2007; Sims et al., 2008; Wakefield et al., 2018); and even large-scale observer programmes may only include a small proportion of the total fishing effort in a given area (Lewison et al., 2004; Moore et al., 2008). Unless observer coverage includes 100% of the fishing effort in a given area, we must assume that the observed fishing effort is representative of the unobserved, and the more limited the observer coverage the broader the assumptions made (Wakefield et al., 2018). Logbooks remain the most comprehensive source of information on catch and effort by fishing vessels. However, beyond where and when fishing occurred, much of the key information required to estimate bycatch such as gear type, fishing effort and mesh sizes (Cosgrove et al., 2016; Northridge et al., 2017a; Luck et al., 2020), is lacking or unreliably reported.

Grey seals (*Halichoerus grypus*) are distributed across the North Atlantic with three recognised population centres in the Northwest and Northeast Atlantic, and the Baltic Sea. In Europe, this species is protected as an Annex II species under the European Union Habitats Directive (92/43/EEC), which obliges member states to ensure populations are maintained at "favourable conservation status". Home to approximately 38% of the global grey seal population, the United Kingdom (UK) provides annual estimates of total seal bycatch by UK vessels, based on observer data (Northridge et al., 2017b; Russell et al., 2019). The UK however, is the only European country with a long-running, dedicated observer programme for bycatch of protected, endangered or threatened species (ICES, 2018), and considerable data gaps exist regarding bycatch levels among non-UK fishing fleets and those operating in neighbouring countries, including the Republic of Ireland. Ireland's population of grey seals, on the western edge of the Northeast Atlantic population's range, is estimated to be between 7284 and 9365 individuals, representing approximately 6% of grey seals in Western Europe (Ó Cadhla et al., 2013; OSPAR COMMISSION, 2017). Recent research has highlighted the risk of seal bycatch in specific static net fisheries in Irish waters (Cosgrove et al., 2016; Luck et al., 2020), however, estimating levels of bycatch mortality has not been possible to date due to limited observer coverage and key information gaps in logbook data.

In this study, we aim to address critical gaps in understanding the scale and sustainability of bycatch of Northeast Atlantic grey seals by estimating the level of seal bycatch across all static net fisheries operating within the Irish Exclusive Economic Zone (EEZ). To maximise the value of limited observer data, we construct a predictive model of seal bycatch including known drivers of seal bycatch, and then apply this model to fishing effort data reported in logbooks. We use expert knowledge to infer key missing or unreliably reported information in the logbooks. We present this as an approach that could be used by maritime nations with comparably limited bycatch data to do more with less, and fill key information gaps required to assess protected species bycatch globally. Finally, we compare these estimates to those produced by simple extrapolation of observed bycatch rates to total fishing effort for comparative purposes.

## 2. Methods

### 2.1. Observed effort and bycatch data

Observations of seal bycatch were recorded by scientific observers and self-reported by skippers on fishing vessels off the west, southwest, and south coasts of Ireland, between January 2010 and December 2018. These data comprised 3118 hauls from 17 vessels ranging in size from <10 m to 22 m length, and accounted for approximately 1.3% of the total reported static net fishing effort within the Irish EEZ over that time. Data were collected as part of separate research programmes along the west and south-west coasts of Ireland, including on-board observations of bycatch by scientific observers and self-reported data by skippers when observers were not present, and one extensive dataset of self-reported data on fishing effort and catch composition, including seal bycatch, in the south of Ireland (Table 1). The presence or absence of a scientific observer was included in our analysis to control for potential bias in self-reported versus observer-collected bycatch data. All fishing vessels used forms of gillnet/entangling nets, as described by the Food and Agricultural Organisation of the United Nations (FAO). The nets used in this study could broadly be classified as "gillnets", "tangle", and "trammel" nets which differed primarily in mesh size (Table 2). All nets were set on the sea floor (bottom-set) with weighted lead lines and buoyant head ropes to keep the nets vertical in the water. Some, but not all, gillnets were fitted with plastic floats on the head ropes for extra buoyancy. All nets

**Table 1**

Description of bycatch data collection programmes included in this study.

Data collection programme	Data collected by	Number of hauls	Number of vessels	Vessel size range
Interactions between seals and entangling net fisheries (see <a href="#">Cosgrove et al., 2016</a> )	Observers	358	3	12–22 m
Bycatch monitoring programme, led by the Marine Institute of Ireland (ongoing)	Observers	63	3	<12 m
	Fishers	477	10	<12 m
Personal fishing records from static net fisher	Fishers	2220	1	<12 m

**Table 2**

Description of nets included in bycatch data collection programmes.

Net type	Mesh size (cm)	Outer mesh size (cm)	Set length (km) – Mean ( $\pm$ SE); Median	Soak time (days) – Mean ( $\pm$ SE); Median	Seals per unit effort (SPUE) – Mean ( $\pm$ SE)
Gillnet	14	NA	2.41 ( $\pm$ 0.09); 1.00	0.94 ( $\pm$ 0.04); 1	0.015 (0.009)
Tangle	$\geq$ 27	NA	1.20 ( $\pm$ 0.03); 0.71	5.04 ( $\pm$ 0.10); 4	0.031 (0.006)
Trammel	27	81	0.72 ( $\pm$ 0.06); 0.52	2.91 ( $\pm$ 0.046); 3	0.055 (0.010)

comprised a single net wall, with the exception of trammel nets, which were three-walled (the mesh size of the outer net walls being substantially larger than the central net).

Recorded data included the date the nets were shot and hauled, GPS location at the beginning and end of each haul, net type, mesh size, catch, and bycatch composition. The net length was taken as the distance between start and end locations of a haul in metres and was measured using the “sf” package ([Pebesma, 2018](#)) in the statistical framework R ([R core team 2018](#)). Any net lengths longer than 10 km (approx. 4% of the data) were assumed errors and were removed from the dataset.

The rate of seal bycatch or seals per unit effort (SPUE) was defined as:

$$SPUE = \frac{\text{number of seals caught}}{\text{net length (km)} \times \text{soak time (days)}}$$

The majority of data included the dates the nets were shot and hauled but not the times. Therefore, nets soaked for less than a day were assigned a soak time of 0.5 days.

## 2.2. Fishing effort data

### 2.2.1. EU fishing effort

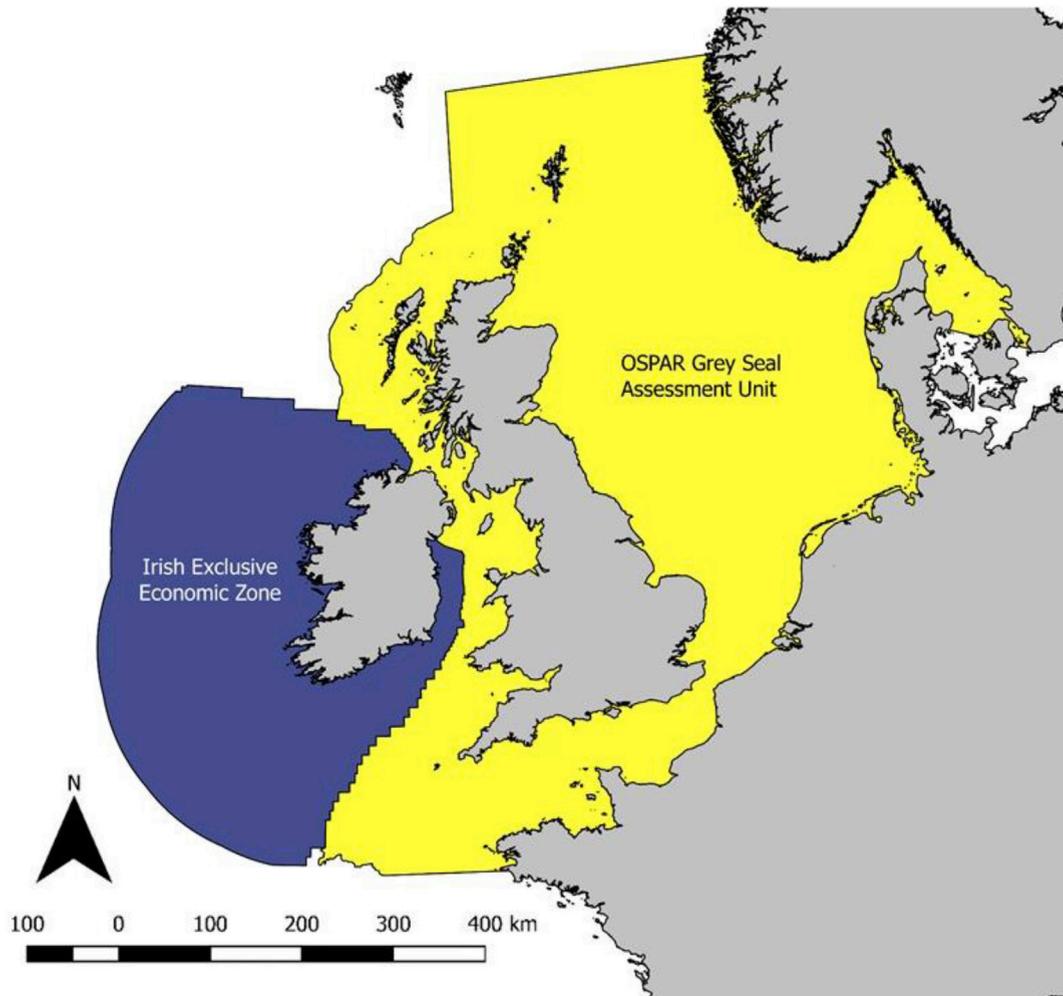
Data on European fishing effort were downloaded from the Joint Research Centre data dissemination tool of the European Commission Scientific, Technical and Economic Committee for Fisheries (STECF; see <https://stecf.jrc.ec.europa.eu/data-dissemination>). STECF effort data were based on logbooks for vessels over 10 m in length only, as vessels smaller than 10 m are not required to keep logbooks. Fishing effort was reported as hours fished and was spatially aggregated by ICES statistical rectangles and temporally by quarter and year. Data could be further differentiated between net type (gillnet or trammel), vessel length category, and vessel home country. Only ICES rectangles that were at least partly within the Irish EEZ were included in the analysis (see [Fig. 2](#)), and fishing effort from Irish vessels were excluded, as these were accounted for in more detailed national logbooks, to which we had access.

### 2.2.2. National fishing effort

The Marine Institute, Ireland, provided aggregated and anonymised logbook data for Irish vessels, which included location (aggregated to ICES statistical rectangles), net type, species landed, and fishing effort in days fished per trip. Only logbook entries from Irish vessels were included. As with EU fishing data, ICES rectangles that were at least partly within the Irish EEZ were included in the analysis ([Fig. 2](#)).

### 2.2.3. Inferring gear type

National and STECF logbook data aggregate gear types differently, but both essentially define all static nets as gillnets (all non-trammel nets) or trammel nets. This was a critical impediment to our analysis as studies have shown mesh size and net type to have significant effects on seal bycatch rates ([Cosgrove et al., 2016](#); [Northridge et al., 2017a](#); [Luck et al., 2020](#)), and we believed combining gears would lead to considerable over and underestimation of bycatch in certain fisheries. To differentiate between gillnet and tangle net effort, we examined the species landed, and through consultation with fisheries experts identified a number of indicator species which were most likely caught with a certain net type ([Table 3](#)). For each logbook entry, we assumed that nets listed as trammel nets were indeed trammels. For nets listed as “gillnets” we identified the top

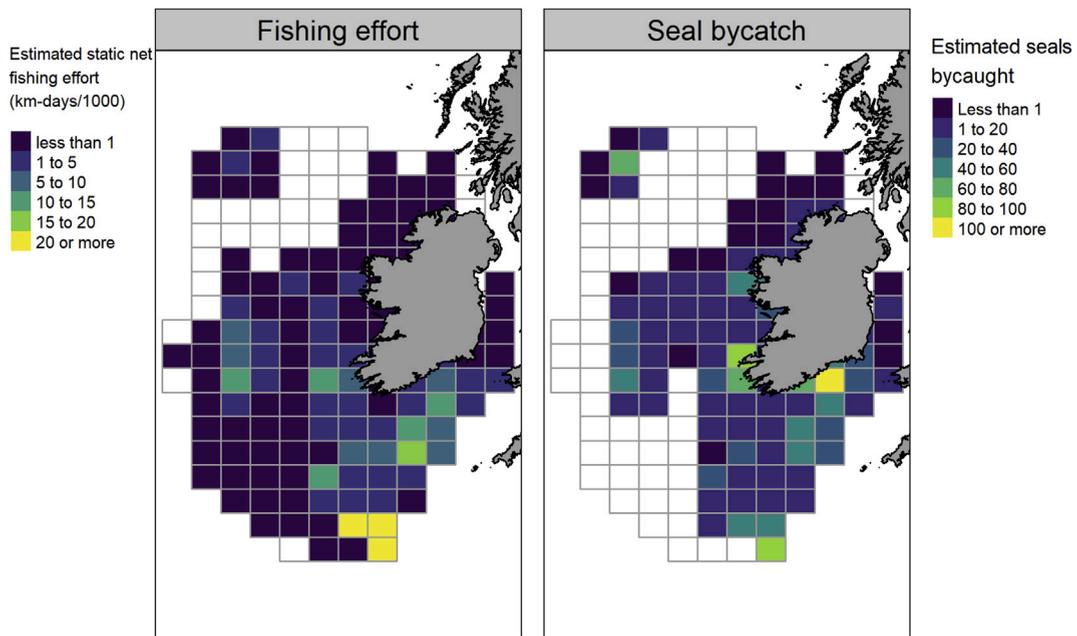


**Fig. 1.** Map of the Irish Exclusive Economic Zone (EEZ) in blue, and the OSPAR Grey Seal Assessment Unit in yellow, inclusive of the Irish EEZ. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

three species landed by weight per logbook entry, and first attempted to infer the net type from the top species landed. If the top species did not include any “indicator” species, we then looked to the second species landed by weight, and then the third if necessary. For national logbook data, this was carried out on a trip-by-trip basis, however, the STECF aggregate landings and effort data as separate datasets. For each STECF record, effort was matched to landings first by combination of ICES rectangle, vessel nationality, vessel length category, quarter, and year. If no gear type could be inferred from these landings, data were then matched by rectangle, nationality, quarter, and year only, and then if necessary, by rectangle, quarter, and year only. Some species, such as shrimp or shellfish, were unlikely to have been caught with static nets, and we excluded these records (<1% of data) from our analysis, as we assumed they were a result of mislabelled gear types. Less than 4% of logbook entries could not be assigned a net type and for these we took the precautionary assumption that unknown nets were tangle nets, as these have been found to have higher levels of seal bycatch in Irish waters (Cosgrove et al., 2016; Luck et al., 2020).

### 2.3. Geo-processing

Following Luck et al. (2020), we incorporated data on water turbidity and seal density in our analysis, as these were found to significantly affect seal bycatch rates. Water turbidity was sourced from the European Space Agency GlobColour project (see <http://www.globcolour.info/>), as 8-day composites at 4 km<sup>2</sup> spatial resolution. Turbidity was averaged along the length of each observed net and across each statistical rectangle. For observed nets we further averaged values between shoot and haul dates where necessary. Seal density was estimated by calculating the minimum distance from each fishing net (observer data) or the centroid of each ICES rectangle (logbook data) to each of the seven major grey seal breeding colonies in Ireland, and dividing by the estimated proportion of the national breeding population (according to Ó Cadhla et al., 2013) at each site.



**Fig. 2.** Static net fishing effort, based on logbook records, within the Irish Exclusive Economic Zone (EEZ) and estimated total levels of seal bycatch between 2011 and 2016.

**Table 3**

Landed species, as listed in logbook entries, used as indicator species to differentiate between gillnets and tangle nets.

Indicator species landed, as listed in logbooks	Inferred gear type
Cod, Haddock, Hake, Pollack, Saithe	Gillnet
Monkfish, Skates and rays, Black Sole, Turbot, Brill, Spiny Lobsters	Tangle net
Nephrops, Scallop, Shrimp, Whelk	Exclusive of static nets

Thus, proximity to a large colony was weighted more heavily than the same distance to a smaller colony. The lowest scaled distance was selected for each haul or rectangle as the distance to nearest colony. We focused on grey seal colonies because of their greater size relative to harbour seal (*Phoca vitulina*) colonies, the larger foraging range of grey seals relative to harbour seals (Cronin et al., 2014), and the higher incidences of grey seal bycatch in static nets (Cosgrove et al., 2016; Northridge et al., 2017b). To estimate the depth at which nets were set, we used bathymetry data from a gridded global terrain model produced by General Bathymetric Chart of the Oceans data portal ([https://www.gebco.net/data\\_and\\_products/gridded\\_bathymetry\\_data/](https://www.gebco.net/data_and_products/gridded_bathymetry_data/)) and identified the minimum depth within each ICES statistical rectangle.

#### 2.4. Predicting bycatch rates

We used a negative binomial generalised linear mixed effects model (GLMM) to construct a predictive model of seal bycatch as a function of predictors known to affect seal bycatch (Luck et al., 2020), modified to only include predictor variables that could be applied to logbook data. The rate of seal bycatch per hundred units of effort (SPUE), rounded to the nearest integer, was included as the response variable to allow for a negative binomial distribution.

$$\log(SPUE \times 100) = Dist + SDD + Net + Obs + (1|vessel\ ID) + (1|year)$$

where, *Dist* is the minimum, scaled distance to colony, *SDD* is water turbidity as Secchi disc depth in metres, *Net* is the type of net (i.e. gillnet, tangle, or trammel) and *Obs* the presence/absence of an observer, both included as categorical predictors.  $(1|vessel\ ID)$  and  $(1|year)$  were included as random effects to account for inter-vessel and inter-annual variation. *Dist* and *SDD* were square-root transformed to allow for model convergence. We employed a backwards step-wise model selection based on second-order Akaike information criterion (AICc), and then used the final model to predict the rate of seal bycatch across all reported fishing effort. We also calculated mean gear-specific bycatch rates for gillnets, tangle nets and trammel nets for comparative purposes.

## 2.5. Extrapolating to fleet level

Once the rate of bycatch had been estimated, extrapolating the total number of seals caught within the EEZ became a relatively simple calculation of bycatch rate multiplied by fishing effort. The logbook data included the number of days fished per trip, the number of nets and the total length of nets deployed. However, in the national logbooks some fields were left mostly blank (net length) or filled with inconsistent or implausible values (number of nets, which ranged from 2 to 475) and were therefore considered unreliable. In these cases, we substituted the net lengths entered in the logbooks with the median observed net lengths and soak times, per net type (Table 2), and for the number of nets deployed per day of fishing effort we first calculated the medians per vessel, before calculating the median of these values to at least partially account for inter-vessel variation. We considered some of the extreme values of the reported number of days fished per trip unlikely to be true (e.g. 72 days fished) and replaced any outliers exceeding the 99th percentile (~1% of the data) with the median reported value. Effort in km-days was then calculated as:

$$E_{km-days} = D * N_{npd} * L_{(g,t,tr)} * S_{(g,t,tr)}$$

where  $D$  is the number of days fished, as reported in the logbooks,  $N_{npd}$  is the median of the median number of nets deployed per day by each observed fishing vessel,  $L_{(g,t,tr)}$  is the median observed net length in km, and  $S_{(g,t,tr)}$  is the median soak time in days for gillnets ( $g$ ), tangle ( $t$ ), or trammel ( $tr$ ) nets. The STECF provided fishing effort as hours fished, so for the EU fleet fishing effort in km-days was estimated as ( $E_H$  = sum of STECF fishing effort in hours fished):

$$E_{km-days} = \frac{E_H}{24} * N_{npd} * L_{(g,t,tr)} * S_{(g,t,tr)}$$

We then estimated total bycatch by multiplying fishing effort by the predicted bycatch rates according to the GLMM ( $E_{km-days} * SPUE_{GLMM}$ ). For comparison, we also estimated total bycatch as the product of fishing effort and mean observed gear-specific bycatch rates ( $E_{km-days} * SPUE_{(g,t,tr)}$ ), hereafter referred to as the “applied average” method.

## 2.6. Confidence limits

Bootstrapping, or random resampling of observer data with replacement, was used to generate confidence limits around bycatch estimates. We resampled the subset of observer data usable in our predictive GLMM 1000 times, recalculated the mean bycatch rates, and refitted the GLMM to each iteration. The 5th and 95th percentiles of these bycatch estimates were taken as the 90% confidence interval for bycatch estimates.

## 2.7. Potential Biological Removal

In the absence of an agreed bycatch limit within EU waters and defined management units for European grey seals, estimates of total bycatch were compared to thresholds of Potential Biological Removal (PBR) for the national grey seal population. PBR is enshrined in the Marine Mammal Protection Act of the USA, which requires mitigation measures to be enacted when “incidental mortality and serious injuries” caused by human activities exceed PBR (Wade, 1998; Taylor et al., 2000). This is designed to allow protected populations to maintain at least half of their estimated population size, given no human-caused mortality. PBR is calculated as:

$$PBR = N_{min} * \frac{R_{max}}{2} * F_r$$

where  $N_{min}$  is the minimum population estimate,  $R_{max}$  is the intrinsic growth potential for the population (0.12 is the standard value for pinnipeds), and  $F_r$  is a recovery statistic set to between 0.1 and 1.0. By using the minimum population estimate in the calculation of PBR, greater uncertainty in population size will lead to wider confidence intervals and lower bycatch limits. We relied on the most recent assessments of the national breeding population of grey seals in the Republic of Ireland for  $N_{min}$ . Carried out by Ó Cadhla et al. (2007, 2013), these assessments used counts of new-born pups and production estimation models to estimate total pup production, and then used standard ratio estimates to produce all-age population estimates. Lower values of  $F_r$  (e.g.  $F_r = 0.1$ ) allow for depleted populations to recover quickly; the default value of 0.5 is recommended to allow for bias in estimates of population size, structure, growth rates, and bycatch removals; while greater knowledge and certainty around these estimates may allow for a higher  $F_r$  value to be used (Taylor et al., 2000). The most appropriate value of  $F_r$  may also depend on specific management objectives for the population, and for this reason, we present bycatch estimates relative to the full range of PBR values, with  $F_r$  ranging from 0.1 to 1.0.

Bycatch estimates are inclusive of both grey seals and harbour seals, however, due to few observations of harbour seal bycatch, and the more inshore distribution and reduced interconnectivity between haul-out sites of harbour seals relative to grey seals (Cronin, 2011; Vincent et al., 2017), we assumed that the majority of bycaught seals were grey seals. As we assumed that seals became entangled while the nets were set on the seafloor, we only included estimates of bycatch from ICES

rectangles with a minimum depth of 500 m or less given the deepest recorded dive by a grey seal in Irish waters of 455 m (Jessopp et al., 2013).

### 3. Results

#### 3.1. Observed effort and bycatch

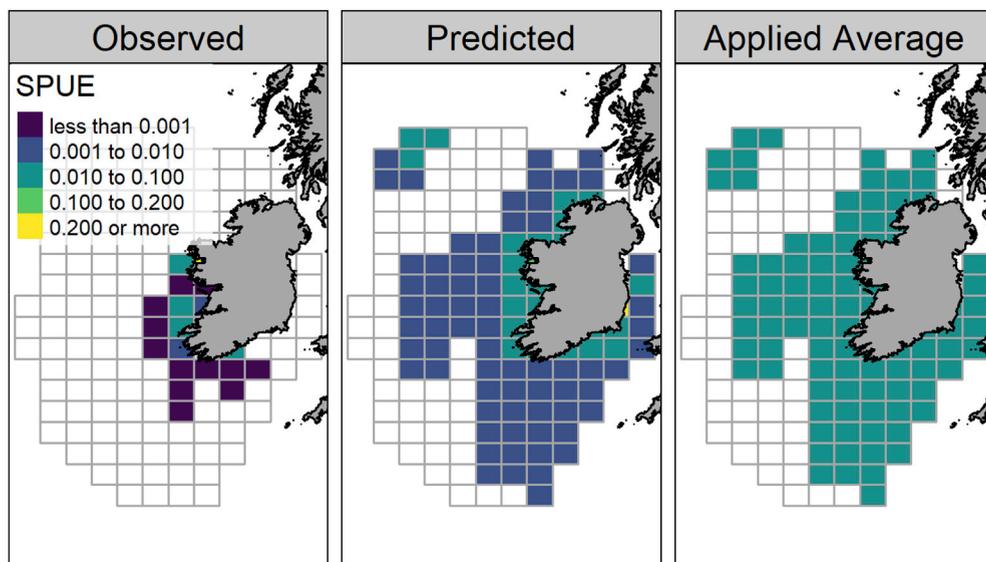
The number of nets deployed per day across all vessels ranged from one to eleven. The median nets per day varied considerably between vessels, ranging between one and seven. The median of these values was two nets per day. The median net length of gillnets was longer than that of tangle and trammel nets and gillnets had the shortest median soak time (Table 2). Seal bycatch was recorded in 197 (6.3%) of the observed hauls, and 257 seals were bycaught in total. Of these, 63 were identified as grey seals, and 10 as harbour seals. Observed SPUE ranged from 0.000 to 13.726, with a median of 0.000 and a mean of 0.040 ( $\pm 0.005$  SE). SPUE was lowest in gillnets and highest in trammel nets (Table 2). Bycatch data were recorded in 35 of the 166 ICES statistical rectangles within the Irish EEZ, and incidences of bycatch were recorded in seven. The highest mean rates of observed SPUE occurred in relatively inshore areas, particularly along the west coast, at the northern extent of our observer coverage (Fig. 3).

#### 3.2. Fishing effort

Between 2011 and 2016, logbooks reported static net fishing activity in 124 of the 157 ICES rectangles within the Irish EEZ (Fig. 2). This involved vessels from nine European countries, with French (43%), English (23%), and Irish (21%) vessels responsible for most of the effort. The highest levels of static net fishing were recorded in the Celtic Sea.

#### 3.3. Predicting bycatch rates

The predictive model failed to converge with both vessel ID and year included as random effects, so year was excluded and the presence/absence of an observer was not retained in model selection, resulting in a final model with a lognormal conditional  $R^2$  value of 0.24 (Table 4). The GLMM predicted values of SPUE between 0.000 and 0.300, with a mean of 0.013 (Fig. 4). Fig. 3 shows the spatial distribution of predicted values of SPUE relative to what was recorded by observers, and the applied average bycatch rates. The GLMM predictions failed to replicate the highest levels of SPUE observed on the west coast, but rather generally predicted moderately high rates of bycatch close to shore and major grey seal colonies, whereas the applied average approach resulted in intermediate bycatch rates applied throughout the study area.

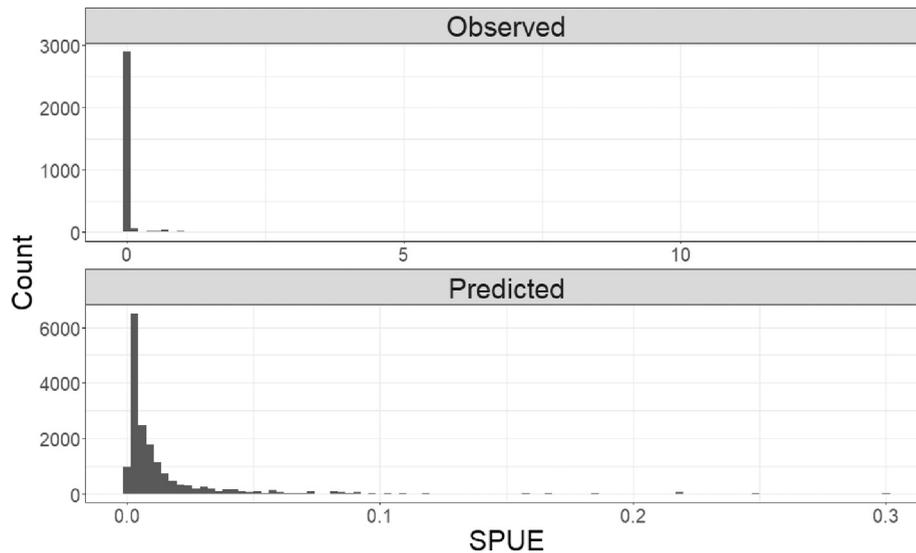


**Fig. 3.** Mean observed seal bycatch per unit effort (SPUE; “Observed”), predicted rates of SPUE using a generalised linear mixed effects model (“Predicted”), and mean gear-specific rates of SPUE applied throughout the EEZ (“Applied Average”). Empty cells indicate areas with no observed or reported fishing effort and all plots share the same colour scale for SPUE. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

**Table 4**

Model averaged estimates, with standard errors (SE), 90% confidence intervals (CI), z values and P values for seal bycatch per unit effort (SPUE) as a function of known predictors of seal bycatch, fitted with a negative binomial generalised linear mixed effects model (GLMM).

	Estimate	SE	Lower CI	Upper CI	z	P
Intercept	5.608	2.838	0.979	10.412	1.976	<0.05
Net type: tangle	1.393	1.172	-1.137	3.098	1.188	0.235
Net type: trammel	0.752	1.166	-1.774	2.356	0.645	0.519
Secchi disc depth	-1.815	0.708	-3.008	-0.664	-2.562	<0.05
Distance to colony	-0.028	0.045	-0.112	0.043	-0.629	0.530



**Fig. 4.** Histogram of observed values of seal bycatch per unit effort (SPUE) relative to predicted values of SPUE across the entire static net fishing fleet within the Irish EEZ.

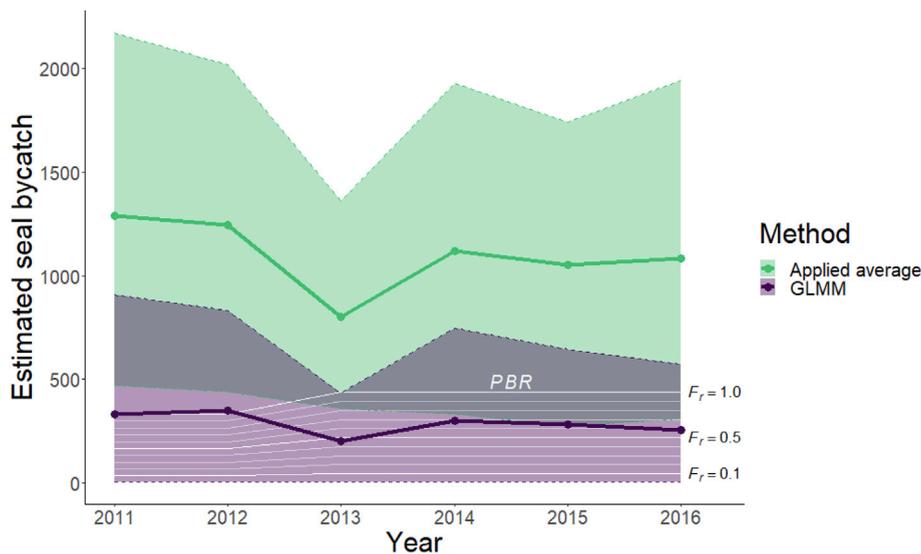
### 3.4. Total seal bycatch within Irish EEZ

The predictive model resulted in estimates of annual total seal bycatch that ranged between 202 (90% CI: 3-433) and 349 (90% CI: 6-833), with no clear trend over time (Fig. 5). The lower bycatch estimates in 2013 coincided with a one-year decrease in reported fishing activity by French vessels using gillnets in the Irish EEZ. These annual estimates exceeded the default value of PBR (165-218;  $F_r = 0.5$ ) for the national grey seal population but were below the least conservative PBR (330-437;  $F_r = 1.0$ ) values, albeit with overlapping confidence intervals (Fig. 5). Alternatively, using the applied average method resulted in bycatch estimates that exceeded the least conservative value of PBR by almost 300% (Fig. 5).

## 4. Discussion

Using a predictive model and expert knowledge allowed us to produce estimates of total seal bycatch in Irish waters from limited observer data, the first of their kind. More direct bycatch extrapolation methodologies have proven effective elsewhere, such as in the UK where observed metier-specific bycatch rates are applied to logbook data (e.g. Northridge et al., 2017b), but this relies on extensive on-board data collection to estimate bycatch rates across a wide range of detailed fishing metiers. This approach is less suitable in regions with limited observer data across a smaller range of fisheries, and coarse resolution of fishing effort data, that occurs in the majority of marine jurisdictions globally. The observed mean seal bycatch rate recorded in the Irish observer programme was 7.5 times higher than what has been observed in the UK (Northridge et al., 2017b), potentially driven by a smaller sample size but comparatively high observer effort on board inshore fisheries in Ireland, close to major seal haul-outs.

Relative to the modelled bycatch rates, applying mean observed rates across the entire EEZ generally underestimated bycatch in inshore areas, overestimated bycatch further offshore (Fig. 3), and inflated overall bycatch estimates compared to using the predictive model (Fig. 5). Whereas applying average bycatch rates assumed a spatially independent relationship between observed and unobserved bycatch, the GLMM method assumed that the observed effects of known predictors of seal bycatch were consistent across observed and unobserved fishing effort (Luck et al., 2020). This method provided a more informative distribution of bycatch risk (Fig. 3) and allowed us to produce predictions of bycatch rates in response to changing



**Fig. 5.** Estimated levels of seal bycatch across all static net fisheries within the Irish EEZ between 2011 and 2016. Point estimates (solid lines) and associated 90% confidence intervals (shaded area) were produced by applying predicted (GLMM) and gear-specific mean (applied average) bycatch rates to fisheries logbook data. White lines indicate the Potential Biological Removal (PBR) thresholds for the Irish grey seal population, with an  $F_r$  statistic ranging from the minimum (0.1) and maximum (1.0) values, in increments of 0.1.

environmental variables (in this case, water turbidity). Given the growing importance of developing dynamic management strategies to protect highly mobile marine species (Dunn et al., 2016; Maxwell et al., 2020), if policy makers wished to reduce bycatch to specified levels, this could provide an invaluable tool for identifying areas or seasons of high bycatch risk to target mitigation measures accordingly.

The national population of grey seals was first assessed by means of a comprehensive national survey effort during the 2005 breeding season and repeated in 2013 (Ó Cadhla et al., 2007, 2013). The most recent survey provides supporting evidence of grey seal population growth since the 1990s, but cautions that “robust statistical data on grey seal population viability or trends in Ireland are not available at present” (Ó Cadhla et al., 2013). The choice of which  $F_r$  value to use depends on the population status and the management objective for the population. There are currently no bycatch limits for grey seals within the EU and the criteria for “favourable conservation status” are poorly defined, leading to unclear management objectives for protected seal populations. PBR provides a precautionary reference point/limit for non-natural mortality designed to allow marine mammal populations to reach a maximum net productivity level, or approximately 50–70% of the carrying capacity for the population (Wade, 1998). These results do not simplistically define bycatch estimates as sustainable or unsustainable, especially considering the relatively wide confidence intervals, but rather highlight that, with the information presently available, bycatch may represent a significant pressure on the national grey seal population.

For any bycatch limit reference point such as PBR to be effective, it is critical that we identify demographically independent management units within a species range, and manage such units independently (Taylor, 1997; Curtis et al., 2015). If the minimum population size used in PBR represents the total of multiple distinct populations, then even the most conservative estimates of PBR may fail to prevent local depletions at this level. Genetic analysis provides an important means of delineating discrete populations (DeYoung and Honeycutt, 2005), however, to date this has not been applied to grey seals at a broad geographic scale in Western Europe. Animal movements can provide a hint at population structure, but tagged grey seals have been shown to regularly move between major colonies around Ireland, the UK, and France (Jessopp et al., 2013; Vincent et al., 2017; Carter et al., 2020), providing no clear evidence of demographic isolation. In the absence of such information, the OSPAR Commission define an Assessment Unit for grey seals in Western Europe that extends from the Atlantic margin to the greater North Sea area, inclusive of Irish waters (OSPAR COMMISSION, 2017, Fig. 1). Future studies on the genetic structure of grey seals in Western Europe will be critical to identifying discrete management units and setting the most appropriate bycatch limits possible.

Providing estimates of bycatch mortality is essential for an ecosystem approach to fisheries management (Garcia and Cochrane, 2005; Bellido et al., 2011). The major obstacle to providing usable estimates is the many uncertainties in the data and assumptions required. This study makes every effort to make justifiable assumptions in response to the absence of detailed data, but it is nonetheless vital that estimates are considered within the context of this uncertainty. One potential source of overestimation could be that bycatch observer programmes with limited resources may focus their observer efforts on fisheries that would be expected to experience bycatch. Without a balanced distribution of observer effort across “low” and “high” risk fisheries, it is possible that applying simple means of bycatch rates from this subset of fisheries could

overestimate bycatch when applied to the wider fleet. Alternatively, by identifying the gear-specific and environmental drivers of bycatch, then using these to predict bycatch rates, as done in this study, we run a lower risk of overestimating bycatch in “low” risk fisheries.

Nonetheless, if we take a precautionary approach to fisheries management, we must consider that given the remaining data uncertainties, we cannot exclude the possibility that even our highest estimates may still underestimate total levels of bycatch. EU logbook data underestimate fishing effort from smaller vessels, as only vessels larger than 10 m in length are obligated to submit logbooks. Consequently, these data fail to capture the fishing effort of 75% of the currently registered Irish fishing vessels smaller than 10 m in length. This is arguably the most important data gap to consider as smaller vessels are typically restricted to inshore waters, with potentially high levels of bycatch, even in excess of larger fisheries (Peckham et al., 2007; Alfaro-Shigueto et al., 2011). Fishing activity by small-scale fisheries is poorly monitored globally, and unless this fishing effort can be accounted for, best estimates will most likely underestimate total levels of bycatch (Lewison et al., 2014).

A key assumption of observer programmes is that unobserved fishing activity can be inferred from the observed, and observed activities approximate to a random and representative subsample of all activities (Benoît and Allard, 2009). However, observer data may not accurately reflect fishing practice, as the presence of an observer may encourage fishers to change their behaviour while the observer is present (observer bias), and the distribution of observers may be determined more strongly by logistical constraints than experimental design (deployment bias; Benoît and Allard, 2009; Faunce and Barbeaux, 2011; Amande et al., 2012). Observer coverage may be further biased if a programme targets only fisheries with expected high rates of bycatch, or conversely if fisheries with the highest levels of bycatch are less agreeable to facilitating observers on board (Cotter and Pilling, 2007; Benoît and Allard, 2009). This study utilises a large amount of bycatch data self-reported by skippers. While it is encouraging that a potential observer effect was not detectable in the data, numerous examples exist in the literature to suggest that self-reporting may underestimate bycatch rates relative to observer-collected data (e.g. Allen et al., 2002; Bremner et al., 2009). As such, we should remember that the fishing activity observed in this study might not reflect the fishing activity of the wider fleet, and “observed” bycatch rates should be treated as minima.

Lastly, observers can only record what they can see and may entirely miss incidences of bycatch where animals fall out of nets or escape with serious injuries (Gilman et al., 2005; Benoît and Allard, 2009). As such, the estimates presented here, all of which were built on observer and self-reported data, may underrepresent this additional bycatch mortality. Peltier et al. (2016) analysed the distribution of stranded common dolphins (*Delphinus delphis*) along the Bay of Biscay and modelled the drift patterns of dolphin carcasses to estimate the total number of bycaught dolphins, inclusive of unobservable bycatch and robust to observer bias. Bycatch estimates inferred in this way suggested unsustainable levels of bycatch; thousands of dolphins per year, compared to the more sustainable estimate, based on limited observer data, of 550 dolphins caught annually.

In conclusion, we present a methodological framework for estimating bycatch mortality from limited observer data. We demonstrate that by using (1) sufficient observational data of bycatch events to identify reliable bycatch predictors, (2) fisheries logbook data to estimate fishing effort, and (3) expert knowledge to address key data gaps, plausible and informative bycatch estimates can be produced. While dedicated scientific observer programmes remain the most reliable means of estimating bycatch rates at sea, the global rarity of extensive, long-running programmes too often precludes the estimation of bycatch mortality by traditional means. Unable to quantify this anthropogenic pressure, it remains challenging to identify populations or species at risk. Therefore, it is vital that we develop complementary methodologies to maximise the value of the limited data that does exist, providing an important starting point for an ecosystems-based approach to fisheries management, bycatch mitigation, and addressing key data gaps in our understanding of fisheries bycatch worldwide.

## Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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