Estimating biomass, fishing mortality and 'total allowable discards' for surveyed non-target fish

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Abstract

Demersal fisheries targeting a few high-value species often catch and discard other 'non-target' species. It is difficult to quantify the impact of this incidental mortality when population biomass of a non-target species is unknown. We calculate biomass for 14 demersal fish species in ICES area VIIg (Celtic Sea) by applying species- and length-based catchability corrections to catch records from the Irish Groundfish Survey (IGFS). We then combine these biomass estimates with records of commercial discards (and landings for marketable non-target species) to calculate annual harvesting rates (*HR*) for each study species. Uncertainty is incorporated into estimates of both biomass and *HR*. Our survey-based *HR* estimates for cod and whiting compared well with *HR*-converted fishing mortality *(F)* estimates from analytical assessments for these two stocks. Of the non-target species tested, red gurnard (*Chelidonichthys cuculus*) recorded some annual *HR*s greater than those for cod or whiting; challenging 'Pope's Postulate' that *F* on non-target stocks in an assemblage will not exceed that on target stocks. We relate *HR* for each species to two corresponding Maximum Sustainable Yield (*MSY*) reference levels; six non-target species (including 3 ray species) show annual *HR*s $\geq HR_{MSY}$. This result suggests that it may not be possible to conserve vulnerable non-target species when *F* is coupled to that of target species. Based on biomass, *HR* and HR_{MSY}, we estimate 'total allowable catch' for each non-target species.

Introduction

Bottom trawl surveys of the Celtic Sea shelf demersal assemblage record over 100 fish species. Commercial fisheries target predominately 10 of the largest and most abundant (Marine Institute, 2013), but survey catch per unit of effort (CPUE) indicates that several other species of little market value have similar overall abundance to these. Such abundant non-target species are frequently caught in commercial fishing gear (Pope *et al*., 2000), typically being discarded dead or dying (Benoit *et al*., 2013). It has proved difficult to quantify the effect of fishing pressure on non-target species, although long-term survey abundance trends in North Sea populations do imply fishing impact (Jennings *et al*., 1999). Fishing exerts mortality at some rate (the instantaneous death rate due to fishing F) on each species. Realized F varies with behaviour and morphology, and can be expressed in terms of susceptibility and selectivity in the fishing gear, and the survival of fish after discarding (Zhou *et al*., 2010). Estimating F for key target species is typically undertaken using analytical stock assessments that are based on the age distribution of commercial catches. Such data are generally collected only for dominant target species and therefore such assessments are precluded as a means of estimating F for non-target species.

In the current absence of empirical estimates of non-target F, some authors (e.g., Le Quesne and Jennings, 2012) suggest applying a precautionary principle such as 'Pope's postulate' (Pope *et al*., 2000). Pope's postulate states that "fishing fleets generate a fishing mortality on non-target species which is less than or equal to that generated on the target species". It thus may provide upper limits on current mortality for ‘sensitive’ species in a fishery and represents a useful management principle for data poor situations (Jennings, 2013). This is a pragmatic approach, but may be too conservative in some fisheries and insufficiently precautionary in others. Only case-specific assessments can produce estimates of non-target mortality rate (Jennings, 2013). Nonetheless, accurate estimates of fishing mortality on non-target species are required to fulfill international commitments on biodiversity conservation (e.g., CBD 2010), and are central to the Ecosystem Approach to Fisheries Management (EAFM) (Garcia *et al*., 2003), which proposes to integrate fishing effects across exploited communities.

As an alternative to age-structured methods, Pope *et al*. (2000) proposed that the volume of catch (discards) in the fished component of a ‘local area’ could be extrapolated to derive an estimate of *F* for the whole of that local area. This method demands broad assumptions about catchability (the probability that an individual encountering fishing gear will be retained) and especially about spatial distribution of fish across the overall area. Where possible, a better approach may be to derive population biomass estimates from standardized fisheries-independent trawl surveys. The primary challenge here is that such surveys are not typically designed to estimate absolute fish population abundance, but to produce CPUE indices of relative abundance. Many aspects of fishing gear selection and fish behaviour will affect catchability, and differences in catchability mean that surveys provide ‘biased perceptions of the actual abundance of different species and size classes at a particular time and location’ (Fraser *et al*., 2007). Estimating absolute abundance from survey data requires a correction for catchability (ideally by species and length).

Survey catchability has been modelled (e.g., Harley and Myers, 2001), but Fraser *et al*. (2007) provide a list of empirically calculated length-based survey catchabilities for the most abundant NE Atlantic demersal fish species. Piet *et al*. (2009) developed a model that uses the catchability parameters from Fraser *et al*. (2007) to raise International Bottom Trawl Survey (IBTS) catches to spatial (ICES rectangle) estimates of biomass for the most abundant species. The model simulates fishing of these species according to known fishing intensity and commercial gear selectivity, to derive model estimates of catch and then *F*. We benefit from access to data from an Irish monitoring programme that records discards of both commercial species and all non-target fish species. We use species catchabilities from Fraser *et al*. (2007) to derive population biomass estimates from survey data, and combine these with discard and landings records to yield empirical estimates of the proportion of biomass removed annually by fishing (harvesting rate *HR*) for 14 fish species in the Celtic Sea. To test the approach, our survey-based estimates of *HR* for ‘Celtic Sea’ stocks of Atlantic cod (*Gadus morhua*) and Whiting (*Merlangius merlangus*) are compared with corresponding *HR*s converted from the *F* estimates produced by analytical stock assessments. *HR*s for non-target species are then compared to *HR*s for the assessed stocks to test Pope’s postulate.

Differences in life history mean that sustainable levels of *F* vary considerably among demersal fish species. It is useful to be able to compare *F* or *HR* with appropriate reference levels for ‘sustainable’ mortality. We present two estimates of sustainable *HR* for each tested species, based on published approaches, and compare these estimated reference levels with observed annual *HR* values. We then derive the ‘total allowable catch' (TAC) for each species that corresponds to a sustainable *HR* given estimated biomass.

Methods

Our study focuses on 15 International Council for the Exploration of the Seas (ICES) statistical rectangles in the Celtic Sea, with total area approx. 52,000 km^2 (Figure 1). This area (approximating to ICES area VIIg) was selected because it comprised the best overlap between available Irish Groundfish Survey (IGFS) data and Irish discard observer data, and captures much of the landings of the analytically assessed 'Celtic Sea cod' stock (ICES, 2012a) and the 'whiting in divisions VIIe-k' stock (ICES 2012b) (Figure 2). Each of these two stocks is assessed annually with an age-structured analytical assessment XSA (ICES, 2013a) to produce estimates of spawning stock biomass (*SSB*), and importantly for this study, total stock biomass (*TSB*) and *F* at age. These assessments are primarily based on commercial landings data from the relevant stock area, but the most recent biomass estimates are adjusted ('tuned') using fisheries-independent survey data from the IGFS. In addition to the assessed cod and whiting stocks, we selected 9 abundant non-target species from the IGFS, and also 3 exploited ray species of conservation interest. All selected species were deemed to be above the survey detection threshold, i.e., present in >5% of all hauls and recording abundance >5 individuals km^{-2} (Trenkel and Cotter, 2009).

Fish population biomass from survey data

The IGFS is a stratified random bottom-trawl survey that includes the Celtic Sea (Figure 1). This survey is operated in autumn (Q4) by the Irish Marine Institute using a GOV trawl fitted with a 20mm codend liner. Standard International Bottom Trawl Survey (IBTS) protocol is followed. In a given year, trawl samples (designed to be 30 min duration at 4 knots) are collected at 1-10 sites within each surveyed ICES rectangle. All fish captured in the IGFS are identified to species and measured (total length; l_{cm}). For cod and whiting, and the selected non-target species, we calculate the annual biomass $B_{s,y}$ of species *s* in the study area in year *y*, by summing over survey rectangles:

$$B_{s,y} = \sum_{k=1}^{K} B_{s,k,y} \qquad (1)$$

where K is the number of rectangles in the study area and

$$B_{s,k,y} = A_k \overline{D}_{s,k,y} \qquad (2)$$

where A_k is the sea surface area (km^2) of the *k*th rectangle and $\overline{D}_{s,k,y}$ is the mean biomass density (kg km^{-2}) of species *s* taken over J_k individual survey trawls in rectangle *k* and year *y*:

$$\overline{D}_{s,k,y} = \sum_{j=1}^{J_k} D_{s,j,k,y} / J_k, \qquad (3)$$

in which $D_{s,j,k,y}$ is the *expanded* biomass density of species *s* estimated from trawl *j*, in survey rectangle *k* and year *y*. Biomass of all species was expanded for survey trawl net catchability, as follows:

$$D_{s,j,k,y} = \sum_{i=1}^{I} (D_{i,s,j,k,y} / q_{i,s}) \qquad (4)$$

in which $D_{i,s,j,k,y}$ is the observed biomass density of the *i*th length class of species *s* from trawl *j*, in survey rectangle *k* and year *y*. The correction coefficient $q_{i,s}$

represents catchability and uses data from Fraser *et al*. (2007) resolved by species and length class.

$D_{s,j,k,y}$ for whiting only was calculated from doorspread (see equation 7) because this species is known to be herded into the path of the net by the sediment cloud stirred up by the otter doors and sweeps (e.g., Main and Sangster, 1981; Wardle, 1986). However, not all individual fish located between the doors are captured in the net. To account for this potential underestimate of biomass, for whiting only we used:

$$D_{s,j,k,y} = \frac{\sum_{i=1}^{I}(D_{i,s,j,k,y} / q_{i,s})}{h_s} \qquad (5)$$

where the coefficient h_s is a length and species dependent herding factor, defined from Piet *et al*. (2009) as:

$$h_{whiting} = \begin{cases} 0.30 \; if \; l_i < 29.5, \\ 0.75 \; if \; l_i \geq 29.5, \end{cases}$$

where h is the probability that an individual fish located between the trawl doors will be herded in between the trawl wings, l_i is the median length of length class i.

Piet *et al*. (2009) assumed minimal herding for flatfish and rays, while the benthic habit and weak swimming characteristics of gurnards (Floeter and Temming, 2005) and dragonets (Takita *et al*., 1983) also suggest very little herding, and so h was not applied for these species. However, some herding was assumed for the small roundfish Norway pout *Trisopterus esmarkii* and Poor cod *Trisopterus minutus*, meaning that wingspread biomass for these species (see equation 7) could be an overestimate. Observed survey biomass for these species was thus expanded as:

$$D_{s,j,k,y} = h((\sum_{i=1}^{I} \frac{D_{i,s,j,k,y}}{q_{i,s}})) \qquad (6)$$

where the coefficient h represents the probability that an individual of these two species that is swept by the trawl doorspread will enter the net: $h = Rp_n + (1 - R)p_d$, where p_n is the probability of entering the net if in front of the net wingspread and p_d is the probability is the probability of entering if in front of the doorspread but not wingspread, and R is the ratio of wingspread to doorspread. Using R = 1/3 and p_n = 1 and p_d = 0.3, h = 0.533.

The observed biomass density estimate is obtained from the survey trawl swept area

$$\text{as} = D_{i,s,j,k,y} \frac{w_{i,s,j,k,y}}{v_{j,k,y}\lambda_{j,k,y}} \qquad (7)$$

where $v_{j,k,y}$ is the length of the jth trawl in rectangle k and year y; $\lambda_{j,k,y}$ is that trawls wingspread (for all species except whiting, where it is that trawls doorspread – following Fraser *et al*., 2007), and $w_{i,s,j,k,y}$ is the total mass caught in that trawl in the ith length class of species s. This mass is estimated from a length-weight relationship for each species as:

$$w_{i,j,k,y} = N_{i,j,k,y}\alpha l_i^{\beta} \qquad (8)$$

where $N_{i,j,k,y}$ is the number of individuals of length class i, caught in trawl j in rectangle k in year y, such that l_i is the median length of length class i, and α and β are species-specific values taken from Celtic Sea data for cod and whiting and from North Sea IBTS survey data for non-target species, which have insufficient weight records in the IGFS.

Harvesting rate

Discard data

Discard data came from an Irish observer programme that serves the Data Collection Regulation (EC No. 1639/2001). Fishing trips are sampled at a rate proportional to métier activity, with sampling coverage of the Irish fleet being approximately 1% during the study period. Sampling trips are selected randomly, and so the distribution of fishing activity sampled is considered representative of the population as a whole (Marine Institute, 2011). Discard data were extracted by species, gear, quarter and year. If a sampled fishing trip included hauls outside study rectangles, then the proportion of the fishing effort inside the area was used. Discard weight was raised to Irish fleet level by dividing it by the proportion of total Irish effort covered by discard sampling.

For cod and whiting, each of the Irish landings and discard values were raised to an international estimate by dividing by the annual proportion of the total catch landed by Irish vessels only in the stock area. This was considered a proxy for the proportion of total (international) effort (and hence discarding) accounted for by Irish vessels. For non-target species, discard records were raised according to the proportion (range = 51-58% in the study period) of annual international effort by mobile gears (kilowatt hours = vessel engine power multiplied by time) in the study area recorded by Irish vessels (STECF, 2013). For years where effort for a given nation was not reported to STECF, the mean annual value for that nation was applied.

Calculation

For each species s in year y, a first HR range for the study area was then estimated as:

$$HR^{B}_{s,y} = \frac{C_{s,y}}{C_{s,y} + (B_{s,y} \pm 1SD)} \qquad (9)$$

where $C_{s,y}$ is the total catch (landings and discards) of species *s* in year *y*, and $B_{s,y}$ is expanded biomass (see above). $HR^{B}_{s,y}$ includes uncertainty in the estimate of survey biomass $B_{s,y}$. To include (in addition) uncertainty in the estimation by the Irish observer scheme of discard rates, a second *HR* range $HR^{B\_dis}_{s,y}$ for the study area was then estimated as $HR^{B}_{s,y}$, but where $C_{s,y}$ = landings + discards * (1 ± 20%); Rochet *et al.* (2002) estimated overall CV in discard estimates for the French fleet in the Celtic Sea as 20%. $HR^{B\_dis}_{s,y}$ may be considered most precautionary because the upper bound accounts for a scenario in which species biomass was over-estimated by the survey, while the observer scheme underestimated discards (by 20%, Rochet *et al.*, 2002).

Validation

As a validation exercise, $B_{s,y}$ and $HR^{B\_dis}_{s,y}$ for cod and whiting were compared to *TSB* and *F* values from respective stock assessments, where assessment *F*s were converted to *HR* (*HR* = 1-exp(-*F*)) for direct comparison. $B_{s,y}$ were smaller than assessment *TSB* for whiting, possibly reflecting the fact that the study covers only part of the assessed stock area. $HR^{B\_dis}_{whiting,y}$ differed by an average of 0.06 (12%) from mean annual assessment *HR* estimates (Table 1). $B_{s,y}$ for cod were closer to assessment *TSB*, while $HR^{B\_dis}_{cod,y}$ differed by an average of 0.05 (16%) from assessment *HR* with no years showing large discrepancy (Table 1). Similarity between survey-based and assessment *HR*s for both cod and whiting suggest that our approach generates estimates that can be reasonably applied to non-target species.

Precautionary reference levels

To gain some insight into the likely ecological significance of observed *HR* for non-target species, estimates for each species were compared to two sets of candidate reference levels:

1. From a meta-analysis of 245 fish species, Zhou *et al.* (2012) suggested that F_{MSY} could be estimated as 0.87M for teleosts and 0.41M for chondrichthyans. For most species, we calculated HR_{MSY} reference points according to Zhou *et al*. (2012), but the value for witch (*Glyptocephalus cynoglossus*) was derived from ICES advice (ICES 2013b). Estimates of natural mortality M are from Gallagher *et al*. (2004) (rays) and FishBase (most other species). For red gurnard (*Chelidonichthys cuculus*), we used: $\ln(M) = 1.44 - 0.982 \ln(t_m)$ (Hewitt and Hoenig, 2005), where t_m is the maximum observed age reported in FishBase. An *HR* limit reference point (HR_{lim}) was also calculated by adding one standard deviation to the $F_{msy}:M$ ratios of Zhou *et al.* (2012), i.e., $(0.41+0.09)M$ for elasmobranchs and $(0.87+0.05)M$ for teleosts respectively.

2. For many of the demersal species in the Celtic Sea, Le Quesne and Jennings (2012) provide estimates of F_{40} (the F that reduces *SSB*-per-recruit to 40% of that in the absence of fishing). We used F_{40} estimates from Table S1 in Le Quesne and Jennings (2012) to derive a list of HR_{40} estimates.

A precautionary mean total allowable catch (TAC) or total allowable discards (TAD) for each species was estimated by averaging the lower bounds of $B_{s,y}$ and then multiplying by *HR*.

Survey trends

Time series of CPUE (kg km^{-2}) in the IGFS and also the UK West Coast Groundfish Survey (WCGFS) were used as a visual descriptor of changes in species relative abundance that might partly reflect *F*. The WCGFS was discontinued in 2004 but has some data for the study region extending to 1986 and is assumed to describe fish abundance at an earlier stage of exploitation history (Shephard *et al*., 2011)

Results

Survey biomass

$B_{s,y}$ varied among species as expected, with low estimates for 'k-strategy' (low *M*) species such as the rays and higher estimates for 'r-strategy' (high *M*) species such as Norway pout and Poor cod (Table 2).

Harvesting rate

Non-target species showed a wide range of $HR_{s,y}^{B\_dis}$, with some missing values due to incomplete discard data (Table 3). Two roundfish (red gurnard and grey gurnard

Eutrigla gurnardus) recorded some $HR_{s,y}^{B\_dis}$ in the same range as estimated for cod, with some values for red gurnard exceeding the maxima recorded for assessed stocks of both cod and whiting (Table 3).

Reference levels

Of the teleost species, dab (*Limanda limanda*), red gurnard and grey gurnard showed at least one annual *HR* greater than one of the proposed *HR* reference levels, while some *HR*s for red gurnard were greater than both reference levels. All tested elasmobranchs recorded at least one $HR_{s,y}^{B\_dis}$ with upper bounds ≥ one of the tested reference levels. Spotted ray *Raja montagui*, which has the lowest l_{95} and greatest M among the elasmobranchs, and might thus be expected to be the most resilient, recorded lowest $HR_{s,y}^{B\_dis}$ in this group (Table 3). Estimated TACs varied strongly among species, with red gurnard TAC estimated at only 56t while Norway pout TAC was >20,000t.

Discussion

The ecosystem approach to fisheries requires information on population size and fishing mortality of non-target species within exploited fish assemblages. We use catchability-expanded survey records and catch (landings + discards) data to estimate biomass and *HR* for two assessed and several non-target demersal fish species. $HR_{cod,y}^{B\_dis}$ and $HR_{whiting,y}^{B\_dis}$ are close to analytical stock assessment estimates for these species, suggesting that our method may produce reasonable estimates for the non-target species. We aim to present a pragmatic method that can yield information on the state of surveyed but unassessed stocks. This requires certain assumptions that we consider with reference to some of the fisheries survey criteria presented by Trenkel and Cotter (2009).

Survey data

Study area vs. fish ‘stock’ distributions

In order to validate our approach, we needed to make a direct comparison between survey-based estimates of biomass and mortality for given stocks, and corresponding estimates from analytical assessments. We identified an area of the Celtic Sea (Figure 2) corresponding to the core area (greatest landings: STECF, 2013) of assessed cod and whiting stocks, and for which survey and catch data were available. Comparison of survey-based and assessment biomass estimates suggested that we probably captured much of the cod population but only a component of the whiting population – this was anticipated from the known range of these stocks. Comparison of survey and assessment *HR*s suggest that the study area allowed quite accurate estimates of fishing mortality for both cod and whiting. For simplicity in continuing the current ‘worked example’ of our approach, we used the same study area for non-target ‘stocks’. Our results thus represent biomass and *HR* estimates for that component of each non-target stock within our study area in a given year. When applying the method at, e.g., MSFD Subregion scale, there may be ecological justification in defining species-specific ranges from survey CPUE data. A potential difficulty is that these ranges are likely to cross national survey and discard sampling boundaries and so synthesis of disparate data series may be required.

In our biomass estimates $B_{s,y}$, we stratify the survey data by ICES rectangle, as in the analytical stock assessments. This simple approach incorporates spatial heterogeneity in population density without demanding complex evaluation of density patterns and temporal changes in these patterns. An alternative approach might be to stratify the survey data in a more dynamic way, based on observed spatial density of the fish within and among years.

Catchability

We applied $q_{i,s}$ from Fraser *et al*. (2007) to raise survey CPUE by species and length class to total biomass within the study area. Fraser *et al*. (2007) calculated catchability parameters for assessed species by quantifying the ratios between catch at length in North Sea survey data and numbers at age (converted to length) in analytical stock assessments for the same stocks. For whiting, we follow Piet *et al*. (2009) in using the relevant Fraser *et al*. (2007) catchability parameters with an additional trawl door herding factor h. Fraser *et al*. (2007) estimated catchabilities for non-target species by comparing catch between beam trawl and GOV hauls undertaken in ‘approximately the same time and place’. This method assumes that $q_{i,s}$ in the beam trawl are fairly stable relative to the GOV, and close to 1. We suggest that this is a reasonable assumption for flatfish and rays (Piet *et al*., 2009), gurnards (Floeter and Temming, 2005) and dragonets (Takita *et al*., 2003), which are unlikely to have a very active or sustained escape response to fishing gear. Many other small non-target demersal species also show benthic behaviour, and probably low herding response. For Poor cod and Norway pout, we assume more active herding and escape behaviour and apply a similar size-based herding factor *h* (Piet *et al*., 2009).

Fraser *et al*. (2007) derived $q_{i,s}$ for the standard GOV trawl used in the Q3 North Sea IBTS. However, the same parameters have been widely applied to the separate Q1 North Sea IBTS (e.g., Collie *et al*., 2013; Heath *et al*., 2013) and result in very similar species biomass trends between surveys (Greenstreet *et al*., In Review). We thus assume that Fraser *et al*. (2007) can be used as a first-order approach to raise catch in similar survey GOV trawls in different areas/seasons. This assumption is upheld by similarity between survey-based and assessment *HR* estimates for cod and whiting in our study area.

Catchability for given species can vary with size, and this is already incorporated into the Fraser *et al*. (2007) catchability parameters, which typically comprise a catchability curve with length. Catchability $q_{i,s}$ may also vary with season due to changes in spatial distribution of stock components (size classes) and in their availability to the survey gear. Without species-level information on such effects in the study area, it is difficult to evaluate whether the Q4 IGFS under-samples certain population components. Trenkel and Cotter (2009) suggest that if species CPUE trends are similar between different surveys in a given area, then $q_{i,s}$ may be similar between surveys. This is the case between the Q1 and Q3 IBTS in the North Sea (Greenstreet *et al*., In Review).

Harvesting rate

Our estimates of *HR* depend on survey, observer and commercial landings data. We have to assume that the landings data are accurate, and so the two major potential sources of uncertainty in our approach are the survey-based estimates of species biomass (which include uncertainty in catchability $q_{i,s}$) and the observer estimates of discarding. We attempt to address both these sources of uncertainty, although uncertainty in q is not explicitly addressed. The survey follows a robust scientific protocol in which samples are located randomly within ICES rectangles. By sub-sampling (random 10% deletion) from the pool of survey hauls in each year, we address the potential effect of haul location and inter-haul variation on annual species biomass estimates, while maintaining spatial stratification by ICES rectangle. The discard observer scheme is more *ad hoc*, with overall fleet coverage stratified by métier, but actual sampling depending partly on access to fishing vessels, weather, seasonality in fishing patterns etc. We thus take a more flexible approach to uncertainty in the discard data, incorporating into *HR* estimates the effect of ±20% error (Rochet *et al.*, 2002) in the estimate of annual discards of a given species. As expected, the range in annual *HR* for each species tends to be wider when both sources of uncertainty are included (Table 3). The most precautionary application of *HR* for management would be to use the upper estimate, and this is the approach we take when proposing TACs.

Validation

We 'validate' our method by comparing $B_{s,y}$ and $HR_{s,y}^{B-dis}$ for cod and whiting to estimates from corresponding analytical stock assessments. We note that while the stock assessments are predominately based on age-structured landings data, they use the IGFS to tune (adjust) biomass series. This means that the two approaches to estimating biomass and mortality are not completely independent. Survey CPUE generally has most influence on terminal estimates of fishing mortality and SSB in the assessments, and so the most recent 3 (approx.) years, assessment *F* and SSB are most strongly influenced by the survey. For earlier years, the assessments tend to converge on the catch data according to the traditional convergence properties of VPA-based assessments. On this basis, the IGFS will not strongly influence cod and whiting assessment outputs (which we took from 2013 ICES WG reports) for the earlier years used in our study (2008-2010), but there will be some non-independence for the final year (2011). Even for 2011, we suggest that possible non-independence does not matter, since our intention is to provide survey-based estimates that can serve, for unassessed populations, the same purpose as age-structured assessments. Correspondingly, we find that we can use survey data directly and derive results that are similar to those produced by those assessments. This result suggests that for other species, survey data may also be sufficient to produce estimates that would be similar to those from age-structured assessment if such assessments were conducted.

Reference levels

Fishing mortality reference points provide insight into the likely ecological significance of observed *HR*s for non-target species. Slower growing, later maturing and less fecund species (e.g., many elasmobranchs) are expected to experience greater negative impact for a given *HR* (e.g., Jennings *et al*., 1999; Garcia *et al*., 2008; Le Quesne and Jennings, 2012). High levels of *F* for the ray species probably reflect strong susceptibility to capture in mobile fishing gears (Cedrola *et al*., 2005). Patches of low fishing effort in the Celtic Sea can act as *de facto* refuges for some elasmobranchs (Shephard *et al*., 2012), but our *HR* estimates suggest that additional conservation measures are probably necessary to reach precautionary reference levels for *F* in these species.

The high *HR*s for both gurnard species are notable (Table 3). Rochet *et al*. (2002) found that red and grey gurnards (and boarfish *Capros aper*) were the most common non-target species discarded in the Celtic Sea; discarding of the highly perishable red gurnard increased with trip length. This high level of discarding for red gurnard, compared to our small abundance estimates for this species (Table 2), helps explain observed high *HR*. Gurnards may be particularly susceptible to capture in trawl gear because of morphology (broad square head and long spiny fins) and behaviour (sit-and-wait or stalk predation with little muscle capacity for sustained escape swimming (Floeter and Temming, 2005).

We estimate a mean TAC (2008-2011) for each study non-target species. These TACs currently refer only to the study area (approximately ICES VIIg) but the method presented may have considerable utility in the context of the upcoming EU landing obligation (EU COM/2013/0889 final - 2013/0436). Understanding fishing mortality of vulnerable species and being able to relate this to a specific volume of landings will enable managers to make informed decisions about conservation of these species. A significant problem in this regard is that *F* of vulnerable species is closely tied to that of target species, and reducing target *F* to a level consistent with conservation of the most vulnerable species may restrict fishing activities to a socially unacceptable degree. A potential solution might be the use of spatial management to de-couple target and non-target F, possibly with particular application to elasmobranchs (e.g., Shephard *et al.*, 2012).

Survey trends

The IGFS time series showed little trend in standardized abundances (haul density) of non-target species. The WCGFS recorded visible declines in several species that we find to have high *HR*s, including red gurnard and cuckoo ray (Figure 3). These data suggest that non-target species experiencing high fishing mortality can show decadal population declines. Spotted ray, for which we observed $HR > HR_{msy}$ in 2008 only, also demonstrated a declining trend in the IGFS.

Conclusions

HR can be estimated for surveyed non-target species, and related to precautionary reference levels. At least two non-target species discarded in the Celtic Sea experience *HR*s greater than those recorded for assessed cod and whiting stocks, meaning that Pope's postulate does not hold in the study area. Some vulnerable non-target species are 'exploited' at rates greater than precautionary reference levels, and some of those show declines in survey CPUE. It is possible to calculate the catch (TAC or TAD) associated with precautionary reference levels and thus monitor exploitation of vulnerable species.

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References

Branch, T. A., Jensen, O. P., Ricard, D., Ye, Y., and Hilborn, R. 2011. Contrasting global trends in marine fishery status obtained from catches and from stock assessments. Conservation Biology, 25(4): 777-786.

Benoît, H. P., Plante, S., Kroiz, M., and Hurlbut, T. 2013. A comparative analysis of marine fish species susceptibilities to discard mortality: effects of environmental factors, individual traits, and phylogeny. ICES Journal of Marine Science, 70(1): 99-113.

CBD. 2010. X/2. The Strategic Plan for Biodiversity 2011–2020 and the Aichi Biodiversity Targets. Convention on Biological Diversity, Nagoya, Japan. 18–29 October 2010. UNEP/CBD/COP/DEC/X/2. 13 pp.

Cedrola, P. V., González, A. M., and Pettovello, A. D. 2005. Bycatch of skates (Elasmobranchii: Arhynchobatidae, Rajidae) in the Patagonian red shrimp fishery. Fisheries Research, 71(2): 141-150.

Collie, J., Rochet, M-J., and Bell, R. 2013. Rebuilding fish communities: the ghost of fisheries past and the virtue of patience. Ecological Applications 23: 374–391.

Floeter, J., and Temming, A. 2005. Analysis of prey size preference of North Sea whiting, saithe, and grey gurnard. ICES Journal of Marine Science, 62(5): 897-907.

Fraser, H. M., Greenstreet, S. P., and Piet, G. J. 2007. Taking account of catchability in groundfish survey trawls: implications for estimating demersal fish biomass. ICES Journal of Marine Science, 64(9): 1800-1819.

Garcia, S. M. (Ed.). 2003. *The ecosystem approach to fisheries: issues, terminology, principles, institutional foundations, implementation and outlook* (No. 443). Food & Agriculture Organization of the United Nations.

García, V. B., Lucifora, L. O., and Myers, R. A. 2008. The importance of habitat and life history to extinction risk in sharks, skates, rays and chimaeras. Proceedings of the Royal Society London B**,** 275(1630): 83-89.

Gallagher, M. J., Nolan, C. P., and Jeal, F. 2004. An investigation of the Irish ray fishery in ICES divisions VIIa and VIIg. Journal of Northwest Atlantic Fisheries Science 35: 1-13.

Greenstreet, S. P. R., Shephard, S., Guijarro-Garcia, E., Oesterwind, D., and Reid, D. G. In Review. Deriving regional-scale assessments of fish community status from disparate subregional-scale data sets. ICES Journal of Marine Science.

Harley, S. J., and Myers, R. A. 2001. Hierarchical Bayesian models of length-specific catchability of research trawl surveys. Canadian Journal of Fisheries and Aquatic Sciences, 58(8): 1569-1584.

Heath, M. R., Culling, M. A., Crozier, W. W., Fox, C. J., Gurney, W. S., Hutchinson, W. F., ... and Carvalho, G. R. 2013. Combination of genetics and spatial modelling highlights the sensitivity of cod (*Gadus morhua*) population diversity in the North Sea to distributions of fishing. ICES Journal of Marine Science: fst185.

Hewitt, D. A., and Hoenig, J. M. 2005. Comparison of two approaches for estimating natural mortality based on longevity. Fishery Bulletin 103(2): 433-437.

ICES. 2012a. Celtic Sea and West of Scotland Cod in Divisions VIIe–k (Celtic Sea cod). ICES Advice document. (doi: http://www.ices.dk/sites/pub/Publication%20Reports/Advice/2012/2012/cod-7e-k.pdf)

ICES. 2012b. Celtic Sea and West of Scotland Whiting in Divisions VIIe–k. ICES Advice document. (doi: http://www.ices.dk/sites/pub/Publication%20Reports/Advice/2012/2012/whg-7e-k.pdf)

ICES. 2013a. Report of the Working Group for Celtic Seas Ecoregion (WGCSE), 8–17 May 2013, Copenhagen, Denmark. ICES CM 2013/ACOM:12. 1986 pp. (doi:

http://www.ices.dk/sites/pub/Publication%20Reports/Expert%20Group%20Report/acom/2013/WGCSE/wgcse_2013.pdf)

ICES. 2013b. Witch in Subarea IV and Divisions IIIa and VIId. ICES Advice Document. (doi: http://www.ices.dk/sites/pub/Publication%20Reports/Advice/2013/2013/wit-nsea.pdf)

Jennings, S., Greenstreet, S.,P.,R., and Reynolds, J.,D. 1999. Structural change in an exploited fish community: a consequence of differential fishing effects on species with contrasting life histories. Journal of Animal Ecology, 68: 617–627.

Le Quesne, W. J., and Jennings, S. 2012. Predicting species vulnerability with minimal data to support rapid risk assessment of fishing impacts on biodiversity. Journal of Applied Ecology, 49: 20-28.

Main, J., and Sangster, G. I. 1981. A study of the sand clouds produced by trawl boards and their possible effect on fish capture. Department of agriculture and fisheries for Scotland. 20 pp.

Marine Institute. 2011. Atlas of Demersal Discarding, Scientific Observations and Potential Solutions, Marine Institute, Bord Iascaigh Mhara, September 2011. ISBN 978-1-902895-50-5. 82 pp.

Marine Institute. 2013. The Stock Book: Report to the Minister for Agriculture, Food and the Marine. Annual 2013 Review of Fish Stocks with Management Advice for 2014. Irish Marine Institute. 534 pp.

Patterson, K. R., and Melvin, G. D. 1996. Integrated catch at age analysis version 1.2. Scottish Office Agriculture, Environment and Fisheries Department. 60 pp.

Piet, G. J., Van Hal, R., and Greenstreet, S. P. R. 2009. Modelling the direct impact of bottom trawling on the North Sea fish community to derive estimates of fishing mortality for non-target fish species. ICES Journal of Marine Science, 66(9): 1985-1998.

Pope, J. G., Macdonald, D. S., Daan, N., Reynolds, J. D., and Jennings, S. 2000. Gauging the impact of fishing mortality on non-target species. ICES Journal of Marine Science, 57: 689-696.

Rochet, M. J., Péronnet, I., and Trenkel, V. M. 2002. An analysis of discards from the French trawler fleet in the Celtic Sea. ICES Journal of Marine Science, 59(3): 538-552.

Shephard, S., Greenstreet, S. P. R., and Reid D. G. 2011. Interpreting the Large Fish Indicator for the Celtic Sea. ICES Journal of Marine Science, 68: 1963-1972.

Shephard, S., Gerritsen, H. D., Kaiser, M. J., and Reid, D. G. 2012. Spatial heterogeneity in fishing creates *de facto* refugia for endangered Celtic Sea elasmobranchs. PLOS ONE. doi:10.1371/journal.pone.0049307.

STECF. 2013. Scientific, Technical and Economic Committee for Fisheries. Doi: http://stecf.jrc.ec.europa.eu/web/stecf/ewg1313. Appendix 05. Effective effort by rectangle by country.

Takita, T., Iwamoto, T., Kai, S., and Sogabe, I. 1983. Maturation and spawning of the dragonet, Callionymus enneactis, in an aquarium. Japanese Journal of Ichthyology, 30(3), 221-226.

Trenkel, V. M., and Cotter, J. 2009. Choosing survey time series for populations as part of an ecosystem approach to fishery management. Aquatic Living Resources, 22(02): 121-126.

Wardle, C. S. 1986. Fish behaviour and fishing gear. *In* The behaviour of teleost fishes. (pp. 463-495). Springer US.

Zhou, S., Smith, A. D., Punt, A. E., Richardson, A. J., Gibbs, M., Fulton, E. A., Pascoe, S., *et al*. 2010. Ecosystem-based fisheries management requires a change to the selective fishing philosophy. Proceedings of the National Academy of Science, 107(21): 9485-9489.

Zhou, S., Yin, S., Thorson, J. T., Smith, A. D., and Fuller, M. 2012. Linking fishing mortality reference points to life history traits: an empirical study. Canadian Journal of Fisheries and Aquatic Sciences 69(8): 1292-1301.

Tables

Table 1. Survey-based ($B_{s,y}$) and assessment (*TSB*) estimates of total biomass and harvesting rate for cod and whiting. $HR^{B}_{s,y}$ is a range incorporating uncertainty in survey biomass estimates ($HR_{s,y}$ at $B_{s,y}$ ± 1SD). $HR^{B\_dis}_{s,y}$ is a range incorporating uncertainty in estimates of both survey biomass $B_{s,y}$ and discards. Assessment HR = 1-exp (-F), using ages 2-5 for cod and 2-7 for whiting (ICES, 2013). International landings and discard estimates are also shown. All weights are tonnes.

Table 2. Survey-based (IGFS) estimates of total stock biomass in tonnes ($B_{s,y}$ ± 1SD), 95th percentile of length (l_{95}, cm) from the IGFS data and natural mortality (M) from Fishbase for non-target demersal fish species in the study area of the Celtic Sea.

Table 3. Harvesting rate estimates for non-target demersal fish in the study area of the Celtic Sea. $HR^{B}_{s,y}$ is a range incorporating uncertainty in survey biomass estimates ($HR_{s,y}$ at $B_{s,y}$±1SD). $HR^{B\_dis}_{s,y}$ is a range incorporating uncertainty in survey biomass and in discard estimates. Species HR_{MSY} (converted from F_{MSY}) is calculated from M (Zhou *et al*., 2012), except for witch, where F_{MSY} is from ICES (2013a). Species HR_{lim} accounts for the SD around the F_{msy}:M ratio in Zhou *et al*. (2012). HR_{40} is from Table

S1 in Le Quesne and Jennings (2012). *HR* estimates that are ≥ a given *HR* reference level are highlighted in red and < in green. Mean TAC (*t*) is the mean of the observed lower bounds of estimated annual biomass for each species multiplied by the more conservative (lower) *HR* reference level for each species.

Table 1.

	Whiting				Atlantic cod			
	2008	**2009**	**2010**	**2011**	**2008**	**2009**	**2010**	**2011**
International landings	9093	6382	8563	9639	3639	3263	3229	4737
International discards	3479	7569	4184	3188	1487	1351	1833	7541
$B_{s,y}$ ±1SD in study area	8385 ±871	17151 ±905	12761 ±635	19674 ±1198	5928 ±399	3638 ±386	14113 ±1016	27293 ±2462
Assessment TSB	29151	44568	61056	61623	9216	9781	23145	23358
HR^{B}	0.58-0.63	0.44-0.46	0.49-0.51	0.38-0.41	0.45-0.48	0.53-0.59	0.25-0.28	0.29-0.33
HR^{B_dis}	0.57-0.63	0.42-0.48	0.48-0.52	0.37-0.42	0.44-0.48	0.53-0.59	0.24-0.29	0.28-0.34
Assessment *HR*	0.51	0.46	0.44	0.32	0.51	0.52	0.38	0.31

Table 2.

				2008	2009	2010	2011
Common name	Latin name	L_{95}	M	B	B	B	B
Lesser spotted dogfish	*Scyliorhinus canicula*	67	0.37	32272 ±8618	25771 ±1326	42734 ±2927	46734 ±4996
Spotted Ray	*Raja montagui*	61	0.41	2832 ±336	7714 ±630	13652 ±666	9728 ±1195
Thornback ray	*Raja clavata*	78	0.20	NA	3709 ±320	6136 ±659	2113 ±234
Cuckoo ray	*Leucoraja naevus*	63	0.37	4506 ±653	2766 ±326	NA	5004 ±453
Witch	*G. cynoglossus*	40	0.18	7070 ±713	31663 ±4299	6920 ±235	12951 ±391
Dab	*Limanda limanda*	26	0.26	1040 ±147	2653 ±270	2084 ±140	4179 ±508
Long rough dab	*H. platessoides*	22	0.32	1299 ±89	1208 ±80	1610 ±66	1539 ±78
Poor cod	*Trisopterus minutus*	21	1.00	83570 ±6797	79816 ±2959	48918 ±2665	39395 ±2157
Grey gurnard	*Eutrigla gurnardus*	31	0.29	3213 ±279	3317 ±328	10261 ±756	13015 ±1144
Red gurnard	*C. cuculus*	32	0.20	280 ±36	385 ±38	484 ±24	398 ±39
Dragonets	*Callionymus spp.*	25	0.91	1041 ±135	1338 ±137	2113 ±79	NA
Norway pout	*Trisopterus esmarkii*	21	1.54	16855 ± 1058	43256 ±2138	75228 ±6294	104805 ±18867

Table 3.

		2008		2009		2010		2011		HR_{msy}	HR_{lim}	HR_{40}	Mean
Common name	Latin name	HR^{B}	HR^{B_dis}	HR^{B}	HR^{B_dis}	HR^{B}	HR^{B_dis}	HR^{B}	HR^{B_dis}	Zhou 2012	Zhou 2012	Le Quesne 2012	TAC
Lesser spot dogfish	*Scyliorhinus canicula*	0.13-0.21	0.12-0.22	0.15-0.16	0.14-0.17	0.13-0.15	0.12-0.16	0.11-0.13	0.11-0.14	**0.14**	**0.17**	**0.11**	**2398**
Spotted Ray	*Raja montagui*	0.07-0.11	0.07-0.11	0.02-0.02	0.01-0.02	0.01-0.01	0.01-0.01	0.05-0.06	0.05-0.06	**0.15**	**0.18**	**0.10**	**777**
Thornback ray	*Raja clavata*	NA	NA	0.04-0.04	0.03-0.04	0.05-0.06	0.04-0.07	0.13-0.16	0.13-0.16	**0.08**	**0.09**	**0.09**	**287**
Cuckoo ray	*Leucoraja naevus*	0.11-0.15	0.10-0.17	0.15-0.18	0.13-0.20	NA	NA	0.10-0.11	0.09-0.13	**0.14**	**0.17**	**0.11**	**398**
Witch	*G. cynoglossus*	0.08-0.10	0.08-0.11	0.01-0.01	0.01-0.01	0.03-0.03	0.03-0.04	0.03-0.03	0.02-0.04	**0.16**	**0.17**	**0.24**	**2119**
Dab	*Limanda limanda*	0.19-0.24	0.17-0.27	0.11-0.13	0.09-0.14	0.13-0.14	0.11-0.16	0.04-0.05	0.03-0.06	**0.20**	**0.21**	**0.24**	**445**
Long rough dab	*H. platessoides*	0.12-0.12	0.09-0.14	0.09-0.10	0.07-0.12	0.08-0.09	0.06-0.10	0.03-0.03	0.02-0.04	**0.24**	**0.25**	**0.29**	**321**
Poor cod	*Trisopterus minutus*	0.01-0.01	0.01-0.01	0.01-0.01	0.01-0.01	0.01-0.01	0.01-0.01	0.01-0.01	0.01-0.01	**0.58**	**0.60**	**0.27**	**18789**
Grey gurnard	*Eutrigla gurnardus*	0.23-0.25	0.18-0.29	0.29-0.34	0.26-0.36	0.06-0.07	0.05-0.08	0.03-0.04	0.03-0.05	**0.22**	**0.23**	**0.27**	**1501**
Red gurnard	*C. cuculus*	0.54-0.60	0.48-0.64	0.54-0.60	0.48-0.63	0.26-0.28	0.22-0.31	0.32-0.36	0.27-0.40	**0.16**	**0.17**	**0.23**	**56**
Dragonets	*Callionymus spp.*	0.01-0.01	0.01-0.01	0.01-0.01	0.01-0.01	0.01-0.01	0.01-0.01	NA	NA	**0.55**	**0.57**	**0.32**	**442**
Norway pout	*Trisopterus esmarkii*	0.01-0.01	0.01-0.01	0.01-0.01	0.01-0.01	0.01-0.01	0.01-0.01	0.01-0.01	0.01-0.01	**0.74**	**0.76**	**0.37**	**22976**

Figures

Figure 1. Map of the Celtic Sea, showing ICES statistical rectangles included in the study.

Figure 2. Spatial distribution of Celtic Sea landings from the cod (left panel) and whiting (right panel) stocks (STECF, 2013) included in the study.

Figure 3. Standardized abundance (mean haul density, kg km^{-2}) for cod, whiting and 12 non-target species in the UK Celtic Sea West Coast Groundfish Survey (WCGFS).

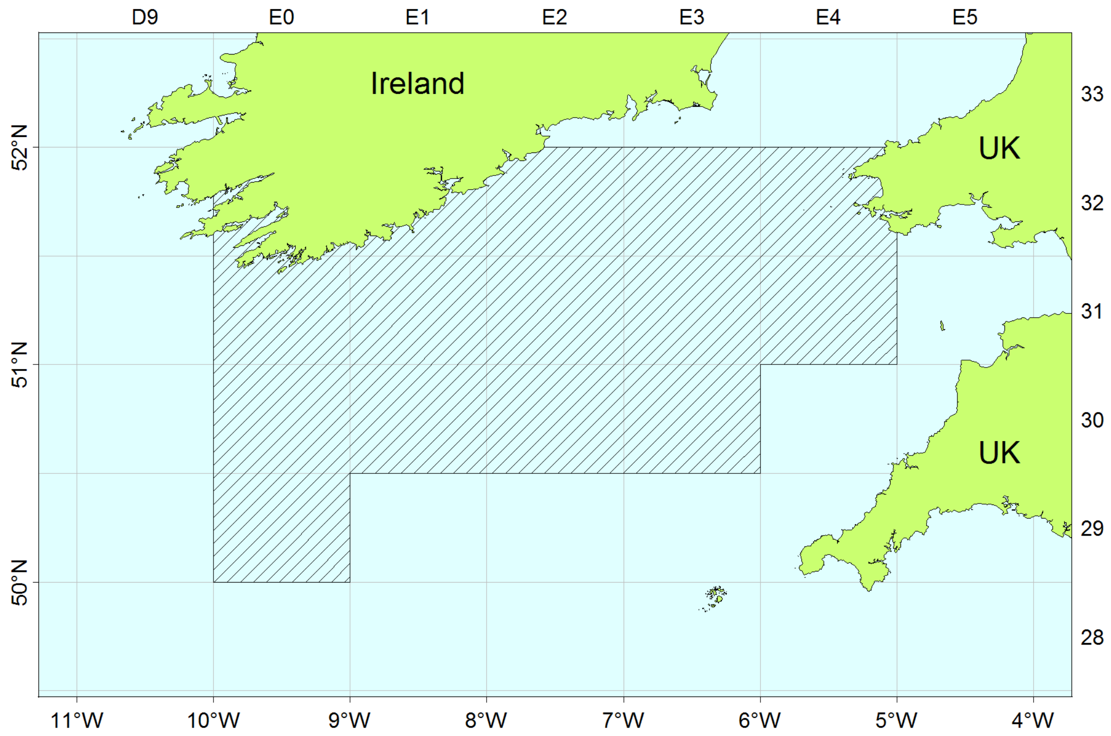


Figure 1.

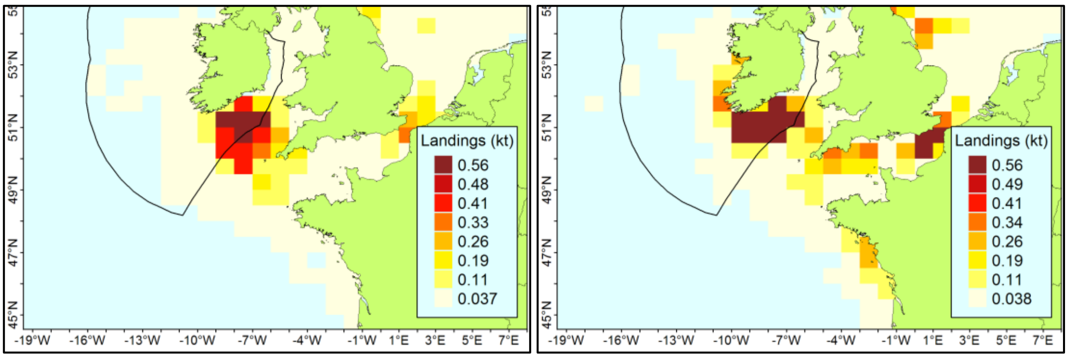
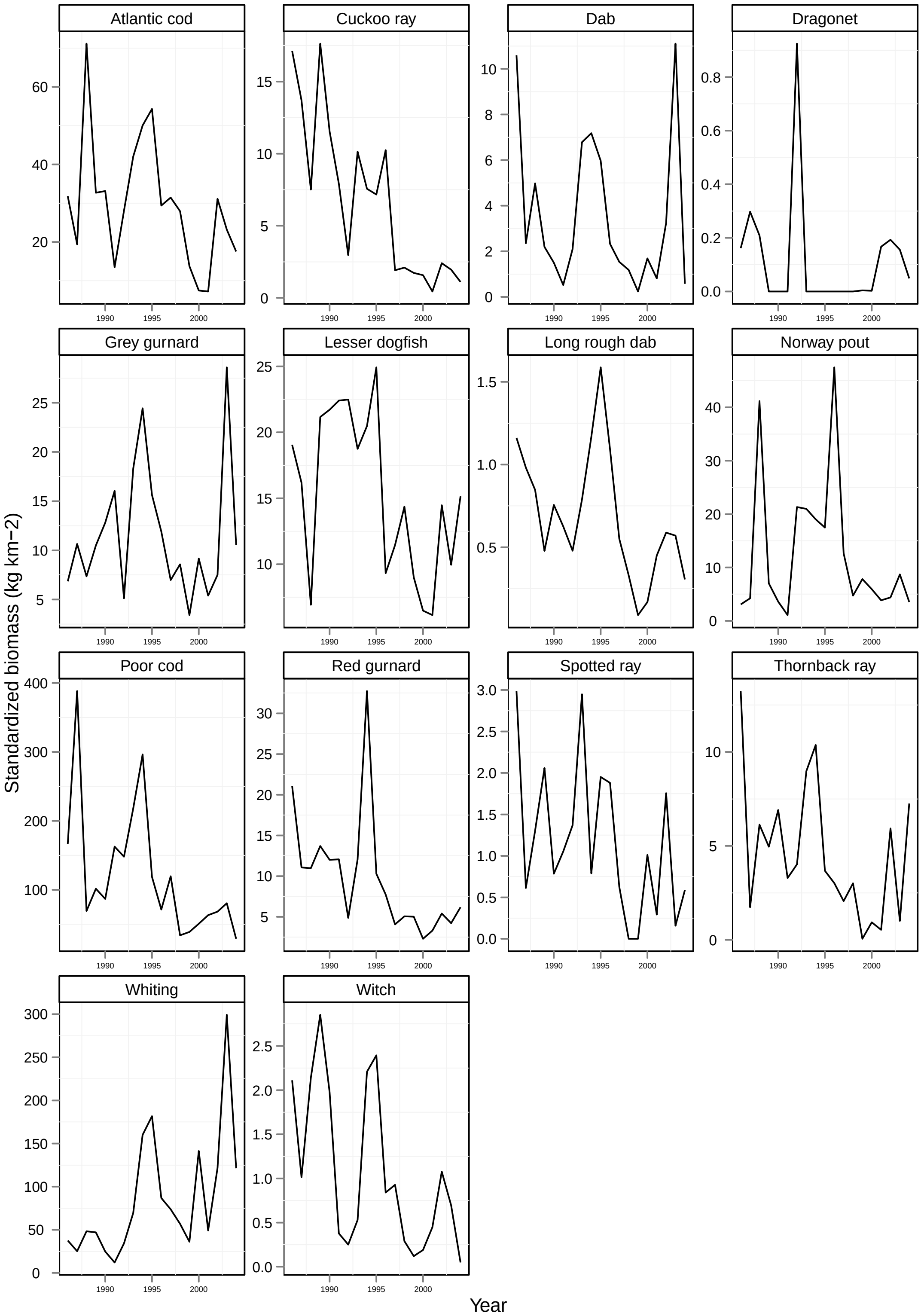

Landings (kt)
0.56
0.48
0.41
0.33
0.26
0.19
0.11
0.037
Landings (kt)
0.56
0.49
0.41
0.34
0.26
0.19
0.11
0.038
-19°W
-16°W
-13°W
-10°W
-7°W
-4°W
-1°W
1°E
3°E
5°E
7°E
45°N
47°N
49°N
51°N
53°N

Figure 2.



Atlantic cod
Cuckoo ray
Dab
Dragonet
Grey gurnard
Lesser dogfish
Long rough dab
Norway pout
Poor cod
Red gurnard
Spotted ray
Thornback ray
Whiting
Witch
Standardized biomass (kg km−2)
Year
1990
1995
2000

Figure 3.