Bycatch of harbour porpoise, harbour seal and grey seal in Norwegian gillnet fisheries

Master thesis

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ABSTRACT

Data from a monitored segment (about 20 vessels) of the fleet of about 6000 small vessels (less than 15 meters length overall) operating bottom-set gillnets for cod (*Gadus morhua*) and monkfish (*Lophius piscatorius*) in the Norwegian coastal zone were used to estimate the bycatch rates of harbour porpoise (*Phocoena phocoena*), harbour seal (*Phoca vitulina*) and grey seal (*Halichoerus grypus*). Bycatch was estimated using the traditional stratified ratio technique and a GLM-based approach. Bycatch rates were then applied to the landing statistics for the target species for the whole fleet to estimate total bycatch. The stratified ratio estimates ranged from 2211 (CV 0.16) to 3218 (CV 0.17) harbour porpoises, 459 (CV 0.24) to 565 (CV 0.18) harbour seals and 68 (CV 0.27) to 128 (CV 0.41) grey seals. In the model-based approaches, estimates ranged from 2317 (CV 0.16) to 3218 (CV 0.17) harbour porpoises, 424 (CV 0.13) to 600 (CV 0.31) harbour seals and 83 (CV 0.36) grey seals. Unless the population of harbour porpoise along the Norwegian coast exceeds 176,500 animals, then current levels of harbour porpoise bycatch are unsustainable. Current levels of harbour seal bycatch (coupled with current hunting quotas) are most likely unsustainable. The grey seal data was insufficient to produce good modelling results.

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1. INTRODUCTION

1.1 HUMAN EXPLOITATION OF MARINE RESOURCES

Today, there are few, if any, oceanic regions where there is a complete absence of commercial fishery (Lewison et al. 2004). Both large scale industrial and small-scale artisanal fisheries are pervasive in their exploitation of the seas, and especially nutrient-rich, coastal waters associated with a high degree of mixing and upwelling are heavily exploited. Humans have a history of overexploitation of marine resources, sometimes with disastrous effects for the species involved. The Stellar's Sea Cow (*Hydrodamalis gigas* Zimmermann, 1780), for example, was hunted to complete extinction sometime in the 16th century. North Atlantic right whales (*Balaena glacialis* Müller, 1776) number less than 300 animals today (Knowlton et al. 1994, Donovan 1995), even though they have been protected for over 100 years. Similarly, blue whales (*Balaenoptera musculus* L. 1758), were hunted nearly to extinction (Chapman 1964), and still have not recovered (Branch et al. 2004). The Canadian cod (*Gadus morhua* L. 1758) fishery collapsed completely in the early 90s (Myers et al. 1997), and despite almost 25 years of strict management measures, has still not reverted back to previous levels (Brander 2005). The Peruvian anchovy (*Engraulis ringens* Jenyns, 1842) stock has similarly been characterized by severe instabilities over the last few decades (Boerema and Gulland 1973).

There are many more such examples. But even well-managed sustainable fisheries may have adverse demographic effects on populations of non-targeted species that reside in the area where the fishing occurs. Marine megafauna, such as whales, sharks, pinnipeds (seals and sea lions) and small cetaceans (dolphins and porpoises), as well as sea turtles and sea birds, and many other groups of animals, are vulnerable to incidental catches in fisheries. Incidental catches, or bycatches, is the taking of animals in fishing gear designed and/or intended to take some other species.

Modern fishing gears are ideally designed in a way that maximizes the taking of the intended catch ("target") species, while simultaneously minimizing the unintended taking of other species. This can be accomplished in a number of ways. Some gears, such as longlines, may use plastic "streamer lines" to scare away seabirds, or special hooks or "escape devices" to reduce bycatch (Dotson et al. 2010). Recently, acoustic deterrent devices (ADDs, or “pingers”) have been used to deter echolocating animals from approaching gill nets. But while these pingers may work as intended for some animals, such as beaked whales (family Ziphiidae) (Gearin et al. 2000, Carretta et al. 2008), bottlenose dolphins (*Tursiopsis truncates* Montagu, 1821) (Waples et al. 2013) and harbour porpoises (*Phocoena phocoena* L. 1758) (Palka et al. 2008), they are not without problems of their own, because they may actually *attract* other groups of animals, such as seals (Mate et al. 1986) and sea lions (Carretta and Barlow 2011), in what is popularly called “the dinner bell effect”.

1.2 ENTANGLEMENT IN GILL NETS

In the case of gill nets, catch is largely determined by the mesh size of the net. To target a particular species, nets with a mesh size that matches the size of the target fish are used. Fish smaller and larger than the intended catch species (including individuals of the target species belonging to younger age classes), are less likely to get trapped, either because they can just swim through the net or because they don’t fit through at all. These measures confer a degree of *size specificity*, or *selectivity* on the gear.

But this selectivity is not perfect. Gill nets intended to take one type of fish may still incidentally catch many other types of fish in the same size category, or even other animals such as marine mammals. These animals may get stuck in the net themselves, or get entangled in mooring ropes used to hold the nets in place. Among marine mammals (a loose non-taxonomic term which includes a variety of animals, such as otters and polar bears), pinnipeds and cetaceans are the taxonomic groups that are most vulnerable to bycatch in fisheries, owing to their aquatic lifestyle. It is not currently well known how these animals wind up getting stuck in the nets, but nondetection, curiosity and possibly social organization may all be contributing factors.

Once entangled, animals that get stuck in the nets are unable to return to the surface to breathe, and subsequently drown and die. Sometimes, trapped animals may be able to break loose, but in the process of struggling to get free, they may incur serious wounds, and even if they are able to successfully free themselves, they may still be partially entangled in ropes or other parts of the net. This could render them unable to swim to the surface, hinder navigation or make them unable to catch prey or socialize properly with conspecifics. Entanglement is almost always fatal to marine mammals, even if it doesn't kill them until days, weeks or even months later (Moore and van der Hoop 2012).

1.3 THE PREDISPOSITION OF MARINE MAMMALS TO ENTANGLEMENT

The geographic distribution, range, feeding habits and life history strategy of marine mammals make them particularly vulnerable to interactions with commercial fishing fleets. Being fairly large animals (compared to, say, some gadoid fish), and characterized by long life spans (~20 years for small cetaceans, 30+ years for pinnipeds, 50+ for large cetaceans) and late sexual maturity (3+ years), marine mammals have K-selected life histories. Pinnipeds and cetaceans typically only produce a single pup/calf once every few years, and a large proportion of young ones never make it to adulthood due to high neonatal mortality (Mattlin 1978, Barlow and Boveng 1991, Gabriele et al. 2001, Hall et al. 2001, Hanson et al. 2013). For those that make it to adulthood however, mortality rates are much lower, as both seals and small cetaceans occupy high trophic levels in the food chain and have few natural predators (the main predators being killer whales, polar bears and humans). The demographic make-up of many marine mammal populations can therefore be expected to be skewed towards a higher abundance of individuals in higher age classes. In other words, these populations tend to consist of a smaller number of younger and a larger number of older individuals. Marine mammals are extensively distributed in all the world's oceans, and preferentially range in the relatively rich coastal waters, where prey is more abundant and fishing pressure the highest. Seals and harbour porpoises have been known to seek out the same good fishing sites as commercial fishing vessels, presumably because of a localized high abundance of a common catch/prey species (Roche et al. 2007, Goetz et al. 2015).

This may be a contributing reason for why they get stuck in fishing gear so often. While marine mammals may get entangled in a variety of different fishing gears, bottom set gill nets constitute the greatest single cause of this kind of incidental killing (Hall et al. 2000, Read and Wade 2000, Dans et al. 2003, Read et al. 2006). 84% of cetacean and 98% of pinniped bycatches occur in gill net fisheries (Read et al. 2006). Populations (or subpopulations that do not readily interbreed) that are subject to high bycatch pressures can decline over short periods, because the taking of a few individuals in a small population with the characteristics described above, can be expected to have large adverse demographic effects on the remaining animals in the population. Population declines may occur so rapidly, that once management authorities notice the decreasing trend, it may already be too late to take effective measures, as evidenced in several of the historical examples mentioned above. Once the damage is done, it may take a very long time for affected populations to recover.

Incidental bycatches of non-targeted animals in fisheries is a global issue concerning all nations that engage in fishery activities. One conservative US estimate of global bycatch revealed that as many as 653,365 marine mammals (307,753 cetaceans and 345,611 pinnipeds) may be killed in fisheries every year (Read et al. 2006), but the authors speculate that actual bycatch may be higher. It is abundantly clear that even if global bycatch were only a fraction of this estimate, given the fragility of many cetacean and pinniped populations, bycatch would nonetheless still represent a serious problem in fisheries today. Norway is a nation with a long and strong tradition of coastal fisheries. In 2014, Norwegian fishing vessels landed a total of 2.3 million tons of fish, crustaceans and molluscs, estimated to a monetary worth of NOK 14.4 billion (Statistics Norway 2015). Fish and fish products constitute Norway's second largest export commodity.

1.4 BYCATCH OF MARINE MAMMALS IN NORWAY

In Norway, the largest bycatch of marine mammals occurs in the bottom set gill net fisheries targeting cod and monkfish (Scans II 2008, Bjørge et al. 2013). Harbour porpoises, harbour seals (*Phoca vitulina* L. 1758) and grey seals (*Halichoerus grypus* Fabricius 1791) are frequently killed in these fisheries. The harbour porpoise is one of the smallest porpoises, with adults typically growing to 1.4 to 1.7 m and weighing 60 - 76 kg. They are limited to continental shelf waters, and often venture into fjords and up into estuaries (Fontaine et al. 1994). They typically feed on squid, crustaceans and various types of fish, most important among them mackerel, herring and lesser sand eel. Most of the time, porpoises occur either alone or in small groups, but may occasionally hunt in packs and herd fish together. The abundance of harbour porpoise in Norwegian waters is not known, but incidental killing in fishing gear is thought to be a great risk to North Atlantic populations (Caswell et al. 1998). As of this writing, the national whale counting expedition in Norway in 2016 is conducting aerial surveys in an attempt to improve our estimates of the population size of porpoises along the Norwegian coast.

The harbour seal population in Norway distributed along the entire coastline and numbers 7568 animals, according to counts of moulting seals in 2011-2015 (Nilssen et al. 2016). The overall population has increased slightly since the previous count (2003-2006), but local depletions were observed. Harbour seals seldom venture more than 20 kilometres off shore. They are generally considered a solitary species, but gather in large colonies for hauling out and during the breeding season. They feed on small fish, especially saithe, pout and herring.

The grey seal has one small breeding colony in Rogaland and the species is continuously distributed from Trøndelag to Varanger. The overall population has been slowly growing over the past two decades and some models estimate the population at 8740 animals (95% CI 7320-10170) in 2011 (Øigård et al. 2012). However, recent surveys aimed at counting grey seal pups have revealed a dramatic decline in pup production in the recent five years in some areas (Nilsen and Bjørge 2015). Grey seals are gregarious and reside in large colonies. They feed on fish, especially monkfish, cod, saithe and pollock.

The harbour porpoise is legislatively protected from harvesting in Norway through national laws and international agreements, such as CITES (Utenriksdepartementet 2002). The harbour and grey seals however, are harvested in a quota-regulated harvest, but this harvest should not be the reason for the dramatic decline in abundance in some areas. The extent of the bycatch of these species in fisheries is not known, but it is believed to be not negligible.

1.5 HYPOTHESES

In this paper, I hypothesize that current levels of fishery-caused incidental killing of harbour porpoises, harbour seals and grey seals have the potential to significantly impact the population trajectories of the Norwegian populations of these species. The objective of the paper is to explore the extent of this bycatch using bycatch rates derived from a coastal reference fleet. My analyses will be based on a similar work by my supervisor Arne Bjørge et al. (2013), which estimated harbour porpoise bycatch in 2006 – 2008. This manuscript therefore, is both a supplement to and an extension of that work.

2. MATERIALS AND METHODS

2.1 MONITORING BYCATCH: THE COASTAL REFERENCE FLEET

Bycatch data was obtained from fishing logs from the Coastal Reference Fleet (CRF), which is administered by the Institute of Marine Research (IMR). The CRF constitutes a monitored segment of the Norwegian coastal fleet of fishing vessels. The entire coastal fleet is comprised of approximately 5500 – 6500 fishing vessels (this number varies from year to year) with a length overall less than 15 m. But regular fishing vessels in the fleet do not keep and maintain detailed catch information beyond species and landed weight. Fishing vessels in the CRF on the other hand keep very detailed records of their fishing activities. The particular information recorded is briefly outlined in Section 2.2.

In forming the CRF, candidate vessels were selected from among the coastal fleet based on criteria such as fishing gears used, species targeted and geography (seeking at least two vessels in each of nine statistical areas, see Figure 2.1), in an effort to ensure that the CRF as a whole should be as representative as possible for the < 15m section of the commercial fleet. Reference vessels were selected randomly from among the candidate vessels that met these criteria. These reference vessels were contractually obligated to provide detailed information about their fishing activities using S- and T-forms, which will be described in the next section. In the ten-year period in this study, a total of 49 different vessels served in the fleet. The contracts were one-year engagements. Over time, some contracts were renewed, while others were not. Occasionally new vessels were contracted. At any time, the fleet was comprised of 16 – 24 vessels. These vessels were distributed across nine statistical fishing areas (Figure 2.1), collectively covering the entire Norwegian coastline. Each vessel was primarily associated with one specific fishing area, distributed so that there were two vessels associated with each fishing area. The data shows, however, that vessels were also fishing outside their primary area.

2.2 DATA FORMAT

Each fishing trip in the CRF was recorded in "S-forms". The S-forms included the following information fields: *year*, *vessel type*, *vessel name*, *vessel length*, *month*, *year*, *station number*, *serial number*, *station type*, *station latitude*, *station longitude*, *north/south/east/west*, *system*, *fishing area*, *sector*, *location*, bottom *depth*, *number of gears*, *gear code*, *gear number*, *start time*, *start log*, *stop time*, *condition*, *quality*, *max fishing depth* and *min fishing depth.* Not all of the S-form data fields were populated with information, and a number of them (such as system/quality) were not directly relevant to this analysis. Real vessel names in the data had been mapped to anonymous identifier codes to protect the identity of the CRF vessels by the CRF data curator. The vessels were given handles such as "KY1", "KY2", etc. to uniquely identify them in the data set.

Each separate type of catch item (i.e. different species of catch) obtained at a fishing station was recorded in T-forms. Thus, every S-form had a number of T-forms associated with it (nine on average) specifying details regarding the total catch for that fishing trip. The T-forms included the following information fields: *species code*, *species name*, *sample number*, *group*, *conservation*, *measure*, *weight/volume*, *weight* and *count*. The *species code* field specified the type of species name used in the corresponding *species name* field, e.g. whether the species was specified by its Latin, English or Norwegian name. All the T-forms in the data set used Norwegian names. The *condition* field specified how the catch had been conserved on board the vessel, e.g. stored fresh, on ice, in formalin or in alcohol. Additional information on the S- and T-form format specification is available in IMR's field manual for sampling fish and crustaceans (Mjanger et al. 2014), but only in Norwegian.

2.3 PRE-PROCESSING OF DATA

Incomplete data records (such as T-forms missing species information) and S-forms (along with associated T-forms) from areas that were not part of the nine coastal fishery statistics areas were deleted. 86 S-forms were missing positional information about the fishing station, but these were retained because the catch information was intact. Data fields with information that was not relevant to the analysis were removed (e.g. *start time, condition, quality*) to alleviate the computational load of data treatment and calculations. To be able to model temporal variation on scales coarser than month, two factor variables *quarter* and *season* were added, with four and two levels, so that each quarter factor level corresponded to a three-month period and each season factor level corresponded to a six-month period. Neighboring areas were combined into a factor variable *region*, with four levels according to the following scheme. Region 1 consisted of areas 03, 04 and 05; region 2 consisted of area 00; region 3 consisted of areas 06 and 07; and finally region 4 consisted of areas 08, 28 and 09, as indicated in Fig 2.1. Because of space limitations on the physical S-forms that are filled out, presumably by hand, by the fishers, gear counts in the CRF data set were specified as a somewhat confusing composite of two fields: number of gears and gear number, the former being an integer (assumed 0 if missing) and the latter being an adjustment factor (also assumed 0 if missing). Letting *GC* be the actual gear count, *NG* be the number of gears and *GN* the gear number, the gear count calculation can be expressed:

$$GC = NG * 10^{GN}$$

The final data set consisted of the following fields: *year*, *month*, *quarter*, *season*, *area*, *region*, *vessel*, *serial*, *gear type*, *gear.count*, *lat*, *lon*, *species*, *catch*, *count*. The factor variables *quarter*, *season* and *region* were added to the landing data for the whole commercial fleet.

2.4 THE STUDY AREAS

The study area spanned the entire Norwegian coastline, from the Varanger fjord near the Russian border in the far north, to the Skagerrak Sea and the Hvaler archipelago near the Swedish border in the south. Due to a high degree of heterogeneity with regards to both catch and bycatch in different parts of the Norwegian coast, in order to be able to predict bycatches with increased spatial resolution, the study area was divided into nine statistic areas, as shown in Figure 2.1. However, due to low bycatch numbers in some areas, the areas, were recombined into four distinct regions (as represented by the shaded lines in Figure 2.1).

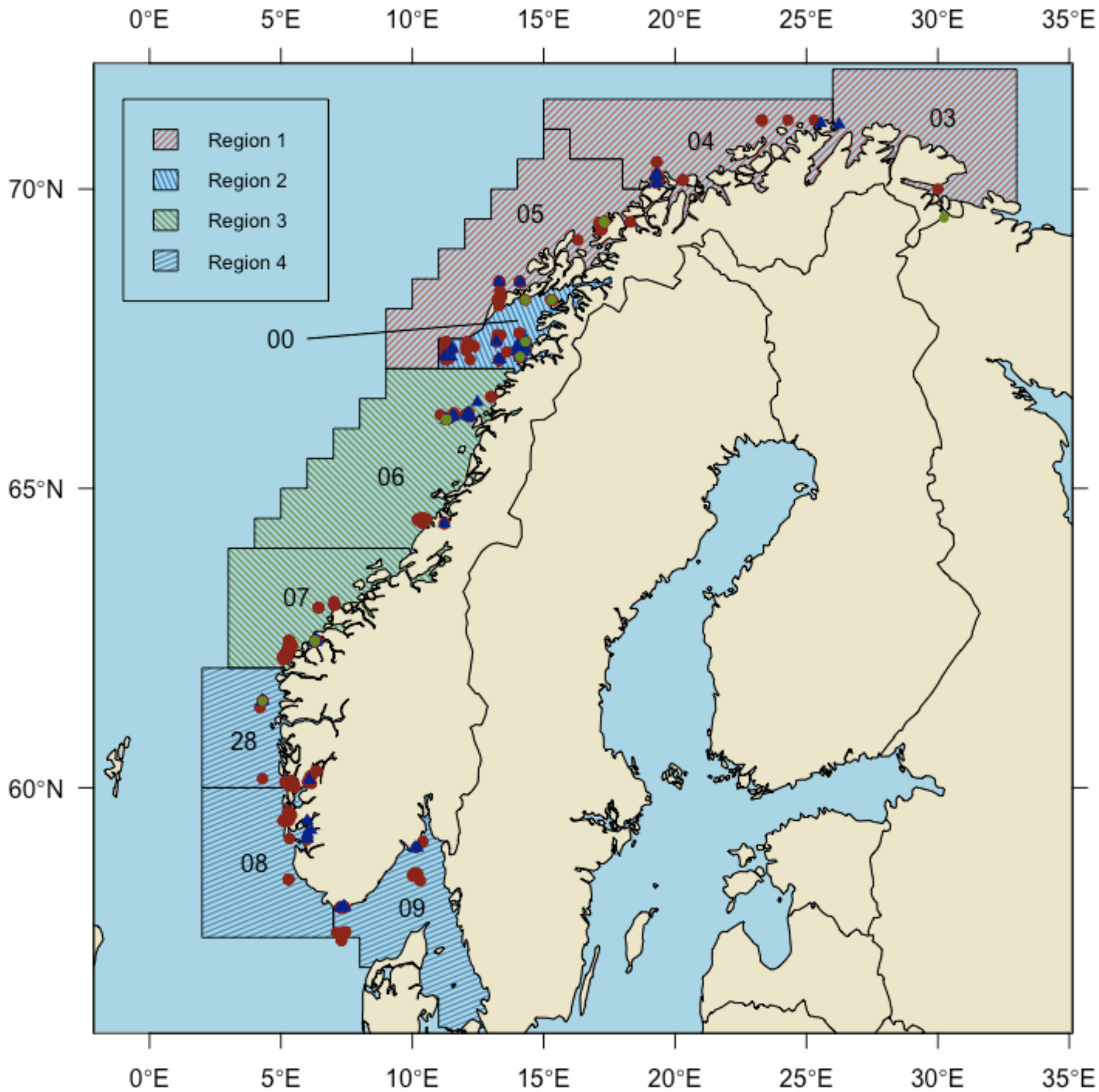

Figure 2.1*: The nine fishery statistic areas along the Norwegian coastline. Shaded background lines represent the designation of areas into four different regions (see plot legend). Red dots, blue triangles and green dots represent bycatches of harbour porpoise, harbour seal and grey seal, respectively, in the CRF between 2006 and 2015. Each symbol represents a fishing station associated with some bycatch, but icon sizes are not scaled according to the number of animals bycaught. See figure 3.7 for an overview of the temporal and spatial variation in bycatch.*

2.5 DEFINING AND DISTINGUISHING FISHERIES

The CRF did not target cod and monkfish exclusively; other commercially important species (such as saithe (*Pollachius virens,* L. 1758), mackerel (*Scomber scombrus*, L. 1758), herring (*Clupea harengus*, L. 1758), haddock (*Melanogrammus aeglefinus*, L. 1758), and many more) were frequently fished as well. In the period 2006 – 2015, cod catches constituted 44.9% of total landings, and monkfish a mere 2.7%. Therefore, to be able to extrapolate the CRF derived bycatch rates to the entire coastal fisheries, it was necessary to distinguish the cod and monkfish fisheries from other fisheries. Bycatch rates can be reasonably expected to differ between the two fisheries, due to the use of different fishing gears, fishing sites, timing, not to mention the clear seasonality of each fishery. Thus, the cod and monkfish fisheries had to be distinguished from each other and from other fisheries. On the premise that the most abundant catch at each fishing station represented the species targeted, one approach would be to define the fisheries as catch obtained at fishing stations at which the ratio of cod or monkfish catches exceeded some minimum threshold value, e.g. 0.05. However, exploratory analyses of the data revealed that this approach would include in the cod and angler fisheries definitions catches obtained in a wide variety of gear types, such as lobster traps, seine nets and hook lines. Since none of these gears are likely to be associated with the entanglement of marine mammals, this catch proportion based approach seemed an inappropriate fisheries definition.

A previous study on bycatch of harbour porpoise in Norway, similarly based on CRF data, used a gear-based fisheries definition. In the study, bottom set gill nets with mesh sizes between 75 and 105mm were defined as cod nets and bottom set gill nets with a mesh size of 180mm as monkfish nets (Bjørge et al. 2013). The CRF data does however contain a large number of S-forms that use a general gear specification; specifying a type of gill net without specifying mesh size. It is not clear how Bjørge et al. treated these S-forms in their data processing, i.e.: whether they were included in either definition or not.

In the present study, the latter definitions for cod and monkfish fisheries were used, and additionally, gill nets of unspecified mesh size were included in the cod fishery. Table 2.1 lists the gear types used in the final fishery definitions and the ratio of total catch taken in the gear specified. By landed fish weight, this extended definition captured 91% of the monkfish and 78% of the cod in the data set. The CRF data was then aggregated in the same way as the landing data for the whole commercial fleet, i.e. cod and monkfish catches and harbour porpoise, harbour and grey seal bycatches were summed for every year + month + area + fishery combination. The resulting data set consisted of 10 years × 12 months × 9 areas × 2 fisheries = 2160 data points.

Table 2.1: *Overview of gears included in each fishery. Codes refer to gear codes specified in IMR's field manual for sampling fish and crustaceans (Mjanger et al. 2014). The catch ratios refer to the ratio of catch taken with each of the listed gears to the total catch taken of that species across all gears in the CRF (by weight).*

Code	Mesh size	Description	Catch ratio
Cod fishery			
4110	Unspecified	Bottom set net	7.19%
4111	Unspecified	Nylon	0.08%
4112	Unspecified	Monofilament.	0.17%
4113	93mm	Monofilament	15.78%
4114	90mm	Monofilament	10.24%
4120	90mm	Multimono	0.43%
4123	76mm	Monofilament	2.11%
4125	84mm	Monofilament	1.06%
4127	96mm	Monofilament	2.10%
4128	105mm	Monofilament	1.00%
4141	80mm	Bottom set net	2.30%
4142	90mm	Bottom set net	13.03%
4143	100mm	Bottom set net	22.07%
			Total 77.6%
Monkfish fishery			
4129	180mm	Monofilament w/ pingers	0.07%
4149	180mm	Bottom set net	90.96%

Landings statistics for the entire commercial fleet of gillnetters less than 15 m were provided by the Directorate of Fisheries. These statistics are based on fish landed in harbors and were not specified by gillnet type. They therefore include fish taken by all types of bottom set gillnets. To account for this in the bycatch estimates, catch data from the commercial fleet was adjusted by the ratio of catch obtained in the cod/monkfish fisheries (as defined above) to the total catch for cod/monkfish, estimated from the CRF data.

$$C_{a,f} = C_{a,F} \times \frac{C_{b,f}}{C_{b,F}}$$

In this equation, *C* denotes catch, the subscripts *a* and *b* represent the commercial fleet and the reference fleet, respectively, while *F* and *f* indicates "all gears" and gears according to the above definitions, respectively. Thus, $C_{a,f}$ would represent the catch for fishery F in the commercial fleet, but corrected for only cod/monkfish gear use. The landing statistics were aggregated by year, month, and statistical area, in the same way as the CRF data.

2.6 ESTIMATING BYCATCH RATES

2.6.1 ESTIMATING BYCATCH RATES BY A RATIO-BASED APPROACH

Bycatch rates in the CRF were estimated using both ratio-based and model-based approaches. In the ratio-based approaches, the data was stratified according to five different stratification schemes, by *month*, by *area*, by *region*, and by each possible combination of *area* × *month* and *region* × *month*. The areas and regions used were the same ones as described previously. Two different measures of fishing effort were used: catch (tons of landed fish) and number of fishing trips. If we consider the $r_{i,j}$ as the ratio of bycatch to fishing effort, then a generalized ratio estimate can be expressed mathematically in the form:

$$r_{i,j} = \frac{t_{i,j}}{1 + e_{i,j}}$$

Where $t_{i,j}$ is the observed takes (bycatches) in stratum *i* for the fishery *j* and $e_{i,j}$ is the fishing effort in stratum *i* for fishery *j*. The constant 1 in the denominator was added because some stratification schemes yielded strata (such as the cod fishery in the latter half of the year) that had no bycatch, and division by zero is not defined. The bycatch ratios were then applied to the nation-wide fleet by multiplication with the unobserved effort. Letting B be the predicted number of takes in both fisheries across all strata and $E_{i,j}$ be the unobserved effort in stratum *i* for fishery *j*, we have:

$$B = \sum r_{i,j} \times E_{i,j}$$

This equation represents the final ratio-based bycatch estimate. To obtain a measure of the variance of the estimate, CVs and confidence intervals were calculated. Letting *CV* be the coefficient of variation, *s* the standard deviation of the sample and $\bar{x}$ e the mean of the sample, then:

$$CV = \frac{s}{\bar{x}}$$

Letting *CI* be the 95% confidence interval, *B* the predicted bycatch and $SE_{\bar{x}}$ the standard error of the mean, then we have:

$$CI = [B - 1.96 * SE_{\bar{x}}, B + 1.96 * SE_{\bar{x}}]$$

The term $SE_{\bar{x}}$ can be expressed as the ratio between the standard deviation of the mean *s* and the square root of the total number of strata *N*:

$$SE_{\bar{x}} = \frac{s}{\sqrt{N}}$$

The CVs and 95% confidence intervals were calculated by bootstrapping. In each bootstrap iteration, a number of observations N equal to the number of strata in each stratification scheme were sampled at random with replacement from the CRF data. A bycatch estimate was calculated from the new sample, and extrapolated using data from the commercial fleet, as explained above. This procedure was replicated 1000 times, and the CVs and 95% CIs were calculated from the resulting distribution of B values (i.e. predicted bycatch).

2.6.2 ESTIMATING BYCATCH RATES BY A MODEL-BASED APPROACH

In the model-based approach, generalized linear additive models (GAMs) were used. Table 2.2 lists the variables (model terms) that were considered for inclusion in the model along with a short description of each. The *catch* and *trips* variables were used as proxies (measures) of fishing effort, and were thus mutually exclusive. The model formula should logically only include one or the other. Catch and trips thus represented two main "lines" of models. Catch and trips were log-transformed and then entered as the first term of the model as an offset. Entering the effort term as an offset in the context of a GAM, assumes that the coefficient associated with that term is a constant one, instead of estimating it. When using an offset then, the value of the offset is simply added to the linear predictor (hence the name, "offset") and would be appropriate if there was a strong correlation between the response variable (bycatch) and the effort variable.

The other variables were added in a forward step-wise process, by trying all the available variables in turn and choosing the one that resulted in the greatest model AICc reduction. This process was "sanity-checked" (e.g.: to make sure the model made biological sense and that no mutually exclusive variables were used together) by visual inspection of the variance in the

Table 2.2*: List of candidate variables for inclusion in the model. An asterisk (*) denotes that the variable has been more accurately described in the text. Other variables are deemed self-explanatory. Identical numbers in superscript indicate mutually exclusive terms.*

Variable name	Description
CATCH[1]	The landed catch of the target species in kilograms
TRIPS[1]	Number of fishing trips
YEAR	Year, 2006 – 2015
MONTH[2]	Month, 1 – 12
QUARTER[2] *	Quarter of the year, Jan-Mar, Apr-Jun, Jul-Sep, Nov-Dec
SEASON[2] *	Season of the year, January-June, July-December
AREA[3] *	Fishing statistics area (see Figure 2.1)
REGION[3] *	Combination of fishing areas into regions, 1 – 4
BOTTOM DEPTH	Average bottom depth at fishing station
FISHERY	The type of fishery, either “cod” or “monkfish”
GEAR COUNT *	The number of nets set (GC)
GEAR MINIMUM DEPTH	The shallowest depth that the gear has fished in
GEAR MAXIMUM DEPTH	The deepest depth that the gear has fished in
PINGER	N/A, data insufficient

data. Bycatches were grouped into each categorical variable and the means and variances between groups were inspected. The resulting tables were too numerous and lengthy to include in this manuscript. A high variance in data within individual group means was considered an indication that that variable may have a good explanatory potential, and the magnitude of this variance was also taken into consideration when determining which and in what order terms were added to the model.

When fitting a GAM, the modeling function needs to make assumptions about the probability distribution of the data. It is therefore necessary to specify the family of statistical distributions that should be used. In their 2006 – 2008 analysis of the CRF data, Bjørge *et al*. (2013) assumed that the bycatch followed a Poisson distribution, an assumption which is often made for non-negative, countable (e.g. bycatch) data. The Poisson distribution applies when the event that is under observation results in something that can be counted in whole numbers, and that each observed event is independent, so that the result of one event does not affect another. We must also know the average frequency of occurrence for the time period, and we must be able to count how many events have occurred. So, applying these criteria to the data at hand: Assuming each fishing trip is associated with the hauling of one set of nets (that have soaked for approximately 24 hours) in one location, then we may consider each fishing trip as one "event". We can count the number of fishing trips. Bycatch associated with one fishing trip cannot realistically be expected to change the probability of getting bycatch on the next trip (the change in probability would be infinitesimal). We can calculate the average rate of bycatch per fishing trip, and we can count the number of trips undertaken. The number of takes cannot be negative numbers, and so there is a lower limit on the data. In theory, therefore, we can expect the data to be a subset of the set of nonnegative integers, denoted $\mathbb{Z}_{\geq 0}$, which corresponds to the "counting numbers", including zero. There is no upper limit on the counts, but realistically, given the size of the vessels and the nature of the fisheries in the Norwegian coastal fisheries, we should not expect takes much greater than ten animals, in the case of the harbour porpoise, and fewer still, in the case of the seals. On this basis, the Poisson distribution seems like a reasonable choice.

Usually, the choice of distribution incorporates information about the population which is being modeled, and in particular the expected distribution of that population based on biological and ecological knowledge as outlined above. While the Poisson distribution seems like a good fit, if the data suffers from over-dispersion and/or zero-inflation, distributional assumptions may be violated. This could potentially lead to uncertainty regarding parameter estimates (Martin et al. 2005). Therefore, it seemed prudent to test the Poisson assumption by exploring the extent of over-dispersion and zero-inflation in the fitted models, and additionally examining results of models using other candidate distributions.

The Poisson distribution assumes that the mean and variance are the same, but if the data show extra variation that is greater than the mean, the data is per definition over-dispersed. Over-dispersion in the Poisson GAMs was tested using the method outlined in Cameron and Trivedi (1990) and Cameron and Trivedi (1998), using *overdispersion* from the R package *AER* (Kleiber and Zeileis 2008). In a Poisson model, we have the mean $E(Y) = \mu$ and $var(Y) = \mu$ (i.e. the mean and the variance are equal). The overdispersion test simply tests a null hypothesis of Poisson variation in the model against an alternative hypothesis that the variance has some particular form depending on the mean. Mathematically,

$$var(Y) = \mu + c + f(\mu),$$

where *c* is some constant, and *f(μ)* is some monotonic transformation function of the mean. If the test results in a c value greater than one, this indicates over-dispersion and conversely, values less than one indicate under-dispersion. In running this test, a *trafo* parameter of 1 was used, corresponding to an alternative hypothesis that the true dispersion was greater than 1.

The appropriateness of the Poisson distribution was further examined by inspection of regression rootograms, as in Kleiber and Zeileis (2016b), using the countreg R package (Kleiber and Zeileis 2016a), which contains an implementation of their method. For data where over-dispersion is present, the negative binomial (NB) distribution may be more appropriate than the Poisson, as pointed out by Hedeker (2005). The NB distribution can be considered a generalization of the Poisson distribution, but has an extra parameter to model the over-dispersion. Correspondingly, the zero-inflated Poisson distribution *may* be better able to account for the large number of zeroes in the CRF data set, assuming that a regular Poisson model is zero-inflated. The candidate distributions that were tested were thus the Poisson, the negative binomial (NB), and the zero-inflated Poisson (ziP). The best GAM from Bjørge *et al*. (2013) (model 1.10 in their paper) and associated nested models (models 1 – 13, listed in Table 2.3) were run using each of the statistical distribution families listed. In all models, "number of takes" (number of porpoises/seals taken) was entered as the response variable and the log-transformed effort proxy was specified as an offset. Nested models within each family were compared using AIC (Akaike 1974) and AICc scores.

Distributional model support in terms of AIC/AICc scores was determined heuristically for each investigated species by trying a set of preliminary models and comparing their AIC scores. For each species, if the best (lowest) AIC scores associated with different model formulations were consistently associated with one particular distribution family, then that family was used for that species. If no such consistency was found, then the best distribution was decided by model "majority voting".

Table 2.3: *The various GAM models evaluated for modelling bycaught porpoises. "+" separates main effects, ":" denotes interactions and "*" indicates all main effects and interactions. REGION, FISHERY, SEASON and QUARTER are factor variables. Model 13 corresponds to model 1.10 in Bjørge et al. (2013).*

#	Model formulation
	No smoothing terms (only factor variables):
1	offset(LOG.CATCH)
2	offset(LOG.CATCH) + REGION
3	offset(LOG.CATCH) + FISHERY
4	offset(LOG.CATCH) + SEASON
5	offset(LOG.CATCH) + FISHERY + SEASON
6	offset(LOG.CATCH) + FISHERY + REGION
7	offset(LOG.CATCH) + FISHERY + QUARTER
8	offset(LOG.CATCH) + SEASON + REGION
9	offset(LOG.CATCH) + QUARTER + REGION
10	offset(LOG.CATCH) + FISHERY + SEASON + REGION
11	offset(LOG.CATCH) + FISHERY + SEASON + REGION + FISHERY : SEASON
12	offset(LOG.CATCH) + FISHERY + SEASON + REGION + FISHERY : REGION
13	offset(LOG.CATCH) + FISHERY + SEASON + REGION + FISHERY : REGION + FISHERY : SEASON

QQ-plots, or quantile-quantile plots, as in Augustin et al. (2012), of the best models within each family were used to evaluate the distributional assumptions of each model. Ordered theoretical quantiles based on the distribution selected were plotted against the ordered observed quantiles in the CRF data. If the distributional assumptions were correct, then the resulting points were expected to fall roughly on a straight line. The linearity of the resulting points was used as a measure of the suitability of that distribution.

CVs for the GAM models were calculated by bootstrapping in the same manner described previously. In each bootstrap iteration, a number of observations n = 2160 (10 years × 12 months × 9 areas × 2 fisheries) were sampled at random with replacement from the CRF data. A bycatch estimate was calculated from the new sample, and extrapolated using data from the commercial fleet. This procedure was replicated 1000 times, and the CV was calculated from the resulting distribution of predicted values.

Analyses were conducted in RStudio version 0.99.887 with R version 3.1.1 (R Core Team 2014), running on OS X El Capitan 10.11.3. Packages used include *mgcv* (Wood 2006), *nlme* (Pinheiro et al. 2014), *AER* (Kleiber and Zeileis 2009), *AICcmodavg* (Mazerolle 2016) and *countreg* (Kleiber and Zeileis 2016a) for modelling and model evaluation, *maps* (Becker et al. 2013) and *maptools* (Bivand and Lewin-Koh 2014) for map making and finally, *rgdal* (Bivand et al. 2014) and *geosphere* (Hijmans 2016) for GIS estimates and analyses.

2.7. ANALYSIS SOURCE CODE

For increased reproducibility and in the spirit of transparency and openness, all the R scripts used in the analyses conducted and in producing the figures in this paper are publicly available at a permanent github repository located at https://github.com/supermoan/bycatch. There is an explanatory "readme"-file included in the repository that describes the setup (file paths, etc.) necessary to reproduce the analyses. CRF data can be obtained by request from the Institute of Marine Science in Norway (www.imr.no) and national catch landings data can be obtained from the Directorate of Fisheries in Norway (http://www.fiskeridir.no).

3. RESULTS

3.1 TRENDS IN THE FISHERIES

The ten years of CRF data contained records (S-forms) for 20,115 fishing trips/stations, 10,633 of which belonged to the collective cod and monkfish fisheries. There was a distinct seasonal shift in both fisheries (Figure 3.1, top panels). In the CRF, 97% of cod catches were taken in the first half of the year, while 74% of monkfish catches were taken in the second half of the year. The corresponding numbers for the nation-wide fleet were 95% and 76%, respectively. The annual cod fishery starts in January, reaches a top in March, before decreasing rapidly over the next months and ending by May. The monkfish fishery starts in April, increases steadily, reaching a maximum in September, and then decreases gradually until settling at a low level in December/January, and reaching a minimum in March.

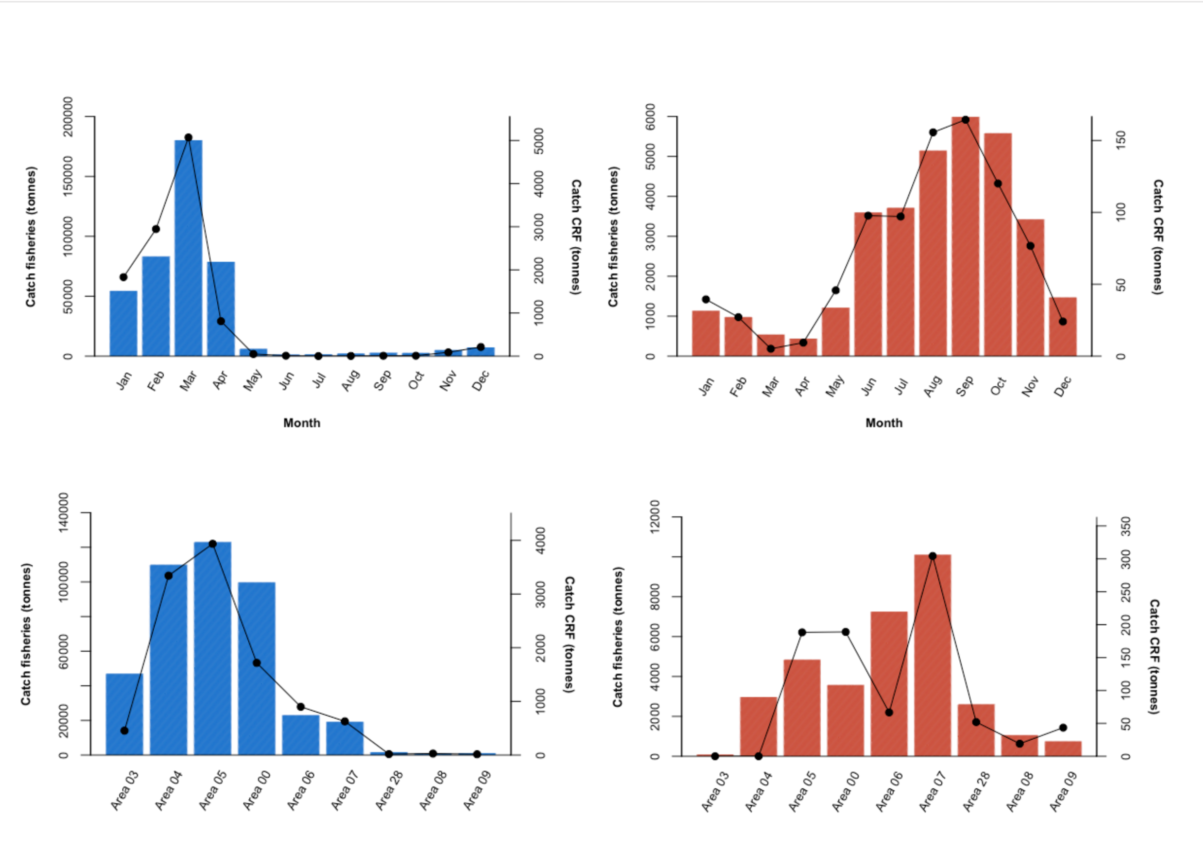

Figure 3.1: *Cod (left panels, blue bars) and monkfish (right panels, red bars) landings in the nation wide coastal fleet (bars) and in the CRF (lines and dots), summed by month (top panels) and area (bottom panels).*

The majority of cod fisheries occurred in the neighboring areas 04, 05 and 00, with intermediate levels in area 03 (further north) and low levels in the other areas further south along the coast (Figure 3.1, bottom left panel). Monkfish fisheries were more spread out along the coast, but the majority occurred in areas 06 and 07, with intermediate levels of fishing in areas just north (areas 04, 05, 00) and south (area 28) of that region (Figure 3.1, bottom right panel). There were low levels of monkfish fishery in areas 08 and 09 (both further south) and almost none in area 03 (far north).

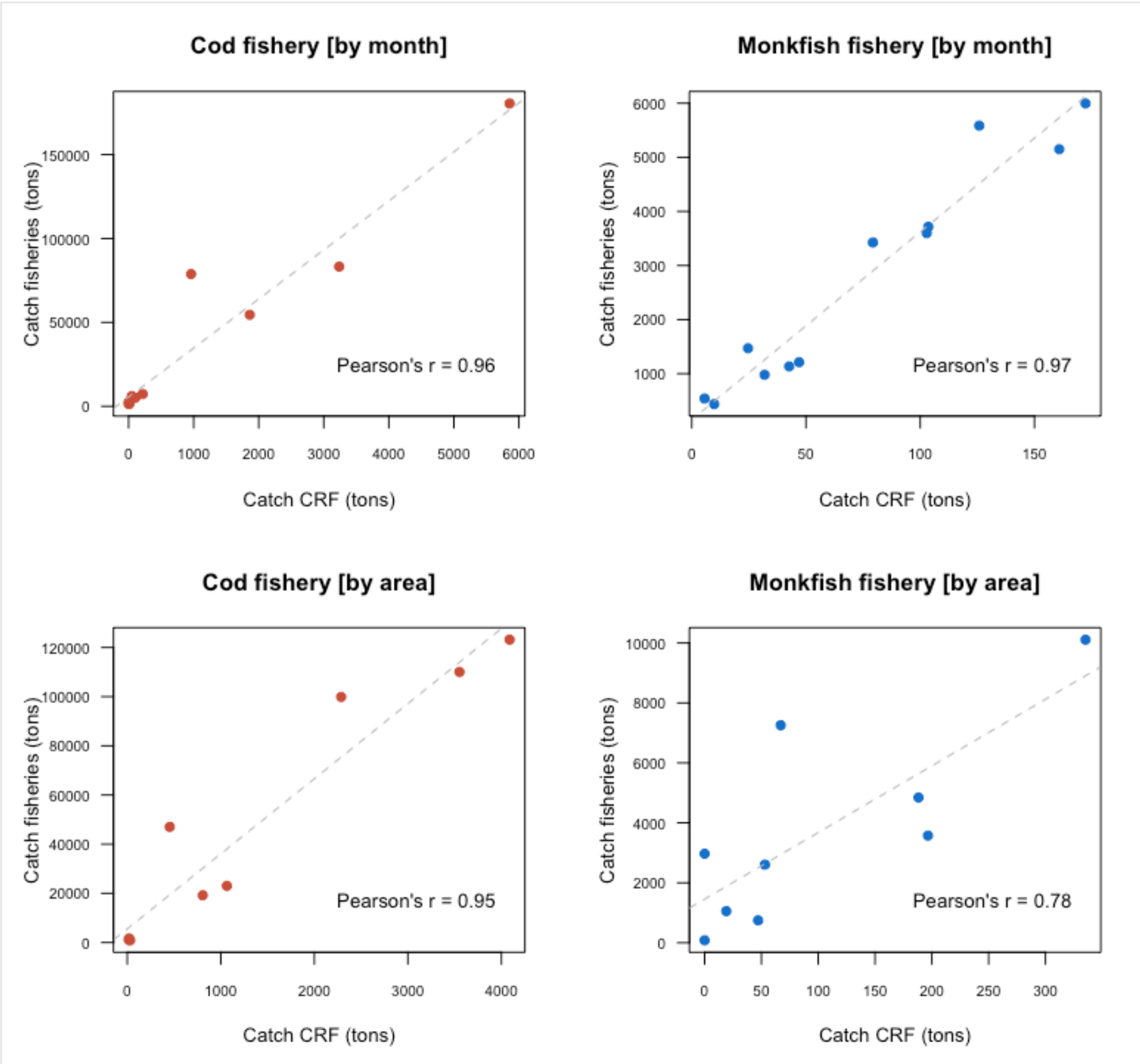

Figure 3.2*: Catches in the CRF plotted against catches in the fisheries, after having been aggregated by month (top panels) and area (bottom panels). Based on total catches for the years 2006 – 2015.*

The CRF landings followed the nation-wide landings very closely (Figure 3.2) when summed both by month (Pearson's $r = 0.96$, $p < 0.001$ (cod) and $r = 0.97$, $p < 0.001$ (monkfish)) and by area ($r = 0.95$, $p < 0.001$ (cod) and $r = 0.78$, $p = 0.013$ (monkfish)), which suggests, at least in terms of the monthly and areal variation in landed cod and monkfish, that the CRF is representable for the nationwide fleet. The correlations between catch in the CRF and in the fisheries were slightly lower when summed by year and month combinations, both for cod (Pearson's $r = 0.82$, $p << 0.001$) and monkfish (Pearson's $r = 0.81$, $p << 0.001$) fisheries, but still highly significant.

3.2 TRENDS IN THE BYCATCH

In all the fisheries added together (i.e. in the entire unfiltered CRF data set), a total of 877 harbour porpoises, 151 harbour seals and 20 grey seals were taken. In most cases, takes were of a single animal, but takes of up to 13 harbour porpoises, up to seven harbour seals and up to two grey seals were observed (Figure 3.7). 331 harbour porpoises (38%), 17 harbour seals (11%) and eight grey seals (40%) were caught in the cod fishery, and 405 harbour porpoises (46%), 112 harbour seals (74%) and 12 grey seals (60%) were caught in the monkfish fishery. Thus, a total of 84% of the CRF bycatch was taken in the cod and monkfish fisheries. The remaining 12% were caught in other fisheries. Figure 3.3, Figure 3.4 and Figure 3.5 show the catch locations of harbour porpoises, harbour seals and grey seals, respectively, taken in the CRF. Each red triangle represents a fishing station at which bycatch of the corresponding species occurred, but does not indicate the number of animals taken at that station (except in Figure 3.3). Since individual fishing stations in some cases were associated with takes of more than one animal, there are fewer catch locations on the maps than total bycatch figures would indicate. Figure 3.3 shows that harbour porpoises were taken along almost the entire Norwegian coastline. There are a few gaps in the catch sites (represented by triangles, diamonds, squares and circles) that dot the map, notably from the outermost Oslo fjord to Lindesnes, from Lindesnes west- and northwards to Stavanger. There are also a couple of gaps along the Nordland coast (mid-Norway, 64-67°N). There is a long stretch of coastline at which there were no bycatches along the north-facing Finnmark coast, from 20°E and eastwards, interrupted by a few takes at 71°N 24-25°E. Based on the map, most porpoise takes occurred in the Lofoten, Senja and Stavanger/Bergen areas. These regions correspond roughly to the statistical fishing areas 00, 05 and 28/08 (Figure 2.1), the former two of which are areas where there were a very high level of fishing activity targeting cod (Figure 3.1) during parts of the year.

Figure 3.4 and Figure 3.5 include the geographical distributions of the two seal species, based on previous national seal counts (Nilssen and Bjørge 2015). The species range is designated in dashed dark blue color. Known regions of high abundance are designated in dashed red color. Figure 3.4 shows that the great majority of harbour seals were taken in the Lofoten area and southwards along the coast (approximately 66 - 68°N). There were also many takes in the

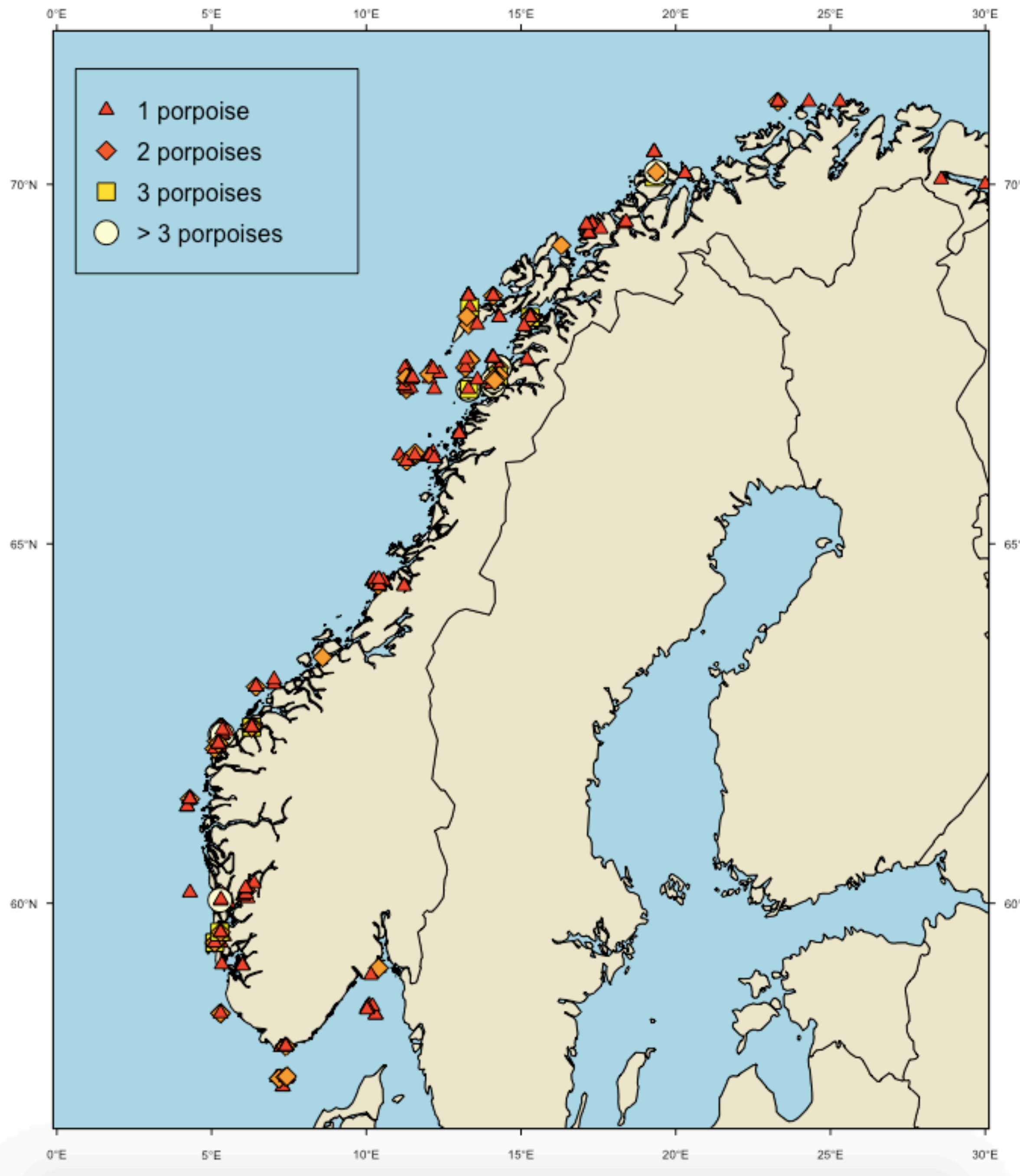

Figure 3.3*: Catch locations for harbour porpoises (N=225) in the combined cod and monkfish fisheries in the CRF in the period 2006 – 2015. Different plotting characters indicate the number of porpoises taken at each location, according to the plot legend. Triangles are used to indicate takes of one porpoise, diamonds for takes of two porpoises, squares for three porpoises and circles for takes of more than three porpoises. The size and color intensity of the plotting characters correspond to the size of the bycatch, with symbols for larger bycatches being larger and of a more intense color.*

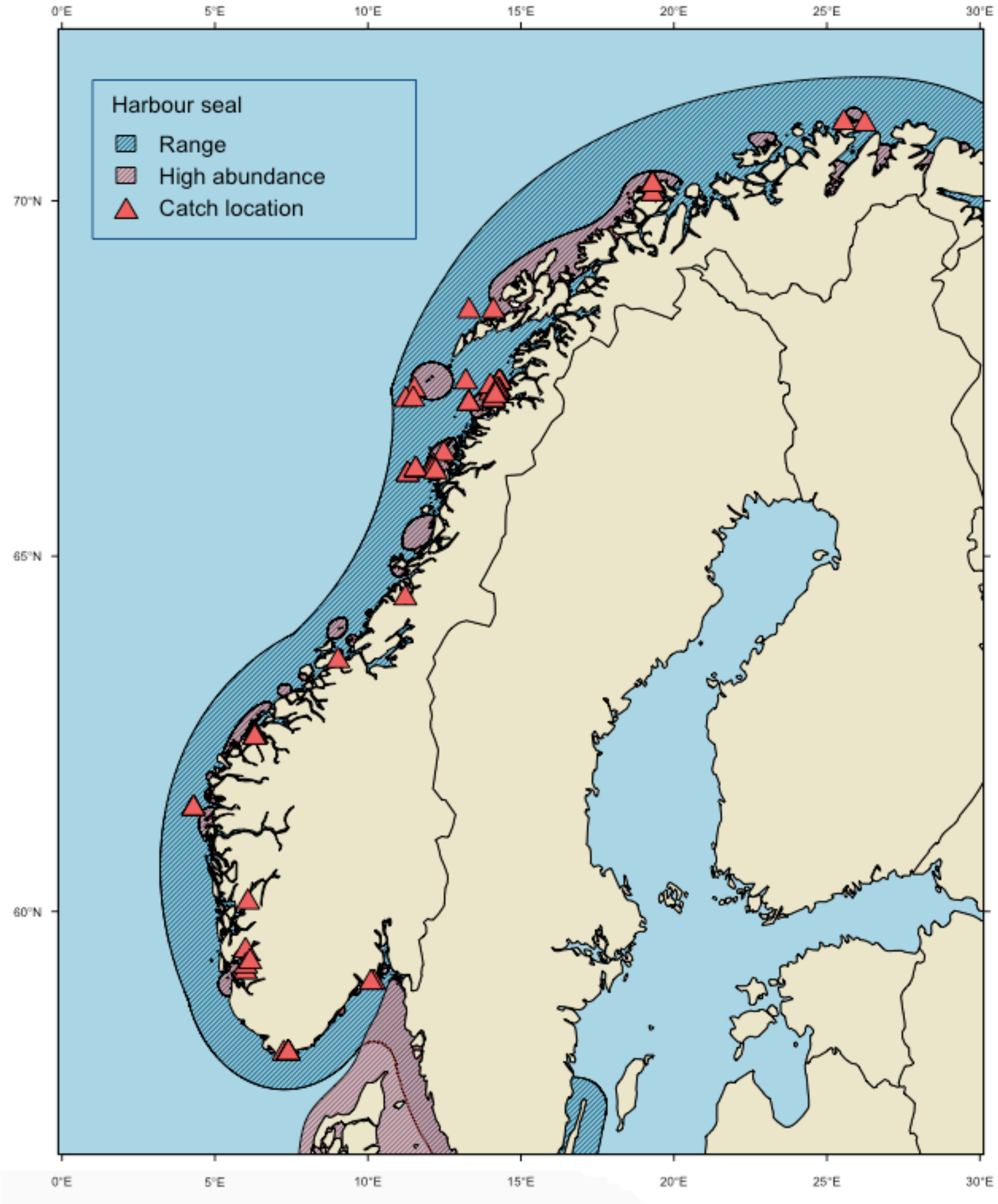

Figure 3.4: *Catch locations (red dots) for harbour seals (N=78) in the combined cod and monkfish fisheries in the CRF in the period 2006 – 2015. The ranging distribution of the coastal harbour seals is designated with a dashed dark blue background. Known areas of high abundance are designated with dashed red. Note that several animals may have been taken at a single location, hence the mismatch between the number of locations and the total number of animals taken, as described in the text.*

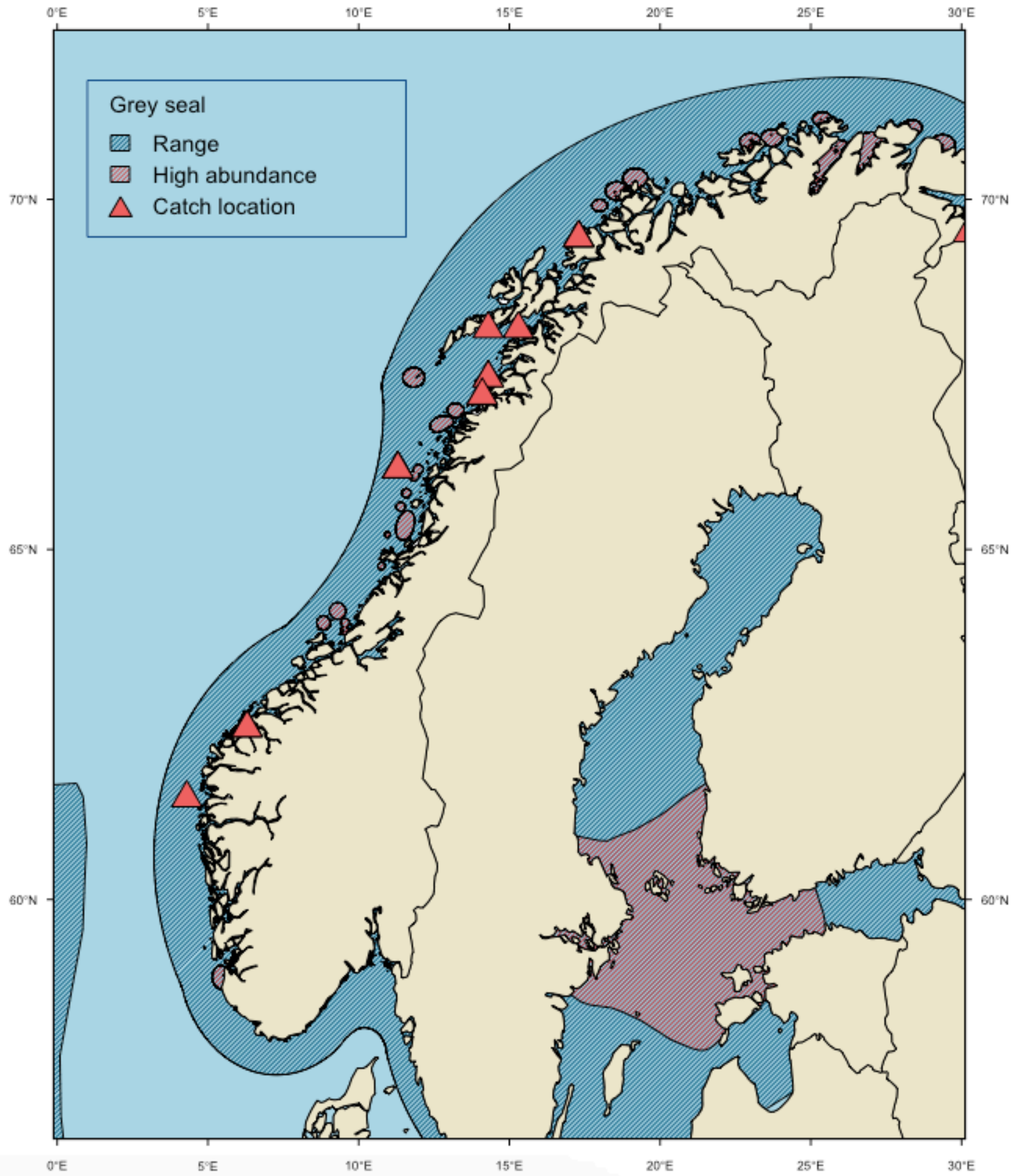

Figure 3.5*: Catch locations (red dots) grey seals (N=16) in the combined cod and monkfish fisheries in the CRF in the period 2006 – 2015. The ranging distribution of the coastal harbour seals is designated with a dashed dark blue background. Known areas of high abundance are designated with dashed red. Note that several animals may have been taken at a single location, hence the mismatch between the number of locations and the total number of animals taken, as described in the text.*

Bokna fjord (*Boknafjorden*, in Norwegian), outside Rogaland/Stavanger (approx. 59°N). The remaining catch locations were scattered along the coastline at 100 – 200km intervals. A number of the harbour seal catch locations were in close proximity to known areas of high seal abundance. In the Lofoten area, where the most harbour seals were taken in the CRF, for example, there are three known high abundance harbour seal areas. The Bokna fjord itself also comprises a high abundance harbour seal area. But by visual inspection of the maps, there seems to be little correlation between the size (in terms of area) of these high abundance regions and the number of seals taken nearby. The biggest known harbour seal high abundance region (centered at approximately 69°N 16°E, stretching from mid-Hinnøy to the north of Kvaløya), for example, only has four bycatch locations nearby. Figure 3.4 also shows that many of the takes of harbour seals occurred in fjords, in some cases many kilometers away from the open ocean. The 20 grey seals that were taken in the CRF in the ten-year period were caught at only nine different locations. Most grey seal takes occurred in the Lofoten area. The southernmost takes occurred outside of Sogn og Fjordane county (62°N). Takes also occurred outside of Senja (69°N) and in Varangerfjorden, in the far northeast (69°N 30°E). Visual inspection of Figure 3.5 suggests that the tendency for grey seals to be taken near known high abundance grey seal areas was less prominent than for harbour seals.

GIS analyses show that only 36 of the 129 harbour seals in the cod and monkfish fisheries were taken in high abundance harbour seal regions. A further 80 harbour seals were taken less than 20.0 km away from the nearest high abundance region (Figure 3.6, left panel). The remaining 13 harbour seals were taken at sites more than 20.0 km away from the nearest high abundance region. The average distance of harbour seal catch locations from the nearest high abundance region (not including catches taken inside these regions) was 16.5 km. GIS analyses therefore support the notion that most harbour seal takes occurred near these high abundance regions. Grey seal high abundance regions are much sparser and less expansive in area than harbour seal high abundance regions. On average, grey seal catches occurred farther away from the grey seal high abundance regions, as shown in the right panel in Figure 3.6. Only two grey seals were taken inside known high abundance areas. Another further seven animals were taken at sites less than 20.0 km away from the nearest high abundance region. The average distance of grey seal catch locations from the nearest high abundance area was 41.5 kilometers. In the most extreme case, two grey seals were taken during a single haul at a fishing station 122.0 km away from the nearest grey seal high abundance region. The temporal and spatial variation in the bycatch of these three species in the cod and monkfish fisheries are illustrated in Figure 3.7.

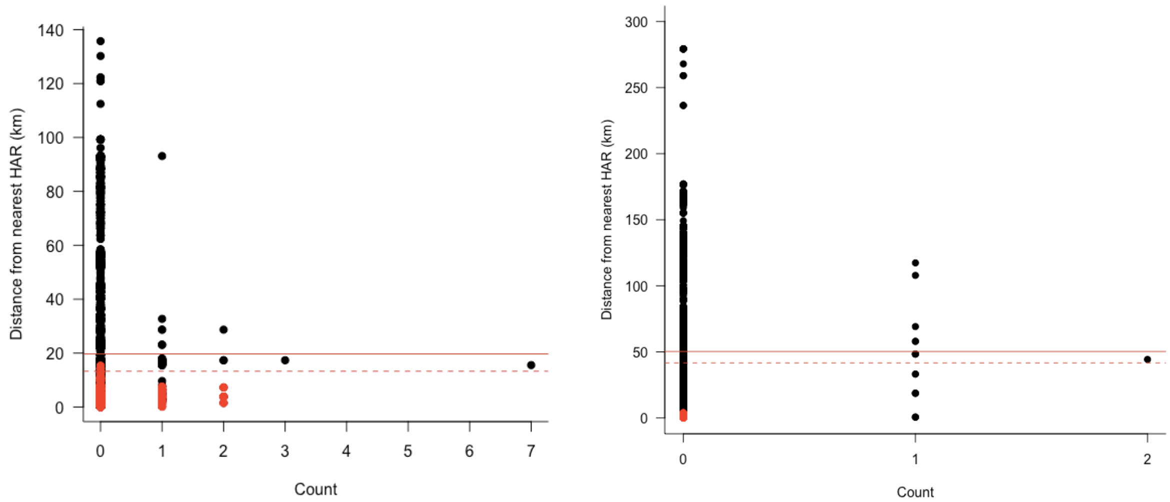

Figure 3.6*: Number of seals taken at each fishing station plotted against the distances of fishing stations to the nearest harbour seal (left panel) and grey seal (right panel) high abundance region (HAR). Red dots represent stations located inside HARs. The solid red horizontal line represents the average distance of fishing stations to the nearest HAR. The dashed line represents the average distance of fishing stations at which bycatch of harbour porpoise occurred to the nearest HAR.*

The number of bycaught harbour porpoise (Panels A + B, Figure 3.7) seems to coincide with the most intense fishing periods. Harbour porpoise bycatch was greatly elevated in February-March, when the cod fishery is at its most intense (Figure 3.1, top left), and in September-October, when the corresponding peak in the monkfish fishery occurs (Figure 3.1, top right), with intermediate levels in between. Harbour seal bycatch (Panels C + D, Figure 3.7) was highest in the second half of the year, reaching a peak in November and veering off towards the end of year through to April, before starting to rise again. This indicates that the majority of harbour seal bycatch occurred in the monkfish fishery, a conjecture supported by the fact that 87% of harbour seals were taken in monkfish gears. Grey seal bycatch (Panels E + F, Figure 3.7) varied less from month to month over the course of a year, and was greatest in June and October. Areas 07 and 00 were both associated with elevated levels of harbour porpoise and harbour seal bycatch. There were intermediate levels of harbour porpoise and harbour seal bycatch in adjacent areas, and comparatively low levels in areas further north and south. Grey seal bycatch was similarly slightly elevated in areas 00 and 06, and decreasing with increasing distance away from these areas.

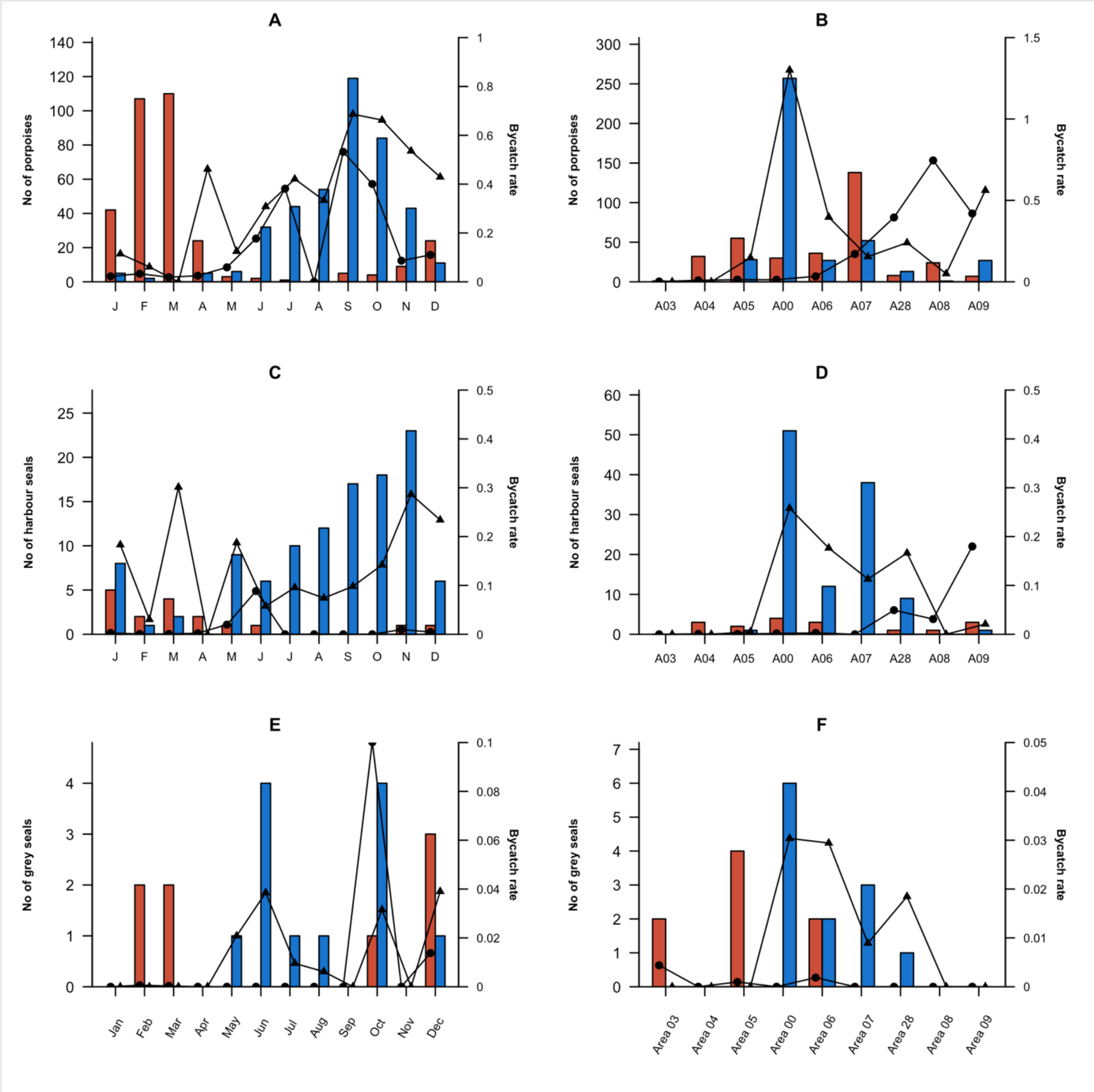

***Figure 3.7**: TOP PANELS (A + B): Bycatch of harbour porpoise in cod (red bars) and monkfish (blue bars) fisheries and associated catch rates (circles for the cod fishery and triangles for monkfish fishery, N animals × tons landed fish^{-1}) by month (left) and area (right). MIDDLE PANELS (C + D): Corresponding figures for harbour seal bycatch. BOTTOM PANELS (E + F): Corresponding figures for grey seal bycatch.*

3.3 CORRELATION OF BYCATCH AND FISHING EFFORT

The relationships between the number of takes of harbour porpoises, harbour seals and grey seals and fish landed, the number of fishing trips (TRIPS) and the number of gears used (GEARS) are outlined in Tables 3.1 – 3.3 and Figures 3.8 – 3.11. Only correlations for takes of harbour porpoise and the effort candidate variables are shown graphically; correlation results for the seals have been tabulated in Tables 3.2 and 3.3. Figure 3.8 shows the relationship between HP and CATCH after summing these variables by strata, so that each data point represents the total catch and bycatch in one specific year + month + area combination, resulting in a

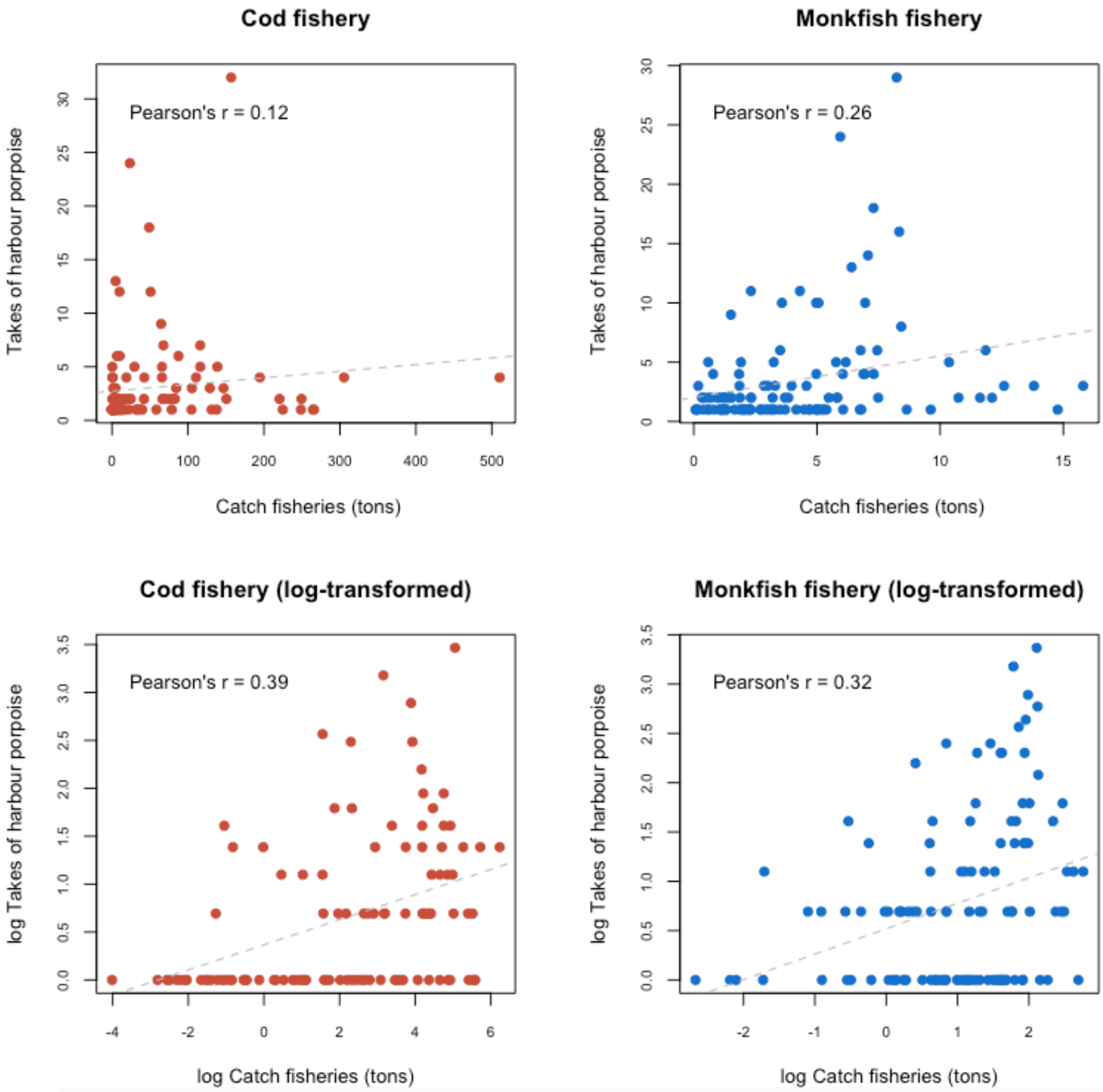

***Figure 3.8**: Correlation between takes of harbour porpoise and catch in the cod (red dots, left panels) and monkfish (blue dots, right panels) fisheries in the CRF before (upper panels) and after log-transformation (lower panels). Each point represents one of the 1080 year + month + area combinations in the CRF data. The dashed grey lines are lines of best fit.*

total of 1080 data points for each fishery. The upper panels show the correlation between monkfish landings (Pearson's r = 0.32, p << 0.001, Figure 3.8, bottom panels). The correlation improved drastically when aggregating data by month (n=12 combinations), and by year and month (n=120 combinations), yielding in all cases a highly significant correlation between the raw, summed catch and bycatch values, while the lower panels show the correlation between the log-transformed values. Takes of harbour porpoise in the CRF were significantly correlated both with cod (r = 0.39, p << 0.001, Figure 3.8, top panels) and catch and takes of harbour porpoise in the cod and monkfish fisheries (Table 3.1). Areal totals, however, did not yield significant correlations (Table 3.1). Figure 3.10 and Figure 3.11 show the relationship between takes of harbour porpoise and other fishing effort candidate variables(fishing trips

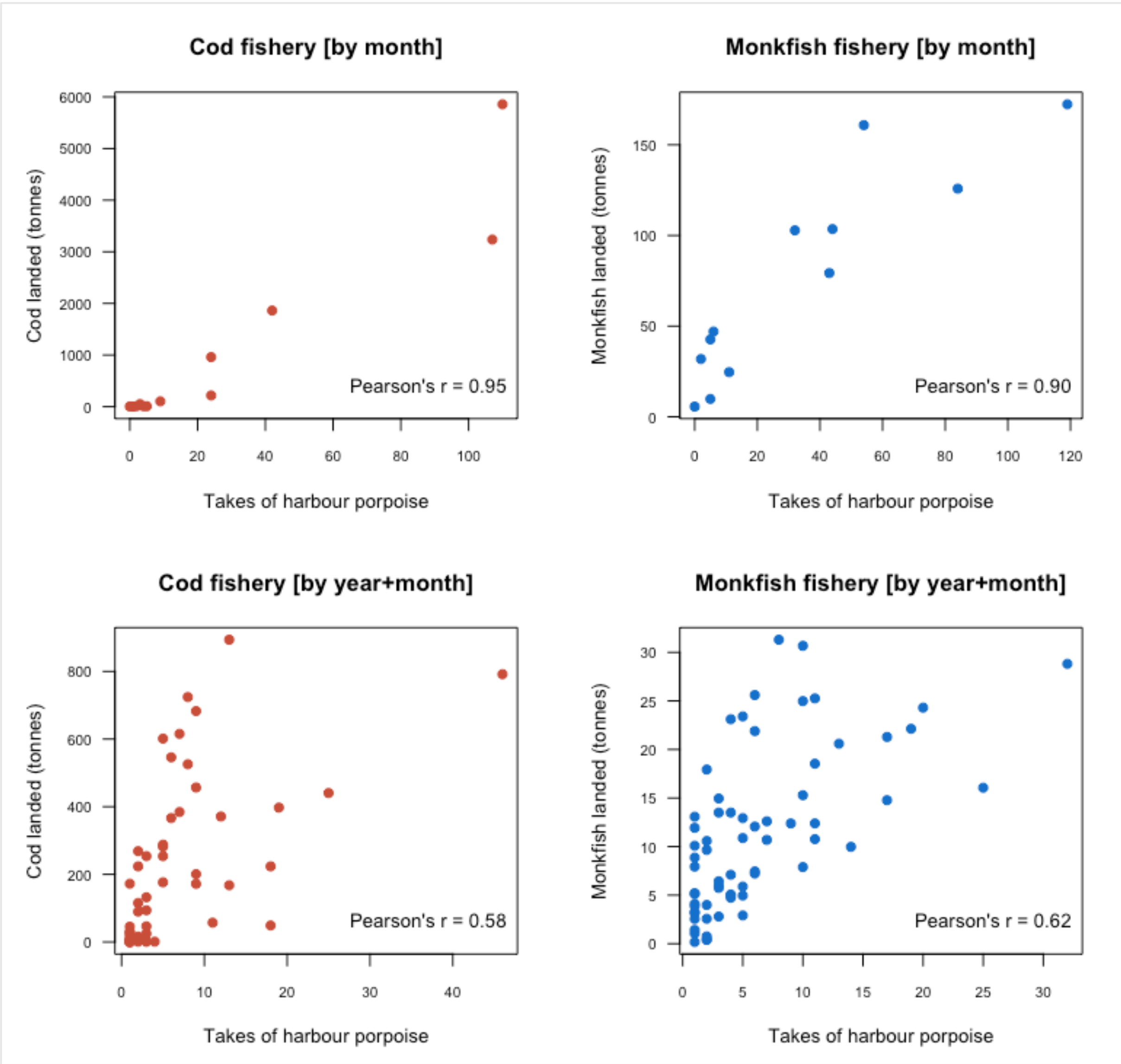

Figure 3.9*: Correlation between takes of harbour porpoise and catch in the cod (red dots, left panels) and monkfish (blue dots, right panels) fisheries after pooling by month (top) and year + month (bottom) combinations.*

and number of gears, respectively). Takes of harbour porpoise were significantly correlated with trips in the cod fishery (Pearson's r = 0.42, p << 0.001), but not in the monkfish fishery (Pearson's r = 0.01, p = 0.94). Takes of harbour porpoise was significantly correlated with the number of gears used in the cod (Pearson's r = 0.34, p << 0.001) and the monkfish (Pearson's r = 0.21, p < 0.05) fisheries.

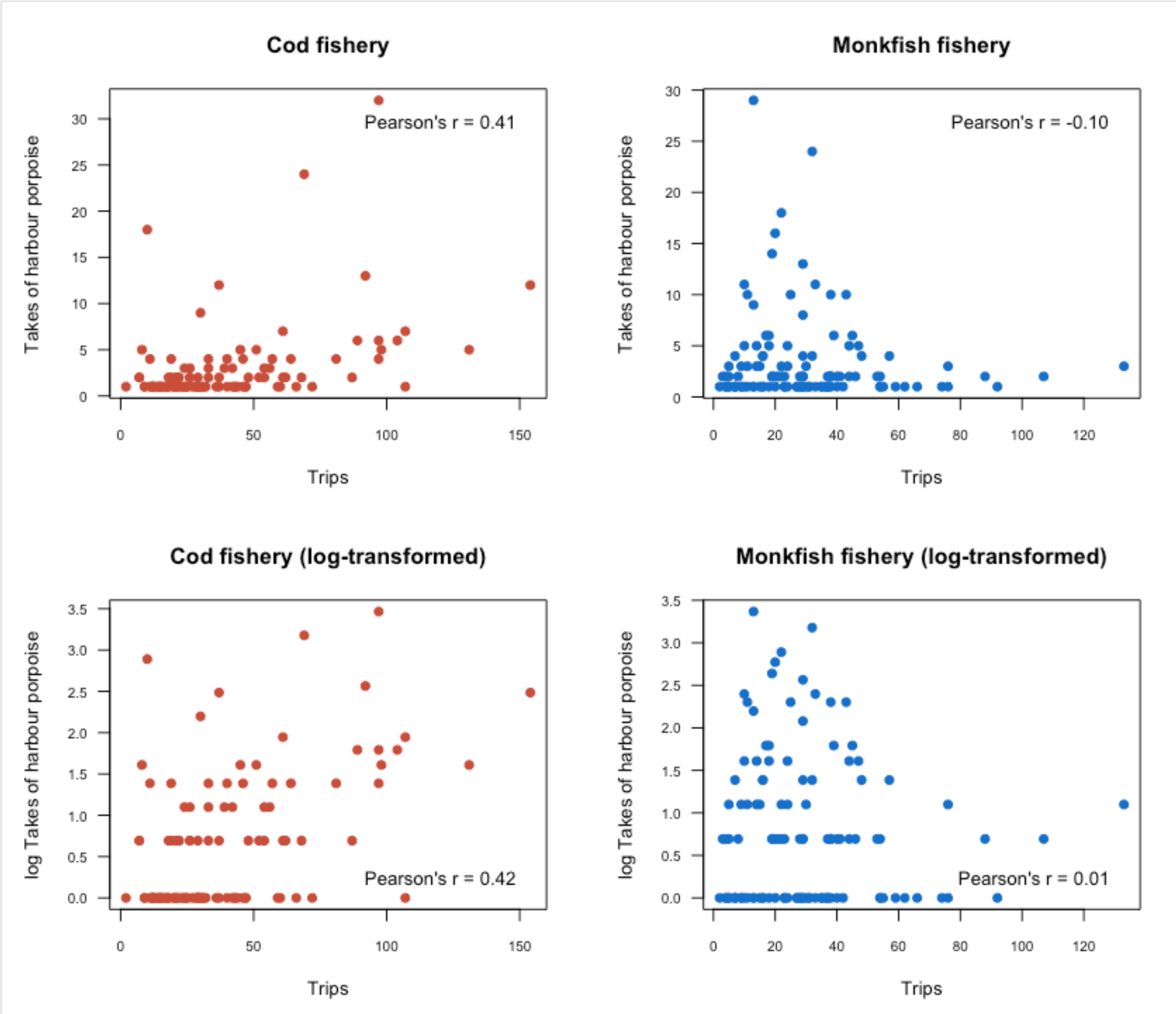

Figure 3.10*: Correlation between takes of harbour porpoise and the number of fishing trips in the cod (red dots, left panels) and monkfish (blue dots, right panels) fisheries in the CRF before (upper panels) and after log-transformation (lower panels).*

Harbour seal bycatch was significantly correlated with monkfish landings (Pearson's r = 0.39, p << 0.001) but the correlation with cod landings was very weak (Pearson's r = 0.16, p << 0.001). The correlation between takes of harbour seal and the number of fishing trips was weak, but significant in both cod (Pearson's r = 0.11, p << 0.001) and monkfish (Pearson's r = 0.12, p << 0.001) fisheries. There were no significant correlations between takes of harbour seal and the number of gears used in either fishery in any of the aggregation schemes. A number of correlation tests between the number of harbour seal takes and the number of gears used resulted in *negative* correlations (Table 3.2). Grey seal bycatch was significantly, but very weakly correlated with monkfish catches (Pearson's r = 0.15, p << 0.001, but not significantly correlated to cod catches (Pearson's r = 0.05, p = 0.099). The correlation between grey seal bycatch and the number of fishing trips was slightly better, but not by much Table 3.3). Only when aggregating by year + month + area and using log-transformed data was the correlation significant at the 0.05 level or better for both fisheries.

Figure 3.11*: Correlation between takes of harbour porpoise and the number of gears used in the cod (red dots, left panels) and monkfish (blue dots, right panels) fisheries in the CRF before (upper panels) and after log-transformation (lower panels).*

Table 3.1*: Overview of the correlations between harbour porpoise (HP) bycatch and fishing effort candidate variables. The letters in the "Aggregation schemes" column represent year, month and area. Asterisks represent significance codes, with *** indicating p << 0.001, ** indicating p < 0.01, * indicating p < 0.05 and the absence of an asterisk indicating no significance.*

Relationship tested	Aggregation scheme	Pearson's r (cod fishery)	Pearson's r (monkfish fishery)
HP vs CATCH	Y + M + A	0.12 ***	0.26 ***
	Y + M	0.58 ***	0.62 ***
	M	0.95 ***	0.90 ***
	A	0.20	0.49
log(HP) vs log(CATCH)	Y + M + A	0.39 ***	0.32 ***
	Y + M	0.68 ***	0.53 ***
	M	0.95 ***	0.89 ***
	A	0.48	0.83 **
HP vs TRIPS	Y + M + A	0.41 ***	-0.10
	Y + M	0.47 ***	-0.12
	M	0.77 **	-0.17
	A	0.71 *	0.29
HP vs GEARS	Y + M + A	0.27 **	0.11
	Y + M	-0.07	-0.06
	M	-0.32	0.82 **
	A	-0.28	-0.20

Table 3.2*: Overview of the correlations between harbour seal (HS) bycatch and fishing effort candidate variables. The letters in the "Aggregation schemes" column represent year, month and area. Asterisks represent significance codes, with *** indicating p << 0.001, ** indicating p < 0.01, * indicating p < 0.05 and the absence of an asterisk indicating no significance.*

Relationship tested	Aggregation scheme	Pearson's r (cod fishery)	Pearson's r (monkfish fishery)
HS vs CATCH	Y + M + A	0.28 ***	0.37 ***
	Y + M	0.39 ***	0.48 ***
	M	0.74 **	0.69 *
	A	0.44	0.75 *
log(HS) vs log(CATCH)	Y + M + A	0.16 ***	0.39 ***
	Y + M	0.33 ***	0.41 ***
	M	0.91 ***	0.80 **
	A	0.19	0.64
HS vs TRIPS	Y + M + A	0.11 ***	0.12 ***
	Y + M	0.23 **	-0.10
	M	0.57	-0.18
	A	-0.09	0.56
HS vs GEARS	Y + M + A	-0.01	-0.01
	Y + M	-0.05	-0.07
	M	-0.45	0.44
	A	-0.28	-0.13

***Table 3.3**: Overview of the correlations between grey seal (GS) bycatch and fishing effort candidate variables. The letters in the "Aggregation schemes" column represent year, month and area. Asterisks represent significance codes, with *** indicating p << 0.001, ** indicating p < 0.01, * indicating p < 0.05 and the absence of an asterisk indicating no significance.*

Relationship tested	Aggregation scheme	Pearson's r (cod fishery)	Pearson's r (monkfish fishery)
GS vs CATCH	Y + M + A	0.05	0.08 **
	Y + M	0.11	0.11
	M	0.49	0.37
	A	0.46	0.62
log(GS) vs log(CATCH)	Y + M + A	0.07 *	0.14 ***
	Y + M	0.19 *	0.14
	M	0.49	0.46
	A	0.45	0.52
GS vs TRIPS	Y + M + A	0.00	0.00
	Y + M	0.00	-0.08
	M	0.17	-0.20
	A	-0.41	0.41
GS vs GEARS	Y + M + A	0.00	0.00
	Y + M	0.00	0.00
	M	-0.26	-0.06
	A	-0.31	-0.13

In summary, the correlations of takes of harbour porpoises and harbour seals and catch and trips were moderately to highly significant and therefore good candidates for use either as the basis for calculating a catch-per-effort measure or as predictors in a GLM/GAM estimation approach. Correlations for the number of gears used varied greatly under different aggregation schemes, and were mostly not significant. In the case of the grey seal data, catch seems to be the only feasible effort measure.

3.4 FREQUENCY DISTRIBUTION OF BYCATCH

Figure 3.12 shows the number of fishing trips that resulted in a bycatch of 0, 1, 2, ..., N animals, where N is the maximum number of animals taken in a single trip. There were 10,633 observations, with each data point corresponding to one fishing trip. The distribution is clearly asymmetrical and positively skewed. The great majority of trips did not result in any bycatch. Most of the trips that did result in bycatch, only resulted in the taking of a single animal. Trips that resulted in increasing number of takes were increasingly fewer. This general trend was the same for all the species discussed. There appeared to be a couple of outliers in the harbour porpoise data. Among the 527 fishing trips that resulted in porpoise bycatch, only

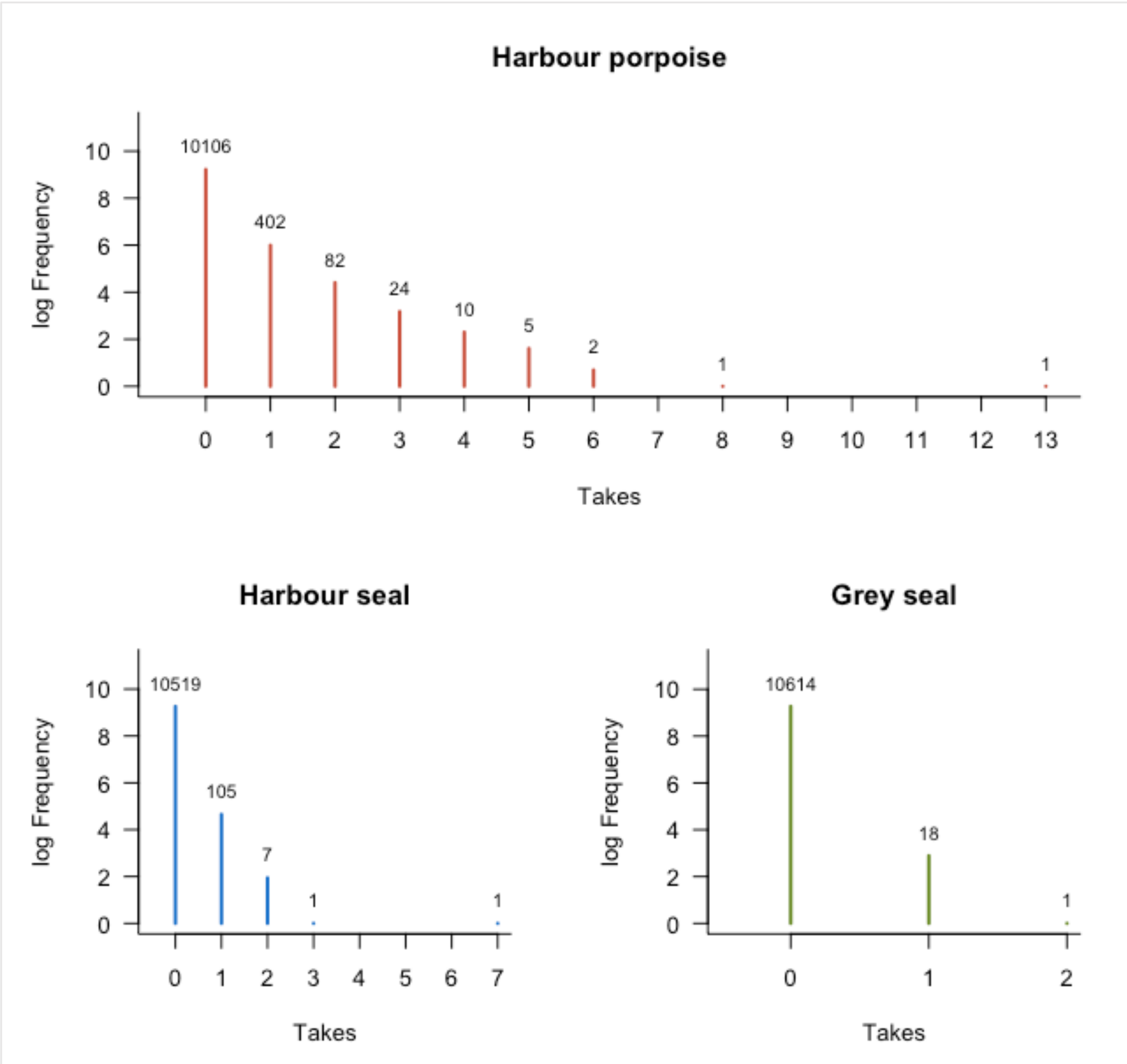

Figure 3.12*: Frequency distribution of bycatch in the unaggregated CRF data (n=10633) for each of the three species. Frequencies on the y-axes are log-transformed, but the numbers above each bar represent the absolute frequency for the corresponding number of takes, specified on the x-axes below the bars.*

two trips resulted in (relatively) extreme takes, and so we can consider the probability of getting extreme values to be fairly small. Figure 3.13 similarly shows the frequency of strata with an associated bycatch $x \in < 0, 1, 2, \ldots, N >$ animals in the aggregated CRF data set, where *N* is the maximum number of animals taken in a single stratum, excluding empty classes. A stratum here refers to one of the 2160 possible combinations of year, month, area and fishery in the aggregated data set, as described previously. Note that for increased clarity, the frequencies have been log-transformed. Otherwise, the plot would not be very informative, because there would only be one large bar representing trips with zero takes and horizontal lines a single pixel high representing the bars of other frequencies. But the numbers on top of each bar are the absolute frequencies.

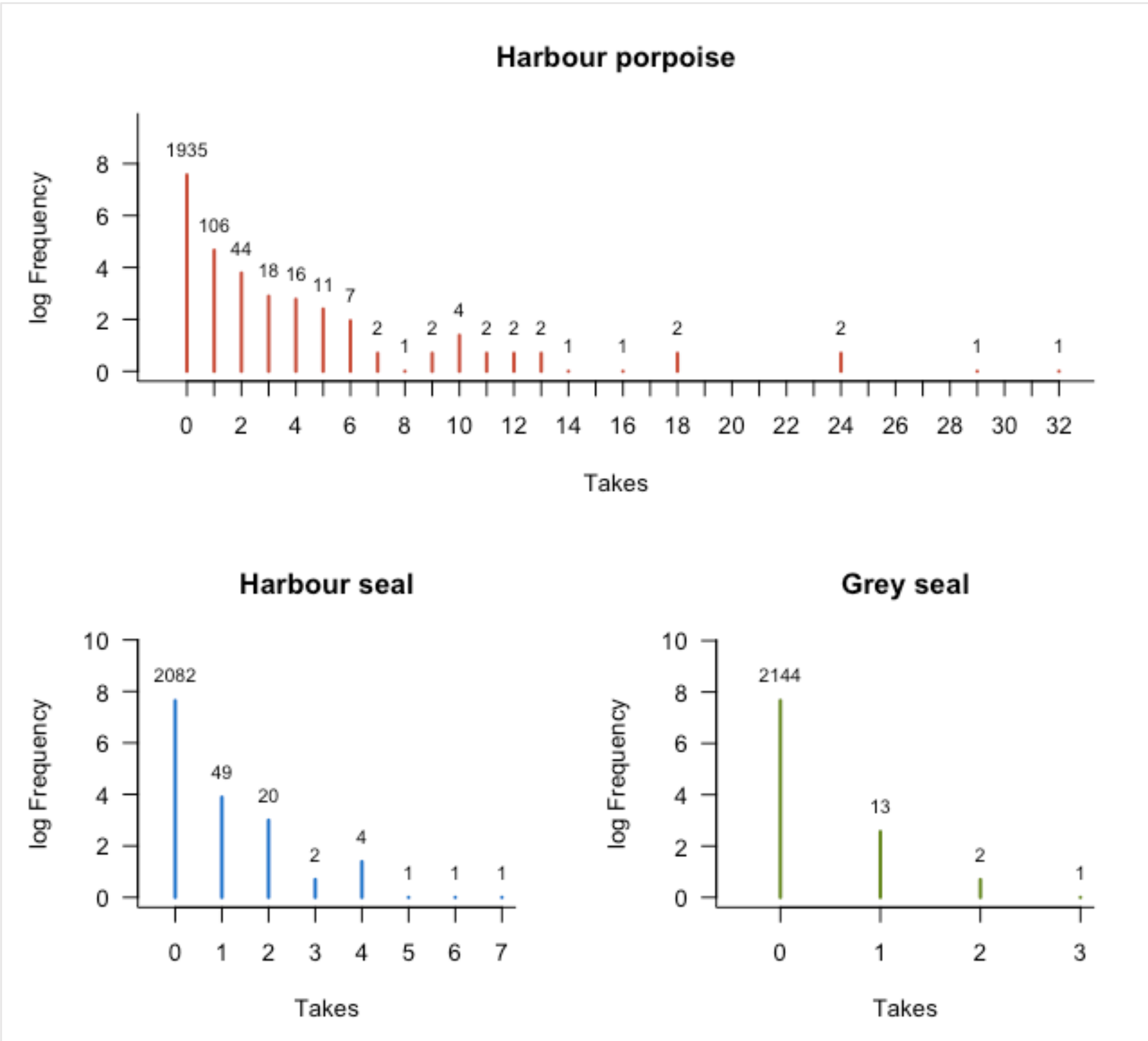

Figure 3.13*: Frequency distributions of bycatch in the aggregated CRF data (n=2160) for each of three species. Frequencies on the y-axes are log-transformed, but the numbers above each bar represent the asolute frequency for the corresponding number of takes, specified on the x-axes.*

3.5 STRATIFIED RATIO ESTIMATION

Table 3.4 summarizes annual bycatch estimates for the entire coastal fleet derived using the traditional stratified ratio estimation technique. The use of different stratification schemes (five in total) resulted in different absolute bycatch values, but all estimates for each of the species investigated were within the same order of magnitude and the 95% confidence intervals of these estimates were overlapping within each species group. Thus, using this method, porpoise bycatch is estimated to be in the range 2211 to 3218 animals. Similarly, national harbour and grey seal bycatch is estimated to be in the ranges 176 – 849 animals and 31 – 210 animals, respectively.

Table 3.4: *Stratified ratio bycatch estimates with associated CVs and 95% CIs for harbour porpoise, harbour seal and grey seal in the combined cod and monkfish fisheries for 2006 - 2016. All values represent averages over ten years of data, rather than a single year. Bycatch refers to yearly bycatch. Bycatch per unit effort is not shown, as it varies between strata.*

Species	**Stratification**	**Bycatch**	**CV**	**95% CI**
Harbour porpoise	Area × month	2211	0.16	1569 – 2926
	Region × month	2366	0.12	1854 – 2909
	Area	2313	0.27	1108 – 3671
	Month	3218	0.17	2279 – 4409
	Region	2347	0.14	1742 – 2952
Harbour seal	Area × month	459	0.24	265 – 677
	Region × month	482	0.22	296 – 700
	Area	502	0.34	176 – 849
	Month	565	0.18	387 – 775
	Region	463	0.35	189 – 811
Grey seal	Area × month	128	0.41	39 – 246
	Region × month	97	0.41	32 – 186
	Area	92	0.31	40 – 153
	Month	105	0.45	31 – 210
	Region	68	0.27	32 – 104

Some stratification schemes were associated with lower CVs than others. For the harbour porpoise, stratifying by region × month produced the lowest CV, of 0.12. For the harbour seal, the best stratification scheme was to stratify by month, yielding a CV of 0.18. The best stratification scheme for the grey seal was by month, yielding a CV of 0.27. The average value of all the CVs associated with the five estimates for each species, decreasing from grey seal

($\overline{CV} = 0.37$), through harbour seal ($\overline{CV} = 0.27$) and to harbour porpoise ($\overline{CV} = 0.17$), reflect the amount of bycatch data available for each of the modelled species. Figure 3.14 shows the predicted bycatch for months and areas for each of the three species based on the corresponding stratification schemes. Like the bycatch values given in Table 3.4, these predictions are averages over the entire period. Comparing the trends of the predictions in the left panels in Figure 3.14 (panels A + C + E) with the corresponding observed bycatch values, the predicted and observed monthly variations agree fairly well. This is a natural mathematical consequence of the high correlation of cod and monkfish catches in the CRF and the nation-wide fleet. Some months, especially towards the end of the year (from August and on) are associated with greatly increased bycatch of all three species. Generally speaking, bycatch can be expected to be higher during the fall and the winter. This seasonal pattern coincides with the seasonality of the cod and monkfish fisheries, and also with the presence of young of the year for all three species. Most animals were taken during the monkfish fishery season, which occurs in the second half of the year. The annual variation in predicted harbour porpoise bycatch looks distinctly bimodal, with most takes occurring in late winter/early spring and during the fall. There is a distinct drop in January and in May, which coincides well with low levels of fishery activities. There are takes of both seal species throughout the year, but the number of takes increase greatly after August. Areas 03, 06 and 07 are associated with the largest numbers of porpoise and harbour seal takes. In area 04 on the other hand, (and additionally, 05 and 00 for the harbour seal) there are comparatively much lower levels of bycatch of these two species. Grey seal bycatch however, is at its highest in Areas 03, 04, 00, 06 and 07. These areas all coincide with the areas along the Norwegian coast where the known abundance of grey seal is the highest, and where it is known to range extensively. In the other areas, there are virtually no takes of grey seal. Figure 3.14 does not show predicted bycatch per stratum for the other stratification schemes, because there are too many strata for such figures to be informational (9 areas × 12 months = 108 strata, 9 areas × 4 regions = 36 strata) and because those strata incorporate temporal and spatial dimensions together.

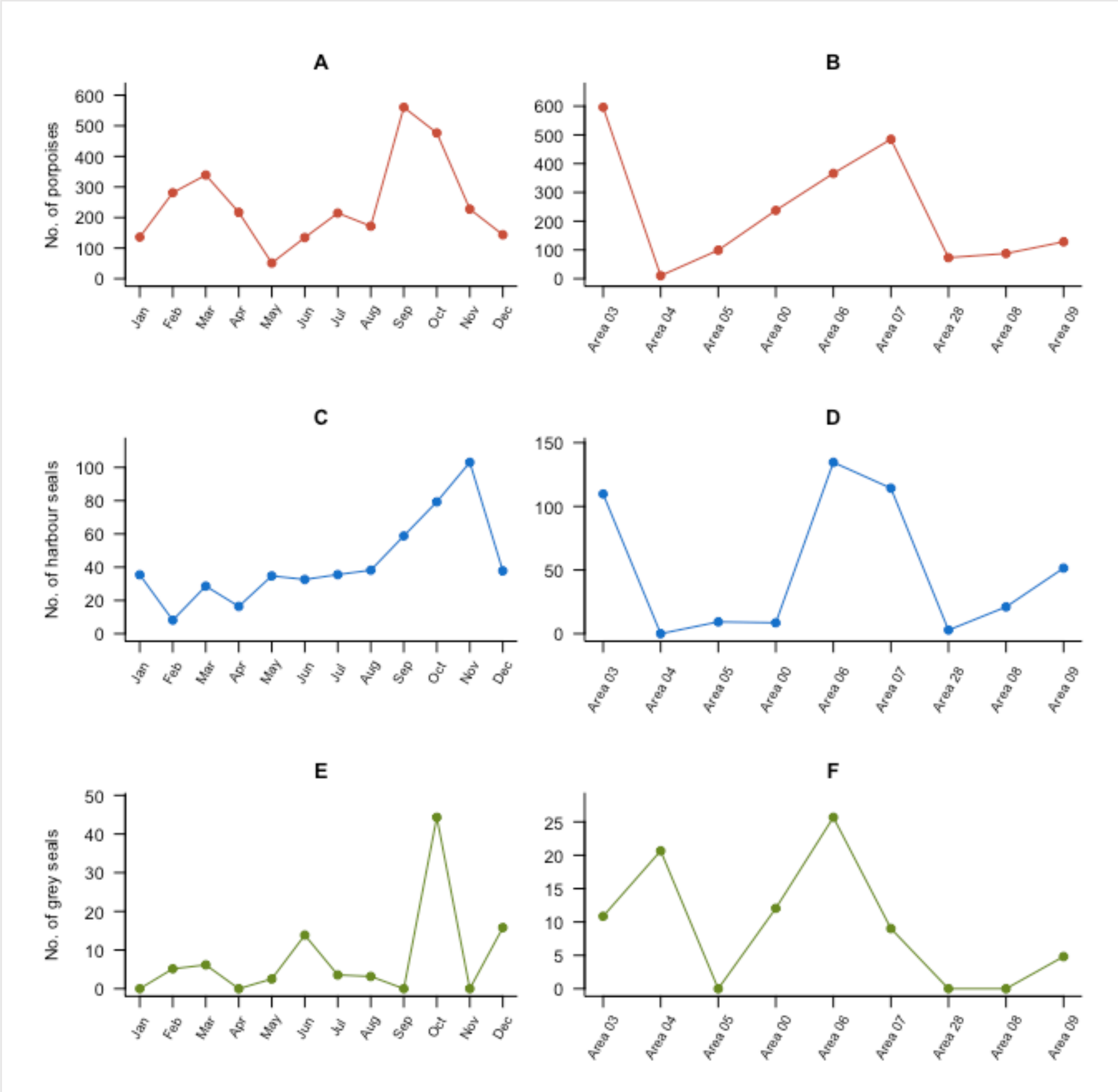

Figure 3.14*: Predicted total bycatch of harbour porpoise (top panels, A + B), harbour seal (middle panels, C + D) and grey seal (bottom panels, E + F) per month (left panels, A + C + E) and area (right panels + B + D + F) in the combined cod and monkfish fisheries, using bycatch rates derived using the traditional stratified ratio method.*

3.6 MODEL ESTIMATION

As reported in Section 3.3, bycatch was significantly correlated with catch in the CRF, although some particular correlations were very weak. The correlation between bycatch and catch was nonetheless stronger and more significant than the correlations of bycatch and other potential effort proxies, such as the number of trips undertaken, or the number gears used (Tables 3.1 – 3.3). Preliminary Poisson model runs showed that arbitrary CATCH based models were significantly better than models based on either TRIPS or GEARS, with a model AIC reduction of over 1000 units for one CATCH based model. Additionally, the unavailability of TRIPS/GEARS data in the commercial fishery supported the choice of CATCH as a measure of effort in the models.

Table 3.5*: Overview of correlations between bycatch and three different measures of effort along with AIC scores for basic GLM model based on these effort variables. The correlation column refers to Pearson's r.*

Species	Effort variable	Correlation	Model AIC
Harbour porpoise	CATCH	0.51 ($p << 0.01$)	2,771
	TRIPS	0.23 ($p << 0.01$)	3,868
	GEARS	0.07 ($p << 0.01$)	35,223
Harbour seal	CATCH	0.27 ($p << 0.01$)	724
	TRIPS	0.12 ($p << 0.01$)	902
	GEARS	0.04 ($p = 0.06$)	7,571
Grey seal	CATCH	0.11 ($p << 0.01$)	264
	TRIPS	0.04 ($p = 0.12$)	243
	GEARS	0.003 ($p = 0.90$)	1,011

Fit statistics for the model formulations listed in Table 2.3 are shown in Table 3.6 for the harbour porpoise (HP), Table 3.7 for the harbour seal (HS) and Table 3.8 for the grey seal (GS). Each table tabulates results from Poisson, zero-inflated Poisson and negative binomial models. The predicted bycatch column refers to yearly averages, calculated over the ten years of CRF data. The models are reported as GAMs, but only models with factor variables have been included in this manuscript. The models listed are thus functionally equivalent to GLMs. Coefficient estimates are not shown because of the number of models evaluated, but all main predictors used were statistically significant at the 0.05 level. Some particular combinations resulting from interaction terms, however, were not. The best Poisson models converged after

8 – 13 IWLS[1] iterations. The best zero-inflated Poisson and negative binomial models converged after only two iterations.

3.6.1 HARBOUR PORPOISE MODELS

Table 3.6 shows that the addition of further predictor variables to the base model including only the offset (model 1) resulted in drastic improvements to the fit statistics. The AICc scores were reduced by 2092 points (Poisson), 1523 points (ziP) and 158 points (NB) going from model 1 to model 13. Model 13 had the best (lowest) AICc score. Models with more predictors were also better able to account for the variation in the bycatch. Deviance explained for model 13 was > 0.5 for all model families. Expected bycatch from models with more predictors also exhibited a higher degree of correlation with observed bycatch. The CVs for the Poisson and ziP models had a tendency to decrease as predictors were added, resulting in CVs of 0.10 and 0.13, respectively, for the best models. The CVs for NB models 12 and 13 on the other hand were 1.14 and 1.08, respectively, but only 0.34 for model 11. The predicted yearly bycatch were considerably higher in ziP and NB models than in Poisson models. Predicted bycatch ranged from 2317 to 2556 animals (Poisson), 3479 to 4325 animals (ziP) and 4203 to 16745 animals (NB). NB models had lower AICc scores than corresponding ziP models, which in turn had lower AICc scores than corresponding Poisson models.

Comparing the variations in expected (as per the best models) and observed bycatch over months and areas (Figure 3.15), the Poisson model has the best agreement of the three model families (Pearson's r = 0.66 vs. 0.64 and 0.40). In Figure 3.15, panels A and B, we can see that bycatch estimates (red lines) roughly follow the CRF bycatch (black lines) without any extreme deviations. Deviations of note though are the months March and September, for which the Poisson overestimated and then underestimated bycatch by approximately 50% and 30%, respectively. The ziP and NB models had difficulties fitting the months February, March and September in particular. The best ziP model severely overestimated bycatch in March (by approximately 500%) and less severely, but still by about 200%, in areas 00 and 06. The NB curves are very similar to the Poisson curves, but the estimates in March and September are off by a larger amount.

[1] Iteratively reweighted least squares, the method used in a GLM/GAM to fit data

Table 3.6: *Harbour porpoise model fit statistics for the GAM formulations listed in Table 2.1, modelled using different statistical distributions. Predicted bycatch values refer to average annual bycatch. Cor is the correlation between observed data and model predictions.*

Model #	DF	Deviance explained	Scale	AICc	Predicted bycatch	Corr	CV	95% CI
		Poisson models						
1	1	0.00	1.70	4279.91	2556	0.18	0.14	2377 - 2747
2	4	0.21	1.35	3508.85	2609	0.20	0.18	2265 - 3007
3	2	0.33	1.15	3077.36	2629	0.33	0.11	2373 - 2910
4	2	0.30	1.20	3196.38	2988	0.28	0.13	2699 - 3306
5	3	0.36	1.10	2971.59	2726	0.37	0.11	2417 - 3079
6	5	0.41	1.01	2786.29	2561	0.34	0.12	2187 - 3000
7	5	0.36	1.09	2961.44	2823	0.37	0.10	2393 - 3340
8	5	0.45	0.94	2641.25	2956	0.38	0.11	2518 - 3473
9	7	0.45	0.93	2620.56	3096	0.38	0.10	2545 - 3774
10	6	0.46	0.93	2610.87	2787	0.39	0.12	2336 - 3333
11	7	0.46	0.92	2593.44	2857	0.40	0.12	2357 - 3481
12	9	0.57	0.74	2209.84	2244	0.66	0.09	1782 - 2849
13	**10**	**0.57**	**0.73**	**2187.59**	**2317**	**0.66**	**0.10**	**1809 - 2997**
		Zero-inflated Poisson models						
1	1	0.20	1.23	3492.22	4325	0.16	0.19	4039 - 4631
2	4	0.38	0.92	2846.30	5042	0.11	0.33	4455 - 5735
3	2	0.46	0.77	2535.88	4507	0.29	0.13	4050 - 5014
4	2	0.43	0.86	2695.93	4574	0.23	0.17	4114 - 5084
5	3	0.48	0.75	2486.12	4379	0.32	0.14	3851 - 4994
6	5	0.55	0.64	2268.95	4501	0.25	0.16	3827 - 5305
7	5	0.48	0.75	2484.44	4585	0.32	0.13	3831 - 5516
8	5	0.58	0.62	2195.31	4753	0.29	0.13	4026 - 5630
9	7	0.58	0.60	2178.98	5179	0.29	0.13	4220 - 6378
10	6	0.58	0.61	2177.64	4534	0.30	0.15	3763 - 5500
11	7	0.58	0.60	2168.33	4679	0.30	0.16	3817 - 5796
12	9	0.65	0.54	1989.41	3281	0.64	0.13	2527 - 4319
13	**10**	**0.65**	**0.52**	**1968.75**	**3479**	**0.64**	**0.14**	**2629 - 4670**
		Negative binomial models						
1	1	0.72	0.24	1821.11	16745	0.18	0.12	13635 - 20562
2	4	0.76	0.24	1711.68	13425	0.08	0.24	9135 - 19738
3	2	0.68	0.25	1809.93	9884	0.20	0.29	7468 - 13082
4	2	0.69	0.25	1797.90	10454	0.22	0.20	7960 - 13730
5	3	0.68	0.25	1797.39	9006	0.23	0.27	6604 - 12326
6	5	0.73	0.25	1707.67	9558	0.10	0.39	6248 - 14631
7	5	0.67	0.24	1791.39	9449	0.24	0.26	6197 - 14488
8	5	0.71	0.25	1685.39	7278	0.15	0.30	4855 - 10919
9	7	0.70	0.25	1683.07	6755	0.17	0.27	4240 - 10773
10	6	0.71	0.25	1687.42	7280	0.15	0.35	4746 - 11181
11	7	0.71	0.25	1687.40	7113	0.15	0.34	4548 - 11181
12	9	0.70	0.25	1667.26	4578	0.37	1.14	2758 - 7645
13	**10**	**0.70**	**0.25**	**1662.67**	**4203**	**0.40**	**1.08**	**2498 - 7135**

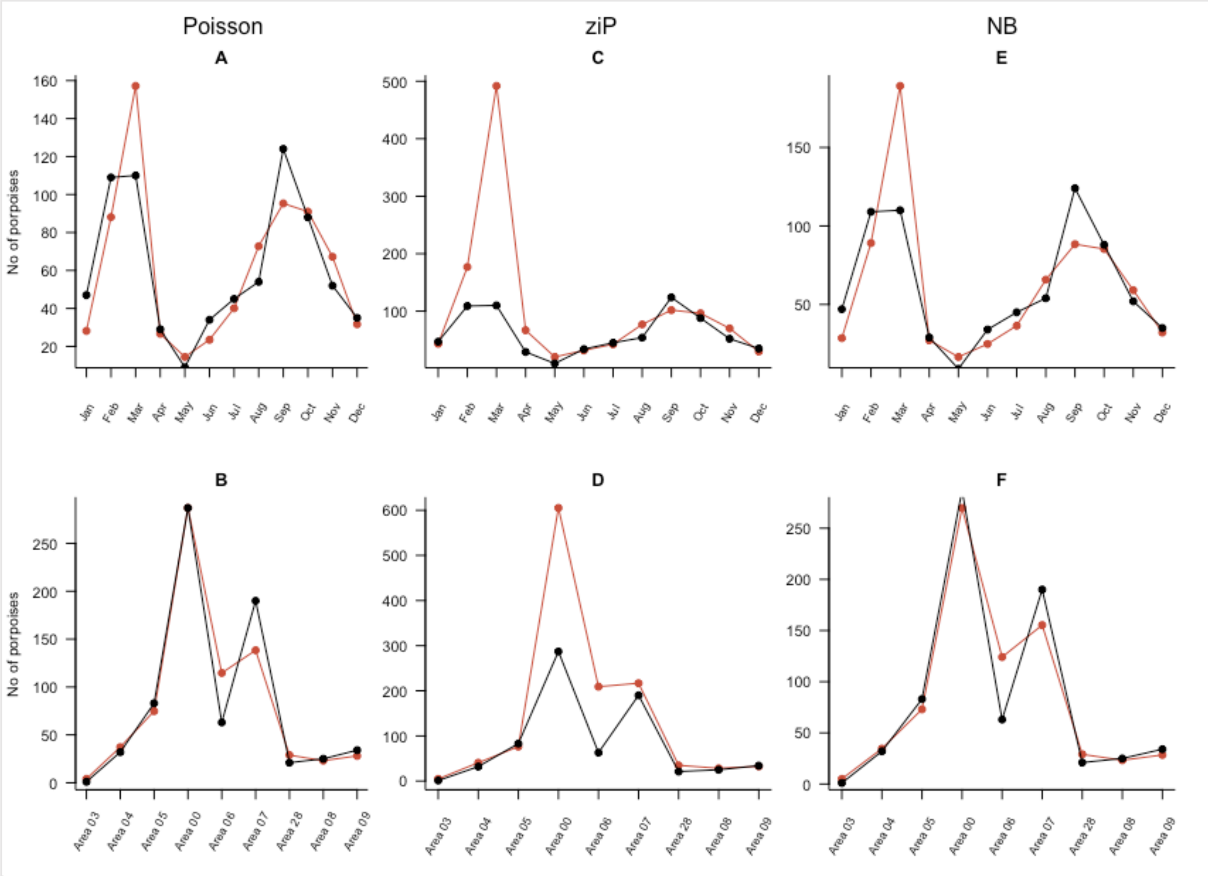

Figure 3.15: *Predicted (red dots) and observed (black dots) porpoise bycatch for month totals (above) and area totals (below), based on Poisson (Panels A and B), ziP (panels C and D) and NB (Panels E and F) model #13.*

3.6.2 HARBOUR SEAL MODELS

Harbour seal model fit statistics are given in Table 3.7. Again, models with more explanatory predictors had lower AICc values. The AICc score differences between the base models and the best models were 584 (Poisson), 327 (ziP) and 121 points (NB). In terms of AICc, model 12 was the best model, with an AICc of 650 (Poisson), 618 (ziP) and 621 points (NB). Comparing each model across the three model families, the NB models had the lowest AICc scores, followed by the ziP models and finally the Poisson models. The best models could explain 57% (Poisson), 82% (ziP) and 55% (NB) of the deviance. The correlation between observed and predicted bycatch was 0.50 (Poisson), 0.47 (ziP) and 0.49 (NB). Predicted bycatch for the best models were 424 animals (Poisson), 600 animals (ziP) and 483 animals (NB). CVs for the best model were 0.13 (Poisson), 0.31 (ziP) and 0.32 (NB). The best models in all three model families were able to capture very closely the spatial variation (i.e. variation between areas) in CRF bycatch. In Figure 3.8, the Poisson prediction and the observed bycatch curves are virtually overlapping for about half of the areas. In other areas, the Poisson slightly

Table 3.7: *Harbour seal model fit statistics for the GAM formulations listed in Table 2.1, modelled using different statistical distributions. Predicted bycatch values refer to average annual bycatch. Cor is the correlation between observed data and model predictions. The best model is highlighted in bold.*

Model #	DF	Deviance explained	Scale	AICc	Predicted bycatch	Corr	CV	95% CI
		Poisson models						
1	1	0.00	0.49	1233.61	448	0.18	0.17	376 - 532
2	4	0.20	0.39	1024.92	454	0.20	0.24	330 - 634
3	2	0.48	0.25	733.55	469	0.34	0.13	377 - 588
4	2	0.28	0.35	937.75	547	0.27	0.17	433 - 691
5	3	0.48	0.25	734.30	472	0.35	0.13	360 - 625
6	5	0.54	0.23	678.93	424	0.46	0.13	296 - 619
7	5	0.49	0.25	722.36	494	0.36	0.12	346 - 717
8	5	0.43	0.28	786.01	531	0.42	0.14	369 - 775
9	7	0.47	0.26	752.58	602	0.41	0.15	386 - 950
10	6	0.54	0.23	678.51	433	0.47	0.13	291 - 659
11	7	0.54	0.22	675.25	443	0.47	0.13	288 - 738
12	**9**	**0.57**	**0.21**	**649.81**	**424**	**0.50**	**0.13**	**273 - 715**
13	10	0.57	0.21	650.89	426	0.50	0.13	269 - 761
		Zero-inflated Poisson models						
1	1	0.36	0.26	945.33	1394	0.18	0.60	1142 - 1700
2	4	0.54	0.18	787.64	1932	0.25	0.66	1493 - 2629
3	2	0.75	0.15	648.48	966	0.33	0.21	739 - 1295
4	2	0.59	0.18	756.20	1714	0.18	0.26	1326 - 2234
5	3	0.75	0.15	648.33	963	0.33	0.19	693 - 1388
6	5	0.80	0.15	624.66	649	0.46	0.32	406 - 1092
7	5	0.76	0.15	649.13	947	0.36	0.21	600 - 1565
8	5	0.71	0.16	693.90	1125	0.33	0.64	724 - 1810
9	7	0.72	0.15	676.86	1382	0.31	0.48	810 - 2427
10	6	0.80	0.15	625.76	661	0.46	0.32	390 - 1179
11	7	0.80	0.15	627.16	657	0.46	0.33	385 - 1259
12	**9**	**0.82**	**0.15**	**618.02**	**600**	**0.47**	**0.31**	**358 - 1175**
13	10	0.82	0.15	620.26	599	0.46	0.31	354 - 1232
		Negative binomial models						
1	1	0.64	0.10	743.52	3906	0.18	0.22	2732 - 5585
2	4	0.66	0.11	682.07	2440	0.11	0.22	1348 - 4438
3	2	0.45	0.15	673.11	540	0.35	0.27	392 - 747
4	2	0.55	0.11	731.06	2085	0.25	0.41	1296 - 3355
5	3	0.45	0.15	673.08	564	0.36	0.21	380 - 842
6	5	0.50	0.14	641.65	501	0.43	0.29	301 - 848
7	5	0.46	0.15	669.65	584	0.37	0.30	344 - 1005
8	5	0.55	0.11	669.24	1142	0.24	0.27	607 - 2163
9	7	0.53	0.12	665.54	1055	0.28	0.30	502 - 2229
10	6	0.51	0.14	641.14	525	0.45	0.27	299 - 934
11	7	0.51	0.14	639.53	520	0.46	0.24	287 - 1002
12	**9**	**0.55**	**0.14**	**620.60**	**486**	**0.49**	**0.32**	**269 - 938**
13	10	0.55	0.14	622.13	483	0.49	0.49	261 – 981

overestimated or underestimated bycatch. ziP and NB model predictions also very closely agreed with the observed bycatch, but the deviations were slightly larger than those of the Poisson, especially in areas 00 and 07. The general temporal variation (monthly increase and decrease) in bycatch was captured by all models, but estimates diverged from observed bycatch values by varying amounts (from 5-50%). So predictions were fairly precise, but not very accurate. None of the models were able to capture the increased bycatch in January, flanked by lower bycatch levels in December and February.

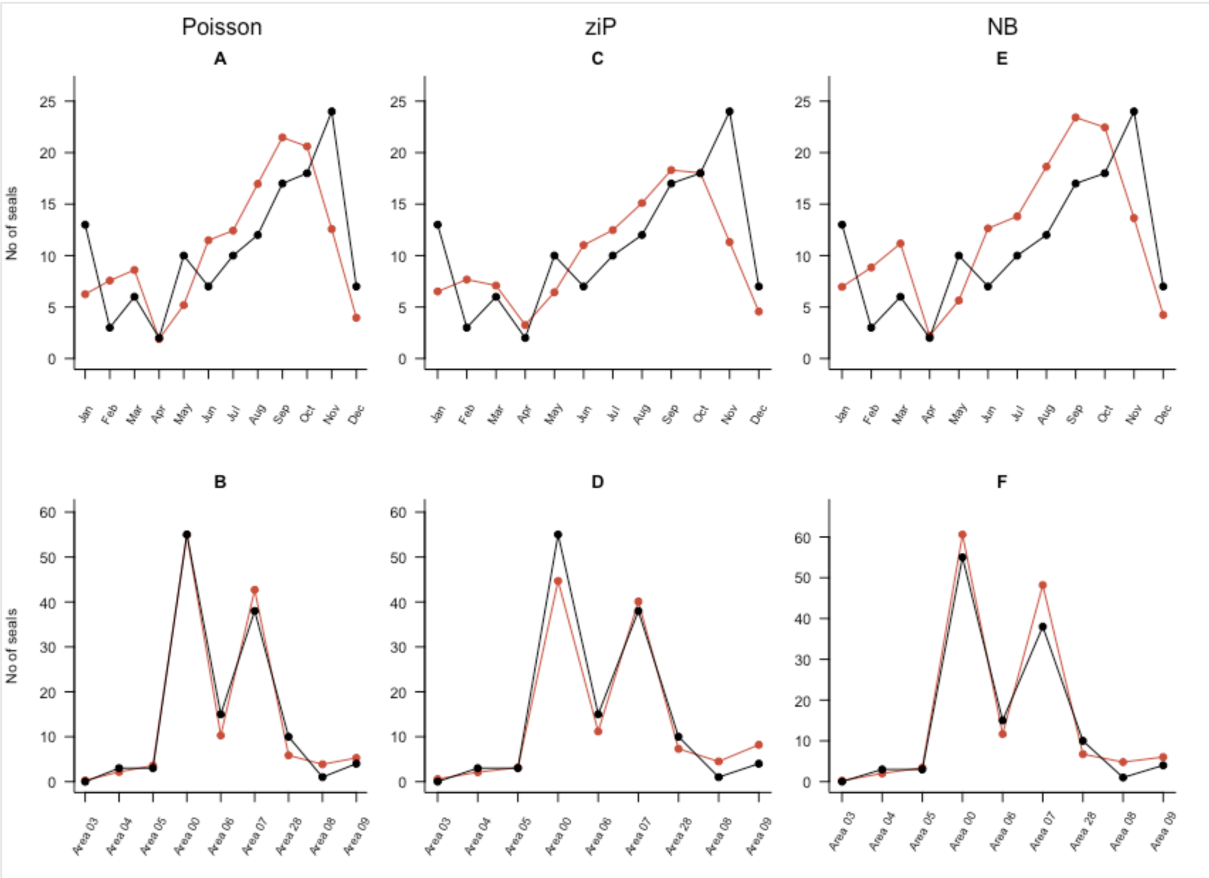

Figure 3.16: *Predicted (red dots) and observed (black dots) harbour seal bycatch for month totals (above) and area totals (below), based on Poisson (Panels A and B), ziP (panels C and D) and NB (Panels E and F) model #12.*

3.6.3 GREY SEAL MODELS

Table 3.8 lists the fit statistics for the grey seal models that were able to converge. Convergence was reached after 10 – 13 iterations. Models #12 and 13, which included the interactions fishery × region and fishery × season, are not listed, because they resulted in step failure during the iterative modelling process. We can see from the predicted bycatch numbers and their associated confidence intervals that there is a large spread in predictions between different models, especially for the ziP and NB models. The best model, in terms of deviance explained and AICc score, was model 11 for all model families. The predicted bycatch for model 11 was 83 animals (Poisson), 17517 animals (ziP) and 7329 animals (NB). The confidence intervals associated with these predictions were 26 – 287 animals, 14141 – 28751 animals and 1216 – 45187 animals.

There was very little agreement between grey seal model predictions and observed bycatch. Both the low degree of correlation (Pearson's r ranging from about 0.1 to 0.3) between observed and predicted bycatch, and the variation plots in Figure 3.17 show this very clearly. The models were unable to capture the variation in bycatch, as demonstrated by the disjoined bycatch curves for areas and months. There is a slight tendency of agreement for certain periods/models (e.g. January through to July for the Poisson model), but this is the exception rather than the rule. Some of the predictions were so far off that they could not fit in the plotting window without greatly distorting the remaining data points, rendering the figures completely uninformative. These extreme data points are indicated by the red arrows in Figure 3.17. Numbers under each arrow indicate the height of the curve at that value of x, i.e. the predicted bycatch for that month/area. While Poisson model predictions were within the same magnitude as the observed bycatch, we can see that both the ziP and NB models severely overestimated bycatch for particular months and areas. In the most extreme case, the NB model predicted a bycatch of 794 animals in December, when the observed bycatch was four animals. This corresponds to an overestimate of 19850%. The tendency to overestimate was the most severe for the northernmost areas (areas 03, 04, 05, 00 and 06) and the last quarter of the year, corresponding to the monkfish season.

Table 3.8: *Grey seal model fit statistics for the GAM formulations listed in Table 2.1, modelled using different statistical distributions. Predicted bycatch values refer to average annual bycatch. Cor is the correlation between observed data and model predictions. Models #12 and 13 missing because of insufficient data.*

Model #	**DF**	**Deviance explained**	**Scale**	**AICc**	**Predicted bycatch**	**Corr**	**CV**	**95% CI**
		Poisson models						
1	1	0.00	0.12	298.88	69	0.18	0.28	44 - 107
2	4	0.04	0.12	295.62	69	0.24	0.33	29 - 171
3	2	0.15	0.10	262.36	72	0.35	0.27	38 - 132
4	2	0.12	0.11	270.40	81	0.28	0.29	44 - 150
5	3	0.15	0.10	262.70	73	0.37	0.29	36 - 158
6	5	0.15	0.10	266.81	70	0.40	0.28	28 - 188
7	5	0.21	0.10	250.78	89	0.32	0.31	35 - 237
8	5	0.14	0.11	270.72	80	0.39	0.28	31 - 221
9	7	0.20	0.10	257.94	100	0.29	0.37	33 - 322
10	6	0.16	0.10	267.17	72	0.42	0.31	26 - 223
11	**7**	**0.24**	**0.09**	**248.96**	**83**	**0.31**	**0.36**	**26 - 287**
		Zero-inflated Poisson models						
1	1	0.42	0.05	230.37	856	0.20	NA	475 - 1538
2	4	0.46	0.04	224.73	2866	0.03	NA	1964 - 5060
3	2	0.47	0.04	219.67	808	0.37	NA	288 - 2292
4	2	0.46	0.04	222.66	1210	0.22	NA	557 - 2651
5	3	0.49	0.04	221.07	844	0.39	NA	282 - 2991
6	5	0.52	0.04	221.75	666	0.41	NA	142 - 3456
7	5	0.48	0.03	214.65	5574	0.08	NA	3592 - 13100
8	5	0.50	0.04	223.74	1227	0.16	NA	480 - 3999
9	7	0.49	0.03	214.47	27785	0.09	NA	22768 - 54829
10	6	0.52	0.04	223.30	705	0.45	NA	148 - 4448
11	**7**	**0.55**	**0.03**	**210.20**	**17517**	**0.09**	**NA**	**14141 - 28751**
		Negative binomial models						
1	1	0.53	0.03	229.58	3542	0.18	1.63	1352 - 9273
2	4	0.80	0.02	218.88	54186	0.08	1.10	12893 - 230225
3	2	0.69	0.02	226.84	27026	0.16	1.38	7534 - 96981
4	2	0.59	0.02	221.91	3653	0.25	1.28	1032 - 12958
5	3	0.59	0.02	223.86	3599	0.21	1.21	698 - 18965
6	5	0.80	0.02	220.89	50390	0.09	1.21	9623 - 266816
7	5	0.73	0.02	211.76	5029	0.11	1.17	894 - 28595
8	5	0.75	0.02	216.21	9607	0.17	1.05	1592 - 60212
9	7	0.81	0.02	209.58	8570	0.15	1.04	1262 - 61105
10	6	0.75	0.02	218.29	9728	0.17	1.11	1203 - 82019
11	**7**	**0.81**	**0.02**	**204.91**	**7329**	**0.10**	**1.06**	**1216 - 45187**

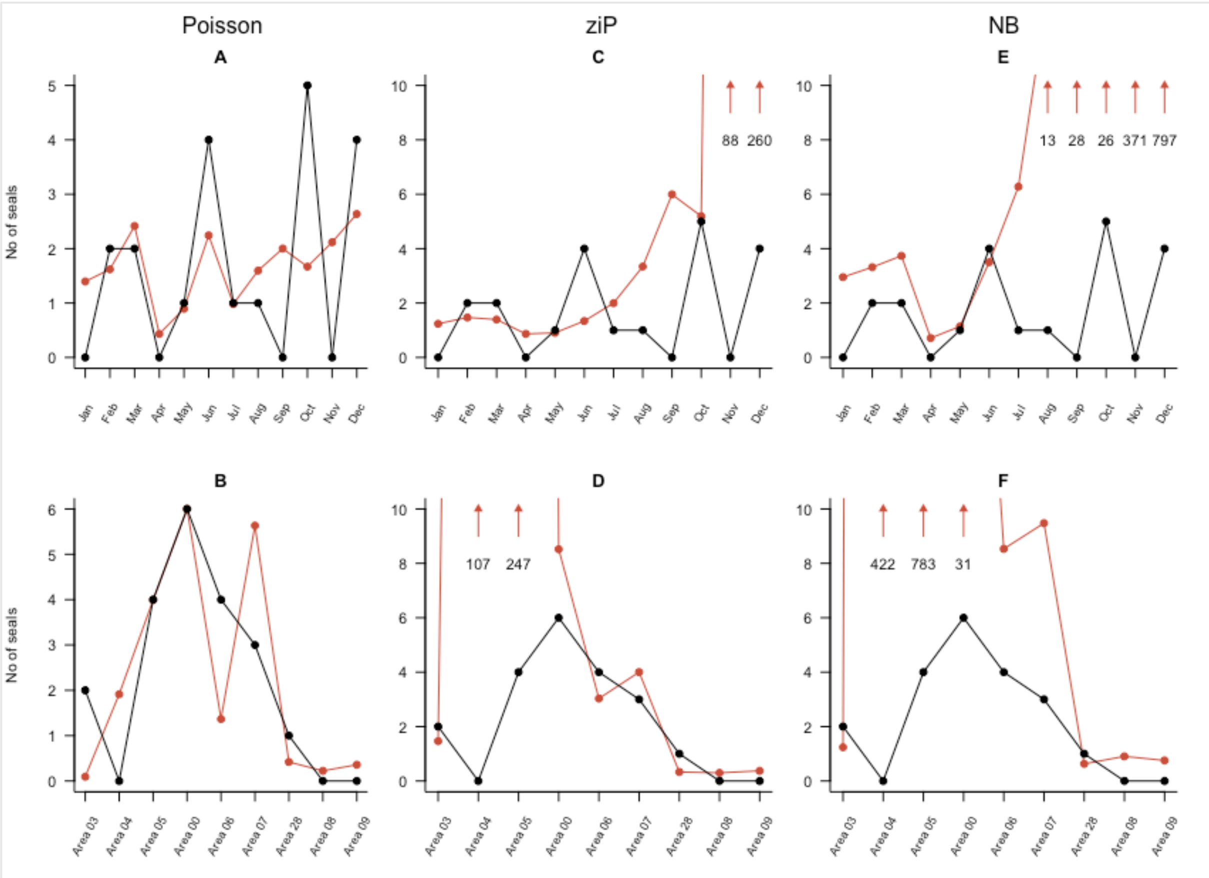

Figure 3.17: *Predicted (red dots) and observed (black dots) grey seal bycatch for month totals (above) and area totals (below), based on Poisson (Panels A and B), ziP (panels C and D) and NB (Panels E and F) GAM #11. Red arrows with numbers below indicate the values of predictions that did not fit into the plotting window.*

3.7. COMPARISON OF MODEL FAMILIES

The differences in fit statistics within and between model families have been summarized in Tables 3.6 through 3.8 in the section above for each of the species investigated. Using AICc scores as a comparison of the relative goodness of fit of the different nested models within each family, the best models were model 13 for harbour porpoise, model 12 for harbour seal and model 11 for grey seal. The best model formulations were thus:

HP ~ offset(LOG.CATCH) + FISHERY + SEASON + REGION + FISHERY : REGION + FISHERY : SEASON

HS ~ offset(LOG.CATCH) + FISHERY + SEASON + REGION + FISHERY : REGION

GS ~ offset(LOG.CATCH) + FISHERY + SEASON + REGION + FISHERY : SEASON

where *HP* corresponds to harbour porpoise, *HS* to harbour seal and *GS* to grey seal. These models consistently achieved the best AICc score among corresponding models from different distribution families. NB family models achieved the lowest AICc scores, followed in turn by ziP models and finally Poisson models (although scores for the best harbour seal ziP and NB models differed by only two points). While it may be statistically justifiable to use AICc scores to compare analogous models from different statistical families, aspects of the ziP and NB models fit statistics dictate that in this case, isolated AICc scores are not sufficient to determine the best model. The initial objection to a conventional Poisson-based regression was that a Poisson model may be overdispersed and/or zero-inflated.

Table 3.9 shows the results of the AER overdispersion test for the best Poisson models. The estimated c values are highly suggestive of overdispersion, especially in the case of the harbour porpoise and the grey seal models (for which the c is estimated to be approximately 34 and 86, respectively). However, the p-values associated with these estimates are both about three times above the 0.05 significance level. The null hypothesis that the models comply with Poisson variation therefore *cannot* be rejected on the basis of this test. The c statistic for the harbour seal model is similarly greater than, but much closer to 0 with an associated p-value < 0.05. In this case, the Poisson model can be rejected. The AICc scores of the harbour seal NB models, which are consistently lower than the corresponding Poisson models (sometimes by several hundred points), further add support to the added suitability of using an NB model for modelling harbour seal bycatch. Generally, the ziP and NB harbour seal models performed better than the corresponding harbour porpoise models in predicting bycatch.

Table 3.9: *Overdispersion estimates for the best models. Model numbers refer to those specified in Table 2.3. c is the test statistic. Overdispersion corresponds to c > 0 and underdispersion to c < 0. The z statistic is a standard t statistic under the null hypothesis.*

Model	Response	Estimated c	z	p-value
Poisson model #13	Harbour porpoise	34.01	1.09	0.1372
Poisson model #12	Harbour seal	0.32	1.65	0.0496
Poisson model #11	Grey seal	85.83	1.00	0.1583

Figure 3.18 shows rootograms generated based the best Poisson models for each species. In all the rootograms, the curves representing predicted frequencies closely track the observed frequencies. We can see that some bars extend slightly beyond the horizontal reference lines (where y = 0), but no consistent pattern is apparent. Other bars are raised, or “hanging” above the line. In the case of the harbour porpoise model, the data exhibited slightly too many zeroes, fours, fives and tens for a very good Poisson fit, but these deviations were very small. We can see that counts of one, two, three and seven through to nine were slightly overfitted. Underfitting for count 0 and overfitting for counts 1 and 2 is characteristic of data with excess zeroes, but this tendency was only very weak in the harbour porpoise rootogram. The harbour seal rootogram was very similar to that of the harbour porpoise, but the number of different count classes was much smaller. Counts of zero, two and four were underfitted, while counts of one and three were overfitted. But as with the harbour porpoise model, this tendency was very weak. In the grey seal rootogram, we can see a perfect fit for zeroes, a slight overfit for count 1 and slight underfits for counts 2 and 3.

Figure 3.19 shows QQ-plots for the residual deviance of the best Poisson, ziP and NB harbour porpoise (top), harbour seal (middle) and grey seal (bottom) models. QQ-plots provide a visual comparison of sample quantiles to the corresponding theoretical quantiles. Departures from linearity indicate that distributional assumptions are violated. Thus, if the model distributional assumptions are met, then the points on the QQ-plots should fall on the red reference lines. Considering first the harbour porpoise models, we see that in the Poisson model, only a short interval of quantiles in the center fall on the red line. Moving away from the center in either

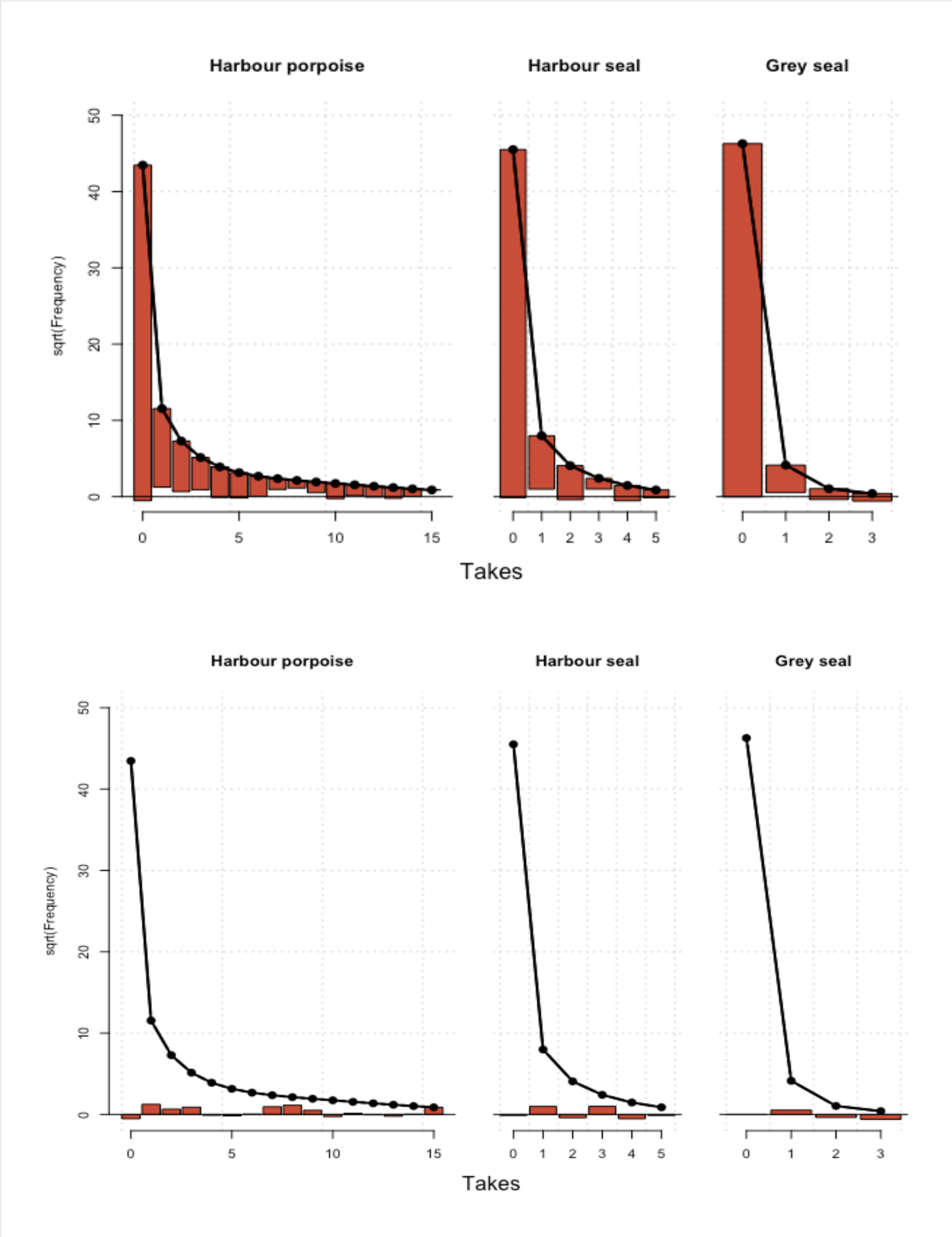

Figure 3.18: *Comparison of observed (red bars) and predicted values (black dots/lines) for the best Poisson models (model 1.13 for the harbour porpoise and seal and model 1.11 for the grey seal) using hanging (top) and suspended rootograms (bottom).*

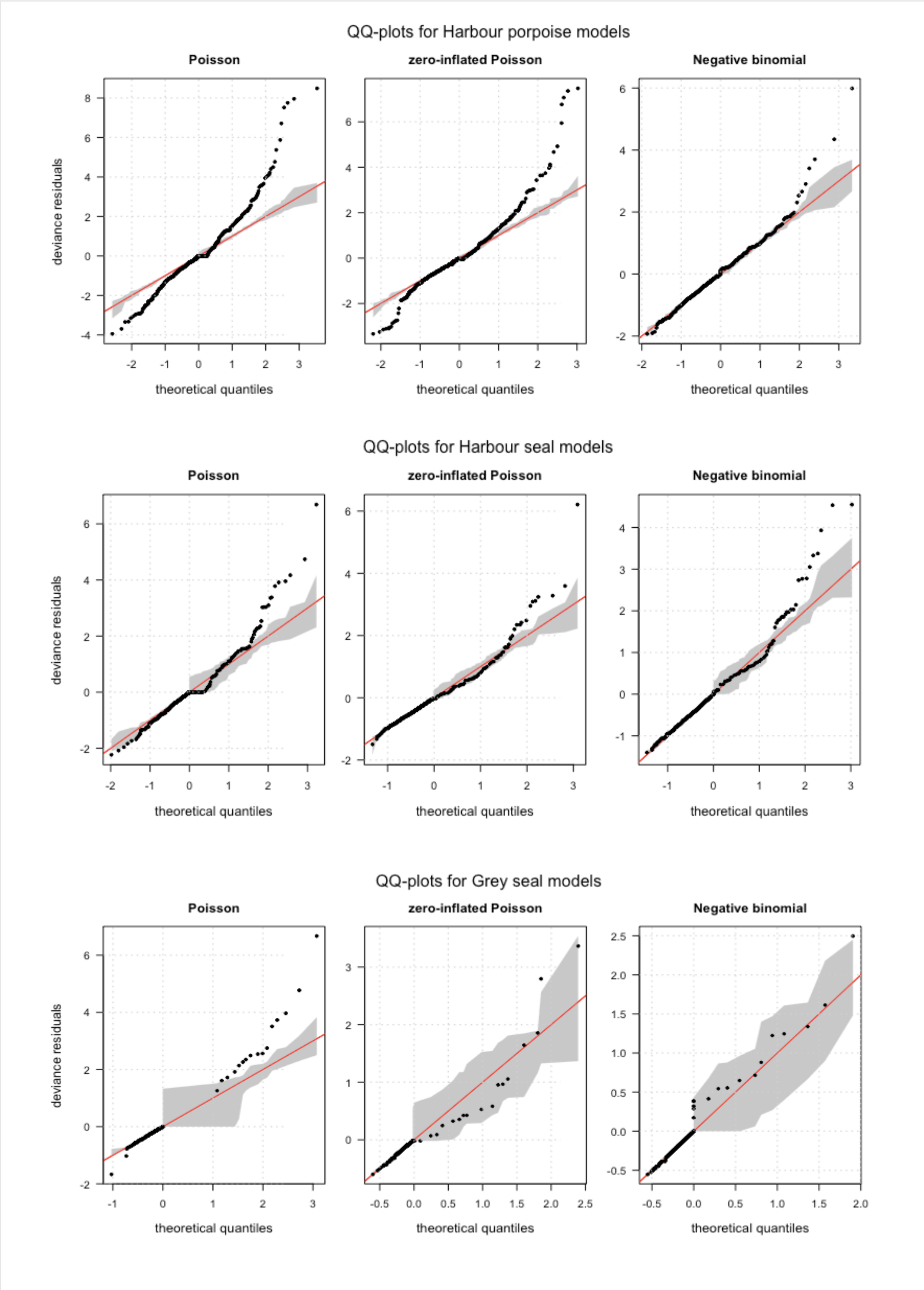

Figure 3.19*: QQ-plots for the best models. The red line is the reference line. Black dots are the points of intersection between theoretical quantiles and the model deviance residuals. The grey areas represent 95% confidence bands, generated by simulation.*

direction results in large departures from the reference line. Departures are especially large for larger quantiles. The ziP model performs a little better. A greater proportion of lower quantiles fall on the reference line, but large and very low quantiles both show large departures. In comparison, the NB model quantiles fall quite neatly on the reference line for quantiles < 2. Larger quantiles on the other hand depart greatly from the reference line. The harbour seal model QQ-plots all show a closer match for small quantiles, but considerable variation for high quantiles. The Poisson model shows small departures for very small quantiles and for quantiles 0 – 0.5, and large departures for quantiles > 1.7. The zero-inflated Poisson performs marginally better, especially for small quantiles, but large quantiles still show large departures from the reference line. The NB model performs worst of the three, with quantiles > 0 showing intermediate to large departures. In the grey seal QQ-plots, only very small quantiles fall on the reference line, and quantiles > 0 show very large departures in all the model families tested. Based on these QQ-plots, it seems that the harbour porpoise bycatch best matches a negative binomial distribution and harbour seal bycatch best matches a zero-inflated Poisson distribution. The QQ-plots for the grey seal model fits were inconclusive.

4. DISCUSSION

4.1 SUITABILITY OF MODELLING CRF BYCATCH WITH GLMS

This analysis has revealed that given the quality of the CRF data, a modelling approach for calculating harbour porpoise bycatch is problematic, and even more so for harbour and grey seal bycatch. Model AIC scores suggest that the NB model (rather than a Poisson/ziP model) may provide the best fit for modelling porpoise bycatch in terms of distributional assumptions, but there are considerable fitting difficulties associated with all three model families, as shown in the QQ-plots in Figure 3.19. Model predictions were skewed by the large deviance residuals associated with higher quantiles. The NB models resulted in additional large values that were not expected under the negative binomial distribution. This could possibly be caused by the use of a constant dispersion parameter across the ten years in the study (θ was estimated by R to approximately 2.732 in the best NB model). Estimating a dispersion parameter for each year in the study might improve the NB fit, but this was not done in this analysis. The deviance residuals of the NB porpoise model in particular greatly influenced model predictions, causing a great positive bias in predicted bycatch for specific areas and months (Figure 3.15). Bycatch predicted by the NB model in March, for instance, was quintupled compared to the observed bycatch. Extrapolations based on a model suffering from deviances of this magnitude cannot be considered to be reliable. This is reflected in the relatively high NB model CV of 1.08 (compared to 0.10 and 0.14 for Poisson/ziP models) and the wide 95% confidence interval, ranging from 2498 to 7135 porpoises. The size of the NB porpoise model confidence interval was 4637, compared to 1188 and 2041 for the Poisson and ziP models, respectively. The unsuitability of the NB model was also apparent in the predictions based on model formulations 1 – 12, which were unreasonably high, ranging from 4240 to 20562 porpoises. Predictions for model formulations 1 – 7 were all greater than 9000 porpoises. Even though total model deviance for the Poisson and ziP models were greater in both cases than the NB model, model predictions of the former models were less influenced by the deviances, as evidenced by a closer correction with observed bycatch.

These same problems were present and greatly accentuated in the harbour and grey seal models, presumably because they were based on even fewer bycatch observations than the corresponding porpoise models. It therefore seems that the CRF data do not wholly support the models developed in this manuscript, and that it would be appropriate to defer to the design-based estimator of the traditional stratified ratio approach until such a time as a better model can be developed. The annual bycatch estimated in the stratified ratio approach ranged from 2211 to 3218 porpoises, depending on the stratification scheme used (Table 3.4), and the 95% confidence intervals associated with these estimates were all overlapping. The corresponding estimates for the other two species ranged from 463 to 565 harbour seals and from 68 to 128 grey seals. The associated confidence intervals were overlapping. This makes a good point estimate difficult. However, as the results show, bycatch rates varied greatly between fisheries, areas and months, and this variation has been incorporated in the area × month models listed in Table 3.4. On this basis, annual bycatch was estimated to 2211 porpoises (CV=0.16), 459 harbour seals (CV 0.24) and 97 grey seals (CV=0.41)

4.2 SUITABILITY OF STUDY DESIGN

4.2.1 SAMPLING METHOD

The CRF data constitute a sample of observations of a number of variables (as outlined in Section 2.1), most important among them catch [of target species], bycatch and associated temporal and spatial variables. The sample was taken from a subset of the commercial fleet of fishing vessels with an overall length less than 15 meters. One assumption underlying the entire analysis is that the data collected by this segment of about 20 of the approximately 6000 fishing vessels was representative for the whole fleet. This, however, is an unrealistic assumption. We may expect that different fishing vessels exhibit unique fishing patterns. Different vessels may tend to frequent the same particular fishing sites, use one specific kind of gear, fish at particular depths, specialize in one particular catch species, etc. A consequence of these vessel-specific fishing patterns is that observations associated with the same vessel most likely are correlated, and not independent, as is assumed. Since there are only 16-24 different vessels every year, and in total only 40 different vessels represented in the CRF, a great number of observations will be correlated. The potential effects of this correlation between vessels were not included in the models. The experimental design therefore does not actually constitute a simple random sample, but rather a clustered random sample. Because of this, we should not assume that each observation is independent of the others. When working with clustered random samples, it is imperative to use statistical methods appropriate for this kind of sampling. One way that the predictions and confidence intervals in the GLM/GAM models could have been improved, would be to include vessels as a random effect in a mixed effects model. This would likely more accurately account for the variation among vessels in the CRF data. GAMs, being a type of *fixed effects* model, may not be appropriate for certain types of data, in particular clustered or longitudinal data (Hedeker 2005). For analysis of clustered data, random cluster and/or subject effects can be added into the regression model to account for the correlation of the data. This results in a *mixed effects* model, or in our case; a generalized linear mixed model (GLMM), which includes the fixed effects for the regressors and the random effects. Generalized linear mixed models are called "mixed" because they contain both fixed and random effects. Random effects are useful when one or more of some conditions are met: 1) lots of levels (e.g. many species or blocks), 2) relatively little data on each level and 3) uneven sampling across levels (Fox et al. 2015). In the CRF data set all of these criteria are met. Mathematically, the impact of this term then, is that it measures the difference between the average bycatch for a vessel, and the average for the entire fleet. This would have allowed us to incorporate the variability in the vessel effect that is due to picking a set of roughly 20 vessels out of a population containing 6500. On a related note, in calculating CVs and 95% CIs, the primary sampling unit (PSU) should ideally be the fishing vessel, and not the totals across all vessels in each area + month combination. But the observations in the CRF data set were not numerous enough to accomplish this kind of bootstrap. Separating observations into different strata resulted in a large number of strata with observations from only one single vessel. But for this approach to be viable, there would have to be at least two vessels in every stratum.

4.2.2 CRF DATA LIMITATIONS

Some research suggest that fish logbooks may significantly underreport the magnitude of bycatch (Stevens 1992, Johnson et al. 1999, Rowe 2006, Vanhatalo et al. 2014). Establishing that current bycatch levels are unsustainable may have repercussions for fishing activities by way of down-regulated quotas, gear restrictions, closed off fishing areas or mandatory use of pinger devices. All of these consequences would most likely entail added work and/or economical expenses for the fishing industry, so it is conceivable that fishers may have a vested interest in reporting bycatch. The CRF is based primarily on self-reporting. Additionally, independent observers (i.e. IMR personnel) visit each vessel at least once every year, and fish logs with and without an observer present are cross-checked in an effort to make sure that logs are accurate. However, Hall (1999) outlines three motivations for why even independent observers may plausibly choose to underreport bycatch. First, observers may develop a friendship with the crew, and this may affect their report. Second, the observer might be intimidated by the crew and underreport out of fear. Third, observers may be bribed to falsify reports. Whether any of this occurs in the CRF is not known, but in any case, bycatch in CRF fish logs should be considered the *minimum* number of animals killed by the CRF.

The basis for modelling grey seal bycatch in the cod and monkfish fisheries was only 16 observations spread over nine areas, twelve months and ten years. That means that 2144/2160 (or 99.3%) of observations were zeroes. This is a very poor basis for modelling bycatch. Because grey seal takes are so few, each single take would greatly influence bycatch rates, and in turn, disproportionately influence predicted bycatch when applied to fisheries statistics. This was demonstrated by the fact that the grey seal GLMs were not able to converge.

4.2.3 USING FISH LANDED AS A PREDICTOR OF BYCATCH

Conventionally, when estimating bycatch rates, bycatch is compared against some measure or proxy of fishing effort, yielding a catch-per-unit effort (CPUE) estimate, which is then used as a basis for extrapolation. The underlying assumption is that bycatch is proportional to the chosen measure of fishing effort. But as pointed out by Rochet and Trenkel (2005), this is often not the case. We might not expect there to be a strong correlation between catch and bycatch. We can easily imagine a scenario in which the catch yield associated with a fishing trip is not representative of the fishing effort. Consider two independent fishing trips, one lasting two hours and one lasting ten hours. Let us assume the same fishing gears were used on both trips. Further assume that the first trip took place in Lofoten during the cod spawning season and that the second trip took place on the west coast outside of Bergen. It is not impossible, nor even unlikely, that the catch yield for both of these trips be roughly the same, despite the fact that the second trip lasted five times as long as the first. Perhaps on the first trip, within a couple of hours, the fisher had hauled in ten big cod, each weighing five kilograms, for a total catch of 50 kgs. The fisher on the other trip might have spent ten hours at sea, and hauled in fifty cod, weighing one kg a piece, again for a total of 50 kgs. The total catch would be the same for both trips, but the fishing effort on the second day would be five times greater than on the first day. Clearly, in this case, catch yield would not be a good measure of fishing effort.

In this study, we chose to use landed catch as a proxy of fishing effort, despite the weak correlation between catch and bycatch (Section 3.3), for reasons outlined below. Other alternative measures of effort were the number of fishing trips undertaken and the number of gears set. The number of trips could be extracted from the CRF data set by pooling T-forms (Section 2.2) under each unique year and serial number combination, and then counting the total number of combinations[2]. But in light of the two-hours vs. ten-hours fishing trip example given above, it would seem the number of trips similarly would not be representative of fishing effort. Both fishers in the example undertook only one fishing trip, so the problem would seem to remain. As it turns out, however, the great majority of fishing trips undertaken by the small vessels that make up the CRF and the rest of the coastal fleet targeting cod and monkfish tend to last for the same duration of time. That is, gill nets are set one day, and then left to soak for approximately 24 hours before being hauled. If this assumption is correct, then the number of trips would likely represent a much more representative measure of fishing effort than catch yield, and we should expect a higher correlation between bycatch and trips than bycatch and catch. The same rationale applies to the relationship between the number of gears used and bycatch. Intuitively, the number of gears deployed should affect the probability of entanglement of marine mammals.

Unfortunately, the data that we received from the Norwegian Directorate of Fisheries only included cod and monkfish landing data. So a trips or gear based bycatch model could not have been extended to the entire coastal fishery fleet. Unaggregated landing statistics (i.e.: catch per fish log, or per fishing trip) was requested from IMR, who were more than willing to provide the data, but because of technical difficulties, it was not possible to obtain the data in time before the deadline of this thesis. One way to bypass this data deficiency would be to calculate the average catch per trip from the CRF data, and then use this factor to estimate the number of trips in the entire coastal fleet based on reported catch. This way, bycatch models based on the number of trips could have been applied to the entire coastal fleet. But

[2] This assumes a single haul for each fishing trip, and so would be wrong if the fishers did a second haul.

this approach would introduce an array of new assumptions in itself, and while the assumptions of the model itself would be more realistic from a biological point of view, the extrapolation would still be inherently linked to the catch variable.

4.3 BIOLOGICAL SIGNIFICANCE OF FINDINGS

4.3.1 HARBOUR PORPOISE

The ASCOBANS (Agreement on the Conservation of Small Cetaceans in the Baltic, North East Atlantic, Irish and North Seas) recommends that harbour porpoise bycatch not exceed 1.7% of the best available population estimate. This means that in order for an annual bycatch of 3000 harbour porpoises to be sustainable, the coastal populations must number at least 176,500 animals. Unfortunately, reliable estimates for the harbour porpoise population off the coast of Norway do not currently exist. In previous large scale, international surveys, the population of harbour porpoises in the North Sea was estimated to 341,366 porpoises (CV 0.14) in 1994 (Hammond et al. 2002) and 335,000 porpoises (CV 0.21) in 2005 (Scans II 2008). The highest abundance was found in the proximity of the British Isles, off the west coast of Denmark and in the Skagerrak/Kattegat Strait. A new survey (SCANS III), which in addition to the North Sea also included the Norwegian coastline northwards all the way to the Vestfjorden area (south of Hinnøya, approximately 67-68°N 12-16°E, the area in which regional bycatch pressure was highest), was carried out in the summer of 2016. But the results of this survey are not available yet. Based on the findings of this manuscript and the currently available population estimates, it is not possible to state conclusively whether harbour porpoise bycatch in Norwegian coastal fisheries is sustainable or not.

The annual bycatch of harbour porpoises in the cod and monkfish fisheries in Norway in 2006 – 2008 was previously estimated to 6900 animals (Bjørge et al. 2013). But in this study, it turned out that the statistics provided by the Norwegian Directorate of Fisheries, which was used to extrapolate CRF derived bycatch rates to the entire fisheries, erroneously contained landings of cod and monkfish taken with *all* gears. In addition to bottom set gill nets, this included hand jigs, long lines, purse seines, Danish seines and demersal trawls, none of which are typically associated with bycatch of harbour porpoise. Thus, landed tonnage was much exaggerated. For the month January 2006, for example, cod landings totaled 4,079 tons, but cod caught in gill nets was only 2,447 tons. This corresponds to an overestimation of approximately 66% for this month. Specific catch numbers for the other months vary, but the trend is consistent. So the estimated bycatch of 6900 animals was a substantial overestimate, and the actual annual bycatch rate, given correct landing data, would probably have been closer to about half that.

Tregenza et al. (1997) estimated an annual bycatch of 2200 porpoises (95% C.I. 900 – 3500) in the Celtic Sea, corresponding to 6.2% of the estimated population in the region (36,280 animals, CV=0.57). More recently, the combined annual porpoise bycatch in the Celtic and the Irish Seas was estimated to between 1137 and 1472 animals (range estimate), corresponding to 1.07% – 1.39% (i.e.: less than the 1.7% limit established by ASCOBANS) of the total population of 106,000 porpoises in the area (ICES 2015). While the latter study also included the Irish Sea, the combined bycatch estimate of the two areas is still less than the earlier, higher estimate that included only the Celtic Sea. Therefore, bycatch in the Celtic Sea must have been reduced over the last few years. This reduction in bycatch is likely due to the EU Resolution 812/2004, which required that vessels of specific size categories fishing in specific zones use pingers on their nets.

In the early 1990s, annual harbour porpoise bycatch in the Bay of Fundy and the Gulf of Maine in Canada was estimated to approximately 1800 animals, corresponding to 3-4% of the total population size (Trippel et al. 1996). The use of pingers on nets in 1996 – 1997 reduced bycatch levels in the Bay of Fundy by 77% compared to non-pinger nets (Trippel et al. 1999). The annual bycatch of harbour porpoise in the Northwest Atlantic (primarily the Gulf of Maine) from 1993 to 1997 was 1,704 animals (CV=0.09) and the best population estimate at the time was 54,300 animals (CV=0.14) (Waring et al. 2000). Bycatch thus corresponded to about 3% of the total population size. The US has since introduced a Harbour Porpoise Take Reduction Plan, most recently amended three years ago (NOAA 2013). Bycatch reduction measures in the plan include the use of pingers and restrictions on where, when and how gear can be set.

In the latest Marine Mammal Stock Assessment Reports (SARs) by Species/Stock) the harbour porpoise population in the Gulf of Maine/Bay of Fundy was estimated to 77,883 animals (CV=0.32), with a concurrent total annual bycatch estimate for the same area of 564 porpoises

(521 in US fisheries (CV=0.15) and 43 in Canadian fisheries), corresponding to 0.7% of the total population, well below the 1.7% ASCOBANS limit.

While we do not have good estimates for the size of the harbour porpoise population in Norwegian waters, management schemes should strive to err on the side of caution. IMR is currently (September – early December 2016) conducting an experiment with pingers on gillnets targeting monkfish. In addition to the cases just discussed, such pingers have been known to reduce bycatches of harbour porpoise in gillnet fisheries in Denmark and the USA with 80-100%. Pending the results from this experiment (which will be available by January 2017), it remains to be seen how the use of pingers affect bycatch of harbour porpoise and other species. But whether pingers can be a part of the solution to the bycatch problem in Norwegian fisheries also depend on the susceptibility of Norwegian fishers to their use. Adding pingers to the nets and making sure they are working properly may involve both economic costs and extra work on the part of the fishers.

4.3.2 HARBOUR SEAL

An annual bycatch of 424 harbour seals and 83 grey seals corresponds to a total bycatch of 507 coastal seals, 16% of which were grey seals. A mark and recapture study (Bjørge et al. 2016) showed that about 46% of bycaught seals along the Norwegian coast were actually grey seals. Another study (Bjørge et al. 2002) indicated that the majority of bycaught grey seals were taken within four months of birth. This proportionate discrepancy (16% vs. 46%) might be explained by a seal misidentification hypothesis. While the adult harbour and grey seals are easily distinguishable, it is possible that a proportion of young grey seals have been misidentified as harbour seals in the CRF data. The coastal harbour seal population has been estimated to 7568 animals. With a yearly hunting quota of 455 animals and a yearly bycatch of 424 animals, the total anthropogenic mortality of the harbour seal is 879 animals per year, or about 12%. This roughly corresponds to the grey seal's maximum growth rate (Heide - Jørgensen and Härkönen 1988). The mark and recapture study by Bjørge et al. (2016) suggested an annual bycatch of 555 harbour seals. The predicted yearly bycatch estimates by the best harbour seal GLMs were 424, 486 and 600 with overlapping confidence intervals. Estimates using the stratified ratio derived bycatch rate ranged from 459 – 565 animals with overlapping confidence intervals. It therefore seems that actual yearly harbour seal bycatch might be slightly greater than 424 animals. This level of bycatch, in addition to the legally established catch quotas, has the potential to affect the population trajectory of harbour seals in Norway.

4.3.3 GREY SEAL

Surveys between 2006 – 2008 showed that the grey seal had a yearly pup production of 1269 young, the great majority (943) of which were located between 62°N and 68°N. New surveys in 2014 and 2015 showed that pup production in this area was more than halved. The hunting quota in this area has previously been set to 250 animals, but was reduced in 2014 to 105 animals, and then again in 2015 to 0 animals. Most bycaught grey seals were taken in this region (areas 00, 06 and 07), but it is not known whether incidental catches of grey seals in fisheries may have been the cause of the reduced pup production.

It is also worth noting that all the bycatch estimates given in this manuscript were based on an absolute *minimum* number of animals caught. There are several potential reasons for why actual bycatch most likely is higher than that which is reported in the CRF fish logs. When the nets are being hauled, for example, entangled porpoises and seals may come free from the net and never be taken on board the fishing vessel. These “losses” would not be easily detected, because animals may become disentangled while still submerged, and sink or float away, or hauling may occur when it’s dark outside, and the attention of the fishers is on handling the gear). Losses during hauling have been reported by observers on board fishing vessels in other monitoring programmes (Tregenza et al. 1997).

4.4 FUTURE STUDIES

Data from the coastal reference fleet should be used in a continued effort to monitor bycatch of harbour porpoises, harbour and grey seals, as well as sea birds in Norwegian fisheries. One aspect of bycatch reporting that can potentially be easily improved in future studies is related to species identification. Some effort should be made to make sure that the crew on CRF vessels are able to properly distinguish between different bycaught species, especially young coastal seals, which seem to be subject to misidentification. This could potentially be addressed at annual CRF meetings, where owners and crew from CRF vessel usually are present.

Future studies should take into consideration that the CRF sampling method constitutes a clustered random sampling, and not a simple random sampling. Analyses should therefore take into account that observations from each vessel are inherently correlated. Future modelling studies should also consider whether harbour porpoise bycatch might be better modelled using a negative binomial rather than a Poisson distribution, assuming that the problems of a NB model can be solved (e.g. by not constraining k, but allowing it to very in a hierarchical way). Trips-based models (as opposed to catch-based models) should also be explored.

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