Assessing extinction risk: a Population Viability Analysis of the Norwegian population of harbour seals *(Phoca vitulina)*

Master thesis

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Assessing extinction risk: a Population Viability Analysis of the Norwegian population of harbour seals *(Phoca vitulina)*

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ABSTRACT

The harbour seal population in Norway is managed through a quota regulated hunt to ensure a viable population within their natural habitat along the Norwegian coastline. The government has determined that this aligns with the registration of 7 000 individuals during the molting season. But how can scientists and the government be ascertained that the long-term of this plan is sustainability. In this study, the viability of the harbour seal population in Norway was investigated under different rates of hunting and bycatch, together with the probability of disease outbreak and reduction in fertility due to pollutants. A parameterized Population viability analysis (PVA) was used to simulate the harbour seal population trajectories over 100 years. The PVA was adjusted to test the viability of the population under a bycatch rate that was equally distributed between the different age classes, and a bycatch rate that was higher towards younger individuals. The method resulted in 100 different scenarios with all the possible combination of the hunting and bycatch rates. There were not many scenarios that fulfill the management requirements. Only 9.9% of the scenarios had a final mean population size that was either equal to or over 7 000 individuals. An evenly distributed bycatch rate among the age classes resulted in a greater number of population trajectories experiencing quasi extinction risk compared to an age-distributed bycatch rate towards younger individuals. According to the results, the annal removal of harbour seals due to hunting and bycatch is not sustainable. However, a great discrepancy exists between the population estimates provided in this study and those based on field observations conducted by the Institute of Marine Research (IMR). Hence, adjustments to the model, and more data on the study species, are needed to enhance the representativeness of the PVA simulations and to reduce the discrepancy.

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1 INTRODUCTION

1.1 PINNIPEDIA

Pinnipedia is one of three main groups of marine mammals and consists of 35 different species of seals (Bjørge et al., 2010a; Jefferson et al., 2015a). The suborder consists of three families: Odobenidae (walrus), Otariidae (sea lions and fur seals), and Phocidae (true seals) (Van Bonn, 2015). Pinnipeds can be found in every part of the globe, including the Arctic and Antarctic regions (Biolsi, 2017). Pinnipeds are carnivorous mammals with a diet made up mainly of fish (Biolsi, 2017). They have therefore evolved to be excellent swimmers with a streamlined body form to reduce friction (Davis, 2018). To survive in water, where the temperature often is lower than on land, pinnipeds have evolved features that limit the heat loss from the body to the water (Bjørge et al., 2010a). They are covered in fur and have a thick layer of blubber underneath the skin that provides the animal with good insulation. The insulation of the blubber assists in keeping the internal temperature stable at 37° C while the skin can have the same temperature as its surroundings (Bjørge et al., 2010a). Although pinnipeds spend most of their life span in aquatic habitats, they still need to haul out on land or ice. All members of this subgroup give birth on land or drifting icefloes and are therefore still able to move on land (Bjørge et al., 2010a; Jefferson et al., 2015a).

Pinnipedia contains K-selected species. Organisms within this group share many similar features. They produce few offspring over a long lifespan, and normally just one pup per litter (Bowen, 2018). The parents invest much time and resources to raise their young. The generation time can be several years, and the mortality rate for adults is very low compared to pups (Lewison et al., 2004). These life history traits can be favorable against environmental and demographic stochasticity, but make the organisms vulnerable to extinction when facing overexploitation and intense hunting pressure (Lewison et al., 2004).

Seal hunting is not a new phenomenon. Pinnipeds, with their high energy and nutritional density, have ensured the survival of hunters for thousands of years (Nye et al., 2020). However, several species are now extinct or threatened due to intense hunting by humans (Van Bonn, 2015). A large reduction in species' population will result in the loss of genetic diversity which can affect the long-term survival of the population (Nye et al., 2020; Van Bonn, 2015). However, many pinniped species are stable and increasing in numbers, which often results in more frequent interaction with humans and human activities (Van Bonn, 2015). Both seals and humans eat and hunt fish, resulting in conflicts. Some fishers believe that the seals cause damage to their fishing gear, consume large amounts of fish, and spread parasites (Koss et al., 2023). The seals can also become entangled or entrapped in fishing gear, and wind up as bycatch (Pemberton et al., 1994). A species that is exposed to both hunting and bycatch, is the harbour seal (*Phoca vitulina).*

1.2 THE STUDY SPECIES – THE HARBOUR SEAL

The harbour seal is a species belonging to the true seal family, Phocidae (Bigg, 1981; Bjørge et al., 2010a). The 19 species that are part of this family do not have external ears, and their hind-flippers are pointed backwards and can only be used for swimming (Bjørge et al., 2010a). The harbour seal is a small seal, where the males can be 1.5 meters and 100kg. The females are slightly smaller and can weigh up to 80kg (Bigg, 1981; Bjørge et al., 2010a).

1.2.1 Status and distribution

Today the harbour seal has one of the largest distributions among all the pinnipeds. The species can be found in the North Atlantic and North Pacific, from temperate to Arctic waters (Bigg, 1981). The global population is estimated to range from 610 000 – 640 000 individuals (Bjørge et al., 2010b), and is listed as "Least concern" on the International Union for Conservation of Nature's (IUCN) red list on both a global and European scale (European Mammal Assessment team, 2007; Lowry, 2016). Being categorized as a "Least concern" species implies that the IUCN has evaluated it against the red list criteria but does not qualify as endangered or vulnerable (IUCN, 2022). The species is distributed among five subspecies divided primarily based on the geographical distribution (King, 1983):

- *P. vitulina* subsp. *vitulina* – The Northeast Atlantic harbour seal (113 000)
- *P. vitulina* subsp. *concolor* – The Northwest Atlantic harbour seal (38 000)
- *P. vitulina* subsp. *mellonae* – The Ungava harbour seal (few animals)
- *P. vitulina* subsp. *richardii* – The Northeast Pacific harbour seal (376 000)
- *P. vitulina* subsp. *stejnegeri* – The Northwest Pacific harbour seal (10 000)

P. vitulina subsp. *vitulina* is located from North of Portugal to the Barents Sea, Iceland and into the south of the Baltic Sea (Jefferson et al., 2015b). The northernmost population of harbour seals belongs to this subspecies and can be found in Kong Karls Forland at Svalbard (Lydersen & Kovacs, 2005). The combined Northeast Atlantic population is estimated to be between 113 450 – 134 200 individuals (Bjørge et al., 2010b).

The harbour seal is the most numerous coastal seal species in Norway with an estimated population of approximately 10 000 individuals (not including Svalbard) (Bjørge et al., 2007). The population can be found along the entire coast from Østfold to Finnmark (Nilssen et al., 2010). The Norwegian coastline is 103 000 km long, including all the islands and fjords, and spans 13 latitudinal degrees (Breili, 2022). It is the second-largest coastline after Canada and hosts a variety of different habitats such as fjords, islands, sandbanks, open rocky coasts, etc. (Breili et al., 2017; Ministry of Climate and Environment, 2015). All these habitat types are environment that the harbour seal inhabits (Bjørge, 1991). The harbour seal is considered a coastal seal because the species is only located in the coastal zone throughout the whole year (Bigg, 1981). They do not migrate, and local movements are associated with breeding and searching for food (Bigg, 1981).

In 2010 the species was listed as "vulnerable" on the national red list in Norway because high harvest quotas had reduced the population from a minimum estimate of 7700 to 5800 individuals (Eldegard et al., 2021; Nilssen et al., 2006). In 2010, the hunting quotas were reduced following recommendations from scientists, which led to the population returning to the preferred stock level. After 2010, the population has remained stable, and therefore listed as "least concern" on the national red list in 2015 and 2021 (Eldegard et al., 2021).

1.2.2 Annual cycle

The breeding season occurs in June and early July, with a peak in the middle of June (Thompson, 1988; Venables et al., 1955). The females give birth to one pup annually on haul-out sites (Bigg, 1981). Before the pups are born, they moult their white lanugo fur which is typical for many pups of other *Phoca* species (Bigg, 1981; Bjørge, 1993). This means that the harbour seal pups are born without the white birth coat, and have the same grey and brown, irregular patch fur as the adults (Bigg, 1981; Bjørge et al., 2010a).

The newborn pups are highly dependent on their mother during the lactation period, which lasts three to five weeks (Bjørge, 1993). Just minutes to hours after the females have given birth, the pups follow their mother into the water (Lawson & Renouf, 1985). This behavior may have evolved to avoid predators on land, and due to the exposure to tides on the haul-out sites (Bowen et al., 1999).

When the lactation period is over, and the pup is weaning, the mother will abandon her pup (Bigg, 1981). The females start to ovulate, and mating begins (Bigg, 1981; Thompson, 1988). The majority of the Phocidae species, including the harbour seal, have an aquatic-mating system (Bigg, 1981; Boeuf, 1991). Females become sexually mature after 3-4 years, while males reach their mature age after 5-7 years (Bjørge, 1993). Males have shown territorial behavior to increase their mating success, but mating might be promiscuous (Bigg, 1981; Hayes et al., 2004).

After the breeding season, every seal one year and older starts their annual moulting period (Thompson, 1988; Venables et al., 1955). The moulting period last from mid-July to mid-September, with a peak in August (Thompson, 1988). This is the time of the year when the seals spend the most time at the haul out sites. Each individual uses approximately one month to moult, and usually, immature seals moult before the adults (Thompson & Rothery, 1987). To speed up the process, the skin temperature increases (Bjørge & Nilssen, 2020a). They can allow a higher skin temperature since the heat loss is less on land than in the water (Bjørge & Nilssen, 2020a).

Harbour seals are social and central place foragers animals (Orians & Pearson, 1979). When they gather on land, they form small groups that lie ashore in the littoral zone (Bjørge et al., 2010a). When not hauled out, the harbour seal spends the time in water searching for food (Bigg, 1981). They often feed near their haul-out site, although feeding trips that last several days and extend tens of kilometers away from their haul-out site are not uncommon (Wiig & Øien, 1988). These kinds of feeding trips occur outside the breeding and moulting season (Bjørge et al., 2010a)

1.3 STRESSORS ON THE NORWEGIAN HARBOUR SEAL POPULATION

A natural population is exposed to many stressors at the same time (Silva et al., 2021), and the harbour seal is not an exception. The stressors can have both positive and negative effects on the population and be caused by humans or arise naturally within their ecosystem. Some of the key stressors that affects harbour seal population dynamics are described below.

1.3.1 Hunting

Records show that harbour seals have been hunted for many years in Norway. During the stone age, the harbour seal was the second most important mammal in humans' diet (Olsen, 1976). In the 20th century, harbour seals were hunted to near extinction because the species was regarded as a vermin (Bjørge, 1991). Up to 1973, the harbour seal hunt in Norway was unregulated except for some local protection initiatives in the 1960s (Anon, 1990; Bjørge, 1991) In 1973 the population in Østfold to Sogn & Fjordane became protected. Further north (from Møre & Romsdal to Finnmark), free hunting was allowed from the 1st of December to the 30th of April. There were no quotas in place to regulate the number of seals that could be hunted, and no obligation to report catch numbers (Bjørge & Øien, 1999).

In 1997 the management regime that is used today was employed (Nilssen et al., 2010). The management objective is to ensure a viable population of harbour seals within the species' natural habitats (Bjørge et al., 2021; Nilssen et al., 2020). To ensure this goal, the harbour seal is managed through a quota-regulated hunt that intends to stabilize the population on a level where 7000 seals are recorded during the moulting season (Nilssen et al., 2020). The hunting quotas are distributed between the counties along the coast. A Total Allowable Catch (TAC) is calculated based on the population's status, size, and production. The TAC is supposed to remain stable between every count (± 5 years), and is adjusted based on the most recent counting results (Fiskeri- og kystdepartementet, 2010). The Norwegian Fishery management can determine annual quotas on harbour seals in areas where colonies are defined as huntable. In recent years, the annual hunting quota for harbour seals has been set at approximately 450 individuals. The actual reported harvest, however, have been slightly lower at around 350 individuals (Nilssen et al., 2020).

1.3.2 Bycatch

While the harvest of the harbour seal population in Norway is well regulated and documented, the species faces another threat caused by humans. Commercial fisheries are conducted in almost every part of the ocean today, including the Norwegian coastline (Moan, 2016). Fishery products are Norway's second largest export goods, and the Norwegian fisheries are widespread (Bjørge et al., 2013). The fishing pressure is highest in the rich coastal waters where the abundance of the number of fish species is greatest. However, fishermen are not the only ones who seek out the best fishing territories. Harbour seals and other marine mammals have a diet containing many of the same fish species as the fishing vessels harvest. The seals may be unfortunate and interact with the fishing gear which can result in fatal outcomes.

The harbour seals in Norway are exposed to incidental bycatch throughout their habitat (Bjørge et al. 2002). 98% of all the pinniped bycatches occur in gillnet fisheries (Read et al., 2006). How the seals get entangled or caught in the net is unknown. The outcome, however, can be that the animals can’t return to the surface to breathe and will drown (Koss et al., 2023). If a seal manages to get untangled, the process may cause the animal serious injuries that may inhibit the search for food or other life-important tasks (Moan, 2016).

Since 2006 the IMR has monitored the bycatch of marine mammals (Bjørge & Nilssen, 2020a). The IMR use the Norwegian reference fleet to monitor marine mammal bycatches. The reference fleet is a monitored group of vessels from the Norwegian fishing fleet, that is designed to be as representative as possible of the whole fleet (Clegg & Williams, 2020). Their duties include reporting catches, bycatches and other details pertaining to their fishing activities to the IMR (Clegg & Williams, 2020). Analyses indicate that the bycatch rate of harbour seals along the Norwegian coast lies at around 500 individuals annually (Bjørge & Nilssen, 2020a). This is consistent with the capture/recapture study of harbour seals along the Norwegian coast (Bjørge et al., 2006).

1.3.3 Disease outbreak

The population in the Skagerrak area, and other parts of Europe, went through a big decline in 1988 and again in 2002. The Phocine Distemper virus (PDV) killed approximately 50% of the population in Skagerrak (Bjørge et al., 2010b; Härkönen et al., 2006). The population recovered quickly after the outbreaks with a 13% increase per year (Olsen et al., 2010). The cause of the high growth rates may be the high survival rates of mature females compared to other age classes and gender (Bjørge et al., 2010b). The PDV did not spread from the Skagerrak area and further north meaning that a large proportion of the Norwegian population did not get affected by the PDV epidemics (Andersen & Olsen, 2010).

1.3.4 Pollutants

The harbour seal, and all marine mammals, are exposed to chemical compounds (O'Shea et al., 1999). Polychlorinated biphenyls (PCBs) are toxic chemicals that resist environmental degradation, can bioaccumulate in ecosystems, and potentially cause health problems to marine mammals (O'Shea et al., 1999; Rashed, 2022). High PCB levels in harbour seals, and other seal species, have been linked with reproductive failure (Helle et al., 1976; Reijnders, 1986). Despite the ban on PCBs in the EU in the 1880s, reports indicate that PCB levels in marine mammals are still high (Jepson et al., 2016; Shawa et al., 2014; Williams et al., 2021). The persistently high levels of PCBs in marine mammals suggest that this stressor still has a negative impact on the harbour seal population.

1.4 LOOKING INTO THE FUTURE – POPULATION VIABILITY ANALYSIS

As mentioned above, humans have gathered extensive knowledge regarding the harbour seal over an extended period. The collected information has provided insight into previous and current population status. This data material has facilitated the formulation of a comprehensive management strategy aimed at protecting the species in the years to come. However, the question arises as to how scientists and the government can ascertain the long-term sustainability of this plan. Who can predict the development of the harbour seal population 100 years from now?

Population Viability Analysis (PVA) is widely used to calculate and predict the survival prospects of natural populations by using quantitative methods (Boyce, 1992). A PVA is a parameterized simulation model based on data regarding demography, life history, management activities, and environmental variability (Roberts et al., 2016) With this information used in a PVA, it is possible to simulate the population size over a chosen time horizon into the future (Roberts et al., 2016). It is also possible to enable the simulation of various population dynamics, including time to extinction, the probability of extinction, and survival likelihood over 100 years, for example, (Boyce, 1992).

A simulation is repeated multiple times, during which the population is tracked concerning a predetermined threshold, such as the quasi-extinction threshold. By changing the parameter values in the PVA, it is possible to assess the influence of extinction from the different factors. This information provides the foundation for assessing alternative management strategies to save the population from possible extinction (Boyce, 1992; Lee et al., 2020; Roberts et al., 2016)

1.5 AIMS AND OBJECTIVES.

The overall aim of this thesis was to investigate how different hunting pressures and bycatch rates would affect the Norwegian population of harbour seals over time. The analysis was conducted using a parameterized PVA model. The following objective was made:

1. Assess extinction risk for harbour seas along the Norwegian coast.

Data from a mark/recapture study on Norwegian harbour seals conducted by Bjørge, et al. (2006) was used to create an age-distributed bycatch rate. The age-distributed bycatch rate was incorporated into a second PVA model to assess the viability of harbour seals and pursue the following objective:

2. Compare the viability of Norwegian harbour seals under scenarios with age-distributed bycatch rates and bycatch rates that were constant for all age classes.

2 MATERIALS AND METHODS

2.1 STUDY POPULATION

Data from the latest seal surveys were used as population estimates for harbour seals in each region (Table 2.1) (Institute of Marine Research, unpublished data). The data from the IMR contained regional county-specific total counts from 1997 to 2022. The most recent population estimates were used as the starting population size for each region in the Population Viability Analysis (PVA).

Table 2.1: List of numbers of harbour seals (N) per county. The year column indicates when the latest count was completed in the corresponding region. K represents the calculated carrying capacity.

Region	**Year**	***N***	***K***
Møre & Romsdal	2017	633	1266
Sogn & Fjordane	2017	643	1286
Rogaland	2017	410	820
Finnmark	2020	1121	2242
Troms	2020	759	1518
Nordland	2020	1570	3140
N-Trøndelag	2020	124	248
S-Trøndelag	2020	788	1576
W-Agder	2022	51	102
E-Agder	2022	43	86
Telemark	2022	234	468
Vestfold	2022	662	1324
Østfold	2022	695	1390
Total		**7733**	**15 467**

In this study, the Norwegian harbour seal population was divided into subpopulations based on the original county divisions (before new counties were created after 2018 and 2020 (Regjeringen, 2019)). This decision was undertaken with consideration for the fact that the Norwegian government was still using the original counties to estimate population size in the most recent aerial surveys. Using old counties would also facilitate the comparison of the results with previously published studies.

2.2 POPULATION VIABILITY MODEL

The population viability model (PVM) was based on an age-structure population model for harbour seals developed by Carroll (2021) (https://github.com/DaireCarroll2023/Harbour-Seal-PVA/tree/main). Carroll tested the viability of Norwegian harbour seals under different hunting scenarios combined with decrease in birth rate due to pollutants, and the possibility of disease outbreaks. Carroll's model was built upon the initial framework by Silva et al. (2021), who investigated the viability of the Swedish-Danish harbour seal population under several scenarios of single and combined effects of hunting, pollutant, and disease outbreaks. Comprehensive details are provided in his corresponding research paper (Silva et al., 2021). In this study, the model was adjusted to explore the viability of the Norwegian population of harbour seals when confronted with varying hunting and bycatch levels (Table 2.2). Additionally, the model was extended to allow age-specific bycatch rates.

Each scenario was simulated 1000 times to increase the accuracy and reliability of the results and allow for the calculation of confidence intervals. The 95% confidence intervals were calculated from the 2.5th and the 97.5th quantiles of the resulting distribution of the replicates. The projection time of 100 years was selected deliberately to ascertain the population level where the mean growth curves would stagnate, enabling an examination of the long-term effects of hunting and bycatch.

The quasi-extinction probability is the likelihood that a population will decrease to a given threshold or a fraction of the initial population size where extinction is certain (Harding et al., 2003). The quasi-extinction threshold (QE) was set at 100 individuals for each subpopulation, resulting in a functional quasi-extinction threshold of 1300 individuals for the whole population in Norway. If the final population size was below this threshold after 100 years, it would be defined as extinct.

The structure of the model was similar to Carroll's model, although a few changes were made to include different bycatch rates. The population was analyzed under two different bycatch settings. One where the bycatch rates were constant across all the 38 age classes (Setting A), and the second where the bycatch rates were higher for younger individuals (Setting B). Each setting included modifications to the PVA model and the population model. This resulted in the making of two unique sets of PVAs and population models. The alterations associated with each of the two settings are described below.

2.2.1 Setting A

The hunting pressures examined for the harbour seals remained consistent with those in the original model (Carroll, 2021). The hunting delay (number of cycles before hunting was implemented) in the model used was set to zero as this phenomenon was not to be discussed in this study.

Some modifications in the population model for harbour seals were necessary to make the PVA simulate the population under different values of bycatch instead of just one value. The age-specific bycatch $E_{c,a}$ was calculated for every age group a in county c:

$$E_{c,a} = \frac{1}{38} * N_c * e \qquad \text{(eq. 1)}$$

Here, N_c refers to the population size in county c in each time step before bycatch is applied. e represents the bycatch rate associated with the current scenario (Table 2.2). The term 1/38 means that total bycatch in each county was evenly distributed between the 38 age classes. For each age class a, $E_{c,a}$ was subtracted from that age class.

2.2.2 Setting B

Modifications to the population model were again necessary to incorporate age-distributed bycatch. The procedure was similar to the age-biased hunting created by Silva et al. (2021). Some adjustments to equation 1 were needed to distribute the total bycatch between the age classes. Equation 2 calculated the total bycatch $E_{c,a}$ in age group a and county c:

$$E_{c,a} = N_c * e * \frac{Ep_a}{\sum_1^{38} Ep_a} \qquad \text{(eq. 2)}$$

In equation 2, Ep_a represented the age-specific bycatch rate in Table 2.3 to age group a. $E_{c,a}$ individuals were then removed from the age group a. If $E_{c,a} > a$, the age group would be reduced to a value less than zero. To prevent negative individual values in the age groups, individuals from other age groups were removed from the county population until $a = 0$. This process was done for every age group in all the counties with every value of e, based on N_c.

The parameter Ep was then inserted into the PVA along with all the other model parameters that represented the characteristics of the study population. When Ep was included, the age-specific probability of entanglement would be incorporated into the population simulations.

2.3 ENVIRONMENTAL STRESSORS

Since a natural population is exposed to many stressors at the same time, the hunting quotas were simulated in combination with five levels of bycatch (Table 2.2). The different hunting and bycatch rates were simulated together with the probability of disease outbreak *P_D* and fecundity reduction due to pollutants *f* (Table. 2.5). These two parameters, however, were constant throughout all the scenarios. Except for the baseline scenario where all these four stressors (*H, E, f,* and *P_D*) were set to zero.

Table 2.2: Overview of the hunting and bycatch rates used in this study. The numbers indicate the proportion of the population that is being removed due to hunting and bycatch.

	Hunting *H* and bycatch *E* rates										
H	0%	1%	2%	3%	4%	5%	6%	7%	8%	9%	10%
E	0%		2.5%			5%		7.5%			10%

As mentioned in chapter 2.2, the population was simulated under two settings of bycatch. A study conducted by Bjørge et al. (2006) was used to obtain a realistic distribution of the bycatch rate between the different age classes. This study exclusively utilized data pertaining to cases where the cause of death was relevant to bycatch. Based on their findings, it was possible to calculate the relative bycatch rates for the age classes that were used in the simulations for Setting B (Table 2.3).

Table 2.3: Age-specific recoveries of harbour seals tagged as pups during the period 1978-1998 (Bjørge et al., 2006).

Cause of death	**Age class (year)**				
	0-1	**2**	**3-4**	**5-38**	**Total**
Set nets	24	4	1	1	30
Cod traps	2	1	1		4
Other gear	2	1		1	4
Total	28	6	2	2	38
Bycatch rate	0.737	0.158	0.053	0.053	1.0
Ep	0.737	0.158	0.0265	0.0016	

The combination of different hunting and bycatch rates, with two bycatch settings, resulted in a total of 100 different scenarios (Table 2.4). The various scenarios consisted of one baseline scenario with no environmental stressors present. Eleven scenarios with the indicated hunting levels, but where bycatch was not present. 44 scenarios with different hunting and bycatch levels where the bycatch rate was spread equally between the age classes (Setting A). And at last, 44 scenarios with the same composition of hunting and bycatch levels, but where the bycatch rate was biased towards younger individuals (Setting B).

Table 2.4: An overview of every scenario (not including the baseline scenario). The table illustrates the different scenario IDs with unique combinations of hunting H and bycatch E. The table refers to both Setting A and Setting B.

***H* (%)** \ ***E* (%)**	**0**	**2.5**	**5**	**7.5**	**10**
0	1	12	23	34	45
1	2	13	24	35	46
2	3	14	25	36	47
3	4	15	26	37	48
4	5	16	27	38	49
5	6	17	28	39	50
6	7	18	29	40	51
7	8	19	30	41	52
8	9	20	31	42	35
9	10	21	32	43	54
10	11	22	33	44	55

2.4 PARAMETER VALUES

This study used many of the same parameters as the original model (Table 2.5). The life history parameters were gathered from Härkönen et al. (2002). Both survival rates (S) and fecundity rates (B) (female pups/ surviving females) were age-specific and are shown in Table 2.6. To reflect the stochasticity in the system, a 5% random variation was added every year (Silva et al., 2021). The carrying capacity K was calculated in the same manner as Carroll (2021):

$$K = N * 2 \qquad \text{(eq. 3)}$$

Table 2.4: An overview of the parameters used in the population viability analysis of harbour seals. Introduced or changed parameters are marked in green. Complex parameters with multiple values or complex structure are given in a table form.

Parameter	Value	Description
A	38	Maximum age
B	Table 2.5	Birthrates (age specific) (Vector)
E	Table 2.2	Proportion of population lost due to bycatch (Vector)
Ep	Table 2.3	Relatively entanglement risk by age class (Vector)
f	0.1	Reduced fecundity due to pollutants
H	Table 2.2	Proportion of population lost due to hunting (Vector)
H_delay	0	Number of cycles before hunting is implemented
K	Table 2.1	Population carrying capacity
M	Table A1	Migration between local populations (Matrix)
P	Table A3	Initial population structure (Matrix)
P_D	0.07	Probability of disease outbreak in each cycle
O_D	0.65	Maximum proportion of an age class killed during outbreak
W_D	Table A2	Proportion of O_D experienced by age class (Vector)
S	Table 2.5	Survival rate (Vector)
R	0.05	Level of random variation per cycle
t	100	Number of cycles (years)

Table 2.5: Age-specific parameters of survival and fecundity in harbour seals (Heide-Jørgensen & Härkönen, 1988; Härkönen et al., 2002).

Age	Survival rate ***S***	Age	Birthrate ***B***
0-1	75%	4	17%
1-4	89%	5	33%
4+	95%	6-27	47%
		27-37	35%

2.5 SENSITIVITY ANALYSIS

A sensitivity analysis was conducted to investigate the sensitivity of the estimated population growth to changes in fecundity reduction and probability of disease outbreak (parameters *f* and *P_D*). This analysis was done for the whole population of harbour seals. Only scenario 27 and 28 was tested. Scenario 27 had a 4% hunting quota and a 5% bycatch rate. These values corresponded best with the actual reported harvest, and the estimated bycatch rate in Norway. Scenario 28 had a hunting quota and a bycatch rate equal to 5%. These values corresponded best with the hunting quotas set by the IMR in recent years, and the estimated bycatch rate of harbour seals in Norway (Nilssen et al., 2020). The scenarios were analyzed for both Setting A and Setting B. The parameter *P_D* was tested with two values (0.0, 0.07), while parameter *f* was tested with three values (0.0, 0.1, and 0.2). The 20% reduction in fecundity was added in this analysis to account for reported 15% reduction in fecundity (Helle, 1980). Combinations of the parameters was tested and resulted in ten different combinations of each scenario (Table 2.6)

Table 2.6: Different combinations of the values of parameters f and P_D that were tested in the sensitivity analysis.

Setting A	**Setting B**	***P_D***	***f***
A_a	B_a	0.00	0.00
A_b	B_b	0.07	0.00
A_c	B_c	0.00	0.10
A_d	B_d	0.00	0.20
A_e	B_e	0.07	0.10

2.6 STATISTICAL ANALYSIS

The mean population curve with 95% confidence intervals and the quasi-extinction risk for every scenario and county were stored after the simulation was done. Each of the thirteen counties (based on pre-2020 administrative county borders) represented a subpopulation (Table 2.1). This study only analyzed the largest (Nordland), an intermediate (Sogn & Fjordane), and the smallest (East-Agder) subpopulations under eight different scenarios (Scenarios 1, 4, 6,11, 23, 26, 28, and 33). These subpopulations and scenarios were selected to assess the impact of varying population under the absence of bycatch and the present of a 5% bycatch rate. The simulations and figures for alle the subpopulations with alle the scenarios are provided in the Appendix.

Since the PVA model was initially designed to simulate the predictions exclusively for the 13 subpopulations, adjustments were necessary to analyze the overall population viability. To achieve this, a specific possess was followed. Firstly, for each simulation replicate, the total population size was calculated. Once the total population size was computed, statistical analysis had to be conducted. Statistical measures such as standard deviation and confidence intervals were derived. These statistics were determined based on the distribution of the total population values obtained from each individual replicate.

The quasi-extinction probability $P_{QE,s,c}$ was calculated at the end of the simulation (i.e., after 100 simulated years) for each possible combination of scenario *s* and county *c* using equation 4 below:

$$P_{QE,s,c} = \frac{N_{QE,s,c}}{N_{s,c}} \qquad \text{(eq. 4)}$$

Here, $N_{QE,s,c}$ refers to the number of replicates in scenario *s* and county *c* that had a total population size below the quasi-extinction threshold QE, which was set to 100 animals. $N_{s,c}$ refers to the total number of replicates in scenario *s* and county *c*, i.e., 1000. This gives a total of 1300 distinct scenario-county specific P_{QE} estimates (13 counties/subpopulations x 100 scenarios). The population-wide quasi-extinction probability $P_{QE,s}$ was calculated for each scenario *s*:

$$P_{QE,s} = \frac{N_{QE,s}}{\sum_{c=1}^{13}\left(N_{s,c}\right)} \qquad \text{(eq. 5)}$$

Here, $N_{QE,s}$ represents the number of replicates in scenario *s* where the summed total of the population sizes of all counties/subpopulations was lower than the population-wide quasi-extinction threshold, which was defined as 13*QE = 1300 animals.

All analyses were performed using RStudio (RStudio Team, 2021) with R Statistical software version 4.1.1 (R Core Team, 2021), running on OS Ventura 13.5. Packages used included *data.table* (Dowle & Srinivasan, 2021) to handle large data sets. *tidyverse* (Wickham et al., 2019) and *reshape2* (Wickham, 2007) were used to reshape data frames and read CVS files. *progress* (Csárdi & FitzJohn, 2019) was used to create progress bars that provided feedback on the duration of the simulations. To generate plots and data visualizations *ggplot2* (Wickham, 2016) and *patchwork (Pedersen, 2023)* were downloaded. ChatGPT was used to assist in resolving errors related to plot creations, and to enhance sentence structure (OpenAI, 2023).

Every R script used to conduct analysis and create figures to visualize the results are available at https://github.com/celinanl/Master_thesis_H2023.git

3 RESULTS

3.1 POPULATION GROWTH WITHOUT ENVIRONMENTAL STRESSORS

The baseline scenario did not include any environmental stressors. This resulted in a typical logistic growth curve (Figure 3.1). The population increased rapidly in the first 22 years with an exponential growth rate. In this period the population almost doubled. The following years show that the growth rate started to stagnate as the population got closer to the carrying capacity, which was reached after 60 years. In the last 40 years of the simulation, the population of harbour seals was stable at the carrying capacity, which was calculated to be at a population size of 15 467 individuals (Table2.1). When the population reached the carrying capacity, the death rate was equal to the birth rate, meaning that the growth rate was zero.

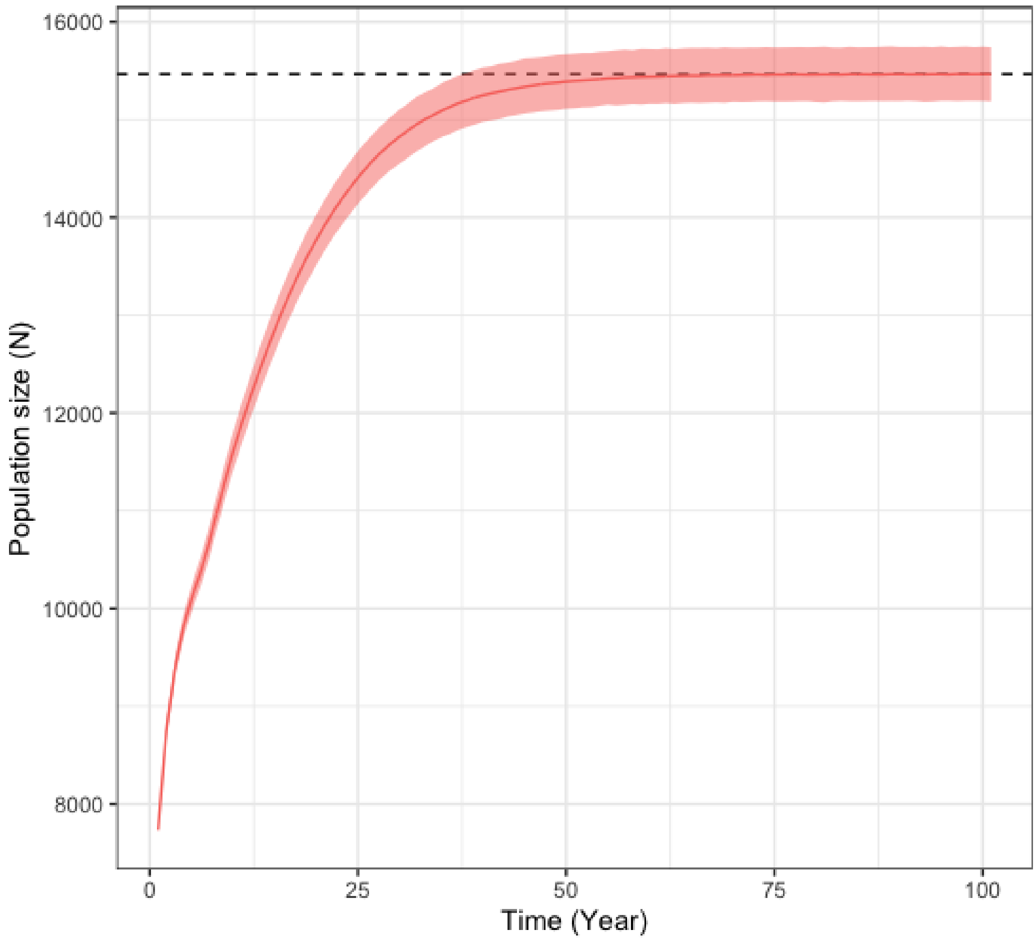

Figure 3.1: The plot illustrates the baseline scenario with no environmental stressors. The graph shows how many individuals the population of harbour seals contains over 100 years. The red line indicates the mean development while the shaded area shows the 95% confidence interval. The dotted black line represents the calculated carrying capacity for the population.

3.2 INTRODUCING HUNTING

In the scenarios that included hunting, disease, and fecundity reduction (scenarios 1-11), none of the populations reached the calculated carrying capacity (Figure 3.2). The absence of hunting pressure revealed that the reduction in fecundity and the probability of a disease outbreaks had a negative impact on the population growth of harbour seals (Figure 3.2, Scenario 1). Although population growth followed the same logistic growth pattern as illustrated in the baseline scenario (Figure 3.1), the simulation of scenario 1 ended after 100 years at a mean population size of 12 761. A 19% reduction from the carrying capacity.

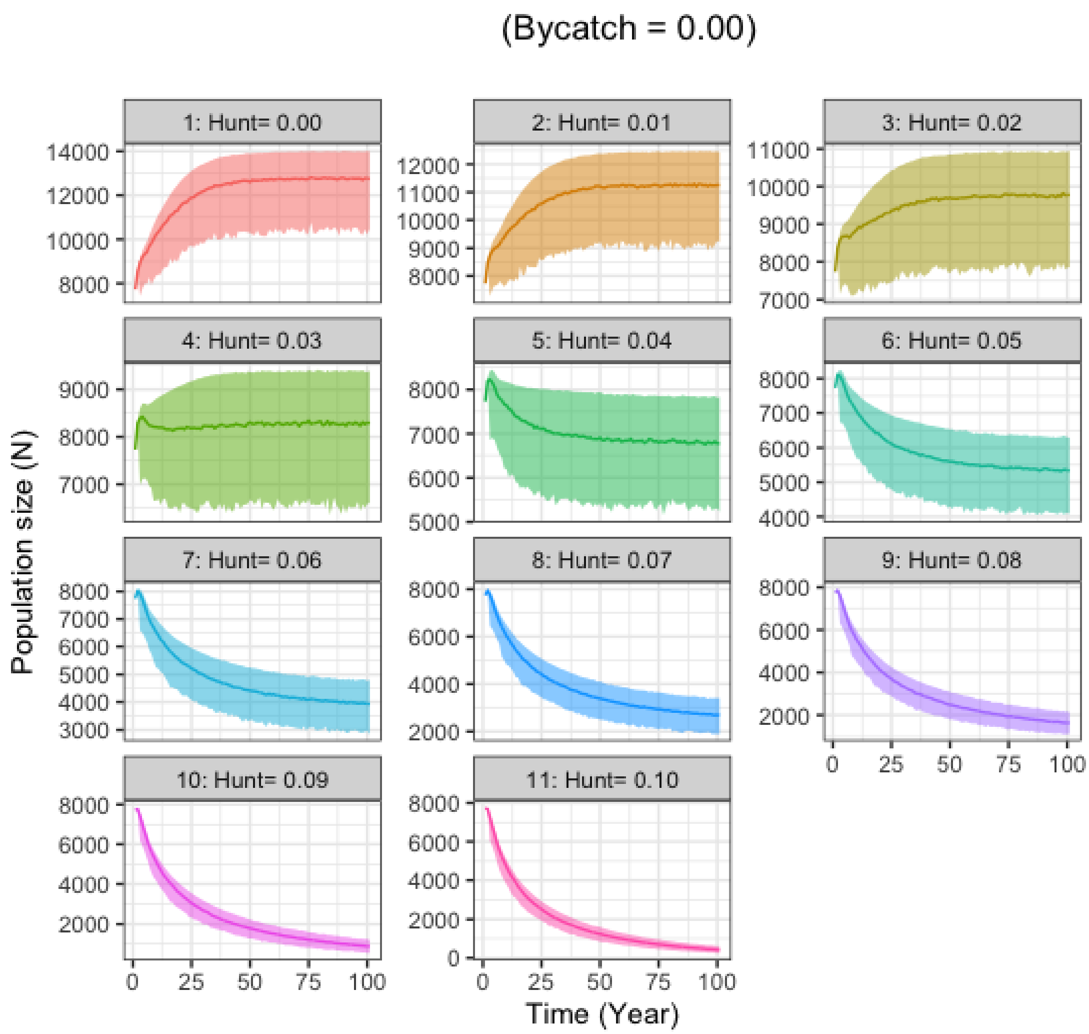

Figure 3.1: Estimated development of the Norwegian harbour seal population over 100 years under the eleven different levels of hunting while bycatch = 0.0 (scenarios 1-11). The graphs are represented with the mean (line) and the 95% confidence intervals (the shaded area). P_D = 0.07 and f=0.1.

The population was also affected by hunting. Figure 3.2 shows that in scenario 2 and 3, the population growths lead to mean population sizes of 11 258 and 9 786 individuals after 100 years. Indicating that more intense hunting pressure did result in a smaller population of harbour seals. The development no longer followed a logistic growth when the hunting level was increased to 3%, as shown in scenario 4. The population growth had a small increase in the first years before it quickly decreased and remained stable on a population size of 8 278 individuals. If hunting pressure was intensified even further, the population of harbour seals decreased, as illustrated in scenarios 5-11. The most extreme scenario is illustrated in scenario 11 when hunting is 10% of the population size. The population decreased rapidly in the first years and then at a slower pace towards a population size of 0, leading towards extinction.

Figure 3.3 reveals that the 95% confidence intervals to the population estimates represented in Figure 3.2 overlapped with each other. Scenario 1-3 had overlapping confidence intervals, meaning that there may not be a statistically significant difference between the scenarios. It is not possible to assert that there is a significant difference between 0%, 1%, and 2% hunting pressure based on these graphs alone. The figure clearly shows that confidence intervals decreased as the hunting level increased, and the population size decreased.

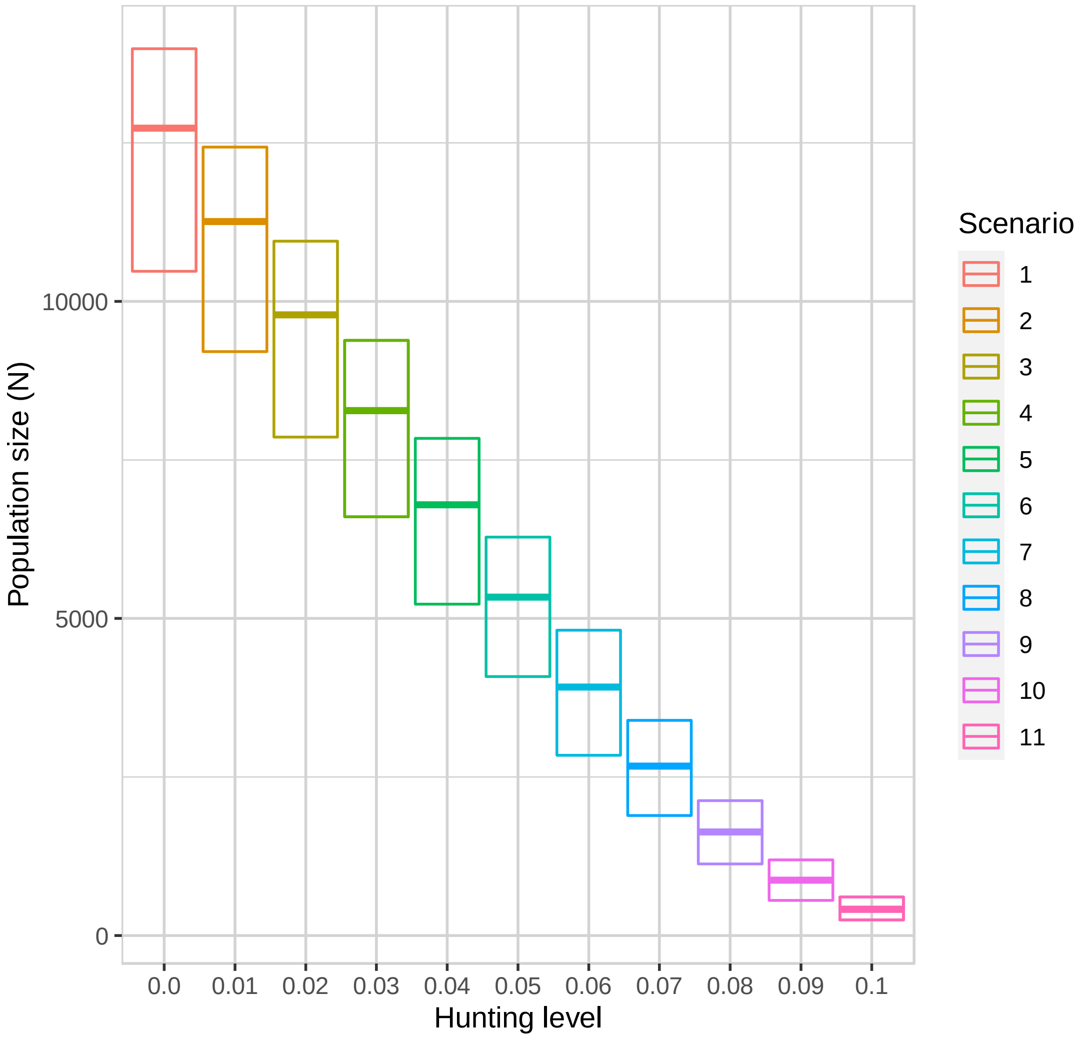

Figure 2.3: Boxplot of the mean population size and the 95% confidence intervals after 100 years in scenarios 1-11. The thick line in the middle of the box represents the mean, while the upper and lower lines of the box show the upper and lower limits of the confidence intervals.

The estimated populations in scenarios 1-8 (hunting pressure 0-7%) had a 0% probability of experiencing quasi-extinction after 100 years (Figure 3.4). If the hunting pressure reached 8%, there was a 10.2% probability that the population would experience quasi-extinction within 100 years. With a 9% hunting pressure, the quasi-extinction probability would increase to 99.5%. Among the 11 scenarios in Figure 3.2, the scenario with the most severe outcome was when the population was exposed to a hunting pressure of 10%. This situation led to a 100% probability of quasi-extinction.

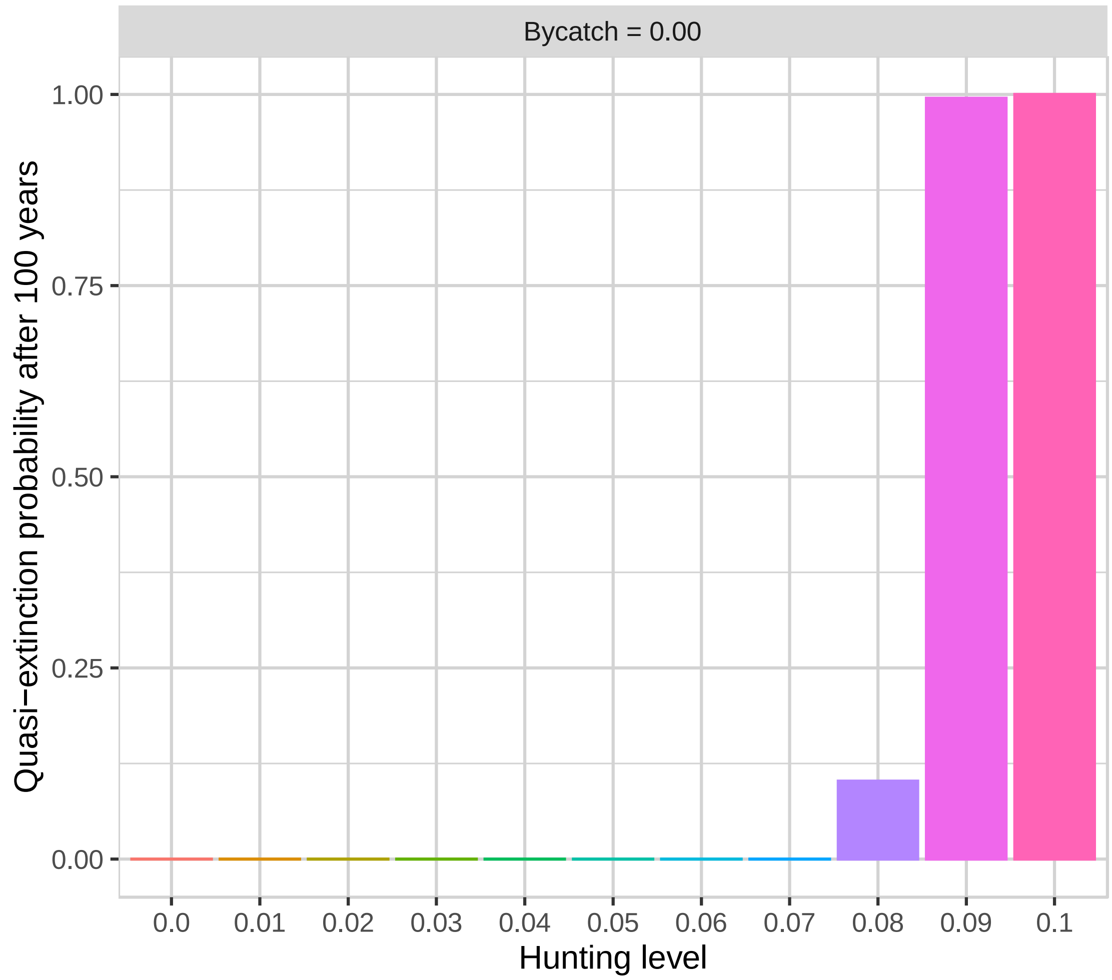

Figure 3.4: Bar plot showing the probability of quasi-extinction in scenarios 1-11 after 100 years.

3.3 INTRODUCING BYCATCH

The inclusion of bycatch resulted in reduced population sizes over the simulation period of 100 years (Figure 3.5). The mean population sizes in scenario 12 reached 8 998 in Setting A, and 10 431 in Setting B, which signifies a decrease in population size of 42.8% and 33.8% compared to the baseline scenario. The population also started to decrease at a lower hunting pressure when bycatch was introduced.

The mean population growth curves differed between the two bycatch settings. Setting B always ended at a higher mean population size after the simulations were done. Even when the hunting pressure was 10%, Setting B had a mean population size of 111.3 individuals while Setting A reached a mean population size of 55.2 individuals after 100 years. When the hunting pressure intensified, the distance between the two mean curves decreased. The difference in the simulated population size of Setting A and Setting B was not statistically significant due to the overlapping confidence intervals.

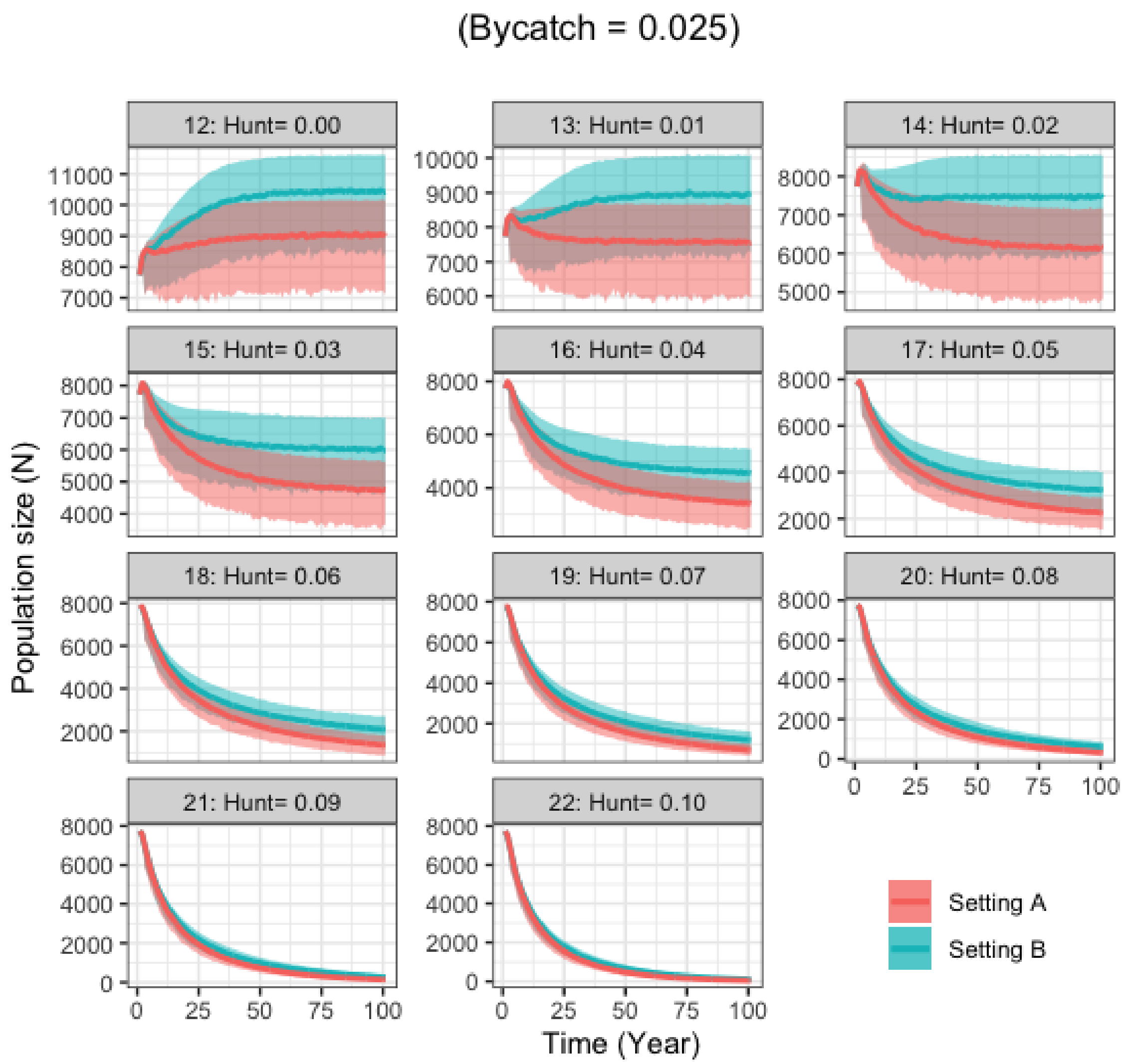

Figure 3.5: The plot illustrates the development of the population of harbour seals under increasing hunting levels when the bycatch rate = 0.025 (Scenarios 12-22). The line represents the mean, while the shaded area shows the 95% confidence intervals. The red curve signifies the population where the bycatch rate was spread equally between the age classes (Setting A). The blue curve represents the population where younger individuals had a higher bycatch probability than adults (Setting B).

The quasi-extinction risk for the estimated population differed between Setting A and Setting B in scenarios 12-22 (Figure 3.6). Both settings experienced a 0% probability of quasi-extinction with a hunting pressure of 0-4% of the population size (Scenarios 12-16). When the hunting pressure reached 5%, Setting A had a 0.3% probability of quasi-extinction, while Setting B still had zero chance. The contrast between the two populations was greatest with hunting levels at 6% and 7%. Setting A had a quasi-extinction risk at 45.1% and

100%, while Setting B had 0.8% and 66.4%. However, when the hunting pressure reached 8%, both settings had a 100% probability of experiencing quasi-extinction.

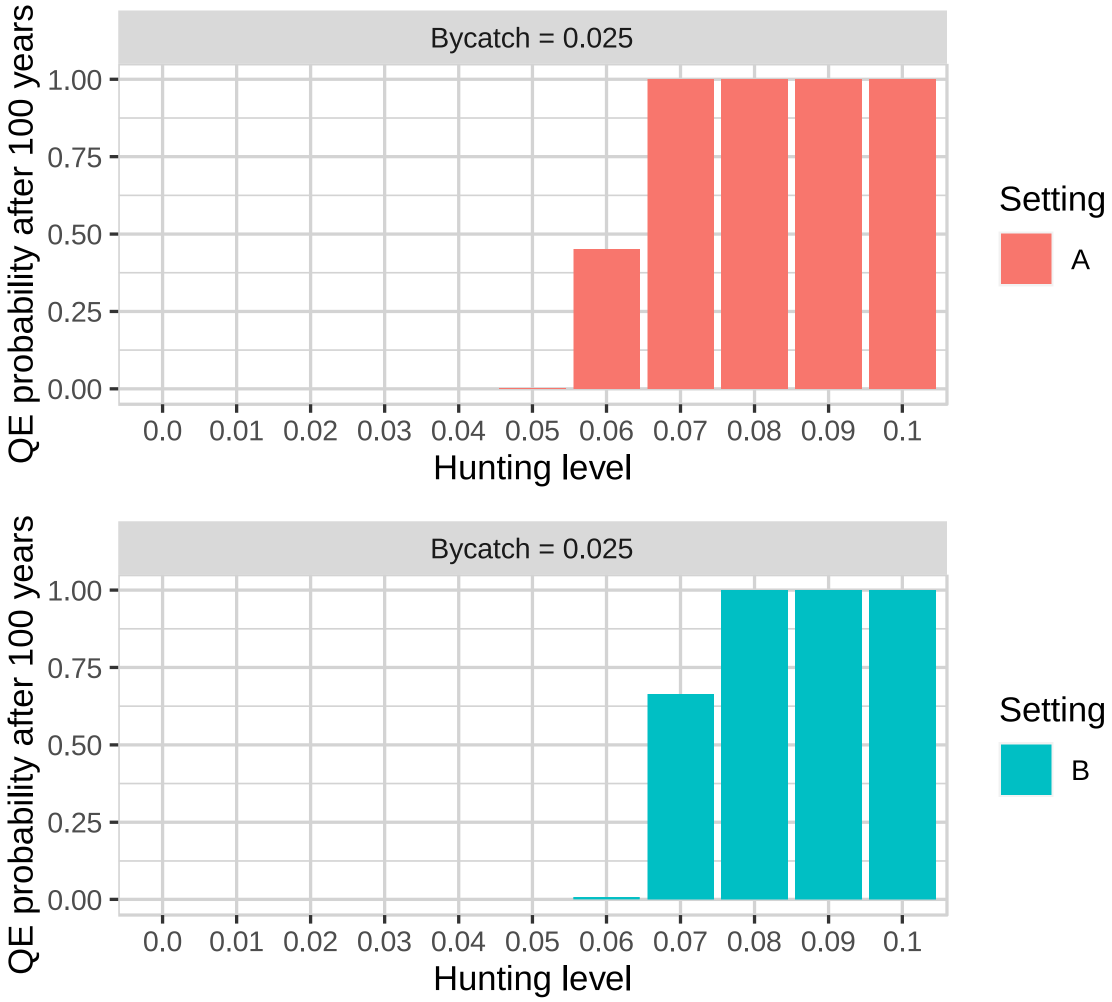

Figure 3.6: Bar plot showing the probability of quasi-extinction for scenarios 12-22 after 100 years based on the mean population size. The red bars indicate a population where the bycatch probability was equally distributed between the age classes (Setting A). The blue bars represent a population where younger individuals had a higher chance of getting caught in fishing gear than adults (Setting B).

When the bycatch rate was intensified to 5%, the contrast between the two settings became even more substantial (Figure 3.7). Even in the absence of hunting (scenario 23), the harbour seal population in the Setting A scenario would decline to a population level of 5 336 individuals. The harbour seal population in Setting B, scenario 23, didn’t experience much growth or loss as the curve had a flat development during the 100 years. The population ended at 8 161 individuals, which was a bit above the original population size. Still, Setting B also started to decrease when hunting was increased to 1%, which resulted in a final population size of 6 723 individuals (scenario 24). The 95% confidence intervals of settings A and B did not overlap with each other in scenarios 23-29, but after hunting reached 7% the growth curves exhibited a similar development.

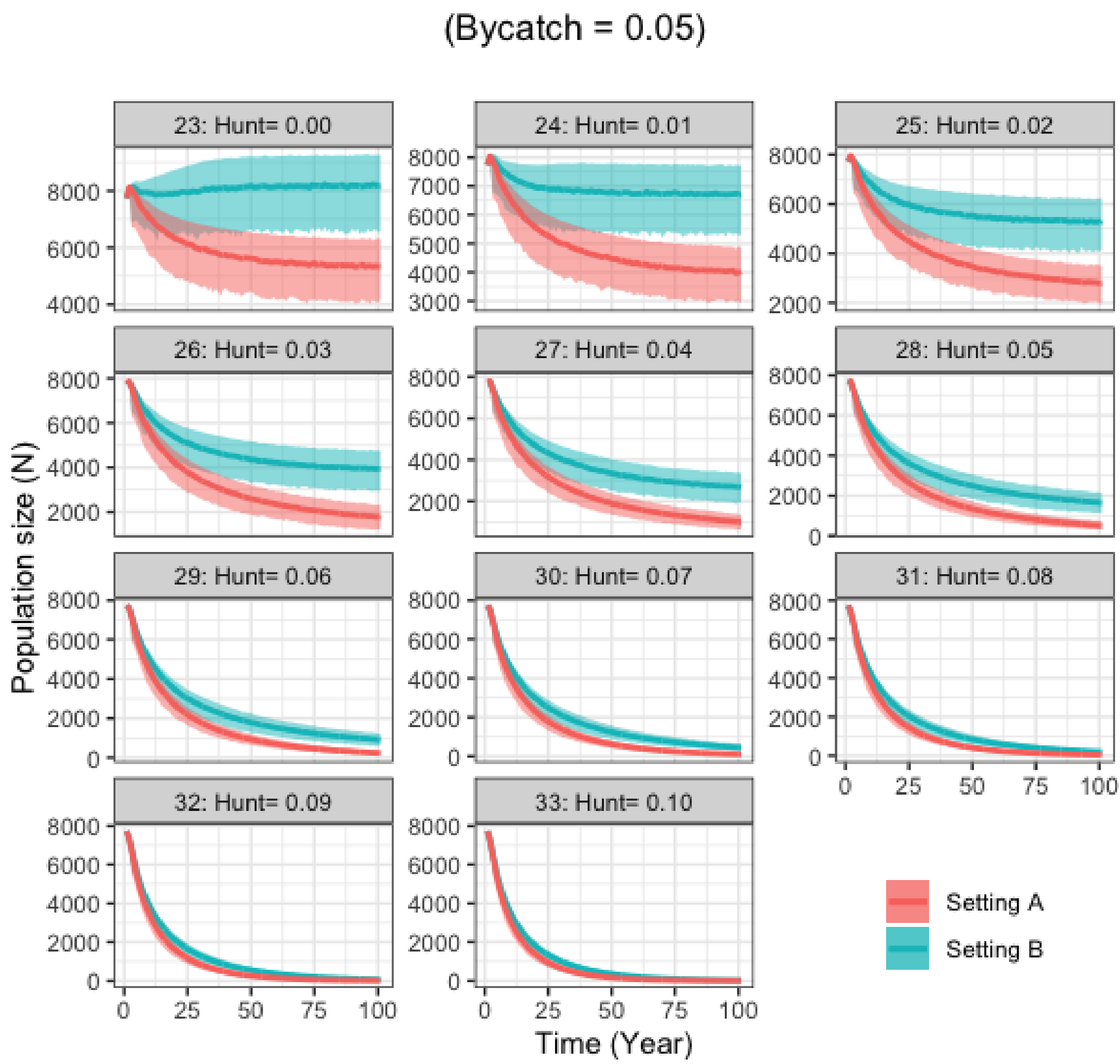

Figure 3.7: The plot illustrates the development of the population of harbour seals under increasing hunting levels when the bycatch rate = 0.05 (Scenarios 23-33). The line represents the mean, while the shaded area shows the 95% confidence intervals. The red curve signifies the population where the bycatch rate was spread equally between the age classes (Setting A). The blue curve represents the population where younger individuals had a higher bycatch probability than adults (Setting B).

The increase in the bycatch rate substantially increased the likelihood of populations facing quasi-extinction. 7/11 scenarios had a high or 100% probability of facing quasi-extinction in Setting A, while 5/11 scenarios experienced a high quasi-extinction probability in Setting B. Only the first three scenarios in Setting A, and the five first scenarios in Setting B had a 0% quasi-extinction probability.

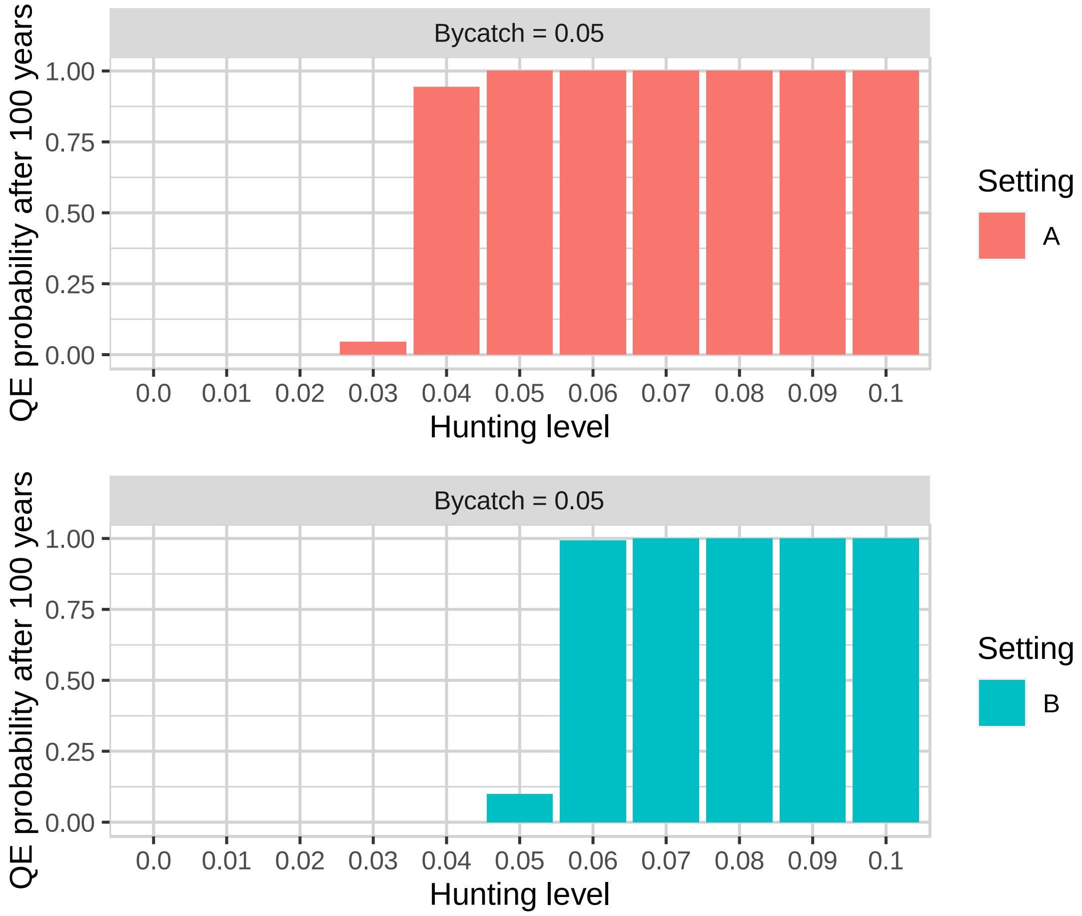

Figure 3.8: Bar plot showing the probability of quasi-extinction for scenarios 23-33 after 100 years based on the mean population size.

All the population estimates had a decreasing growth curve when bycatch was increased to 7.5% in both Setting A and Setting B (Figure3.9). The difference between the mean population size in Setting A and Setting B in scenario 34 had expanded even more. The final mean population sizes had a difference of 3 750 individuals, but the difference between the two growth curves decreased as the hunting level increased. There was a statistically significant difference observed in scenarios 34-39, whereas no such difference was evident in scenarios 40-44.

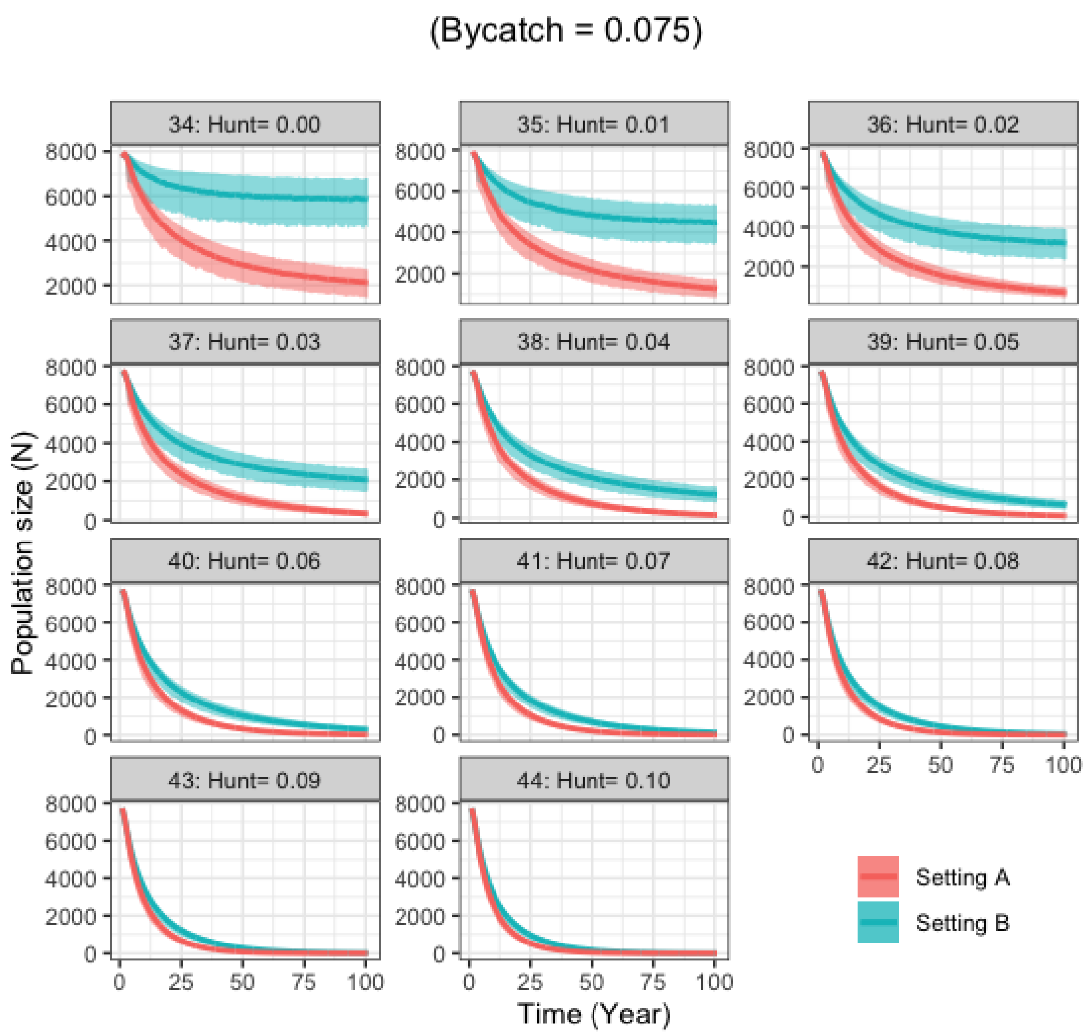

Figure 3.9: The plot illustrates the development of the population of harbour seals under increasing hunting levels when the bycatch rate = 0.075 (Scenarios 34-44). The line represents the mean, while the shaded area shows the 95% confidence intervals. The red curve signifies the population where the bycatch rate was spread equally between the age classes (Setting A). The blue curve represents the population where younger individuals had a higher bycatch probability than adults (Setting B).

A bycatch rate of 7.5% of the population size resulted in a larger proportion of scenarios with a high probability of quasi-extinction (Figure 3.10). 9 out of 11 scenarios exhibited a 100% probability of quasi-extinction in Setting A, while 6 out of 11 scenarios faced the same faith in Setting B. The estimated quasi-extinction risk was nonzero for Setting A in all these scenarios (Scenarios 34-44). The quasi-extinction risk for Setting B in the same scenarios, on the other hand was zero for low hunting pressures (Scenarios 34-36, H=0-2%).

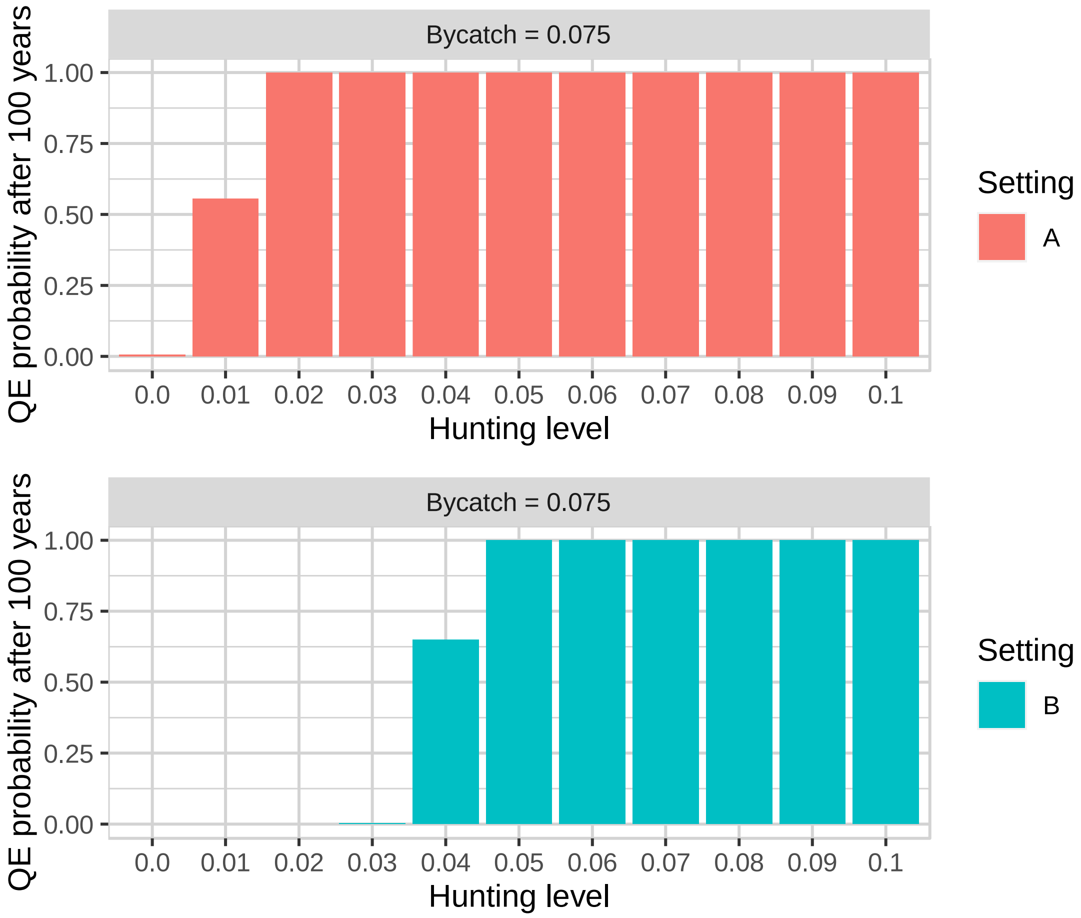

Figure 3.10: Bar plot showing the probability of quasi-extinction for scenarios 34-44 after 100 years based on the mean population size.

The most extreme population developments are presented in Figure 3.11 with a bycatch rate of 10% of the population size (Scenarios 45-55). All the population estimates in Setting A and Setting B had a negative growth rate, but the curves were steeper for the Setting A scenarios than for the Setting B scenarios. In other words, the Setting A scenarios always ended at a smaller population size than the Setting B scenarios did. Scenario 45 provided the strongest example on this matter, as the final mean population sizes were 415 (Setting A) and 3 539 (Setting B). Again, the same pattern occurred where the difference between the mean population size in Setting A and Setting B decreased as hunting pressure intensified. In scenario 55 the difference between the two growth curves was only 1.4 individuals.

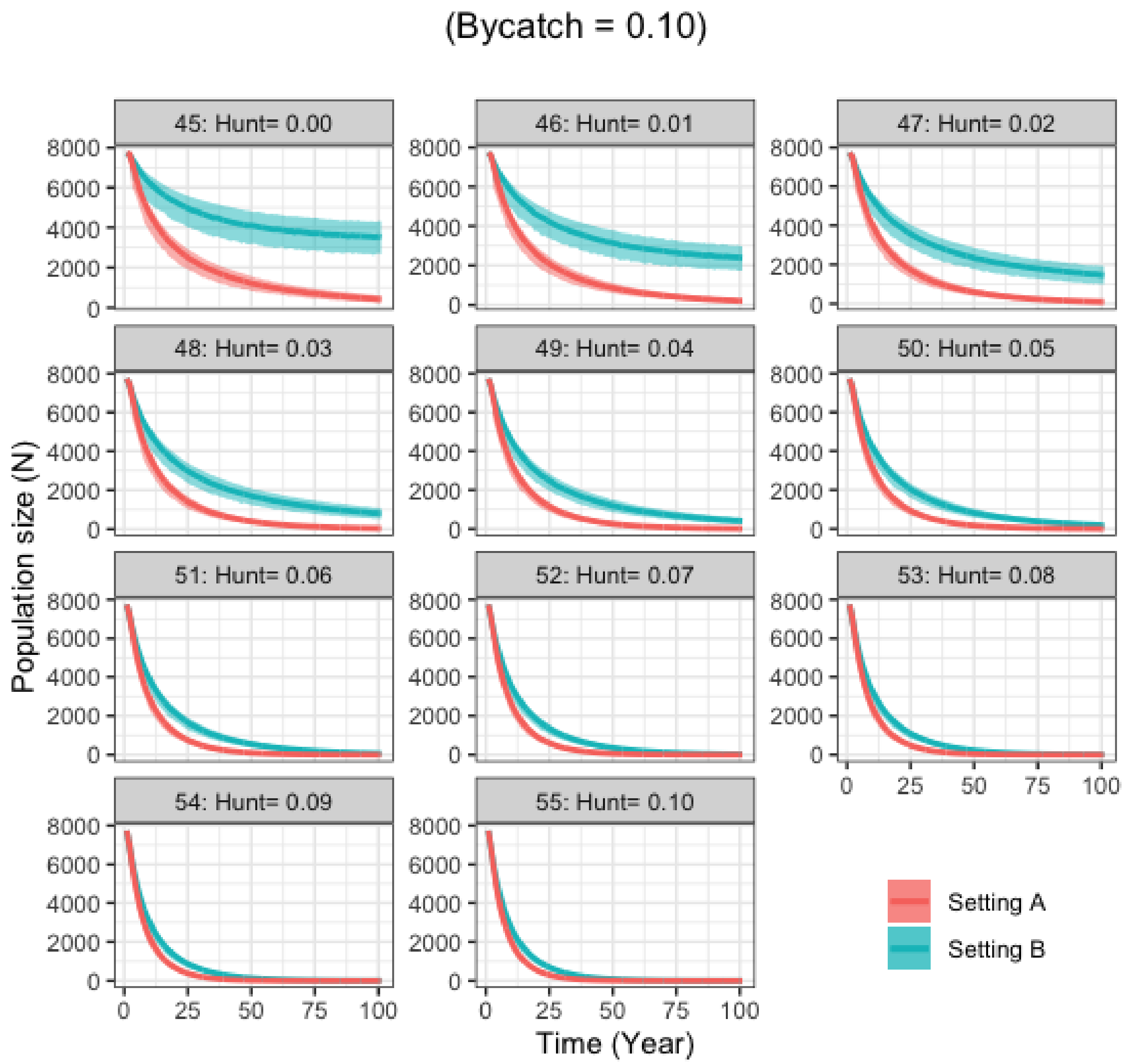

Figure 3.11: Figure 3.11: The plot illustrates the development of the population of harbour seals under increasing hunting levels when the bycatch rate = 0.10 (Scenarios 45-55). The line represents the mean, while the shaded area shows the 95% confidence intervals. The red curve signifies the population where the bycatch rate was spread equally between the age classes (Setting A). The blue curve represents the population where younger individuals had a higher bycatch probability than adults (Setting B).

The consequences of the intense hunting and bycatch rate were also well illustrated when considering the quasi-extinction risk (Figure 3.12). All the population estimates in Setting A had a 100% probability quasi-extinction. 8 out of 11 scenarios had the same outcome in Setting B (Scenarios 48-55, H=3-10%).

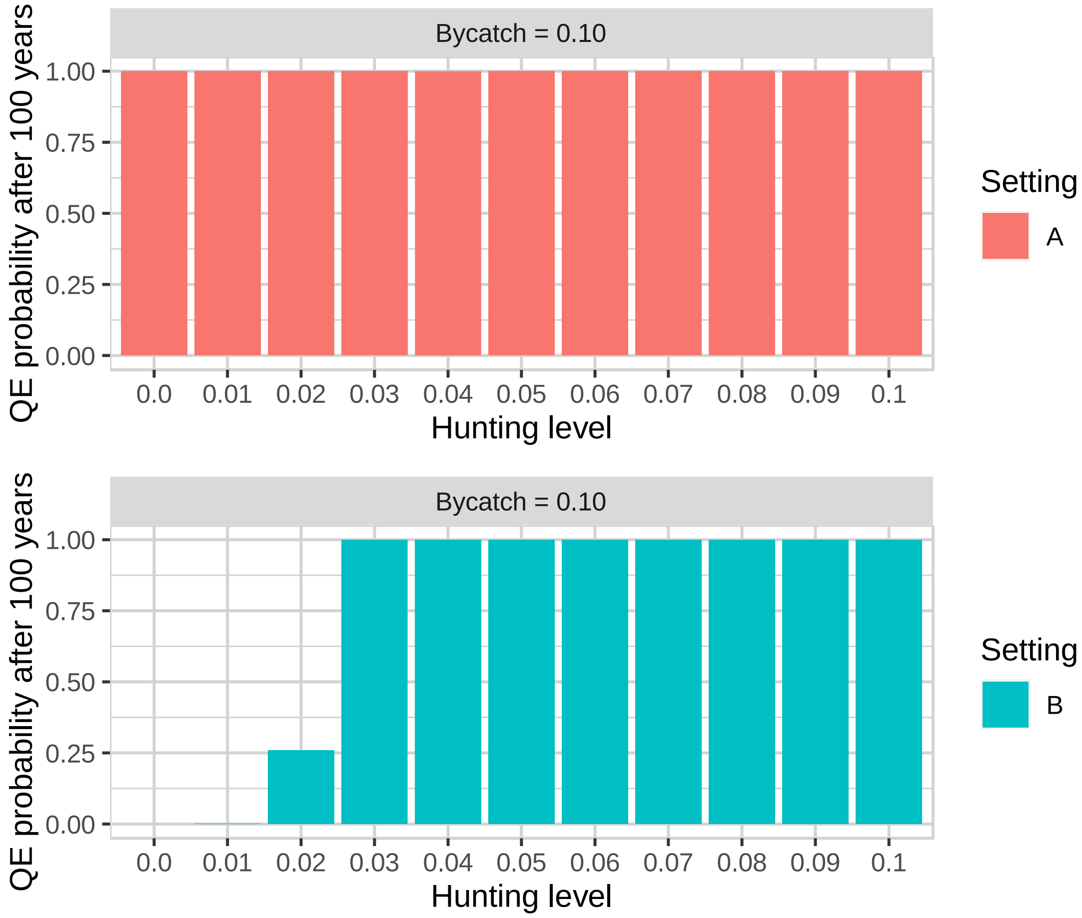

Figure 3.12: Bar plot showing the probability of quasi-extinction for scenarios 45-55 after 100 years based on the mean population size.

3.4 SENSITIVITY ANALYSIS

The estimated population growth was affected by the fecundity reduction and disease outbreak (parameters *f* and *P_D*) in both scenarios and both settings (Figure 3.13). Increasing values of *f* had a decreasing effect on estimated population size after 100 years in scenario 27 and 28. There was greater variation in the data when *P_D* equaled 0.07, as the confidence intervals were wider. In scenario 27 and 28A the confidence intervals of $_d$ and $_e$ overlapped, indicating that there was not a statistically significant difference in the presence of *f* together with *P_D.* In Setting A, each combination of Scenario 27 and 28 deviated substantially from the initial population size. 27B and 28B had a greater difference between B_a (*f=0.0, P_D=0.0*) and B_e (*f=0.10, P_D=0.07*).

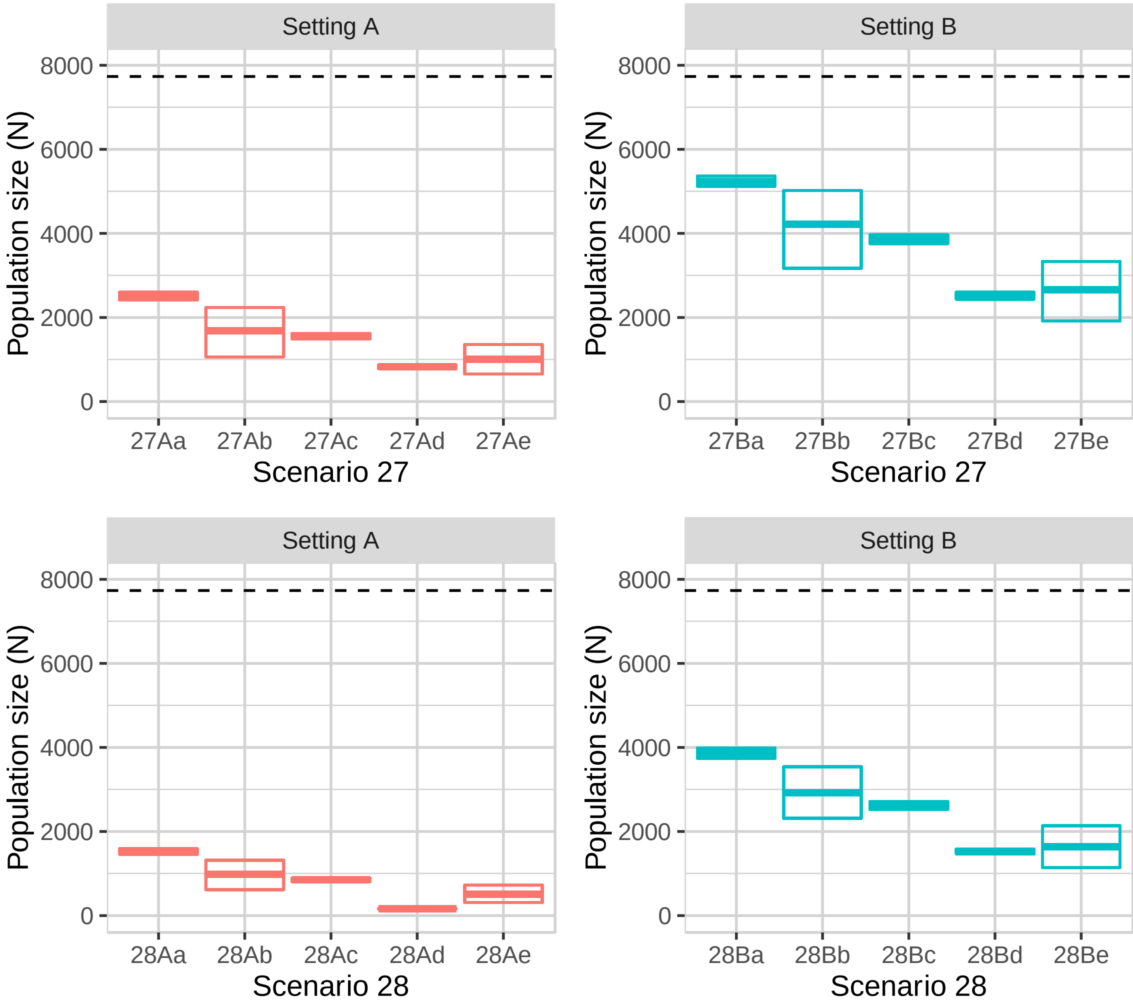

Figure 3.13: Sensitivity analysis of Scenario 27 and 28 presented in a box plot. Every box represents the final mean population size with 95% confidence intervals after 100 years. The different boxes represent a unique combination of the parameters f and P_D (Table 2.6). The dotted line represents the initial population size (7733 individuals).

3.5 STUDYING THE SUBPOPULATIONS

The initial population sizes varied among the three counties presented in Figure 3.14. However, the population trajectories for the different subpopulation were similar. When bycatch was absent (Figure 3.14a), a 0% hunting pressure resulted in a 64% increase in the population size in all 3 counties. With a 3% hunting pressure, all the subpopulations remained stable with a growth rate close to zero. There was a 31% decrease in the population size when hunting pressure reached 5%, and a drastic fall of 94% in the population sizes when the hunting pressure was doubled to 10%. None of the confidence internals overlapped with each other, indicating that there may be a statistically significant difference between the scenarios.

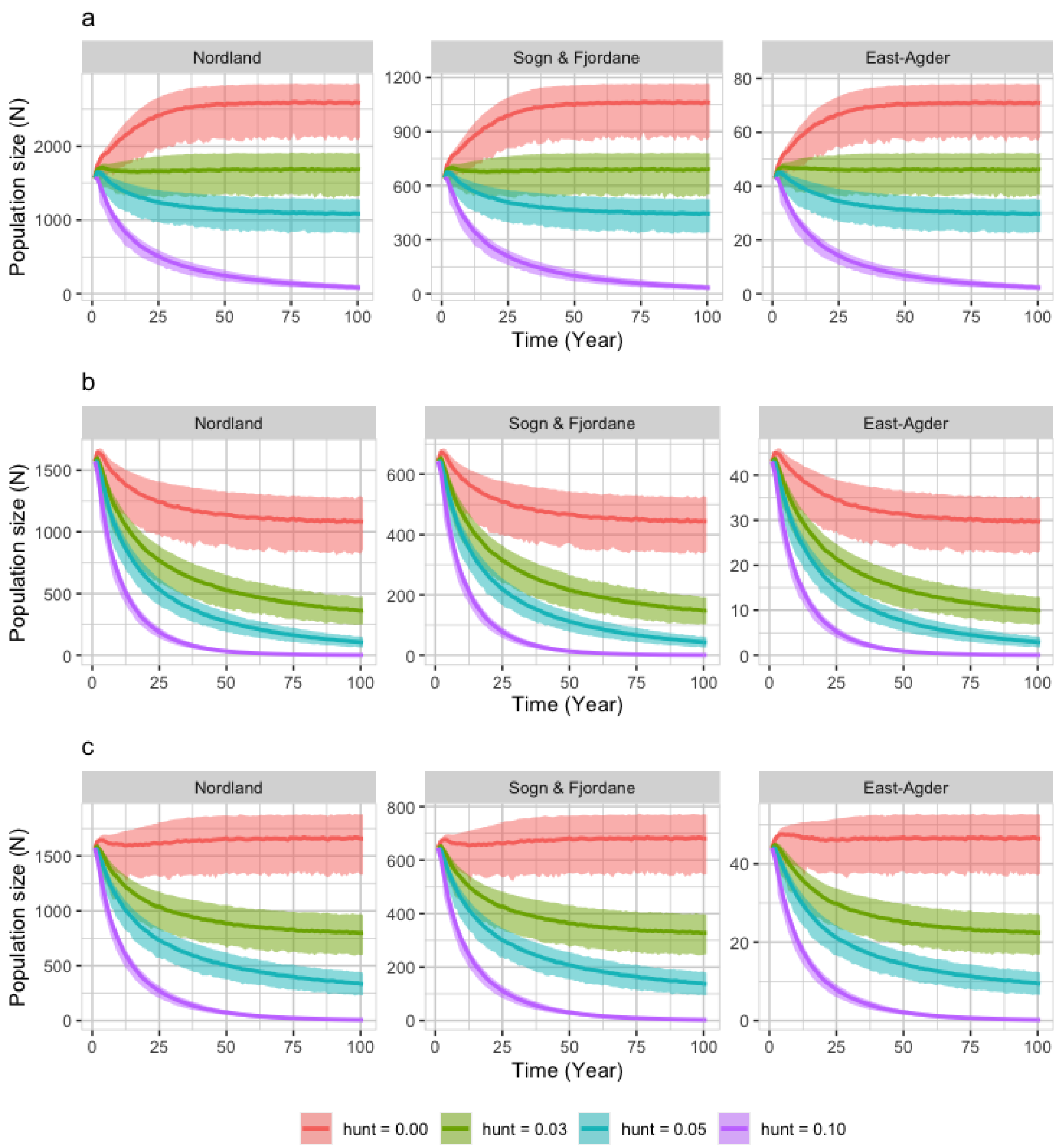

Figure 3.14: Development of the Norwegian harbour seal by county with four different hunting levels when bycatch is not present (plot a) and when bycatch equals 5% with Setting A (plot b) and Setting B (plot c). Red lines = 0% hunting. Green lines = 3% hunting. Blue lines = 5% hunting. Purple lines = 10% hunting. The lines indicate the mean growth of the population and are represented with 95% confidence intervals (shaded area).

There were great variations between the counties when looking at the quasi-extinction probability (Figure 3.15). East-Agder had the smallest initial population size with 43 individuals. The populations never managed to exceed the set quasi-extinction threshold, resulting in a 100% quasi-extinction risk in all the scenarios. The population in Nordland, which had the largest initial population size with 1570 individuals, exhibited the least likelihood of facing quasi-extinction. Even at a 10% hunting pressure, and no bycatch, the population only had a 78% probability of facing quasi-extinction, while the others had a 100% probability when bycatch was not present (Figure 3.15a).

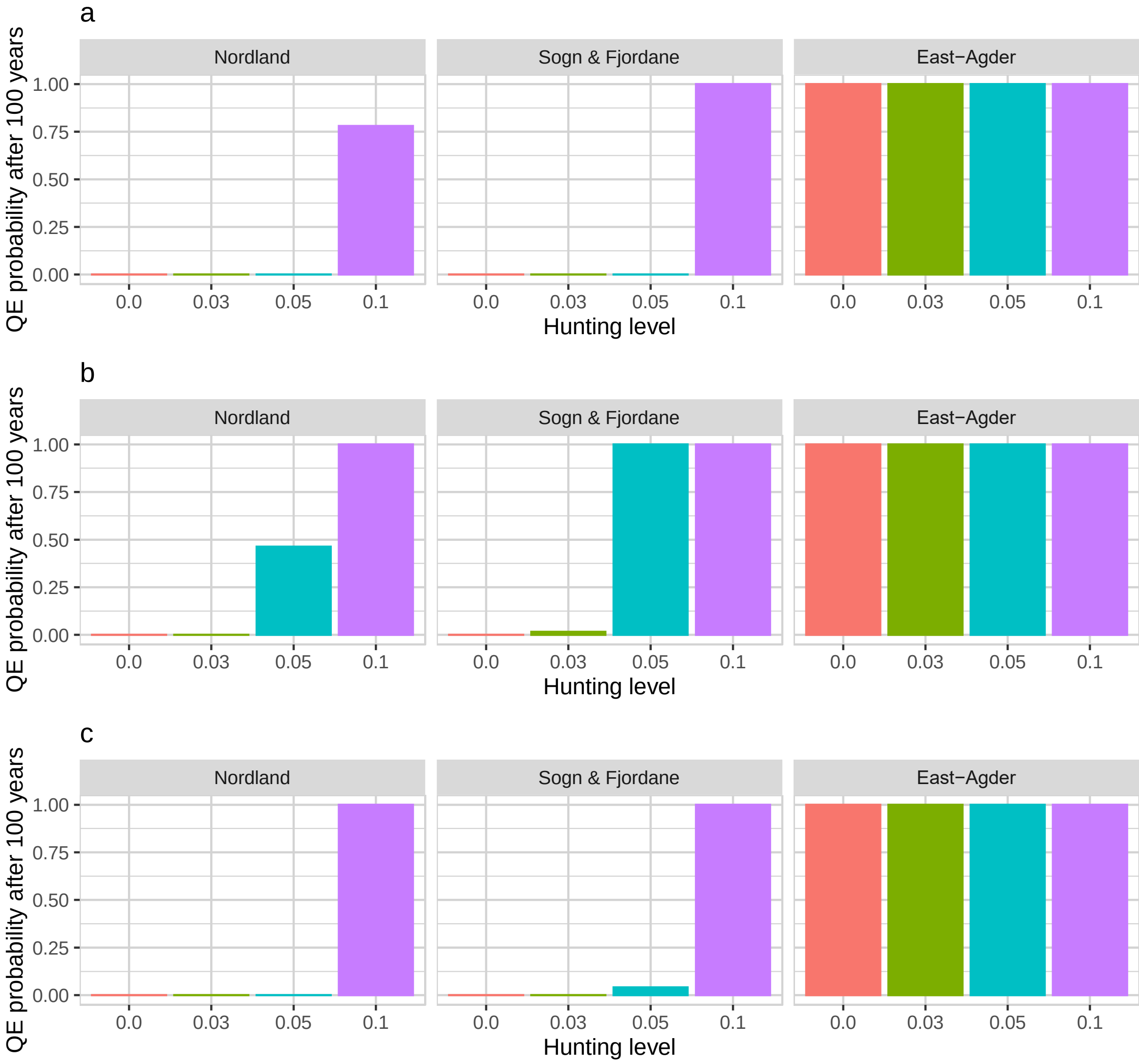

Figure 3.15: Bar plot showing the probability of quasi-extinction for each scenario in each county represented in Figure 3.14 after 100 years based on the mean population size. Plot a illustrates the quasi-extinction probability without the presence of bycatch. Plot b represents the quasi-extinction probability with a 5% bycatch rate in Setting A. Plot c represents the quasi-extinction probability with a 5% bycatch rate in Setting B.

4 DISCUSSION

4.1 THE EFFECT OF ENVIRONMENTAL STRESSORS ON A LARGE POPULATION

A removal of 455 individuals, which is the recommended national harvest quota for harbour seals in Norway, is equivalent to a hunting quota just below 5% of the estimated population. The results from this study indicate that neither a 4% nor a 5% hunting quota correspond with the government's management goals for harbour seals. Scenario 5 and 6 in Figure 3.2 clearly illustrates a declining growth trend, with a mean population size of 6 794 and 5 338 individuals after 100 years. The actual reported removal of 350 individuals, corresponds with a hunting quota of 3.5%. It has already been established that the 4% hunting quota did not meet the desired requirements. A hunting removal of 3%, however, resulted in an increase of 8 278 individuals after 100 years (Figure 3.2, Scenario 4). It is not possible from these results to establish the exact hunting quota that will result in a final population size of 7 000 individuals. However, these population estimates did not consider bycatch.

The average annual bycatch of harbour seals in Norway has been estimated to be 555 seals (Bjørge et al., 2017). This corresponds to a bycatch rate of 5.5% of the estimated abundance. Scenario 23 in Figure 3.7 shows the population estimate when the bycatch rate was equal to 5% and the hunt was 0%. Setting A (red curve) shows a declining population trend with a mean population size of 5 336 individuals after 100 years. The population in the Setting A scenario declined 31% over 100 years and did not meet the terms of the management goals. The blue curve, which represents Setting B, shows a straight curve, and therefore a stable population growth over 100 years. The result implies that a bycatch of 555 seals annually complies with the management goals that Norway has agreed to. However, this raises the question of how a bycatch level of 5% with Setting B can be considered sustainable when a hunting quota of 5% is not. The explanation is that the hunting resulted in a higher mortality rate of adult seals than the bycatch in Setting B. The removal of older seals has a more negative effect on the population than the loss of younger individuals. This will be discussed further in chapter 4.2.

Up to this point, the impact of hunting and bycatch have been discussed separately. In nature, however, these two factors co-occur. This means that a total of 905 harbour seals are removed from the population annually when considering the reported harvest. Scenario

27 in Figure 3.7 presents the population growth with a hunting quota of 4% and bycatch rate of 5% relative to the population size. This scenario best represents the annual harvest of 350 harbour seals and the annual bycatch of 555 harbour seals. The population was heavily reduced, containing 1 006 individuals (Setting A) and 2 665 individuals (Setting B) after 100 years with an estimated 94% probability of experiencing quasi extinction in Setting A and 0% probability in Setting B (Figure 3.8). With the Setting A scenario, the population was reduced to 87% of the initial population size after 100 years. The species would then be categorized as "critically endangered" with an extremely high risk of going extinct. The Setting B scenario resulted in a 66% reduction in mean population size after 100 years, classifying the population as "endangered" on the red list, and it would be at a very high risk of going extinct according to the IUCN red list criteria. If the reported hunting had been equal to the recommended hunting quota (455 harbour seals annually), the outcome would have deteriorated (Figure 3.7, Scenario 28). The final population size decreased 93% to 508 individuals (Setting A) and 76% to 1 637 individuals (Setting B). The quasi-extinction probability increased to 100% (Setting A), and 10% (Setting B).

Based on the findings of this study, it is recommended to decrease the hunting quota to a removal between 3% and 4% to be able to sustain a stable population of harbour seals in Norway. This will correspond with an annual harvest quota of 350 individuals, which resembles the reported annual removal of harbour seals. However, this only applies when looking at hunting alone. When bycatch at today's level (5%) is considered, the results reveal that hunting harbour seals in Norway with the objective of maintaining a stable population of 7 000 individuals is not sustainable. This applies to both settings. To allow hunting on harbour seals in Norway, the bycatch rate should be reduced to 2.5% based on the results provided in this study (Figure 3.5). Even then, the hunting quota can only be set at 1% (Setting A) or to a maximum of 2% (Setting B) to maintain a stable population of harbour seals along the Norwegian coast.

4.2 AGE-DISTRIBUTED BYCATCH, OR NOT?

The age-specific bycatch rate was used to assess how the population was affected by distributing bycatch equally among the 38 age classes, versus when it had the greatest impact on younger individuals. When comparing Setting A and Setting B, the results showed that the age-specific bycatch rate had an impact on population growth. In the scenarios that included bycatch (Scenarios 12-55), Setting B always had a higher population growth than Setting A after 100 years. The impact of the age-specific bycatch rate on the population trajectories became more significant as the bycatch rate increased (Scenarios 12, 23, 34 and 45). However, there was an overlap in the confidence intervals when the bycatch rate was set to 2.5%. This indicates that there may not be a statistically significant difference between the simulated trajectories in setting A and setting B in scenarios 12-22 after 100 years. It is therefore not possible to assert this matter based on these graphs alone (Figure 3.5). Nevertheless, the pattern that is observed between Setting A and B can be explained by some ecological concepts.

An explanation for the observed pattern can be the number of females that are removed from the population. The most critical group within a population is the prime-aged females since they have the greatest potential to affect population growth and size (Gaillard et al., 1998; Van de Walle et al., 2021). In Setting A, there is a bigger proportion of fertile females that were removed due to bycatch than in Setting B. A selective removal of fertile females has proven to severely affect the population dynamic (Van de Walle et al, 2021). That may also explain why the difference in Setting A and B decreased as hunting increased. The hunting rates in this PVA were not age-specific, which signifies that a proportion of fertile females were lost in hunting in both settings. As the hunting quota rose, there was a corresponding increase in the mortality of fertile females, which in turn had a greater negative impact on the population.

Now that the cause of disparity in estimated population growth between Setting A and B has been explained, it is pertinent to inquire about what the most realistic bycatch distribution within a harbour seal population is. The bycatch model in Setting B assumed that seal pups were more exposed to bycatch incidents compared to adults. This trend has also been observed in another coastal seal species, the grey seal *(Halichoerus grypus) (*Murray et al., 2021*)*. The age distribution in the population may explain the strongly biased bycatch rate. Studies of harbour seals reveal that the highest occurrence within a population is among individuals aged less than one year (Lydersen & Kovacs, 2005). This age distribution was also reflected in this study (Figure A1).

Another study revealed that most bycatch incidents on harbour seals in Norway happened between August and October (Bjørge et al., 2017). After newborn pups are abounded by their mother and during the moulting season. Since harbour seals do not moult their first year, the high bycatch numbers in these months may consist of a high number of pups, as most of the adults and juveniles spend most of their time on land (Bigg, 1981). The pups are abandoned when they are approximately one month old. They have not gathered much experience and may therefore not understand the danger of fishing vessels. Instead, their curiosity and lack of experience may make them more exposed to bycatch incidents. It may also be the fact that young seals lack the strength to break free from the nets (Murray et al., 2021). While previous studies reveal that younger seals are more commonly taken as bycatch, and the results from this study align better with field observations when using an age-distributed bycatch rate, further research is needed to enhance and make realistic age-specific bycatch rates.

4.3 DIVERGING DATA: DISCREPANCIES BETWEEN STUDY RESULTS AND IMR SURVEYS

The results provided in this study do not coincide with the results from recent coastal surveys conducted by the Institute of Marine Research (IMR). The IMR has reported a stable population of harbour seals of approximately 7 000 individuals since the last outbreak of the PDV virus (Nilssen et al., 2020), with the last counting survey showing a population of 7 733 harbour seals. The hunting harvest has not fluctuated much since 2010, with an average removal of 350 annually (Nilssen et al., 2020). There has not been reported a change in the annual bycatch of harbour seals either, but unfortunately, there exists not much data on this matter. Due to a lack of data, it is possible that the bycatch rate has decreased, which could explain the discrepancy. However, the results have shown that the bycatch rate had to be close to zero to maintain a stable population size of ≥ 7 000 individuals with the harvest that has been registered in the past years. A bycatch rate at such a low level, however, is highly unlikely.

4.3.1 Reduction in fecundity and probability of disease outbreak

Another potential cause of the discrepancy between the results and the IMR data may arise from the incorporation of the possibility of disease outbreak P_D in the PVA model, together with the reduction in fecundity due to pollutants f. In accordance with anticipations, the sensitivity analysis conducted in this study provided evidence that *f* reduced the estimated population size when increased from 0% to 10% (Figure 3.13). Furthermore, the estimated population underwent an even greater decline when *f* was adjusted to 20%. This could mean that the value used for *f* in the simulations (10%) was too high for the studied population of harbour seals. It is reported that PCB pollutants have had a decreasing trend in Norwegian waters (Frantzen et al., 2022). However, the concentrations of pollutants in aquatic animals are still high, as they are transferred and accumulated through the food web and through breast milk (Frantzen et al., 2022; Jepson et al., 2016). Although PCB concentrations in harbour seals decreased after it was banned from Europe, reports confirm that the concentration has stagnated since the 1990s (Jepson et al., 2016; Shawa et al., 2014). This proves the appropriateness of including the parameter *f* in the simulations, although the discrepancy is reduced when the parameter is omitted. Hence, the source of the discrepancy must be situated elsewhere.

The possibility of disease outbreaks could be the factor contributing to the discrepancy. High death rates were reported after the two outbreaks of the Phocine Distemper virus (PDV) (Härkönen et al., 2002), but no outbreaks of any kind have been observed since the last outbreak in 2002. The three disease parameters in this study had values that would correspond with the reported outcome of the two PDV epidemics (Silva et al. 2021). Incorporating such a catastrophic event in the model could result in the estimated population growth having a more declining trend than what has been observed in nature since 2002. Figure 3.13 shows that the inclusion of *P_D* causes an estimated population with a lower final mean population size after 100 years than if *P_D=0*. This result holds true whether *f* is included or not. The results show that incorporating the possibility of a disease outbreak in the simulations causes discrepancies compared to field observations. However, a substantial disparity between the estimated population size and the initial population size persists when *P_D=0* and *f=0.1*. This means that there are additional factors, aside from *P_D,* that contribute to the observed discrepancy.

4.3.2 Migration

Migration from other populations outside of Norway has not been considered in this study. If the Norwegian population is experiencing immigration from foreign populations, it would positively influence population growth by introducing a greater number of new individuals.

DNA analysis of European harbour seals has unveiled the existence of six population units in European waters (Goodman, 1998). While migration within these groups is high, migration between these groups is low (Goodman, 1998). Norway was part of the western Scandinavian population together with Skagerrak, Kattegat, and the West Baltic (Goodman, 1998). Immigration from these parts is therefore likely to occur, whereas immigration from other populations in Europe is improbable. While it is established that harbour seals immigrate to Norway, it is important to consider that harbour seals also emigrate from Norway. If the immigration is equal to the emigration there will be no net positive effect on the population growth. Should this be the circumstance, migration cannot explain the discrepancy between the results presented in this study and the data obtained from the IMR.

Because migration from other populations was not considered in this study, all the seals removed from the population due to hunting and bycatch were assumed to be Norwegian harbour seals. Given documented migration between Norwegian and Swedish/Danish territories (Goodman, 1998), some of the seals harvested or caught as bycatch may have been migrating from these regions. Should this be the case, it would positively influence population growth because it would result in fewer Norwegian harbour seals being removed from the population. However, the average migration distance for harbour seals is 58km away from their haul-out site (Wiig & Øien, 1988). Based on this, the migrating seals from Sweden and Denmark will normally stay within the Norwegian Skagerrak area. Most of the bycatch scenarios in Norway occur in areas where the frequency of seals is highest (Moan, 2016). The highest hunting quotas are also given out in these regions (Nilssen et al., 2020). Data from the Coastal Reference Fleet (CRF) indicates that most bycatch scenarios occur on the west coast of Norway. More specifically areas outside of Lofoten, Trøndelag, Møre & Romsdal, and Stavanger (Bjørge et al., 2017; Moan, 2016). This means that most of the seals caught as bycatch or harvested happen in areas where there is very little migration activity from Swedish/Danish regions. Hence, there is a minimal possibility that seals removed from the population in these regions originate from locations other than Norway.

Bycatch scenarios have also been registered in areas where migration activity occurs (the Norwegian Skagerrak area) (Bjørge et al., 2017; Moan, 2016). There is therefore a possibility that seals from Swedish/Danish regions are caught as bycatch or harvested in this area.

However, the effect of foreign seals removed in this area is likely to have an insignificant impact on population growth. First of all, the number of bycatch scenarios in the Skagerrak area is notably lower in comparison to the remainder of Norway (Bjørge et al., 2017). Second, harbour seals normally stay close to their haul-out site (Bigg, 1981; Wiig & Øien. 1988), so the proportion of foreign seals in relation to Norwegian seals will likely be very small. This implies that the likelihood of capturing or harvesting a substantial number of foreign seals in the Norwegian Skagerrak area, is quite low.

The cause of the discrepancy between the results presented in this study and the field observations done by the IMR cannot be explained by one factor alone. There must be several factors that lead to the disparity. The presence of the probability of a disease outbreak and a not age-distributed bycatch rate have proven to be two of these factors. If the model had included migration from foreign populations together with a possible lower bycatch rate, the results would perhaps be more consistent with the observed field data.

4.4 THE DANGER OF BEING FEW

Under the different hunting and bycatch rates, the PVA predicted that the subpopulations would undergo a similar growth pattern (Figure 3.14). Despite this, there exists a substantial disparity in the results that will lead to varying impacts on the subpopulations. For instance, in the scenario with 10% hunting and 0% bycatch (Figure 3.14a), all three subpopulations experienced a 94% decline in population size. However, the final mean population size was 87 (Nordland), 35 (Sogn & Fjordane), and 3 (East-Agder) individuals. The difference in the final mean population size influenced the viability of the population. This is shown in Figure 3.14a where Nordland experienced a 78% quasi-extinction probability, while the two others had a 100% probability. The subpopulation in Nordland had a better tolerance for hunting and bycatch than East-Agder and Sogn & Fjordane (Figure 3.15), indicating that small populations have a higher extinction risk than large populations when facing stressors. The same was found in a similar study carried out by Silva et al. (2021). The study showed that smaller populations of harbour seals reached the quasi-extinction threshold at lower hunting pressures than larger populations (Silva et al., 2021). In summary, the results indicate that even though the total population of harbour seals that resides in Norway is large, it consists of subpopulations that differ in abundance. Hunting and bycatch may therefore result in vastly different outcomes in different regions.

East-Agder had the smallest subpopulation of harbour seals in Norway with only 42 individuals present. The small number of seals lead to the worst outcome when looking at the quasi-extinction probability (Figure 3.15). According to the PVA, the subpopulations would have a 100% probability of experiencing extinction in all the scenarios. Even in the scenario where hunting and bycatch were absent. This is because the subpopulation never managed to exceed the quasi-extinction threshold of 100 individuals. However, the results are again inconsistent with the data from the IMR. Nationwide censuses show that the subpopulation in East-Agder has had a gradual increase since the last outbreak of PDV in 2002 and started to stagnate in 2015 (Nilssen et al. 2020). So why do the simulations predict an already extinct population of harbour seals, while field data show the opposite?

An epidemic outbreak like the PDV has not been reported along the Norwegian coast since 2002. Ever since the government introduced annual hunting quotas and obligations to report on affected animals in 1997, there have not been given out hunting quotas in East-Agder (Nilssen et al, 2020). This means that the subpopulation in East-Agder has lived without two significant stressors for more than two decades. The results in this study show that the populations (even the small ones) would increase without stressors. Data also show that many populations of harbour seals in Europe had a quick increase after the epidemic. However, it was 14 years between the two outbreaks of PDV in 1988 and 2002. It is uncertain when or if a similar outbreak may occur. Nonetheless, there remains a potential threat to the Norwegian harbour seal population in the form of a new disease outbreak. Consequently, it is imperative to implement vigilant population management strategies. Therefore, despite the reported increase in the East-Agder subpopulation an local concern about seal disturbance, the possibility of a new disease outbreak must be taken into account (Løvland, 2023; Nilssen et al., 2020). Due to its potential occurrence, a new outbreak has the capacity to inflict severe consequences on these small subpopulations.

4.5 VIABILITY OF THE METHOD

The population viability analysis (PVA) presented in this thesis made it possible to estimate the development of the Norwegian population of harbour seals over a period of 100 years under different scenarios. A PVA model, such as this, is a valuable tool to assess the extinction risk for harbour seals, and can also help guide management decisions . (Morris & Doak, 2002). It is important, however, to keep in mind that models are only a simplification of reality. To be critical when analyzing them is therefore crucial.

4.5.1 Demographic parameter values

PVAs have been criticized for the amount of data required to validate the underlying model(s). If sufficient data on the target species do not exist, a quality PVA cannot be performed (Keedwell, 2004). Sparse data may cause downwardly biased estimates of variability and imprecise population parameters, which can cause incorrect viability measures in population growth rates and probability of quasi-extinction (Morris & Doak, 2002). This is often a problem when studying rare and endangered species, as there may not exist much data on their demographic traits (Keedwell, 2004; Morris & Doak, 2002; Tuberville et al., 2012). When this is the case, an Individual-based model (IBM) may be a better approach to assess future population projections (Tuberville et al., 2012). The harbour seal, however, is a common, well-studied species on a national-, and international scale. The life-history parameters used in this survey were defined based on data collected over a ten-year period (Härkönen et al., 2002; Silva et al., 2021). This supports the use of a PVA model in this survey.

The life-history parameters (survival and birthrates) used in this survey, however, were based on data from the Swedish/Danish population in the Kattegat and Skagerrak area, and not for the Norwegian population due to a lack of data. There exists, however, data on the population of harbour seals in Svalbard (Lydersen & Kovacs, 2005), which is Norwegian territory. Despite Svalbard being a part of Norway, the population of harbour seals in Svalbard is more isolated due to its location as an island in the Barents Sea, ten degrees of latitude further north than the northernmost point of mainland Norway (Wiig, 1989). The strait of Skagerrak, however, stretches from the southeast coast of Norway to the west coast of Sweden (Svansson, 1975). As previously noted, cross-border migration occurs (Goodman, 1998), resulting in a population of harbour seals divided by unnatural borders rather than DNA and life-history traits. In conclusion, there is little evidence suggesting that the parameter values calculated for the Swedish/Danish harbour seal population are not a good representation of the Norwegian population.

However, to enhance the representativeness of the PVA simulations for the Norwegian harbour seal population, parameters (survival and birthrates) should be calculated and used in the model. The Norwegian coastline is long, and harbour seals at different parts of the coast may face different environmental challenges. Vital rates may vary from region to region. Life-history parameters should therefore also be calculated on a regional basis, with regions defined based on genetic differences.

4.5.2 Projecting subpopulations vs the whole population

Projections for the subpopulations and the whole populations in Norway have been presented in this study with the use of a PVA model. All the population growths show similar patterns when exposed to the same hunting and bycatch pressure, regardless of whether simulation was done for the whole population, a big subpopulation, or a small subpopulation. The reason is probably due to the setup of the model. Beyond the different initial population sizes and internal migration patterns, all factors are equal. Consequently, the effect of hunting and bycatch is equal in all the subpopulation regions. Based on this, the structure of the PVA model may be more suited to create population projections for the whole population in Norway rather than for the thirteen subpopulations.

4.5.3 Quasi-extinction threshold

The quasi-extinction threshold (QE) was set at 100 individuals for the subpopulations as this is the minimum number of individuals recommended to account for demographic stochasticity (Hernández-Camacho & Trites, 2018). However, a study carried out on small harbour seal populations in Norway, calculated the Minimum Viable Population (MVP) (the population size that ensured ≥95% survival over a 100 years period) to be 120 individuals when exposed to hunting and no migration (Bjørge, 1993). This means that the MVP is almost the same as the set QE in this study. This study also included migration between subpopulations, which means that the MVP would probably have been below the QE. On the other hand, Bjørge did not include bycatch and catastrophic events, such as epizootic outbreaks, which would have exceeded the MVP (Bjørge, 1993).

4.6 FUTURE WORK AND IMPROVEMENTS

4.6.1 Age-distributed bycatch rates

The age-distribution of bycatch rates was derived from one single survey. To get the most precise parameter values, several similar surveys should be conducted. There has, however, not been much flipper tagging over the last years. This is unfortunate, as the mark-recapture data best reflect the true bycatch levels (Bjørge et al., 2017). This method may also be the easiest way to determine the age distribution of bycaught animals, if the tags include the date they were applied, and every tag is placed on newborns. It would then be easy to calculate the age of the bycaught seal. Data from the reference fleet have shown to be a less efficient method for determining bycatch rates as there is more sources of error regarding species and age identification (Bjørge et al., 2017). Additionally, more data is essential to get a better estimation of the total bycatch rate of harbour seals in Norway. This improved data would enable the integration of a precis bycatch rate into the PVA model, facilitating a rigorous assessment of harbour seal viability.

4.6.2 Subpopulations based on DNA and not county borders

This study divided subpopulations based on current management units, which are the old division of counties in Norway. Although county borders have shown to be a practical classification, they may not be biologically representative or relevant (Bjørge & Nilssen, 2020a; NAMMCO-North Atlantic Marine Mammal Commission, 2021). There is an ongoing investigation to determine if DNA analysis can reveal population structures that can be used as new borders for management units (NAMMCO-North Atlantic Marine Mammal Commission, 2021). This has been done for grey seals along the Norwegian coast, which are divided into three management units based on genetic differentiation (Bjørge & Nilssen, 2020b). Implementing similar measures for the harbour seal could reduce the risk of eradicating genetically distinct populations (Bjørge & Nilssen, 2020a). When, or if, this work is completed for the harbour seal, the PVA model should be modified to divide subpopulations based on the DNA analysis.

4.6.3 Include migration from other populations

As previously discussed, the harbour seal population in Norway is experiencing migration from regions in Sweden and Denmark (Goodman, 1998). For further refinement of this model, it is imperative to incorporate this migratory factor into the PVA model. This step serves to evaluate both the potential impact of migration on the viability of the Norwegian harbour seal population and the reduction of the existing discrepancy. If this is to be carried out, it is imperative to gather requisite data by monitoring the migration patterns between the countries.

5 CONCLUSION

The use of a demographic PVA model made it possible to explore the harbour seal population trajectories over 100 years under different hunting and bycatch rates. The results provided by the PVA indicate that hunting and bycatch, together with environmental stressors such as the probability of disease outbreaks and fecundity reduction due to pollutants, have a negative impact on population growth. According to the results, the annual reported harvest, and the annual harbour seal bycatch rate in Norway are not sustainable and fail to meet the management aims to ensure a viable population along the Norwegian coastline. The results present evidence that a bycatch rate distributed equally between the age classes leads to a greater risk of extinction, primarily due to a higher loss of fertile females, compared to an age-distributed bycatch rate towards younger individuals. An age-distributed bycatch rate has shown to be more consistent with previous surveys and field observations. However, a great discrepancy exists between the population estimates provided in this study and those based on field observations conducted by the IMR. To conclude, this study has proven that a demographic PVA model can be used to assess the extinction risk of the harbour seal population in Norway under different environmental stressors. However, adjustments to the model, and more data on the study species, are needed to enhance the representativeness of the PVA simulations and to reduce the discrepancy. This does not undermine the importance of PVAs in the field of conservation biology. When used right, PVAs can assist scientists and governments in the management of wildlife populations, ensuring their sustainability in the future.

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APPENDIX

Complex parameter values

Table A1: Migration. Proportion of a population which is present in a population at the end of each cycle.

	Finnmark	Troms	Nordland	N-Trøndelag	S-Trøndelag	Møre & Romsdal	Sogn & Fjordane	Rogaland	West-Agder	East-Agder	Telemark	Vestfold	Østfold
Finnmark	1	0	0	0	0	0	0	0	0	0	0	0	0
Troms	0	1	0	0	0	0	0	0	0	0	0	0	0
Nordland	0	0	1	0	0	0	0	0	0	0	0	0	0
N-Trøndelag	0	0	0	0.92	0.08	0	0	0	0	0	0	0	0
S-Trøndelag	0	0	0	0.08	0.87	0.05	0	0	0	0	0	0	0
Møre & Romsdal	0	0	0	0	0.05	0.89	0.06	0	0	0	0	0	0
Sogn & Fjordane	0	0	0	0	0	0.06	0.94	0	0	0	0	0	0
Rogaland	0	0	0	0	0	0	0	0.94	0.06	0	0	0	0
Vest-Agder	0	0	0	0	0	0	0	0.06	0.74	0.09	0.06	0.05	0
Aust-Agder	0	0	0	0	0	0	0	0	0.0.9	0.60	0.15	0.10	0.06
Telemark	0	0	0	0	0	0	0	0	0.06	0.15	0.39	0.30	0.10
Vestfold	0	0	0	0	0	0	0	0	0.05	0.10	0.30	0.40	0.15
Østfold	0	0	0	0	0	0	0	0	0	0.06	0.10	0.15	0.69

Table A2: The weighting of disease per age class (W_D).

Age class	W_D	Age class	W_D
1	1.0000	**20**	0.0110
2	0.1155	**21**	0.1100
3	0.0995	**22**	0.1100
4	0.0915	**23**	0.1100
5	0.0900	**24**	0.1100
6	0.1025	**25**	0.1100
7	0.0900	**26**	0.1100
8	0.0850	**27**	0.1100
9	0.0725	**28**	0.1100
10	0.0700	**29**	0.1100
11	0.0560	**30**	0.1100
12	0.0260	**31**	0.1100
13	0.0190	**32**	0.1100
14	0.0020	**33**	0.1100
15	0.0300	**34**	0.1100
16	0.0170	**35**	0.1100
17	0.0010	**36**	0.1100
18	0.0000	**37**	0.1100
19	0.0020	**38**	0.1100

Table A3: The values of parameter P (the initial population structure).

Age-class	**Finnmark**	**Troms**	**Nordland**	**N-Trøndelag**	**S-Trøndelag**	**Møre & Romsdal**	**Sogn & Fjordane**	**Rogaland**	**West-Agder**	**East-Agder**	**Telemark**	**Vestfold**	**Østfold**
1	147	100	206	16	104	83	85	54	7	6	31	87	91
2	106	72	148	12	75	60	61	39	5	4	22	63	66
3	90	61	126	10	64	51	52	33	4	4	19	53	56
4	77	52	108	9	54	44	44	28	4	3	16	46	48
5	66	44	92	7	46	37	38	24	3	3	14	39	41
6	60	40	84	7	42	34	34	22	3	2	12	35	37
7	54	37	76	6	38	31	31	20	3	2	11	32	34
8	49	34	69	5	35	28	28	18	2	2	10	29	31
9	45	31	63	5	32	25	26	17	2	2	9	27	28
10	41	28	57	5	29	23	24	15	2	2	9	24	25
11	37	25	52	4	26	21	21	14	2	2	8	22	23
12	34	23	47	4	24	19	19	12	2	1	7	20	21
13	31	21	43	3	22	17	18	11	1	1	6	18	19
14	28	19	39	3	20	16	16	10	1	1	6	17	17
15	26	17	36	3	18	14	15	9	1	1	5	15	16
16	23	16	33	3	16	13	13	9	1	1	5	14	14
17	21	14	30	2	15	12	12	8	1	1	4	13	13
18	19	13	27	2	14	11	11	7	1	1	4	11	12
19	18	12	25	2	12	10	10	6	1	1	4	10	11
20	16	11	22	2	11	9	9	6	1	1	3	9	10
21	15	10	20	2	10	8	8	5	1	1	3	9	9
22	13	9	19	1	9	7	8	5	1	1	3	8	8
23	12	8	17	1	8	7	7	4	1	0	3	7	7
24	11	7	15	1	8	6	6	4	1	0	2	6	7
25	10	7	14	1	7	6	6	4	0	0	2	6	6
26	9	6	13	1	6	5	5	3	0	0	2	5	6

27	8	6	12	1	6	5	5	3	0	0	2	5	5
28	8	5	11	1	5	4	4	3	0	0	2	4	5
29	7	5	10	1	5	4	4	3	0	0	1	4	4
30	6	4	9	1	4	4	4	2	0	0	1	4	4
31	6	4	8	1	4	3	3	2	0	0	1	3	4
32	5	3	7	1	4	3	3	2	0	0	1	3	3
33	5	3	7	1	3	3	3	2	0	0	1	3	3
34	4	3	6	0	3	2	2	2	0	0	1	3	3
35	4	3	5	0	3	2	2	1	0	0	1	2	2
36	4	2	5	0	2	2	2	1	0	0	1	2	2
37	3	2	5	0	2	2	2	1	0	0	1	2	2
38	3	2	4	0	2	2	2	1	0	0	1	2	2

Scenarios under or over 7 000 individuals

Table A4: Summary of the scenarios that are either over or below the 7000 individual limit after 100 years. Not including the baseline scenario

Setting A		**Setting B**	
Final pop ≥ 7 000	**Final pop < 7 000**	**Final pop ≥ 7 000**	**Final pop < 7 000**
1-4	5-11	1-4	5-11
12-13	14-22	12-14	15-22
	23-33	23	24-33
	34-44		34-44
	45-55		45-55

Harbour seal age distribution

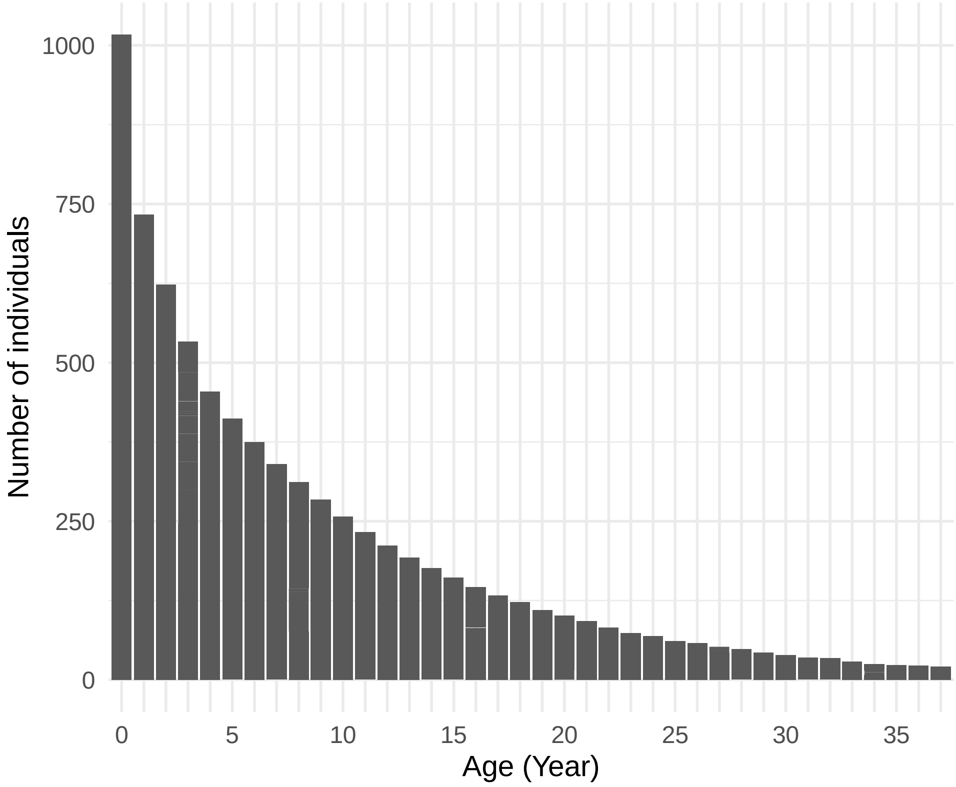

Figure A1: Age distribution of the harbour seal population in Norway calculated in the population model and used in the PVA model.

Subpopulation Growth Curves Scenario 1-11

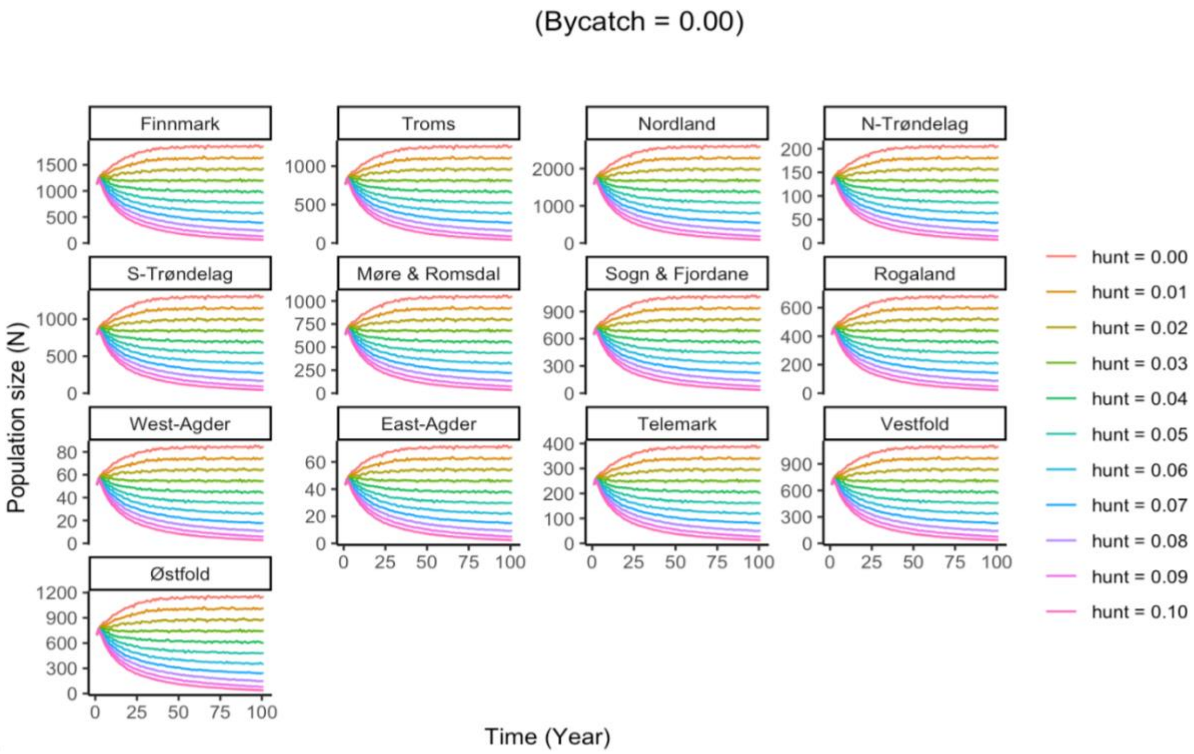

Figure A2: Population development for harbour seals in 13 counties over 100 years with hunting pressure from 0-10% when bycatch equals 0%.

Subpopulation quasi-extinction probability Scenario 1-11

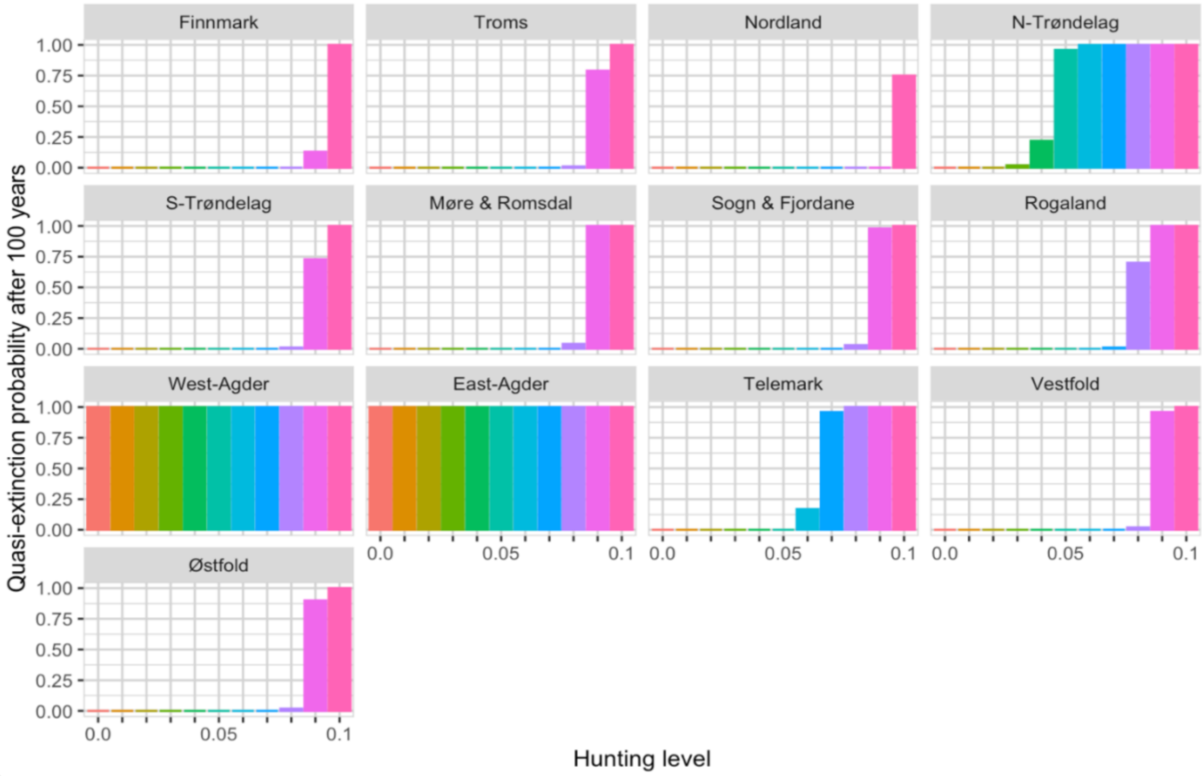

Figure A3: Bar plot showing the probability of quasi-extinction for scenarios 1-11 in each county after 100 years based on the mean population size when bycatch equals 0%.

Subpopulation Growth Curves Setting A

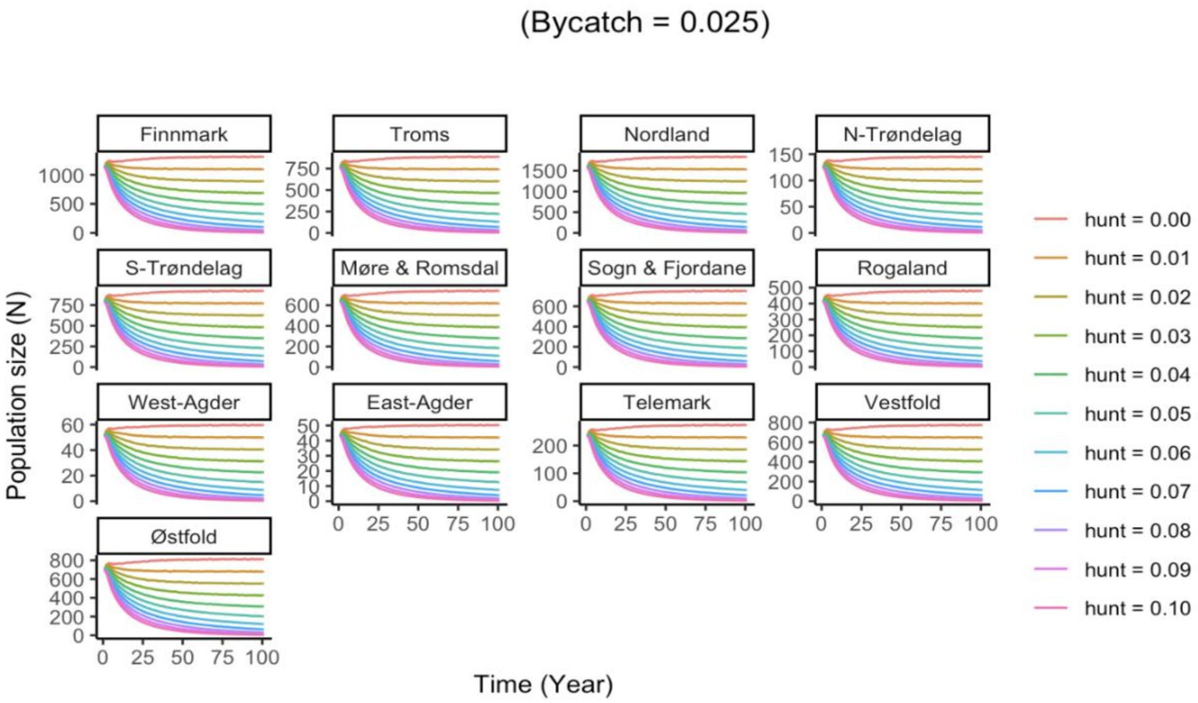

Figure A4: Population development for harbour seals in 13 counties over 100 years with hunting pressure from 0-10% when bycatch equals 2.5%.

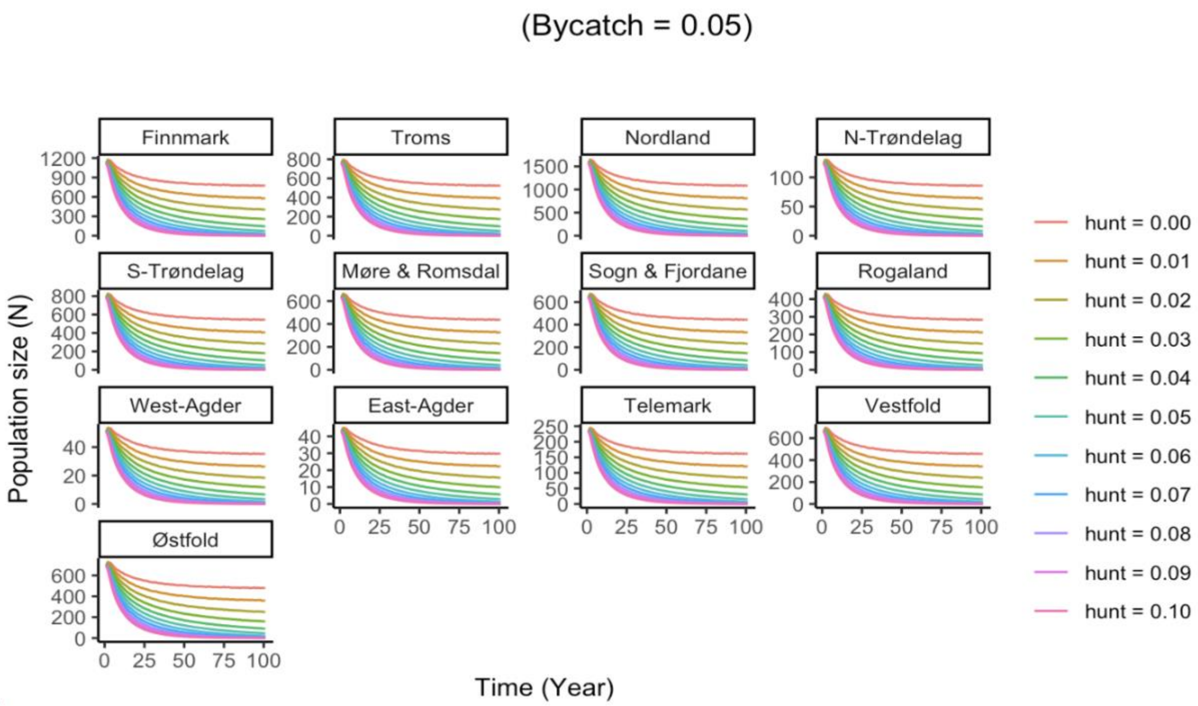

Figure A5: Population development for harbour seals in 13 counties over 100 years with hunting pressure from 0-10% when bycatch equals 5%.

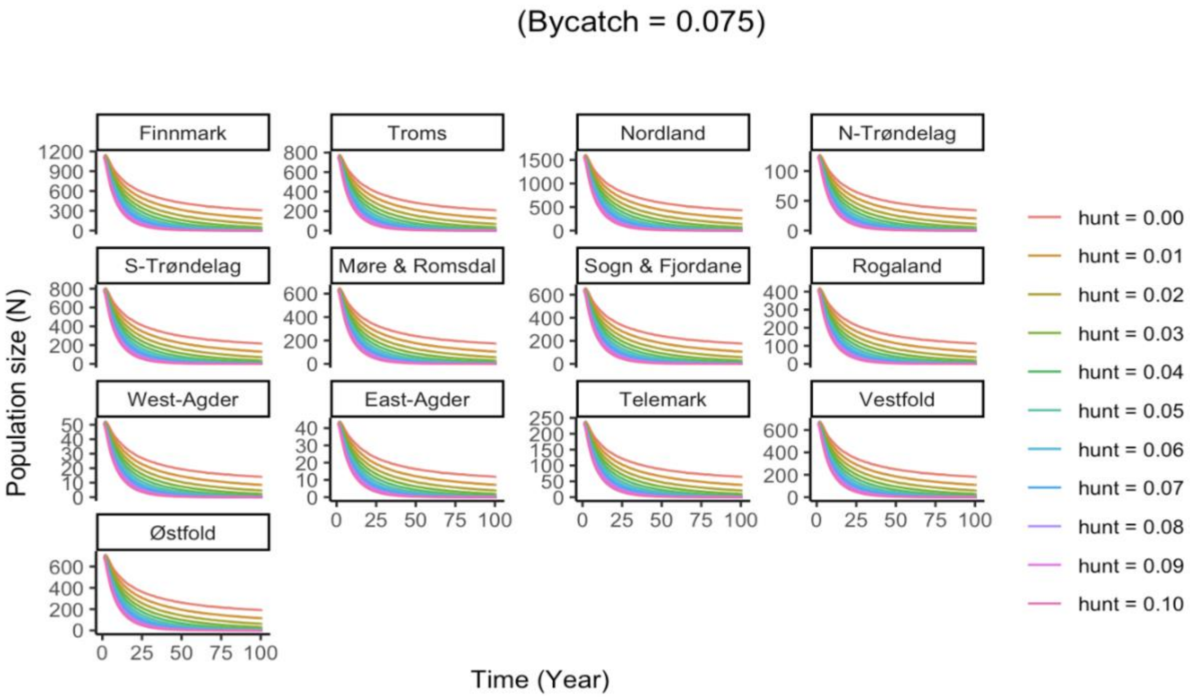

Figure A6: Population development for harbour seals in 13 counties over 100 years with hunting pressure from 0-10% when bycatch equals 7.5%.

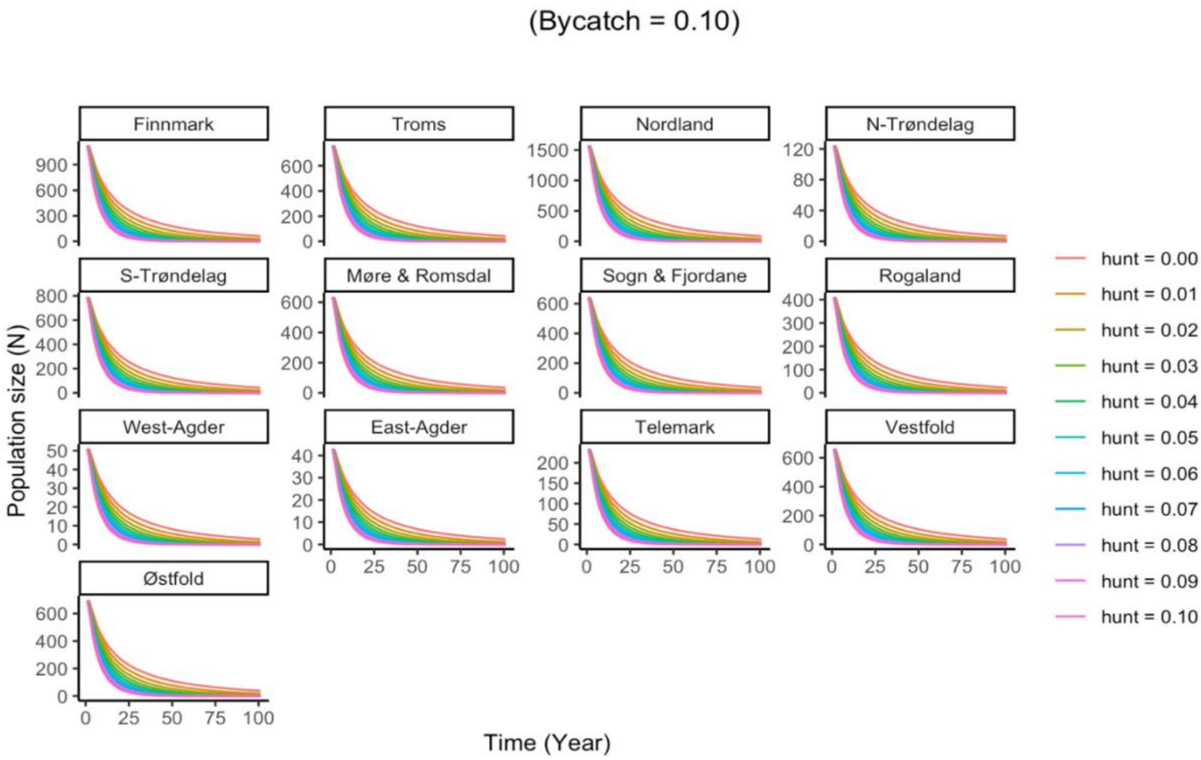

Figure A7: Population development for harbour seals in 13 counties over 100 years with hunting pressure from 0-10% when bycatch equals 10%.

Subpopulation quasi-extinction plot Setting A

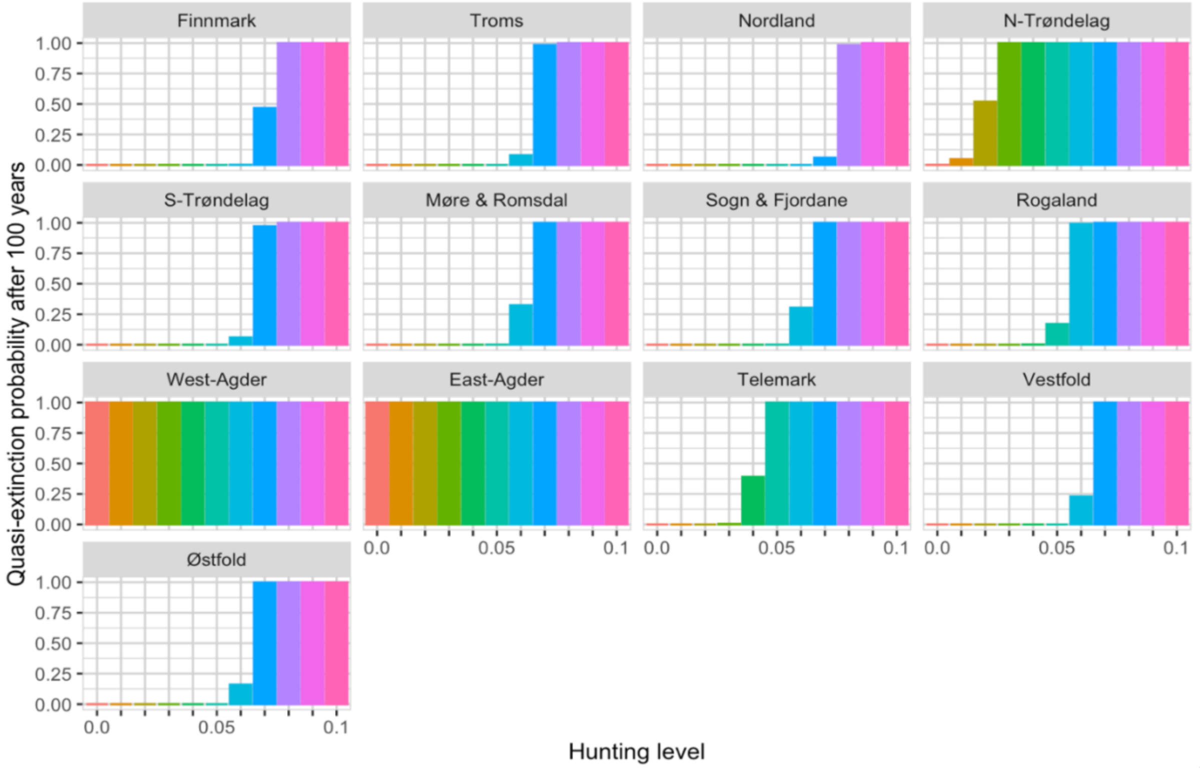

Figure A8: Bar plot showing the probability of quasi-extinction for scenarios 12-22 in each county after 100 years based on the mean population size when bycatch equals 2.5%.

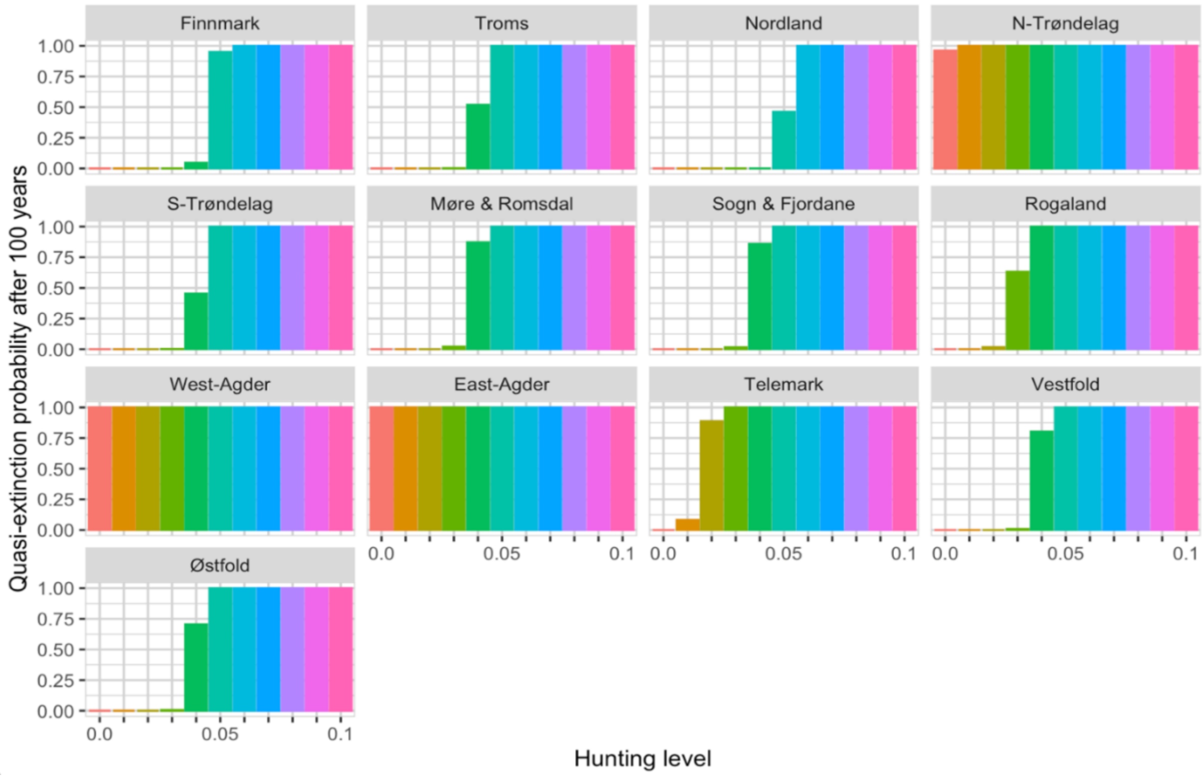

Figure A9: Bar plot showing the probability of quasi-extinction for scenarios 23-33 in each county after 100 years based on the mean population size when bycatch equals 5%.

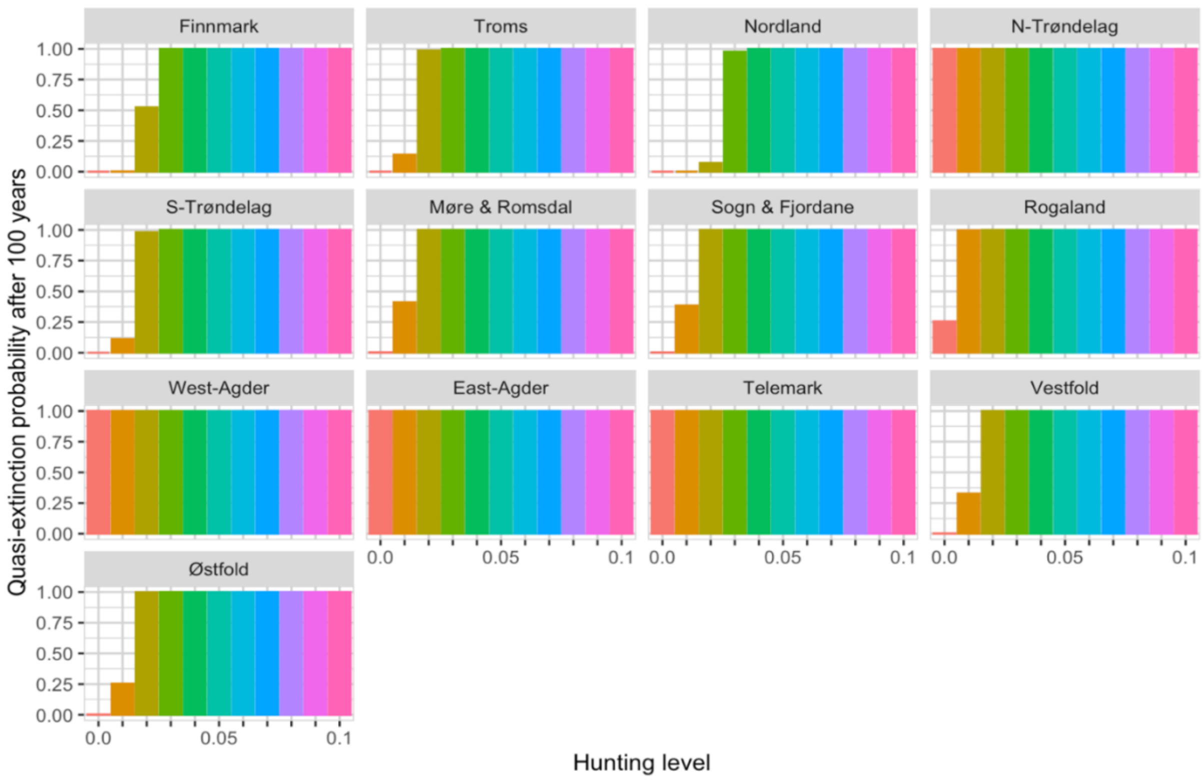

Figure A10: Bar plot showing the probability of quasi-extinction for scenarios 34-44 in each county after 100 years based on the mean population size when bycatch equals 7.5%.

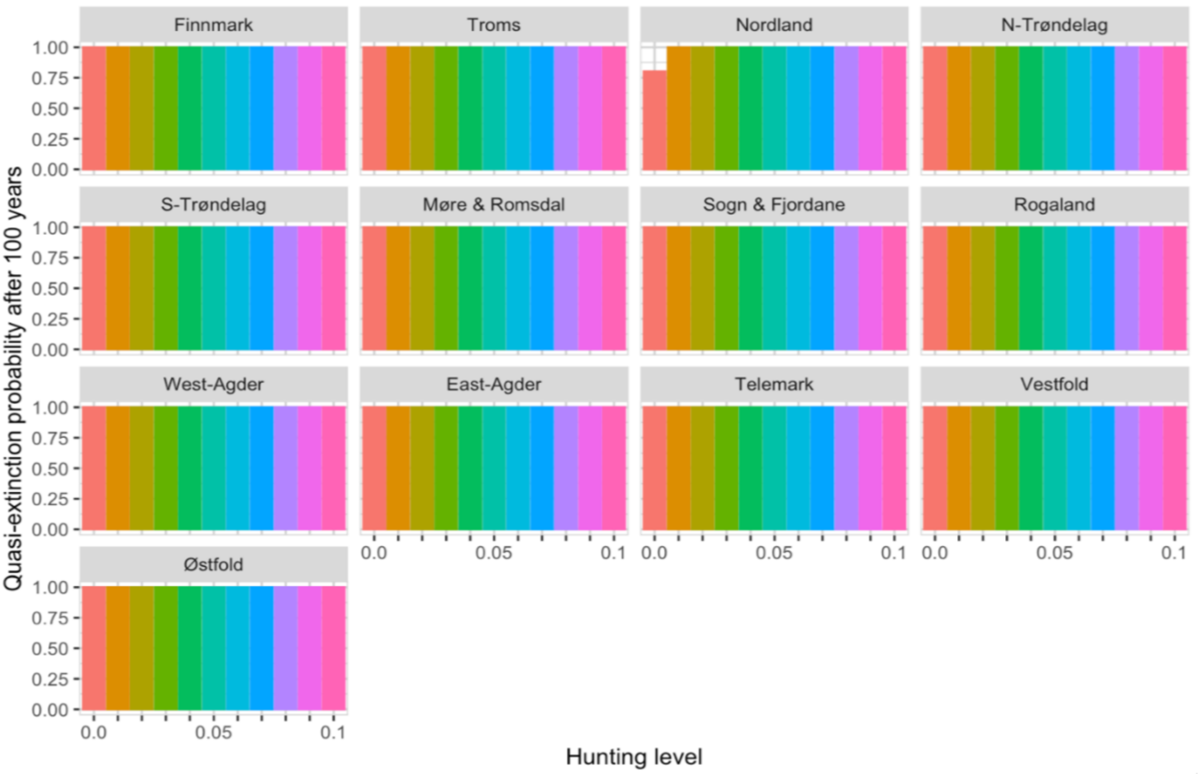

Figure A11: Bar plot showing the probability of quasi-extinction for scenarios 45-55 in each county after 100 years based on the mean population size when bycatch equals 10%.

Subpopulation Growth curves Setting B

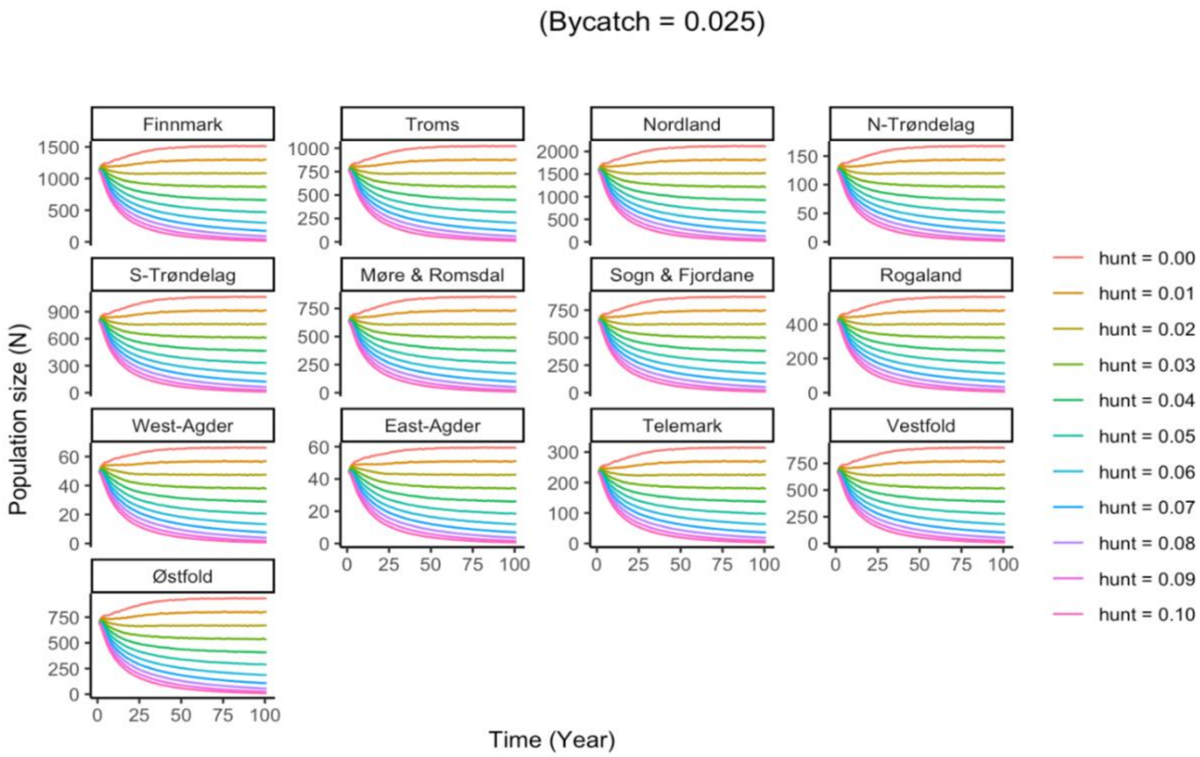

Figure A12: Population development for harbour seals in 13 counties over 100 years with hunting pressure from 0-10% when bycatch equals 2.5%.

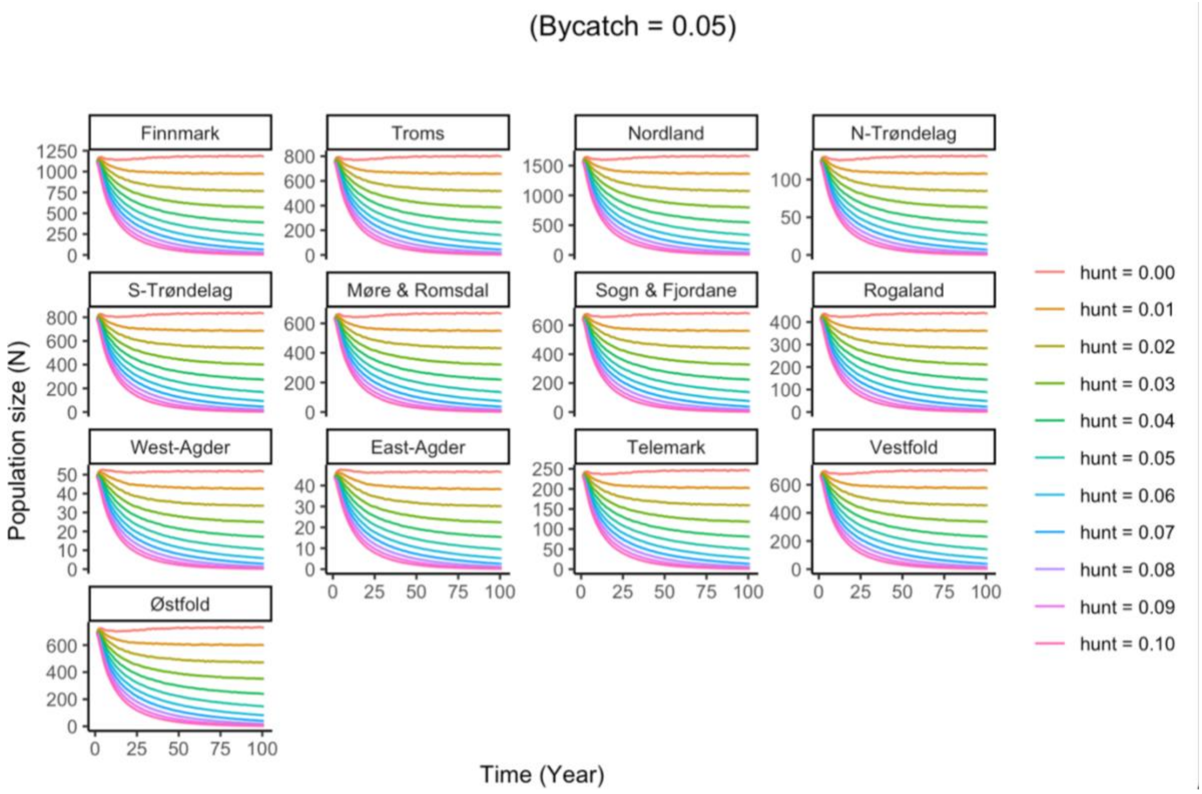

Figure A13: Population development for harbour seals in 13 counties over 100 years with hunting pressure from 0-10% when bycatch equals 5%.

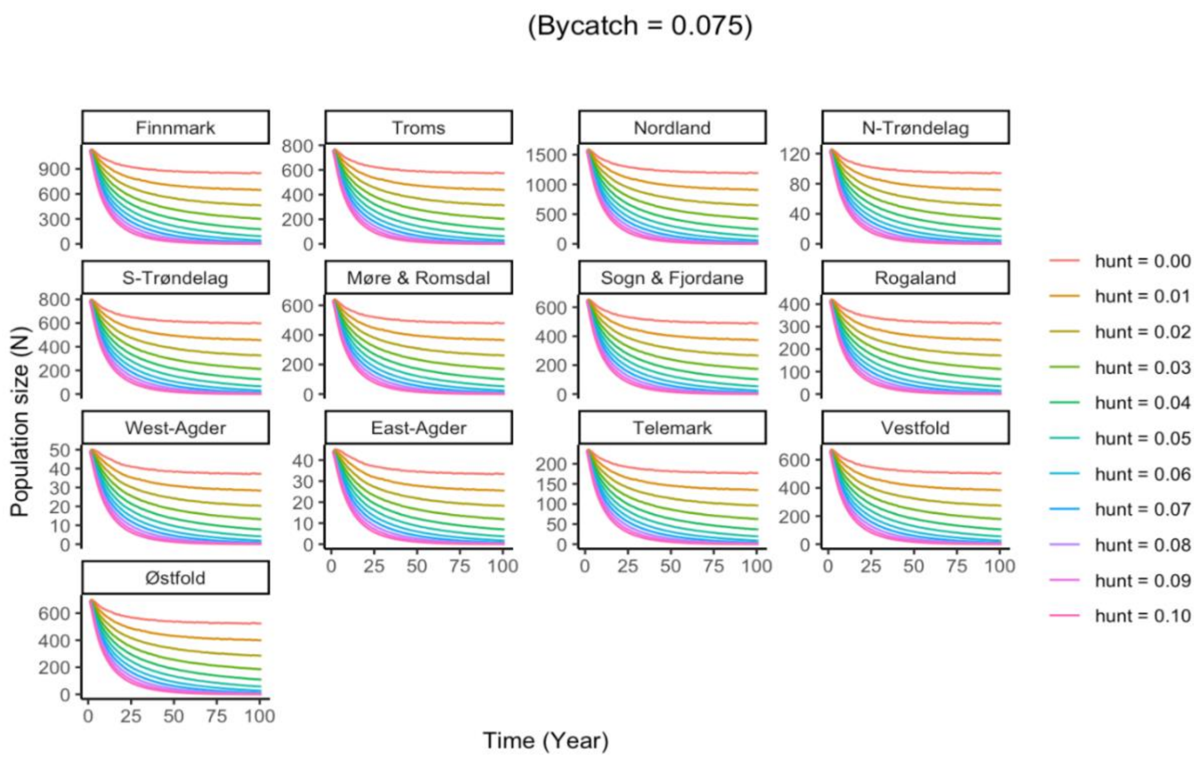

Figure A14: Population development for harbour seals in 13 counties over 100 years with hunting pressure from 0-10% when bycatch equals 7.5%.

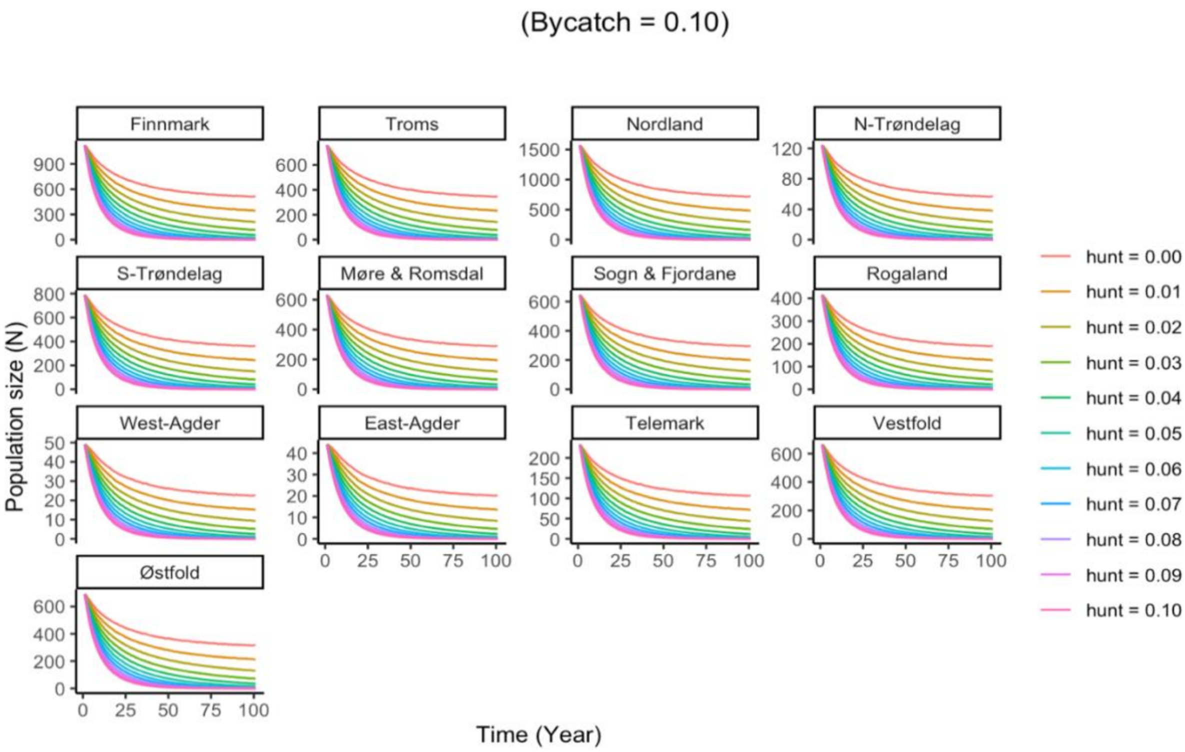

Figure A15: Population development for harbour seals in 13 counties over 100 years with hunting pressure from 0-10% when bycatch equals 10%.

Subpopulation quasi-extinction probability plots Setting B

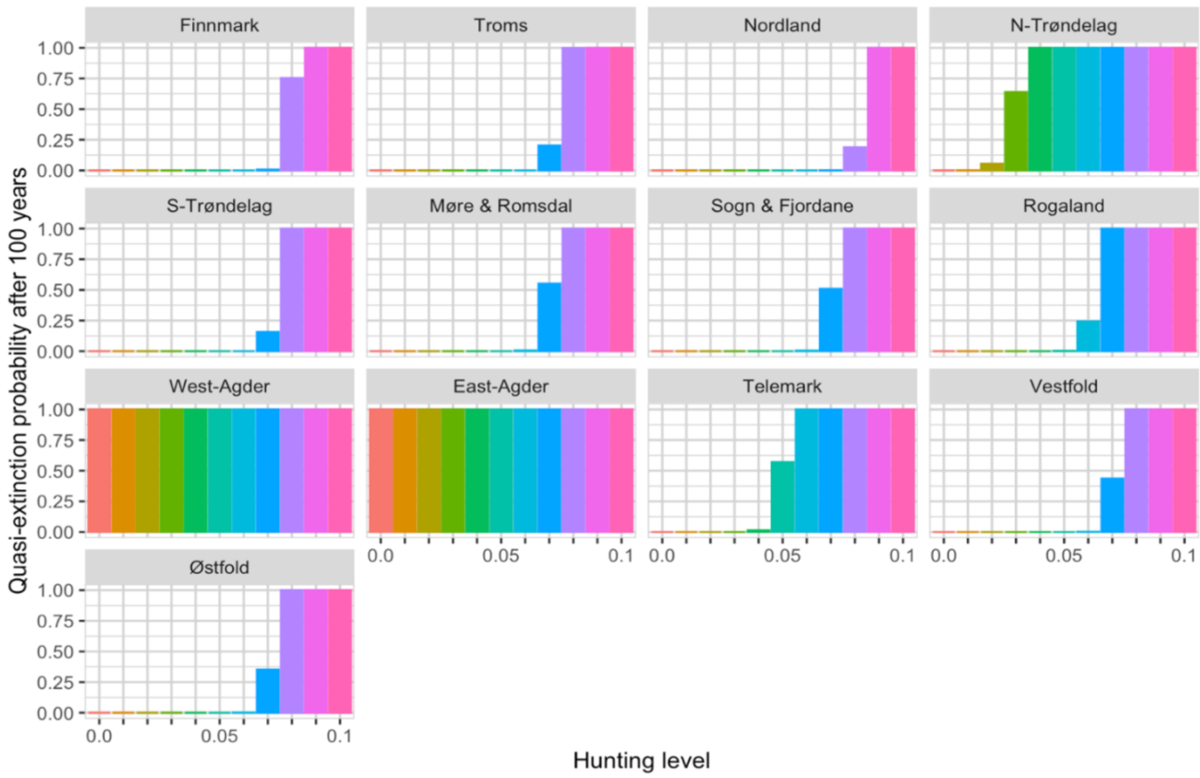

Figure A16: Bar plot showing the probability of quasi-extinction for scenarios 23-33 in each county after 100 years based on the mean population size when bycatch equals 2.5%.

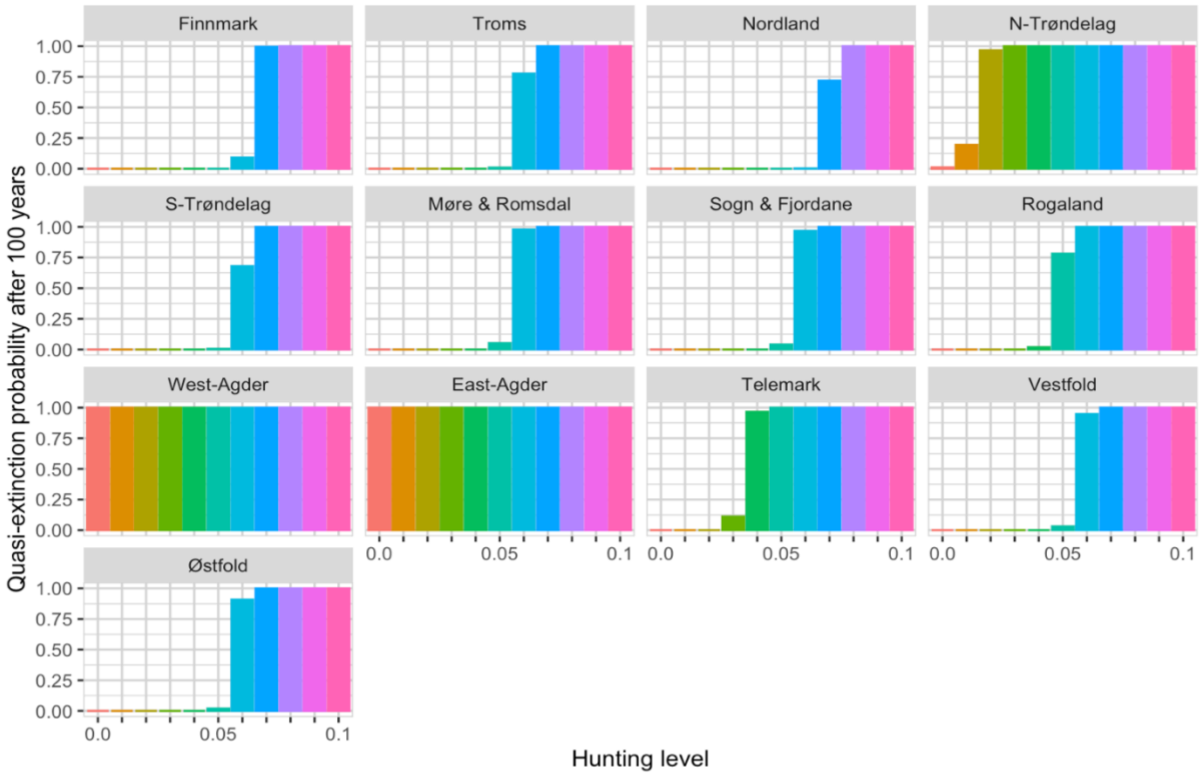

Figure A17: Bar plot showing the probability of quasi-extinction for scenarios 23-33 in each county after 100 years based on the mean population size when bycatch equals 5%.

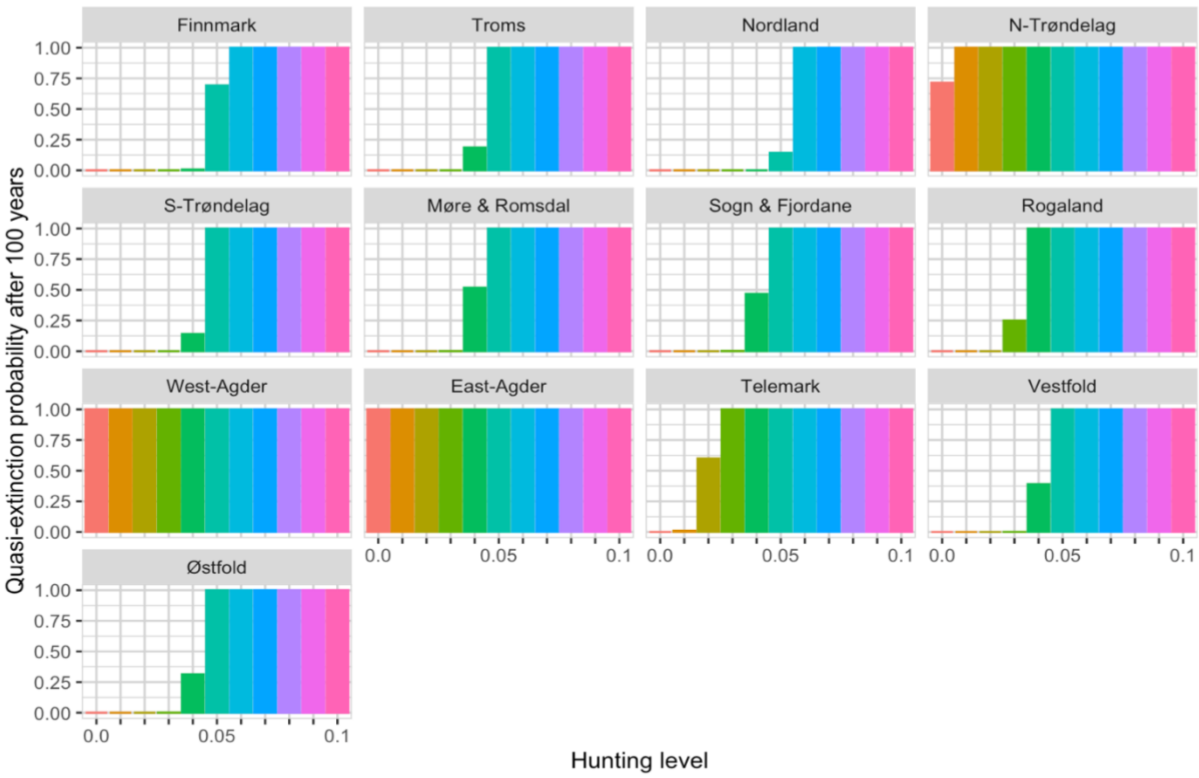

Figure A18: Bar plot showing the probability of quasi-extinction for scenarios 34-44 in each county after 100 years based on the mean population size when bycatch equals 7.5%.

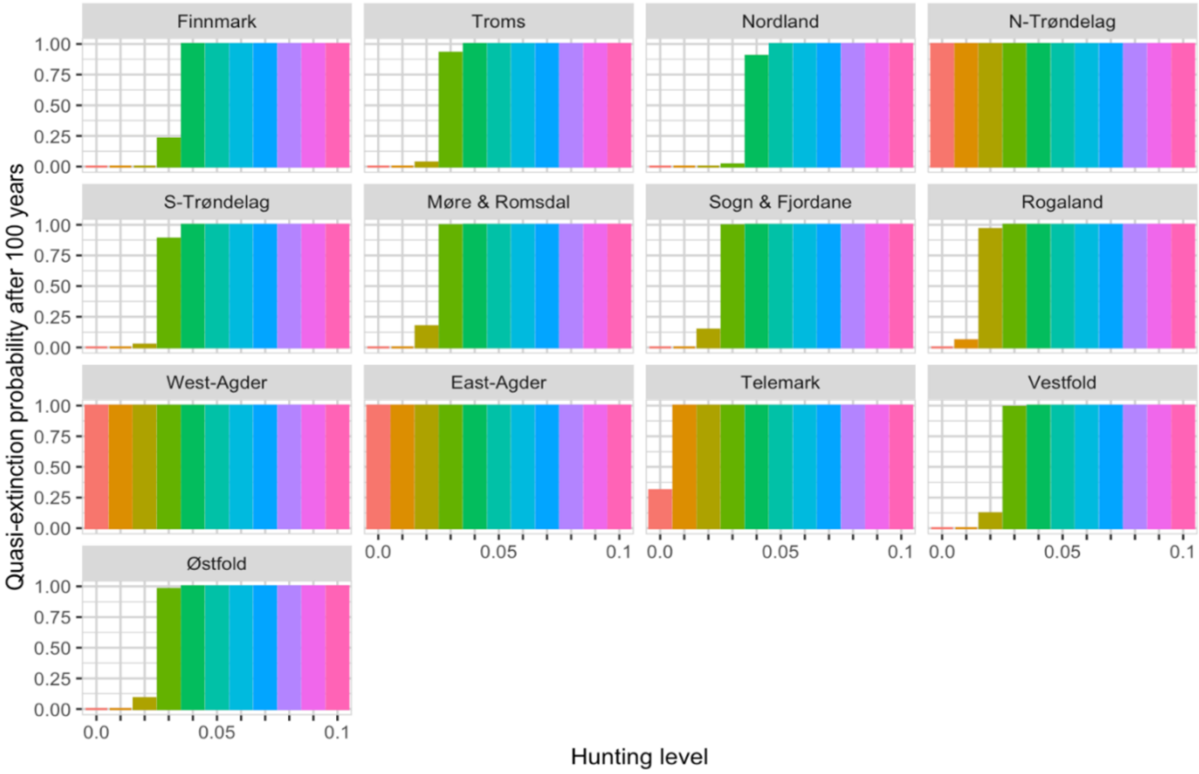

Figure A19: Bar plot showing the probability of quasi-extinction for scenarios 45-55 in each county after 100 years based on the mean population size when bycatch equals 10%.