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Long-term effects of no-take zones in Swedish waters

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Abstract

Marine protected areas (MPAs) are increasingly established worldwide to protect and restore degraded ecosystems. However, the level of protection varies among MPAs and has been found to affect the outcome of the closure. In no-take zones (NTZs), no fishing or extraction of marine organisms is allowed. The EU Commission recently committed to protect 30% of European waters by 2030 through the updated Biodiversity Strategy. Importantly, one third of these 30% should be of strict protection. Exactly what is meant by strict protection is not entirely clear, but fishing would likely have to be fully or largely prohibited in these areas. This new target for strictly protected areas highlights the need to evaluate the ecological effects of NTZs, particularly in regions like northern Europe where such evaluations are scarce. The Swedish NTZs made up approximately two thirds of the total areal extent of NTZs in Europe a decade ago. Given that these areas have been closed for at least 10 years and can provide insights into long-term effects of NTZs on fish and ecosystems, they are of broad interest in light of the new 10% strict protection by 2030 commitment by EU member states.

In total, eight NTZs in Swedish coastal and offshore waters were evaluated in the current report, with respect to primarily the responses of focal species for the conservation measure, but in some of the areas also ecosystem responses. Five of the NTZs were established in 2009-2011, as part of a government commission, while the other three had been established earlier. The results of the evaluations are presented in a synthesis and also in separate, more detailed chapters for each of the eight NTZs. Overall, the results suggest that NTZs can increase abundances and biomasses of fish and decapod crustaceans, given that the closed areas are strategically placed and of an appropriate size in relation to the life cycle of the focal species. A meta-regression of the effects on focal species of the NTZs showed that CPUE was on average 2.6 times higher after three years of protection, and 3.8 times higher than in the fished reference areas after six years of protection. The proportion of old and large individuals increased in most NTZs, and thereby also the reproductive potential of populations. The increase in abundance of large predatory fish also likely contributed to restoring ecosystem functions, such as top-down control. These effects appeared after a 5-year period and in many cases remained and continued to increase in the longer term (>10 years). In the two areas where cod was the focal species of the NTZs, positive responses were weak, likely as an effect of long-term past, and in the Kattegat still present, recruitment overfishing. In the Baltic Sea, predation by grey seal and cormorant was in some cases so high that it likely counteracted the positive effects of removing fisheries and led to stock declines in the NTZs. In most cases, the introduction of the NTZs has likely decreased the total fishing effort rather than displacing it to adjacent areas. In the Kattegat NTZ, however, the purpose was explicitly to displace an unselective coastal mixed bottom-trawl fishery targeting Norway lobster and flatfish to areas where the bycatches of mature cod were smaller. In two areas that were reopened to fishing after 5 years, the positive effects of the NTZs on fish stocks eroded quickly to pre-closure levels despite that the areas remained closed during the spawning period, highlighting that permanent closures may be necessary to maintain positive effects.

We conclude from the Swedish case studies that NTZs may well function as a complement to other fisheries management measures, such as catch, effort and gear regulations. The experiences from the current evaluation show that NTZs can be an important tool for fisheries management especially for local coastal fish populations and areas with mixed fisheries, as well as in cases where there is a need to counteract adverse ecosystem effects of fishing. NTZs are also needed as reference for marine environmental management, and for understanding the effects of fishing on fish populations and other ecosystem components in relation to other pressures. MPAs where the protection of both fish and their habitats is combined may be an important instrument for ecosystem-based management, where the recovery of large predatory fish may lead to a restoration of important ecosystem functions and contribute to improving decayed habitats.

With the new Biodiversity Strategy, EUs level of ambition for marine conservation increases significantly, with the goal of 30% of coastal and marine waters protected by 2030, and, importantly, one third of these areas being strictly protected. From a conservation perspective, rare, sensitive and/or charismatic species or habitats are often in focus when designating MPAs, and displacement of fisheries is then considered an unwanted side effect. However, if the establishment of strictly protected areas also aims to rebuild fish stocks, these MPAs should be placed in heavily fished areas and designed to protect depleted populations by accounting for their home ranges to generate positive outcomes. Thus, extensive displacement of fisheries is required to reach benefits for depleted populations, and need to be accounted for e.g. by specific regulations outside the strictly protected areas. These new extensive EU goals for MPA establishment pose a challenge for management, but at the same time offer an opportunity to bridge the current gap between conservation and fisheries management.

Svensk sammanfattning

Allt fler marina skyddade områden inrättas globalt som en åtgärd för att skydda och restaurera ekosystem som är negativt påverkade av mänskliga aktiviteter. Hur starkt skyddet är varierar mellan områden och påverkar även effekterna på arter och livsmiljöer. I fiskefria områden, så kallade no-take zones (NTZs) på engelska, är allt fiske förbjudet. Nyligen beslutades nya mål för EUs biodiversitetsstrategi, där 30 % av havsytan ska skyddas till 2030, varav en tredjedel ska utgöras av strikt skyddade områden. Strikt skydd innebär sannolikt att fiske är helt förbjudet eller kraftigt begränsat. Detta nya mål gällande strikt skydd aktualiserar behovet av att utvärdera de ekologiska effekterna av fiskefria områden, särskilt i norra Europa där dessa områden är få. För 10 år sedan utgjorde de svenska fiskefria områdena ungefär en tredjedel av Europas totala yta av fiskefria områden. Intresset för de långvariga effekterna av dessa områden på fisk och ekosystem är därför stort för resten av Europa, särskilt med tanke på de nya EU-målen gällande strikt skydd till 2030.

I Sverige har det införts totalt åtta fiskefria områden i kust- och utsjöområden. Fem av områdena infördes 2009-2011 som en del av ett regeringsuppdrag, medan övriga tre inrättats tidigare. Effekterna av de fiskefria områdena utvärderas gemensamt i denna rapport, i första hand avseende fokusarter för fredningen men i vissa fall även på ekosystemeffekter. Två områden har öppnats upp igen för fiske efter att bestånden i dessa fiskefria områden hade återhämtat sig under 5 år. Resultaten i rapporten presenteras i en sammanfattande syntes och mer i detalj i enskilda kapitel för de enskilda områdena. Sammanfattningsvis visar resultaten att fiskefria områden kan öka antalet och storleken på de arter som är målarter för fisket, framförallt när man tagit hänsyn till storlek och placering av det skyddade området i relation till arternas ekologi. En meta-analys av effekterna visar att mängden fisk var i medeltal 2.6 gånger högre efter 3 års skydd och 3.8 gånger högre efter 6 års skydd än i referensområdena som var öppna för fiske. Storlek- och åldersstrukturen hos fokusarterna förbättrades också inom de fredade områdena, med en ökning av större och äldre fisk och kräftdjur och därmed också en ökning av reproduktionspotentialen hos populationerna. Ökningen av stora rovfiskar kan även bidra till att återupprätta ekosystemfunktioner genom s.k. top-down kontroll av näringsväven. De positiva effekterna blev synliga redan inom 5 år och i många fall fortsatte utvecklingen under längre tid (>10 år). För de populationer som varit rekryteringsöverfiskade under lång tid, vilket gällde torsk i Kattegatt och Havstensfjorden, var effekterna svaga eller obefintliga. I Östersjön har predation av säl och skarv i några fall varit så hög att de initiala positiva effekterna av skydd har dämpats eller helt uteblivit med tiden. För de två fiskefria områdena som öppnades upp för fiske igen efter 5 år av skydd, minskade mängden fisk snabbt till nivåer motsvarande de innan skyddet infördes, trots att områdena fortsatte vara fredade under lekperioden. Detta belyser att permanenta fiskeförbud kan behövas för ett mer långsiktigt skydd av vissa arter.

Sammantaget visar dessa fallstudier att fiskefria områden kan bidra till en återhämtning av populationer av fisk och kräftdjur och kan fungera som ett komplement till andra fiskeåtgärder, som fångst- och redskapsbegränsningar. Fiskefria områden kan vara ett viktigt förvaltningsverktyg framför allt för lokala kustfiskbestånd och för områden med blandfisken, liksom för att motverka negativa ekosystemeffekter av fiske. Fiskefria områden behövs också som referensområden för miljöövervakningen och för att kunna studera fiskets påverkan på arter och ekosystem. Marina skyddade områden där fiskeförbud kombineras med habitatskydd kan utgöra ett viktigt verktyg för ekosystembaserad förvaltning, där en återhämtning av rovfiskbestånden kan leda till att ekosystemfunktioner i livsmiljöer återställs.

EUs ambitionsnivå för skydd av haven har ökat betydligt i och med den nya biodiversitetsstrategin, där ett mål är att 30 % av kust- och havsområdena ska vara skyddade till 2030, och att en tredjedel av dessa områden ska ha ett strikt skydd. När skyddade områden införs står ofta sällsynta, känsliga eller karismatiska arter i fokus, och man strävar vanligen efter att inte göra mer inskränkningar i fisket än nödvändigt och att minimera förflyttning (displacement) av fiskeansträngning. I och med målet om strikt skyddade områden, kan stärkta fiskbestånd bli en viktig del av det marina områdesskyddet i Europa. Det innebär i sin tur att man behöver fokusera på att skydda områden med ett starkt fisketryck och att de skyddade områdena utformas så att de skyddar de livsstadier som utsätts för fiske på ett effektivt sätt, vilket kan innebära förflyttning av fiskeansträngning är nödvändigt och måste beaktas, och att fiskeregleringar behövs även i kringliggande områden. För att de strikt skyddade områdena ska ge önskade effekter på svaga fiskbestånd krävs alltså att områdena placeras och utformas så att fisketrycket på bestånden som helhet minskar väsentligt. Dessa nya omfattande mål för områdesskydd inom EU är en utmaning för förvaltningen, men utgör samtidigt en möjlighet att samordna och gynna såväl områdesskyddet som fiskförvaltningen.

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1. Introduction

Charlotte Berkström, Ulf Bergström, Mattias Sköld, Edmond Sacre

1.1. No-take zones (NTZs)

Marine protected areas (MPAs) are increasingly established worldwide to protect and restore degraded ecosystems. However, the level of the protection varies among MPAs where some are strictly protected with no extraction or other activities causing local disturbance allowed, while others allow some level of disturban333ce, including fisheries (Grorud-Colvert et al., 2021). The level of protection has, however, been found to affect the outcome of the closure (Lester and Halpern, 2008; Motta et al., 2021; Ferreira et al., 2022). In no-take zones (NTZs), no fishing or extraction of marine organisms is allowed. In the literature, NTZs are often referred to as no-take marine reserves or just marine reserves by many authors (Wells et al., 2016). Originally, NTZs were used as a simple way to manage fish associated with coral reefs in tropical seascapes and enforce regulations, often in countries with little resources for fisheries management (Alcala and Russ, 1990; Russ and Alcala, 2003). Gradually, the use of NTZs increased also in temperate regions, but they are still rare in northern temperate waters (Costello and Ballantine, 2015; Wells et al., 2016). Only about 2,8 % of the ocean is fully or highly protected from fishing, and most of them are found in remote areas with little human impact (mpaatlas.org). Areas with partial or seasonal closures are common, however, with restrictions still allowing many species to be fished, e.g. by allowing low impact fishing gear only or closed during spawning.

Global targets to protect 10% of the ocean by 2020 was previously set by the Aichi Biodiversity Targets in the strategic plan of the Convention on Biological Diversity and adopted by EU member states. Recently the EU Commission committed to protect 30% of European waters by 2030 through the updated Biodiversity Strategy (O'Leary et al., 2016; EuropeanCommission, 2020; Jones et al., 2020). Importantly, one third of these 30% should be of strict protection, which is a very ambitious target. Exactly what is meant by strict protection is not entirely clear, but NTZs would be an important part of this category. This new target for strictly protected areas highlights the need to evaluate the ecological effects of NTZs, particuarly in

regions like northern Europe where such evaulations are scarce. The Swedish NTZs, evaluated in the current report, made up approximately two thirds of the total areal extent of NTZs in Europe a decade ago (Fenberg et al., 2012). The total areal extent of these NTZs was around 1200 km^2 at the time of establishment of most of them in 2011, corresponding to less than 1% of Swedish coastal and marine waters. Given that these areas have been closed for at least 10 years and can provide insights into long-term effects of NTZs on fish and ecosystems, they are of broad interest in light of the new 10% strict protection by 2030 commitment by EU member states (EuropeanCommission, 2020).

1.2. Effects of NTZs on target species and fisheries

Positive effects of NTZs on abundances, biomasses and sizes of individuals within the closed areas have been found worldwide for fish and crayfish (Halpern, 2003; Lester et al., 2009; Fenberg et al., 2012). The increase in large fish may benefit fish production, since large individuals produce disproportionally higher numbers, and at the same time more viable, eggs than smaller counterparts (Beldade et al., 2012). Large individuals also have an important structuring function in the ecosystem, where at high densities they may control populations of prey species, such as smaller fish and crustaceans, through predation. Studies show that adult fish from stationary species benefit the most from NTZs, where there is a match in geographical scale between local stock sizes and sizes of NTZs (Baskett and Barnett, 2015). Life-history traits including home ranges of protected species therefore have a profound effect on NTZ success (Palumbi, 2003; Claudet et al., 2010; Green et al., 2015). Several studies have found that size, duration and placement of NTZs are important for positive outcomes (Halpern and Warner, 2002; Claudet et al., 2008; Molloy et al., 2009; Vandeperre et al., 2011).

Fishing often targets large individuals and can distort the size structure of a population (Garcia et al., 2012; Law et al., 2015). This may have evolutionary consequences since size-selective fishing may alter selection on growth rate, timing of maturation, and reproductive investment of target species (Conover and Munch, 2002; Baskett et al., 2005). Hence, targeted species may grow slower and become sexually mature at an earlier age. The establishment of NTZs can avoid this by increasing the fitness of fast-growing and late-maturing individuals, allowing local populations to develop a more natural age and size structure (Baskett et al., 2005; Pereira et al., 2017a).

While there are numerous examples of positive effects on relatively stationary target species for fisheries within NTZs, the potential effects on fish in surrounding areas, and thus the effects on fisheries, are difficult to evaluate and have been much

less studied. There are still a fair number of studies showing positive responses in surrounding areas through the export of larvae and migration of adult fish into open areas, thus contributing to the fishery (e.g. Gell and Roberts, 2003; Halpern et al., 2009; Vandeperre et al., 2011; Di Lorenzo et al., 2020). An important question for fisheries management is to what extent potential spillover effects compensate for the loss of fishing areas to the NTZ. Spillover is defined as "*the net movement of fish across the boundary of a reserve into the fished ground, which would be expected to occur on the basis of fundamental physical principles of random movement. This is in contrast to net spillover benefit which involves spillover of sufficient magnitude to compensate for lost productivity due to the closure of fishing grounds, resulting in an overall benefit to the fishery through higher catch or economic yield*" (Buxton et al., 2014). Some studies indicate that spillover effects may indeed be large enough to offset the negative effects of lost fishing areas (Halpern et al., 2009; Goñi et al., 2010; Pelc et al., 2010). However, the relationships are complex, for example, changes in density-dependent growth of populations (Gårdmark et al., 2006) and in fishing patterns (Hilborn et al., 2004) makes it difficult to establish the magnitude of spillover effects, or if the displacement of fisheries due to an NTZ have a negative impact on populations in other areas. However, NTZs may have other important positive effects on fisheries. They may serve as an insurance against mistakes in fisheries management by decreasing the risk of stock collapses (Hilborn et al. 2004, Baskett and Barnett 2015), and they may counteract a depletion in genetic diversity through size-selective fisheries (Roberts et al., 2005). In addition, protecting habitats and biodiversity is part of an ecosystem approach to fisheries management (EAFM), which can reduce for example, physical disturbance by gears like bottom trawls, and protect birds and marine mammals from bycatch and disturbance by fisheries activities.

1.3. Ecosystem effects of NTZs

Most studies evaluating the effectiveness of NTZs have focused on harvested species. However, many policy instruments, such as the Natura 2000 program of the European Union Habitats Directive, call for comprehensive evaluations of biological compartments within MPAs rather than evaluating single species (Kriegl et al., 2021). Particularly, effects of MPAs on ecosystem functioning, food-web interactions and trophic cascades are less understood (Baskett and Barnett 2015). Ecosystem effects typically take longer to develop than direct effects on targeted species. A study analysing a long-term time series of ecological data inside and outside NTZs found that direct effects on targeted species were seen after an average of 5 years, while indirect effects on other taxa took on average 13 years to become measurable (Babcock et al., 2010). The long time span before detecting

changes may be a reason why evaluations of ecosystem effects are rarer, coupled with a need for more comprehensive sampling requiring time and effort.

Fishing may have both direct and indirect effects on other species than those targeted (Langlois et al., 2012; Collie et al., 2017). Some fishing gears are more destructive for the benthic environments than others, e.g. bottom trawling. When dragged along the bottom, the trawl causes direct physical disturbance on sensitive species and indirect effects by for example increased turbidity (Thrush and Dayton, 2002; Sköld et al., 2018). This in turn will reduce the biomass and biodiversity of the benthic ecosystem and may also reduce the complexity of seabed habitats (Buhl-Mortensen et al., 2016). Food-web interactions may be affected by trawling in cases when the diet of benthic-feeding predators has changed (Hinz et al., 2017). The ecosystem effects of bottom trawling are determined by a number of factors including the type of gear used, seabed structure, trawling frequency, and the sensitivity of the species comprising the benthic community (Collie et al., 2017; Eigaard et al., 2017). Also, bycatch of fish, birds, mammals and other organisms may affect population and ecosystem structure of benthic communities (Kumar and Deepthi, 2006).

Establishing NTZs in heavily fished areas can restore and maintain ecosystem structure and function (Babcock et al., 2010; Baskett and Barnett, 2015; Pereira et al., 2017b). Particularly the increase of large fish can benefit ecosystem function (Baskett and Barnett, 2015). In coastal areas, a decrease in large predatory fish may cause negative ecosystem effectsdue to trophic cascades (Donadi et al., 2017). A loss of large predatory fish can result in an increase of small mesopredators (e.g. fish and crabs), which in turn reduce the number of small epifaunal grazers. With a decrease in grazers, epiphytic algae will increase, and the trophic cascade thereby enhances eutrophication effects (Eriksson et al., 2009; Östman et al., 2016). The epiphytic algae will overgrow and smother large canopy-forming plants and algae, and decrease the cover of these important habitats. Since these habitats are important spawning and nursery areas for large predatory fish, a viscous cycle further depressing predatory fish populations is created (Baden et al., 2012; Donadi et al., 2017) Furthermore, hypoxic conditions may occur when mats of epiphytic algae decompose (Norkko and Bonsdorff, 1996).

In the Baltic Sea, such a cascade is evident in the coastal ecosystem, where the reduction of predatory fish like pike and perch is linked to an increase in three-spined stickleback. In this case, stickleback may reinforce a low-predator state by eating eggs and juveniles of the predator, thus causing a regime shift involving an overproduction of filamentous algae and a loss of habitat-forming vegetation (Donadi et al., 2017; Eklöf et al., 2020). The effects of losing predatory fish on the

production of filamentous algae are thus similar to fertilizing the system (Östman et al., 2016). In open sea ecosystems, a reduction of large predatory fish can lead to similar regime shifts, with an increase in pelagic mesopredators. These changes may cascade down the food web to increase the biomass of phytoplankton, thus enhancing the effects of eutrophication. Such cascades have been observed in offshore pelagic environments of the Black Sea, Nova Scotia and the Baltic Sea (Daskalov, 2002; Frank et al., 2005; Scheffer et al., 2005; Casini et al., 2008). These large-scale ecosystem changes demonstrate how strong populations of large predatory fish may counteract the negative effects of eutrophication. As, NTZs may provide an efficient means of increasing abundances of predatory fish at a local scale (Baskett and Barnett, 2015), they may thus potentially be a tool for restoring disturbed ecosystems.

2. Areas and species included in the evaluation

2.1. Establishment and evaluation of NTZs commissioned by the Swedish Government

In 2005, the Swedish Board of Fisheries (which later became part of the Swedish Agency for Marine and Water Management) was commissioned by the Swedish Government to establish no-take zones (NTZs) in the Baltic Sea and in Skagerrak/Kattegat, representing both coastal and offshore areas, and to evaluate the biological effects of closing these areas to all fisheries. These NTZs were intended to be used for fisheries management of declined fish stocks and not primarily for nature conservation purposes. The NTZs were protected using fisheries legislation and not the Swedish Environmental Code, which is normally the case when establishing marine protected areas (MPAs). The NTZs were to be established around 2010 and evaluated in 2015. Five NTZs were established; one during 2009 in Kattegat, one in 2010 in Havstensfjorden at the Swedish Skagerrak coast, one at Gålö in the Stockholm archipelago at the Swedish Baltic Proper coast, and one in 2011 at Storjungfrun-Kalvhararna in the Bothnian Sea (Figure 1.). The fifth NTZ in Vinga, in the Gothenburg archipelago in Kattegat, was first established in 2003 when artificial reefs and protected areas were created, and slight changes of the areal extent were made in 2010.

The established NTZs ranged in size from 426 km^2 in Kattegat to 1,7 km^2 in Gålö with 4,4 km^2, 13km^2, and 147km^2 of protection in Vinga, Havstensfjorden, and Storjungfrun-Kalvhararna respectively (Table 1.). For all areas but Vinga, buffer zones where fishing was partly regulated, e.g. closure during spawning of focal species, were established around the NTZs at the same time. This included Storjungfrun-Kalvhararna (3980 km^2), Gålö (1,7 km^2), Kattegat (2721 km^2) and Havstensfjorden (139 km^2, Figure 1). The five NTZs were evaluated after five years of closure in a Swedish report in 2016 (Bergström et al. 2016). Since then, two NTZs (Storjungfrun and Gålö) have been reopened to fishing five years after closure, while the remaining three are still in effect. Although reopened to fishing,

they are still closed during spawning season, like the surrounding buffer zones. This partial protection has not been removed.

The county administrative boards, municipalities, non-governmental organisations (NGOs), as well as local small-scale and commercial fishers, participated in the process involving the choice and design of the NTZs and focal species. This process and the suggested NTZs were presented in a 2008 report to the Swedish Government (Sköld et al., 2008). However, the process in the Kattegat was different and stemmed from a proposal from Swedish and Danish scientists followed by a political process by the Swedish and Danish Governments (see chapter 10). The focus for all areas was on populations that had a poor status, primarily or likely as an effect of overfishing, and where the establishment of NTZs would likely have the potential to contribute to rebuilding abundance, biomass and size structure of focal species. Initially there were three suggestions for offshore NTZs, involving areas where neighbouring countries were also allowed to fish. Only the Swedish/Danish scientists' proposal to protect the cod population in Kattegat was established, however, with significant deviations from the proposal following the political process. The other two proposals were for closures to protect cod in the Baltic Proper and herring in the Bothnian Sea. However, these suggestions were rejected and instead a coastal NTZ at Storjungfrun-Kalvhararna protecting whitefish (*Coregonus maraena*) was established (Sköld et al., 2008). The other three NTZs were all placed in coastal areas with the intention to protect pikeperch (*Sander lucioperca*), pike (*Esox lucius*) and perch (*Perca fluviatilis*) in Gålö, lobster (*Homarus gammarus*) and demersal fish in Vinga, and cod (*Gadus morhua*), turbot (*Scophthalmus maximus*) and European plaice (*Pleuronectes platessa*) in Havstensfjorden (Figure 1.).

During the designation of the NTZs and buffer zones, goals, objectives, indicators and target criteria (Goals, Objectives, Indicators, Success criteria; GOIS) were defined for focal species, and sometimes for broader ecosystem effects, i.e. other components of the fish assemblage and benthic fauna, for each area. With this as a starting point, monitoring programmes focusing on the target species of fisheries in the areas were set up in order to evaluate the ecological effects of the NTZs with respect to the area-specific objectives during a five-year follow-up period. These studies primarily estimated changes in abundance, as well as in size and age distributions, of the focal species. The monitoring programmes were continued after this period, although not always on a yearly basis, in order to evaluate the long-term effects of protection as well as what may happen to populations when NTZs are reopened to fishing. Additional methods were used to evaluate the indirect food-web effects on non-targeted species and various life-stages of fish and crayfish. Non-destructive methods like hydroacoustics, traps, fykenets, trapnets and systems

with stereo-video cameras were applied when possible to avoid additional pressure. Sampling was, if feasible, conducted in collaboration with commercial fishers, both for practical reasons and because collaboration with fishers may increase the acceptance and facilitate the process of establishing and maintaining NTZs (Voyer et al., 2015a; Voyer et al., 2015b).

The ecological effects of fishing closure on food-web interactions and effects of apex predators on targeted fish species was evaluated in study areas, where the impact of seals and birds was a major concern for some stakeholder groups. The number of great cormorant (*Phalacrocorax carbo sinensis*) and grey seal (*Halichoerus grypus*) have increased strongly in the Baltic Sea during the last decades, starting from the 1980s, and there are indications that these predators may have had significant effects on fish populations in some areas (Östman et al., 2014; Hansson et al., 2018; Bergström et al., 2022). To account for predation pressure by cormorants and seals, diet composition was described and combined with information on predator abundances from national and local monitoring programs to estimate fish removals for NTZs in Gålö and Storjungfrun-Kalvhararna, where these large predators are common and may impact fish populations.

Model calculations of effort displacement of the commercial fishing fleet were done for the NTZ in Kattegat as this was the main purpose of the NTZ, i.e. to displace all fisheries from the main spawning area for cod. For the remaining NTZs effort displacement was not studied, as they are situated in coastal areas where recreational and small-scale commercial fisheries dominate. For these types of fisheries, there is no information available at a resolution that would allow estimates of displacement effects. Assessment of spillover effects of the closures were not directly assessed since this was not the primary aim of the studies. In addition it is difficult to quantify such effects in a reliable way. However, calculations on the increase in reproduction potential due to an increase in size and abundance of species has been done for some areas, since this can be expected to increase the larval production and hence indicate spillover. Hydrodynamic modelling of larval dispersal from the NTZ in Gotska Sandön has also been done, as well as tagging-recapture studies at Gålö, Kåvra and Vinga, which may give some indication of spillover effects from NTZs.
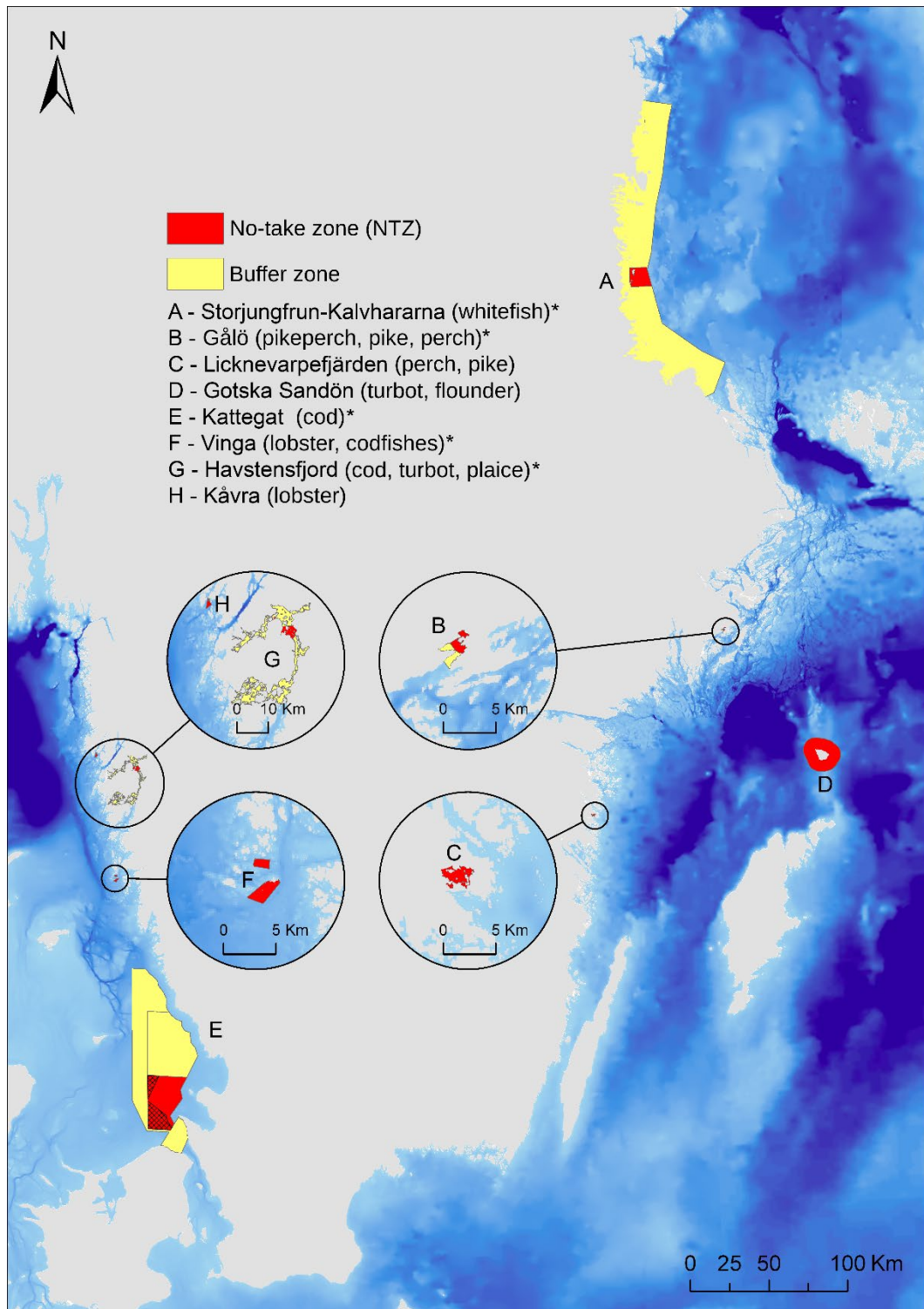
*Figure 1. Map of Sweden's eight no-take zones (NTZs) in coastal and offshore waters. Target species for which NTZs were designed are listed in brackets. Checkered area in the NTZ in Kattegat (E) is fished by a small German fishing fleet, and cannot be considered NTZ area. It is therefore removed from the analyses. * denotes areas that were established around 2010 as part of a government commission, and which were subject to a more in-depth monitoring program than the other areas, which were established earlier.*

2.2. Previously established NTZs

In addition to the NTZs described in the previous section, three NTZs were established prior to the above commission. One small NTZ (3.7 km^2) was established in 1979 as a nature reserve in Licknevarpefjärden on the Swedish Baltic coast (Figure 1). The fishing ban was introduced in order to minimize disturbance of the threatened white-tailed eagle, and not specifically to protect fish. However, being closed to fishing, it has become an area where the effects of NTZs on fish communities can be studied. To evaluate the effects of protection on large predators like pike (*E. lucius*) and perch (*P. fluviatilis*), surveys have been performed in the NTZ and a reference area from 2005 onwards, but not on a yearly basis. Predation by seals and cormorants were also included in the analysis since these predators have increased in abundance in the area during the last years.

Another NTZ was established in 1989 in Kåvra, on the Swedish west coast, in order to study the development of lobster (*H. gammarus*) populations in the absence of harvesting (Figure 1.). This NTZ is also small in size, only 2.6 km^2. The closure benefits have been clearly documented for lobster (Moland et al., 2013), however, no control fishing has been conducted outside the NTZ in reference areas, until recently. Fish surveys were done in 2017-18 within the Kåvra NTZ and in two reference areas where fishing is permitted, in order to determine if differences due to protection can be found in size, diversity, and/or abundance in the lobster population and fish communities.

Finally, a NTZ was established in 2006 around the island of Gotska Sandön, north of Gotland in the middle of the Baltic Proper, in order to protect populations of turbot (*S. maximus*) and European flounder (*Platichthys flesus*, Figure 1). The NTZ is much larger, 360 km^2, than the other two. This was the result of a special commission to the Swedish Board of Fisheries and the Swedish Environmental Protection Agency in 2002 to establish an NTZ in a marine nature reserve (M2002/731/Mk). The development of fish populations in the NTZ and two reference areas was monitored in 2006-2009 (Florin et al., 2013) and again in 2021 (this report). Potential effects of predation by seals and cormorants have been included in the analyses. Hydrodynamic modelling of larval dispersal was previously done to study spillover effects of the NTZ (Florin et al., 2013).

2.3. Meta-analysis

To assess the general effects of Swedish NTZs on fish and crustacean focal species, we performed a meta-analysis. The following coastal NTZs were considered for inclusion in the analysis: Gotska Sandön, Havstensfjorden, Kåvra, Gålö,

Licknevarp, Storjungfrun, and Vinga. However, Kåvra and Licknevarp could not be included in the meta-analysis due to insufficient data (see details below). The meta-analysis followed a before-after-control-impact (BACI) design. We performed two separate meta-analyses, one based on the Hedges' *g* measure of effect size and one meta-regression based on a time-series of log response-ratios (LRRs), for which explanations are provided below.

Table 1. All datasets sampling NTZs and reference areas considered for inclusion in the meta-analysis. LRR = log response-ratios.

NTZ name	Focal Species	Area (km²)	Implemen tation year	Inclusion in Hedge's meta-analysis	Inclusion in LRR meta-regression
Gotska Sandön	European flounder (*Platichthys flesus*)	360	2006	x	x
Gotska Sandön	Turbot (*Scophthalmus maximus*)	360	2006	x	x
Havstensfjorden	European plaice (*Pleuronectes platessa*)	13	2010		x
Havstensfjorden	Turbot (*Scophthalmus maximus*)	13	2010		x
Kåvra	European lobster (*Homarus gammarus*)	2.6	1989		
Gålö	European perch (*Perca fluviatilis*)	1.7	2010	x	x
Gålö	Northern pike (*Esox lucius*)	1.7	2010	x	x
Gålö	Pikeperch (*Sander lucioperca*)	1.7	2010	x	x
Licknevarp	European perch (*Perca fluviatilis*)	3.7	1979		
Licknevarp	Northern pike (*Esox lucius*)	3.7	1979		
Storjungfrun	European whitefish (*Coregonus lavaretus*)	147	2011	x	x
Vinga	Atlantic cod (*Gadus morhua*)	4.4	2003	x	x
Vinga	European lobster (*Homarus gammarus*)	4.4	2003	x	x

Meta-analysis using Hedges' g

The first meta-analysis measured effect size using Hedges' g (Hedges, 1981), which compared catch per unit effort (CPUE) in NTZs to reference areas before and after implementation of the NTZ for the first 3 years. Effect sizes using Hedges' g are generally considered to be small if g is less than 0.2, moderate if g is around 0.5, and large if g is around or greater than 0.8. To measure the NTZ-reference difference before implementation, we included data from the year before its implementation or, if not available, the same year that the NTZ was enforced. We assumed that no effect would be found the same year as the NTZ was implemented. To measure the NTZ-reference effect size after implementation, we used data from the year after 3 years of protection had elapsed. We chose to measure the "after" effect size after 3 years of protection because CPUE was sampled for these years across all of the datasets, allowing a more robust comparison of effects between the NTZs. Importantly, it should be noted that to calculate Hedges' g, it is necessary to have data on means, sample size, and standard errors for that year in both the NTZ

and reference areas, and to have sampled in the year of, or year prior to, enforcement. As such, data from Havstensfjorden, Kåvra, and Licknevarp were not included in this analysis, as this information was not available.

To assess the overall effect size, we used the "meta" package in R (Balduzzi et al., 2019). To account for differences in sampling gear, time period, and study species, we used a random-effects model. Between-study variance was estimated using the Hartung-Knapp-Sidik-Jonkman (HKSJ) method, which produces more conservative estimates, and generally outperforms other methods (IntHout et al., 2014).

Meta-regression using log response-ratios

The second meta-analysis we performed measured effect size over time using log reponse-ratios (LRRs). In this meta-analysis, LRRs were calculated as follows:

$$LRR_p = \ln \frac{CPUE_t}{CPUE_{before}}$$

Where p is the protection type (NTZ or reference area), and t is the number of years since implementation of the NTZ. For reference areas, the number of years since implementation of the corresponding NTZ was used. LRRs were calculated as such for NTZs and reference areas for all years in which data were available. It should be noted that only datasets for which "before" data were available for either the NTZ and/or reference area could be included. As such, data from Kåvra and Licknevarp were not included in this analysis, as no before data were available in either the NTZ or the reference area.

We compared the temporal trend in NTZs to reference areas using a generalised linear model (GLM), where the LRR served as the response variable, protection type (NTZ or reference area) served as a categorical predictor variable, and the number of years of protection served as a continuous independent variable. To test for potential differences in slope between NTZs and reference areas, we performed a GLM model with an interaction between protection type (NTZ or reference area) and the number of years protected.

3. Results

In this section we give a general overview of results from all NTZs and the meta-analysis, while detailed reports from each NTZ is presented in separate chapters (6 - 13). Summaries of the results from the evaluations in relation to goals and indicators (GOIS) for the NTZs, commissioned by the Swedish Government in 2005, are presented in tables in these separate chapters.

To provide an overview of the effects in all eight NTZs included in the report, a table is presented, summarising short-term effects from the previous evaluation in 2016, corresponding mostly to a five-year period after the NTZs were established, and long-term effects from the current report, corresponding to ten years or longer after establishment (Table 2). We also provide results for the two areas that have been reopened to fishing (Storjungfrun-Kalvhararna and Gålö). We summarise results on the effects of NTZs on focal species and on other species within the ecosystems when available. This includes both fish communities and benthic fauna. For focal species we evaluate the effects on three fundamental population parameters including abundance (catch per unit effort, CPUE), size (mean size or CPUE of large individuals), and growth (size at a given age). To evaluate effects on ecosystems, we examine indirect effects on non-focal species through food-web interactions, and on benthic faunal communities through a decrease in disturbance from trawling. We also present results from the meta-analysis, which summarise the effect sizes observed in the case studies.

3.1. Effects of NTZs on focal species

Short-term effects

Of the seven NTZs for which there were data on the short-term development, all but Havstenfjorden displayed increases in CPUE of focal species during the initial 5 years of closure (Table 2). In most cases, body sizes and/or ages of focal species also increased over time in NTZs compared to reference areas. This shows that removing fishing pressure in these areas leads to strong positive responses in abundance and size of focal species in the NTZs, including species subjected to commercial fisheries (e.g. cod and Norway lobster), and species caught mainly in recreational fisheries like pike, perch, pikeperch and lobster. Growth rates in NTZs were lower in some populations, showing that density-dependent effects may decrease the effect of the NTZs. However, one population showed a clearly higher

growth rate in the NTZ than in the reference area, indicating that the negative size-selectivity of fisheries may have been counteracted.

The meta-analysis confirms the initial positive effects in NTZs within the first six years. There was an overall significant positive effect of NTZs when comparing NTZs to reference areas after implementation (Figure 2, $p = 0.019$). The overall effect size for NTZ-reference areas after three years of implementation was moderate to large (Hedges' $g = 0.638$). Before implementation, there was no difference between NTZs and reference areas (Figure 2, $p = 0.119$). This meta-analysis thus shows an overall moderate to strong positive effect of the NTZs.
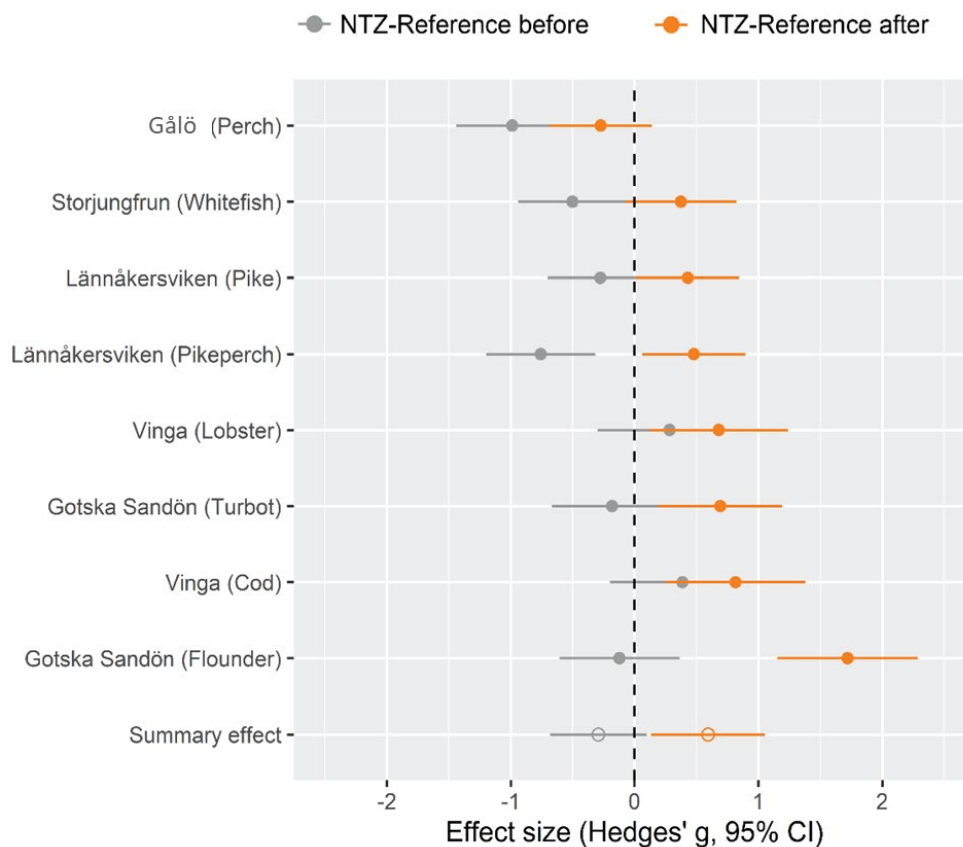
Figure 2. Effect size, measured using Hedges' g, comparing NTZs and reference areas before implementation of the NTZ (grey) and 3 years after implementation of the NTZ (orange). Bars represent 95% confidence intervals. The summary effect sizes of the meta-analysis are represented by open circles (bottom).

Table 2. Summary of the short- and long-term biological effects of no-take zones (NTZs) in Sweden on focal species (abundance, size, growth) and on other parts of the ecosystem (fish and benthic communities). + depicts a positive effect, - a negative effect, 0 no effect and an empty cell means that aspect has not been evaluated for that particular area. The areas marked with an asterisk (*) have been reopened to fishing after 5 years. The area marked with ** has only been evaluated in 2016, with no new evaluation in 2022. Short-term effects in the table include studies up to 5-6 years after establishment of the NTZs, and long-term effects comprise effects 10 or more years after establishment.

| Area | Year Est | Reopened | Year Eval | Size (km²) | Species | Short-term effects | | | | | Long-term effects | | | | |
| | | | | | | Effects on focal species | | | Ecosystem effects | | Effects on focal species | | | Ecosystem effects | |
						Abund	Size	Growth	Fish commun	Benthic fauna	Abund	Size	Growth	Fish commun	Benthic fauna
Storjungfrun-Kalvhararna, Gävleborg*	2011	2016	2011-2021	147	Whitefish	+	+								
					Sea trout	+									
Gålö, Stockholm*	2010	2016	2010-2015, 2020-2021	2	Pike-perch	+	+	0	+						
					Pike	+	+								
					Perch	0	0								
Licknevarpefjärden, Östergötland	1979		2005, 2013, 2020-2021	4	Pike						+	+	-	+	+
					Perch						+	+	+		
Gotska Sandön, Gotland	2006		2006-2009, 2021	360	Turbot	+	+	-			+	+			
					Flounder	+	+	-			+	+			
Kattegatt, Halland, Skåne	2009		2009-2021	650	Cod	+	+		+	+	0	0	0	+	+
					Norway lobster	0	+				+				
					Dab						+				
					Lemon sole						+				
					Turbot						+				
Vinga, V. Götaland**	2002		2002-2006, 2008-2010, 2014-2015	5	Lobster	+	+		+	+	+	+		+	+
					Cod	+	+				0	+			
					Poor cod	+					+				
					Brown crab	-					-				
Havstensfjord, V. Götaland	2010		2010-2021	13	Cod	0	0				+/0	-/0	0		
					Turbot	0	0				0	-	-		
					Plaice	0	0				0	0	0		
Kåvra, V. Götaland	1989		1994-2007, 2017-2018	3	Lobster	+	+				+	+		0	-

Long-term effects

Positive long-term effects on focal species were seen for many areas and species, but not for all. The lobster population in Kåvra was still larger and sizes of lobster continued to increase in the NTZ compared to the reference areas 30 years after the area was closed, suggesting a long-term build-up of the focal species in this area (Chapter 8). This extensive period of closure has resulted in approximately four times higher CPUE, significantly larger individuals, and 7-9 times higher reproductive potential within Kåvra NTZ compared to surrounding reference areas (Chapter 8). However, there was no continued increase in CPUE, calculated by numbers of individuals, since the values yielded from the historical sampling ending in 2007 (Moland et al., 2013), suggesting that Kåvra is approaching carrying capacity for lobster after three decades of closure. Despite the positive response of the focal species *H. gammarus* to the NTZ, no differences in the fish community between the NTZ and the reference area could be observed. Similar long-term effects on lobster were found in Vinga, where the catches of lobster continued to increase over time, and 13 years later were three to six times higher in the NTZ than in the reference area (Chapter 11). Sizes of lobster also continued to increase over time within the NTZ.

CPUE of both large and small flounder and turbot in the NTZ at Gotska Sandön have also remained significantly higher in the NTZ compared to the reference areas over time (2006-2021). However, CPUE generally decreased for all but small turbot in both the NTZ and reference areas from 2009 to 2021, suggesting other factors than fishing pressure are causing the recent decline in flatfish.

The meta-regression using log-response ratios of focal species from all the coastal NTZs, confirmed these long-term effects. In the GLM of log response ratios there was a significant interaction effect between protection type (NTZ or reference area) and the number of years of protection (Figure 3, $p = 0.047$), indicating a difference in population development over time in NTZs compared to reference areas. In NTZs, there was a clear positive trend in CPUE as the number of years of protection increased, whereas in reference areas there was no trend (Figure 3). According to the model predictions, the LRR for NTZs is 0.95 after three years of protection, and 1.34 after six years of protection, which means that the CPUE was 2.6 times higher after three years of protection, and 3.8 times higher than in the fished reference areas after six years of protection. Longer-term effect size estimates are uncertain, as data is scarce, but indicate that the effect sizes may continue to increase.
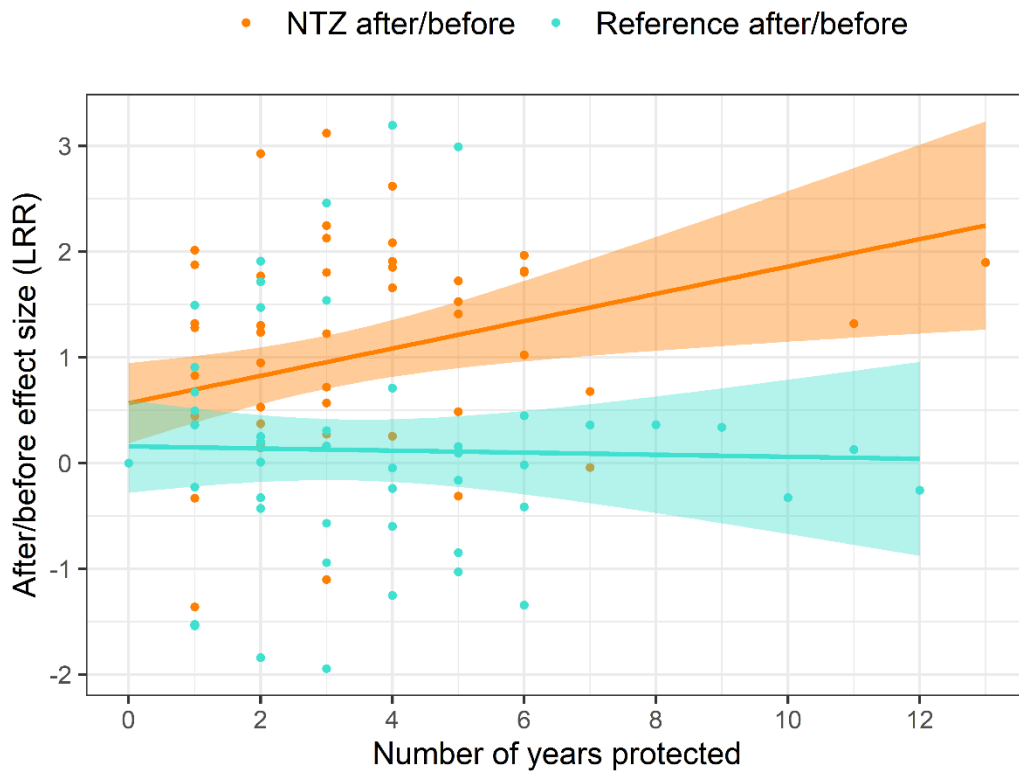
Figure 3. Regression lines for the relationship between the number of years protected and the after/before log response-ratios (LRR) for NTZs (orange) and reference areas (turquoise). Shaded areas represent the 95% confidence intervals of the regressions.

CPUE and sizes of focal species have, however, declined in some of the NTZs after more than 5 years of protection. For example, in the Kattegat (which was not part of the meta-analysis) as a whole, CPUE of cod has declined, despite two years of relatively good recruitment during the studied period and an initial increase in cod CPUE between 2009 and 2015 (Chapter 10). The decline is due to increased fishing effort and a reduction in the use of selective gear outside the NTZ following abolishment of effort regulations in 2016, and successive implementation of the EU landing obligation after the first evaluation of the NTZs in 2016. Bycatches in the trawl fisheries have, and are still causing, high fishing mortality of cod in the Kattegat outside the NTZ (Sköld et al., 2022). CPUE of Norway lobster (*Nephrops norvegicus*), on the other hand, increased in the NTZ compared to the reference and partially protected areas, suggesting long-term positive effects for this target species for the fisheries in Kattegat. In comparison to the mobile cod, Norway lobster has more limited adult dispersal with a home range of less than 9 km (Chapman, 1980). Additionally, a positive response to the establishment of the NTZ was found in several other species with significant increases in biomasses and abundances of dab (*L. limanda*), lemon sole (*Microstomus kitt*) and turbot (*S.*

maximus) over time in the NTZ compared to the reference or partially protected areas, where bottom trawl fisheries continue.

In Licknevarpefjärden, CPUE of large perch and pike were much higher in the NTZ than the reference area in the first survey conducted 25 years after the area was closed. In the following surveys, up to 40 years after closure, large perch and pike had both declined since the first survey 25 years after closure. These declines are likely explained by the large populations of grey seal and cormorant in the area during recent years, leading to a high predation pressure (Chapter 9).

In Havstensfjorden, there is still no detectable recovery after 12 years of full protection in NTZs and strict fishing regulations in the large surrounding buffer zones. The cod and flatfish populations in this area were highly diminished when the NTZ was put in place, and it may take longer than 12 years to recover due to impaired reproduction.

Effects of reopening the NTZs

The CPUE of whitefish declined after reopening the NTZ in Storjungfrun-Kalvhararna and the CPUE of pikeperch and pike declined after reopening the NTZ in Gålö, despite that protection during spawning was still in effect, resulting in CPUEs similar to those before the NTZs were established in both areas. This implies that a spawning closure is not sufficient to maintain these populations, instead a year-round fishing ban is needed. In the Gålö area, the declines are likely explained by a combination of a high predation pressure from grey seal and cormorant and from reopened recreational fishing.

3.2. Effects of NTZs on other ecosystem components

In Kattegat there were significant shifts in the macrofaunal assemblage within the NTZ over the 12 years following the enforcement of the NTZ, indicating a long-term recovery pattern following paucity of bottom trawling. Closer analysis indicated that individual species contributed to the shift, mainly by increasing in abundance over time, but a few species also decreased. The dominant brittle stars *Amphiura filiformis* and *A. chiajei* displayed an initial decrease within the NTZ followed by a lowered abundance and biomass over the 12 years of protection in comparison to the reference stations where trawling continued. This may be a result of increased predation on the brittle stars by demersal fish and the target species for bottom trawling Norway lobster, *Nephrops norvegicus,* in the area, which increased in the NTZ over time.

In Vinga, the benthic fish and crustacean assemblages rapidly shifted in the NTZs compared to the fished reference area. These differences were mainly driven by increased abundances of Atlantic cod (*G. morhua*), poor cod (*Trisopterus minutus*), lobster (*H. gammarus*) and wrasses (corkwing wrasse *Symphodus melops* and rock cook *Centrolabrus exoletus*) in the NTZs, and a decline in the abundance of small decapod crustaceans. Higher abundances of smaller crustaceans were found in the reference area. The decline of small decapod crustaceans was likely due to increased predation on these decapods in the NTZ caused by the higher abundances of large predators.

Long-term ecosystem effects of closure were detected in Licknevarpefjärden, and the fish community composition differed significantly between the NTZ and reference area when first studied 25 years after the NTZ was established. These differences were mainly an effect of higher abundances of large predatory fish and lower abundances of cyprinid fishes, where the latter was likely an effect of increased predation by large predators. However, the fish community of the NTZ has displayed a pronounced shift during the last five years, mainly caused by the decline of predatory fish in combination with an increase in cyprinid species such as roach, bleak and white bream. The increase in cyprinids is likely an indirect effect of the decline in pike and perch in the NTZ over time. This decline, in turn, is likely a result of the increasing abundances of grey seal and cormorant in the area since the beginning of the 2010s. These two top predators now exert a substantial predation pressure on the large predatory fish (Chapter 9).

Similar to Licknevarpefjärden, the fish species composition in the NTZ and reference area diverged somewhat over time in Gålö. The change was partly caused by the increase in abundance of pikeperch in the NTZ. Catches of large predatory fish (pike, pikeperch and perch) decreased after reopening the NTZ after five years of closure and catches of other fish species were highly variable with no clear patterns in the NTZ and reference area. A longer period of closure is likely needed until the increased numbers of predatory fishes and their subsequent increase in predation on other species will cause detectable more substantial changes on fish community level (Babcock et al., 2010; Eger and Baum, 2020).

In Kåvra, there were no significant differences in fish communities between the NTZ and reference areas. However, the brown crab *Cancer pagurus* was three to four times more abundant in the reference areas than in the NTZ in the 2018-19 surveys. This is likely a result of an interaction between the brown crab and the European lobster, where brown crab is outcompeted by the more aggressive European lobster when the lobster population increases in size and abundance in

the NTZ. Similar patterns have been observed by (Rossong et al., 2006; Skerritt et al., 2020).

In summary, the effects on ecosystem components, other than the focal species for the establishment of the NTZs, are often related to interactions between species, in particular as an effect of predation, where prey species of crustaceans and fish decrease when the predators increase. When bottom trawling is stopped, the benthic fauna composition changes and most species contributing to the shift increase in abundance, but some also decrease. For example, the dominant species, brittle stars, was found to decrease in the NTZ, which can be linked to predation from benthic feeding predators that increase in the NTZ.

3.3. Effects of predation from grey seal and cormorant

In all the case studies in the Baltic Sea, the effects of predation from grey seal and great cormorant on the focal species were also evaluated. In discussions with stakeholders during the planning of the NTZs, a concern that was repeatedly raised was that predation from these top predators would override any positive effects of removing fisheries. To account for the potential effects of predation, information on diet composition, bioenergetic needs and predator abundances were combined to estimate fish removals in the NTZs Storjungfrun-Kalvhararna, Gålö, Licknevarpefjärden and Gotska Sandön.

The estimated removals of biomass of the focal species of the NTZs by grey seal and cormorant were high in all these areas, and were estimated to be similar in magnitude to fisheries landings prior to the establishment of some of the NTZs (Florin et al., 2013; Berkström et al., 2021a). In Gålö and Licknevarpefjärden there were also indications of negative effects of seal and cormorant predation on perch and pike populations. In Licknevarpefjärden, a strong decline in the populations of these two predatory fishes was observed, despite that the area had been closed to fishing since 1980. This decline coincided with a pronounced increase in seal and cormorant abundances in the area, which suggests that the growing populations of top predators may have a considerable effect on coastal fish in the Baltic Sea. This is in line with previous studies (Östman et al., 2014; Hansson et al., 2018; Bergström et al., 2022), but the results from these NTZs are unique in that we have been able to separate the effects of fishing and seal and cormorant predation by following the population development in areas with or without fishing, but with similar abundance of predators.

3.4. Redistribution of fishing activity

When establishing a NTZ, a consequence is often that fishing is redistributed to other areas outside the NTZ. Displacement may thus result in increased fishing effort in surrounding areas and if overall fishing mortality is not reduced, the intended protection on focal species may not be achieved.

In Kattegat, the only area with a large commercial fishery of bottom trawlers, the NTZ led as intended to displacement of the fisheries to other areas in Kattegat. The aim of establishing the NTZ and the buffer zones was to redistribute the targeted cod and non-selective fisheries for Norway lobster and flatfish away from the cod spawning grounds including areas where mature cod were abundant. Analysis of this displacement indicate, as expected, that the negative impact of the fisheries on larger cod in the Kattegat decreased as a consequence of the NTZ. For small cod, 12 - 24 cm, the spatial regulation measures instead slightly increased the impact from the fishery, mainly since the small cod individuals were distributed in higher densities in areas outside the NTZ including the buffer zones, where relative fishing effort increased. The introduction of more selective gears in combination with a reduced overall fishing pressure did however also decrease the mortality of small cod during this first period. The resulting mortality of small cod for the year 2015 was estimated to be reduced to 37% of the mortality in 2008. Taken together, the measures of establishing the NTZ in combination with the reduction in total fishing effort and increased gear selectivity decreased cod mortality and coincided in time with the temporary recovery of the cod stock.

For most of the NTZs, it was difficult to evaluate the effects on fishing redistribution as they are situated in the coastal zone, where recreational and small-scale commercial fisheries dominate. For these fisheries, the spatial resolution and availability of data is too low to make detailed analyses of displacement effects. However, some data are available indicating redistribution or reduction of fishing activity.

In the NTZ and buffer zone at Storjungfrun-Kalvhararna, commercial fishing using gillnets and push-up traps occurred prior to the closure. During the period of the NTZ (2011-2016) the fishing effort with gillnets and push-up traps decreased with 23-39% compared to the period 2006-2010, before the establishment of the NTZ. During 2017-2020, when the NTZ was re-opened, the fishing effort continued to decrease by another 35%, suggesting an overall decrease in fishing effort in the area and likely not redistribution of fishing following protection. The overall decline in fishing effort is likely connected to the low profitability of small-scale fisheries, and a decreasing number of active coastal fishers within this region (Bergenius et al., 2018). The spatial resolution of the catches reported from small-scale coastal

fisheries is, however, low and does not allow any analysis of potential spatial shifts in fishing efforts related to the protection level. For recreational fisheries, data is not available to allow any analyses of effects on fisheries.

In Gålö and Licknevarpefjärden, commercial and recreational catches of pike, perch or pikeperch were not quantified. At Gotska Sandön, fishing effort (both commercial and recreational) was reported to be low at the time of the NTZ establishment and major shifts in spatial fishing effort were not expected. The positive effects of the closure on flatfish CPUE and sizes suggests, however, that fishing pressure was still high enough to have caused declines in the local fish assemblages. However, it is not possible to quantitatively assess fishing effort redistribution due to the lack of data collection.

Information on fishing effort in the NTZ in Kåvra and Vinga is lacking. However, the closed areas are very small (2,6-4,4 km^2) and any displacement of fishing effort would have a marginal impact on the overall fishing pressure in the surrounding areas. In Havstensfjorden, fish stocks of focal species were already depleted and commercial fishing effort was very low 5-10 years before the closure of the NTZ. A possible shift in effort or redistribution in Havstensfjorden would thus likely be by the recreational fishery, but it is not possible to quantitatively assess this due to a lack of data.

4. Discussion - The use of no-take zones in management

Overall, the results from this report suggest that NTZs can increase abundances and biomasses of fish and decapod crustaceans targeted by fisheries, given that the closed areas are strategically placed and of an appropriate size. NTZs may also aid in recovery of benthic communities through the removal of disturbance from bottom trawling. The size and age structure of fished assemblages recovered in the NTZs evaluated in this report, increasing the proportion of old and large individuals, and thereby also the reproductive potential of populations. Increasing abundances of large predatory fish may also contribute to restoring ecosystem functions, such as top-down control. These effects appeared after a 5-year period and in many cases remained and continued to increase over time (10+ years). For stocks that are heavily overfished, positive responses may, however, be very slow or absent, e.g. cod in the Kattegat and Havstensfjorden. However, in the Baltic Sea, predation by increasing populations of grey seal and cormorant can in some cases, e.g. Licknevarpefjärden, be so high that it overrides the positive effects of removing fisheries, leading to stock declines in the NTZs.

In Kattegat, reliefs in fishing regulations outside the NTZ diminished the initial positive effects on the abundance and biomass of the focal species, cod. In areas that were reopened to fishing after 5 years, the positive effect of the NTZ on fish stocks eroded quickly despite that the areas remained closed during the spawning period, which highlights that permanent closure may be necessary to maintain the positive effects.

Taken together, well-designed NTZs may provide several positive effects, and function as a complement to other fisheries management measures, but cannot be implemented in isolation from other regulations to manage diminishing fish stocks. NTZs may be particularly useful in management of overfished local populations, and for counteracting negative ecosystem effects of fishing, such as habitat degradation due to disturbance by bottom contacting gear, and loss of trophic control by large predatory fish. By providing benefits to both fisheries and conservation, NTZs may constitute an important tool in ecosystem-based management. They may also be valuable as reference areas in management, to better understand the effects of fisheries on target species and ecosystems and to aid in setting targets for the ecological status of coastal and marine environments.

4.1. Fisheries management

NTZs increased the abundance and biomass in species with small home ranges like pike, perch, pikeperch, lobster and Norway lobster and also in more mobile demersal fish species like flatfish and cod, suggesting NTZs may be a useful tool in fisheries management for a variety of species with varying home ranges, as long as these home ranges are matched with the sizes of the NTZs. The NTZs in this report were designed for specific species being depleted to various extent and accounting for home ranges and spawning seasons, likely contributing to their positive outcomes.

An initial loss of fishing grounds occurs when establishing a NTZ, but these losses may be offset to some extent by positive effects on fisheries catches in surrounding areas through larval export and adult migrations out of the NTZ if densities increase inside the protected area (Halpern et al., 2009). The magnitude of these positive spillover effects is difficult to quantify in relation to the loss of fishing opportunities. Higher densities and larger mean sizes of the target species can be expected to increase the potential for spillover of adults due to intraspecific interaction, and also increase the larval production with export of larvae from NTZs to fished areas, as large individuals produce disproportionally larger numbers and more viable eggs than smaller ones (Beldade et al., 2012). However, NTZs may improve fisheries also in other ways than through spillover effects. NTZs may decrease the risk of stock collapses by acting as an insurance against overfishing and may also counteract the negative effects of size-selective fisheries, such as early maturation and slow growth (Conover and Munch, 2002; Baskett et al., 2005; Roberts et al., 2005; Baskett and Barnett, 2015). However, most importantly NTZs is a useful tool in rebuilding overfished stocks.

Indeed, for most areas an increase in density and sizes of the focal species were found during the first 5-6 years after closure, suggesting an increase in the reproductive potential, which may lead to an increase in spillover of both adults and larvae over time. This is in line with a review by Babcock et al. (2010) that found that the initial positive effects of NTZs occur after 5 years. In addition, the benefits of marine protection have been found to be greatest when accumulating five key features; the protected area is fully no-take, well enforced, has been in place >10 years, covers a large area ($>100km^2$) and is isolated (Edgar et al., 2014). The NTZs in our current report are fully no-take, fairly well enforced, and have been in place 10+ years (except Gålö and Storjungfrun-Kalvhararna that were reopened to fishing after 5 years). Five of the eight studied NTZs are rather small in size ($<13km^2$) but sizes are matched to focal species' home ranges and thus should be of adequate size. Regarding isolation, this is mainly described for NTZs in reef areas where

isolation refers to reefs being surrounded by deep (>25m) water or large expanses of sand and is not applicable for the NTZs in the current report.

At Gotska Sandön, hydrodynamic modelling of larval dispersal showed that there is a net transport of turbot and flounder larvae from the NTZ into fished areas around the large island of Gotland, which together with the build-up of the spawning stock biomass within the NTZ likely contribute to substantial larval export to the fished area (Florin et al., 2013). Although the extent of spillover from NTZs to surrounding waters, and the potential to compensate for lost fishing areas cannot be quantified from the case studies, it is clear that in most cases the populations of focal species that resided inside the NTZ increased after the areas were closed. Abundances of focal species were almost four times higher after six years (see meta-analysis). In combination with the increase in mean sizes, this suggests that the build-up of biomass inside the NTZs can contribute to the recovery of depleted stocks and in the long run function as an insurance for fisheries management failures outside the NTZs. While a single NTZ may lead to limited spillover, a network of connected NTZs may contribute to maintaining viable populations of species targeted by fisheries since many species may disperse far, either passively via larvae or actively swim between foraging, spawning and recruitment areas (Berkström et al., 2021b). Besides protecting 30% of our ocean by 2030, the EU Biodiversity Strategy also states specific objectives for a connected and ecologically coherent MPA network. Further, a central objective is strict protection of a third of the protected areas (i.e. 10% of EU coastal and marine waters), which implies that a considerable part of the expanded MPA network will likely be protected as NTZs. From a conservation perspective, unique, sensitive and/or charismatic species or habitats are often in focus when designating MPAs, and displacement of fisheries is then considered an unwanted side effect. However, if the establishment of strictly protected areas also aims to rebuild fish stocks, these MPAs should be placed in heavily fished areas and designed to protect depleted populations by accounting for their home ranges to generate positive outcomes. Thus, extensive displacement of fisheries is then required to reach benefits for depleted populations, sometimes accompanied by specific regulations outside the strictly protected areas.

Buffer zones with lighter fisheries restrictions, i.e. partially protected areas (PPAs) with seasonal protection like in our studies, surrounding the NTZs may be a way of increasing or speeding up the build-up of biomass within the closed areas. A recent meta-analysis by Ohayon et al. (2021) on edge effects of MPAs, found that population sizes were on average 60% smaller at the edge of the protected areas compared to central parts, but that MPAs with buffer zones did not display these edge effects. They suggest that extending NTZs beyond the target habitats with buffer zones and managing activities around MPA borders is critical for boosting

MPA performance. Sciberras et al. (2013) also found PPAs to be effective in increasing abundance and biomass of targeted species, although NTZs are the most effective. Enforcement and compliance by commercial and recreational fishers is also of great importance for MPA success (Gall and Rodwell, 2016; Arostegui et al., 2021).

Temporary NTZs may be an additional way of restoring fish populations to the benefit of fisheries. This approach was tested in the Storjungfrun-Kalvhararna and Gålö NTZs, as the two areas were opened up to fisheries after being closed for 5-6 years. Economic cost-benefit analyses indicated that these two temporary NTZs likely resulted in positive net effects for fisheries (Bostedt et al., 2020). These analyses tested a number of scenarios related to different responses of the target populations after closure. However, the follow-up studies after reopening the areas later revealed that the populations in both areas showed a more pronounced decline than anticipated in the scenarios, despite that the areas were still closed to fishing during spawning season. Rapid declines in abundance and biomass after reopening areas closed to fishing have been found elsewhere, although evaluations like these are rare. In California, densities of red abalone declined by 65%, the size structure changed and egg production drastically declined only three years after opening up the marine reserve to fishing (Rogers-Bennett et al., 2013). The reserve had been mostly closed since 1917 and fishing for abalone was only allowed during a few days a year by an invited number of guests (10-30). In the Philippines, the density of large predatory fish declined after reopening two NTZs that were periodically closed for 2-5 years (Russ and Alcala, 2003). Each time the area was closed to fishing, the density and biomass of predatory fish increased by 70-95% and when reopened decreased with the same magnitude, with similar low densities and biomasses as before closure within 2-5 years. These results indicate that while positive effects appear soon after areas are closed to fishing, these effects may also erode very quickly. Thus, temporary closures may be a way of increasing depleted populations and may provide some benefits after the areas are reopened, but these benefits may not last for long unless other regulations limit fishing mortality. In summary, if a long-lasting insurance effect for fisheries and restoration of ecosystems is the goal, strictly regulated NTZs are most beneficial if they are permanent.

4.2. Marine conservation

NTZs may benefit other ecosystem components than the target species of fisheries, with positive effects on ecosystem structure and function. Thus, NTZs contribute to fulfill goals in the EU Marine Strategy Framework Directive and the EU Habitats Directive, in addition to national and local nature conservation objectives.

Removing fisheries can for example benefit benthic fauna by reduced disturbance from destructive fishing gear as demonstrated in the NTZ in the Kattegat (Sköld et al. 2018, this study). Removing fisheries may also have indirect effects on benthic communities by changes in species interactions as a result of higher densities of species targeted by fisheries. A common effect of NTZs is often an increase in large predatory fish, as these are often highly affected by size selective fisheries. These higher predator densities may restore food-web functions of coastal ecosystems, which may counteract the effects of eutrophication and decrease the risk of regime shifts (Eriksson et al., 2011; Baden et al., 2012; Östman et al., 2016; Donadi et al., 2017; Eklöf et al., 2020). Although we did not have the resources to test causal effects directly in our NTZs, we observed changes in both fish communities and benthic fauna, suggesting indirect ecosystem effects occurred in these areas as a consequence of the increase in large predatory fish.

In Sweden, like elsewhere, fishing is often the most prominent human pressure in coastal and marine waters. However, because the legislation and management sectors for nature conservation and fisheries are often dealt with separately, the protection of biodiversity and fisheries management is rarely combined (Sørensen and Thomsen, 2009; Grip and Blomqvist, 2020). Regulating fishing in MPAs is often pcontroversial as it may affect the livelihood of local fishers. The separation in management between conservation and fisheries management has also provided an easy way out since conflicts between conservation targets and ongoing fisheries can be minimized e.g. by adjusting borders for areas and allowing less damaging, or species selective, gear within MPAs. There is often strong resistance against NTZs from fishers which partly stems from a lack of experience from positive effects on fish populations from closing specific areas to fishing.

Hence, fisheries are often not regulated in MPAs and NTZs are rare. For example, trawling is allowed in the majority of European MPAs, and average trawling intensity is actually higher inside the MPAs than in non-protected areas (Dureuil et al., 2018; Perry et al., 2022). Also, if NTZs are regulated through fisheries legislation, it may be difficult to keep areas closed for longer periods of time if the populations of focal species have recovered. This was the case for both Storjungfrun-Kalvhararna and Gålö in the Baltic Sea, where populations of the focal species whitefish and pikeperch had recovered within the protected areas and NTZs were therefore reopened to fishing after 5-6 years as the Swedish Fisheries Act can only regulate fisheries that are harvested unsustainably. When reopened to fishing, the populations declined to similar levels as before the protection within a few years.

Nonetheless, most ecosystems would benefit from combining nature conservation and fisheries management, which has been advocated in ecosystem-based fisheries management (Halpern et al., 2010; Baskett and Barnett, 2015; Grip and Blomqvist, 2020; Sala et al., 2021). This has also been acknowledged in the Swedish marine management action plan (SwAM, 2016) and the new Swedish framework for MPA network design and management (SwAM, 2021). The new EU goal of protecting 30% of our ocean by 2030 is promising, but there is a risk that the race to protect 30% may detract from the importance of quality and adequate design of MPAs in order to achieve conservation and fisheries management goals. Focus on quality, not only quantity, is needed, where protecting enough of the right habitats and species in the right places is prioritised, and where protection means areas are closed to fishing in cases where this is necessary to reach conservation objectives.

4.3. Future research needs

To develop the use of no-take zones as a tool for ecosystem-based management, the following areas of research need further attention:
- *Indirect effects on ecosystems*: Removing fisheries may also have indirect effects on fish and benthic communities by changes in species interactions and ultimately food web functioning, which may affect habitat quality. These effects are important to identify and understand. Studies on indirect effects are, however, scarce, and a better understanding is needed to advance the use of NTZs in ecosystem-based management.
- *Impacts of top predators*: we have seen indications of negative effects of seal and cormorant predation in several NTZs, but calculations are uncertain. We need better estimates of abundances of the predators in the NTZs, as well as of their diet composition, to estimate their potential population-level effects on fish. A related question is whether top predators aggregate in protected areas due e.g. to a lower level of human disturbance, and how these effects interact in the food web.
- *Long-term effects on focal species and ecosystems*: Studies on long-term effects of NTZs are limited. Available examples indicate that population-level and ecosystem effects may build up over several decades.
- *Effects of NTZs on fish behaviour and genetic selection*: the absence of fishing may potentially gain bold, fast-growing individuals, as fishing is known to induce timidity in fish (Arlinghaus et al., 2017), as well as slower growth (Conover and Munch, 2002). These behavioural and genetic changes may have substantial ecosystem effects, and may also impact fish catches and profitability.
- *Optimising design of NTZ, e.g. size, placement in MPA networks*: To maximise recovery and resilience in populations, a network of strictly

protected MPAs may be more efficient than individual MPAs/NTZs. Creating a network of different sized MPAs/NTZs with appropriate distances related to the ecology of the species of interest may hence be an efficient way of protecting populations and metapopulations. However, knowledge on effective network design is limited.

- *Effects on fisheries*: Spillover effects is one of the corner stones of spatial protection of fisheries, but the magnitude of spillover effects are poorly understood. NTZs may also provide an insurance effect against overfishing, and can counteract adverse genetic and behavioural effects of fishing. These mechanisms need to be better understood for optimising the use of NTZs in management

- Socio-economic effects of NTZs: Few studies link ecological effects of protection with socio-economic benefits, highlighting the ecosystem services that may be improved and the monetary value of population recoveries.

- Efficiency of NTZs in relation to, or combined with, other fisheries regulations: A management strategy where NTZs are combined with various fisheries regulations may be an efficient way of restoring overfished stocks and manage ecosystems. Here, a review of case studies of different combinations of NTZs with other measures may be useful to improve future management strategies.

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6. No-take zone for whitefish in the Bothnian Sea

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Photo: Ulf Bergström

Summary

In 2011 a no-take zone (NTZ) of 147 square kilometers was established at Storjungfrun/Kalvhararna in the archipelago of Söderhamn, Bothnian Sea. At the same time, a larger area with fishing closure during the spawning period was established along the coast of Gävleborg county and the northern parts of Uppland county. The establishment of the regulated areas aimed to increase the populations of sea-spawning whitefish in the southern parts of the Bothnian Sea.

In order to monitor the effects of the NTZ and spawning closure on sea-spawning whitefish populations in the Bothnian Sea, gillnet surveys were carried out on a yearly basis from 2011 to 2021 during the whitefish spawning season (Oct-Nov) to avoid inclusions of river spawning whitefish. In addition, during late spring 2011-2015 young-of-the-year whitefish were sampled in potential coastal spawning and nursery habitats in the NTZ and the spawning closure area.

A significant increase in adult (>30 cm) whitefish was found in both the NTZ and the spawning closure area compared to the reference area in 2011-2016. Further, the increase in CPUE of adult whitefish was significantly higher in the NTZ than in the spawning closure area. A similar pattern was found for sea trout, indicating that stricter protection (i.e. no take) is more effective than only spawning closure to strengthen coastal fish populations.

In 2016, the NTZ was reopened to fishing and in the following period (2017-2021) catches of whitefish were sparse and there were no longer any significant differences in the development of CPUE of adult whitefish over time between the regulated areas and the reference area. CPUE decreased in all areas over time. These results indicate that NTZs can be effective and provide rapid positive effects on coastal fish population, but that these positive effects can rapidly diminish if the regulations are removed.

CPUE in whitefish commercial landings increased significantly (2011-2020) in the coastal area covered by the spawning closure area and NTZs compared to the adjacent Åland Sea, where the reference area was situated during the study period, indicating an overall positive effect of this large (4000 km^2) spawning closure.

Overall, the results in this study indicate that the goal of regaining attractive whitefish fisheries in the southern Bothnian Sea has been, at least to some extent, fulfilled.

6.1. Background

European whitefish (*Coregonus maraena*) is one of the most important species for both the commercial and the recreational fisheries in the Bothnian Sea. However, commercial whitefish landings have been decreasing steadily and are now only about half of the landings in the mid 1990s (Swedish Agency for Marine and Water Management, 2018). The majority of the commercial fishery for whitefish in the area takes place during summer and autumn using gillnets and push-up traps. The recreational fishers in the Bothnian Sea mainly catch whitefish during spawning season in late autumn. The recreational fishery for whitefish is substantial, and landings are at least of the same magnitude as the commercial fishery. The majority of the recreational catches are classified as subsistence fishing, i.e. fishing carried out for personal consumption (Swedish Agency for Marine and Water Management 2016).

Due to the decreasing catches of whitefish, a NTZ was established in 2011, aimed to increase the populations of sea-spawning whitefish in the southern parts of the Bothnian Sea (Berkström et al. 2021). NTZs are used worldwide as a management tool to increase declining fish populations and restore degraded habitat (Gaines et al. 2010, Halpern et al. 2010). Before the establishment a number of meetings with stakeholders were held in 2010 - 2011 and most participants agreed that the decreasing catches of whitefish was worrying and that regulations were needed. However, they also believed that the consumption of whitefish by seals was a major problem for whitefish fisheries and were worried that the expected positive effects of the regulations may not be obtained due to high predation from grey seal and cormorants.

On the basis of the stakeholder meetings and a survey using gillnets and beach seines in 2010, the Swedish Board of Fisheries, together with local fishermen, suggested that a NTZ should be established at Storjungfrun-Kalvhararna to improve the situation for the sea-spawning whitefish. In addition, it was also suggested that a larger spawning closure along the coast of the county of Gävleborg and the northern parts of the county of Uppland should be established. In this spawning closure fishing for whitefish should be prohibited between 15 October and 30 November. The suggestions were accepted and on the 15th of October 2011 both the NTZ in the area Storjungfrun-Kalvhararna, with an extent of 147 km^2, and the spawning closure area, covering an area of 3980 km^2, were established (Figure 1).

Five years later, in 2016, the NTZ was reopened to fishing and converted into a spawning closure only, as it was concluded that the whitefish stock had increased and that a full fishing ban was no longer needed. The aim of the present study was

to evaluate both the effects of the 5-year NTZ and the effects of reopening the area to fishing on CPUE of fish.
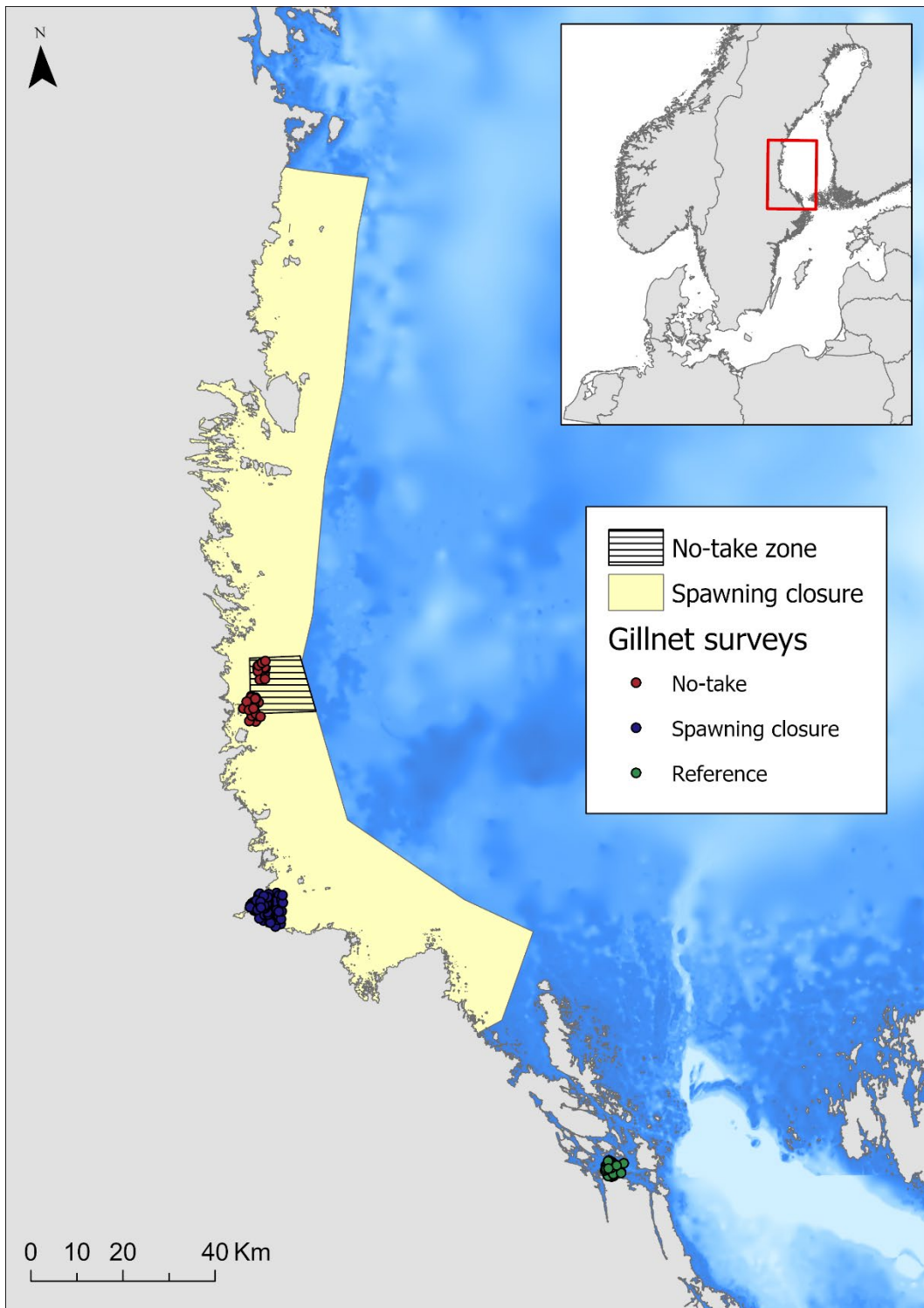
Figure 1. The geographical location of the no-take zone (NTZ) and the spawning closure in the Bothnian Sea for mainly whitefish protection, and the location of the gillnet surveys in the NTZ, spawning closure and reference area.

6.2. Methods

In 2011, a no-take zone (NTZ) of 147 square kilometers was established at Storjungfrun/Kalvhararna in the archipelago of Söderhamn, Bothnian Sea. At the same time, a larger area with fishing closure during the spawning period was established along the Coast of Gävleborg County and the northern parts of Uppland County. In the NTZ no fishing was allowed with an exception, that fishing with hand-held equipment from land was allowed during 1 June - 31 August. In the spawning closure area no fishing targeting whitefish was allowed during 15 October - 30 November and in addition, net fishing and other catching gear were not allowed during this period.

In order to monitor the effect of the NTZ on sea-spawning whitefish populations in the Bothnian Sea, gillnet surveys were carried out on a yearly basis from 2011 to 2021 with multimesh gillnets during the whitefish spawning season (Oct-Nov) to avoid inclusions of river spawning whitefish. The multimesh gillnets were 45 meter long and consisted of nine sections, with a mesh size ranging between 10-60 mm. The survey applied standardized methodology for coastal fish monitoring" (Swedish Agency for Marine and Water Management 2020) and was performed in 3 locations; 1) within the NTZ (Storjungfrun-Kalvhararna), 2) in the spawning closure area (Gävlebukten) and 3) in a reference area open to fishing (Galtfjärden, Figure 1). In each location 30 - 40 random stations were sampled using gillnets; 5 within the depth range 10-20 m and the rest equally distributed within the three depth strata 0-3, 3-6 and 6-10 m. The multimesh gillnets consisted of 9 sections, 5 m long, with a geometrical mesh size series from 10 to 60 mm. Each station was fished with one gillnet overnight, and depth, temperature and salinity were measured at each station. The whole catch was identified to species level and lengths were measured to the closest cm. Otoliths from all whitefish caught, except from the reference area in 2021, were collected and used for age determination.

During late spring 2011-2015 sampling for young-of-the-year whitefish was performed in potential spawning and nursery habitats in the NTZ and the spawning closure. Sampling for young-of-the-year whitefish was performed according to a standardized method (SLU ID: SLU.aqua.2016.5.4-4) using a beach seine with 6 m long arms and a mesh size of 5 mm and a cod end with 2 mm netting. The seine was set at a maximum depth of 2 meters and then hauled back to shore, covering an area of 60-120 sqm. The whole catch was identified to species level and lengths were measured.

Whitefish commercial landings in gillnet fishing for the southern part of the Bothnian Sea (SD 30), corresponding to the spawning closure area was compared

to the neighbouring Åland Sea (SD29). Data for the analysis was collected from logbook data provided by the Swedish Agency for Marine and Water Management. A linear regression analysis was used to evaluate the changes in CPUE and other indicators over time. In order to compare changes in CPUE over time between the NTZ, the spawning closure and the reference area a generalized linear model, using individual stations as replicates, was applied. Area was included as a fixed factor in the analyses, Year as a covariate and Area*Year as an interaction term to explore differences in the development over time between areas. Focusing on the interaction term, rather than on differences between areas and years, allowed testing the hypothesis that the difference in fishing pressure between the two areas would be evident as differences in the catch trajectories over the evaluation period. A negative binomial error distribution was applied, as this distribution was found to fit the zero-inflated data best. All analyses were performed in R, version 3.6.1 (R Core Team 2019) and the generalized linear model was fitted using the "glm.nb" function available via the MASS package (Venables, 2002). Some fishing stations were disturbed, e.g. by seals, and were removed from the statistical analyses (Table 1). Whitefish larger than 30 cm were considered mature and classified as adults and whitefish larger than 35 cm were classified as larger individuals. The mortality rates of the whitefish populations were analysed with an age-based catch curve analysis using the R-package FSA (Ogle et al, 2022).

Apart from analyses on whitefish, changes in CPUE over time between the NTZ and spawning closure was evaluated for sea trout. Catches of sea trout in the reference area were too sparse to be included in the analyses.

Predation on whitefish by grey seals and cormorants was estimated by combining abundance data with information on diet composition and bioenergetic requirments. For cormorants, nest count data from 2014 was used to estimate abundance (Ageheim & Lindqvist 2015, Hjertstrand 2015, Alf Sevastik unpublished data). It was assumed that cormorants were present in the area during 180 days per year, in line with the estimate in Hansson et al. (2017). Estimation of total fish consumption by cormorants were based on food requirements for breeding adults during breeding season (80 days) from Gremillet et al. (1995) and a total consumption of 500 g per day for breeding birds and fledglings outside the breeding season (100 days) as well as for non-breeding birds throughout the season (Keller & Visser 1999; Ridgway 2010). Each nest was assumed to represent two breeding adults, two fledglings and one non-breeding bird, and the average daily food intake was estimated to 1987 g per nest and day (Gremillet et al. 1995). It was assumed that cormorants forage within 20 km from their nests (Grémillet 1997; Nelson 2005), and a kernel density function with a 20 km radius was applied in ArcGIS to estimate cormorant densities across the study areas. Whitefish was not found in the diet of cormorants feeding

in the reference area, based on diet samples collected in Singöfjärden (approximately 8 km southeast of the reference area) 2013-2014 during the nesting period in May-June (n = 469, Ovegård et al. unpubl.). For cormorants feeding in the NTZ and the spawning closure area, the proportion in weight of whitefish in the diet was estimated to 1.75%, based on average estimates from diet samples collected in Lövstabukten April-August 2005 (n = 3002, Boström et al. 2012) and Gävlebukten in April 2014 (n = 34, SLU unpublished data) in the middle of the study area.

For grey seal, abundance data from 2010-2015 from the yearly national count from airplanes in May-June, carried out by the Swedish Museum of Natural History, was used. Seals were assumed to forage within a 60 km radius (Sjöberg & Ball 2000; Oksanen et al. 2014) and densities were estimated using a kernel density function, similar as for cormorants but with a search radius of 60 km. A daily total fish consumption of 4.75 kg per seal was applied (Hammond & Grellier 2006; Hammond & Harris 2006). It was assumed that predation by seals was absent during 120 days/year due to ice cover. The weight proportion of whitefish in the seal diet was estimated to 11%, based on prey remains in grey seal digestive tracts collected in the Bothnian Sea 2001-2012 (n=151, SLU unpublished data).

For the evaluation of the NTZ a set of Goals, Objectives, Indicators and Success criteria were defined for the whitefish population (table 2).

Table 1. Number of undisturbed sampling stations included in the analyses.

	Gillnet			Beach seine	
År	No-take	Spawning closure	Reference	No-take	Spawning closure
2007			30		
2008			28		
2009			29		
2010	28		30		
2011	41	41	30	26	
2012	40	40	30	26	27
2013	34	38	27	26	25
2014	40	38	30	25	25
2015	39	38	30	25	25
2016	40	31	30		
2017	35	40	30		
2018	36		30		
2019	40	40	30		
2020	40	41	30		
2021	38	41	29		

6.3. Results

Table 2. A summary of the results in relation to the goals and objectives set for the no-take zone. The table also refers to relevant figures or sections of the report. CPUE= Catch Per Unit Effort.

Goal	Objective	Indicator	Success criteria	Method	Results - during no-take (2011-2016)	Results - after reopening (2017-2021)	Reference
Strengthening the whitefish population in the No-take area	Decrease in mortality	CPUE, abundance	CPUE increases over time	Gillnet survey	Significant increase in CPUE of mature whitefish copmpared to both the reference area and spawning closure.	No significant trends or differences.	Fig. 2
	Decrease in mortality	Total mortality	Z-value deacrases over time or z-value is low	Gillnet survey	Not enough data for a temporal analysis.	Mortality has increased	Chapter 6.3.1
	Diverse size structure	Size structure	Increase in proportion of mature (>30 cm) and large (>35 cm) individuals	Gillnet survey	No significant trends.	No significant trends.	Chapter 6.3.1
	Diverse age structure	Age structure	Increase in proportion of older specimens	Gillnet survey	No significant trends.	No significant trends.	Fig 4
	Increase in recruitment	Young of the year production	Increased density of young of the year whitefish. Increased number of sites with spawning.	Beach seine	No indication of increased density or increased number of sites with spawning.	No further sampling after 2015	Fig.5
Strengthening of whitefish populations in the spawning closure in the southern Bothnian Sea	Increase in density	CPUE, abundance	CPUE increases over time	Gillnet survey	Significant increase in CPUE of mature whitefish copmpared to reference area.	No significant difference between spawning closure and reference area.	Fig. 2, 3
	Increase in density	CPUE, abundance	CPUE increases over time	Commercial fishing logbooks	The commercial fishery data shows a positive trend in the area of the spawning closure.	The commercial fishery data still shows a positive trend in the area of the spawning closure.	Fig. 7
	Decrease in mortality	Total mortality	Z-value decreases over time or z-value is low	Gillnet survey	Not enough data for a temporal analysis. Spatial comparison indicates low mortality in the no-take zone and a higher mortality in the spawning closure.	Total mortality in the former No-take area has increased. Still high mortality in the spawning closure.	Chapter 6.3.1
	Increase in recruitment	Young of the year production	Increase in density of young of the year whitefish. Increased number of sites with spawning.	Beach seine	No indication of increased density or increased number of sites with spawning.	No further sampling after 2015.	Fig. 5
Regain an attractive whitefish fishing in southern Bothnian Sea	Increase in landings	CPUE	CPUE increases over time	Commercial fishing logbooks	Data on commercial landings indicate a positive effect from the spawning closure.	Data on commercial landings indicate a continued positive effect of the spawning closure.	Fig. 7
		Average size	Average size increases over time	Gillnet survey	No significant trends.	No significant trends.	Chapter 6.3.1

6.3.1. Strengthening the population of whitefish in the southern Bothnian Sea

CPUE

During 2011-2016, a significant increase in CPUE of mature whitefish (>30cm) was found over time in both the NTZ (factor Area*Year, $F = 38.04$, $p = < 0.001$) and the spawning closure (factor Area*Year, $F = 12.09$, $p < 0.001$) compared to the reference area (Figure 2). There was also a significant increase in CPUE of mature whitefish between the NTZ and the spawning closure (factor Area*Year, $F = 7.09$, $p = 0.008$). By the end of the full fishing ban and just before the NTZ was reopened to fishing, the whitefish populations had increased by a factor of 9 in the NTZ and a factor of 2 in the spawning closure compared to the start of the regulations. At the same time, the whitefish population had decreased by a factor of 3 in the reference area. After the reopening of the NTZ, catches of whitefish were sparse and variable, both in the regulated areas and in the reference area. No significant trends in CPUE were found during this period (2017-2021) in the former NTZ (regression $r=0.40$, $p=0.48$), nor in the spawning closure ($r=0.20$, $p=0.74$) or reference area ($r=0.64$, $p=0.20$). Following the reopening of the NTZ, the development of CPUE between the regulated areas and the reference area differed with CPUE rapidly decreasing in the NTZ and spawning closure while CPUE slightly increased in the reference area. However, the last couple of years the CPUE was similar among all areas with generally low catches (Figure 2).
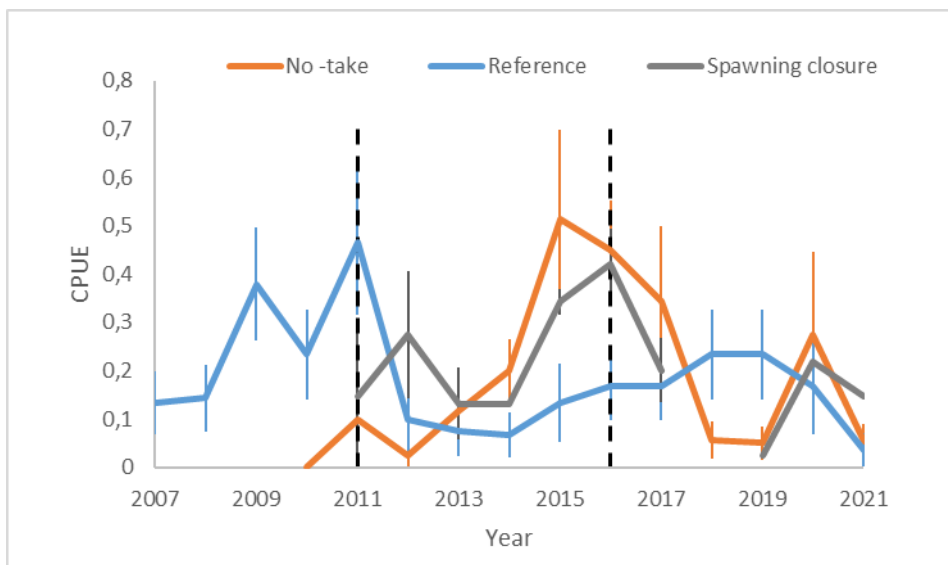
Figure 2. CPUE (mean±SE) of adult whitefish in gillnet surveys within the no-take zone, spawning closure and reference area during spawning season in October - November 2011-2021. Dashed lines show the period of the no-take zone.

Gillnet surveys at Långvindsfjärden (within the spawning closure area) and Gaviksfjärden (outside the regulated areas), from the regional environmental monitoring program in the Bothnian Sea, showed no significant trends in CPUE of adult whitefish prior to the establishment of the fishing regulations (2002-2010, Figure 3). During 2011-2016 there was a significant increase in CPUE in Gaviksfjärden (r=0.59 p<0.05). When including the last five years in the analyses after the NTZ had reopened, there were no significant trends in neither Långvindsfjärden, nor Gaviksfjärden. There was, however, a significant decrease over time in Lagnö in the Åland Sea close to the reference area (r=0.53, p<0.05) during the monitored period (2002-2021).
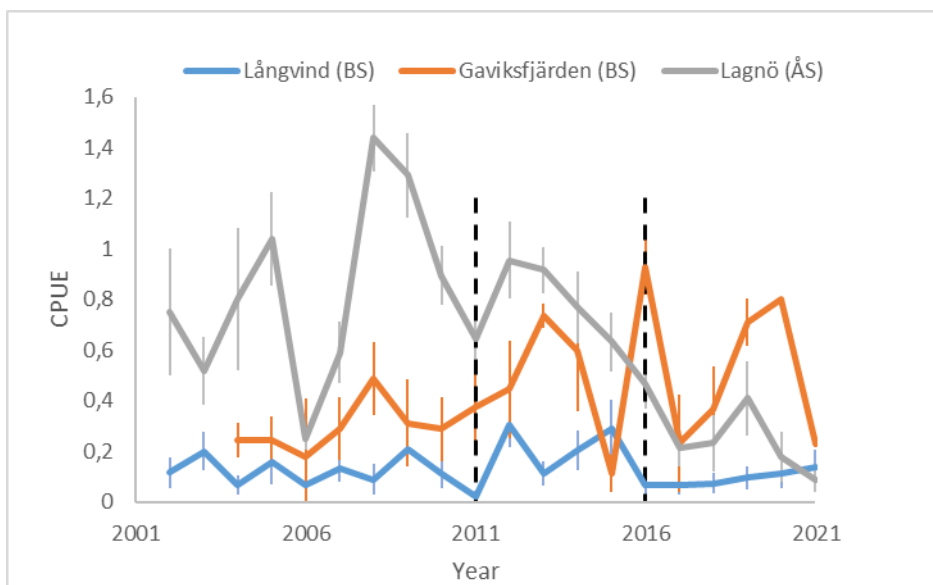
Figure 3. CPUE (mean±SE) of adult whitefish in gillnet surveys in the Swedish coastal fish monitoring program. Gaviksfjärden and Långvindsfjärden are situated in the Bothnian Sea (BS), within the coastal area covered by the spawning closure, while Lagnöfjärden is situated in the Åland Sea (ÅS). Dashed lines show period of the no-take zone.

Age and size distribution

Overall, older whitefish were more common in the reference area compared to the regulated areas and there were no clear changes over time in age distribution during the monitored period (2002-2021). There were no significant trends in the proportion of adult (>30 cm), large (>35 cm) or median length of whitefish larger than 30 cm. Neither were there any significant trends in older (>4 years) whitefish during this period. During 2017-2021, there were no significant trends over time in any of the measures based on age and size. However, the analysis on age and size are based on a limited number of whitefish, which can make it difficult to detect possible trends.
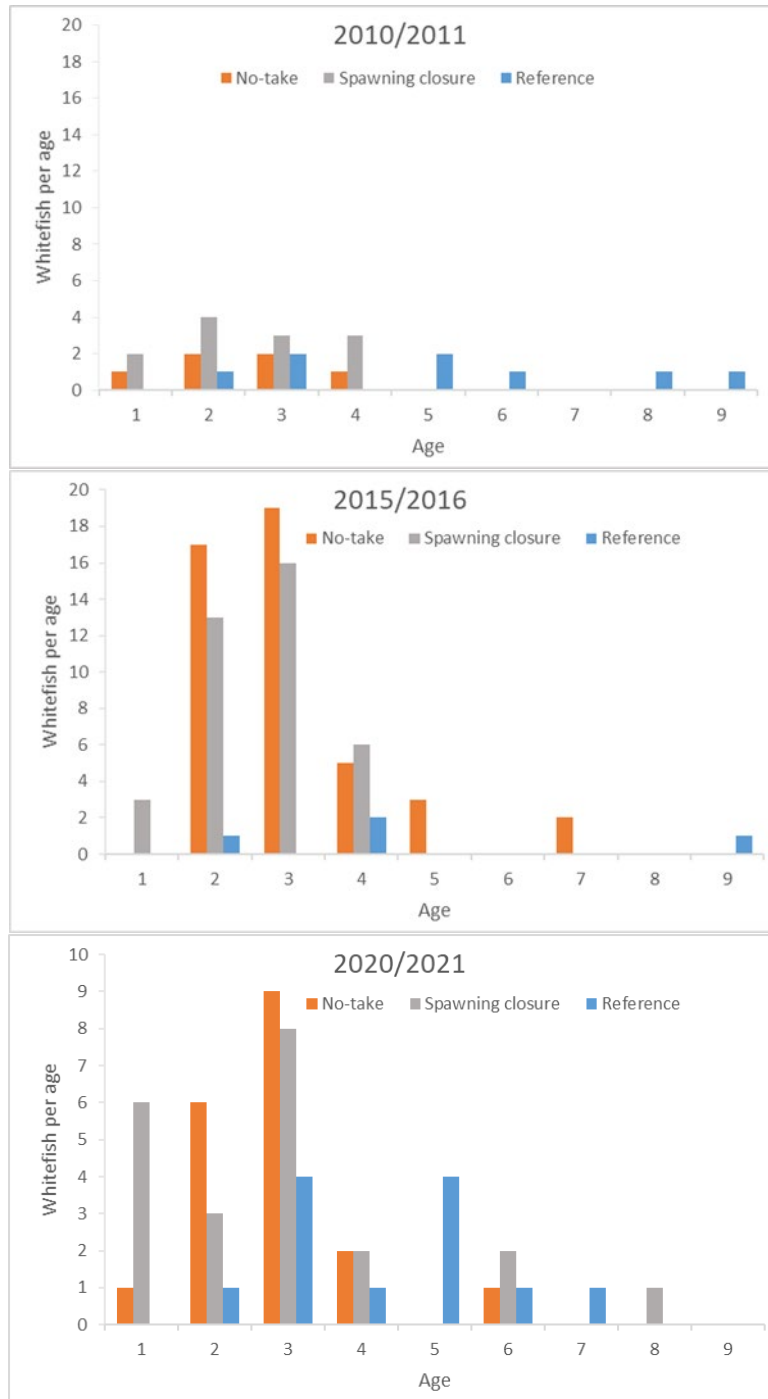
Figure 4. Total number of whitefish per age class in all gillnet surveys at the start of the regulations (2010-2011), during the full closure of the NTZ (2015-2016) and seasonal closure of the spawning area, and after reopening the NTZ to fishing, but closed during spawning (2020-2021).

Mortality

Since the numbers of whitefish caught in the gillnet surveys were so low, the data available for mortality analysis was limited and does not allow for analysis of yearly mortality. Instead, analyses of total mortality were made based on two time periods, more specifically 2011-2016 during the full no-take protection and 2017-2020 after

the reopening of the NTZ. Only whitefish between 2-5 years were included. In 2011-2016, the total mortality in the NTZ was estimated to be 0.13, slightly lower than in the reference area (0.17). Mortality in the spawning closure was considerably higher with a total mortality value of 0.6. During 2017-2020, the mortality in the spawning closure was still high, with a total mortality value of 0.72. In the former NTZ, the total mortality value increased to 0.26. The higher mortality in the NTZ in 2017-2020 compared to 2011-2016 during the time of full protection could indicate that the initial positive effect from the protection in the NTZ had faded when reopened. Unfortunately, the low number of samples may lead to uncertainties in the results.

Recruitment

Sampling of young-of-the-year whitefish during April and May in 2011-2015 indicated that there were suitable spawning and nursery habitats in both the NTZ and in the spawning closure. There was no significant trend over time in number of young-of-the-year whitefish. No further sampling of young of the year whitefish was conducted within the project.
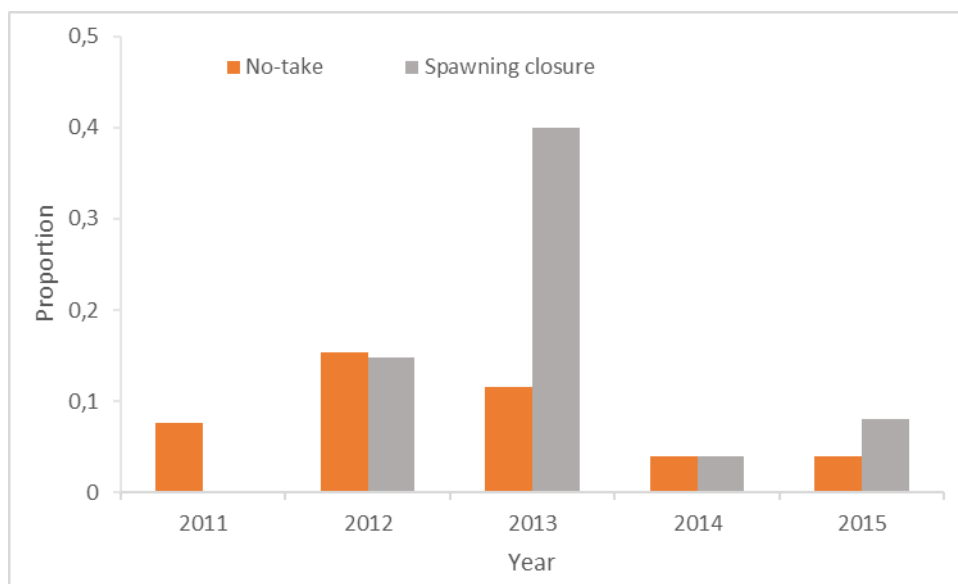
Figure 5. Proportion of sampling stations with young of the year whitefish in the no-take zone and the spawning closure. Note: no sampling was done in the spawning closure in 2011.
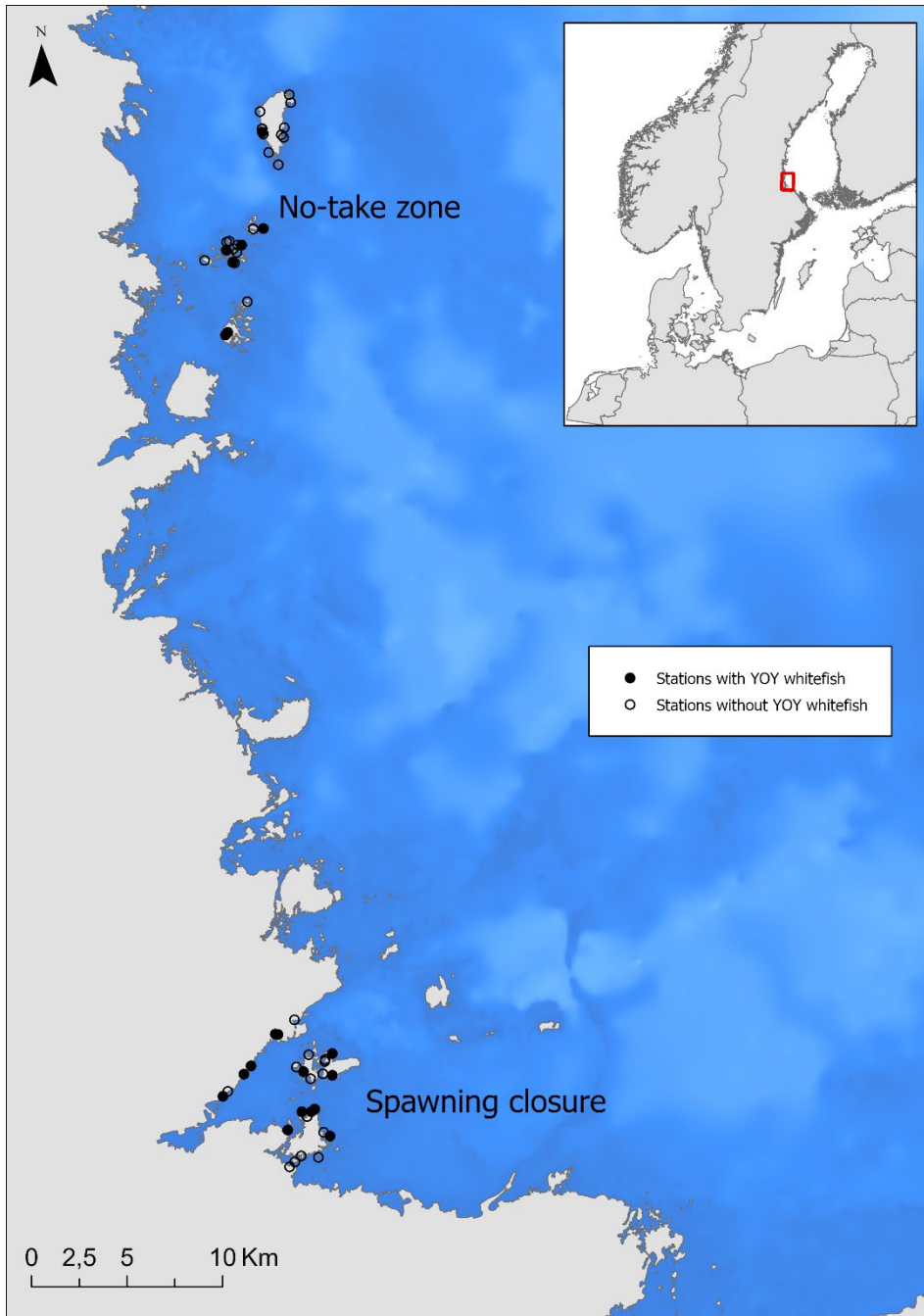
Figure 6. Stations sampled with beach seine 2011-2015 for young-of-the-year (YOY) whitefish. Stations with occurrence illustrated as black dots and stations without as hollow dots.

6.3.2. Relative impact of seals, cormorants and commercial fisheries

Disturbance by seals in gillnet surveys

Overall, the number of sampling stations affected by seals in the gillnet surveys was low and no obvious trend could be detected. The mean number of disturbed stations per year was 0.6 in the NTZ, 0.5 in the spawning closure and 0.2 in the reference area.

Removal of whitefish by seals, cormorants and commercial fisheries

Due to the long distance to breeding colonies, the kernel density function estimated no predation from cormorants in the NTZ. In the spawning closure and reference area, the estimated total fish consumption by cormorants was 9 and 6 kg per hectare and year respectively. Based on available diet data, cormorants extracted whitefish only from the spawning closure area (0.15 kg whitefish per hectare and year). The estimated total fish consumption by grey seals was 0.8 kg per hectare and year in the NTZ, 0.2 in the spawning closure area and 2.6 in the reference area. Based on diet data, the extraction of whitefish by grey seals was estimated to 0.1 kg per hectare and year in the NTZ, 0.03 in the spawning closure and 0.3 in the reference area. The total estimated extraction of whitefish from seals and cormorants (kg per hectare and year) was lower in the NTZ (0.1) compared to the spawning closure (0.18) and the reference area (0.3). In comparison, estimated extraction of whitefish from the commercial fisheries was lower than that of seals and cormorants. The commercial fisheries extracted 0.03 kg per hectare and year of whitefish in the NTZ, 0.09 in the spawning closure and 0.07 in the reference area. There were no data available to estimate the extraction of whitefish by recreational fisheries.

6.3.3. Regaining attractive whitefish fisheries in the southern Bothnian Sea

During the period 1999-2010, there was a significant decline in commercial landings of whitefish per gillnet and night, both in the Bothnian Sea close to the NTZ and spawning area and in the adjacent Åland Sea (r=0.95, p<0.001; r=0.61, p<0.05), where the reference area is situated. However, since the NTZ and spawning closure were established, commercial landings have increased significantly in the coastal area covered by the spawning closure compared to the Åland Sea (factor Area*Year, F = 9.77, p = 0.006) between 2011 and 2020.
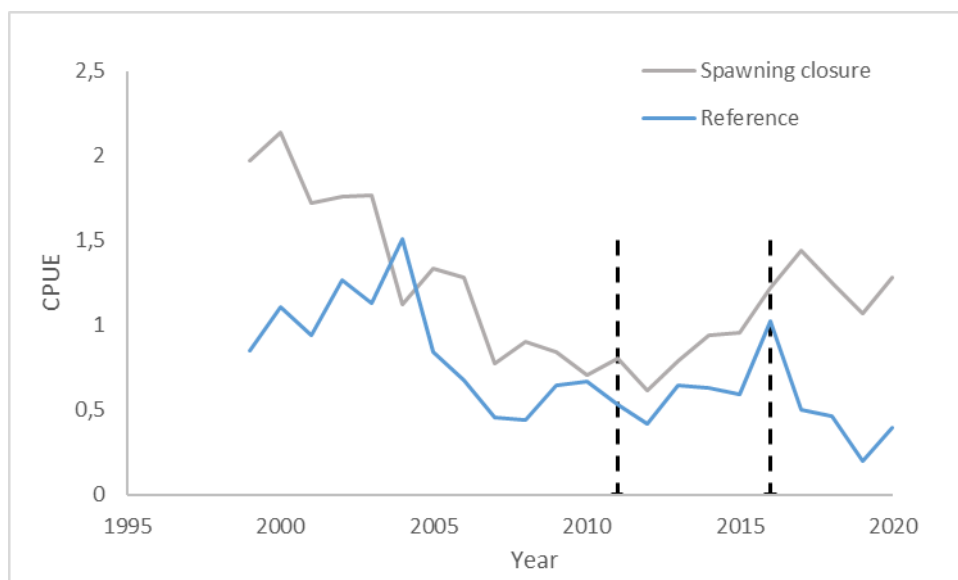
Figure 7. Commercial landings of whitefish in gillnets in the part of the Bothnian Sea (SD 30) that is covered by the spawning closure and the adjacent Åland Sea (SD29), where the reference area is situated. Dashed lines show period of the no-take zone.

6.3.4. Shift in commercial fishing effort

The temporal distribution of fishing effort and the main gears used in the commercial fishery for whitefish in the Bothnian Sea are the same both during and after the establishment of the NTZ. Approximately 50% of the total catches in the Bothnian Sea are caught with gillnets during May-October and 25% with push-up traps in June to July. During the period of the NTZ (2011-2016) the fishing effort in the Bothnian Sea with gillnets and push-up traps decreased with 23-39% compared to the period 2006-2010, before the establishment of the NTZ. During 2017-2020, when the NTZ was re-opened, the fishing effort continued to decrease another 35%. The spatial resolution of the catches reported from small-scale coastal fisheries is low and does not allow any analysis of potential spatial shifts in fishing efforts related to the protection level.

6.3.5. Effect on other species

In addition to the analysis on the target species whitefish, the potential effect of the NTZ and spawning closure on sea trout (*Salmo trutta*) was evaluated (figure 8). During 2011-2016, a significant increase in CPUE in sea trout was found in the NTZ (regression r=0.87, $p < 0.05$), however, there was no significant difference in CPUE over time between the NTZ and the spawning closure. During 2017-2021, there was a significant increase in CPUE over time for trout between the former NTZ reopened for fishing (factor Area*Year, F = 6.22, $p < 0.05$) compared to the spawning closure. Catches of trout in the reference area were too small to be included in the analysis.

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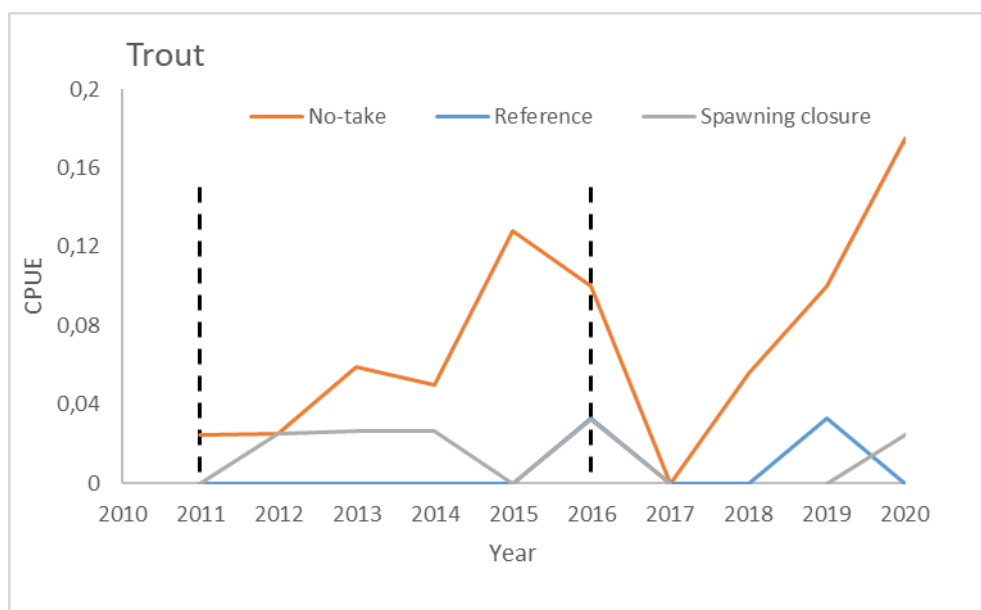
Figure 8. Mean CPUE of trout in gillnet surveys within the no-take zone and spawning closure during October – November 2011-2021. Dashed lines show period of the no-take zone.

6.4. Discussion

A significant increase in adult (<30 cm) whitefish was found in both the NTZ and the spawning closure compared to the reference area in 2011-2016. Further, the increase in CPUE of adult whitefish was significantly higher in the NTZ than in the spawning closure. These results were also reported in Berkström et al. (2021). A similar pattern was seen for sea trout in the current report, indicating that a stricter protection is a more effective measure in strengthening coastal fish populations.

Since the main goal of the NTZ had been achieved by 2016, i.e. a strengthening of the whitefish population, the responsible authority came to the conclusion that it was not legally feasible to maintain the NTZ. The area was therefore reopened to fishing, apart from during the spawning season, i.e. the area had the same level of protection as the surrounding spawning closure. The gillnet surveys continued in the former NTZ and in the spawning closure and reference area, in order to study the long-term effects of the regulations and what would happen when reopening the NTZ to fishing.

The socio-economic effects of the temporary closure were estimated separately in cost-benefit models prior to reopening the NTZ and showed that the net benefits of the NTZ would be positive even after reopening to fishing for most scenarios tested (Bostedt et al. 2020). During the five-year period following the reopening of the NTZ closure, there were no significant differences in the development of CPUE of

adult whitefish over time between the areas, but CPUE declined in all areas over time and the positive effect of the NTZ was lost. Neither were there any significant differences in any of the studied indicators based on age and size. The CPUEs of pikeperch and pike were also found to decline after reopening the NTZ in Gålö, south of Bottenhavet, despite that protection during spawning was still in effect (chapter 7, this report). In both NTZs, CPUEs a few years after reopening to fishing returned to levels similar to those before the NTZs were established. These results indicate that NTZs can be effective and give rapid positive effects on coastal fish populations, but that these positive effects can disappear quickly if the regulations are removed. A year-round fishing ban hence seems necessary to protect populations in areas where fishing pressure is relatively high.

Evaluations of areas reopened to fishing are rare, however, rapid declines in abundance and biomass after reopening areas closed to fishing have also been found for red abalone in California (Rogers-Bennett et al. 2013) and large predatory fish in the Philippines (Russ and Alcala 2003). For red abalone, the size structure was also found to change and egg production drastically declined only three years after opening up the marine reserve to fishing (Rogers-Bennett et al. 2013). In the Philippines, the density and biomass of predatory fish increased by 70-95% when the area was closed to fishing and when reopened decreased with the same magnitude, resulting in similar low densities and biomasses as before closure within 2-5 years (Russ and Alcala 2003).

The analysis on consumption of whitefish by cormorants and grey seal in relation to the catches from the fishery in the present study, show that these predators could potentially have an effect on the whitefish populations. Before the establishment of the NTZ, a number of meetings with stakeholders were held in 2010 - 2011 and they expressed concern that the consumption of whitefish by seals was a major problem for whitefish fisheries and were worried that the expected positive effects of the regulations may not be obtained due to high predation by these predators. Estimated predation on whitefish by cormorants and grey seal was higher in the reference area than the NTZ and the spawning closure and may explain some of the differences in CPUE of adult whitefish among areas. However, due to uncertainties in the abundance and diet data, the predation estimates are rather coarse. Extended monitoring is needed in order to get better data on the spatial distribution of feeding areas of these predators, while more diet data is needed to improve the precision of prey composition estimates. In the Gålö area, where more recent data on grey seal and cormorant seal is available, the declines are likely explained by a combination of a high predation pressure by grey seal and cormorant and from reopening the area to recreational fishing (chapter 7, this report).

In Berkström et al (2021), it was shown that the whitefish CPUE in the commercial fishery significantly increased during 2011-2016 in the coastal area covered by the spawning closure compared to the adjacent Åland Sea, where the reference area is situated. In this study, we show that this trend continues until 2020, with an increase in the commercial gillnet catches over time in the same part of the Bothnian Sea as the spawning closure, compared to the Åland Sea. These results indicate that the goal of regaining attractive whitefish fisheries in the southern Bothnian Sea has been at least to some extent fulfilled. However, the declines of adult whitefish to levels similar to that before the establishment of the NTZ after reopening the area to fishing are worrying, and suggest that partial protection, i.e. during spawning season, may only be enough in areas where fishing is concentrated to the spawning season.

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7. No-take zone for pikeperch, pike and perch in the Stockholm Archipelago (Gålö), Baltic Sea

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Photo: Emilia Bergström

Summary

A no-take zone (NTZ) where all fishing was prohibited was established in 2010 in the southern parts of the Stockholm archipelago in the Gålö area. The main purpose of the fishing regulations was to protect the resident population of pikeperch, but also the local populations of pike and perch. The NTZ covered an inner archipelago bay with an area of 1.7 km^2. A buffer zone of the same size, with a spawning closure only, was located outside the NTZ. The location is one of the most important spawning- and nursery grounds for these three species in the Stockholm archipelago. A substantial recreational fishery is present in the area, but no commercial fishery.

To evaluate the development of the fish communities over time, annual fishing surveys were conducted using trapnets, a type of large fish trap, during the spawning season (May-June), and multimesh monitoring gillnets were used during late summer (August). These two quantitative fishing surveys were conducted in the NTZ and in an adjacent reference area.

The results show that the NTZ had a positive effect on the spawning stock of pikeperch and pike during the closed period of five years; the catches of adult fish increased over time by a factor of 5-11 compared to the reference area Askviken, where recreational fishing was unregulated. Moreover, for pikeperch, there was an indication that abundances of juvenile individuals increased, which contributed to the stability of the community size- and age structure, even though there was an increase in abundance of adult individuals. For pike, catches increased by a factor of 4-6 until 2014. Thereafter catches dropped again, following disturbance from grey seal in the area. The low and variable catches made it difficult to evaluate effects on size structure of pike. For perch, there were no positive effects on densities or size structure, in spite of the previously high fishing pressure by recreational fishers in this area. A high predation pressure by cormorants, overriding the effects of the closure, is a probable explanation. Estimations of removal of perch by cormorants, based on bird inventories and diet data, showed that predation pressure was higher in the NTZ compared to the reference area, potentially because of disturbance by boat traffic in the latter.

Because the communities of both pikeperch and pike were considered to have recovered, the total fishing ban was removed in 2016, with only a temporal fishing ban during the spawning period remaining. A Before-After Control-Impact (BACI) follow-up study was conducted in 2020 and 2021, when fishing surveys using trapnets during spring were conducted in the spawning closure and the reference area. Adult pikeperch had decreased significantly in both areas since the removal of the total fishing ban, but this decrease was less pronounced in the spawning

closure compared to the reference area. For pike, abundances decreased significantly in both areas after 2015, and there was no difference between the areas. Perch showed a similar trend as pikeperch, with abundances decreasing significantly in the spawning closure after the NTZ was lifted, whereas there was no clear trend in the reference area before and after 2015.

In summary, the study shows that a NTZ of a limited size can generate positive effects on local abundances of pikeperch and pike already after a few years. This may be applicable to perch as well, as long as predation by seals and birds does not limit population sizes significantly. Based on the results from the present study, smaller NTZs may constitute an important tool for management and conservation of local populations of coastal species w the Baltic Sea.

The negative trends observed after the opening of the NTZ in 2015, indicate that a temporal fishing ban was not sufficient for maintaining the population levels that were reached during the fully closed period. This may be especially true in areas that harbour high densities of seals and cormorants, which together with fishing exert a high pressure on fish populations.

7.1. Background

The coastal areas around the Gålö peninsula in the inner part of the Stockholm Archipelago is famous among recreational fishers due to the rich abundances of pikeperch (*Sander lucioperca*), pike (*Esox lucius*) and perch (*Perca fluviatilis*). Especially the connected bays Lännåkersviken and Blista fjärd constitute an important spawning ground for these three species. Regarding pikeperch, this is one of the most important reproduction areas in the southern Stockholm archipelago (Gunnartz et al., 2011; Bergström et al., 2013). This location has also been one of the most frequently visited areas by recreational fishers in the archipelago. For example, the Stockholms Sportfiskeklubb, who leased these fishing waters until the open access reform in 1985 have had active sport fishing activities here from the 1930s to 2000s (SLU Aqua, unpubl.). Fishing guides and many other sport fishers also frequent the area. The target species within the fishery are primarily pikeperch, pike and perch. Fishing right owners have practised net fishing targeting pikeperch and perch, but there is no active commercial fishery, nor has there been, in the area. An overview of catches from a larger archipelago area where Gålö is included indicates that the fishing pressure on predatory fishes was relatively high already during the 1990s (Svedäng et al., 1998).

By the end of the 2000s, there were several reports from recreational fishers about decreasing catches of pikeperch and pike. The status of pikeperch and pike along

the coast in the northern Baltic Sea was is generally poor (Mustamäki et al., 2013; SLU Aqua, 2015). The decrease in fish stocks around Gålö were thought to be caused by an intense recreational fishing pressure in combination with low recruitment and potential increased predation by the great cormorant (*Phalacrocorax carbo sinensis*) and grey seal (*Halichoerus grypus*). Fish surveys in 2009 using multimesh gillnets and, for young-of-the-year fish (YOY), small detonations, strengthened previously made assumptions that Lännåkersviken and Blista fjärd in Gålö constitute both an important spawning and nursery ground for pikeperch, pike and perch, and a key area for adult pikeperch (Bergström, 2009). Although the population size structure of perch was stable in the area, the size structure for pikeperch and pike were skewed with a significantly lower proportion of adult fish than could be expected for a healthy fish community. Based upon the YOY surveys in 2009, the recruitment in the area was considered sufficient, why impaired recruitment was not considered to be a likely cause of the skewed size structure (Bergström, 2009). The abundances of juvenile fish were high, and the fish populations were considered to have a good potential for recovery during a five year fishing ban. Fish surveys conducted in the bay Askviken on the eastern side of the Gålö peninsula showed that this site had similar environmental conditions as Lännåkersviken-Blista fjärd and was therefore used as a reference site (Fig. 1.).

The Gålö area is considered a representative site for the Swedish Baltic coast – an inner archipelago environment where the majority of the fishery is conducted by handheld gears. Pikeperch, pike and perch are among the most important species for this type of coastal fishery. These species have highly localized populations, which means that even rather small-sized protected areas can generate positive effects on the local fish stocks (Laikre et al., 2005; Bergström et al., 2007; Dannewitz et al., 2010). Pike rarely moves over longer distances than 5 km, perch 10 km and pikeperch seldom more than 15 km (Saulamo & Neuman, 2002), which sets the spatial scales within which potential effects of the closed areas can be expected.

Prior to the implementation of the NTZ in Gålö, a number of meetings were held with different stakeholder groups to discuss the potential consequences of closing an area to fishing. During the end of 2009, there were meetings jointly organised by the Swedish Board of Fisheries and the County Administrative Board of Stockholm involving organisations representing (among others) fishing rights owners, commercial fishers, sport fishers, fishing guides, and nature conservation organisations. The reactions to the proposed fishing regulations were mainly positive, although the commercial fishers conveyed that they were generally against total fishing bans. In addition to these meetings with organisations, an open meeting was organised where approximately 40 additional stakeholders joined, mainly

fishing rights owners and recreational fishers. The participants agreed that the decrease of pikeperch and pike in the area was worrying. Also, there were concerns regarding the impact of cormorant predation on the local fish populations. Overall, there were few objections against the proposed five year fishing ban, although many stakeholders were worried that the fishing ban would be transformed into a permanent one.

Based on information derived from fish surveys and data analyses conducted during 2009 and different stakeholder workshops, the Swedish Board of Fisheries decided to implement a no-take zone in Gålö, with the following motivation:

"The proposed no-take zone is a very limited area in the archipelago, which constitutes a key spawning and nursery area for the local pikeperch population. The high abundance of pikeperch below the minimum allowed catch size, i.e. subadult individuals, in the area contributes to favourable conditions for a fast recovery of the population. Other management strategies/options, such as an increase in minimum allowed catch size or restricted gear use area are assumed to be insufficient within the core area, considering the low abundances of adult fish. The current fishery is mainly conducted using gear that could accidentally catch pikeperch or pike. The proposed fishing regulations should include all types of fishing. A recovery of the populations of pikeperch and pike is considered to have effects on a larger spatial scale, because the resident fish populations are expected to increase and disperse to adjacent areas, and the core area will function as a spawning area for fish in a larger area than the no-take zone only. The proposed regulations will also contribute to increased knowledge of management strategies involving the use of no-take zones. The Swedish Board of Fisheries has the right to restrict the fishing on privately owned fishing waters following the same procedures as in common waters, given that there are reasons for conservation measures. The Swedish Board of Fisheries considers that there the proposed fishing regulations are necessary to restore the local fish populations and that no monetary compensation to the fishing right owners will thus be provided. From a socio-economic perspective, the largest values with the proposed fishing regulations are the contribution to the recovery of the local pikeperch population as well as increase the local biodiversity. The socio-economic value of recreational fishing is generally large, and recovery of the local fish stocks has the potential to generate substantial socio-economic values in the area."

The fishing regulations were implemented on the 1st of February 2010. Lännåkersviken and parts of Blista fjärd, an area of 1.7 km^2, was closed to all fishing during 2010-2015. Outside the no-take zone, a buffer zone where a temporal fishing ban during the spawning season was implemented, from the 1st of April – 15th of June. Substantial discussions with the concerned fishing right owners and a

property survey were conducted in order to get approval for the scientific follow-up study (necessary according the Swedish law when a scientific fish survey is conducted on privately owned fishing waters). Even though the NTZ was rather small, 150 different real estates were concerned. This illustrates the complexity of the juridical situation that concerns fishing regulations and scientific studies along the Swedish coastal stretch between the counties of Uppsala and Blekinge.

On the 1st of July 2015, the NTZ was replaced with a spawning closure only, from 1st of April- 15th of June. Since then, both the previous NTZ and the buffer zone are open to fishing except during the spawning period. The total fishing ban was abolished because the recovery of the focal species was considered successful and there was therefore no need for a total closure from a fisheries management perspective.
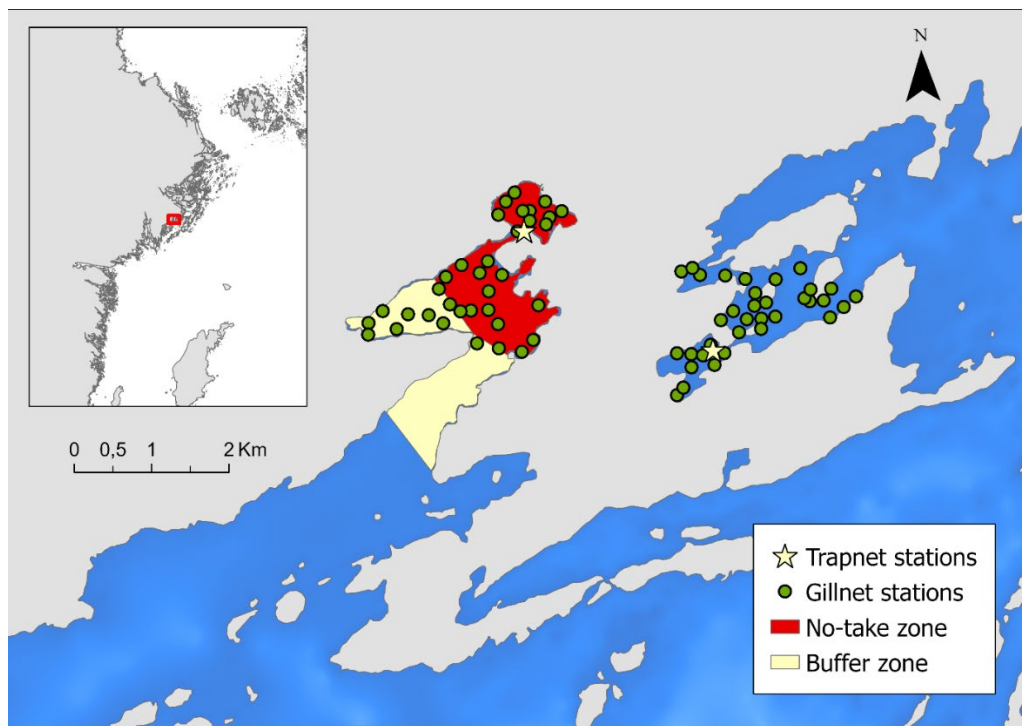
Figure 1. The no-take zone in Lännåkersviken and Blista fjärd and the adjacent buffer zone located in the western parts of the Gålö peninsula in the Stockholm archipelago, and the reference area Askviken on the eastern side. The fish monitoring stations are indicated by circles. The inserted map shows the location of Gålö on the Swedish Baltic coast.

7.2. Methods

7.2.1. Definition of goals

During the implementation of the NTZ, goals, objectives, indicators and success criteria (GOIS) were defined for the local populations of pikeperch, pike and perch (Table 2). A monitoring programme was designed based on these objectives and indicators, aiming to assess whether the goals of the fishing regulations were met.

7.2.2. Fish monitoring

Two different quantitative survey methods were used; eel trapnets and multimesh gillnets. These were used during the yearly follow-up survey of the fish populations in Lännåkersviken and the control area Askviken (Fig. 1). The trapnet survey was conducted in April-June during 2010-2015 (total fishing ban) and during 2016, 2020 and 2021 (spawning closure only). Trapnets are normally used for catching eel, but are efficient for many species. They consist of a main lead of 45 m and two wing leads, each 10 m long. All leads have a height of 4 m and 18 mm mesh size. The trapnet was emptied every second to third day, and all caught fish were identified to species level, counted and length measured before being released. The exact time of the survey varied among years depending on ice cover. To achieve as comparable data as possible among years, only samples from May-June were included in the statistical analyses that investigated catch over time.

An annual survey using multimesh gillnets was conducted in August during the total fishing ban period (2010-2015) following a standardised protocol (Havs- och vattenmyndigheten 2020). Each gillnet was 45 m long and was composed of nine different sections with mesh sizes ranging between 10-60 mm. Sampling was conducted at 36 stations in each of the two sites (Lännåkersviken and Askviken) at a maximum depth of 10 m. Every station was fished using one net during one night. All fish were identified to species and length was measured. In the analyses, only samples from undisturbed stations were included (Table 1). Disturbances rarely occurred, and when they did, they mainly consisted of seal damaged catch or nets clogged with drifting vegetation. On average, one station per year and area was excluded due to disturbances.

The aim of the gillnet survey was to monitor the population status of the focus species (pikeperch, pike and perch) and the local fish communities over time. The survey also collected individuals for analyses of sex, gonadal status and age. Aging from otoliths was conducted for pikeperch (all years) and perch (2009 and 2013), but not for pike, due to low catch rates. To investigate potential changes in ecosystem effects, such as alterations in food web structure over the years in the NTZ and reference area, all fishes were categorised into functional groups;

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cyprinids, non-piscivores and piscivores, according to classifications in Bergström et al., 2019.

Table 1. The number of fishing days for fishing surveys using fish traps (May-June) and the number of stations sampled using gillnets for standardised monitoring (August) in the no-take zone (Lännåkersviken) and in the reference area (Askviken).

	Trapnet survey		Gillnet survey	
Year	Lännåkersviken	Askviken	Lännåkersviken	Askviken
2010	42	43	37	30
2011	35	36	36	37
2012	32	32	36	37
2013	45	46	36	36
2014	44	42	35	35
2015	43	44	36	36
2016	35	36		
2020	37	37		
2021	42	42		

7.2.3. Tagging study

In addition to the sampling described above, a tagging study of pikeperch and pike was conducted using individuals from the fish trap. The aims of the tagging study were to 1) obtain information about the recreational fishery on these species, 2) collect information about the distribution and dispersal patterns of the concerned fish stocks by recapture reports of tagged individuals by recreational fishers outside the protected area, and 3) estimate the size of the fish stocks, using the recaptured individuals in the fish traps. All individuals of pikeperch and pike that were caught during the spring fishing surveys 2010-2015 (except for 2013), were tagged with a t-bar anchor tag in the dorsal fin (for pikeperch, in the second dorsal fin). Catch reports of tagged individuals included recapture date, site, length and injuries (if any).

7.2.4. Environmental variables

Water temperature and visibility were noted for each sampling day, both for gillnets and trapnets. During 2009-2014, temperature loggers were deployed in both the NTZ Lännåkersviken and the reference site Askviken to monitor the seasonal temperature. These loggers were used for the majority of the ice-free period. Both visibility and temperature can have large effects on fish communities, and this

information was used to investigate potential differences or changes over time between the two sites.

7.2.5. Predation by cormorants and seals

An estimation of the removal of fish by cormorant and grey seal in the two sites was conducted to evaluate the potential effects on the focal species. Both cormorants and seals occur in the area, sometimes in high numbers. These estimations were based on a combination of count data (derived from the national monitoring program of grey seal and from inventories of cormorant in the study sites during the fishing surveys) and calculations of fish removal per seal and cormorant. By combining count and diet composition data with information on foraging behaviour and fish consumption based on bioenergetic models, the total consumption per fish species and year in the two sites was assessed.

The fish consumption by cormorants was calculated by combining bird counts from the study area that were conducted in association with the trapnet fishing, during April-June 2015, 2016, 2020 and 2021. Diet data was collected in 2012-2014, based on regurgitated prey that was collected during the nesting season from bird colonies located within normal foraging distance of the sites, i.e., < 20 km. Consumption was estimated by multiplying the observed number of cormorants (standardised per ha) with the estimated energetic need per individual and number of feeding days per year. The consumption per individual was set to 500 g per day and the total number of feeding days per year was set to 185 days (Gremillet et al., 1995; Hansson et al., 2018).

For grey seal, data on abundances from the national seal monitoring programme (by The Swedish Museum of Natural History) were used. The data consisted of seal counts at haul-out sites during the moulting period (May-June). To compensate for the numbers of grey seals that were in water during the surveys, it was estimated that 70 percent of the population was included in the actual counts (Hiby et al., 2007). The approximate density of grey seal in the NTZ and the reference site were estimated using an interpolation analysis in ArcGIS by a kernel density function. The maximum distance was set to 60 km, which corresponds to normal foraging distance of grey seals (Sjöberg & Ball, 2000; Oksanen et al., 2014). This interpolation method builds on the notion that the highest numbers of seals are found close to the haul-out sites and decrease isotropically with increased distances. The total fish consumption of grey seal in the area was estimated by combining the seal density estimates from the interpolated maps with an estimated total consumption of 4.75 kg fish per day (Hammond & Grellier, 2006; Hammond & Harris, 2006). It was estimated that seals were present in the area for 305 days per

year and were prevented from foraging in the area during two months because of extensive ice cover.

7.2.6. Young-of-the-year sampling

To estimate recruitment of fish in the area a survey targeting YOY fish was conducted. The sampling was done using a standardized method of small underwater detonations (Bergström et al. 2021) in the area in 2009, prior to the implementation of the NTZ, and in 2012. All fish were identified to species and length was measured.

7.2.7. Statistical analyses

Evaluation of the no-take zone

Prior to the analyses, all fish < 12 cm were removed from both the eel trapnets and the gillnet survey, to exclude variation due to YOY individuals. To test for potential differences in the development of fish populations between the NTZ and the reference area, an analysis of covariance (ANCOVA) was used, which included 'site' as a factor, 'year' as a covariate and an interaction between 'site' and 'year'. The interaction between site and year is the key aspect of the analysis in this study; a statistically significant ($p < 0.05$) interaction term indicates that the there is a difference between the NTZ and the reference area, and thus that the fishing ban has had an impact. To analyse potential differences in the size and age structure between the two sites, an ANCOVA testing for differences in the development of the proportion of large individuals (> 40 cm for pikeperch and pike, > 20 cm for perch) was applied, based on the data from the gillnet surveys in August. Using a similar setup, differences in the proportion of predatory fish in gillnet catches between the two areas was also analysed by ANCOVA.

Mortality rates were estimated using catch curve analysis in R, using the packages FSA (Ogle et al., 2021) and NCStats (Ogle, 2021). All pikeperch caught in the multimesh gillnet survey, which captures fish from the full size spectrum, were aged. For perch, only a subsample of the fish caught in the gillnet survey were aged. To be able to conduct the mortality analysis for the total catch, the subsample was upscaled to the total catch by the use of a length-age key that estimated age from length distribution. For pikeperch, the years 2010-2012 were tested against 2013-2015, and for perch the year 2009 was tested against 2013. Because there was insufficient data to analyse cohorts or to compare the mortality prior and post the implementation of the MPA, we used these two time periods to evaluate potential differences between the two treatments.

To test for differences in mean trophic value in the NTZ and the reference area, an ANCOVA analysis was conducted. The mean trophic value was calculated as a weighted mean within each trophic level (see HELCOM, 2012). Information about each trophic level was derived from Fishbase (www.fishbase.org, Froese and Pauly 2004).

Differences in species composition were analysed using an index of similarities (Bray-Curtis similarity index) and was evaluated with a multivariate Principal Coordinates Ordination (PCO) analysis. The aim of the PCO analysis was to identify potential differences in species composition between sites and years, and identify which species contributed to these differences. The analysis was conducted using PRIMER 6.0.

To estimate the effect size of the NTZ in terms of catch development during the closed time period (2010-2015), linear regression lines were fitted to the data on large individuals of the main target species of fisheries in the area, i.e. pikeperch and pike. The effect of the NTZ was estimated as the relative difference in development between the two areas.

Evaluation of reopening of the area to fishing

To evaluate effects on the focal fish species (pikeperch, pike and perch) after 2015 when the total fishing ban was replaced with a spawning closure, a Before-After Control-Impact (BACI) design was used, applying a generalized mixed model (GLMM) with negative binomial distribution (to fit the assumptions of a linear model) from the package lme4 (Bates et al., 2015). Catch per unit effort (CPUE) of each target species (only adults; pikeperch > 40 cm, pike > 40 cm, perch > 20 cm) (derived from the trapnet data) was used as response variable. An additional analysis was conducted for roach, due to an observed decrease in CPUE of this species during the fishing closure. All individuals > 12 cm were included in the analysis. For all models, 'site' and 'time period' (before/after) were included as fixed factors. 'Year' was included as a random factor and nested under 'time period'. Similarly to the ANCOVA, it is the interaction of 'site' and 'time period' that determines if there is a significant effect of the treatment; i.e. a BACI effect (Pardini et al., 2018; Fisher et al., 2019). All models were evaluated using ANOVA type 2 tests in the car package. Type 2 tests account for potential confounding effects by removing the variation explained by confounding variables. In addition to the BACI analyses, a before-after (BA) analysis was conducted, using data from the NTZ only. The aim of this analysis was to exclude potential site-dependent variation, because there was only one reference area included in the study. This was achieved by fitting a mixed linear model from the package lme4 (Bates et al., 2015) for CPUE of each of the focal fish species. The models included 'time period' and

'year', with 'time period' as a fixed factor and 'year' as a random factor nested under 'time period'. In the analysis for pike, time period consisted of the years 2013 and 2014 (2015 was omitted from this analysis because of major disturbances by seals that targeted pike specifically) and the years 2020 and 2021 as "after re-opening".   For pikeperch and perch, 2014 and 2015 was defined as the closed period and 2020 and 2021 as the open period. All analyses were conducted in R version 4.0.1 (R Core Team 2020)

7.3. Results

Table 2 summarises the results of the study in relation to the goals and objectives that were defined for the NTZ in Gålö.

Table 2. Overview of the results in relation to the goals and objectives of the no-take zone in Gålö, which was in place in 2010-2015. The table also includes references to figures and tables where results are presented.

Goals	Objectives	Indicators	Success criteria	Surveying methods	Results	Ref.
Recovery of the pikeperch population in the Gålö area	Increase in spawning stock size in the NTZ	CPUE spawners	CPUE of spawning individuals increases over time	Trapnet	Increase in spawning stock in the NTZ compared to reference area	Fig. 2
	Increase in stock size in the NTZ	CPUE	CPUE increases over time	Multimesh gillnet	A trend towards increased densities of small and large pikeperch in the NTZ compared to the reference area, although not statistically significant. Analysis of biomass shows an increase of large pikeperch in the NTZ compared to the reference area	Fig. 3

	Decrease in fishing mortality	Total mortality	Z decreases over time	Multimesh gillnet	Few older individuals of pikeperch in the reference area does not allow for a comparison between the sites. No change in mortality over time in the NTZ, but catches of older individuals were too few to give reliable estimates.	Section 7.1.3
	A diversified age/size structure	Size structure	The proportion of large individuals increases over time	Multimesh gillnet	A tendency towards an increase in the proportion of large pikeperch in the NTZ compared to the reference area, but not statistically significant as small pikeperch also increase in the area	Section 7.1.2
	Increase in recruitment	Production of young-of-the-year	Increased densities of young-of-the-year pikeperch	Young-of-the-year survey	No signs of impaired recruitment in the area. No time-series data of the production of young-of-the-year	Section 7.1.4.
Recovery of the pike population in the Gålö area	Increase in spawning stock size in the NTZ	CPUE spawners	CPUE for spawning individuals increases over time	Trapnet	Increase in the spawning stock of pike within the NTZ compared to the reference area 2010-2014. A decrease in 2015 due to disturbance from seals	Fig. 2
	Increase in stock size in the NTZ	CPUE	CPUE increases over time	Multimesh gillnet	A trend towards increased densities of large pike in the NTZ compared to the reference area	Fig. 3
	Decrease in fishing mortality	Total mortality	Z decreases over time	Multimesh gillnet	Not evaluated because of no available age data	

	A diversified age/ size structure	Size structure	Increase in the proportion of large individuals	Multimesh gillnet	No change in the proportion of large pike in the NTZ or reference area	Section 7.1.2
	Increase in recruitment	Production of young-of-the-year	Increased densities of young-of-the-year of pike and perch	Young-of-the-year survey	No signs of disturbances to reproduction in the area. No time-series data of young-of-the-year production	Section 7.1.4
Recovery of the perch population in the Gålö area	Increase in spawning stock in the NTZ	CPUE spawners	CPUE for spawning individuals increases over time	Multimesh gillnet	No difference in spawning stock of perch between the NTZ and the reference area	Fig. 2
	Increase in stock size	CPUE	CPUE increases over time	Multimesh gillnet	No difference in abundances of large and small perch between the NTZ and reference area	
	Decrease in fishing mortality	Total mortality	Z decreases over time	Multimesh gillnet	No difference in mortality over time in the NTZ	Section 7.1.3
	A diversified size/age structure	Size structure	The proportion of large individuals increases over time	Multimesh gillnet	An increase in the proportion of large perch in the reference area compared to the NTZ	Section 7.1.2
	Increase in recruitment	Production of Young-of-the-year	Increased densities of young-of-the-year perch	Young-of-the-year survey	No signs of reproductive disturbances in the area. No time-series data of the production of young-of-the-year	Section 7.1.4
Restoration of a predatory fish dominated system in Lännåkersviken	Increase in the proportion of predatory fish	Trophic level, mean	Increase in mean trophic level	Multimesh gillnet	No difference in mean trophic level over time between the NTZ and reference area	Section 7.2
	Increase in the proportion of predatory fish	Biomass predatory	Increase proportion predatory fish in	Multimesh gillnet	No difference in the proportion of predatory fish over time between	Section 7.2

		fish/total biomass	the total fish community		the NTZ and reference area	
	Increase in the proportion of predatory fish	Multivariate analysis of fish community composition	Changes in community composition in the NTZ depending on predatory fish	Multimesh gillnet	Species composition diverges over time between NTZ and reference area, party driven by pikeperch	Section 7.2
Restoration of an attractive recreational fishery of pikeperch and pike in the Gålö area	Increase in catches of pikeperch and pike	CPUE of large pikeperch and pike	CPUE of large pikeperch and pike increases compared to control	Multimesh gillnet and trapnet	CPUE of pikeperch increases with a factor 5-11 and CPUE of pike by a factor 4-6 in the NTZ compared to reference area	Section 7.4
	Increase in densities of pikeperch compared to other areas	CPUE of large pikeperch	CPUE of pikeperch larger than in other comparable coastal areas	Multimesh gillnet	CPUE of pikeperch >minimum catch size on average 4 times higher in the NTZ compared to other coastal pikeperch populations	Section 7.4

7.3.1. Recovery of the populations of pikeperch, pike and perch in the no-take zone (2010-2015)

Population densities

The catch per unit effort (CPUE) in the trapnet survey in spring (May-June) was generally higher in the NTZ for all three focal species, compared to the reference area (Fig. 2). For an evaluation of reopening of the area to fishing, see section (9.3.5). Figure 2 illustrates the CPUE of the total spawning stock, i.e., includes individuals that spawn in this specific area, but spend the rest of the year outside the NTZ. For pikeperch and pike, only individuals of a length > 40 cm are included, and for perch only individuals of a length > 20 cm. This generally corresponds to sizes at sexual maturity for the three species, and for pikeperch and pike, it is also the smallest allowed catch size.

For pikeperch, significant differences in the CPUE between the two sites were detected during 2010-2015 and 2010-2016 (ANCOVA, interaction Site * Year, $F = 25.79$, $p < 0.01$), whereas CPUE did not differ between the sites for pike and perch. Because of the substantially lower catches of pike during 2015, differences in

CPUE for the time period 2010-2014 were also tested in a separate analysis. When the year 2015 was excluded, a significant difference in CPUE between the sites was detected for pike (ANCOVA, interaction Site*Year, F= 10.43, p < 0.05).

The increases in CPUE of the focal species during the time period 2010-2015 differed substantially between the NTZ and the reference area. The linear regressions show that for pikeperch, CPUE increased by a factor 4.9 and for pike by a factor of 4.4 in the NTZ compared to the reference area. For perch, there were no differences in CPUE between the NTZand the reference area.
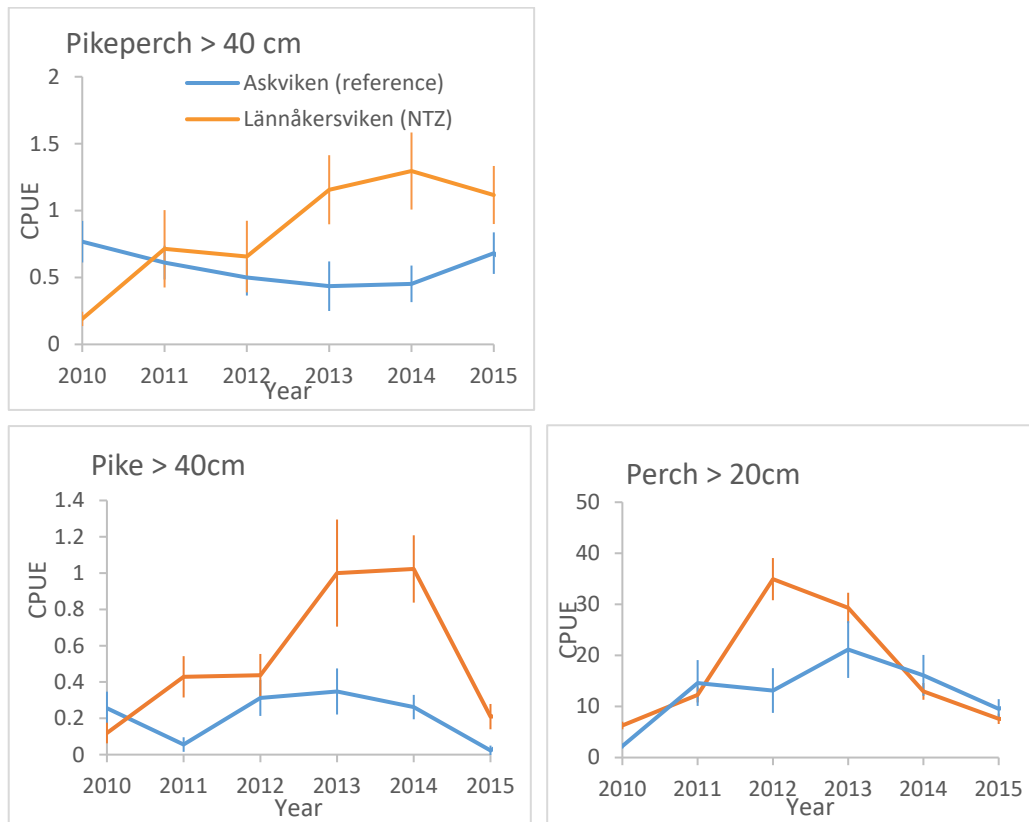
Figure 2. Catch per unit effort (CPUE, mean ± standard error [SE]) of adult pikeperch, pike and perch in the trapnet survey. The survey was conducted during May-June in 2010-2015 and targeted spawning fish.

The gillnet survey offers a representative view of the total fish community composition, as well as of abundance and size structure among the focal species during the summer months. During the summer, some of the adult fishes migrate from the NTZ, which is why there is a higher ratio of younger fish in the catches in August compared to the trapnet survey in spring. Figure 3 shows the catch development derived from the gillnet survey in August 2010-2015. The catch was

categorised into small and large individuals, for pikeperch large individuals were defined as > 40 cm, for pike > 40 cm, and for perch > 20 cm.
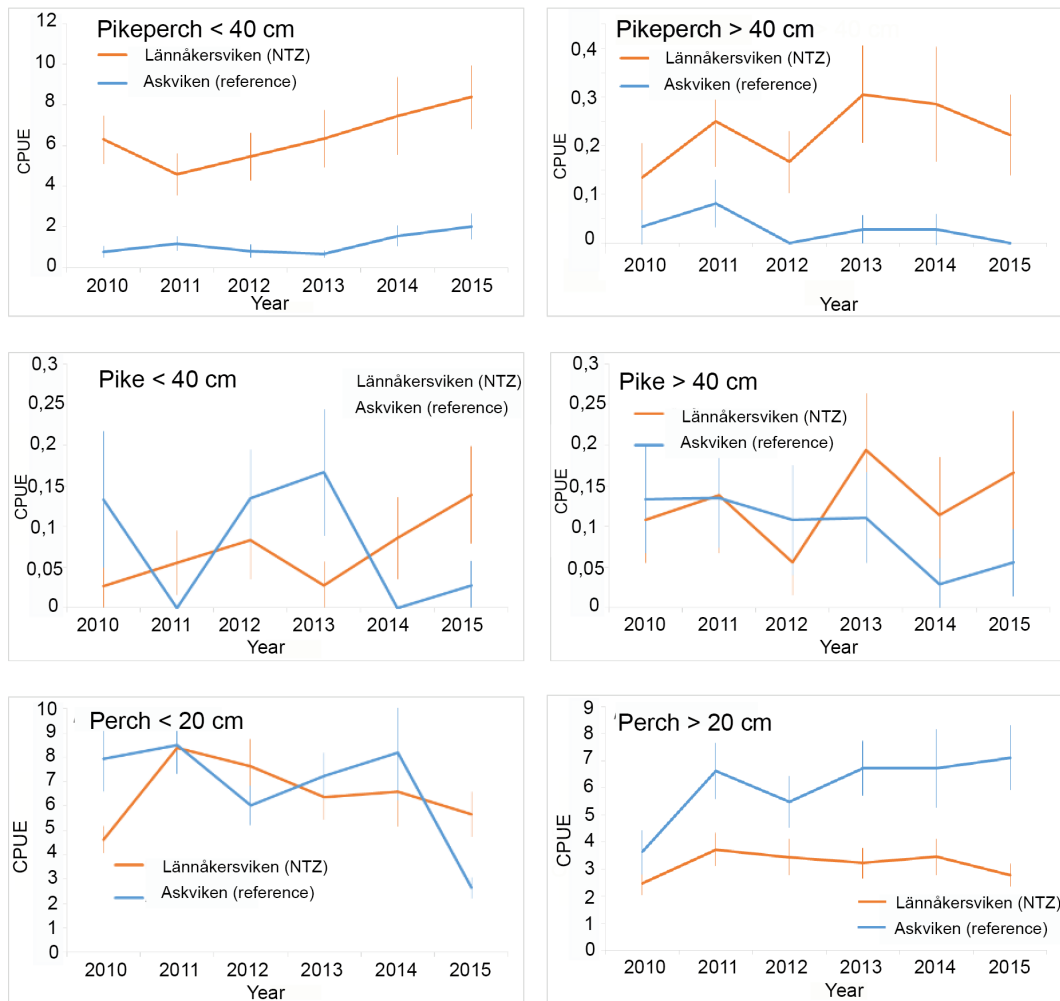
Figure 3. Catch per unit effort (CPUE, mean ± SE) during the gillnet survey. Fishing was conducted in August, 2010-2015. The catch is categorised into small and large fish where large pikeperch individuals were defined as > 40 cm, pike > 40 cm, and perch >20 cm.

The results indicate that the catches of both small and large pikeperch and large pike have increased over time in the NTZ Lännåkersviken compared to the reference area Askviken. However, this increase was not significant (ANCOVA; small pikeperch: F = 2.12, p = 0.18, large pikeperch: F = 2.89, p = 0.13, pike: F = 5.22, p = 0.052). A similar ANCOVA was conducted using biomass of fish as the response variable instead of catch numbers. Using this analysis biomass of large pikeperch was significantly higher in the NTZ (ANCOVA, interaction Site*Year, F = 2.89, p = 0.021).

Size and age distribution

Differences in the occurrence of the proportion of large individuals over time (> 40 cm for pikeperch and pike, > 20 cm for perch) between the two areas were evaluated using ANCOVA. The analysis showed a significantly higher proportion of large perch in the reference area (ANCOVA, interaction site*year, $F = 8.90$, $p = 0.018$) (Fig. 4). For pikeperch and pike, there were no significant differences between sites (ANCOVA interaction site* year, pikeperch: $F = 1.97$, $p = 0.20$, pike: $F = 0.47$, $p = 0.51$).
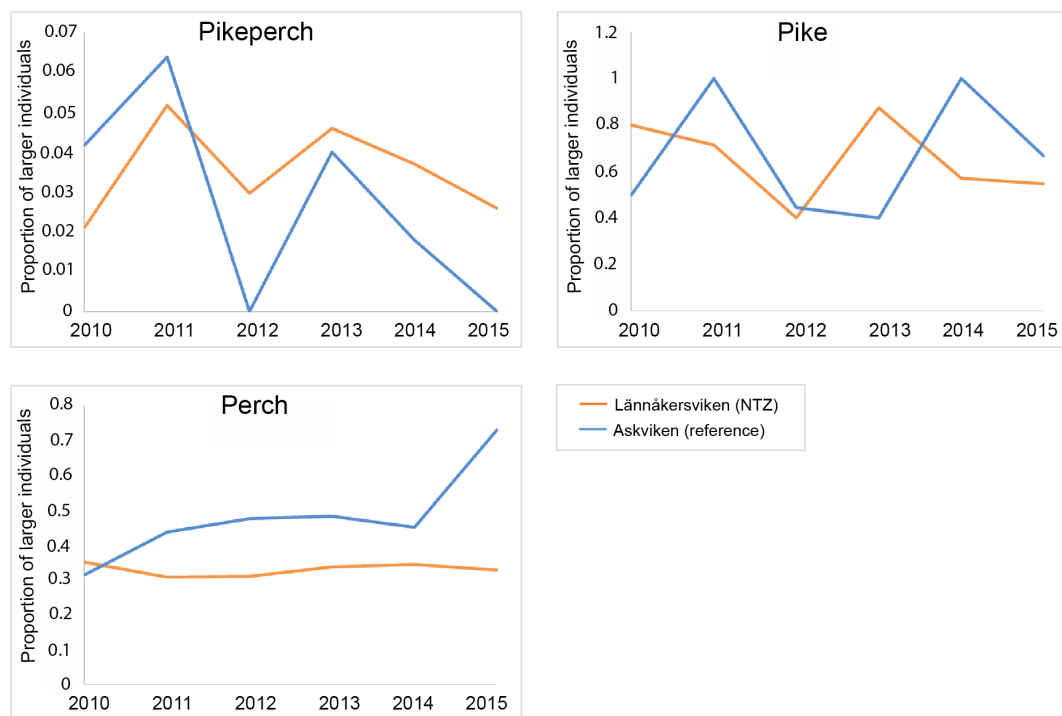
Figure 4. The proportion of large individuals (> 40 cm for pikeperch and pike, > 20 cm for perch) in the gillnet survey conducted in August 2010-2015.

For pikeperch, the development in the proportion of individuals older than four years was also evaluated. The results indicated that the trend is in opposite directions in the two sites; an increase of older individuals in Lännåkersviken and a decreasing trend in Askviken. However, these trends were not significant (ANCOVA, interaction site*year, $F = 0.76$, $p = 0.41$). For perch, there was not sufficient age data to conduct such an analysis.

The absence of an increase in the proportion of older pikeperch and pike in the NTZ compared to the reference area can potentially be explained by the small sample sizes of larger individuals in the fishing surveys, resulting in low statistical power. Additionally, there was an increase in the number of small individuals in the NTZ (Fig. 3), whereas the proportion of larger (and hence older) fish was relatively stable over time.

Mortality

The statistical analysis indicated very small differences in mortality rates of both pikeperch and perch between the different periods (2010-2012 versus 2013-2015 for pikeperch and 2009 versus 2013 for perch). In Lännåkersviken for pikeperch aged 4-8 years, the estimated mortality rate was 0.63 and 0.62, for the time periods 2010-2012 and 2013-2015 respectively. In a similar analysis including younger fish (1-8 years), the corresponding numbers were 0.95 and 0.89. Similar calculations were not possible to conduct for Askviken, due to the low numbers of older individuals in the samples. Generally, NTZs increase the survival of older fish due to the absence of fishing mortality, which was also the case in Lännåkersviken where the monitoring data indicated an increase in the densities of older pikeperch (Fig. 2). The low numbers of pikeperch in the catches made this analysis highly uncertain. Further, the absence of a difference in mortality between the time periods in Lännåkersviken can probably partly be explained by the close proximity of the time periods to each other. A longer study period would be needed to draw conclusions of NTZ effects on mortality.

For perch (4-8 years) in Lännåkersviken, the estimated mortality was 0.32 for both 2009 and 2013. The corresponding numbers for Askviken were 0.65 for 2009, and 0.35 for 2013. This indicates a reduction in mortality for perch.

Recruitment

The results from the young-of-the-year survey in 2009 (Fredriksson et al., 2010). and 2012 did not indicate any reduction in recruitment of the focal species. Both these studies showed normal reproduction rates for pikeperch, pike and perch in the area. This is in line with other studies which show that the recruitment in inner archipelago areas, like Lännåkersviken and Askviken is generally high (Bergström, 2009). This is in contrast with the low recruitment success for predatory fishes along open coastal areas in the Baltic Sea, which may be due to increased abundances of sticklebacks that feed on the eggs and larvae of predatory fishes (Ljunggren et al., 2010; Bergström et al., 2015; Byström et al., 2015, Eklöf et al., 2020).

7.3.2. Ecosystem effects

Increased proportion of predatory fish

Because the fishery in Lännåkersviken mainly targets predatory fish, a fishing ban can be expected to result in an increase in the proportion of predatory fish on a local scale. Differences in the proportion of predatory fish between the NTZ Lännåkersviken and the reference area Askviken, based on both numbers and biomasses, were evaluated for the gillnet data using ANCOVA. The analyses

showed no differences in predatory fish between the two sites, or differences in mean trophic level (based on biomasses); in Lännåkersviken the mean trophic level during the period was 3.7 whereas it was 3.9 in Askviken.

The multivariate PCO analysis showed that the fish community composition among sites and years differed significantly (Fig. 5). The largest differences were mainly attributed to the higher numbers of white bream/common bream, sprat and pikeperch in Lännåkersviken whereas perch was the most abundant species in Askviken. During 2013-2015 the divergence between the two sites was largest. The increasing differences were partly driven by the higher numbers of pikeperch in the NTZ (Fig. 5).
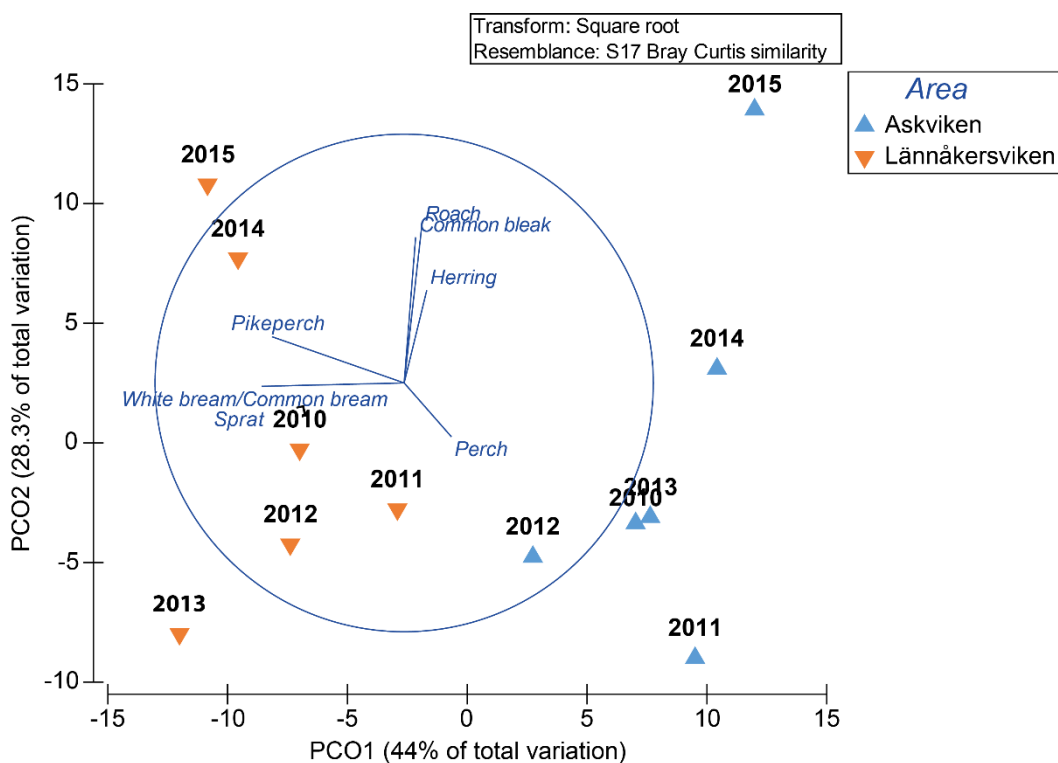
Figure 5. Development of the fish community composition over time in the no-take zone Lännåkersviken and reference area Askviken analysed by a multivariate PCO analysis. The blue vectors denote the species contributing most to the observed differences, as well as in which direction. For example, pikeperch contributes substantially to the difference in community composition (increasing in the no-take zone), which is illustrated by the blue vector pointing left. The two sites did differ from each other, which is indicated by their position on opposite sides on the PCO1. Each triangle represents the fish community composition in a specific year and site.

Catch development for additional fish species over the years (2010-2015) in the two sites were highly variable and there was no clear pattern (Fig. 6). The most common fish species (excluding the focal species pikeperch, pike and perch) were white bream, ruffe, roach and herring (all sizes included) (Fig. 6). In the NTZ, catch rates of all four species were similar to catch rates at the time (2010) when the NTZ was

established, however, catches increased in the reference area over time. An exception was roach, which showed a negative trend in the NTZ since its establishment. This was not observed in the reference area, where roach catch rates were similar across years. The occurrence of herring in the catches was highly variable due to occasions where large schools were caught, and therefore these results should be interpreted with caution.
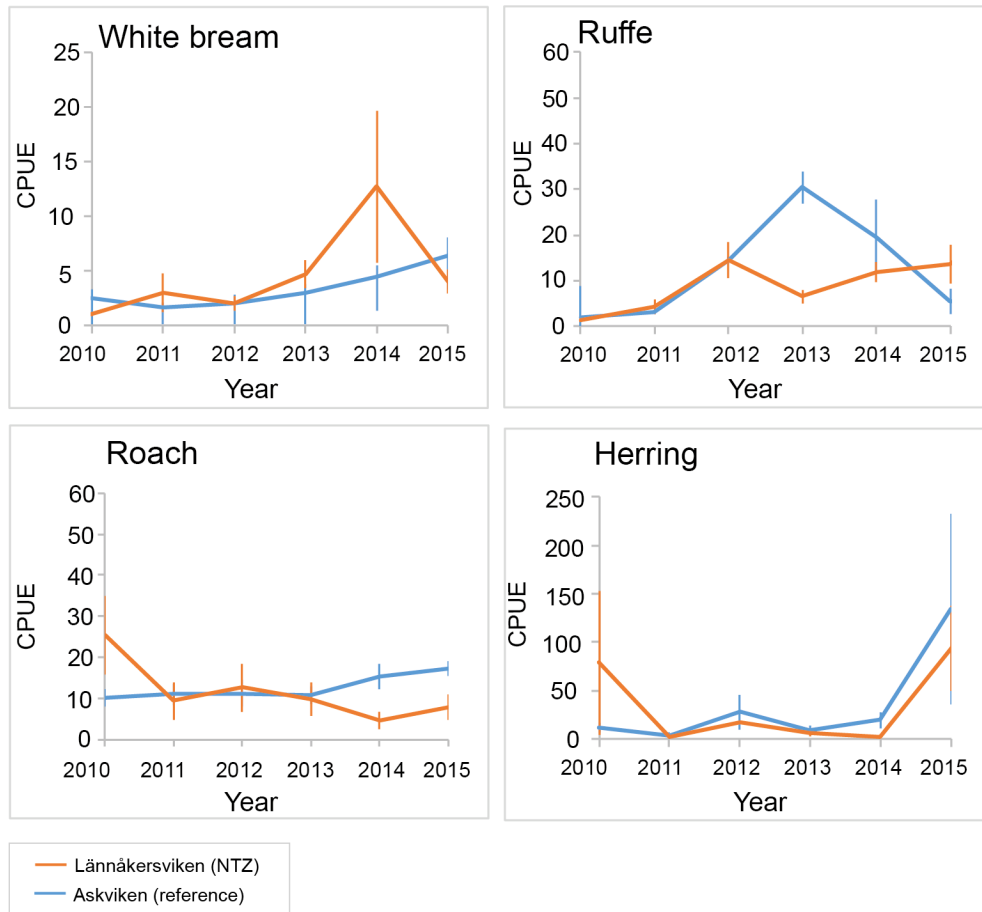
Figure 6. Catch per unit effort (CPUE, mean ± SE) of white bream, ruffe, common roach and herring in the trapnet survey. The survey was conducted during May-June 2010-2015. Lännäkersviken was a NTZ from 2010-2015 and Askviken is the reference site.

7.3.3. Impact from top predators

Injuries caused by cormorant and grey seal

Figure 7 shows the proportion of fishes with bite wounds caused by cormorant and grey seal during the trapnet fishery in spring (Fig. 7). There was a trend indicating the proportion of wounds increased up to 2016. For pikeperch and pike, this trend was interrupted in 2020, whereas the proportion of wounds on perch continued to increase. The large proportion of wounded pikeperch, pike and perch during 2015-2016 may have contributed to the observed drop in trapnet catches (Fig. 9). During the gillnet surveys in August, fishes with cormorant or seal injuries were few. The only reported injuries by cormorants were one pikeperch in 2014 and one in 2015, both in Lännåkersviken. This suggests that the problems with disturbance from cormorant or seal mainly occur during spring and early summer, i.e., during the trapnet survey.
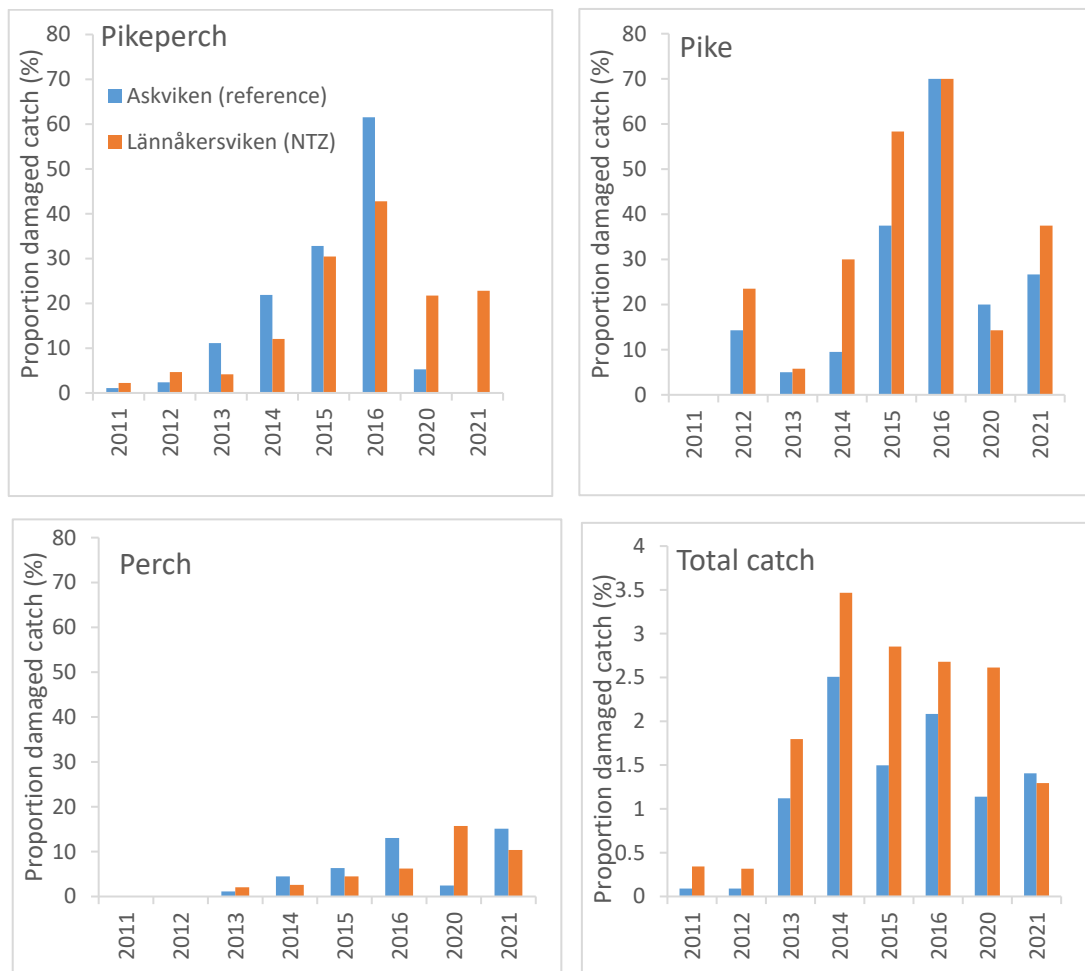
Figure 7. The proportion of the catch of pikeperch, pike, perch and total catches with bite wounds from cormorant or grey seal during the trapnet surveys in 2011-2016 and 2020-2021. The scale on the y-axes differs between the graphs.

Predation by cormorant and grey seal

During the trapnet survey in 2015, cormorants were counted. In total, 3197 cormorants were observed in Lännåkersviken, and 189 in Askviken. During the same time in 2020, 1442 and 39 cormorants were observed in Lännåkersviken and Askviken, respectively. In 2021, which was the last year of the study, 1637 cormorants were counted in Lännåkersviken and 61 in Askviken. The total removal of fish in Lännåkersviken was estimated as 79.1 kg per ha in 2015, and 63.6 kg per ha in 2021. The corresponding numbers for Askviken were 21.8 and 7.4 kg per ha, respectively. The potential effect on the fish stocks is thus considerably higher in NTZ Lännåkersviken than in the reference area.

Figure 8 shows the estimated species-specific removal of fish by cormorants during spring time, when the densities of cormorants are at its highest in the area. These results indicate that especially perch in Lännåkersviken may be severely impacted by cormorant predation, which is a possible explanation to the absence of a positive effect of the no-take zone on this species. Predation by cormorants can also potentially have a negative effect on the local abundance of other species, such as pike, perch and eel.
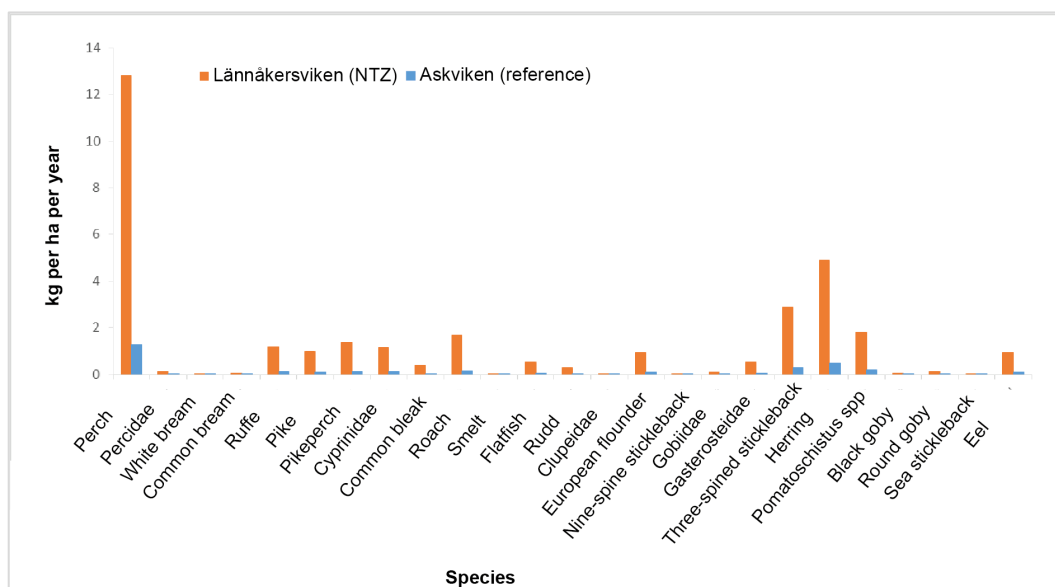
Figure 8. Estimates of the species specific fish consumption by cormorants, based on bird counts in the Gålö area during April-June 2015, combined with diet composition data from nearby colonies in 2012-2014.

The analysis of fish consumption by grey seal in the area around Gålö resulted in an estimated total consumption of 4.7 kg per hectare per year in both Lännåkersviken and Askviken. This is of the same magnitude as the fish

consumption by cormorant in Askviken, but is considerably lower than the estimated fish consumption by cormorants in Lännåkersviken.

The estimated consumption by grey seal of pikeperch, pike and perch in Lännåkersviken and Askviken was based on diet data from faecal samples collected in the central parts of the archipelago. There are large differences in diet composition between different parts of the archipelago; in the intermediate archipelago, diets were dominated by perch, cyprinids, pike and whitefish, whereas herring, sprat, cod and eelpout were the most important species in the outer archipelago. In the samples from the intermediate archipelago, which is most similar to the study area, perch constituted 50 percent (95 percent confidence interval: 41-59 percent) and pike 11 percent (95 percent confidence interval: 7-16 percent) of seal total consumption by biomass. Based on these intervals and the calculated total consumption, the annual removal of perch was estimated to be 2.0-2.8 kg per ha per year in both Lännåkersviken and Askviken. The removal of pike was calculated as 0.3-0.8 kg per ha and year in both areas. Data on pikeperch in the seal diet samples were too scarce to allow any estimates. The low abundance of pikeperch is probably a result of the lack of diet samples from the inner archipelago, where pikeperch is mainly found.

7.3.4. Additional analyses

Environmental conditions in the surveyed areas

Water temperature and water transparency were monitored in the two study sites over time, in order to see if differences in environmental conditions may have affected the development of the fish populations. The temperature conditions in the two bays were very similar and it is unlikely that this had any effects on the results in the present study (Fig. 9). Water transparency, on the other hand, differed between Lännåkersviken and Askviken (Fig. 10). The more turbid water in Lännåkersviken compared to the reference area may partly explain some of the observed differences in the results of the fishing surveys between the two sites, such as the higher catches of white bream, common bream and pikeperch in Lännåkersviken. These all contributed to the differences in fish community composition between the areas in the PCO-analysis (Fig. 5). There were, however, no trends in the transparency in the two areas over the course of the study, suggesting that changes in water transparency can be excluded as a potential explanation for the observed fish population changes.
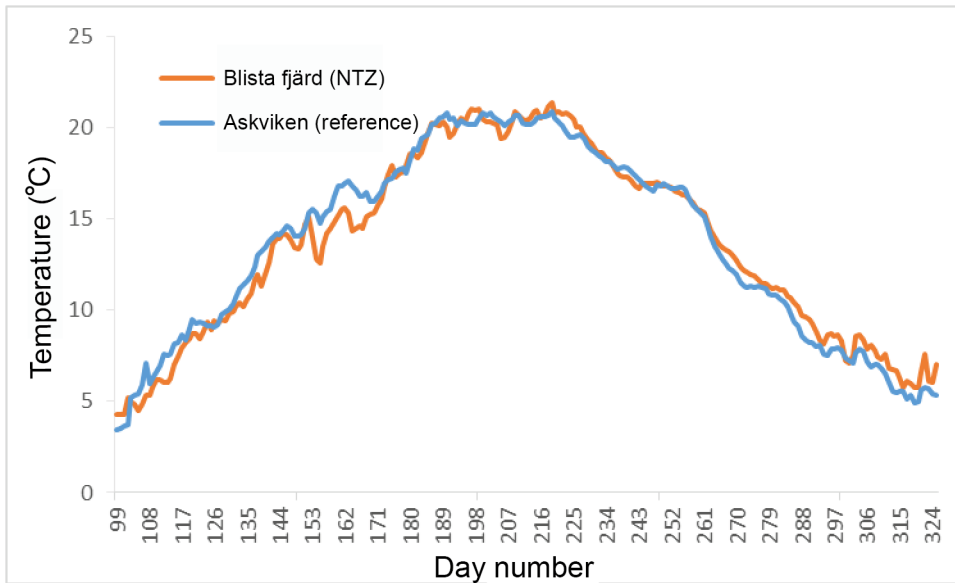
Figure 9. Mean temperatures derived from temperature loggers in Blista fjärd and Askviken deployed during 2009-2014.
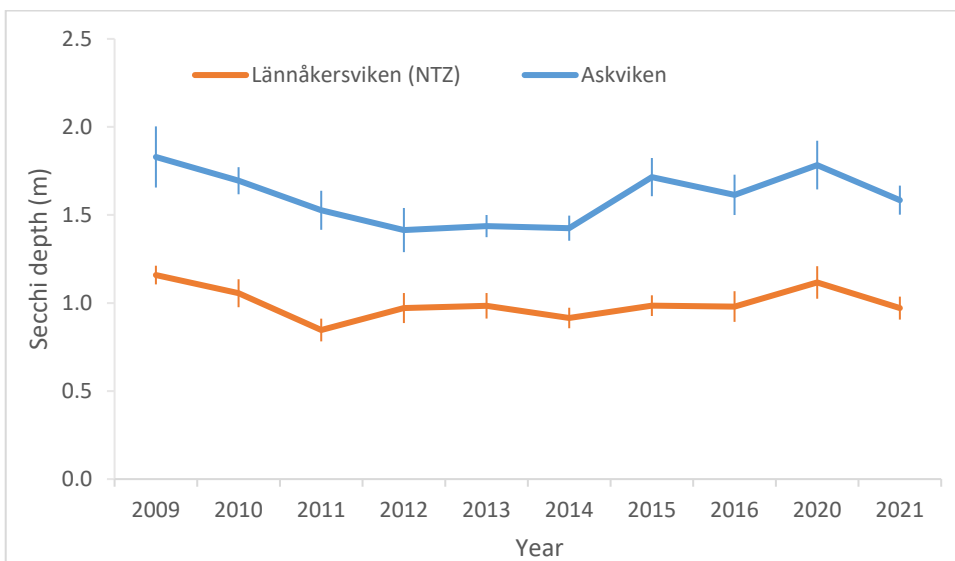
Tagging studies

During 2010-2015, a total of 403 pikeperch and 105 pike were tagged in Lännåkersviken. Of these, 11 pikeperch and three pike were recaptured, which equates to approximately three percent of the tagged individuals of both species. The recaptured individuals were either caught in the trapnet survey or reported by

recreational fishers. A total of 240 pikeperch and 66 pike were tagged in Askviken during the same time period. Of these, 37 pikeperch (15 percent) and two pike (three percent) were recaptured. Of the recaptures in Lännåkersviken, three pikeperch were caught outside the tagging study area (Fig. 11). The corresponding number for Askviken was one individual. This individual was caught approximately 5 km (distance over water) from the tagging area. The three individuals that were recaptured outside Lännåkersviken were found at a distance of 4, 16 and 20 km from the tagging area, which is within the home range of pikeperch living in the archipelago (Saulamo & Neuman, 2002). None of the recaptured pike were caught outside the tagging study area.
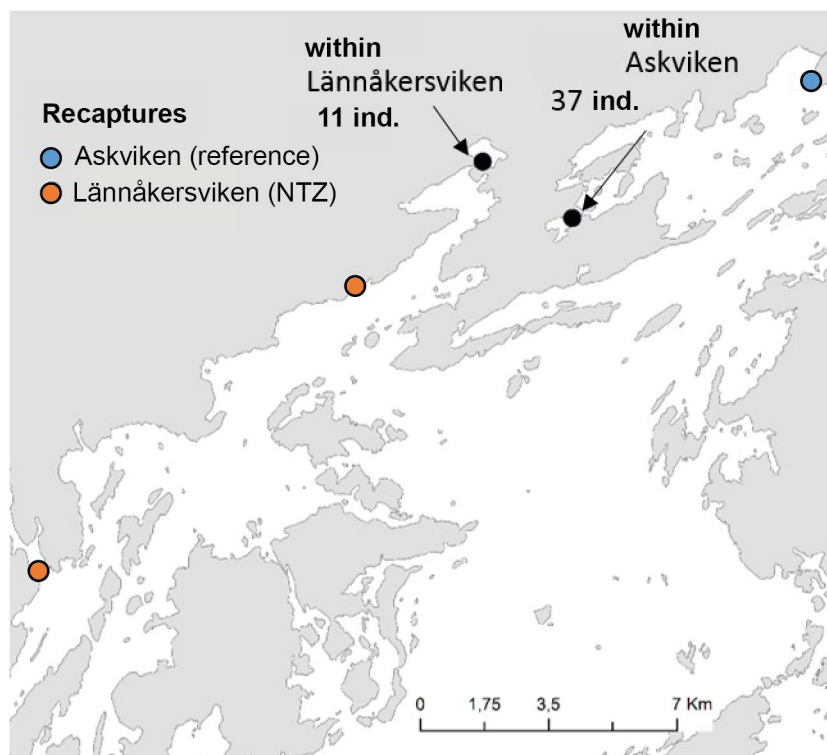
Figure 11. Map over the study area and the location of recaptured pikeperch individuals that were tagged in Lännåkersviken and Askviken. For Lännåkersviken, three individuals were recaptured outside the tagging study area by recreational fishers, and for Askviken, 1 individual was recaptured outside the tagging study area. All other individuals were recaptured in the trapnet survey.

An annual population density estimation was conducted for pikeperch in Askviken and Lännåkersviken, based on the capture-recapture data from the trapnet survey using a Schnabel calculation, a method used for repeated tagging and recapture during the study period. The recapture rate of pike was so low in both Lännåkersviken and Askviken that analysis was not possible. Only recaptured pikeperch individuals caught in the trapnets during the same year as being tagging were included in the analysis, to minimise the effects of mortality between tagging

and recapture. A population value per area and year was calculated. Based on these calculations, the spawning population in Lännåkersviken was estimated as 2477 ± 1559 pikeperch (mean $\pm$ standard error for the years 2010-2014), and to 239 ± 126 individuals in Askviken. The recapture rates were low, especially in Lännåkersviken, which contributes to large uncertainties in the estimates.

7.3.5. Development of pikeperch, pike and perch after reopening the no-take zone to fishing (2015-2021)

In the summer of 2015, the previous NTZ was opened to fishing, leaving only a spawning closure during 1 April-15 June, i.e. the same fishing regulations as in the buffer zone. The area was fished in 2020-2021 using trapnets to evaluate the effects of reopening of the area. During 2020, there was a substantial decrease of pikeperch in the NTZ and the reference area. Similarly, in the NTZ catches of pike have been decreasing since 2016, and the catches of perch have also decreased over time (GLMM pike, time period; chisq = 11.2, Df = 1, p > 0.001; GLMM perch, time period; chisq = 6.5, Df = 1, p = 0.01) (Fig. 6). A probable reason is disturbance by seals at the fishing stations (Section 9.3.3). One grey seal was accidentally caught during the fishing surveys, but injuries on fishes induced by seals were continuously observed also after this event. For perch, one potential explanation for the observed decrease in CPUE could be predation by cormorants (Section 9.3.3).

For pikeperch, abundances decreased from an already low level in the reference area Askviken after 2015, and in the NTZ Lännåkersviken after 2016 (Fig. 12). The Before-After Control-Impact (BACI) analysis showed a significantly stronger decline in CPUE in Lännåkersviken compared to the reference area after the reopening (GLMM, interaction site*time period, chisq= 256.4, p < 0.001). The CPUE was still higher in Lännåkersviken, but the steeper decline here shows that opening the area to fishing had a strong negative effect on the pikeperch population, despite the area remaining closed during the spawning period. Similarly, pike decreased in abundance after the removal of the complete fishing ban (Fig. 6) but there was no significant BACI-effect (GLMM, interaction site*time period, chisq = 1.40, Df = 1, p = 0.24). For perch, there was also a decrease in CPUE in Lännåkersviken after the reopening of the site compared to the reference area (GLMM, interaction site*time period, chisq = 9.64, Df = 1, p = 0.002).
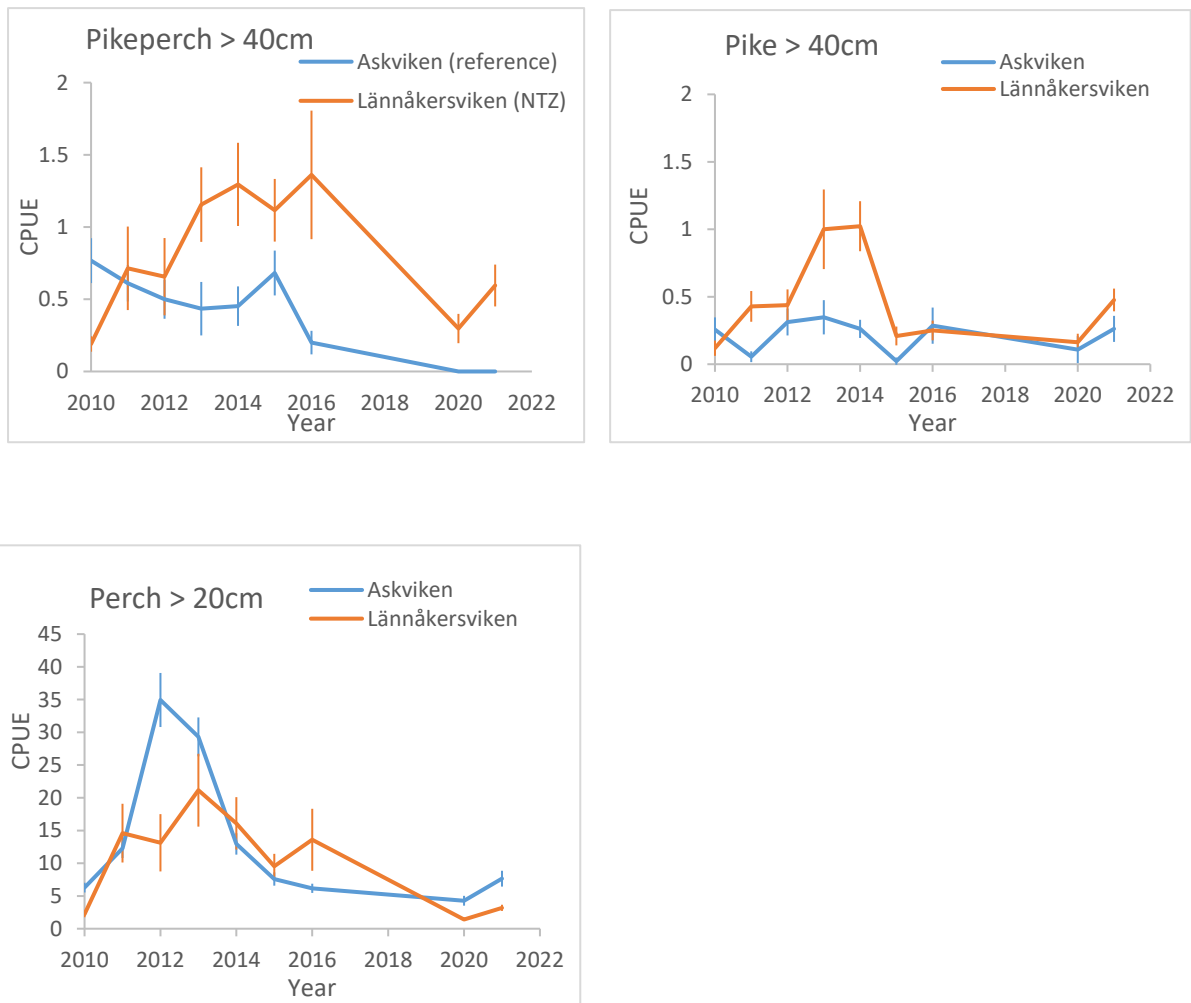
Figure 12. Catch per unit effort (mean ±SE) of large pikeperch, pike and perch in the trapnet survey, which was conducted during May-June in 2010-2016 and 2020-2021 and targeted spawning (adult) individuals. In June 2015, the no-take zone in Lännåkersviken was reopened to fishing, leaving only a spawning closure during 1 April – 15 June.

For roach, there was an opposite pattern with an increase in CPUE in the NTZ after the reopening, and there was a significant interaction effect of site and time period (GLMM, interaction site*time period, chisq = 5.1, p = 0.02). This implies that there was an effect upon the abundances of roach depending on the closure, which was actually positive. The Before-After (BA) analysis confirmed these results; for pikeperch there was a significant difference in CPUE before and after reopening Lännåkersviken (ANOVA, chisq = 13.8, Df = 1, p < 0.001), with lower CPUE during the years following reopening and this was also significant for pike (ANOVA, chisq = 14.23, Df = 1, p < 0.001). For perch, there was also a significant

difference in CPUE after the reopening of the NTZ (ANOVA, chisq = 12.3, df = 1, p < 0.001), with a decrease in CPUE in Lännåkersviken after 2015.

Catch development for additional fish species over the years (2010-2021) in the two sites were highly variable and there were no clear patterns (Fig. 13).
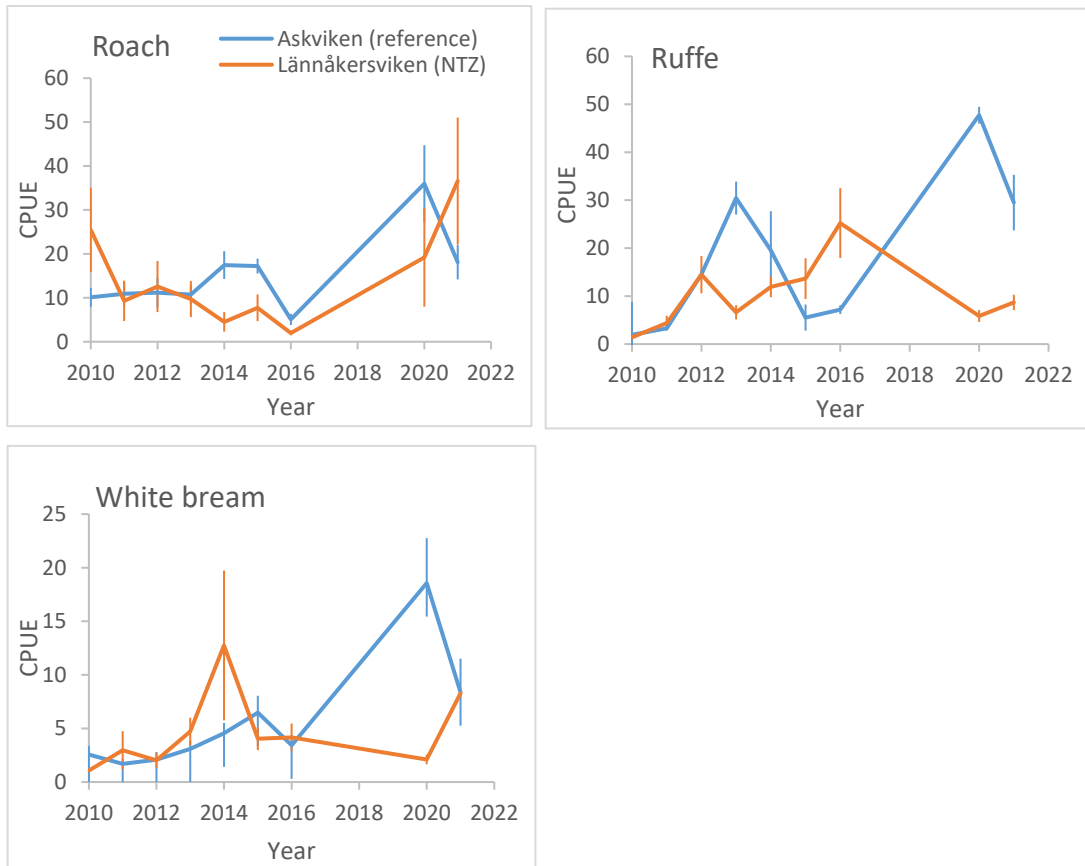
Figure 13. Catch per unit effort (mean ± SE) of roach, ruffe and white bream in the trapnet surveys, which were conducted during May-June 2010-2015 (total closure in Lännåkersviken) and 2016-2021 (spawning closure).

However, the proportion of functional groups changed in the NTZ Lännåkersviken after the reopening of the area (2016-2021), where the rate of cyprinid species increased at the expense of piscivorous species (Fig. 14). Piscivores were decreasing also in Askviken, but there was not a clear increase in cyprinid species (Fig. 14).
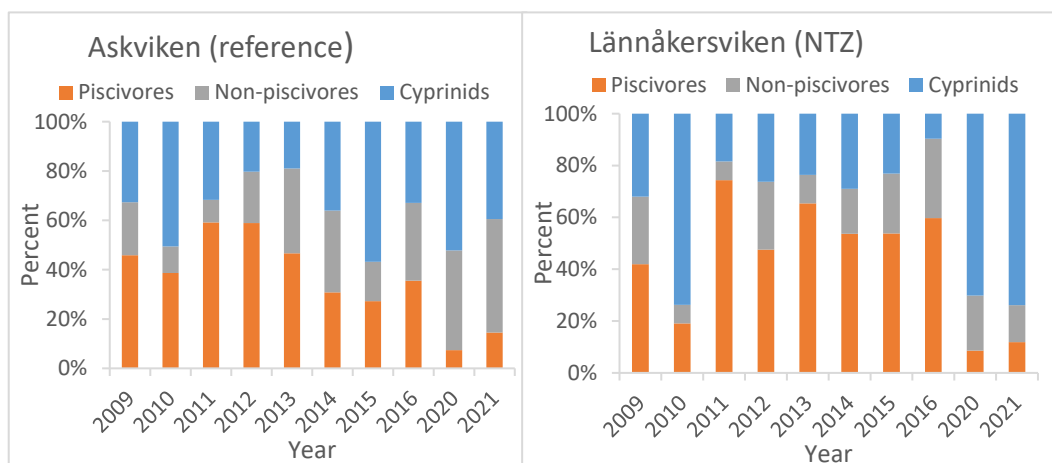
Figur 14. Functional groups of fishes in Askviken and Lännåkersviken prior to the implementation of the no-take zone (2009), during the implementation of the no-take zone (2010-2015) and after the reopening of the area (2016-2021).

7.3.6. Reestablishment of an attractive recreational fishery of pikeperch in the Gålö area

The catches of adult pikeperch had a strong positive development in the NTZ compared to the reference area during 2010-2015, as shown by both the trapnet and gillnet data. Linear regressions showed that pikeperch in the NTZ increased by a factor of five in the spring survey and by a factor of 11 in the late summer survey compared to the reference area Askviken, where no area-specific regulations of fisheries have been in place. For pike, there was a similar development as for pikeperch during 2010-2015 (with the exception of the drop in catches in the trapnet in 2015, which were attributed to seal disturbance); pike increased fourfold in the spring survey and sixfold in the late summer survey.

The trapnet catches in 2020 and 2021 indicated a decrease of pikeperch in Askviken, but also a substantial decrease in Lännåkersviken after opening the area to fishing. The catches of pike during 2020 and 2021 were also substantially lower than during the total fishing closure. In summary, the data indicates that the NTZ likely reversed the negative trend (that was observed in the reference area), and contributed to significantly stronger local stocks of pikeperch and pike during the five years of total protection. Further, the results from the last two years, 2020-2021, show that the fish populations decreased again after the reopening of the area to fishing (Fig. 12).
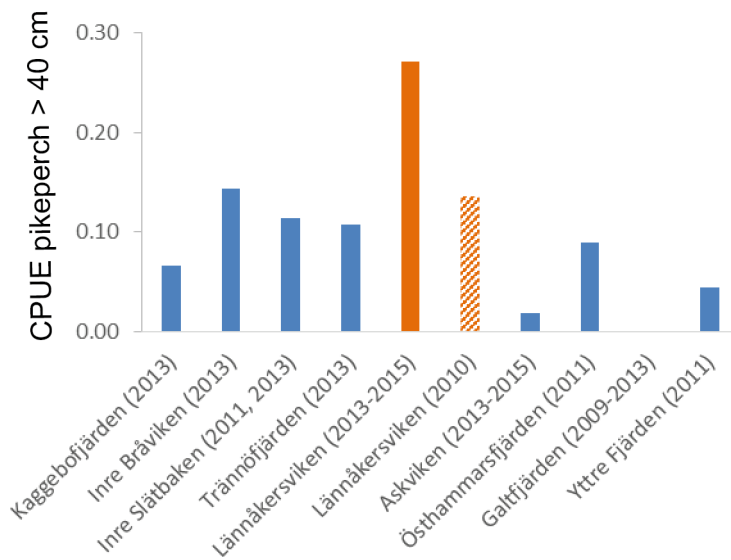
Figure 15. Catches per unit effort of large pikeperch (> 40 cm) in gillnet surveys in Lännåkersviken when the area was closed (orange dashed bar) and after the area had been closed for 3-5 years (orange solid bar). The blue bars show CPUE of pikeperch in other areas located along the Baltic coast within a 200 km radius from the Gålö area. Values in parantheses denote years when surveys were conducted.

A comparison with other coastal areas, which host abundant stocks of pikeperch, shows that the NTZ had considerably higher catch rates than all other locations (Fig. 15). These results support the conclusion that the NTZ had a strong positive effect on the focal species, pikeperch. Lännåkersviken constitutes the most important spawning area for pikeperch in the Stockholm archipelago. Because pikeperch have a rather large home range in archipelago environments, up to 15 km, according to tagging studies (the present study and Saulamo & Neuman, 2002), the pikeperch fishery could be expected to benefit the surrounding archipelago areas due to migration of individuals from the NTZ (see Bostedt et al. 2021). The availability of suitable spawning grounds is a limiting factor for pikeperch and pike in the Baltic Sea (Sundblad et al., 2011; 2014). During spawning, these species aggregate in relatively small areas, and an intense fishing pressure during this time can have a negative effect on fish stocks that normally reside over larger areas. Therefore, the NTZ implemented in this area likely benefitted the total stock, even outside the borders of the area, although this was not observed in the reference area Askviken.

After reopening the area to fishing in 2015, leaving only a spawning closure, declines in the pikeperch and pike stocks were evident, indicating that the populations were not able to withstand the total pressure from fishing and predation from seal and cormorant.

7.4. Discussion

The populations of pikeperch and pike responded quickly to the establishment of the NTZ in Lännåkersviken in Gålö. In the NTZ, catches per unit effort of large pikeperch increased with a factor 5-11 during the five-year closure compared to the reference area Askviken, which was open to fishing. For pike, the catches of adult individuals, increased with a factor of 4-6 up to 2014, compared to the reference area. In 2015 pike catches dropped, likely as a result of seal disturbance. Not all the different individual trends (summarized in Table 2) were statistically significant, even though the changes in mean values over time could be large. Because of the short time series, the statistical power in the tests were low, but because the observed patterns were similar in both surveys, and the effect size, i.e., the change in percent of catches in the NTZ compared to the reference area, was high, the conclusion seems robust. In 2015, the NTZ was reopened to fishing, apart from during the spawning period, and the results from the survey conducted in 2020-2021 indicated a decrease in abundances of all three species of large predatory fish (pikeperch, perch and pike) as a result.

There were tendencies towards an increased proportion of large and old individuals of pikeperch in the NTZ, but these trends were not statistically significant. The mortality estimates for pikeperch showed no decrease in the NTZ. The weak effect on the size and age structure and the mortality is probably explained by the increase in catches of younger and smaller fish, which increased almost to the same extent as adult pikeperch in the area. The higher recruitment of pikeperch following the increased amount of spawning individuals can indirectly be an effect of the NTZ. This pattern indicates that there may previously have been "recruitment overfishing", i.e., such a strong fishing pressure that the reproduction of pikeperch was negatively affected. Similarly, for pike, there was a tendency towards increased densities of young fish in the NTZ, which could explain the lack of effect of the fishing regulations on pike size- and age structure.

For perch, the NTZ did not result in any positive effects on population densities or size structure. This is probably because of the strong predation pressure of cormorants on the perch population in the NTZ. The estimated removal of perch by cormorant, based on bird count and diet data, was high in the NTZ and considerably lower in the reference area. Hence, the predation pressure likely had a strong negative impact on perch abundance and probably masked any positive effects of a reduction in fishing pressure. Top predators can have large effects on prey densities, both in terrestrial and aquatic systems (Berger et al., 2008; Baum & Worm; 2009). There are several studies from the Baltic Sea indicating that grey seal and cormorant can have a major impact on coastal fish populations (e.g. Östman et al., 2013; Hansson et al., 2018; Svensson 2021; Bergström et al. 2022). Site-specific

conditions may further affect predation patterns. For example, in Askviken, boating activities are frequent, which may disturb top predators like grey seal and cormorant (Strong and Morris, 2010; Platteeuw and Henkens, 2013). In the NTZ, disturbances originating from boats were low, partly because recreational fishers did not use the area and partly because there are no marinas in the area. Consequently, the lower anthropogenic activities in the NTZ may indirectly benefit these top predators.

Diet analyses of grey seal and cormorant in the area show that especially perch constituted a large part of the total diet. For cormorants residing in the vicinity of Gålö, perch constituted 36 percent of the total weight of ingested prey, whereas for grey seal perch constituted 50 percent of the total prey biomass in the central parts of the Stockholm archipelago. The corresponding numbers for pike were 3 and 11 percent, respectively, and for pikeperch 4 and 0 percent, respectively. The estimated removals, based on the diet proportions above and estimates of total removals of fish in the study areas, indicate that cormorant likely had a substantial effect on the local perch population. For pikeperch and pike, these estimates suggest that predation effects were likely weaker. However, the documentation of injuries/bite wounds induced by cormorants and seals on these fish indicated that pikeperch and pike can still be exposed to a high predation pressure, which may imply further disturbance for surviving individuals (Harris et al., 2008). During 2016, 50 percent of all pikeperch and 67 percent of all pike that were caught in trapnets had bite wounds. Although some injuries could have been inflicted inside the traps, most of them were likely caused before the fish entered, as the fish inside the traps are difficult to access for predators. In 2020 and 2021, the numbers of injured pikeperch and pike were lower, whereas the proportion of wounds on perch inflicted by cormorants had increased substantially. The proportions of wounded pikeperch, pike and perch were much higher than the corresponding numbers for other fish species, which indicates that grey seal and cormorant actively select these larger fish during foraging.

The grey seal population in the Stockholm archipelago has increased substantially since the 1990s (Bergström et al., 2022), which has led to an increase in predation pressure on coastal fish, but other ecosystem changes may also have contributed. A study from the northwestern Atlantic found that seal-induced injuries on salmon were related to the abundance of their preferred prey (herring), rather than to seal abundance. When abundances of herring increased, the rate of injuries on salmon decreased and vice versa, irrespective of seal densities (Leach et al., 2022). In the Baltic proper, the abundance of herring has declined since the 1980s (SLU 2020), why a shortage of food may force seals to forage closer to shore and switch prey to coastal species, such as pike and perch. Currently, it is suggested that predation by seals constitutes the highest mortality factor of large individuals of pike in coastal

areas of the Baltic Sea, where seals remove much more pike than commercial and recreational fisheries combined (Bergström et al., 2022).

The overall goal to strengthen the predatory fish community in and around Lännåkersviken and Blista fjärd by establishing an NTZ was considered successful. In doing so, the conditions for a high-quality recreational fishery was considered fulfilled for pikeperch and pike after the five-year closure (2010-2015). The stock development was much stronger in the NTZ compared to the reference area on the other side of Gålö. Moreover, the densities of pikeperch, which was the main focal species of the management program, were several times higher in the NTZ than observed for other coastal populations of pikeperch. The socio-economic effects of the temporary closure have been evaluated separately, and showed that the net benefits of the NTZ were positive (Bostedt et al. 2020). During the following five-year period, when the site was reopened to fishing and only remained closed during the spawning season, the results from the survey showed that the populations of pikeperch and pike had decreased substantially. This shows that a spawning closure was not sufficient to maintain the populations of predatory fish, as these are subject not only to fishing but also to a substantial predation pressure. A similar pattern was found for whitefish in the Bothnian, Sea, in a NTZ that was reopened to fishing after a five year closure (chapter 6, this report).

Regarding ecosystem effects induced by the fishing regulations, there were limited effects on the fish community as a whole during the total closure in 2010-2015. The proportion of predatory fish in the surveys did not increase in the NTZ compared to the reference area. On the other hand, the species composition in the two sites diverged over time. This shift was mainly caused by the increased abundances of pikeperch in the NTZ. The lack of clear effects on the total fish community can probably be explained by the relatively short study period of five years. A longer time period is likely needed before the increasing abundances of predatory fish will cause pronounced changes on a fish community level (Eger & Baum 2020). One exception was roach, which decreased in abundance in the NTZ during the total fishing ban, likely reflecting an increase in predation pressure from large piscivorous species. For other common prey species, such as white bream and ruffe, catches were variable and it was difficult to distinguish a clear pattern.

After the reopening of the area (2016-2021), the proportion of predatory fish showed a pronounced decrease in the NTZ simultaneously as the proportion of cyprinid fish species increased, while the proportions remained unchanged in the reference area. This suggests a loss of top-down control of the cyprinids following the declines in perch, pike and pikeperch. A similar effect from predatory fish on cyprinids has previously been observed in the NTZ Licknevarpefjärden further

south at the Swedish coast (Bergström et al., 2019). Top-down controlled coastal habitats, i.e. systems dominated by large piscivores such as pikeperch and pike, are linked to reduced eutrophication effects and favourable conditions for canopy-forming benthic vegetation (Östman et al., 2016; Donadi et al., 2017). In the absence of large piscivores, a system may instead be dominated by cyprinid species such as roach and bream, which can drive the aquatic conditions towards a more turbid, eutrophicated environment (Iho et al., 2017). In the current study, there was an increase in the ratio of cyprinid species in the NTZ in 2020 and 2021, after the reopening of the area, but no differences in water transparency was observed. However, such effects are likely better observed later in the season (from when no data was available), as in spring turbidity in this area is highly influenced by land runoff.

In summary, the study shows that NTZs that are implemented to protect important spawning grounds can be a key component in fisheries management. Even a NTZ of a very limited size (1.7 km^2), plus an equally sized buffer zone with protection only during spawning time, can induce positive effects on the local populations of pikeperch and pike, even after a few years. This is probably also true for perch, in areas where the predation from cormorant and seal is not limiting population growth (see Bergström et al., 2019). The positive effects can be explained by the removal of a previously high fishing pressure on pikeperch and pike, in combination with the relatively small home ranges of these species, while the closed area offers suitable habitats for reproduction of the target species. Along the same lines, the increased fishing pressure likely explains the negative effects during the last years of the study (2016-2021). The study also shows that only a spawning closure does not provide sufficient protection in this particular area, where fish populations are subject to both fishing and predation from seal and cormorant. To maintain or restore healthy fish stocks and attractive fisheries at the Swedish Baltic Sea coast, fishing regulations may have to be combined with local management of cormorant and seal that forage in key habitats for coastal fish, such as the enclosed bays that are central for their reproduction.

7.5. References

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8. No-take zone for flatfish around Gotska Sandön in the Baltic Sea

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Photo: Ulf Bergström

Summary

A no-take zone (NTZ), where all fishing was prohibited, was established around the island of Gotska Sandön in in the central Baltic Sea in 2006. The main purpose of the new NTZ was to protect the local population of flatfish (flounder and turbot) and cod and to assess the response of these populations to the removal of fishing. To evaluate potential changes in the protected fish communities over time, scientific multi-mesh gillnet surveys of the fish populations were undertaken in the NTZ and in a nearby fished reference area, from 2006 to 2009. In addition, pre-protection survey data collected in the NTZ area from 2003–2005, allowed fish populations within the NTZ (but not in the reference area) to be compared pre and post protection. A follow up survey using the same methods and sampling the same NTZ and reference area was undertaken in 2021.

The key takeaway from the follow up study is that there continues to be higher densities of turbot and flounder, both large (legal retention size) and small (below retention size) in the NTZ compared to the fished reference area. However, in 2021 there is a concerning decrease in densities of large turbot in both areas, and the length of both turbot and flounder was shorter in recent monitoring in both areas compared to early monitoring in 2006–2009. Unexpectedly, within the NTZ, densities of large turbot and of flounder generally were also lower compared to pre-NTZ regulations when fishing was occurring. The drivers of the decreases in density and lengths remain unclear due to uncertainty in the source(s) and magnitude of mortality. Better assessments of predation, compliance with NTZ regulations and more regular monitoring are needed to understand the trends in fish densities and lengths in the Gotska Sandön NTZ.

8.1. Background

Gotska Sandön no-take zone (NTZ) is a fully protected zone located in the central Baltic Sea north of the large island of Gotland. The NTZ predominantly encompasses marine soft-sediment habitats, mostly sand, and was implemented in 2006 with the primary purpose of assessing the response of demersal soft sediment associated flatfish to the removal of fishing. Scientific multi-mesh gillnet surveys of the fish populations were undertaken in the NTZ and in a nearby fished reference area, from 2006 to 2009 (Florin et al. 2011, Florin et al. 2013) and similar pre-protection survey data from 2003–2005 was available for the NTZ. In addition, modelling of potential larval export, estimates of commercial fishing pressure in surrounding waters and of consumption by marine top predators, were also undertaken following the NTZ implementation (Florin et al 2011, Florin et al. 2013).

Florin et al. (2013) reported that after the protected area came into effect, there were higher densities of large (above legal retention size of 30 cm) and small (under 30 cm) turbot (*Scophthalmus maximus*) in the NTZ compared to the fished reference area. Within the NTZ there were higher densities of large turbot after protection began compared to before protection. Despite having a lower average length than in the fished reference area after protection, turbot were older in the NTZ and the sex ratio was more even. In addition, larval exports from the NTZ to fished areas to the south were predicted to be high. Similarly, after the protected area came into effect, both large and small European flounder (*Platichthys flesus)* and Baltic flounder (*Platichtys solemdali;* see methods for further information on these two closely related species) were found in higher densities in the NTZ compared to the fished reference area. Within the NTZ both were found in higher densities after protection came into effect compared to beforehand. Again, after protection, length was shorter in the NTZ compared to the reference area. For both flounder species, lower length at age in the NTZ suggested that growth was density-dependent. As no before pre-protection density or length data was available for the reference area, it was only possibly to directly compare between the reference area and protected area after the NTZ came into effect in 2006. Rising predator populations made the task of disentangling fishing and predation effects difficult. Overall, Florin et al. (2013) reported clear positive effects of the removal of fishing on flatfishes.

Here we update previous evaluations of the NTZ with recent multi-mesh scientific survey data from Gotska Sandön and the fished reference zone collected 12 years (2021) after the last monitoring. We focus on densities and lengths of turbot and flounder and also update estimates of commercial fishing pressure, as well as predator abundance and their diet and consumption of flatfish. Effects of protection on fisheries target species often take some time to manifest. For example, Babcock et al.'s (2010) review suggested that, on average, initial positive effects occurred after 5 years. In addition, the benefits of marine protection have been found to be greatest when accumulating five key features; the protected area is fully no-take, well enforced, has been in place >10 years, covers a large area (>100km^2) and is isolated (Edgar et al. 2014). Given that Gotska Sandön NTZ meets many of these key features (no-take, large and relatively isolated and can also now be considered an old NTZ), and early monitoring immediately after protection showed a positive response by flatfish to protection, then we expect that flatfish populations in the NTZ would continue to show benefits of protection compared to equivalent fished areas. More specifically, as fish densities in the NTZ were either increasing or stable in earlier monitoring, we predict that: 1) fish densities in the NTZ will remain relatively stable if at carrying capacity already or if not, then increase further, 2) fish densities in the reference area will increase if fishing pressure has continued to decrease, however will remain lower than in the protected area. Further, we predict that lengths will follow similar patterns to densities and remain stable or increase

where fishing pressure is reduced. These predictions come with the caveat that large increases in predator populations, and thus in the natural mortality of flatfish, may negate protection effects.

Combined, these 2003–2005, 2006–2009 and 2021 monitoring surveys of Gotska Sandön NTZ represent a rare example of long-term, before and after protection evaluations of NTZ effects on marine soft-sediments fishes. At a global scale, large areas of soft sediment habitat are encompassed within NTZs, however, there has been little investigation of soft sediment associated fish response to protection in fully protected NTZs (Fetterplace 2018). As a result, the effects of protection on marine soft sediment demersal fish, particularly over the medium to long-term, are largely unknown. The Gotska Sandön NTZ monitoring data therefore provides a valuable long-term dataset to investigate soft-sediment associated fish response to protection.

8.2. Methods

The island of Gotska Sandön lies off the coast of Sweden in the central Baltic Sea and is surrounded by the 360 km^2 Gotska Sandön NTZ (Figure 1). The NTZ's level of management is equivalent to a fully protected marine protected area (see Grorud-Colvert et al. 2021), , allowing no construction nor any extraction in the national park and Natura 2000 area that partly overlaps with the NTZ (https://lagen.nu/nfs/2014:7). The Swedish government established the NTZ in March 2006, as a means to assess the impact of fishing on demersal soft sediment associated flatfish and to evaluate the usefulness of NTZs in fisheries management in a Swedish context. Prior to the NTZ implementation, the area already likely had lower fishing pressure than areas to the south around the larger island of Gotland (Florin et al. 2013). The majority of the NTZ at Gotska Sandön overlaps with the marine nature reserve "Gotska Sandön-Salvorev" (https://eunis.eea.europa.eu/sites/SE0340097), although they have been established in separate processes, using different legal instruments. Almost 95% of the protected area is classified as sandbanks (1110) according to EUNIS habitat types. A reference area to the South, off the east coast of the island of Gotland (Figure 1) was chosen because of its similarity to the NTZ in terms of habitat and because fishing pressure was expected to be relatively high.
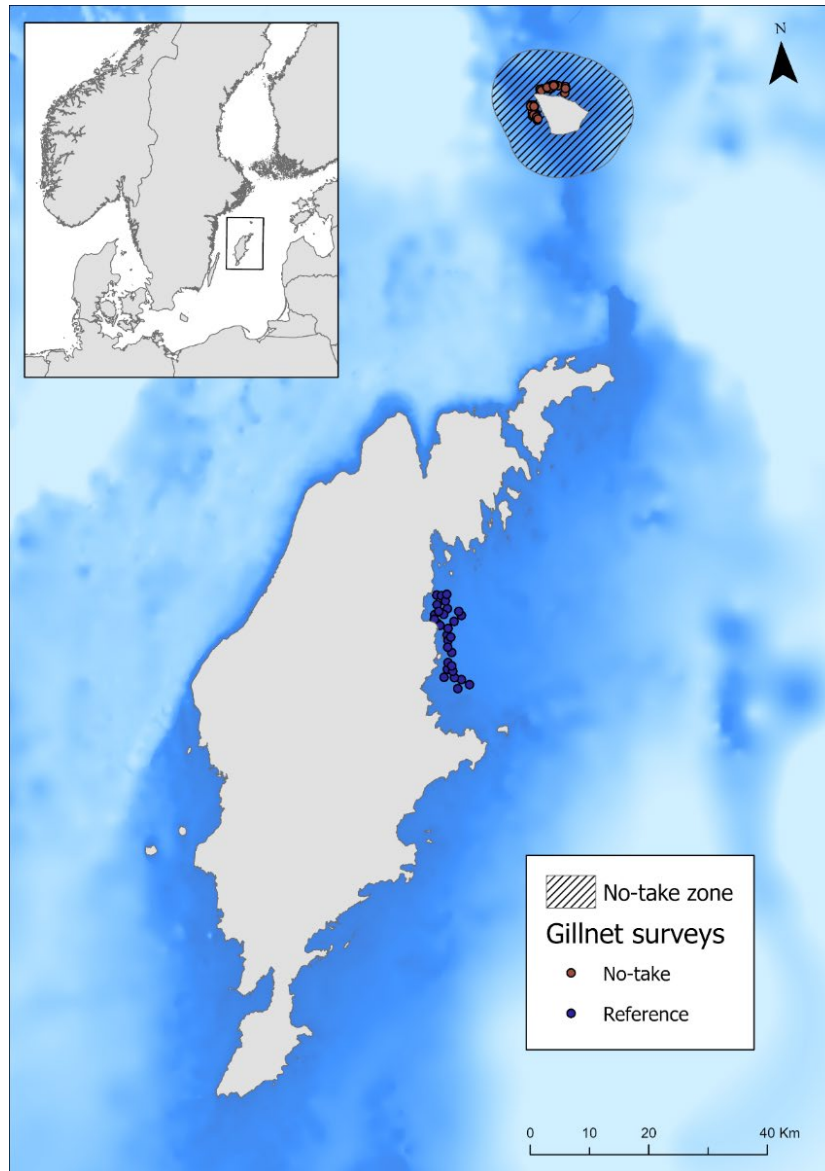
Figure 1. Map of Gotska Sandön no-take zone (NTZ) and the location of each gillnet survey station in the NTZ and in a fished reference area on the eastern side of the island of Gotland.

The main targeted species in the demersal fishery in the area are European and Baltic flounder (*Platichthys spp.*), turbot (*Scophthalmus maximus*), and cod (*Gadus morhua*). The NTZ was put in place primarily for the demersal flatfish, for which the area serves as an important nursery area, and evaluations have focused on these species (see Florin et al. 2011 and Florin et al. 2013). Low salinity in the eastern Gotland basin means that cod do not successfully reproduce in the shallow waters of the NTZ (Nissling and Westin 1997) and therefore were not expected to respond to a spatial fishing closure (Florin et al. 2011, Florin et al. 2013).

Turbot and flounder stock status are not available at the study area scale. More broadly, the International Council for the Exploration of the Sea (ICES) considers

turbot stocks in the Baltic to be stable, although uncertainty about the amount of discards makes it difficult to estimate stock status accurately (ICES 2021b). Targeted commercial fishing of turbot in the Baltic has been very restricted since the mid 1990s, and since then turbot have mainly been caught as by-catch in gillnet fisheries, with catch decreasing until around 2009 and then remaining relatively stable until 2020 (Sundelöf et al. 2022). Discards of turbot are considered to be very high, totalling around 28% of the total turbot catches for the period 2012–2020. In 2020 discards were about three times higher than the average and an increasing number of smaller turbot were caught (Sundelöf et al. 2022). Flounder stocks east of Gotland and in the Gulf of Gdansk (ICES subdivision 26 and 28) have decreased over time since the turn of the century (ICES 2021a). Catches have remained relatively constant over the same period, with the exception of 2019 and 2020 that were substantially lower than previous years. The maximum length of the European flounder has decreased over time in ICES sub-area 28, which may be an effect of an increase in the proportion of coastal spawning flounder (see following paragraph). The extent of recreational fishing is uncertain for both turbot and flounder, however, there are some indications that it may be substantial (Sundelöf et al. 2022).

There are two closely related species of European flounder found in the region; the offshore pelagic spawning flounder (*Platichthys flesus*) and the coastal benthic spawning Baltic flounder (*Platichthys solemdali*). Until recently (and during the earlier 2006–2009 monitoring period in this report), they were considered to be two ecotypes of the same species because their range overlaps geographically and they are morphologically very similar (Jokinen et al. 2019). However, they have different reproductive strategies, do not crossbreed and recent genetic studies have confirmed that they are two different species (see Sundelöf et al. 2022 and references therein for detailed information on the two species). During the monitoring period in early June, the adult flounder (*P. flesus*) are still mainly offshore after spawning and therefore we consider most of the European flounder caught during sampling to be the coastal spawning Baltic flounder (*P. solemdali*).

8.2.1. Fish monitoring

The same scientific sampling methods used in earlier 2006–2009 sampling in Florin et al. 2013 were repeated in 2021 monitoring. A single 195 m multi-mesh gillnet, comprising 5x9 m sections of mesh size (bar length) 25, 30, 38, 50 and 60 mm and 3x50 m sections of 75, 100 and 120 mm meshes, was deployed for one night, in early June, at each sampling station in the NTZ and reference area. Thirty-six fishing stations were to be fished in each area in each sampling year, however, due to weather conditions resulting in some samples not being collected and a small number of samples being excluded due to tampering with the equipment, not all

years have the full 72 samples (Table 1). Catch total for each species, fish length, temperature and depth (10–25 m) were recorded for each station. Catches from the 9 m long sections of the net were standardized to 50 m section length to give the same effort for all mesh sizes.

Some monitoring data was also available from the NTZ area from years 2003 to 2005 (prior to NTZ implementation). The data was collected at the same time of year as the later monitoring data and with similar gear; including the same mesh sizes (although with a 9 m long section of the 75 mm mesh, 25 m sections of the 100 and 120 mm meshes and an extra 85 mm section. By standardizing the catch to 50 m sections and excluding the 85 mm mesh section catch, these data could be used for a before-and-after protection comparison for the Gotska Sandön NTZ. For the before-and-after comparison we include the 2006 data in the before period as sampling was carried out at the start of the fishing season and the fishing closure was unlikely to have time to have an effect. No data from this period were available for the reference area.

Table 1. Number of gillnet surveys completed per year and per area. Each survey station was fished with one 150 m multi-mesh net-link per night. 2003–2005 data only available for the NTZ.

Location	2003	2004	2005	2006	2007	2008	2009	2021
NTZ	32	51	36	36	35	30	36	29
Reference	-	-	-	30	26	36	30	30

Here we use catch per unit effort (CPUE) as a proxy for fish densities, where CPUE is the number of fish per station. CPUE values were fourth-root transformed, following earlier monitoring methods, to obtain homogenous variances and normally distributed residuals. CPUE data was then split into above and below minimum legal landing size for both turbot (legal size 30 cm) and flounder (legal size 21 cm). Comparisons were made across years (2006–2009 vs 2021) and areas (NTZ vs reference area) for large (above legal size) and small (below legal size) fish within each species. Fish activity can be temperature dependent (Linløkken and Haugen 2006) and therefore CPUE may differ based on temperature effects rather than location or management type. To account for potential temperature effects, bottom temperature was included as a covariate in GLM's comparing CPUE (dependent variable) between areas (fixed factor) and years (fixed factor). We focus particularly on the interaction between the factors year and area to evaluate the effect of the NTZ over time, where a significant interaction indicates that the development of CPUE differs between the areas. Where temperature was not significant, temperature was removed and tests were rerun as standard two-way ANOVAs (GLMs and ANOVAs were performed using the aov function in R). Marginal mean plots were constructed and where temperature was significant,

CPUE values were adjusted for temperature, using the emmeans package in R (Lenth et al. 2021), prior to plotting.

Before-and-after comparisons of CPUE in the NTZ area were made across year groups (2003–2006, 2007–2009, 2021) for large (above legal size) and small (below legal size) fish within each species, with a temperature covariate in GLMs. Where temperature was not significant, temperature was removed and tests were rerun as standard one-way ANOVAs. Mean per station for each species by size (large or small) were plotted and where temperature was significant, CPUE values were adjusted for temperature prior to plotting. Data for 2003 and 4 stations in 2004 were removed where temperature was significant, as no temperatures were available. Post hoc pairwise tests using the Holm-Bonferroni method were used to compare differences between each of the year groups.

For visual comparison of length frequency distributions for each species and area, 1 cm length intervals were selected and plotted and the mean for each area overlain. Lengths for 2006–2009 and 2021 were plotted separately to compare length frequency distributions between the earlier and later monitoring periods. To compare mean lengths between areas and between monitoring periods within each area for each species Welch two sample t-tests were used and a Bonferroni correction applied for multiple testing. For these length comparisons using data standardized to 50 m mesh section length, values were rounded to the nearest whole number (e.g. one 24 cm fish per 9 m link becomes 5.56 fish per 50 m link when standardised and in length comparisons here is rounded to 6 x 24 cm fish). All length and catch statistics were calculated using the software R (www.r-project.org).

8.2.2. Fisheries

The impact of the commercial fishery in the studied areas was evaluated by using data on commercial landings from logbook data from 1996–2021, provided by the Swedish Agency for Marine and Water Management. Catch data from the NTZ and reference area were extracted in GIS with a 10 km buffered polygon of the study area, defined by the minimum bounding area (convex hull) of the gillnet survey stations.

8.2.3. Seals and cormorants

Both grey seals and great cormorants are species considered as opportunistic predators, mainly feeding on various fish species. Previous reporting concluded that the consumption of turbot and flatfish by grey seals (*Halichoerus grypus*) and great cormorants (*Phalacrocorax carbo*) around Gotland might exceed the commercial

fisheries landings, however, more information is needed about abundance and prey choice of seals and cormorants (Florin et al. 2013).

Great cormorant

Cormorants around Gotland are monitored annually by counting the number of nests in each colony during the breeding season. The first breeding pairs in the Gotland area established during the early 1990s, and the population increased during the 1990s–early 2000s to more than 10,000 breeding pairs in 2008. Since then the population has decreased and has fluctuated between 7000 and 8000 breeding pairs during recent years (Herrmann et al. 2018, Wirdheim and Green 2021) (Pers. comm. Kjell Larsson) (Figure 2).
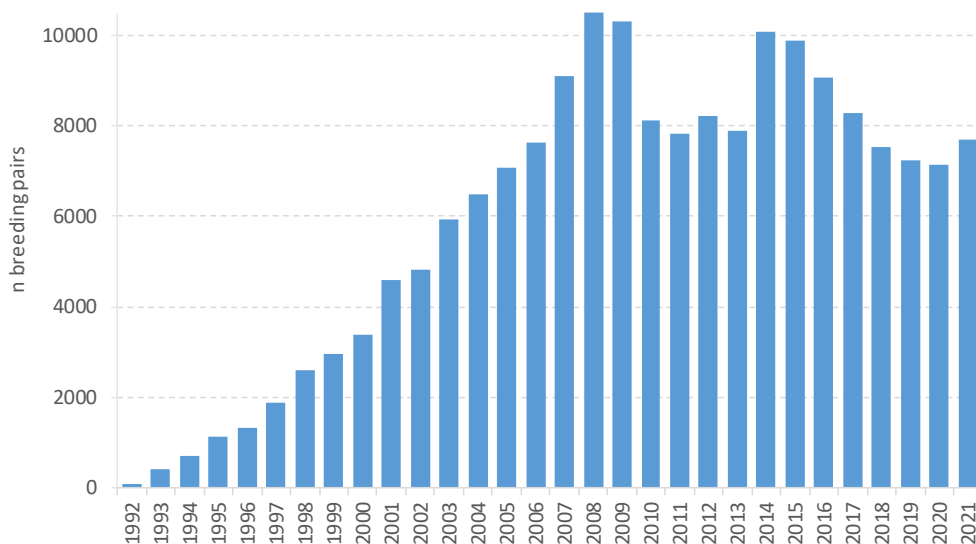
Figure 2. Number of breeding great cormorant pairs in different areas around Gotland 1992-2021. Data from (Herrmann et al. 2018, Wirdheim and Green 2021) (Pers. comm. Kjell Larsson).

Local predation on flatfish by cormorant (i.e. fish consumed per square kilometer per year) in the study areas was estimated by combining abundance data with information on diet composition and bioenergetic needs. For cormorants, available nest count data from 1994–2021 was used to estimate abundance (Pers. comm. Kjell Larsson) and the predation during the periods 2006–2009 and 2019–2021 was estimated by a GIS analysis. It was assumed that cormorants were present in the area during 180 days per year, in line with the estimate in Hansson et al. (2017). Estimation of total fish consumption by cormorant were based on food requirements for breeding adults during breeding season (80 days) from Grémillet et al.(1995) and a total consumption of 500 g per day for breeding birds and fledglings outside the breeding season (100 days) as well as for non-breeding birds throughout the season (Keller and Visser 1999, Ridgway 2010). Each nest was assumed to represent 2 breeding adults, 2 fledglings and one non-breeding bird, and

the average daily food intake was estimated to 1987 g per nest and day (Grémillet et al. 1995). It was assumed that cormorants forage within 20 km from their nests (Grémillet 1997, Nelson 2006), and a kernel density function with a 20 km radius was applied in ArcGIS to estimate cormorant densities across the study areas. Recent data on cormorant diet from Gotland is sparse and diet samples has only been collected from one colony in 2019 and unfortunately the estimated proportion of flounder (3%) and turbot (3%) in the samples were based on number of individuals, not by biomass and therefore it could not be combined with the analysis of total fish consumption. Instead, the estimates from Florin et al (2011 and 2013) was used to estimate the cormorant predation on flatfish (8% flounder, no turbot).

Grey seal

Grey seals are monitored annually during the moulting season in May–June within the national environmental monitoring programme performed by the Swedish Museum of Natural History. On Gotland, the seals are monitored using a mix of methods: from air, from boat and from land, with inter-annual variability in methodology. The number of grey seals counted during the annual survey has varied between 200 and 500 animals over the last 10 years (Figure 3). At Gotska Sandön, there are only seal count data available for 2007, 2012 and 2014 during the study period, with observations of 6, 71 and 18 individuals, respectively. The number of seals counted is assumed to represent 70% of the population size (Hiby et al. 2007). The monitoring data only gives a snapshot of the abundance of grey seals around Gotland. Information on seasonal variability in abundance and the importance of Gotland as a temporary area for seals migrating between the grey seal core areas in Estonia, Finland and Sweden is not available.
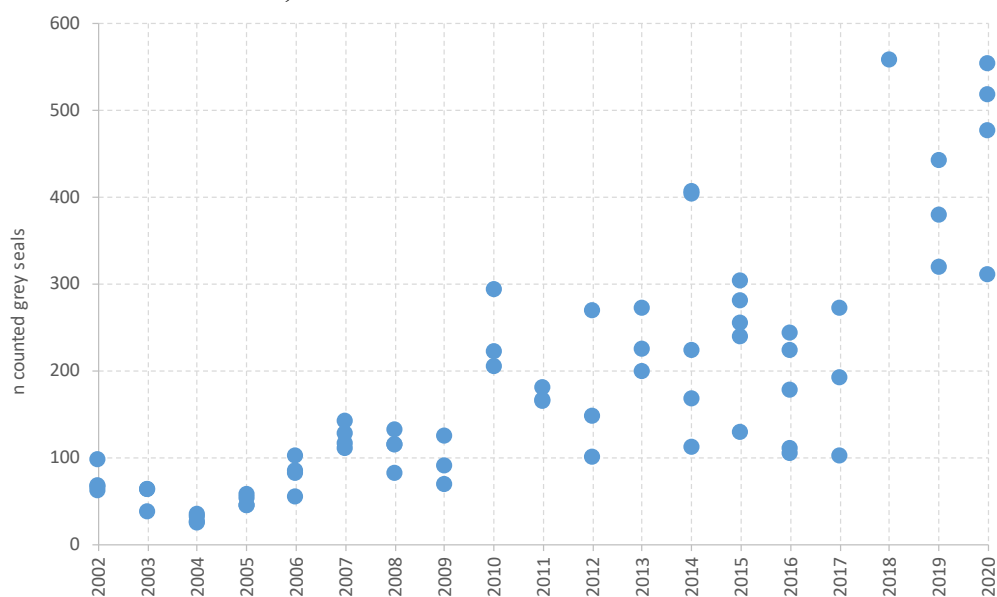

Figure 3. Number of counted grey seals around Gotland during the annual monitoring survey in May–June. Only days when the major haul-outs were covered have been included. Data obtained from Sharkweb (smhi.se) 2022-02-03.

Local predation on flatfish by seals (i.e. fish consumed per square kilometer per year) in the study areas was estimated by combining abundance data (Figure 3) with information on diet composition and bioenergetic needs. Data broken down by each species of flatfish was not available. Seals were assumed to forage within a 60 km radius (Sjöberg and Ball 2000, Oksanen et al. 2015) and densities were estimated using a kernel density function, with a search radius of 60 km. A daily total fish consumption of 4.75 kg per seal was applied (Hammond and Grellier 2006, Hammond and Harris 2006) and flatfish were assumed to make up 22% of diet based on Florin et al. (2013).

8.3. Results

Fifteen fish species were caught during gillnet sampling across all years and areas (2006–2006, 2021). In total, nine of these species were caught at Gotska Sandön NTZ (6–9 per year) and 11 species at the Gotland reference area (6–8 per year). Flounder (*P. solemdali*), cod (*G. morhua*) and turbot (*S. maximus*) dominated the catch in both areas in all years (see appendix Table A9.1 for full list of species caught and catch totals). Shorthorn sculpin (*Myoxocephalus scorpius*) also made up a large proportion of the catch in some years (e.g. 36% of total catch in 2021), however represented a very low proportion of the catch in others (e.g. 1% of total catch in 2007 and 2008). Also of note, is the appearance of the invasive round goby (*Neogobius melanostomus*) in the 2021 surveys at the Gotland reference area (just under 4% of the total reference area catch for 2021). All other species were caught in very low numbers and represented a negligible proportion of the total catch in any area or year.

8.3.1. Catch per unit effort

Large turbot ($\geq$30 cm) catch per unit effort (CPUE) differed by year ($F_{4,308} = 6.64$, $P <0.001$), showed an effect of temperature ($F_{1,308} = 5.88$, $P <0.05$) and there was a significant interaction between year and area ($F_{4,308} = 2.80$, $P <0.01$). CPUE increased in the NTZ over time from 2006 to 2009 and was higher in the NTZ than in the reference area from 2008 onwards (Figure 4). In 2021, CPUE was the lowest of any year for both areas, however, it remained higher in the NTZ compared to the reference area (Figure 4). Small turbot (<30 cm) CPUE differed by area ($F_{1,308} = 28.38$, $P <0.001$) and showed an effect of temperature (temperature: $F_{1,308} = 73.39$, $P <0.001$). CPUE increased in the NTZ over time from 2006–2009 and was higher in the NTZ than in the reference area from 2008 onwards (Figure 5).

There was no effect of temperature on CPUE for large flounder ($F_{1,308}$ = 0.35, P >0.05) or small flounder ($F_{1,308}$ = 0.12, P >0.05), so an ANOVA without the temperature covariate was used (following Florin et al. 2013). Large flounder (≥21 cm) CPUE differed by year ($F_{4,309}$= 12.86, P <0.001) and there was a significant interaction between year and area ($F_{4,309}$ = 9.01, P <0.001), showing an increase over time in the NTZ compared to the reference area. CPUE was higher in the NTZ compared to the reference area after 2006 in all years (Figure 6). The 2021 CPUE was slightly lower than in 2009 for both areas and CPUE remained higher in the NTZ compared to the reference area (Figure 6). Small flounder (<21 cm) CPUE followed a similar pattern and differed by year ($F_{4,309}$= 8.02, P <0.001) and there was also a significant interaction between year and area ($F_{4,309}$ = 10.29, P <0.001). CPUE increased over time in the NTZ and was higher compared to the reference area after 2006 in all years (Figure 7). The 2021 CPUE was slightly lower than in 2009 for the NTZ and higher than in 2009 for the reference area, however, CPUE remained higher in the NTZ compared to the reference area (Figure 7).

Figure 4. Mean catch per unit of effort for large turbot (above minimum legal landing size; ≥30 cm total length) in the no-take zone and reference area. Values are fourth square-root transformed numbers per station, mean±s.e., adjusted for temperature.
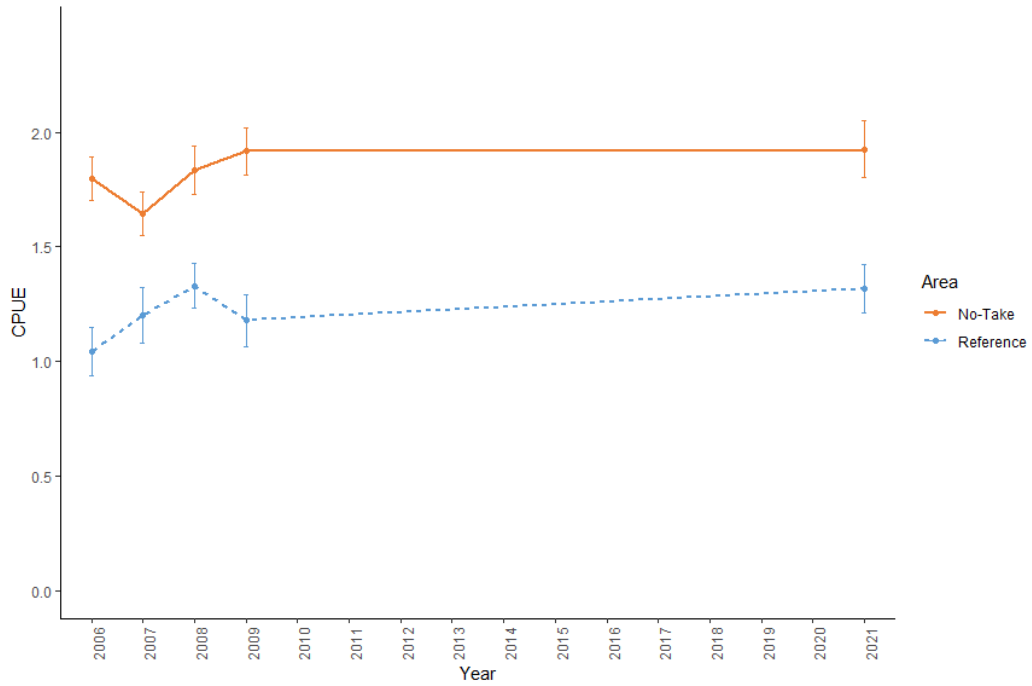
Figure 5. Mean catch per unit of effort for small turbot (below minimum legal landing size; ≤30 cm total length) in the no-take zone and reference area. Values are fourth square-root transformed numbers per station, mean±s.e., adjusted for temperature.
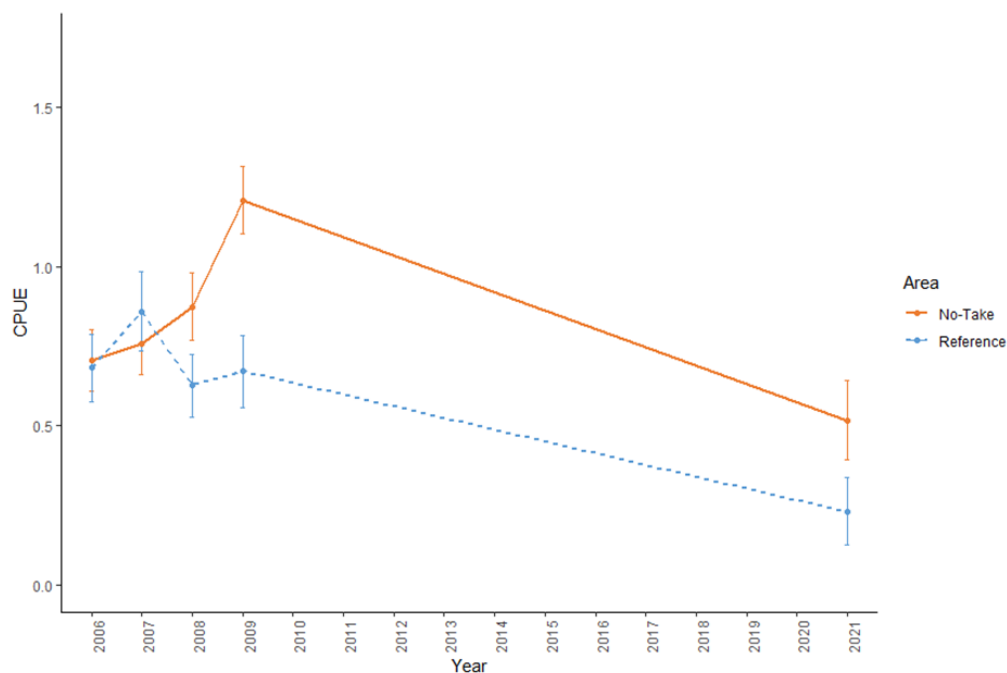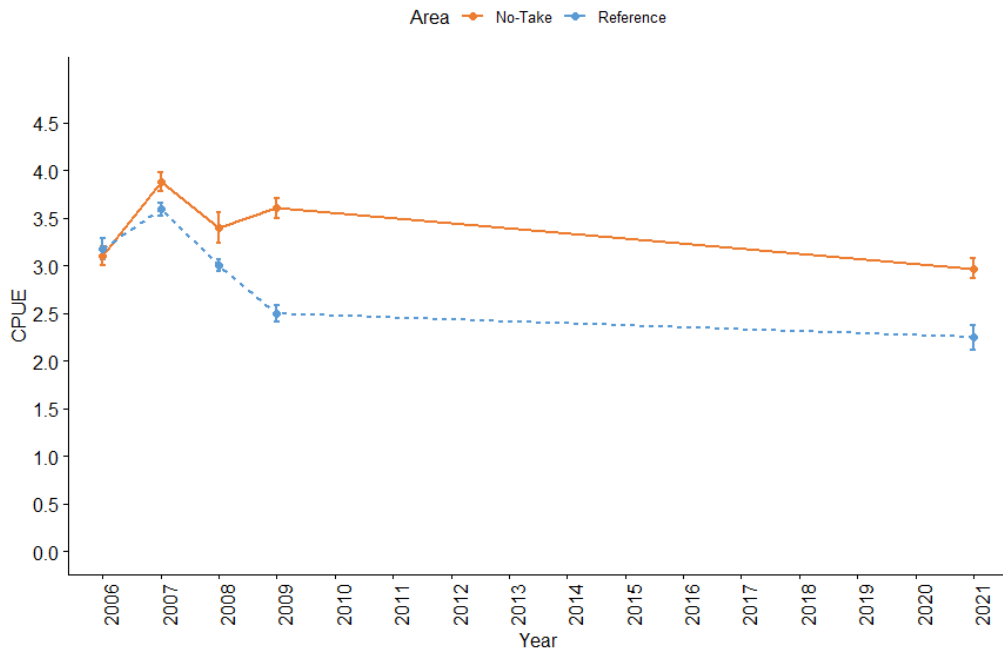

Figure 6. Mean catch per unit of effort for large flounder (above minimum legal landing size; ≥21 cm total length) in the no-take zone and reference area. Values are fourth square-root transformed numbers per station, mean±s.e.

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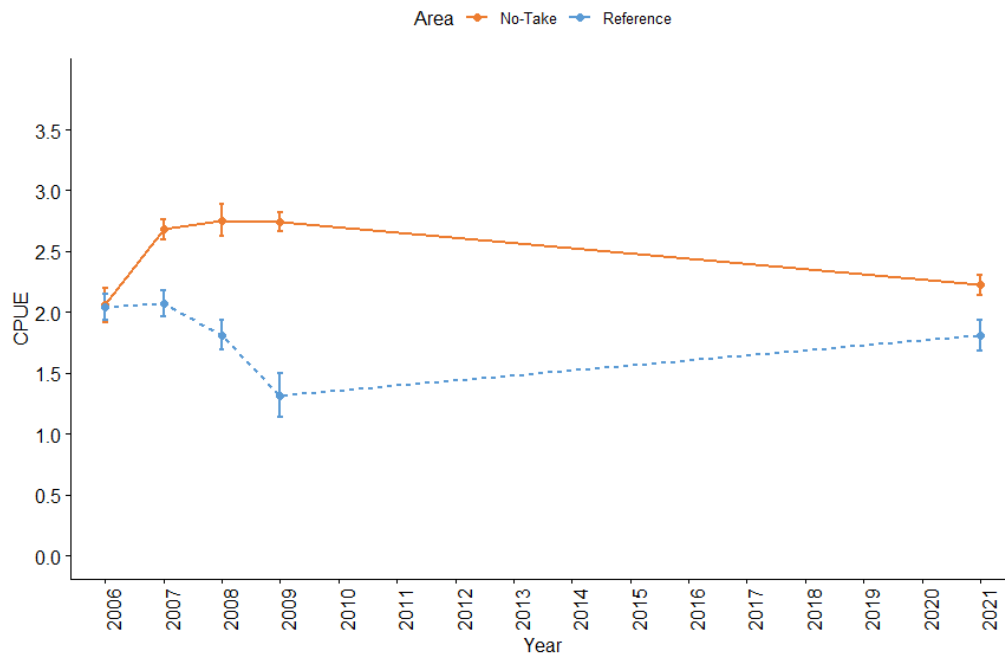
Figure 7. Mean catch per unit of effort for small flounder (below minimum legal landing size; ≤21 cm total length) in the no-take zone and reference area. Values are fourth square-root transformed numbers per station, mean±s.e.

8.3.2. Before-and-after NTZ comparison

Comparisons of CPUE prior to the removal of fishing and in the two post NTZ monitoring periods were made for turbot and flounder (small and large). There was no effect of temperature for large turbot ($F_{1,246} = 0.57$, $P>0.05$), so all data (including stations with no temperature data) were used in comparisons and temperature removed as a covariate. There was a significant difference in CPUE between periods for large turbot ($F_{2,283} = 10.19$, $P<0.001$). Pair-wise comparisons showed there were higher densities of turbot in 2007–2009 compared to 2003–2006 and higher densities in 2003–2006 compared to 2021 (2003–2006 vs 2007–2009: $P<0.001$, 2003–2006 vs 2021: $P<0.05$, 2007–2009 vs 2021: $P<0.001$) (Figure 8).

There was an effect of temperature for small turbot ($F_{1,246} = 13.09$, $P>0.001$), so stations without temperature data were not included in comparisons. There was a significant difference in CPUE between periods for small flounder ($F_{2,246} = 44.01$, $P<0.001$). Pair-wise comparisons showed that CPUE in 2003–2006 was higher than the other two periods, while there were no differences between the latter two periods (2003–2006 vs 2007–2009: $P<0.05$, 2003–2006 vs 2021: $P<0.05$, 2007–2009 vs 2021: $P>0.05$) (Figure 8).

There was no effect of temperature for large flounder ($F_{1,246} = 0.59$, $P>0.05$), so all data were used in comparisons and temperature removed as a covariate. There was

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a significant difference in CPUE between periods for large flounder ($F_{2,283} = 17.13$, $P<0.001$). Pair-wise comparisons showed that CPUE was highest in 2007-2009, while in 2003-2006 they were higher than in 2021 (2003–2006 vs 2007–2009: $P<0.001$, 2003–2006 vs 2021: $P<0.01$, 2007–2009 vs 2021: $P<0.001$) (Figure 8).

There was an effect of temperature for small flounder ($F_{1,246} = 8.22$, $P>0.01$), so stations without temperature data were not included in comparisons. There was a significant difference in CPUE between periods for small flounder ($F_{2,246} = 5.73$, $P<0.01$). Pair-wise comparisons showed that densities were higher in 2007-2009, while there was no difference between the other two periods (2003–2006 vs 2007–2009: $P<0.001$, 2003–2006 vs 2021: $P>0.05$, 2007–2009 vs 2021: $P<0.001$) (Figure 10).
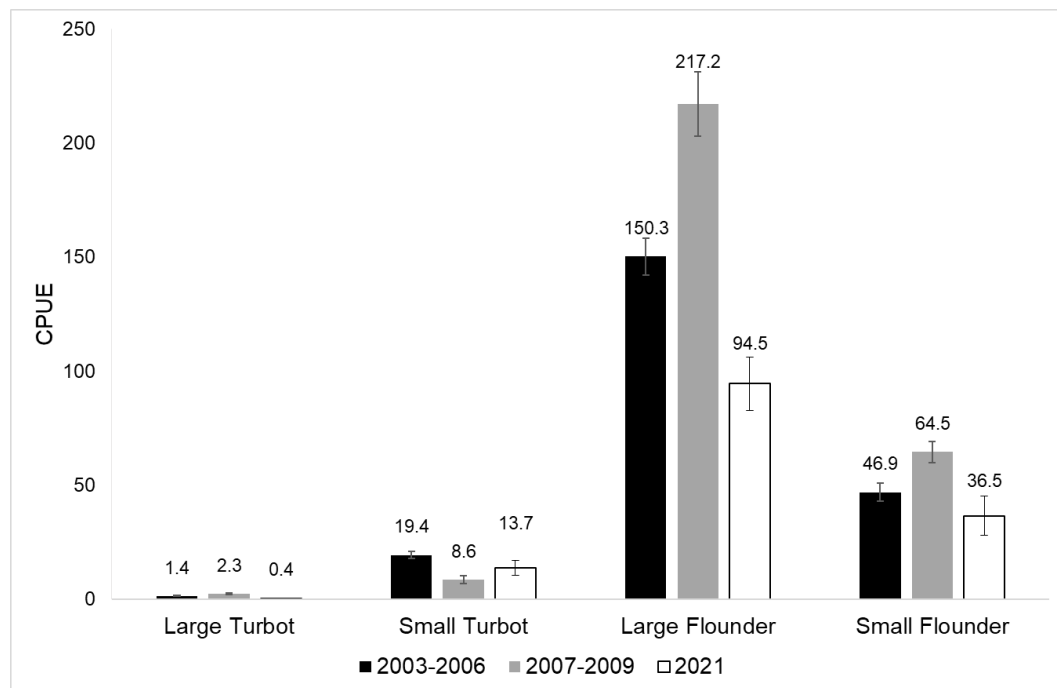
Figure 8. Mean catch per unit of effort (numbers per station) for turbot and flounder above and below minimum legal size (total length) in the no-take zone pre-protection (years 2003–2006;), previous monitoring (years 2007–2009) and recent monitoring (2021) periods.

Lengths

In 2006–2009, the average length of turbot caught in the reference area was significantly longer than in the NTZ (t = 0.87, df = 486.86, $P >0.05$; Figure 9). The shape of the length frequency distribution was unimodal in the reference area and had a somewhat bimodal distribution in the NTZ (Figure 9). In the NTZ there was a large proportion of small fish below 19 cm length and only a very small proportion at these lengths in the reference area. The longest fish, above 42 cm, made up only a very small proportion of the total fish and were only caught in the NTZ.

In 2021, in contrast to the earlier monitoring period, the difference in the average length of turbot between areas was not significant (t = 0.87, df = 486.86, P >0.05) and the shape of the distribution was similar in both areas with smaller fish dominating in both (Figure 9). There were no fish caught above 35 cm in either area in 2021. The average length was significantly shorter in 2021 compared to 2006–2009 in both the NTZ (t = 4.6213, df = 333, P<0.001) and the reference area (t = 15.322, df = 430.3, P < 0.001) (Figure 9).

In 2006–2009, the average length of flounder caught in the reference area was significantly longer than in the NTZ (t = -20.325, df = 23210, P < 0.001). The shape of the length frequency distribution was unimodal in both areas and slightly more skewed to larger sizes in the reference area (Figure 10). The longest fish, above 40 cm, made up only a very small proportion of the total. In contrast, in 2021 average length was longer in the NTZ than the reference area (t = 3.5142, df = 2504.2, P = < 0.01). The average length was significantly shorter in 2021 compared to 2006–2009 in both the NTZ(t = 9.0516, df = 3229, P<0.001) and the reference area (t =18.997, df =1746.4, P<0.001)(Figure 10). In 2021, smaller fish dominated in both areas and there were no fish caught above 37.5 cm in either area.
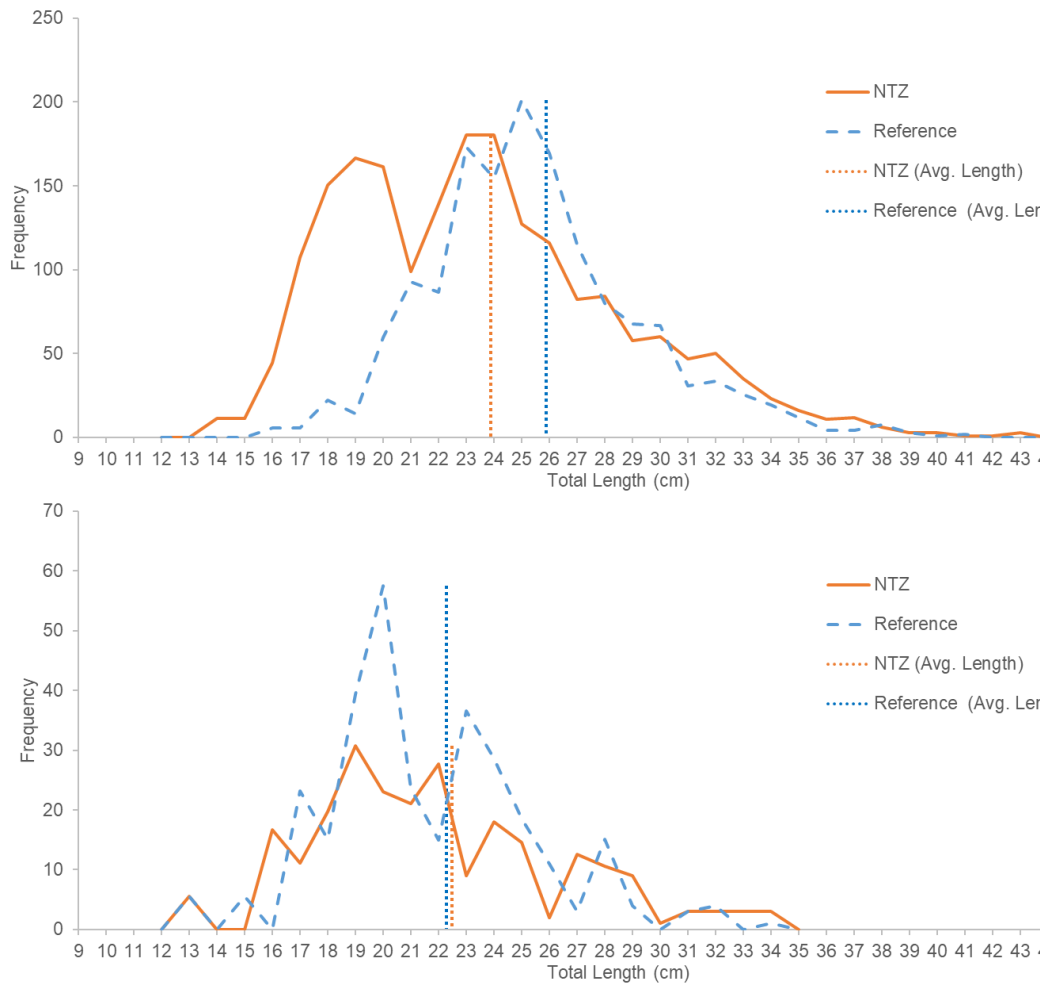
Figure 9. Length frequency distribution of all turbot in the NTZ and in the reference area, for 2006–2009 (top) and 2021 (bottom).
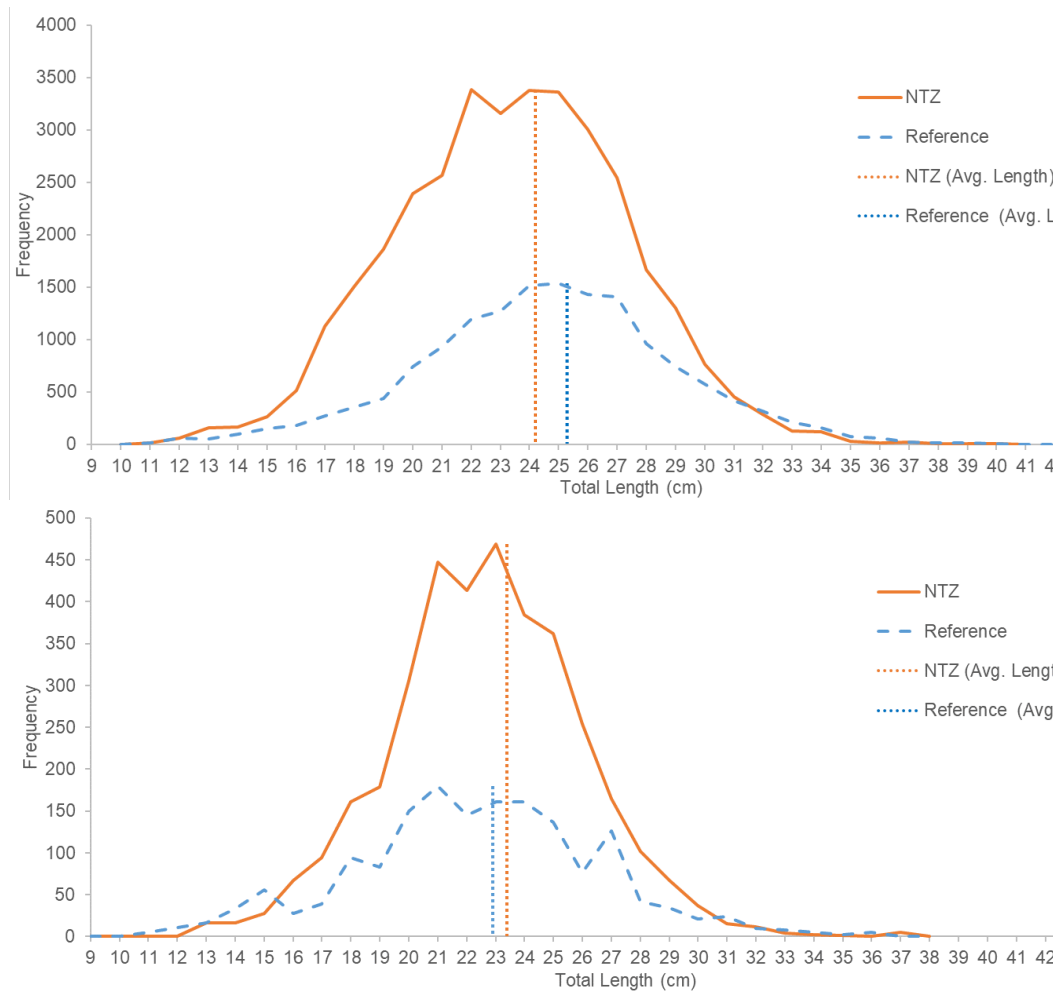
Figure 10. Length frequency distribution of all flounder caught in the NTZ and in the reference area, for 2006–2009 (top) and 2021 (bottom).

8.3.3. Fishing pressure

In Gotland, the commercial fishery targeting turbot did not start until the 1990s, while fishing for flounder has a longer tradition in the area. The commercial fishery at Gotska Sandön was virtually non-existent even before the establishment of the no-take zone, while landings of both turbot and flounder was relatively high in the reference area at the introduction of the regulation, even if the landings of turbot had decreased since the start of the fishery in the 1990s (Figure 11 and 12). Since the last evaluation, the landings of both turbot and flounder has decreased in the reference area. The calculated removal of turbot by the commercial fishery varied between 4–15 kg per square kilometer in 2006–2009 and decreased to 2 kg per square kilometer and year in 2019–2021 and the removal of flounder decreased from 12–27 in 2006–2009 to 6–12 kg per square kilometer and year in 2019–2021.
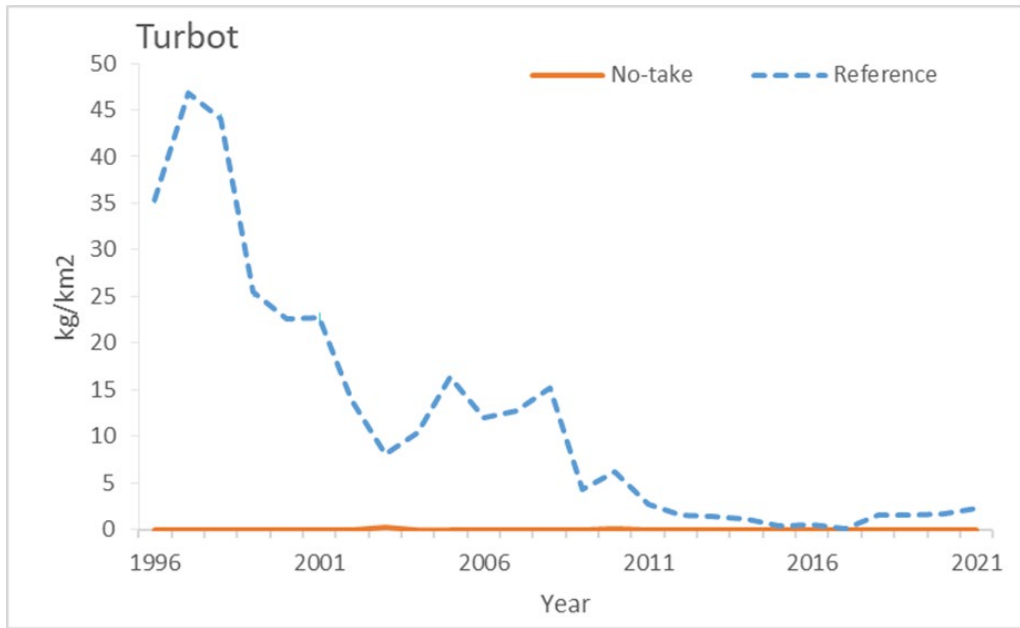
Figure 11. Fishing pressure estimated as commercial landings per unit area per year of turbot (Scophthalmus maximus) in the no-take zone around Gotska Sandön (almost zero) and the fished reference area outside Gotland.
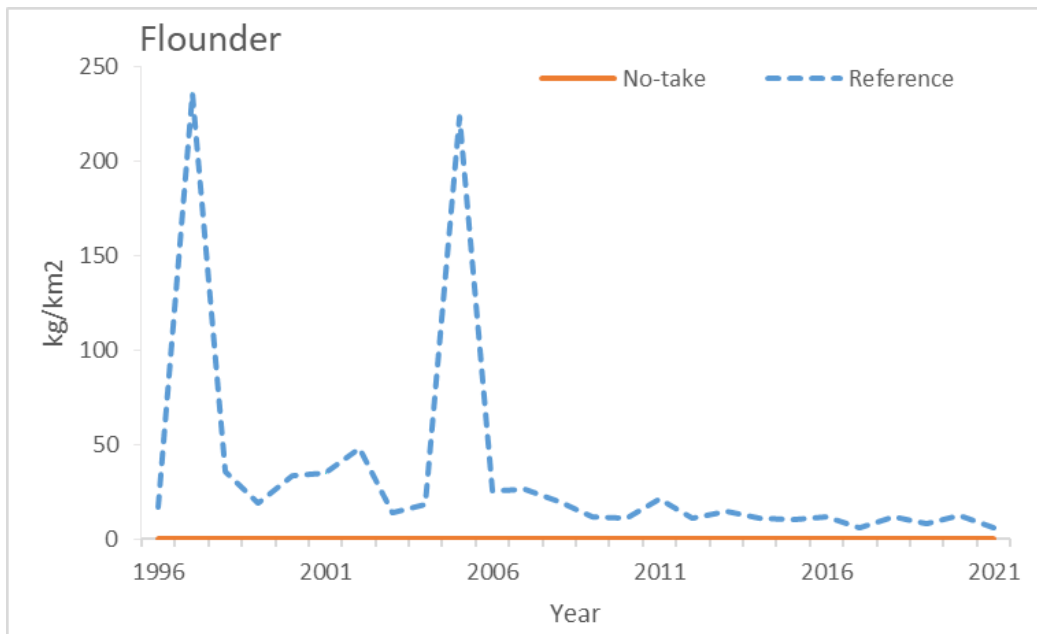
Figure 12. Fishing pressure estimated as commercial landings per unit area per year of flounder (Platichthys solemdali) in the Gotska Sandön no-take zone (almost zero) and the fished reference area outside Gotland.

8.3.4. Total local predation: seals and cormorants

The total predation from cormorants during nesting season calculated in the GIS analysis estimated that cormorants consumed between 395 to 520 kg fish per square kilometer in the reference area. In recent years, the predation from cormorants in the reference area has decreased and was estimated to 95–130 kg per square kilometer in 2019–2021. There were no colonies close to the no-take zone, thus no predation from cormorants was estimated there. Based on flounder making up 8% of the cormorant diet, we estimated the predation of flounder in the reference area to be 32–42 in 2006–2009 and 8–10 kg per square kilometer and year in 2019–2021.

The proportion of flatfish (assumed to be mostly flounder and turbot) in the diet samples from seal were low and thus the estimated predation in the study areas also low. During 2006–2009, the estimated predation from grey seal on flatfish varied between 0–2.3 kg per square kilometer and year in the no-take zone and 0.6–7.6 kg per square kilometer and year in the reference area. In 2018–2020, corresponding predation from grey seal was estimated to 0–2.3 kg in the no-take zone and 15–19 kg per square kilometer and year in the reference area.

8.4. Discussion

The current 2021 study replicated the initial monitoring methods in Florin et al. (2013), demonstrating, in line with our predictions, that densities of turbot and of flounder (large and small), continued to be higher in the Gotska Sandön NTZ compared to the fished reference area. The difference in fish densities, for both species, between each zone was relatively stable between the final year of initial monitoring in 2009 and later 2021 monitoring. As predicted, there was no difference in small turbot densities within the NTZ or reference area from 2006–2009 to 2021. However, in contrast to our predictions, there was a marked decrease in the densities of large turbot in the NTZ in 2021, a trend mirrored in the reference area. Large flounder densities also decreased compared to 2006–2009 in both areas, though the decrease was relatively marginal. Small flounder densities had a small decrease in the NTZ and there was a small increase in the reference area. The decrease in the number of large turbot and flounder was reflected in a lower average length for both species in 2021 compared to 2006–2009 in both areas, although the length reduction was greatest in the reference area. Unlike the earlier monitoring period where turbot and flounder were longer in the reference area, in 2021 there was no difference in length between the areas for turbot, and flounder were longer in the NTZ.

It is not clear why there was a sharp decrease in the densities of large turbot until 2021. The decrease occurs in both the fished reference area and the NTZ, so the decrease is not the result of an increase in commercial fishing pressure. These patterns may be the result of natural variation and more consistent sampling (i.e. more than the single 2021 survey) is needed to more reliable ascertain population trends. We can speculate, however that, given that commercial fishing pressure was considerably lower in 2021 compared to earlier monitoring periods, the large turbot densities in the reference area, all else being equal, should have increased. Predation alone also does not immediately appear to be the driver behind the decrease in large turbot in both areas, since we estimated that predation by seals has increased in the reference area but remained stable in the NTZ, in which case any decrease in fish numbers due to predation should have only occurred in the reference area. Taking lowered commercial fishing pressure and increased reference area predation together, we would expect the result to be stable or reduced densities of large turbot in the reference area and at the same time stable or increasing densities in the NTZ, particularly as fisheries catch of turbot is much higher than that by seals. The similar decrease in large turbot in both areas suggests that predation may have possibly have increased in the NTZ and that predation estimates for Gotska Sandön need to be improved. Alternatively, there may be other sources of mortality that are underestimated in both areas and other sources of natural variation, such as reduced reproduction rates, could be contributing.

The decreases in densities of large flounder in both areas and small flounder in the NTZ are relatively small. The results for 2021 are close to or within the range of previous individual years densities and may be the result of natural variation. However, we only have a single year of recent data and this snap-shot does not allow us to differentiate between small natural year to year variation in densities and a gradual long-term trend of decreasing densities compared to earlier monitoring. As it stands, the lower numbers of flounder in the NTZ are not overly concerning but suggest that there is value in at least two more years sampling in this monitoring period, as this would allow a better assessment of whether there is a downward trend or the decrease is explained by natural variation.

Catch data from Gotska Sandön before the NTZ came into effect allowed us to compare fish densities in the NTZ for before protection (2003–2006), immediately after protection (2007–2009) and after long-term protection (2021). Densities of large turbot, small flounder and large flounder were highest in the period immediately after protection, whilst small turbot had higher densities before protection. Somewhat surprisingly, densities of both species, large and small, within the NTZ in 2021 were either lower than or had no measurable difference

compared to densities before protection. The lower densities in 2021 compared to before protection add further weight to the suggestion that predation in the NTZ is being underestimated or that we are missing other sources of mortality or other sources of natural variation e.g. reproduction rates.

Our results come with two main caveats, 1) we only have a single year of recent data from 2021 and a large gap between monitoring, so fish density trends lines should be treated with caution. Ideally, monitoring should be carried out more regularly and ideally not as single year snapshots with large gaps between samples, 2) the data available on mortality from predation and other sources are uncertain or unknown.

Recreational fishing occurs in the reference area (Florin et al. 2013) and represents a largely unquantified source of turbot and flounder mortality. Prior to 2013, at the Baltic Sea scale, there is some evidence that recreational fishing took a similar amount of flounder and turbot as the commercial fisheries and that on Gotland recreational fishing pressure was greatest around the reference area (Thörnqvist et al. 2009, Florin et al. 2013). There have been no recent estimates of recreational fishing effort in the study area. Recreational fishing in the NTZ should be zero, however, the level of compliance with the prohibition of fishing is unknown and there is no dedicated enforcement of NTZ regulations. Whilst it is assumed that compliance is high, this has not been evaluated and there remains a risk that there is illegal fishing occurring. Assessing the recreational fishing compliance directly in the NTZ would allow this source of mortality to be more confidently excluded or conversely, included if it is occurring.

Information about spatio-temporal dynamics of cormorants in the Baltic Sea and on Gotland is limited, e.g. to what extent birds stay in the area after the breeding season and the importance of Gotland as a stop-over for migrating cormorants breeding in other areas. The limited available data does suggest that the overwintering cormorant population is much smaller than during the breeding season population. In Gotland winter bird count data collected by volunteers (European network for bird monitoring, www.fageltaxering.lu.se), the number of observed cormorants has varied between 1000 and 3000, thus, assuming a population size 4 times the number of breeding pairs, the number overwintering seems to be at a minimum, less than 10% of the breeding population (Engström 2001). It is unclear if any predation occurs by overwintering cormorants in the NTZ area and a better understanding of this mortality component is needed, however, in the case of large turbot, cormorant are not likely to directly affect densities as they do not feed on these size classes (see below).

Further complicating estimates of mortality from predation, is that predator diet data we used here is preliminary and not specific to the NTZ. For cormorant, based on Florin et al. (2013), we used an estimate of 8% flatfish in the diet; however, there is much uncertainty on cormorant diet in the area. For example, a study on the post-breeding data of great cormorants on eastern and northern Gotland 2018–2019 concluded that sticklebacks and gobies dominated the diet, followed by eelpout, flatfish and cod (Hansen, Svanbäck and Lundström, in prep.). The results are based on fish remains, i.e. otoliths and other bones, identified in cormorant pellets collected from roosting sites July–November. The prey group flatfish was almost completely dominated by flounder (93% of the otoliths, whereas 7% could not be identified to species level and only 0.3% was identified as turbot). The cormorant consumption of flounder was dominated by fish between 15 and 25 cm. Each pellet could be considered as the daily prey consumption (Zijlstra and Vaneerden 1995). In the analysed material, flounder occurred in 30% of the pellets, and each of those pellets contained on average 2.7 fish. Thus, an assumed cormorant population of 10,000 birds would consume approximately 8000 flounder specimens per day during the post-breeding season (0.3 x 10,000 x 2.7).

Similarly, for seals, there is a large variation in diet component estimates. In this study we used the estimate of 22% flatfish in seal diet based on Florin et al. (2013) and a previous study on grey seal diet in the Gotland area estimated the weight proportion of flatfish in the diet to make up a similar proportion (20%; Hansson et al. 2017). However, preliminary results from an ongoing study on the prey choice of grey seals off Gotland suggests that the contribution of flatfish to the diet has decreased and that flounder and turbot sum up to <1% of the weight of consumed prey. Instead, the more recent study suggest a importance of sprat, herring and cod as prey resources, representing almost 90% of the diet, by weight (Pers. comm. K. Lundström, SLU). At such low predation levels we would have expected that flatfish numbers would have increased in recent monitoring–the exact opposite of what our results show.

The various ecological roles of grey seals and great cormorants and their impact on turbot and flounder in the fishery-closure area are still uncertain. Thus, conclusions about their impact and to what extent these drive the decrease in large turbot and the decease of turbot and flounder lengths in the NTZ should be treated with caution, primarily due to limited information on the feeding ecology and spatial dynamics of both seals and cormorants in the study region. Future monitoring of the development of fish stocks in the fishery-closure area would benefit from an ecosystem-based approach where, among other factors (e.g. flounder and turbot food availability), monitoring of predators and their local diets was integrated into the overall monitoring of the NTZ.

8.5. References

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Appendix

Table A9.1

Species	Common Name	Swedish Name	2006 NTZ	Ref	2007 NTZ	Ref	2008 NTZ	Ref	2009 NTZ	Ref	2021 NTZ	Ref
Platichthys spp.	European flounder	Skrubbskädda	5188	4397	11460	5191	8096	3824	9508	1797	3607	1653
	Large Flounder		4049	3672	9265	4568	5734	3177	7152	1431	2740	1136
	Small Flounder		1139	725	2194	623	2362	646	2357	367	868	517
Gadus morhua	Atlantic Cod	Torsk	351	1142	278	1848	558	1667	245	1727	50	463
Myoxocephalus scorpius	Shorthorn sculpin	Rötsimpa	644	100	117	78	17	129	317	123	1433	2210
Scophthalmus maximus	Turbot	Piggvar	581	137	366	487	601	456	460	411	244	310
	Large Turbot		34	36	61	103	73	55	116	42	13	8
	Small Turbot		547	102	305	384	527	401	344	369	231	302
Neogobius melanostomus	Round Goby	Svartmunnad smörbult	0	0	0	0	0	0	0	0	0	178
Zoarces viviparus	European eelpout	Tånglake	22	0	22	11	28	11	17	0	0	0
Clupea harengus	Atlantic Herring	Strömming	6	11	29	0	6	11	0	1	17	0
Taurulus bubalis	Long-spined bullhead	Oxsimpa	0	11	0	0	0	6	0	0	0	6
Cyclopterus lumpus	Lumpsucker	Sjurygg	6	0	0	0	6	0	6	0	0	0
Hyperoplus lanceolatus	Great sand eel	Tobiskung	6	0	0	0	0	0	0	0	6	0
Gasterosteus aculeatus	Three-spined stickleback	Storspigg	0	0	0	0	0	0	0	1	0	6
Ammodytes tobianus	Lesser sand eel	Kusttobis	0	0	0	0	0	0	0	6	0	0
Salmo trutta	Brown Trout	Öring	0	0	0	6	0	0	0	0	0	0
Syngnathus typhle	Broadnosed pipefish	Tångsnälla	0	0	0	6	0	0	0	0	0	0
Myoxocephalus quadricornis	Fourhorn sculpin	Hornsimpa	1	0	0	0	0	0	0	0	0	1

9. No-take zone for pike and perch in Licknevarpefjärden, Baltic Sea

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Photo: Ulf Bergström

Summary

Licknevarpefjärden in Östergötland on the Baltic coast is Sweden's oldest no-take zone - it has been mostly closed to fishing since 1979. A very restricted artisanal fishery was allowed until the early 2000s, but after that the area has been fully closed. The no-take zone (NTZ) is situated in a pristine area, with little disturbance from human activities. Altogether, this makes Licknevarpefjärden a unique case in the Swedish Baltic Sea.

To follow the development of the fish community in Licknevarpefjärden over time, two different sampling methods were used; surveys by multimesh gillnets (2005, 2013, 2018 and 2020) and by angling (2005, 2017, 2018, 2020 and 2021). The nearby Kvädöfjärden, which is environmentally similar to Licknevarpefjärden, although open to fishing, was used as a reference area. The results from the gillnet survey showed that in 2005, i.e. 25 years after the area was closed, the abundance of pike was three times higher in the NTZ than in the reference area. For perch, there was a similar pattern, where both the total abundance and the abundance of adult perch (>20 cm) were higher in the NTZ compared to the reference area. In 2020, the abundances of pike and adult perch (> 20 cm) had decreased significantly in both the no-take and reference area. These declines coincided with a marked increase in abundance of grey seal and great cormorant, why their potential impact on fish populations in the area was investigated separately.

In the angling survey, total biomass of pike was substantially larger in the NTZ compared to the reference area. There was also a significant decrease in the biomass of the largest pike (>70 cm) in both the NTZ and the reference area from 2005 to 2020. Simultaneously, there was an increase in smaller pike (<70 cm) after 2005, but only in the NTZ.

Fish community composition differed significantly between the NTZ and reference area, but this difference decreased over time, which resulted in more similar fish communities in the two areas in 2018. In 2020, the fish communities diverged again, mainly driven by increases of certain cyprinid species in the NTZ (common bleak, ide, roach and white bream). Moreover, food-web structure, in terms of functional groups of fishes (cyprinids, non-piscivores and piscivores), changed in the NTZ during 2005-2020, with a marked decrease in the proportion of piscivores and increase in the proportion of cyprinids. In 2020, the proportions between functional groups were similar in the NTZ and the reference area.

Thus, during the first year of study in 2005, 25 years after the establishment, there was a pronounced difference in the fish community between the NTZ and the reference area, characterized by higher densities and biomasses of large predatory fish in the NTZ. The densities of predatory fish decreased over time in both areas,

but more in the NTZ, and the difference in community composition between the areas thus gradually eroded. These changes coincided with an increase in both grey seal and cormorant abundances, starting in the early 2010s. The estimated removals by these predators of both perch and pike, as estimated from abundance and diet composition data, were high. For pike, the removal by grey seal was substantially higher than by cormorant, whereas for perch cormorant predation was higher. These results are in line with previous results showing that cormorant may have an effect on perch populations, whereas grey seal by their preference for larger prey may affect pike. In summary, the results of the study suggest that small and solitary no-takes zones may not provide sufficient protection for piscivorous fish communities in areas of the Baltic Sea coast where abundances of top predators, such as grey seal and cormorant, are high.

9.1. Background

Licknevarpefjärden is an enclosed coastal bay of 3.7 km^2 situated at the Swedish Baltic Sea coast, in the archipelago of Östergötland (Fig. 1). The area is a nature reserve with little development and local physical disturbance, and a local motorboat ban. The nature reserve was established in 1979, when also a fishing ban was implemented (Bergström et al., 2019), originally to protect nesting white-tailed eagle from disturbances. A small artisanal fishery was allowed until the beginning of the 2000s when the area wasere completely closed to fishing, i.e. turned into a no-take zone (NTZ) (Bergström et al., 2016). This makes Licknevarpefjärden a rather unique case in the Baltic Sea, as nor commercial or recreational fishing are allowed, or have been allowed, for decades (Bergström et al., 2019).

Previous studies in Licknevarpefjärden have shown that the local populations of predatory fish, such as northern pike (*Esox lucius*) and Eurasian perch (*Perca fluviatilis*), hereafter pike and perch, differ from nearby areas where fishing is allowed, with both higher densities and larger individuals of fish in the NTZ (Edgren 2005, Bergström et al. 2007). The biomass of perch and pike was 3-4 times higher than in other similar coastal areas, and there were also indications of a top-down effect from predation on cyprinids, which occurred in lower densities than other areas with a corresponding level of eutrophication (Bergström et al. 2019). The growth rate of pike in the area was lower than in fished reference areas, likely caused by density-dependent intraspecific interactions (Edgren 2005, Bergström et al. 2007).

The NTZ in Licknevarpefjärden differs from most of the other case studies in this report in that there is no data available from before the NTZ was established, nor

from the early years after the area was closed to fishing. The first survey data is from 2005, i.e. 25 years after closure. The goal of this study was to investigate how stable the fish community in the area was after these first surveys. While no fishing has taken place in the NTZ, the fishing pressure in the nearby reference area has likely remained on a fairly constant level over time. On the other hand, the fish community in this part of the Swedish coast are subject to predation from increasing populations of both grey seal and cormorant, which may in some areas have effects on densities of prey species, including pike and perch (Östman et al. 2013, Hansson et al. 2017, Bergström et al. 2022). In this study, we follow the development of fish populations in a NTZ and a nearby reference area over time, focusing mainly on the large predatory species pike and perch, but to some extent also on different functional groups and the fish community as a whole. This setup thus provides an opportunity to follow the isolated effect of the predators (in the NTZ) with the combined effect of predation and fishing (in the reference area). Being able to separate the effects from fishing and predation is important to develop knowledge for an ecosystem-based approach to management in the Baltic Sea.
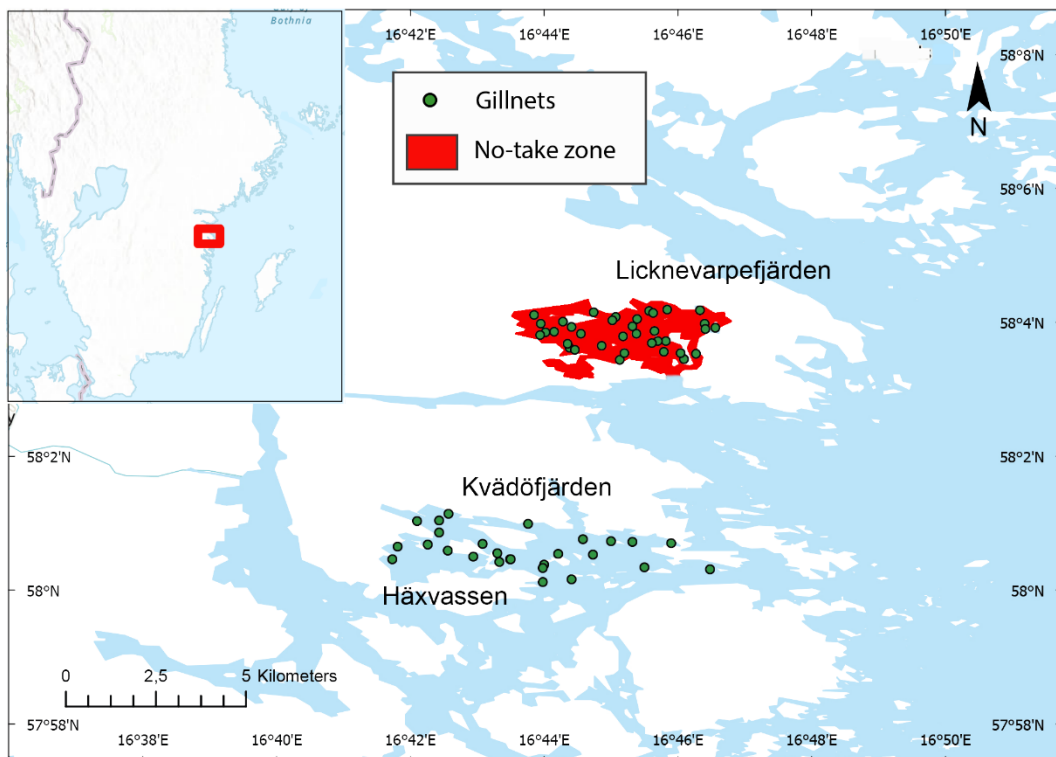

Figure 2. The location of the NTZ Licknevarpefjärden (in red) to the north and the reference area Kvädöfjärden, open to fishing, in the south. Green circles denote the sampling stations of the multimesh gillnet survey.

9.2. Methods

9.2.1. Definitions of goals of the study

Goals, objectives, indicators and success criteria (GOIS) were defined for the long-term study of populations of pike and perch in the NTZ (Table 3).

9.2.2. Fish surveys using gillnets

There is currently no regular monitoring program for fish in the NTZ Licknevarpefjärden, but the Swedish University of Agricultural Sciences, Department of Aquatic Resources (SLU Aqua) has conducted a number of quantitative surveys in the area to collect information about the status of the resident fish populations (Bergström et al., 2016a). Standardised surveys using multimesh gillnets were conducted in late August/beginning of September in 2005, 2013, 2018 and 2020 at fixed stations at a maximum depth of 10 m in the NTZ Licknevarpefjärden and in an adjacent reference area called Kvädöfjärden (Fig. 1). The gillnets were 45 m long and were composed of 9 different sections of mesh sizes ranging between 10-60 mm. Sampling was conducted with one net during one night at each station; 27-29 stations were fished per year in Licknevarpefjärden and 35-36 in Kvädöfjärden. Disturbances during sampling occurred during a few occasions, and these stations were removed from further data analyses (Table 1).

To investigate potential changes in food web structure over the years in the NTZ and reference area, all fishes were categorised into functional groups; cyprinids, non-piscivores and piscivores, according to classifications in Bergström et al., 2019.

Table 2. The number of sampling stations included in the gillnet surveys in Licknevarpefjärden (NTZ) and Kvädöfjärden (reference area).

Gillnet survey Year	Licknevarpefjärden	Kvädöfjärden
2005	27	36
2013	29	35
2018	28	36
2020	29	36

9.2.3. Angling survey

To specifically monitor the pike population and its development over time, an additional survey method was used. Population estimates based on gillnet survey data does not always represent the population accurately, as catchability of pike in gillnets is generally poor. Therefore, pike was also sampled by angling. All angling was conducted from boat during mid April- mid May in 2005, 2017, 2018, 2020 and 2021 (Table 2). The angling survey was conducted in the NTZ Licknevarpefjärden and in an adjacent reference area open to fishing, called Häxvassen, located in the inner parts of Kvädöfjärden. The fishing equipment consisted of spinning rods using artificial baits, and effort was estimated as rod hours to calculate catch biomass per unit effort (BPUE).

Table 3. The number of fishing days in the angling surveys in Licknevarpefjärden (no-take zone) and the reference area Häxvassen, in Kvädöfjärden, plus total number of angling hours in brackets. To achieve as comparable data as possible, only sampling conducted in mid April to the end of May were included in the study. Häxvassen was not sampled in 2020.

Angling survey		
Year	Licknevarpefjärden	Häxvassen
2005	9 (88.3)	3 (15.83)
2017	3 (45.5)	3 (34)
2018	2 (32)	2 (32)
2020	1 (16)	- (0)
2021	1 (16)	1 (16)

9.2.4. Predation by cormorants and grey seals

To evaluate the potential impact of predation by cormorant and grey seal on the fish populations in Licknevarpefjärden, the removal of different fish species by these top predators was estimated. The estimations were based on predator count data (derived from local inventories of cormorant nests and from the national monitoring program of grey seals) combined with diet composition and consumption rates of cormorants and grey seals following methods in Hansson et al., 2018. For cormorant, the energetic need per individual was estimated to 500 g per day and the total number of feeding days per year was set to 185 days for adults and 140 days for juveniles (Hansson et al., 2018). Each nest within foraging distance (20 km) from the study areas was considered to host two adults, one subadult and two juveniles. Bird densities were estimated applying a kernel density function in ArcGIS. Based on these estimates, the total fish consumption (kg/ha) per year was calculated.

To assess the biomass of the focal species (pike and perch) in cormorant diet, proportion of these species in regurgitated prey was estimated. Samples were collected near a cormorant colony at Lilla Källskär, which is the colony that is located closest to Licknevarpefjärden, during April-June in 2018 and 2020. Regurgitates were washed and food items, or parts, were separated. Some fish could be identified visually and when possible length of the fish was noted to the nearest 0.5 cm. Otoliths and other bone remains were dried and identified under a dissecting microscope. Otolith length and width were measured, and by applying species-specific regressions on otolith measurements, lengths and weights of the fish could be attained. For fish where length had been measured, regressions were applied to attain estimates of fish weight. From this data, the biomass of each species in the total cormorant diet was estimated. The biomass proportion of pike and perch in the diet was estimated to 3.9 % and 21.5 %, respectively. These numbers, together with cormorant abundance and energetic needs were used to calculate the total extraction of pike and perch in the NTZ and the reference area.

For grey seal, an estimate of 4800 g fish per day was used (Hansson et al., 2018) and multiplied with the estimated seal density in the area. A maximum foraging distance of 60 km from the haulout sites was applied (Sjöberg & Ball 2000, Oksanen et al. 2014) when estimating densities using a kernel density function in ArcGIS. There were no local estimates of diet composition for grey seal, so instead estimates from the central parts of the Stockholm archipelago, which is a similar environment, was used. According to that study the biomass of pike was estimated to 20 % and perch to 46% of the diet (Svensson, 2021).

9.2.5. Environmental data

Both water transparency and temperature can affect catch data and fish community composition. This information was used in the evaluation of comparable sampling days (temperature, angling survey) and in explaining potential differences or changes over time in the two different sites (water transparency). Water temperature was measured for each sampling station for gillnets and once a day for the majority of the angling survey days. Secchi depth was used to estimate water transparency and was measured on site level once a day for the gillnet surveys (but not for each sampling station).

9.2.6. Statistical analyses

Potential differences in the development of fish communities between Licknevarpefjärden (the NTZ) and Kvädöfjärden (the reference area) were tested using the gillnet survey data. Fish smaller than 12 cm were excluded from all statistical analyses and graphical presentations according to standard methods for this type of survey, due to poor catchability of small fish in the gillnet gear used. A

multivariate principal coordination analysis (PCA) was conducted using the R package vegan (Oksanen et al., 2020). Prior to this analysis, species occurring ≤ 3 times in the data were removed to avoid disproportionally large influence by rare species on the outcome. Similarly, pelagic schooling species such as sprat and herring were excluded from further analyses, as catches are usually very variable. To test for significant differences in fish community composition between the NTZ and reference area, a PERMANOVA was performed using the adonis function in the vegan package, which included 'site', 'year' and the interaction between 'site' and 'year'. The interaction between site and year was the key aspect of the study; a statistically significant ($p < 0.05$) interaction term indicates that the there is a difference between the NTZ and the reference area, and thus that the protection from fishing has an impact.

To evaluate differences in the development of the focal fish species (pike and perch) between the NTZ and the reference area, generalized linear models (GLM) from the package stats was used, with gaussian distribution (R Core Team, 2020). Log(x+1) transformations were applied for both species. Similarly to the PERMANOVA, these models included 'site', 'year' and the interaction term 'site * year'. Fish abundance data was derived from the gillnet surveys and the included years were 2005, 2013, 2018 and 2020. For pike, all sizes were analysed together whereas perch was categorised into adults (> 20 cm) and subadults (< 20 cm).

Population development of pike was further evaluated using data from the angling survey. To account for potential variation in catchability attributed to water temperature and timing of spawning, data from mid-April to mid-May was included in the analyses. The pike were categorised into two size classes; very large (>70 cm) and large (<70 cm). To investigate differences in development over time in the NTZ and the reference area, a GLM with factors 'site' and 'year' and "site*year" was applied with negative binomial distribution. Biomass per unit effort (BPUE) was used as the response variable, because this was considered more representative than CPUE.

To evaluate potential predation effects by seal and cormorant on the pike populations in the NTZ and the reference area, GLMs with negative binomial distribution were used with the same size classes of BPUE of pike from the angling surveys as described above. 'Site' and 'period' were used as fixed factors. 'Period' consisted of two categories: 'before' and 'after' where the first one included all samples from 2005 (prior to the large-scale establishment of predators in the area) and the latter consisted of 2017, 2018, 2020 and 2021 (after the establishment of predators). To evaluate potential predation effects on perch populations in the areas, a similar design was used, but with data from the gillnet surveys, with 2005 and

2013 defined as the period before establishment of predators and 2018 and 2020 as the period after the establishment of predators.

All models were evaluated using ANOVA type III tests by the Anova() function in the car package. Prior to the analyses, normal distributions of predictor and response variables were evaluated based on diagnostic plots. All analyses were conducted in R version 4.0.1 (R Core Team, 2020).

Table 4. Overview of the results in relation to the goals and objectives of the NTZ study including references to figures and tables where results are present.

Objectives	Indicators	Success criteria	Survey methods	Results	References
No decline in stock size of pike in the NTZ	CPUE/BPUE	CPUE/BPUE is higher in the NTZ than the reference area	Multimesh gillnet, angling	CPUE of pike higher in the NTZ compared to the reference area, but an overall decrease over time in the NTZ and reference area (gillnet data). BPUE of pike much higher in NTZ than reference area in angling survey, but a decrease of pike >70 cm and an increase of pike <70 cm over time	9.3.1 Fig. 2, 3
A diversified age/size structure of pike	Size structure	CPUE of large individuals is higher in the NTZ than the reference area	Angling	There was a decrease in BPUE of pike >70 cm and an increase of pike <70 cm over time in the NTZ. In the reference area, BPUE was low and without any obvious trend	9.3.1 Fig. 4
No decline in stock size of perch in the NTZ	CPUE	CPUE is higher in the NTZ than the reference area	Multimesh gillnet	CPUE of perch >20 cm was higher in the NTZ compared to the reference area. There was no difference in CPUE of perch <20 cm between the NTZ and the reference area	9.3.1 Fig. 2
A diversified age/size structure of perch	CPUE/BPUE	CPUE of large individuals is higher in the NTZ than the reference area	Multimesh gillnet	CPUE of adult perch decreased over time in both the NTZ and the reference area. Very large individuals of perch (> 40 cm) were only present in the NTZ	9.3.1 Fig. 2, 4
No decline in proportion of predatory fish.	CPUE/BPUE	Proportion of predatory fish in the fish community is higher in the NTZ than the reference area higher in the NTZ	Multimesh gillnet	There was a decrease in the proportion of predatory fish in the NTZ.	9.3.6 Fig. 9

9.3. Results

9.3.1. Abundances of pike and perch

The catch per unit effort (CPUE) in the gillnet survey for pike and adult perch showed a negative trend both in the NTZ and in the reference area, although there was an overall increase in the CPUE for perch (all sizes) in 2020 that was mainly attributed to an incease in subadults (< 20 cm, Fig. 2).

Catches of pike in the gillnets were higher in the NTZ compared to the reference area (chisq = 21.3, Df = 1, p < 0.001 for factor 'site'). The development over time did not differ between the areas (interaction 'site * year'p = 0.26), but there was an overall decline in CPUE of pike in both areas (GLM, chisq = 39.1, Df = 3, p < 0.001 for factor year).

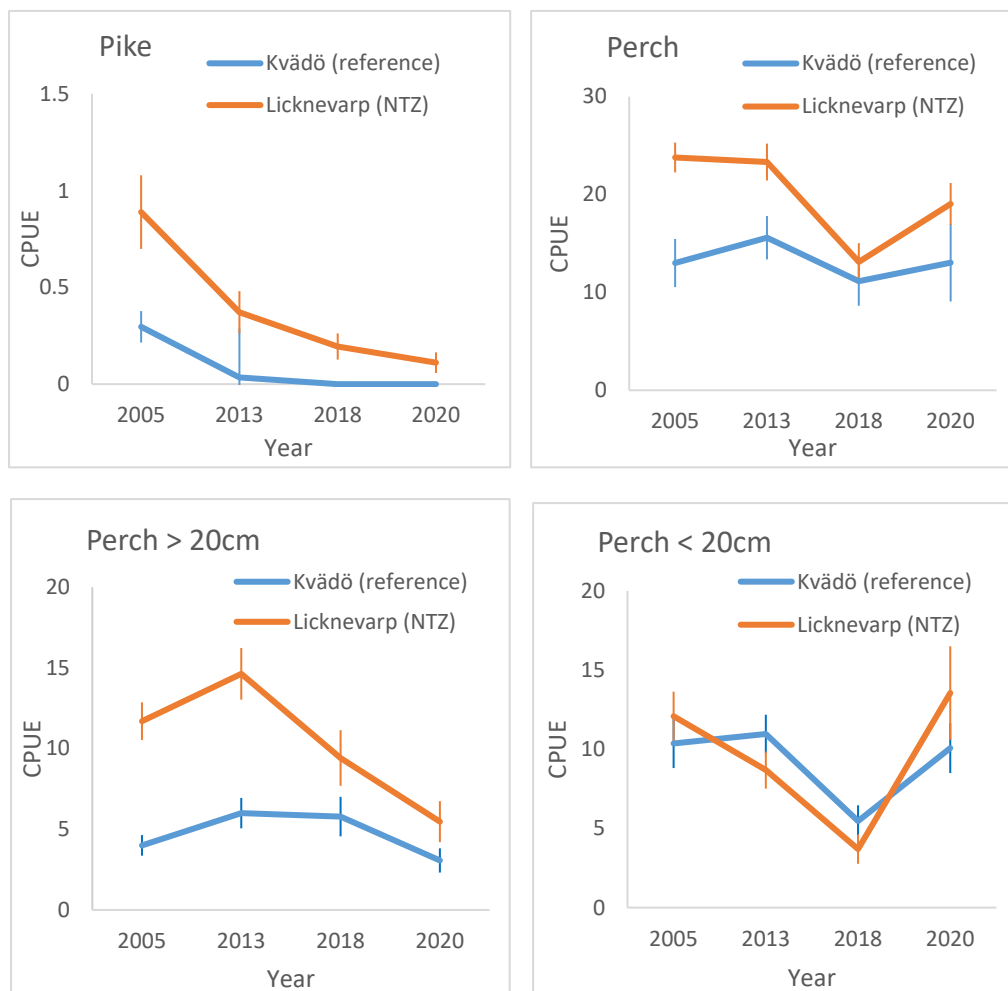*Figure 3. CPUE (total ± SE) of pike (total), perch (total), adult perch (> 20 cm) and subadult perch (< 20 cm) in the multimesh gillnet survey in the NTZ Licknevarp and the reference area Kvädö 2005, 2013, 2018 and 2020.*

For adult perch (> 20 cm), CPUE was also higher in the NTZ compared to the reference area (chisq = 38.2, Df = 1, p < 0.001) and there was a significant difference among years (chisq = 10.0, Df = 3, p = 0.02). For subadult perch (< 20 cm), there were no significant differences between sites (chisq = 0.51, Df = 1, p = 0.48), but among years (chisq = 31.0, Df = 3, p < 0.001). The development over time did not differ between the areas for either adult or subadult perch (interaction 'site * year'; p adults = 0.24, p subadults = 0.52).

In the angling survey, the mean size of pike was considerably higher than in the gillnet survey (mean length 63 versus 33 cm). For this gear too, BPUE of very large pike (> 70 cm) was significantly higher in the NTZ compared to the reference area (GLM; chisq = 17.02, Df = 1, p < 0.001; Fig. 3). There was a trend towards a difference in BPUE of very large pike between years, but this was not significant (GLM; chisq = 6.6, p = 0.16). For large-sized pike (< 70 cm), there were significant differences in BPUE both between the NTZ and the reference area (GLM; chisq = 39.9, Df = 1, p < 0.001) and among years (GLM; chisq = 30.0, Df = 4, p < 0.001; Fig. 3). Due to the limited number of years with data, it was not possible to test for the interaction between 'site' and 'year'.
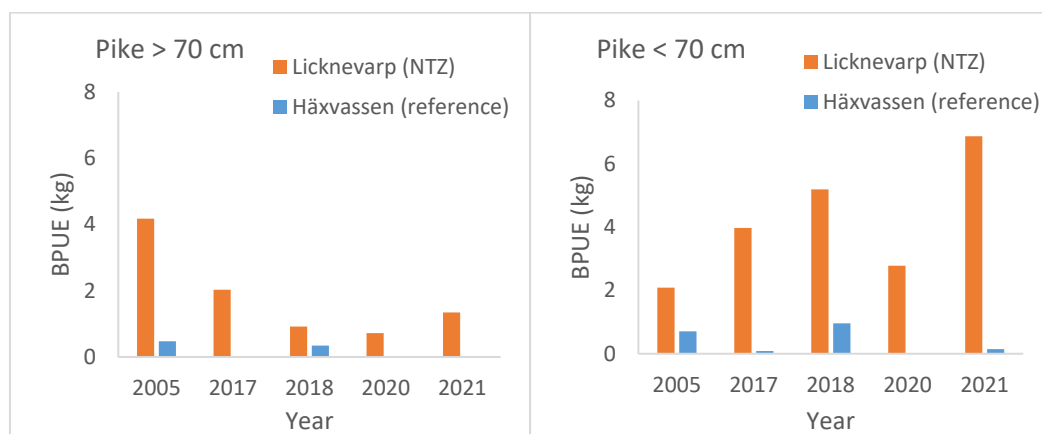
Figure 3. BPUE (mean) of very large (> 70 cm) and large (< 70 cm) pike in the angling survey in the no-take zone Licknevarpefjärden and the reference area Häxvassen. In 2020, Häxvassen was not sampled.

Length distributions of pike and perch did change over time, although this was not tested statistically (Fig. 4). Biomass of respective size class was always highest in the NTZ for perch, but this difference was smaller in 2018 and 2020 (Fig. 4).
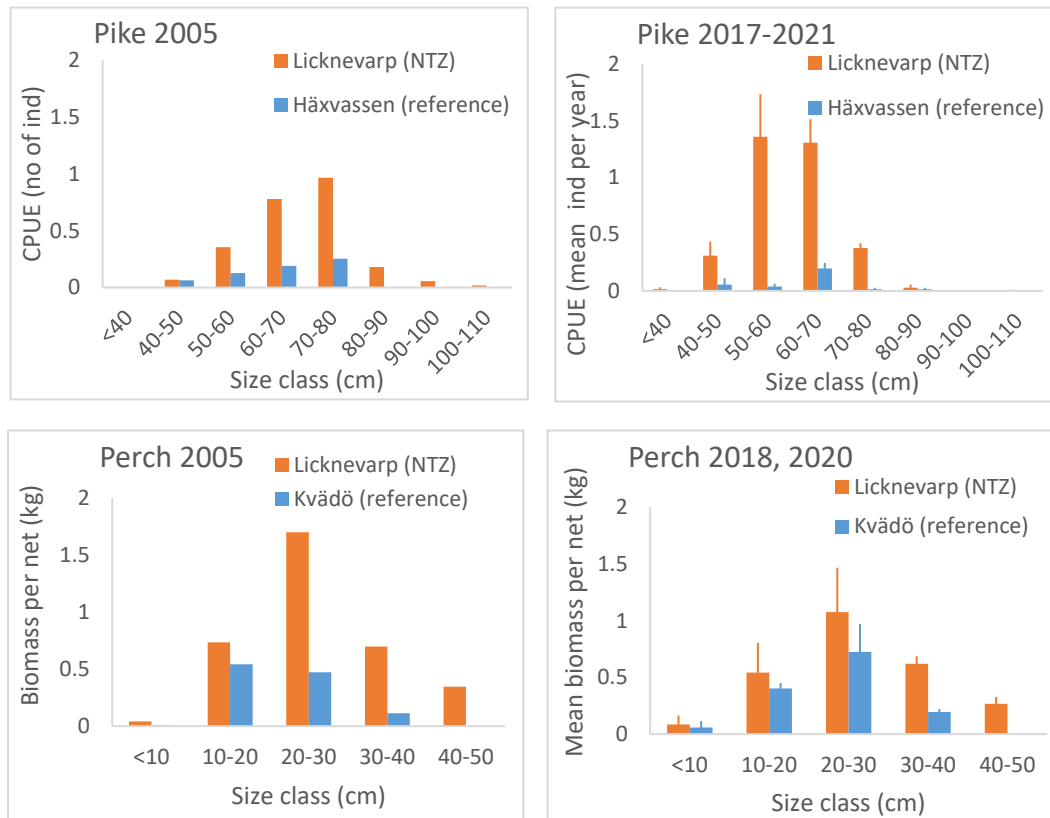
Figure 4. Length histograms (total values and mean ± SE) of pike in the angling survey and perch in the gillnet survey in the no-take zone and the reference area, before (2005) and after (2017, 2018, 2020 and 2021) the establishment of large predators in the area.

9.3.2. Fish community composition

The results from the PERMANOVA and the multivariate PCA analysis (Fig. 5) showed that the development of the fish community composition differed between the NTZ and the reference area (PERMANOVA: Df = 3, MSq = 0.44, p < 0.001; Fig. 5). Further, while community composition was fairly stable in the reference area over time, a strong change could be seen after 2013 in the NTZ (Fig. 5). This difference is mainly driven by a decrease of pike and perch and an increase in cyprinid species (Fig. 5, 6).
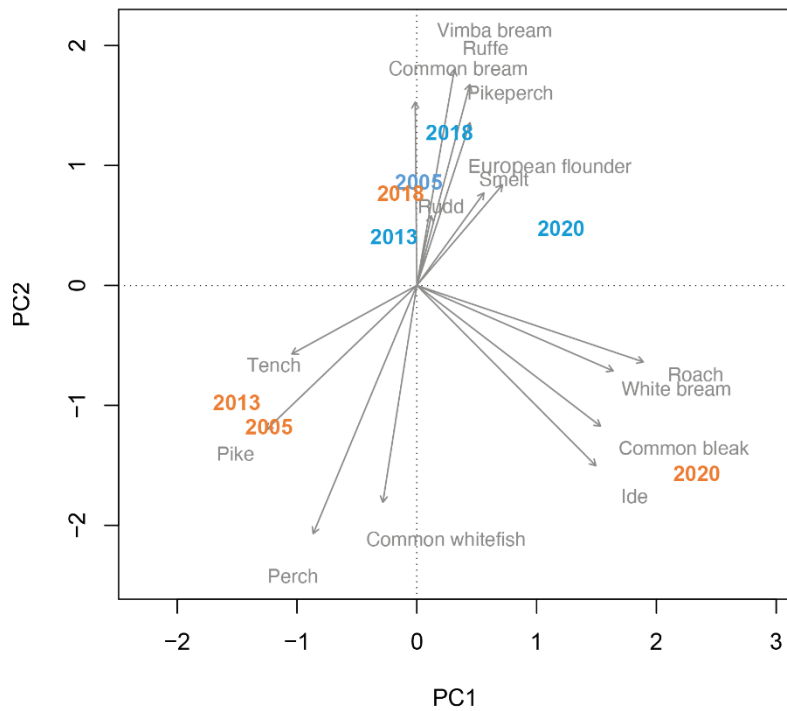
Figure 5. Development of fish community composition (mean values per year caught in the gillnet surveys in the NTZ Licknevarpefjärden (orange) and reference area Kvädöfjärden (blue), in a PCA biplot. Each year represents the fish community composition for that particular year, and the closer together the years are arranged in the plot, the more similar they are. Arrows show which species contribute the most to the dissimilarities and in what direction.
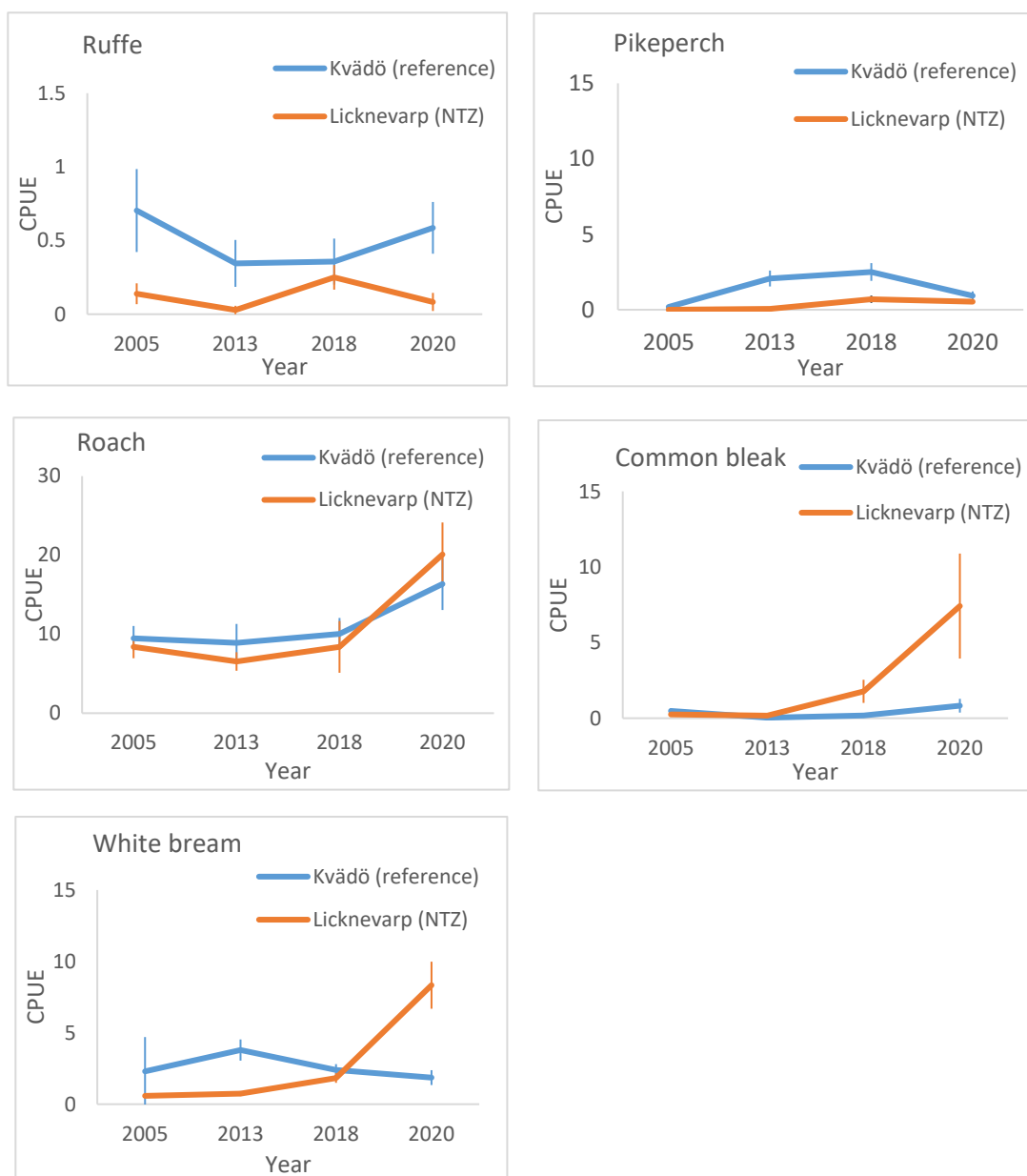
Figure 6. CPUE (total ± SE) of ruffe, pikeperch, roach, common bleak and white bream in the multimesh gillnet survey in the NTZ Licknevarpefjärden and the reference area Kvädö 2005, 2013, 2018 and 2020.

9.3.3. Predation by grey seal and cormorant

The estimated removal of pike and perch by cormorant has increased seven-fold and by grey seal six-fold when comparing the most recent years (2016-2020) to the years preceding the first survey in the area (2000-2004; Fig. 7). There were obvious species-specific differences regarding both prey and predators; grey seal consumed a higher amount of pike, whereas cormorant consumed a higher amount of perch. Overall, the estimated total consumption of perch by seal and cormorant were higher than the total consumption of pike (Fig. 8). Based on the estimations, the predation pressure from both seals and cormorants were generally higher in the NTZ compared to the reference area.
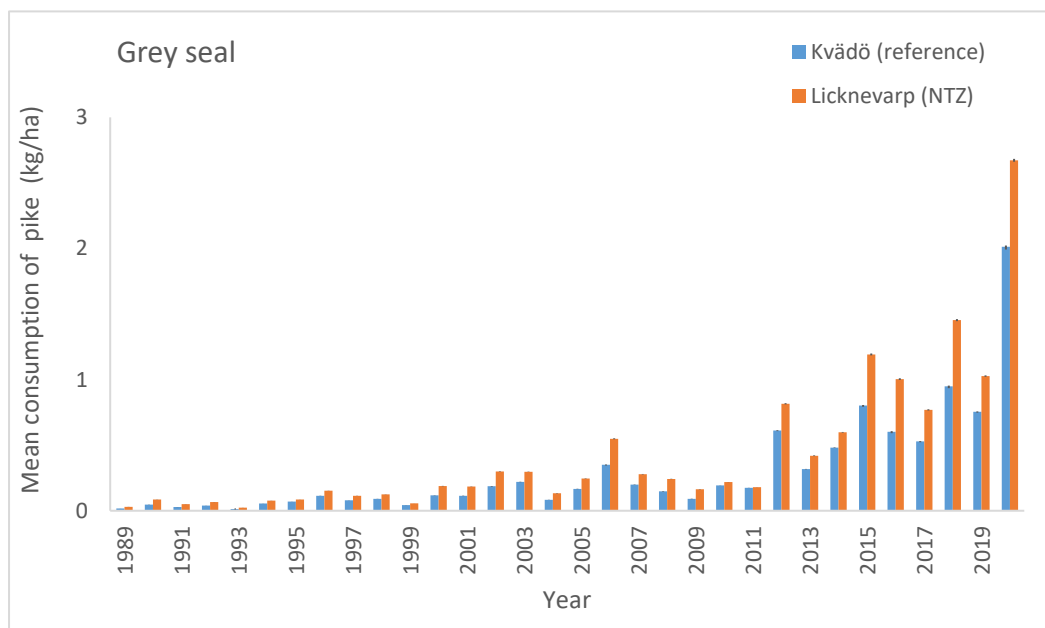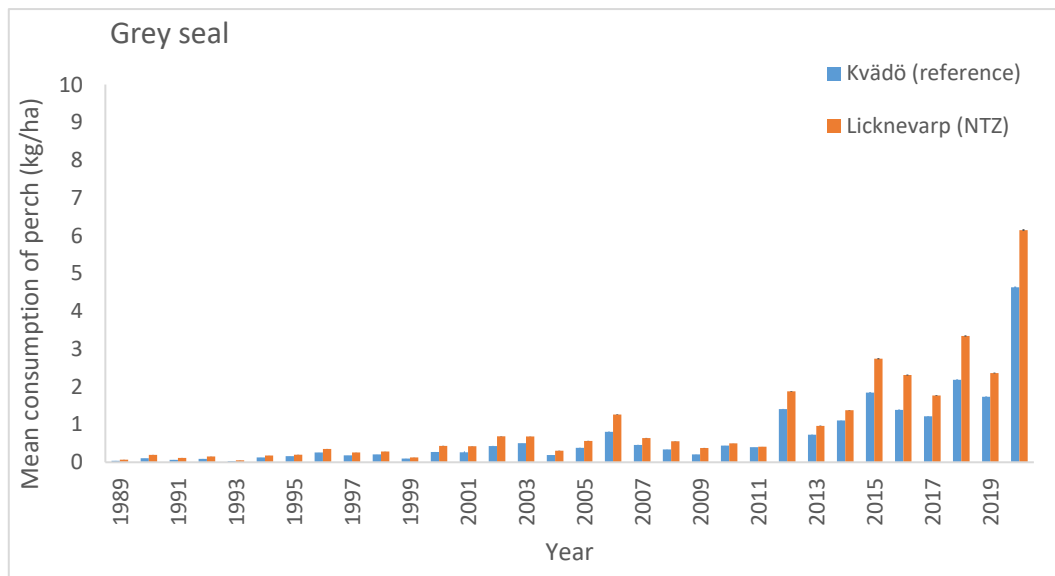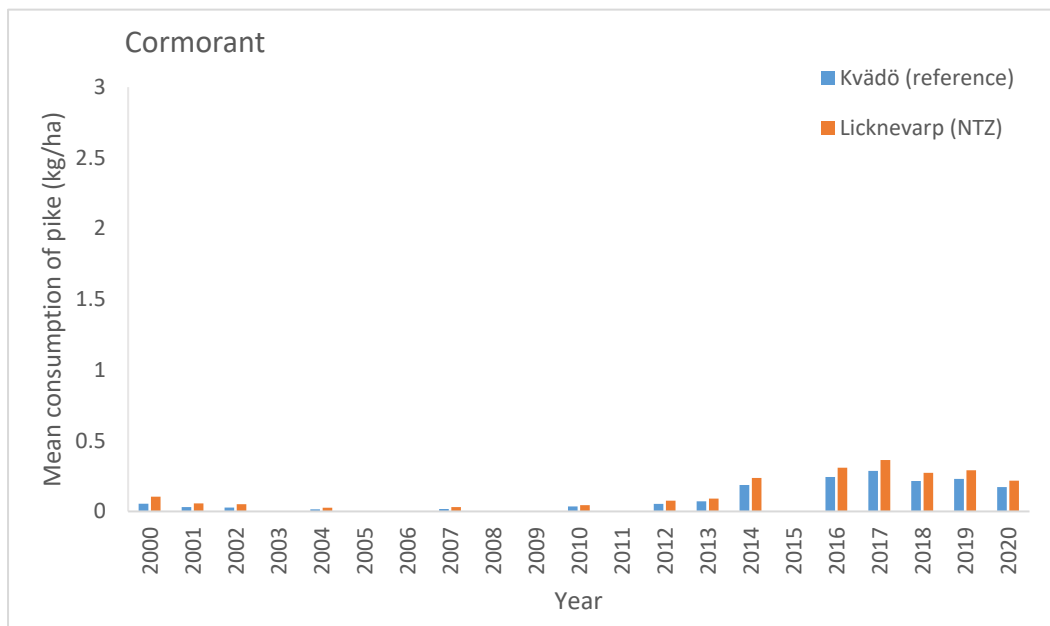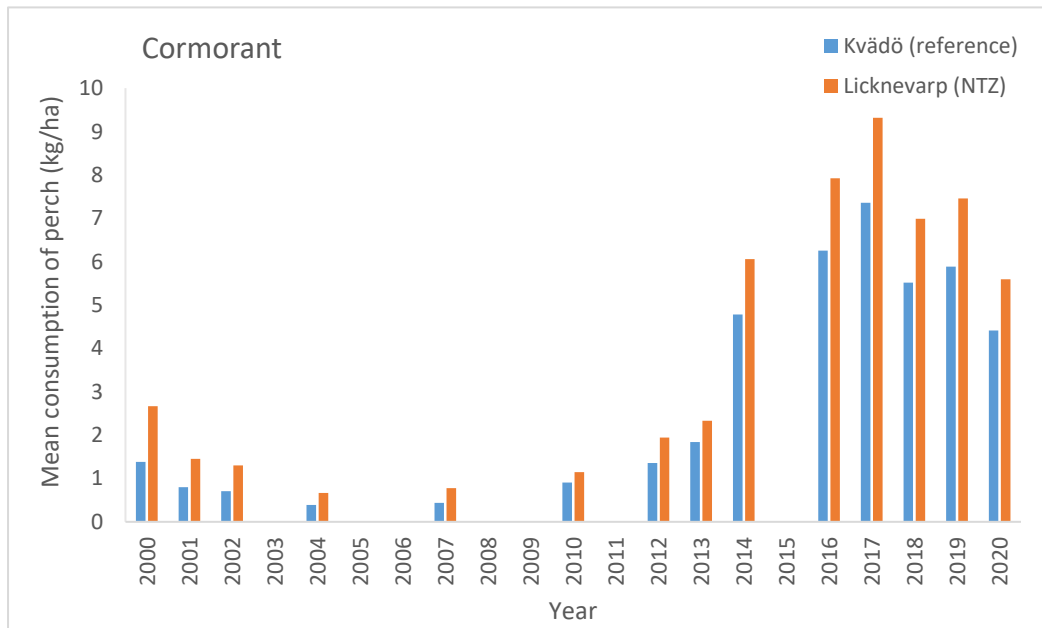
Figure 7. Estimated consumption of pike and perch (kg per hectare) by grey seal and cormorant in Licknevarpefjärden and Kvädöfjärden. Consumption estimates are based on count data at nearby nesting and haulout sites combined with diet composition data and estimates of bioenergetic needs.

According to the diet composition estimates of regurgitated prey for cormorants, the most commonly consumed species were eelpout, perch, roach, ruffe and shorthorn sculpin (Fig. 8).
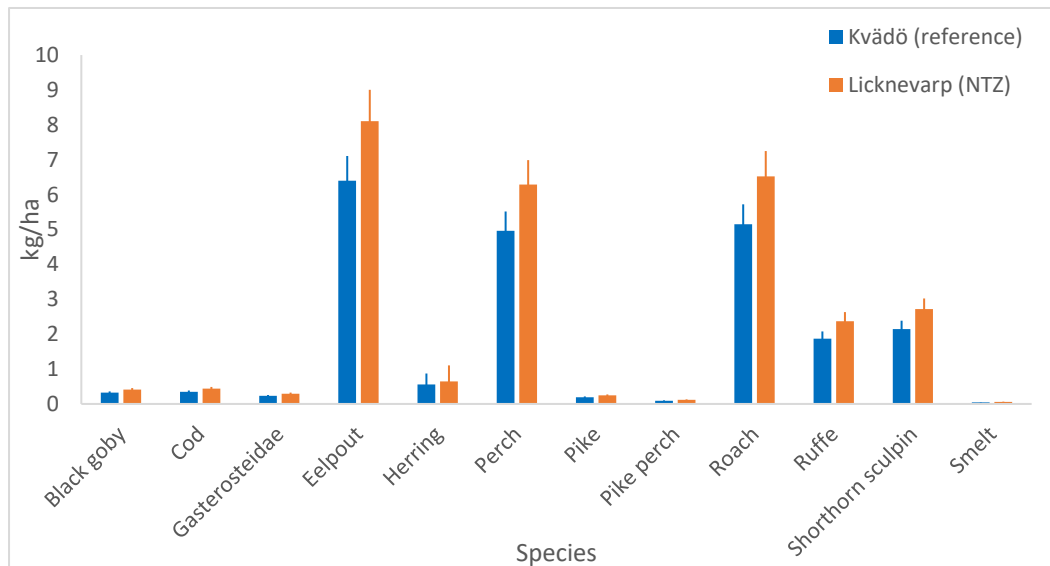
Figure 8. Estimated removal of different fish species from the no-take zone (Licknevarpefjärden) and the reference area (Kvädöfjärden) by cormorant, based on the number of nests within 20 km of the area and diet composition based on regurgitated prey (mean values ± SE) collected in March-August 2018-2020 from the closest cormorant colony, located at Lilla Källskär.

For very large pike (>70 cm) in Licknevarpefjärden there was a decrease in BPUE between the time periods before and after predator establishment, i.e. 2005 compared to the later years (GLM, chisq = 5.1, Df = 1, p = 0.02. Catches were also always higher in the NTZ compared to the reference area (GLM; chisq = 17.4, p < 0.001). For large pike (<70 cm), BPUE was also higher in the NTZ compared to the reference area (GLM, chisq = 28.2, df = 1, p < 0.001), but there was a significant increase in BPUE in the period after predator establishment (GLM, chisq = 12.6, Df = 1, p < 0.001), The increase in angling catches of pike <70 cm may potentially be connected to a more active feeding behaviour as a consequence of the decrease in the abundance of the largest pike individuals.

For adult perch (> 20 cm), CPUE in the latter period, after predator establishment, was significantly lower (GLM, chisq = 5.45, df = 1, p = 0.02). CPUE was consistently higher in the NTZ compared to the reference area (GLM, chisq = 40.9, df = 1, p < 0.0001). For subadult perch (< 20 cm), there was also a significant decline in CPUE (GLM, chisq = 10.6, df = 1, p = 0.001) whereas there were no site-dependent differences (GLM, chisq = 0.25, df = 1, p = 0.61).

9.3.4. Differences in food web structure

In the NTZ Licknevarpefjärden, there was a trend towards an increase in abundance of cyprinids in the gillnet survey, whereas the abundance of piscivores decreased. For the reference area, the proportions of the functional groups remained constant over time (Fig. 9). The proportion of piscivores relative to cyprinids was higher in the NTZ in the first years, but in 2020 the two areas had very similar proportions of the two groups.
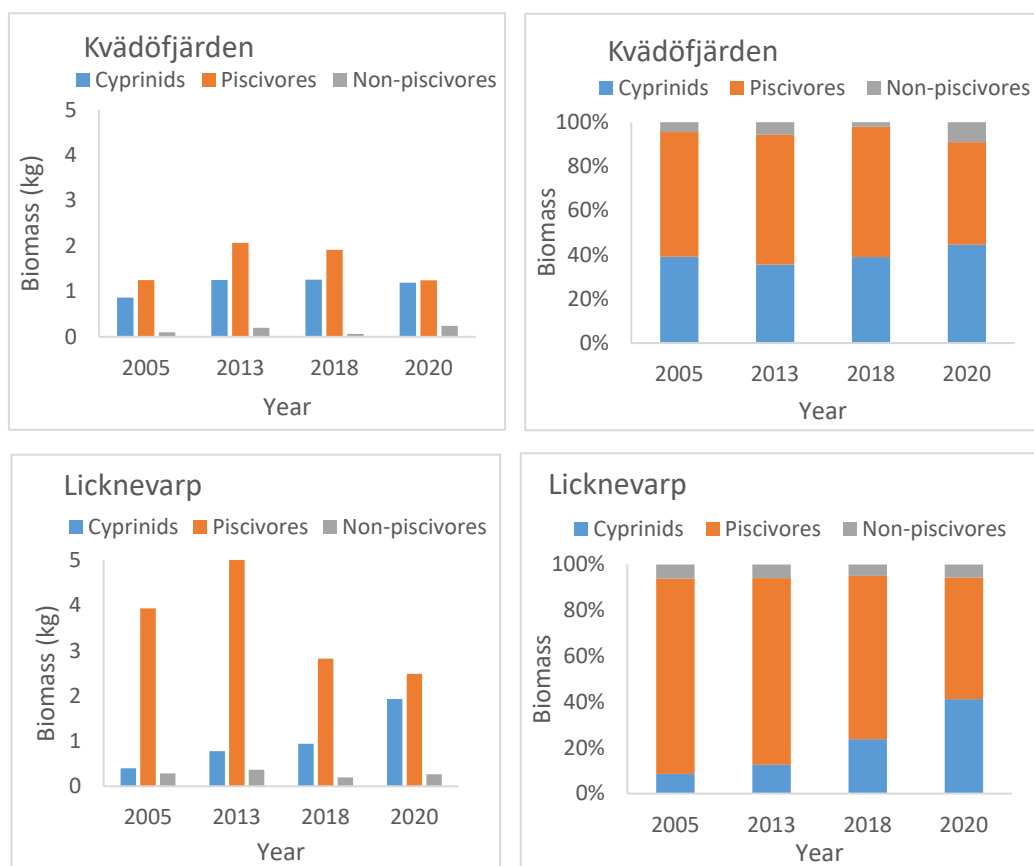
Figure 9. Total BPUE per year in Kvädöfjärden (reference area) and Licknevarpefjärden (NTZ) of different functional groups of fishes. Functional groups are categorised according to Bergström et al. (2019).

9.3.5. Environmental factors

Water temperature and water transparency were monitored in the two study sites over time, in order to see if differences in environmental conditions may have affected the development of the fish populations. Both transparency and temperature were similar in Licknevarpefjärden and Kvädöfjärden (Fig. 10, 11). No trends were observed in the two areas over the course of the study, suggesting that water transparency and temperature dynamics can be excluded as potential explanations for the changes in fish community composition.
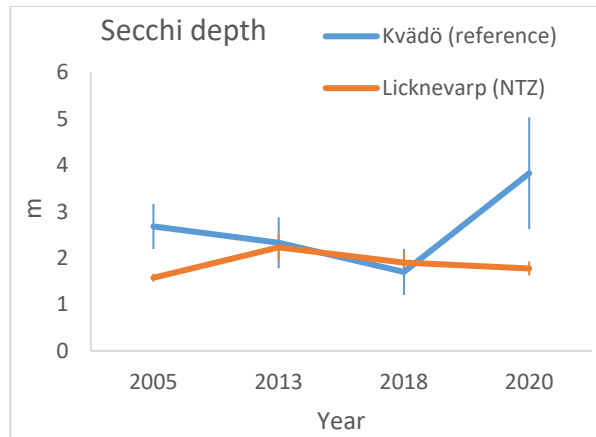
Figure 10. Water transparency (mean values ± SD) in Kvädöfjärden (reference area) and Licknevarpefjärden (NTZ).
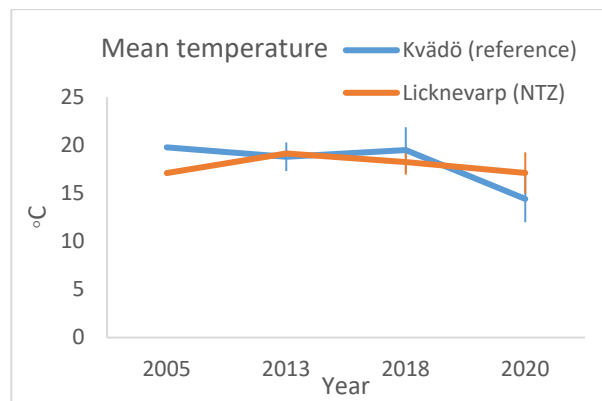
Figure 11. Temperature (mean values ± SD) in Kvädöfjärden (reference area).and Licknevarpefjärden (NTZ).

9.4. Discussion

The NTZ in Licknevarpefjärden hosts larger populations of pike and perch, the two dominating piscivorous fishes of the Baltic Sea coast, than the reference area which is open to fishing. This difference can likely be attributed to the fishing ban that has been in place since 1979. Hence, a top-down driven food-web structure dominated by predatory fish has prevailed in this area (Bergström et al. 2007; Berggren et al. 2022), with lower abundances of prey species like cyprinids (Bergström et al. 2019). For pike, the denser populations are associated with a lower growth rate than in other areas, likely as an effect of interference and competition for food, but the pike in the NTZ were still larger than in areas open to fishing due to their higher age (Bergström et al. 2007, Berggren et al. 2022). For perch, a similar pattern with larger individuals in the NTZ than other coastal areas has also been evident (Bergström et al. 2016b), but for this species the growth rate was actually higher in the NTZ despite the higher abundances

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(Bergström et al. 2007). This may potentially indicate that the lack of fishing pressure favours fast-growing individuals of perch.

However, during the course of the study period (2005-2020), there were negative trends in the abundances of pike and adult perch, both in the NTZ and the reference area. Catch per unit effort (CPUE) of adult perch in the gillnet survey was higher in the NTZ than in the reference area, but this difference decreased over time from being 2.9 times higher in 2005 to 1.8 times higher in 2020. For pike, catches in the gillnet survey were higher in the NTZ compared to the reference area, and declined in both areas over time. Since pike is a sedentary ambush predator (Broughton & Fisher, 1981; Edgren, 2005), gillnet catches were generally low. A similar pattern was, however, also reflected in the angling survey, which caught more pike, especially larger individuals. Here, BPUE of very large (>70 cm) and large-sized pike (32-70 cm) were significantly higher in the NTZ compared to the reference area. Interestingly, in this survey there was a decrease in BPUE of very large pike, and a simultaneous increase in the BPUE of large pike. In contrast, there is an apparent decline in large-sized pike over time in the gillnet survey. We therefore hypothesise that pike <70 cm become more active when their larger conspecifics are lost from the population, decreasing the risk of being eaten Grimm & Klinge, 1996; Nilsson & Brönmark, 1999)why the increased catches of pike <70 cm in the NTZ may be a result of a behavioural change. Such a change in foraging behaviour could thus mask a potential decrease also of pike <70 cm in the angling survey.

The overall fish community composition and food-web structure also displayed pronounced changes over the period in the NTZ compared to the reference area. This was mainly caused by the decline of predatory fish in combination with an increase in cyprinid species. The increase in cyprinids such as roach, bleak and white bream is likely an indirect effect of the lower predation pressure from pike and perch, indicating a loss of top-down control of the fish community following the decline in pike and perch.

Licknevarpefjärden has been closed to fishing during the last four decades. The first fish surveys in the area, in 2005, showed that there were pronounced differences in the main target species of fisheries at the Swedish Baltic Sea coast, perch and pike, compared to nearby reference areas that were open to fishing. Abundances were higher, especially of large individuals, with biomasses of these species being 3-4 times higher in the NTZ compared to the reference area. In the subsequent gillnet and angling surveys, performed in 2013 and 2017-2021, there were, however, clear declines in the abundance of both perch and pike. There are no indications of violations against the fishing ban, why the observed changes in the fish community cannot be attributed to an increase in fishing pressure. Moreover, there are no obvious trends in environmental variables such as temperature and visibility, and it is therefore unlikely that these factors would contribute to the observed changes.

The most obvious change in the area is the strong increase in abundance of the piscivorous top predators grey seal and great cormorant. Top predators can have substantial effects on prey densities, both in terrestrial and aquatic systems, and subsequently also alter food-web

structures (Berger et al. 2008; Baum & Worm 2009; Eklöf et al. 2020). In the Baltic Sea, grey seal and cormorant may in some areas have a substantial impact on populations of coastal fish, such as perch and pike (Östman et al. 2013; Hansson et al., 2018; Svensson 2021; Bergström et al. 2022). Both grey seal and cormorant have increased substantially in the study area since the beginning of the 2010s. In 2013, a cormorant colony established on an island which is well within foraging distance of both Licknevarpefjärden and Kvädöfjärden, although somewhat closer to the NTZ. Around the same time, the number of grey seals started increasing at haulout sites within foraging distance of the two areas. The counts of seal was 88 % higher and cormorant 71 % higher in 2016 compared to in 2000.

The estimated removal rates of pike and perch by grey seal and cormorant included substantial sources of uncertainty, both in predator density and diet composition estimates. Nevertheless, the large estimated removals clearly suggest that the impact can be substantial, and is a plausible explanation to the observed declines. The production of perch has been estimated to around 20 kg/ha and of pike to around 7 kg/ha in productive coastal habitats in the Baltic Sea (Hansson et al. 2018). The estimated total removals by seal and cormorant in the area amounts to 20-50% of the potential production, which for other fish populations in the Baltic have been shown to give rise to adverse population impacts (Hansson et al. 2018). Furthermore, the consumption rates for the study areas are likely underestimated, since they are based on interpolations of seal and bird densities from nesting and haulout sites. These calculations assume that the daily feeding migrations are non-directional, while in practice both predators can be expected to forage specifically in fish-rich habitats (Paillisson et al. 2004; Oksanen et al. 2014), such as the NTZ and the reference area. Also, the estimations of fish species composition in the diet are based on only two years of sampling (2018 and 2020), which may add some bias to the results. Regurgitated prey samples were collected during the nesting season only, which may underestimate the sizes of fish in the overall diet. This is because these samples reflect what adult cormorants feed their chicks, which generally is smaller fish than preferred by the adults. For grey seal, diet composition estimates were not available from the area and instead data from the Stockholm archipelago was used in the calculations.

In a similar study conducted in a NTZ in the Stockholm archipelago (see separate section of this report) the estimated consumption of pike and perch by grey seal and cormorant were very close to the ones of the current study. Also in the NTZ in Stockholm, there were strong indications of a negative impact from predation on both pike and perch. These two studies, where the NTZs have enabled a separation of fisheries and predation-related effects, provide examples of the potential impact of the growing seal and cormorant populations on coastal predatory fish. While cormorant seems to mainly have an effect on perch populations, grey seal by their preference for larger prey may impact also pike. In the current study, there was a decline of very large pike individuals in the NTZ. This corresponds well to a recent study that shows how seal predation seems to be the main factor responsible for the ongoing declines observed in large pike along the Swedish Baltic coast (Bergström et al. 2022). In summary, the results from this study indicate that small NTZs may not have the capacity to buffer impacts on coastal predatory fish populations that are induced by grey seal and cormorant predation in areas where

these top predators are abundant. Given the central importance of perch and pike for ecosystem functioning in the Baltic Sea coastal ecosystem (Donadi et al. 2017; Eklöf et al. 2020), these results suggest that the current level of predation pressure from seal and cormorant in some areas may decrease the provisioning of important ecosystem services. Such services are for example the capacity of the coastal ecosystem to buffer the negative effects from eutrophication, as well as the production of food and opportunities for recreation. Potential solutions to this conflict of management objectives may be to hunt, scare and/or exclude the predators from sensitive areas, such as NTZs, or to increase stocks of alternative prey, such as herring. Strengthening the populations of perch and pike along the coast, through continued implementation of NTZs, spawning closures and other management measures, may also make them more resilient to the effects of predation.

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10. No-take zone and partially protected areas to rebuild cod in the Kattegat

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Photo: Mattias Sköld

Summary

The Kattegat cod stock has been overfished during the last 30 years, and the spawning stock biomass has been very low since the turn of the millennium. Although cod quotas were reduced in the early 2000, cod mortality remained high, and the stock continued to decline. Cod is caught both as a targeted species and as by-catch in the extensive mixed fishery for fish and Norway lobster *Nephrops norvegicus*. Historically, cod fishing was carried out on spawning grounds in the south-eastern Kattegat during the spawning period. Despite the scientific advice by ICES in 2002 to close the cod fishery, only limited fishing restrictions were introduced, and the cod stock continued to decline. To supplement the catch regulations with more effective measures with the aim of rebuilding the cod stock, Swedish and Danish researchers outlined a proposal to close the remaining functional cod spawning grounds to fishing. The purpose of establishing a large year-round No take zone (NTZ) surrounded by a partially protected area (PPA) was to prohibit targeted as well as by-catch fishery of cod on the spawning grounds, and to displace fisheries by-catching cod from areas where mature cod in the Kattegat aggregate outside the spawning season. The Swedish and Danish ministers of fisheries agreed on measures based on the researchers' proposal, however, with major modifications including reduced coverage of the NTZ and the PPA when the measures were enforced in 2009.

The primary aim of this study is to evaluate the effects on the Kattegat cod stock and how the spatial measures including the NTZ have contributed to the recovery of cod in the Kattegat. The secondary aim of the report is to assess the long-term ecosystem effects of the Kattegat NTZ. Here we have analyzed effects on the fish assemblage of fishing prohibition and the potential recovery from bottom trawling of the benthic communities, as bottom trawling and its associated physical disturbance of the seabed has ceased in the NTZ.

The introduction of the NTZ in combination with the PPAs, where only selective gears avoiding cod were allowed, together with an overall decrease in fishing effort initially showed promising results with signs of recovery in the age structure and biomass of the cod stock. However, when the effort regulations were removed in 2016 and the landing obligation was implemented in the Kattegat, the cod stock showed a rapid deterioration to an all-time low observed in 2021. High discard rates, which have affected both recruits and adults of cod since 2016, despite the landing obligation, is considered the major driver behind the halted recovery and additional depletion of the stock. Even with the NTZ in place to protect the main spawning areas, while still allowing for mixed fisheries of Norway lobster and flatfish to continue in other parts of the Kattegat including the PPAs, additional effective management actions to reduce the bycatch of both juvenile and adult cod are needed. Management and stakeholders also need to be patient since rebuilding of an overfished stock takes time and depends on an increased survival of recruits to achieve an age distribution characterizing a healthy cod stock. However, the situation for the Kattegat cod gives at present little hope for this development. To our knowledge, no recovery plan exists, and cod has been defined as a "by-catch species" implying lowered management ambition for this stock that in the past was one of the most important fisheries in the Kattegat.

The local increase in flatfish and Norway lobster biomass in the NTZ relative to control areas, however, show that closing an area has the potential to protect mobile species in an open system such as the Kattegat. In addition, changes in the benthic community composition indicate recovery from physical disturbance by bottom trawling following the enforcement of the NTZ. The dominant benthic taxa, i.e., burrowing brittle stars, have declined in the NTZ which likely results from increased predation by flatfish and Norway lobster, which have increased following implementation of the NTZ.

10.1. Background

The Kattegat cod stock has shown clear signs of overfishing during the last 30 years, and the spawning stock biomass (SSB) has been very low since the turn of the millennium. Since 2000, he International Council for the Exploration of the Sea (ICES) has made the assessment that the cod stock in the Kattegat is below the limit reference point B_{lim} for spawning stock biomass (SSB) and that fishing mortality is too high. (ICES 2021).

Although cod quotas were reduced in early 2000, cod mortality remained high, and the stock continued to decline. The reason for the continued high fishing mortality was that cod was caught both as a targeted species during the spawning period, and despite quotas being full, by-caught and discarded in the extensive mixed fishery for demersal fish and Norway lobster *Nephrops norvegicus*. Historically, targeted cod fishing was carried out on spawning grounds in the south-eastern Kattegat during the first quarter both by Denmark and Sweden, and for several years >70% of the annual Swedish cod quota in the Kattegat was caught in the targeted fishery for spawning cod (Vitale et al. 2008). The stock decline coincided with the disappearance of large spawning aggregations (Cardinale and Svedäng 2004). The high fishing mortality for cod in the Kattegat led ICES to advise a total stop of cod catches in 2002. Despite the clear scientific advice, insufficient fishing restrictions were introduced, and the cod stock continued to decline.

In order to augment catch regulations with more effective measures to rebuild the cod stock, Swedish and Danish researchers outlined a proposal to close the remaining functional cod spawning grounds to fishing (Hjelm et al. 2008). The purpose of establishing a large year-round no take zone (NTZ) surrounded by a partially protected area (PPA) was to prohibit cod fishing on the spawning grounds and displace fisheries by-catch of cod from areas where mature cod in the Kattegat aggregate both during and after spawning (Vitale et al. 2008; Börjesson et al. 2013). In addition, the intention of the proposal was to allow fisheries on other species such as *Nephrops* and flatfish to continue in other areas after the cod spawning season. The Swedish and Danish fisheries ministers agreed on measures based on the researchers' proposal, however, with major modifications including reduced coverage of the NTZ and the PPA. These protection measures in the Kattegat were enforced on 1 January 2009. The measures put in place consisted of a NTZ with adjacent PPAs that are subject to restrictions in fishing both in terms of time and fishing gear (for details see Figure 1). Denmark and Sweden are the Member States mainly involved in fishing activities in the Kattegat. However, German vessels are

allowed to fish outside territorial waters in the Exclusive economic zones (EEZ) and this includes parts of the NTZ and PPA otherwise closed to the Swedish and Danish fisheries. The German vessels fishing in the Kattegat have quotas for cod, flatfish and *Nephrops* and are bottom trawlers active mainly in the southwest part outside the Swedish territorial border (delineated in Figure 1).

The primary aim of this study is to evaluate the effects on the Kattegat cod stock and how the spatial measures including the NTZ have contributed to the recovery of cod in the Kattegat. However, the effects of the NTZ on the highly mobile species cod cannot be interpreted without the perspective of the cod stock, and the overall management of the fishery in the Kattegat.

The secondary aim of the report is to assess the long-term ecosystem effects of the Kattegat NTZ. Here we will analyze effects on the fish assemblage of fishing prohibition and the potential recovery from bottom trawling of the benthic communities, as bottom trawling and its associated physical disturbance of the seabed has ceased in the NTZ (section 1.7).

Parts of this study has been published in the ICES Journal of Marine Science (Sköld et al. 2022), but we give in here in addition a thorough background of the management history and summarize the assessment done by ICES.
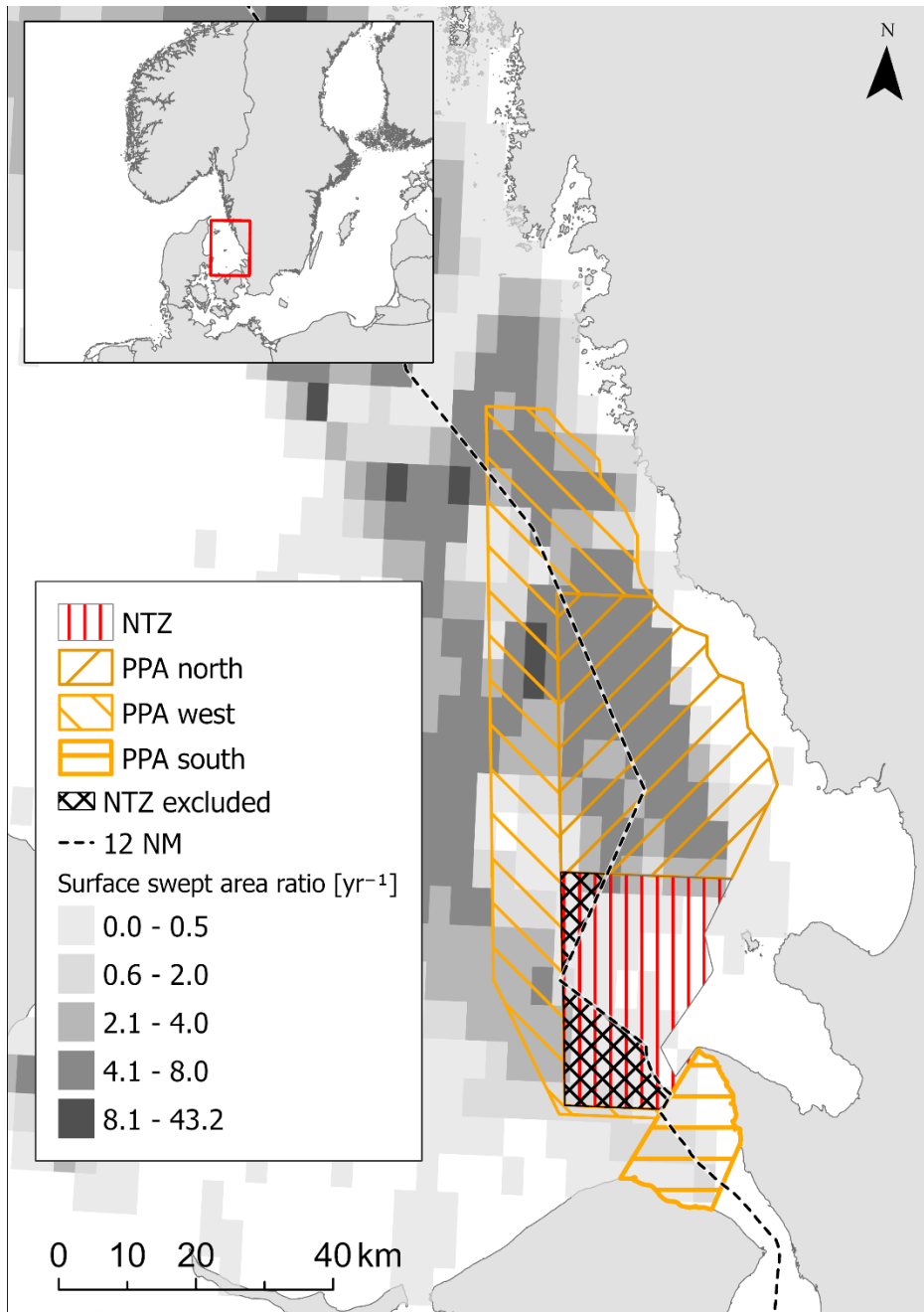
Figure 1. Map view of the enforced NTZ and PPA areas in the southern Kattegat to the Swedish and Danish fishery. Germany have access to EEZ waters outside 12 M of the member states (indicated as the crossed area within the NTZ). In the northern area (PPA-North), fishing is prohibited during the first quarter (January - March), i.e., the spawning period for cod, and during the rest of the year fishing is allowed with selective gear. In the western area (PPA-West), fishing is permitted all year round with the restriction that selective gear must be used in the first quarter. In the southern area (PPA-South) the same premises prevail as in PPA-West but the period is February to March. (for details of regulations see Fiske i Skagerrak, Kattegatt och Östersjön (FIFS 2004:36). Surface swept area ratio estimates (SAR) defines the swept area as the cumulative area contacted by bottom trawlers within a grid cell over one year. The SAR here is from 2017-2020 and averaged per year (ICES 2021). The figure is reproduced from Sköld et al. 2022 ICES Journal of Marine Science under the terms of the Creative Commons Attribution License.

10.2. Summary of ICES stock assessment of Kattegat cod

The Kattegat cod stock is assessed annually by International Council for the Exploration of the Sea (ICES) with respect to status, mortality and recruitment. This stock assessment is the primary source of information on cod status at the population level, both from before NTZ establishment and over time to follow the development of the stock's response not only to the NTZ, but also other changes in fishing regulations. The assessment shows that the stock steadily declined from the mid 1990s and that the stock was at a very low level at the time of the establishment of the NTZ in 2009. From 2010-2015 to the evaluation of the NTZ in 2016, stock size increased (Sköld et al. 2016), an increase that became even more evident after the recent revision of the cod assessment model (Fig. 2). The revised time-series resulted in an upscaling of the relative spawning stock biomass (SSB) the years around 2015 compared to previous assessments but did not change the general perception of the stock as depleted at present (ICES 2022). The stock increase in 2013-2015 can be attributed to the survival of strong year classes in 2011 and 2012 (Fig. 3). Since then, no strong year classes have been observed, and during the period 2016 to 2021 SSB decreased to a historically low level.
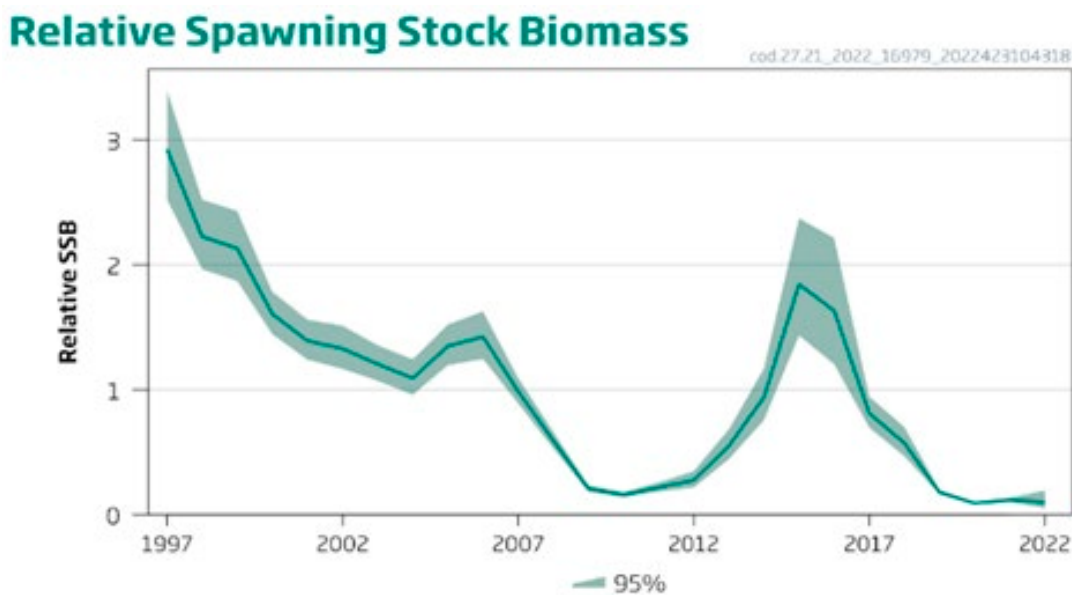
Figure 2 Spawning stock biomass (SSB) of the Kattegat cod from the stock assessment model used by ICES to produce advice on fishing opportunities. Published with courtesy of ICES.
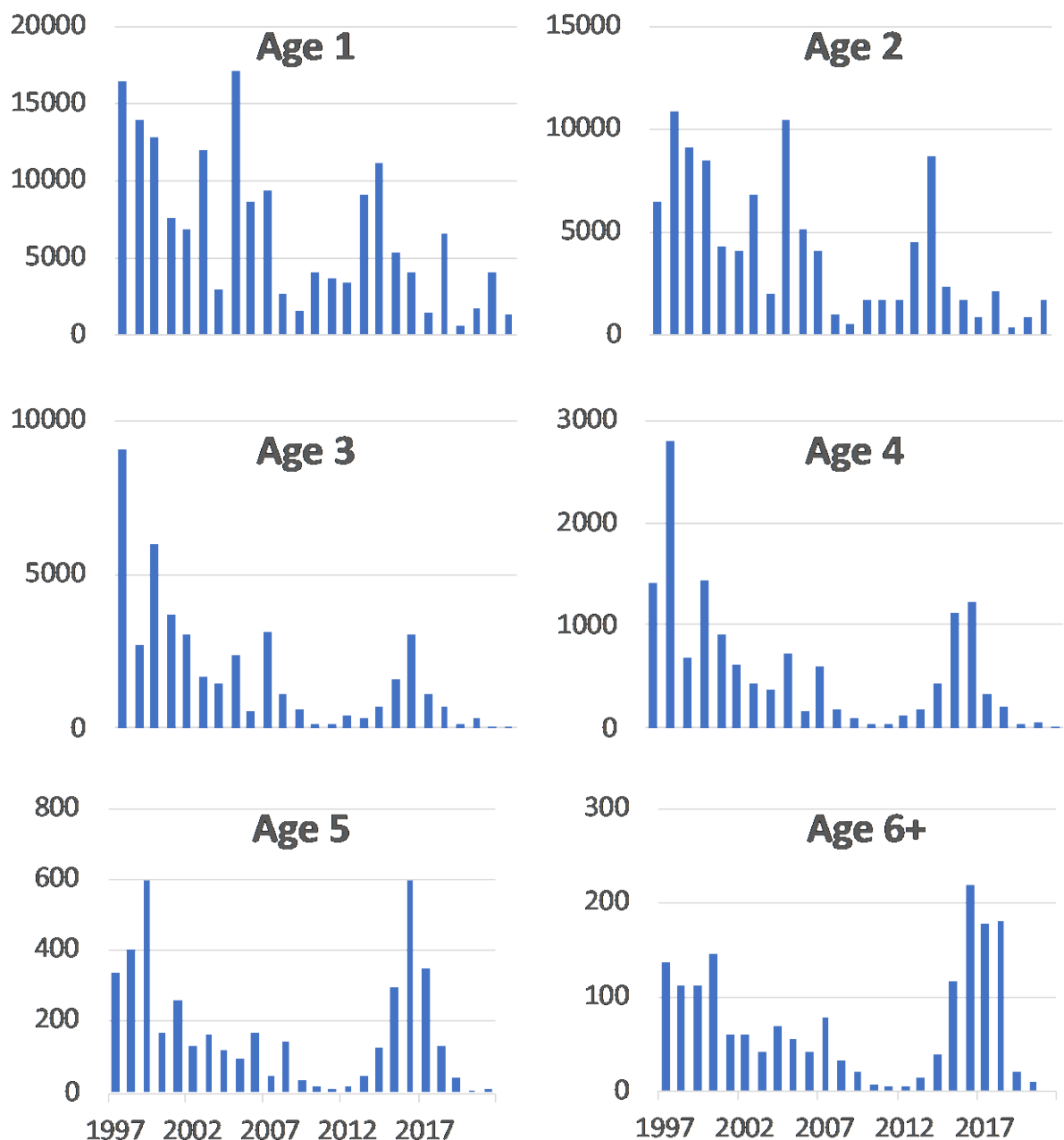
Figure 3. Trends in the abundance of age classes (1 to 6+) over time for Kattegat cod as given by the assessment model published by ICES 2021.

10.2.1.   Recruitment and age distribution

Recruitment in the Kattegat cod stock, here represented by age 1 (see top left panel in Fig 3) has been declining over time, which indicates that recruitment overfishing is occurring. In recent years, the year-classes of 2016 and to some extent 2019 are present to the age of 1-year, but rapidly disappear as they grow older. After the establishment of the NTZ in Kattegat 2009 to protect larger spawning cod, the age distribution recovered to include more age classes with a larger proportion of older fish (Fig. 3). However, in recent years the age distribution has deteriorated, which is closely linked to the decline in spawning stock biomass. In 2020 and

2021 there were very few individuals older than 2 years present in the stock according to the model.

10.2.2. Mortality in the Kattegat cod stock

Relative mortality in the Kattegat cod stock over time is given in Figure 4, as a proxy for the fishing mortality. Relative mortality is used since the reported landings and discards cannot be made to match fishing mortality in the assessment model. This unallocated mortality is mainly attributed to unreported discards, mixing with juvenile North Sea cod and changes in predation by seals. However, recent inclusion of genetic information into the assessment model indicates that stock mixing with North Sea cod has only a limited influence on the Kattegat cod stock mortality estimates, and that fishing mortality in the Kattegat cod stock has remained high except for a period around 2014 when the stock was temporarily recovering (Eero et al 2021).

Figure 4. Relative mortality (Z – 0.2) for the Kattegat cod stock as given from ICES assessment model in 2022. Published with courtesy of ICES

10.3. Modelling contribution of the NTZ to the relative fishing impact on cod

The fishing impact on cod i.e., a proxy for fishing mortality on cod in Kattegat is outlined in Vinther and Eero (2013). The approach combines a predicted spatio-temporal distribution of cod obtained from analysis of scientific survey catch per unit effort (CPUE) with the spatio-temporal distribution of fishing effort obtained from vessel monitoring systems (VMS). The fishing impact is calculated from a combination of spatial stock distribution, spatial effort, and the retention probability of cod sizes for the applied gears (for details see Vinther & Ero 2013; Vinther et al. 2018).

The fishing impact approach has been used to provide management, i.e., the Swedish Ministry for Rural Affairs and the Danish Ministry of Food, Agriculture and Fisheries, with information on how the NTZ and the PPAs put in place in 2009 perform in relation to the cod stock status, the fishing effort deployed and other EU measures. The most recent evaluation of fishing impact is provided in Vinther et al. 2018. This approach has also been used for quantitative analysis of scenarios of opening the NTZ after the spawning season, and further regulating the use of selective gears in the bottom trawl fisheries (Hjelm et al. 2014).

The PPAs and NTZ areas were introduced in 2009 to increase the spawning biomass of the Kattegat cod by protecting cod on historically important spawning grounds in the south-eastern Kattegat. The measures were supplemented with mandatory use of trawls with a lower retention of cod, and until 2016 the effort in fisheries catching cod by member states was regulated by EU (EC No.1342/2008). The Norway lobster (*Nephrops norvegicus*) fisheries are the main demersal fishery in Kattegat and have the largest catches of cod. In the analysis by Vinther et al. (2018) the reduction in fishing impact on cod is quantified from spatially modelled stock distribution and data on fishing pressure and effort from satellite vessel monitoring systems (VMS) and gear selection data from the Norway lobster fisheries. The fishing impact on cod in the Kattegat by the Danish and Swedish fisheries showed a reduction of around 70% from 2007 to 2017. The causes for the reduction in fishing impact was due to a combination of changes in the spatial distribution of effort following enforcement of the NTZ and PPAs, use of more selective gears with a lower retention of cod and an overall decrease in effort. In the Danish fisheries, the NTZ and the PPA displaced the effort which reduced the fishing impact on medium and large sized cod by around 20% over the period 2007 - 2017. Simultaneously the dominant gears changed from 90 mm trawl to the selective gears SELTRA 270/300 (Fig. 5) which reduced the fishing impact on mainly small cod. The reduction of fishing impact by the Swedish fisheries was mainly due to shift to selective gears, i.e., to the standard sorting grid which completely sorts out large cod and has low retention of small cod. For both countries the reduction in fishing impact was strengthened by an overall initial reduction in effort (Fig. 5). However, the downward trend in effort stopped and since 2014 there has been an increase in the Danish effort. Further, in 2017 there has been a shift from the standard grid to SELTRA 270/300 in Swedish fisheries (Fig. 5) which has increased the fishing impact on larger cod (Vinther et al. 2018).
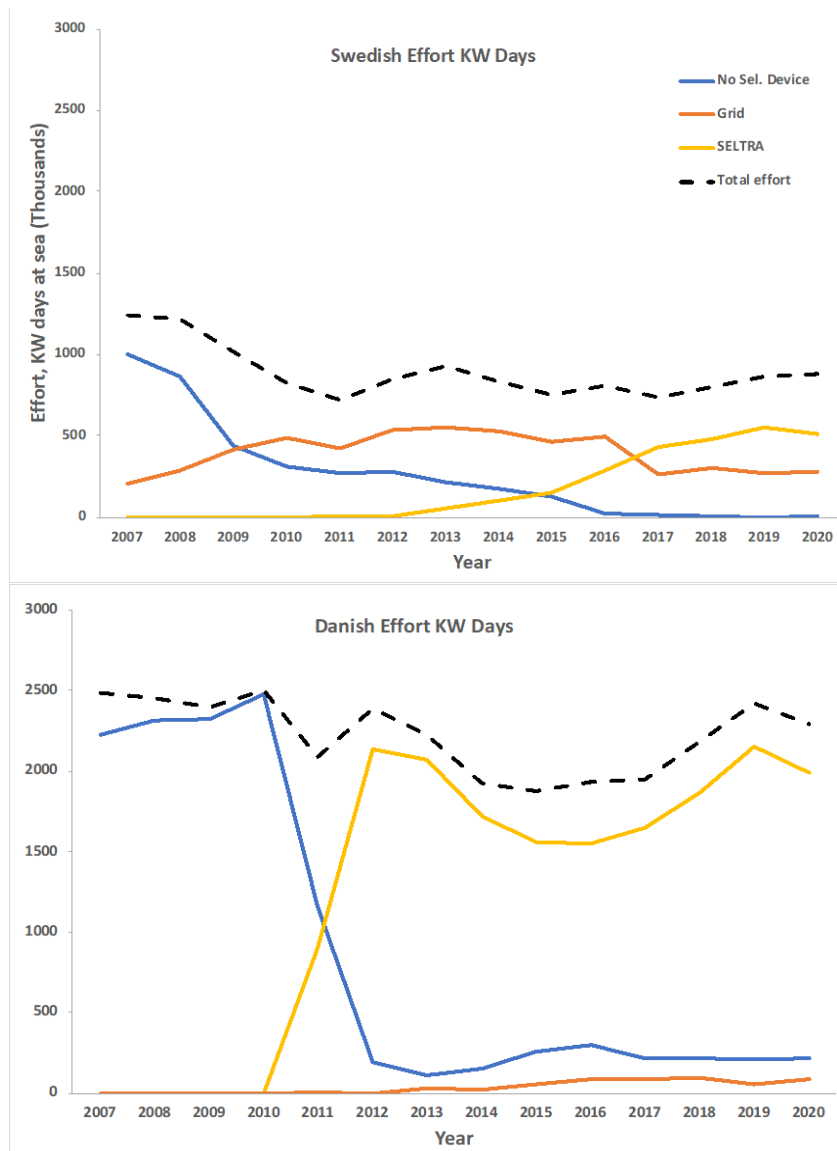
*Figure 5. Fishing effort (kW * days at sea[1]) for the period 2007-2020 for Swedish (top) and Danish (bottom) fishery. All reported effort with grid and SELTRA is included with the "no selectivity device" representing other comparable bottom trawl gears. Total effort gives the sum of all the bottom trawl effort reported.*

The Swedish fishery showed a marked proportional reduction in effort following the cod recovery plan with the effort regulation system in force (EC No.1342/2008), but without a clear counterpart in the Danish fishing effort (Fig. 5). During the years 2010 to 2016 Swedish fisheries used the Standard Grid for 60% of the effort in Kattegat. When the effort regulation was removed and landing obligation implemented in 2017 the use of the grid decreased to 40%. The Danish fishery never adopted the Standard grid in the PPA North and during the spawning season to any extent, but rapidly increased the use of the SELTRA trawls in the fishery making up approximately 90% of the effort from 2012 and onwards. The use of SELTRA has increased in the Swedish fishery, seemingly replacing part of the grid fishery and all the standard 90 mm fishing (now prohibited) following the abolishment of the effort regulation in 2016.

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[1] kW days at sea is a unit for fishing effort deployed calculated by multiplying the sum of engine power in kilowatt per vessel active with number of days.

10.4. Methods for analysing effects of the No-take zone on the fish assemblage

All analysis of fish is based on the Joint Swedish and Danish survey for cod in the Kattegat (Jørgensen et al. 2019). The survey has been conducted in November-December every year since 2008 with the exception of 2012. The survey is based on a stratified random design with 80 hauls distributed within a survey grid of 5×5 M squares. The survey gear is a 34 meter long commercial bottom trawl with 70 mm diamond mesh in the cod-end (for more details see Sköld et al. 2022).
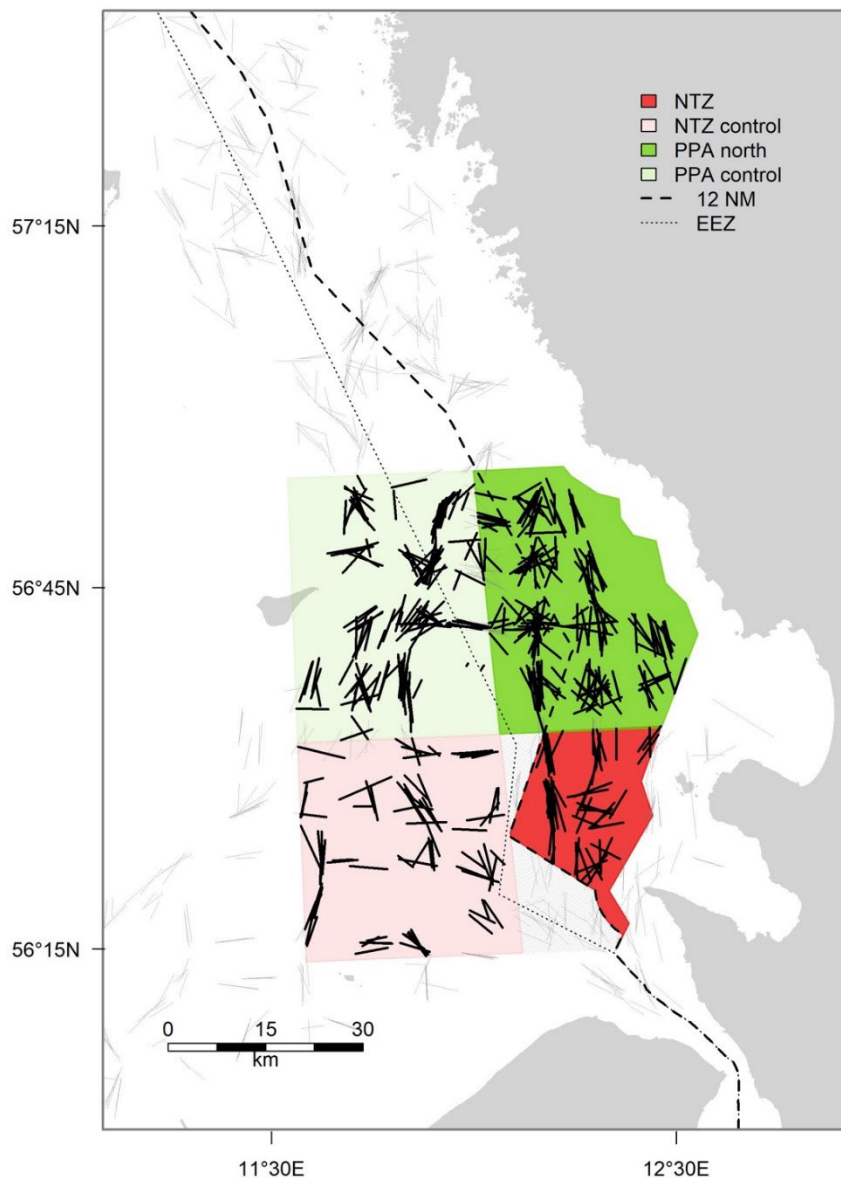
Figure 6. Map showing hauls 2008 - 2021 in the regulated areas (NTZ in red and PPA North in green) and the areas used as control in the analyses. The gray-shaded area outside Swedish territorial waters is part of the no-take zone but has been trawled by German vessels and was therefore excluded from the analyses. The figure is reproduced from Sköld et al. 2022 ICES Journal of Marine Science under the terms of the Creative Commons Attribution License.

Multivariate abundance and biomass of fish for the years 2008 – 2020 were used to evaluate the effect of the NTZ and PPA North on the temporal development of the species assemblage by PERMANOVA using the software PERMANOVA+ for PRIMER (Anderson et al. 2008). Control areas were identified as continuously trawled areas west of the treatment areas within the same depth interval and dominating seafloor substrate as the treatment area (Fig. 6). The effect of the NTZ will be the interaction between year of sampling and treatment. Dependent variables examined were species composition weighted by abundance or biomass divided by the area (km^2) swept by the trawl during the haul. Since this survey uses 70 mm diamond mesh-size nets, only species with a maximum size above 20 cm were included since smaller species are likely to only be caught occasionally. Results were visualized with canonical analysis of principal coordinates (CAP) for the interaction between treatment and year of sampling. Univariate trend analyses were carried out on the differences in average abundance and biomass between areas, by generalized least square regression, including AR-1 autocorrelation between years using the nlme package in R (Pinheiro et al 2020). An increasing difference, i.e., an upward trend between the NTZ and the control, or between the NTZ and the PPA over time indicate that the implementation of the NTZ had a positive impact on the species. For more details see Sköld et al. 2022.

10.5. Results

The multivariate analyses revealed significant shifts in the fish assemblage between 2008-2021. Closer analysis indicated that six to seven fish species, including cod increased in the NTZ relative to control areas depending on if abundance or biomass was used as dependent variable (Fig. 7, table 1). Univariate analysis showed that two flatfish species dab (*Limanda limanda*) and lemon sole (*Microstomus kitt*), and Norway lobster (*Nephrops norvegicus*) significantly increased in biomass in the NTZ, and turbot (*Scophthalmus maximus*) in the PPA relative to the control areas (Fig. 8). These results suggest that the NTZ protected these relatively mobile fish species in the Kattegat, as well as the main target species in the trawl fishery. However, neither cod abundance nor biomass showed a significant increase as an effect of the NTZ and PPA despite two relatively strong year classes in 2012 and 2013 (for more details see Sköld et al. 2022).
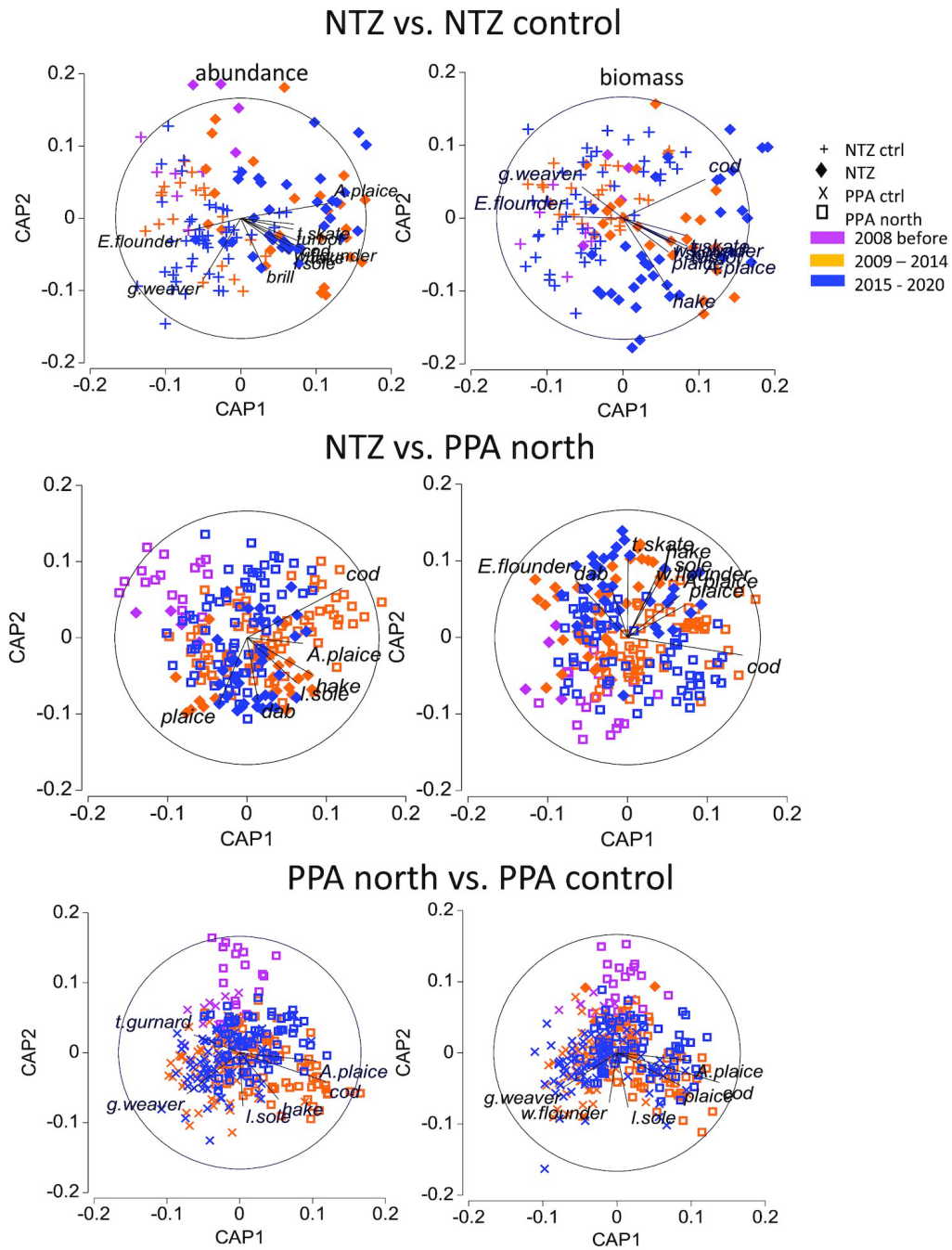
Figure 7. Canonical analysis of principal coordinates CAP plot for the visual presentation of composition by abundance or biomass of species as shaped by the interaction term treatment × year. Species that correlate with r > 0.4 to one or both CAP axes are shown on the plot. See table 1 for further details. Abbreviations for species: A. = American, E. = European, g. = greater, l. = lemon, t. = thorny, and w. = witch. The figure is reproduced from Sköld et al. 2022 ICES Journal of Marine Science under the terms of the Creative Commons Attribution License.

Table. 1. PERMANOVA and correlation of species with principal coordinates of canonical analysis (CAP) of the two most explanatory CAP axes. Species that correlate with r > 0.4 to one or both CAP axes are included. Significant correlations are indicated in red (α level 0.05, df = 24). The table is reproduced from Sköld et al. 2022 ICES Journal of Marine Science under the terms of the Creative Commons Attribution License.

Abundance of species years 2008 - 2020								
NTZ ver. NTZ ctrl			**NTZ ver. PPA north**			**PPA north ver. PPA ctrl**		
pseudo F $_{11.\,113}$ = 1.7503. p = 0.001			pseudo F $_{11.\,203}$ = 1.7079. p = 0.0001			pseudo F $_{11.\,278}$ = 1.9537. p = 0.0001		
Species	**CAP1**	**CAP2**	**Species**	**CAP1**	**CAP2**	**Species**	**CAP1**	**CAP2**
Amblyraja radiata	0.42	-0.05	*Gadus morhua*	0.72	0.39	*Gadus morhua*	0.71	-0.25
Gadus morhua	0.46	-0.19	*Hippoglossoides platessoides*	0.42	-0.04	*Hippoglossoides platessoides*	0.63	-0.07
Hippoglossoides platessoides	0.72	0.13	*Limanda limanda*	0.08	-0.49	*Merluccius merluccius*	0.32	-0.40
Merluccius merluccius	0.49	-0.25	*Merluccius merluccius*	0.49	-0.29	*Microstomus kitt*	0.03	-0.46
Microstomus kitt	0.40	-0.30	*Platichthys flesus*	-0.22	-0.53	*Trachinus draco*	-0.44	-0.35
Scophtalamus rhombus	0.18	-0.41						
Scophthalmus maximus	0.42	-0.09						
Trachinus draco	-0.30	-0.50						

Biomass of species years 2008 - 2020

NTZ ver. NTZ ctrl			NTZ ver. PPA north			PPA north ver. PPA ctrl		
pseudo F $_{1, 113}$ = 1.4138. p = 0.0234			pseudo F $_{11, 203}$ = 2.2335. p = 0.0001			pseudo F $_{1, 113}$ =1.9537. p = 0.0001		
Species	**CAP1**	**CAP2**	**Species**	**CAP1**	**CAP2**	**Species**	**CAP1**	**CAP2**
Amblyraja radiata	0.50	-0.14	*Amblyraja radiata*	0.005	0.62	*Gadus morhua*	0.83	-0.25
Gadus morhua	0.65	0.32	*Gadus morhua*	0.87	-0.14	*Glyptocephalus cynoglossus*	-0.06	-0.42
Hippoglossoides platessoides	0.62	-0.32	*Merluccius merluccius*	0.26	0.56	*Hippoglossoides platessoides*	0.61	-0.07
Merluccius merluccius	0.37	-0.60	*Microstomus kitt*	0.24	0.46	*Microstomus kitt*	0.09	-0.46
Platichthys flesus	-0.44	0.01	*Platichthys flesus*	-0.39	0.45	*Pleuronectes platessa*	0.52	-0.27
Scophthalmus maximus	0.51	-0.23	*Pleuronectes platessa*	0.44	0.26	*Trachinus draco*	-0.44	-0.29

The univariate analyses detected significant trends with increase in biomass over time for the following species identified in the multivariate analysis: *L. limanda*, *M. kitt* and *N. norvegicus* in comparison between the NTZ and the control area; *M. kitt*, *S. maximus*, and *N. norvegicus* in the comparison between the NTZ and PPA north. A positive trend for *G. morhua* was indicated for the NTZ in comparison to the PPA north area albeit only at a significance level of $\alpha = 0.10$ (Fig. 8). A similar pattern was found for abundance although the trends were less distinct than for biomass. No trends were detected for *M. merluccius*, *P. platessa*, *H. platessoides*, *P. flesus*, *T. draco* and *A. radiata* that correlated with the CAP axes for treatment X years in the multispecies analyses (for more details see Sköld et al. 2022).

Amblyraja radiata

Gadus morhua

Hippoglossoides platessoides

Limanda limanda

NTZ vs. NTZ control   **NTZ vs. PPA north**   **PPA north vs. PPA control**

p=0.037   p=0.116   p=0.38

Merluccius merluccius

NTZ vs. NTZ control   **NTZ vs. PPA north**   **PPA north vs. PPA control**

p=0.627   p=0.229   p=0.495

Microstomus kitt

NTZ vs. NTZ control   **NTZ vs. PPA north**   **PPA north vs. PPA control**

p=0.003   p=0.008   p=0.667

Nephrops norvegicus

NTZ vs. NTZ control   **NTZ vs. PPA north**   **PPA north vs. PPA control**

p=0.035   p=0.01   p=0.257

Figure 8. Trend analysis for selected species in 2008-2021. The left panel shows differences in biomass (mean ± 95 % c.i.) between the No-take zone (NTZ) and the control area. Confidence intervals that do not overlap with the zero black dashed line indicate a significant difference. The middle panel shows the difference between the NTZ and PPA north, and the right panel shows differences in biomass between PPA north and the control area. The blue or red line with the associated p value shows the trend over time and is the significance of the generalized least square fitted linear model.

10.6. Discussion

The No take Zone (NTZ) with buffer zones was introduced in the Kattegat in 2009, where the present fishing regulations have been effective for 13 years at the time of this evaluation. The NTZ and the PPAs were primarily introduced to prohibit targeted as well as by-catch fishery of cod on the spawning grounds, and to displace fisheries by-catching cod from areas where mature cod in the Kattegat aggregate outside the spawning season. We find no long-term trend indicating recovery within the NTZ or the PPA North. The ICES stock assessment for Kattegat cod showed that the stock recovered after the introduction of the NTZ and two relatively successful recruitment events. Spatial analysis of the fisheries impact (Vinther & Ero 2013 - discussed below) showed that the displacement of the fisheries areas generated by the NTZ lowered the fishing impact on large cod. The time series from the survey used in our study confirm this, and similar to the ICES assessment, the stock status deteriorated again from 2016 onwards.

Conclusions from the fishing impact model 2008 - 2015

The effect of the NTZ on the cod stock was modelled for the first part of the evaluation period (2008-2015), when the cod stock was showing signs of recovery. Closing the NTZ led to the displacement of the fishery to other areas in Kattegat. Model calculations of this displacement indicate that the negative impact of the fisheries on larger cod in the Kattegat decreased as a consequence of the NTZ. For small cod, 12 - 24 cm total length, the impact from the fishery increased, as small cod individuals were distributed in higher densities outside the NTZ where relative fishing effort increased. However, the introduction of more selective gears in combination with a reduced overall fishing effort decreased the mortality of small cod during 2008-2015, and the mortality of small cod for 2015 was only 37% of mortality in 2008. Establishing the NTZ in combination with the reduction in total effort and increased gear selectivity in the PPAs thus decreased both large and small cod mortality and coincides with the temporary recovery of the cod stock.

The new management regime – introduction of the landing obligation

The new regulations following the stepwise introduction of the landing obligation[2] in the EU also removed other regulations. One important regulation abandoned in 2016 was that use of gear with less than 1.5 % bycatch of cod was exempted from the effort regulation system earlier in force (EC No.1342/2008). The only trawl gear verified to reduce the bycatch of cod to that extent was the Swedish sorting grid (Valentinsson and Ulmestrand, 2006). As a result, the main incentive to use the sorting grid was lost, i.e., unlimited effort for vessels selectively fishing for *Nephrops norvegicus* using sorting grid. Instead, from 2017 the cod in Kattegat came under the landing obligation with the rationale that all bycatch of cod shall be landed and thus would be minimized and kept within a bycatch quota through an incentivized selective fishery for *Nephrops* and flatfish. The recorded landings of cod below minimum conservation reference size (MCRS) have, however, been negligible and discard rates of cod remains at high levels despite the new regulations. ICES estimates that the discards of cod made up about 63% of the catch weight and close to 96% of the individuals caught during 2020 (ICES 2021). Thus, fisheries are not following the landing obligation, but instead dumping cod bycatch at sea to avoid counting this catch against their fisheries quota. Increased bycatch and a diminishing cod stock and have caused high fishing mortality for cod in the Kattegat.

With increased fishing effort and a reduction in the use of selective gear in the Kattegat following the implementation of the landing obligation, the size and location of the NTZ becomes even more important. The NTZ is significantly

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[2] https://ec.europa.eu/oceans-and-fisheries/fisheries/rules/discarding-fisheries_en

smaller than what was proposed by the scientists (Hjelm et al. 2008). Part of the NTZ and the PPAs outside national waters are also fished by German fishing vessels since the NTZ and PPAs are regulated under a bilateral agreement between Sweden and Denmark only. Changes in regulations of the *Nephrops* fishery, being the major source of cod bycatch, has increased the fishing mortality of cod in recent years. Primarily, the discontinuation of the effort regulation system in preparation for the landing obligation removed the cap on the effort. The change in minimum landing size (from 40 to 32 mm carapace length) of *Nephrops* was accompanied by a significant total allowable catch (TAC) increase to account for the fraction of small *Nephrops* previously discarded and now retained in the catches. At the same time an exemption from the landing obligation due to high survival of small *Nephrops*, allowed these to be discarded. Together these changes in regulations may have further released the *Nephrops* fishery from a limitation of the fishing effort, and accordingly contributed to the mortality on cod.

Limitations on recruitment

The recruitment of cod in the Kattegat was relatively strong by juvenile cod born in 2011 and 2012, and the analyses of cohorts (Fig. 3) show that these year-classes were responsible for the initial recovery of the cod stock. After 2012 recruitment has been poor as represented by relatively low numbers of one-year old cod. Depleted fish stocks such as the Kattegat cod stock may be suffering from recruitment overfishing (Myers et al 2004), with a reduced likelihood of producing strong year-classes. Increased fishing mortality of the early juvenile stages when the effort regulations were discontinued in 2016 may also have contributed by reducing the recruitment pulses before they are recorded. Successful recruitment events are necessary to rebuild the cod stock, but under the existing management regime the risk of stock depletion by all incoming year classes through bycatch in the unselective trawl fisheries seems unavoidable.

Stock mixing with North Sea cod

For certain cohorts of cod, stock mixing with North Sea cod is important and may have consequences for the stock assessment. The North Sea cod use Kattegat as a nursery ground and when it is mature it migrates back to the North Sea. A recent report investigated the influence of stock mixing on the perception of the stock as given by the stock assessment model (Eero et al 2021). The genetic analyses underlying the model show that the proportion of juvenile cod of the Kattegat ecotype is close to zero in the Northern Kattegat and close to 100% in the south. The proportion of the Kattegat ecotype increases with age, approaching 73-92% of age 6+ cod sampled in the whole of Kattegat. In summary, the use of genetics data separating the stocks only had a limited effect on the results from the assessment. However, spatial management actions to protect cod, such as the location of the

NTZ in the Kattegat, benefit from the genetic information on the spatial distribution of ecotypes (Eero et al 2021). The genetic data shows that the proportion of older and larger cod of the Kattegat ecotype is high in the South-Eastern Kattegat. This verifies that the locations of the NTZ and PPAs in that perspective were placed correctly to protect the Kattegat cod ecotype.

Response of the fish assemblage to the NTZ

The fish assemblage changed in response to the establishment of the no take zone. A few fish species accounted for the main differences in the multivariate analysis, e.g. American plaice *Hippoglossoides platessoides,* turbot *Scophthalmus maximus*, cod *Gadus morhua*, hake *Merluccius merluccius*, and starry ray *Amblyraja radiata* increased in biomass and abundance over time in the NTZ relative to the NTZ-control and the PPA North. Three of the flatfish species (dab, lemon sole and turbot) showed significant increases in the NTZ relative to the control in biomass. These flatfish species, in addition to plaice, flounder, and sole, are harvested in the Kattegat (Bergenius et al 2018). These findings strengthen the conclusion that NTZs, even in an open system, can lead to an increase in abundance and biomass of some fish species, despite that they are highly mobile and fished elsewhere in an area like Kattegat.

Norway lobster

Norway lobster, *Nephrops norvegicus*, is the most important commercial species in the Kattegat and extensively fished by bottom trawlers. *N. norvegicus* response to the NTZ is interesting not only because of the intensive fishing but also because it is a species with limited patterns of movement. After a pelagic larval stage, the migration of juveniles and adults is relatively limited, as shown in tagging experiments where recaptures were generally reported to be close (< 9km) to the release site (Chapman 1980). Given the size of the NTZ investigated here, a large proportion of the *N. norvegicus* could be regarded as residents in the area. The biomass of *N. norvegicus* in the NTZ increased over time relative to both NTZ control and PPA North and the biomass is presently higher in the NTZ compared to the other areas. The pattern is more variable in the abundance data, but still shows a trend towards more individuals in the NTZ compared to the fished areas. The Kattegat-Skagerrak stock is presently fished at precautionary levels (ICES 2021), and our results show that significant effects of an NTZ can be detected despite being fished at sustainable levels within a management area like the Kattegat.

CONCLUSION

The introduction of the NTZ with buffer zones in combination with incentives to use selective gears initially showed results with signs of recovery in the age

structure and biomass of the cod stock. A local increase in flatfish and Norway lobster biomass in the NTZ relative to control areas also shows that mobile species may benefit from closing of an area even in an open system such as the Kattegat. However, the cod stock showed a rapid deterioration to an all-time low observed in 2020. The reported high discard rates in the mixed fisheries for Norway lobster and fish affecting both recruits and adults of cod, despite the landing obligation, are considered a major driver behind the lack of recovery and reinforced depletion of the stock. When NTZs are used to protect main spawning areas, while allowing for unselective fisheries to continue, effective management actions to reduce the bycatch of both juvenile and adult cod need to be in place. However, the situation for the Kattegat cod gives at present little hope for rebuilding the stock. No recovery plan exists, and cod has been classified as a "bycatch species" implying lowered management ambition for this cod stock (EU, 2018).

10.7. Ecosystem effects of the No take zone in Kattegat – benthic macrofauna

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10.8. Background

Mobile bottom fishing gears disturb the seafloor, impact sediment biogeochemistry including carbon sequestration, damage biogenic structures, and kill benthic invertebrates resulting in impacts on biodiversity and benthic ecosystem functions (Dayton et al. 1995; Thrush and Dayton 2002; O'Neill & Ivanović 2016; Sköld et al. 2018; Bradshaw et al. 2021; De Borger et al. 2021; Sala et al. 2021; Epstein et al. 2022). The most widespread human activity affecting the seafloor is bottom trawling for fish and crustaceans, and the Skagerrak and Kattegat region have among the highest trawling pressure, highest impact from bottom trawling and lowest relative benthic status in the world (Eigaard et al. 2017; Pitcher et al. 2022). Otter trawl is presently the only mobile bottom contacting gear used in Kattegat (ICES 2021), and based on meta-analyses, this gear in general penetrates ca. 2.4 cm down into the sediment and reduces benthic biomass by 6 %, abundance by 3 % and number of species by 9 % (Hiddink et al. 2017; Scibberas et al. 2018). Other documented effects of demersal trawling in soft bottom habitats are shifts in species composition favoring the abundance of mobile and scavenging species, which may in turn affect ecosystem functioning (e.g. Tillin et al. 2006), and shifts from long-lived to short-lived species (Rijnsdorp et al. 2018). Slow-growing, low mobility, erect and fragile species in these habitats, such as sea pens (soft corals), are

particularly sensitive to physical disturbance by demersal trawls (Sciberras et al. 2018; Greathead et al. 2007; McConnaughey et al. 2000).

In the Kattegat, shifts in soft seafloor macrofauna community composition and a general decrease in number of species and indices of diversity at low to medium trawling intensities have been documented (Sköld et al. 2018). Shifts in community composition of benthic fauna is, however, not only a consequence of the physical disturbance but also an indirect effect linked to changes in the food web (Collie et al. 2016). In the Kattegat, the dominant burrowing brittle stars *Amphiura filiformis* and *Amphiura chiajei* are resistant to bottom trawling and even tend to increase with trawling intensity (Sköld et al. 2018). This response was supported by a corresponding trend towards lowered abundance brittle stars abundance following establishment of the NTZ, and increased benthivore density and predation pressure within the NTZ (Sköld et al. 2018).

Using towed video surveys for Norway lobster burrows in Kattegat for years 2015 - 2020 Sköld et al. (2021) found that the seapens *Virgularia* spp. and *Pennatula phosphorea* dominated the larger epifauna, and that *P. phosphorea* had lower abundance in trawled areas compared to the NTZ in Kattegat (see Sköld et al 2021). This study also covered also deeper areas in the Skagerrak and the comparisons between protected and bottom trawled areas confirmed that the sensitivity to physical disturbance from bottom trawling is greatest for large species of seapens while smaller species of seapens are less sensitive.

The aim of this study is to follow-up the study by Sköld et al. (2018) and analyze the long-term effects (12 years) in the benthic macrofauna community as a response to the reduction of bottom trawling within the NTZ.

10.9. Methods

Benthic macrofauna

Macrofauna were sampled in the years 2009, 2010, 2011, 2014, 2017, 2018 and 2020 using a modified Smith-McIntyre grab (0.1 m^2) at 14 stations; 7 inside and 7 outside the NTZ where trawling continued (Fig. 9). The 7 stations outside the NTZ are hereafter named as Trawled reference. One sample was taken at each station and sieved (1.0 mm mesh size) for macrofauna. Further details are outlined in Sköld et al. (2018). Biomass was converted from wet weight to ash-free dry weight (AFDW) using conversion factors from Brey et al (2010) and Gogina et al. (2022).
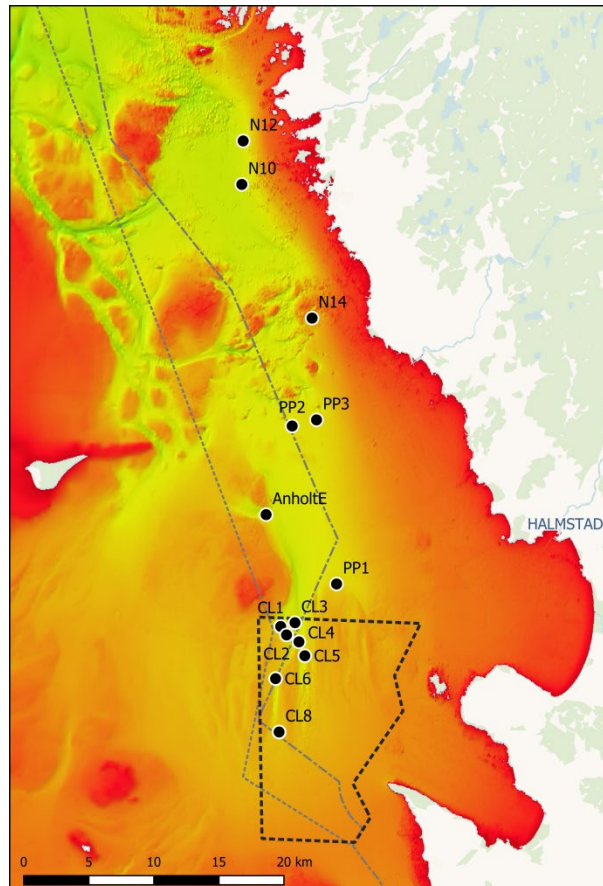
Figure 9. Map of the benthic stations sampled within and ouside the NTZ (black dotted line), The territorial border and the EEZ are illustrated with grey dotted lines.Color shows depth from EMODnet Bathymetry Consortium (2020): EMODnet Digital Bathymetry (DTM).

All stations sampled in 2009, both inside and outside the NTZ, were defined as trawled with trawling intensities above the median trawling intensity (Swept area ratio SAR = 2.5 year^{-1}) calculated across all 58 Kattegat sampling stations (see Sköld et al. 2018 for description of sampling design and SAR calculation method). As expected, a reduction in trawling intensity at the stations inside the NTZ since its establishment in 2009 was seen over time (Sköld et al. 2018). We also assumed that the macrofauna community status in 2009 after the enforcement of the MPA could be considered a reasonable baseline since the impact of bottom trawling is likely to remain longer (>1.9 years according to Hiddink et al. 2017). German vessels are allowed to fish outside territorial waters in the Exclusive economic zones (EEZ) and this includes parts of the NTZ otherwise closed to the Swedish and Danish fisheries. The German vessels fishing in the Kattegat are bottom trawlers active mainly in the southwest part outside the Swedish territorial border, which do not overlap with the stations sampled, as verified by data downloaded from Global Fishing Watch[3].

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[3] https://globalfishingwatch.org

The experimental effect of the NTZ on the temporal development of the species community was in the first step evaluated using PERMANOVA+ for PRIMER (Anderson et al. 2008), with year of sampling and treatment (i.e., NTZ versus Trawled reference) as fixed factors and station as a random factor nested in treatment. The macrofauna community shift is complex and can occur in several directions/dimensions and there were > 250 macrofauna taxa in the multivariate matrix. The effect of the closure on the macrofauna community will be the interaction between year of sampling and treatment. Dependent variables examined were species composition weighted by abundance or biomass (AFDW). If the overall PERMANOVA was significant post hoc comparisons were done using pairwise PERMANOVA, and results were visualized with canonical analysis of principal coordinates (CAP) for the interaction between treatment and year of sampling. Before testing, data were transformed (fourth root) and evaluated with distance-based test for homogeneity of multivariate dispersions using the function PERMDISP.

The community total abundance, biomass (AFDW), number of taxa, Shannon diversity and the dominant species *Amphiura filiformis* and *Amphiura chiajei*, were also examined using univariate factorial ANOVA with year of sampling and treatment as fixed factors and station as a random factor nested in treatment including a repeated measure design with AR-1 autocorrelation between years. Dependent variables were transformed by square root or log (x + 1) and tested for unequal variances by Welch's test. Univariate statistics were analyzed using the statistical package JMP pro 15.

10.10. Results

The premise for the evaluation of the trawling ban within the NTZ is that changes in the macrofauna community composition over time within the NTZ will deviate from changes in the trawled reference area. As expected trawling has continued in the reference area and ceased in the NTZ, however, some indications of illegal trawling activity within the NTZ were present during the first years (Sköld et al. 2012). Significant differences were found for the sought-after interaction term Treatment X Year using both macrofauna community abundance (pseudo $F_{6, 72}$= 1.46, p =0.0016) and biomass (pseudo $F_{6, 72}$ = 1.36, p =0.0103) as dependent variables. Post-hoc comparisons using pairwise PERMANOVA showed, for both abundance and biomass, that the later years (2017, 2018 and 2020) differed from the earlier years (2009−2011) within the NTZ, while fewer and less clear differences occurred between years where trawling continued (trawled reference). The overall multivariate analysis thus indicated more and consistent changes over

time in species composition within the NTZ. The canonical analysis of principal coordinates (CAP) in Fig. 10 shows how individual species correlate to the shift in the macrofauna community as shaped by the sought Treatment X Years interaction i.e. periods 2009-2011 versus 2017-2020. Significance of the correlations are shown for the two most informative CAP axes in table 2.

ntz ver. trawled reference
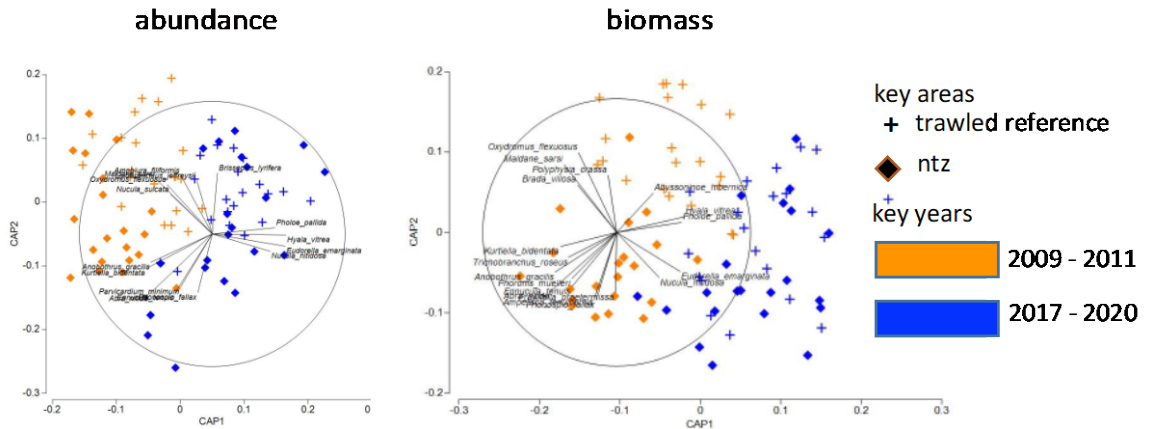
Figure 10. Canonical analysis of principal coordinates (CAP) plot for macrofauna community composition of abundance and biomass of species as shaped by the interaction term Treatment X Year. Species that correlate with r > 0.4 to one or both CAP axes are overlayed the plot. See table 3 for further details.

Table 2. PERMANOVA and correlation of species with principal coordinates of canonical analysis (CAP) of the two most explanatory CAP axes. Species that correlate with r > 0.4 to one or both CAP axes are included. Significant correlations are indicated (in red at α level 0.05 and in Italic at level 0.10, df = 10).

Benthic fauna ntz ver. trawled reference 2008-2010. 2017-2020

Abundance			Biomass		
Species	**CAP1**	**CAP2**	**Species**	**CAP1**	**CAP2**
Hyala vitrea	*0.55*	-0.01	*Hyala vitrea*	*0.50*	0.12
Eudorella emarginata	*0.55*	-0.09	*Pholoe pallida*	*0.48*	0.07
Pholoe pallida	0.47	0.05	*Eudorella emarginata*	*0.47*	-0.29
Nucula nitidosa	0.44	-0.13	Praxillella praetermissa	-0.01	-0.45
Brissopsis lyrifera	0.04	0.46	Polyphysia crassa	-0.06	0.43
Amphiura filiformis	-0.24	0.44	*Prionospio fallax*	-0.15	*-0.50*
Maldane sarsi	-0.42	0.41	Ampelisca tenuicornis	-0.16	-0.49
Anobothrus gracilis	-0.46	-0.21	Oxydromus flexuosus	-0.29	0.60
Abra nitida	-0.47	-0.45	Ennucula tenuis	-0.33	-0.40
Kurtiella bidentata	-0.49	-0.26	*Maldane sarsi*	-0.39	*0.51*
			Kurtiella bidentata	-0.42	-0.10
			Anobothrus gracilis	-0.46	-0.30
			Abra nitida	-0.49	-0.44

The two overall dominant species in the NTZ and the trawled reference areas were the brittle stars *Amphiura filiformis* and *Amphiura chiajei* both in abundance (60 %) and biomass (69 % excluding large mussels *Arctica islandica* and heart urchins). Factorial ANOVA indicated a significant interaction, with a decrease for abundance of *A. filiformis* within the NTZ over time, relative to the trawled reference area albeit only at a significance level of $\alpha = 0.10$ (ANOVA: $F_{6, 12} = 2.16$, $p = 0.057$). A similar pattern was apparent for *A. chiajei* but the ANOVA indicated no difference (Fig. 11).

The total abundance, biomass (AFDW) and diversity of the community showed no significant interaction in the univariate analyses.
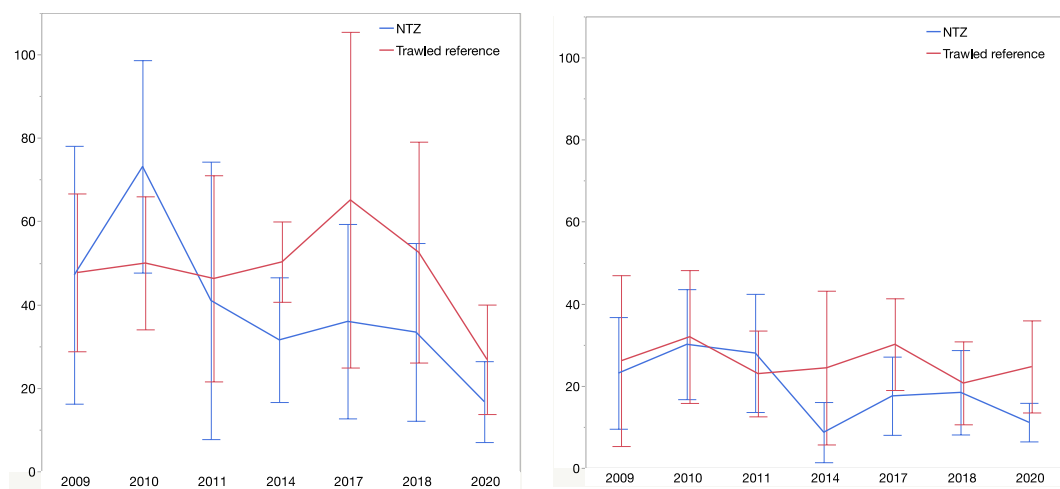
Figure 11. Mean abundance ± 95 % confidence interval of Amphiura filiformis *(left) and* Amphiura chiajei *(right) across years in the NTZ area (blue line, n = 7) and trawled reference area (red line, n = 7).*

10.11. Discussion

Our results show significant shifts in the macrofauna assemblage with species indicating both increase and decrease within the NTZ over the 12 years following the enforcement of the NTZ. Of the species potentially contributing to the shifts we found that 9 out of the 16 species were the same as those correlating in the trawling intensity gradient study of macrofauna by Sköld et al. 2018 in the Kattegat. The response of these species may indicate a long-term recovery pattern following cessation of bottom trawling on the soft seafloor in the Kattegat. The dominant brittle stars *Amphiura filiformis* and *A. chiajei* are evidently resistant to trawling and have shown a positive response to increased trawling intensity, possibly because of reduced predation on the brittle stars by demersal fish and the target species for bottom trawling in the area *Nephrops norvegicus* (Sköld et al. 2018).

The results from the present long-term study using the same initial samples and design as Sköld et al. (2018), support that the decreased abundance and biomass of the brittle stars in the NTZ remains after 12 years. Also, the potential interaction with the main predators on brittle stars i.e., flatfish and *N. norvegicus* are supported by the increase of these predators following the trawling ban within the NTZ. This can be seen in the analysis of the fish assemblage and *N. norvegicus* earlier in this report.

10.12.   Acknowledgements

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11. No-take zone and artificial reefs for lobster and predatory fish at Vinga, Kattegat

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Photo: Baldvin Thorvaldsson

Summary

No-take zones (NTZ) and habitat restoration can be important tools in conservation efforts to strengthen and protect coastal marine populations. In Vinga in the Gothenburg archipelago, Sweden, two small NTZs where no fishing was allowed were established in combination with reconstruction of reef habitats. The first NTZ, Tanneskär, measured 1.2 km^2, and included two artificial reefs. The second area, Buskär, measured 3.2 km^2, and included five artificial reefs (AR). The main purpose of these reefs was to benefit certain species of crustaceans and fish; European lobster (*Homarus gammarus*), edible crab (*Cancer pagurus*), Atlantic cod (*Gadus morhua*), saithe (*Pollachius virens*), pollack (*P. pollachius*) and whiting (*Merlangius merlangus*) by providing habitat and foraging grounds.

The lobster reef project was implemented in 2002. A follow-up/monitoring program (2003–2006) was developed that consisted of quantitative fishing surveys using lobster traps and fyke nets. The monitoring program was officially terminated in 2007, when a final report of the project was compiled, but the fishing regulations remained. Because it was not possible to evaluate a potential reef effect on the recruitment of small lobsters during such a short study period, continuous fishing surveys were conducted using lobster traps during 2008–2010. To evaluate potential long-term effects of the NTZs upon the marine communities, sampling continued with lobster traps in 2014 (to target lobster) and fyke nets in 2015, to target benthic fish and crustacean communities.

The relative abundance and size structure of lobster increased in the NTZ, both in the AR and natural habitats over all years. In 2006, four years after the fishing ban, the catches of lobsters were three times higher than catches prior to the establishment of the NTZs, and in 2015, the catches were six times higher. Moreover, benthic fish and crustacean assemblages rapidly diverged between the fished reference area and the NTZs after establishment. These differences were mainly driven by higher abundances of Atlantic cod, poor cod (*Trisopterus minutus*), lobster and wrasses in the NTZs, and a decline in abundances of small decapod crustaceans, the latter likely indicating cascading effects of increased top-down predation in the NTZ. Abundances of predatory fishes increased in the NTZ following implementation in 2003, but these increases did not continue after 2006. In the follow-up study in 2015, no significant differences in the abundances of predatory fishes in the NTZs compared to the reference area were apparent, except for poor cod which showed a trend towards higher abundances in the NTZs. In summary, the study demonstrates that the implementation of relatively small NTZs of just a few km^2, enhanced by artificial reefs, had the ability to rapidly invigorate local populations of lobster and demersal fishes in the Vinga area. Moreover, the observed changes in species abundances may re-initiate a top-down control of the marine food webs in the NTZs.

11.1. Background

Reef habitats on the Swedish west coast host highly diverse communities, and are important for the production of both fish and crustaceans. In connection with a large operation to develop safer waterways for the maritime traffic destined for or leaving the Gothenburg harbour, the main shipping route was widened and deepened, which led to a removal of shallow hard high-complexity structures considered (among other things) to be valuable for lobster production. To compensate for the negative impact on the marine environment by the nearshore habitat loss, seven artificial reefs were created in the outer archipelago in 2002-2003. The reefs were constructed of rocks from the blasted reef structures, and were deployed in two areas close to the island Vinga in the Gothenburg archipelago. The main purpose of the artificial reefs was to benefit certain species of crustaceans and fish, mainly lobster (*Homarus gammarus*), edible crab (*Cancer pagurus)*, cod (*Gadus morhua*) and other gadoids by providing habitat and foraging grounds.

The locations of the reefs were chosen in collaboration with representatives for both public and private interests, to assure that the establishment of the reefs did not negatively affect any stakeholders. Examples of aspects that were taken into consideration were the needs of the (commercial) fishery and the Swedish military, hydrographical and geotechnical issues, and the already existing high environmental and cultural values in the area. In connection with the construction of the reefs, two no-take zones (NTZs) which were closed to all fishing were established. These fishing regulations were implemented in 2003. One of the areas, Tanneskär, measured 1.2 km^2, and included two artificial reefs, and the other area, Buskär, measured 3.2 km^2, and included five artificial reefs.

A monitoring program was implemented in 2003–2006, consisting of quantitative surveys using lobster traps and fyke nets. The monitoring program was terminated in 2007, but the fishing regulations remained. Because it was not possible to evaluate a potential reef effect on the recruitment of small lobsters during such a short study period, continuous fishing surveys were conducted using lobster traps during 2008–2010. In 2009, the Swedish government made the two NTZs permanent. To evaluate potential long-term effects of the NTZs upon the marine communities, two additional surveys were conducted; one that targeted lobster in 2014 (using lobster traps) and one that targeted benthic fish and crustacean communities in 2015 (using fyke nets).

Before the NTZs were made permanent, several workshops were held with a range of different stakeholders, organised by the Swedish Board of Fisheries and the County Administration in Västra Götaland. Organizations included in these workshops represented the municipalities concerned, commercial and recreational

fisheries, nature conservation, and the tourism sector. All of these stakeholders were positive towards permanent NTZs, but some opposed a suggestion to expand and merge the two areas into one, larger continuous area. The final NTZs that were implemented in 2010 included a small extension eastward from the NTZ at Tanneskär, whereas Buskär remained the same (Fig. 1).

The fish community in shallow hard bottom areas of Kattegat and southern Skagerrak is generally characterised by different species of wrasses, gadoids and sculpins. On shallow soft bottoms, the fish community is slightly different and composed mainly of various species of flatfish and gadoids (Pihl et al., 1994; Pihl & Wennhage, 2002). Cod is the most commonly occurring gadoid, although the cod populations along the Swedish west coast have decreased substantially in density, distribution and size structure since the 1960s. The main reason for the decline in the coastal cod populations is probably an unsustainable fishing pressure (Svedäng, 2003; Svedäng & Bardon, 2003; Swedish Board of Fisheries, 2009). For lobster, there are indications of a long-term decrease since the end of the 19th century. In Kattegat and Skagerrak, the population dynamics are driven mainly by fishing and climate (Sundelöf et al., 2013). Stock analyses based on the rather limited data on lobster indicate that the fishing pressure is high and that the Swedish lobster population is at historically low levels (SLU Aqua, 2020). Consequently, NTZs were hypothesised to have a good potential for contributing to the recovery of local populations of lobster and cod.

The main purpose of the implementation of the NTZs east of Vinga in 2010 was to restore the local lobster population, increasing the density and size of individuals to contribute to a "spillover effect" to the surrounding fishing grounds, both by the export of lobster larvae and by migration of adult individuals. Lobster is dependent on the availability of suitable habitat where they can hide, e.g. environments with numerous holes and crevices, and if lobster densities are too high in an area, individuals may migrate to less crowded habitats to avoid competition. Therefore, by increasing the density of lobster inside an area, there will potentially be an increase in individuals also outside the NTZ, which may benefit the commercial and recreational fisheries of this highly valued species (Lenihan et al., 2021).

An additional purpose of the NTZs was to achieve a more efficient protection for the gadoid populations residing in the areas of Tanneskär and Buskär, potentially including local coastal cod populations. Finally, yet importantly, the increased densities of lobsters and cod could re-establish a top-down controlled food web, which in turn would affect the overall structure of fish and crustacean communities.

A)
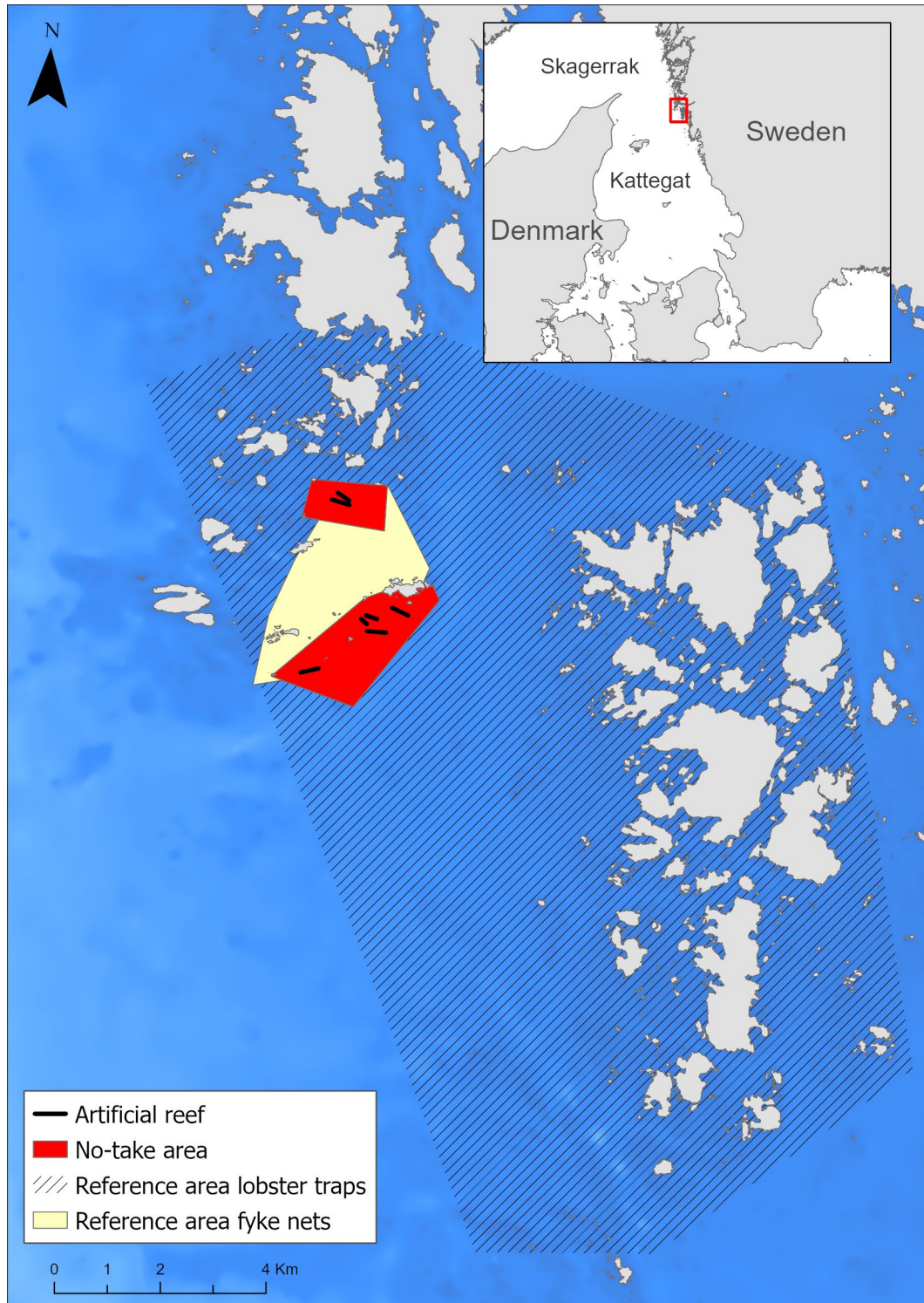
Figure 4 A. The no-take AR represented by red polygons at Vinga outside Gothenburg, Sweden. The black lines represent the locations of artificial reefs. The yellow polygon represents the location of fyke net sampling stations. Fyke net stations outside the no-take AR (the red polygons) constitute the reference, located in the area open to fishing. For the targeted lobster sampling, reference data were additionally collected from a geographically more widespread area by collaboration with local fishers (the striped polygon). Figure from Kraufvelin et al. (2022).

B)
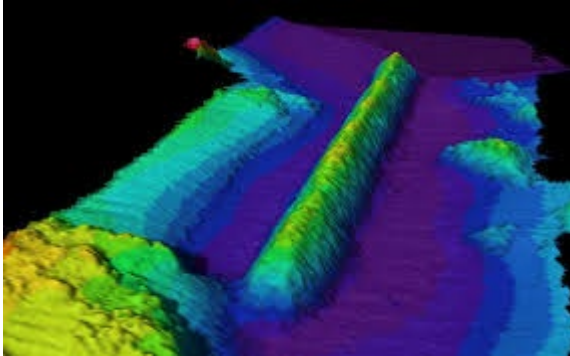
Figur 1B. A three-dimensional multibeam sonar image of reef 1A-1 Buskär at roughly 26–28 m depth. This reef is 350 m long, 45 m wide and 14 m high (above bottom) with its highest point 13.5 m below the surface. The image is reproduced with permission from the Swedish Maritime Association.

11.2. Methods

11.2.1.   Definition of goals

Prior to the establishment of the no-take zones, goals, objectives, indicators and success criteria (GOIS) were defined regarding lobster, the fish community and ecosystem functions (Table 2). To be able to determine if the goals and objectives were fulfilled after the defined time period, a tailored monitoring program was established.

11.2.2.   Quantitative surveys

Two types of quantitative survey methods were used; lobster traps and fyke nets. The lobster trap survey aimed to study the lobster abundance and size structure, and fyke nets aimed to investigate the development of the fish community and its ecosystem functions, while also contributing information on the development of the lobster population. The aim of the ecosystem function study was to determine how the large predators, such as lobster and gadoids, can affect the fish and crustacean communities by top-down control.

11.2.3.   Monitoring of lobster

A lobster trap survey was conducted within the NTZs of Tanneskär and Buskär, on both artificial reefs and in natural hard substrate habitats. A total of 20 lobster traps were used in the study area during 2003. These were of a Swedish circular design, approximately 120 cm long with two entrances, one chamber and no escape exits (Fig. 2). During the years 2004–2006, 2008–2010 and 2014, an additional 20 lobster traps were used, and these were of a Scottish semi-circular model and each trap measured 90 cm in length. Both types of lobster traps had a mesh size of 50 mm and were baited with salted flounder or herring. In both the NTZs, 10 lobster traps (five Swedish and five Scottish) were used on the artificial reefs and 10 in the natural lobster habitats. Two of the artificial reefs in Buskär were partly buried in the bottom sediment and were therefore excluded from the sampling program. In addition, the southernmost reef in Buskär was heavily exposed to wind and waves, which complicated the lobster trap fishing. Altogether, this resulted in a sampling/fishing effort of a total of 20 lobster traps distributed on the four remaining artificial reefs and 20 on natural lobster habitats. The lobster traps were to be emptied twice every week during the fishing period in October-November, but due to weather conditions, this was not always possible, especially not for the more exposed NTZ in Buskär. This resulted in that each lobster trap was hauled on average 8.4 times annually.

Figure 2. The lobster traps that were used in the fishing surveys in the two no-take zoness in Vinga. The left trap is a Scottish semi-circular trap and the right is a Swedish circular trap. The Scottish lobster trap has a length of 90 cm and width of 40 cm. The Swedish one measures 120 cm in length and has a diameter of 40 cm. Both types of lobster traps have two entrances, one chamber, and a mesh size of 50 mm. Photo: Mats Ulmestrand

The lobster trap survey was conducted at 12–18 m depth on the artificial reefs, and at 12–20 m depth on natural hard substrate, in both NTZs. All lobsters that were caught were sexed, carapace length measured, and each individual was tagged with a numbered Floy Anchor Tag (Fig. 3). Tags were attached behind the edge of the carapace, which ensures that the tag remains also after molting. This method is widely used in tagging studies of decapod crusteaceans, and has also successfully been used in a study in the lobster protection area Kåvra further north on the Swedish west coast (Ulmestrand, 1996). All lobsters were released at the same site where they were caught.

Figure 3. Lobsters tagged with Floy Anchor Tags.

Because the initial purpose of the lobster reef project was to document the lobster colonisation of the artificial reefs, there were no fishing surveys conducted in the reference areas outside the NTZs (where fishing is allowed). Therefore, to be able to investigate a potential effect of the fishing ban upon the lobster population, information of catches per lobster trap was derived from two lobster fishers. One of the fishers was a recreational fisher, using a total of 28 lobster traps in the area surrounding the closed areas, and the other fisher was a commercial fisher with a total of 50 lobster traps, active in a wider area in the southern archipelago of Gothenburg. This study design meant that the recaptures of tagged lobsters in the NTZs were obtained from the annual fishing surveys, whereas recaptures outside NTZs were dependent upon reports from the public.

Additional data available were lobster catches using the fyke nets (section 13.2.4), where fishing surveys were conducted both within the NTZs and in the reference area located between them, where fishing is allowed. The difference in catches between the open and closed areas enabled a measure of the combined effects of artificial reefs and fishing regulations upon the lobster populations.

11.2.4. Monitoring of fish communities and ecosystem functions

The quantitative fishing surveys by fyke nets were conducted in October during the years 2002–2006, and in 2015. The overall purpose of the survey was to monitor the development of the demersal fish and crustacean communities over time. The aim of the 2015 sampling was to obtain information of any potential long-term effects of the NTZs. Figure 1 shows the areas for fyke net sampling that were used in the study, within the NTZs and in the adjacent reference area. The fyke nets were standardised Swedish national monitoring fyke nets with a semi-circular opening of 55 cm height, two long arms of five meters and one chamber (Bergström & Karlsson, 2016). Each station was fished by sixfyke nets connected arm-by-arm, with the exception of 2002, when five fyke nets were used at each station. The fishing time at each station was 24 hours, but some stations were fished for 48 h in 2002 due to bad weather conditions. Differences in fishing effort, number of fyke nets and fishing time were accounted for in the result calculations. All individuals caught during the survey period were identified to the lowest taxonomical level (species) and length was measured, after which they were released. All data derived from fishing surveys that were disturbed by bad weather were omitted from the statistical analyses. The number of undisturbed fishing stations per year varied between 20 –34 within the NTZs, and between 16–27 in the neighbouring reference area (Table 1).

Table 5. The number of undisturbed stations that were fished by fyke nets in the two NTZs in Vinga and the neighbouring reference area (open to fishing).

Year	No-take zone	Reference area
2002	24	22
2003	32	27
2004	20	16
2005	30	23
2006	34	25
2015	34	26

11.2.5. Statistical analyses

To assess the development of lobster abundance and size distribution over time in the NTZs, ANCOVA models were applied with 'site', 'year' and the interaction term 'site * year' as predictor variables. The interaction effect is the key aspect of the models and a significant p-value equates to an effect of the NTZ on the response variable (abundance and size of lobsters). Separate models were run to compare

lobster catches in 1) artificial reefs versus natural reef habitats inside the NTZs, 2) artificial reefs in the NTZs versus natural reef habitats in the reference area, and 3) natural reef habitats in the NTZs versus natural reef habitats in the reference area. Furthermore, potential changes in the sizes of female and male lobsters over time on the artificial reefs in the NTZ were investigated by linear regression. To assess changes in species community composition over time, catch data from the fyke net surveys were used from the years 2002-2006 and 2015 (mean values per species and year in the NTZs and the reference area). A multivariate principle coordinates analysis (PCO) was conducted and a two-way PERMANOVA, both using Bray-Curtis similarity index. Prior to analysis, the data was square-root transformed and pelagic fishes and small crustaceans were excluded, because these are generally not representatively caught in fyke nets (Bergström et al., 2016). ANCOVA models were applied to determine potential effects of the NTZs on lobsters (total abundance of lobster and abundance of large lobsters [> 23 cm body length]), certain aspects of the fish communities (total abundance of large fishes [> 30 cm]), abundance of predatory fishes, abundance of large cod [> 30 cm], abundance of poor cod) and additional crustacean communities (abundance of small crustaceans and abundance of edible crabs).

11.3. Results

A summary of the results in relation to the goals (GOIS) that were defined for the NTZs are listed in Table 2.

Table 6. An overview of the results from both the short- and long term surveys of the lobster and fish communities in Vinga. References to the figures and tables where the results can be found are included in the table.

Goals	Objectives	Indicators	Success criteria	Method	Short-term effects	Long-term effects	Reference
Increase in local lobster production	Increase in population densities	Lobster abundance	Increase in lobster CPUE over time	Lobster traps	Increase in lobster CPUE over time in the NTZs compared to the reference area	CPUE continues to increase over time in the NTZs compared to the reference area	Section 11.3.1. Fig. 4
	Increase in spawning stock biomass	Mean length and CPUE	Spawning stock biomass increases over time	Lobster traps and fyke nets	Mean length of females increases over time, the abundance increases over time within the NTZs compared to the reference area, which means that egg production has increased since the implementation of the NTZs	Mean length of females continues to increase over time. Abundance continues to increase over time in NTZs compared to reference area, which means that egg production continues to increase.	Section 11.3.1 Section 11.3.2 Fig. 4-6
	Diversified size structure	Size structure	Increase in abundance of large individuals over time	Lobster traps and fyke nets	Mean length of both males and females increases over time, catches of large lobsters increases compared to the reference area	Mean length of both males and females continues to increase over time, catches of large lobsters continues to increase compared to the reference area	Section 11.3.2 Fig. 6 Section 11.1.3 Fig. 7
	Increase in recruitment	Egg production,	Increase in egg production	Estimate based on abundance and size structure	Increase in egg production over time in NTZs	Continued increase in egg production over time in NTZs	
	Spillover of adults	Recapture of tagged lobster	The rate of tagged individuals increases in the surrounding areas	Tagging, lobster trap and fyke net survey	Repcaptures outside NTZs only 7%	Lobsters are very stationary and migration rates are low	Section 11.3.5
Fish and lobster communities undisturbed by fishing/Re establishment of ecosystem function	Diversified size structure	Size structure	Increase in abundance of large individuals over time	Fyke net survey	Increase in CPUE of large lobsters compared to reference area. Increase in CPUE of large fishes and large cod	Continued increasing CPUE of large lobsters compared to reference area. No difference in development for large fishes or large cod, but consistently larger CPUE in NTZs	Section 11.3.3 Fig. 8 Section 11.2.1 Fig. 10
	Increase in abundance of predatory fishes	Mean trophic level	Increase in mean trophic level	Fyke net survey	No indication of increase in mean trophic level	No indication of increase in mean trophic level	Section 11.3.2
	Increase in abundance of predatory fishes	CPUE of predatory fish	Increased CPUE of predatory fish	Fyke net survey	No indication of increased CPUE of cod or total predatory fish, but a tendency towards increase of poor cod in NTZs compared to reference area	No indication of increased CPUE of cod or total predatory fish, but a tendency towards increase of poor cod in NTZs compared to reference area	Section 11.3.2 Fig. 11
	Decrease in abundance of smaller crustaceans	Abundance decapod crustaceans	Abundance decreases over time	Fyke net survey	The abundance of smaller crustaceans decreases	The abundance of smaller crustaceans were continuously low	Section 11.3.3 Fig. 12

					significantly in the NTZs compared to the reference area	in the NTZs compared to the reference area. No large changes in community composition of fish and crustaceans compared to 2006	

11.3.1.   Increase in local lobster production

Increase in population density

Lobster catches in the trap survey were significantly higher on the artificial reefs than in natural habitats in the NTZs during the autumn surveys in 2004–2006 (Fig. 4). Likewise, the abundance of lobster increased in the natural habitats within the NTZs. Overall, lobster abundance has increased in the NTZs compared to control areas since the establishment of the NTZs. In the follow-up study in 2014 to examine long-term effects, lobster abundance was the highest documented on both the artificial reefs and on the natural hard substrate habitats compared to the previous years (Fig. 4). Since the autumn of 2008, there have been no significant differences in lobster abundance among the artificial reefs and the natural habitats. During the complete study period, 2003-2014, catches of lobster have increased in the no-take zone on both artificial and natural reef habitats compared to the reference area (ANCOVA; interaction Site * Year, $p > 0.001$). In the reference area, which was open to fishing, catches of lobster did not increase over time (Fig. 4). The abundance of lobster within the NTZs was approximately 3–3.5 times higher than in the reference area at the end of the study period in 2014. This difference was attributed to the effects of the fishing ban that was implemented in 2003, when population densities of lobster were equal in all areas.

In line with the trap net survey, the results from the fyke net survey showed that the abundance of lobster increased in the NTZs compared to the reference area (ANCOVA; Interaction Site * Year, $F = 8.08$, $p < 0.01$) (Figure 5).
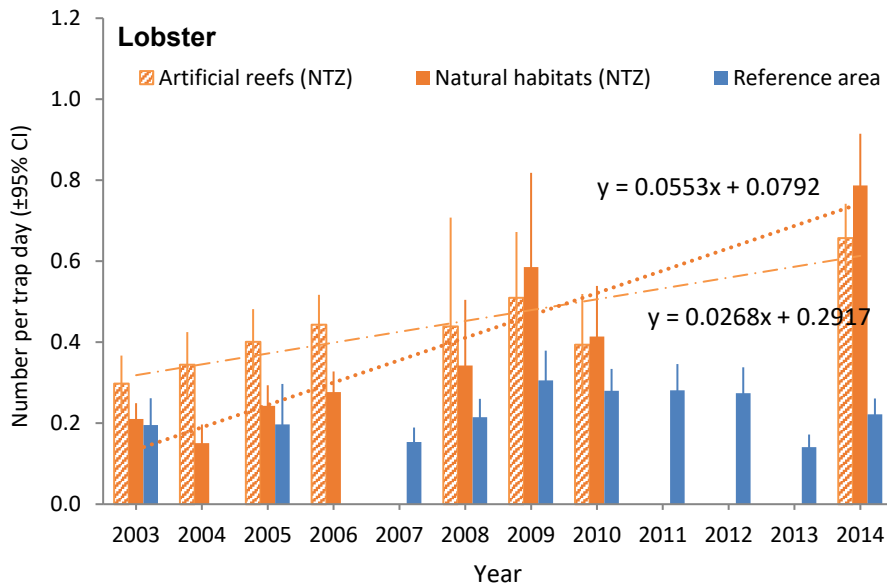
Figure 4. Number of lobster per trap and day (mean ± 95% CI) on the artificial reefs and in natural lobster habitats within the no-take zones and reference area in Vinga during 2003-2006; 2008-2010 and 2014. Dashed line (artificial reefs) and dotted line (natural habitats) are fitted regression lines of the annual mean value.
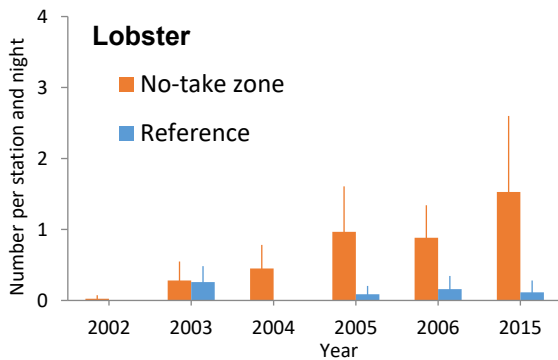
Figure 5. Number of lobster per station and night (mean ± SE) from the fyke net survey in the no-take zones and reference area during 2002–2006 and 2015. Each station was fished by five fyke nets in 2002, and six fyke nets in all other years.

Increase in spawning stock biomass

The mean length of both male and female lobster increased after the establishment of the no-take zones. In 2003, when the NTZs were established, the mean carapace length was approximately 80 mm, which corresponds to the limit for defining an individual as an adult. Since then, lobster size has increased gradually, and in 2014, the mean length was 95 mm for males and 93 mm for females (Fig. 6).

Because the mean length and abundance of females increased simultaneously over time (Fig. 6), this implies that the total spawning biomass has increased (Fig. 4, 5) and as a consequence, also the egg and larvae production.
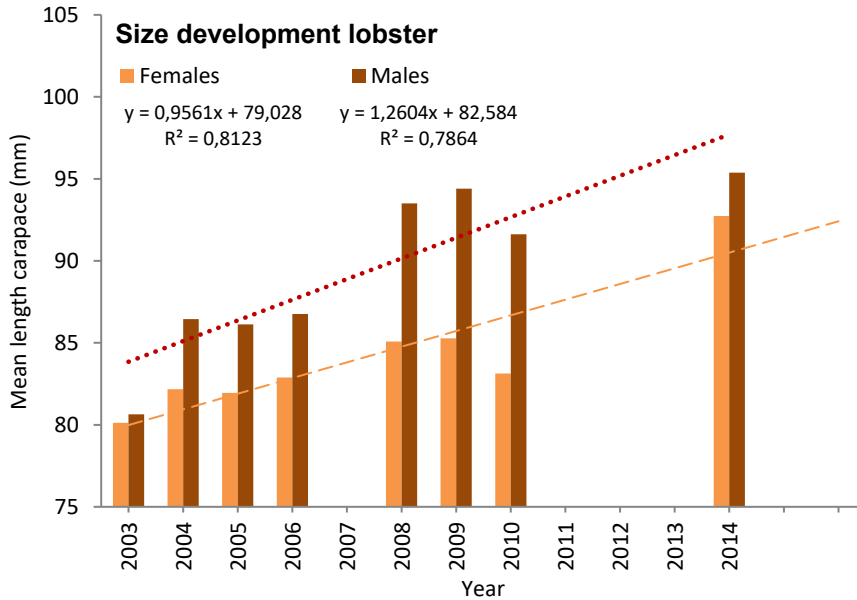
Figure 6. Mean carapace length of male and female lobsters inside the no-take zones in Vinga during 2003–2006, 2008-2010 and 2014. Dotted line (males) and dashed line (females) are fitted regression lines.

Diversified size structure

After the implementation of the NTZs, the lobster population (Fig. 5), and mean size of both males and females increased in the NTZs (Fig. 6). In addition, the abundance of adult individuals ($\geq$ 80 mm) increased within the NTZs over time (Fig. 7).

Catches of adult lobsters (>23 cm total length or >80 mm carapace length) in the fyke net survey increased significantly over time in the NTZs compared to the reference area (ANCOVA, Interaction Site * Year, F = 9.13, p < 0.01, ANOVA NTZs, F = 3.85, p < 0.01). In 2006, the catches of adult lobster were three times higher in the NTZs and by 2015 catches were six times higher than in 2003 when the NTZs were first implemented (Fig. 8).
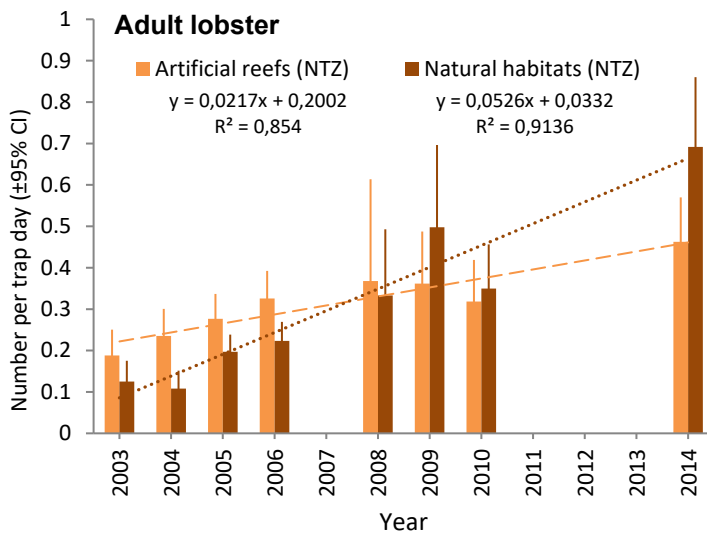
Figure 7. The number of adult lobster (≥ 80 mm carapace length) per trap and day at the artificial reefs and at the natural habitats within the NTZs in Vinga during 2003–2006, 2008–2010 and 2014. Dotted line (artificial reefs) and dashed line (natural habitats) are fitted regression lines. Error bars denote 95% confidence intervals.
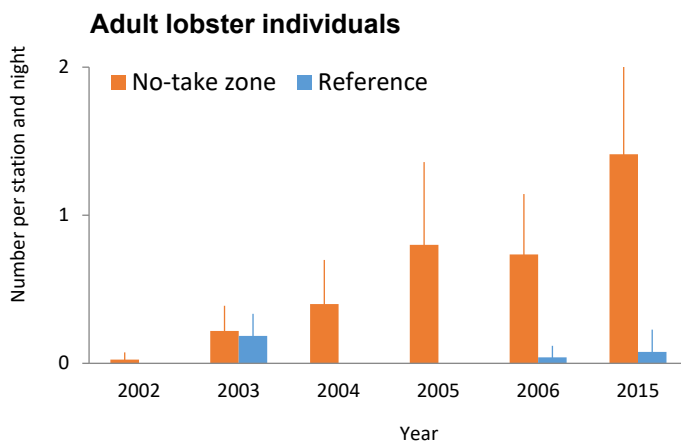

Figure 8. Number of adult lobster (mean ± SE) in the no-take zones and reference areas in the fyke net survey in 2002-2006 and 2015.

Increase in recruitment

Catches of subadult lobster (< 80 mm) did not change during the study period in the NTZs (Fig. 9). Because the trap survey was conducted within the NTZs only, no comparisons were made between the reference area and the NTZs.
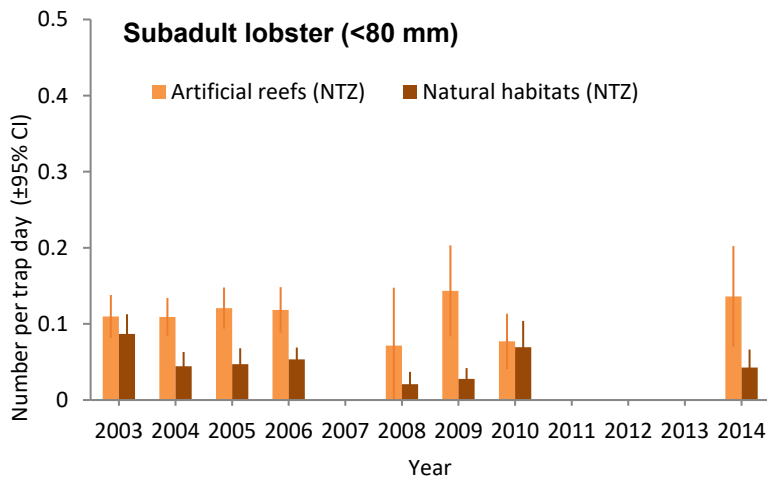
Figure 9. The number of subadult lobster (< 80 mm) per trap and day (mean ± 95 % confidence interval at the artificial reefs and the natural lobster habitats within the NTZs during 2003–2006, 2008–2010 and 2014.

Spillover of adult lobster

The lobsters in the NTZs were rather stationary during the study period (2003–2009). The tagging study resulted in a total of 1540 recaptured lobsters, of which 93.2% were caught within the no-take zones (in the lobster trap and fyke net surveys combined). Only 6.8% had migrated from the NTZs based on catch reports from the public.

11.3.2.   An undisturbed fish community

Diversified size structure

The catch of large individuals of fish (> 30 cm) in the fyke nets did not change over time in the NTZs compared to the reference area (ANCOVA, Interaction Site * Year, F = 0.42, p = 0.52). Nor did the catch of large cod (> 30 cm), show any difference between the areas (ANCOVA, Interaction Site * Year, F = 0.00, p = 1.00), but the catch was generally higher inside the NTZs than in the reference area (Fig. 10). For both total individuals of large fish and cod only, there was an increase in abundance over time during the first three years (2003–2005), which decreased after 2005 and onwards (Fig. 10). In 2015, the catches were almost at the same levels as in the first year of the study (2003).

Increase in abundance of predatory fish

The mean trophic level of the fish community did not change over time in the NTZs (linear regression, F = 1.32, p = 0.32). No significant differences over time in catches of predatory fishes in the NTZs and reference areas were found (ANCOVA,

Interaction Site * Year, F = 1.68, p = 0.20) (Fig. 11). The analysis indicated trophic level increased in the reference area, which was attributed to the large catches of predatory fish in 2015. These catches constituted of high numbers of young saithe in two stations (ANOVA, reference area, F = 4.33, p < 0.01), and was probably an effect of a random occurrence of a large school of fish, rather than a protection effect.

The most influential/important predatory fishes in the area were cod and poor cod. There was no difference in the catch development over time between the two management regimes for cod (all sizes), or for total predatory fishes. For poor cod, there was a trend towards an increase in the NTZs compared to the reference area (ANCOVA; Interaction Site * Year, F = 2.63, p = 0.10).
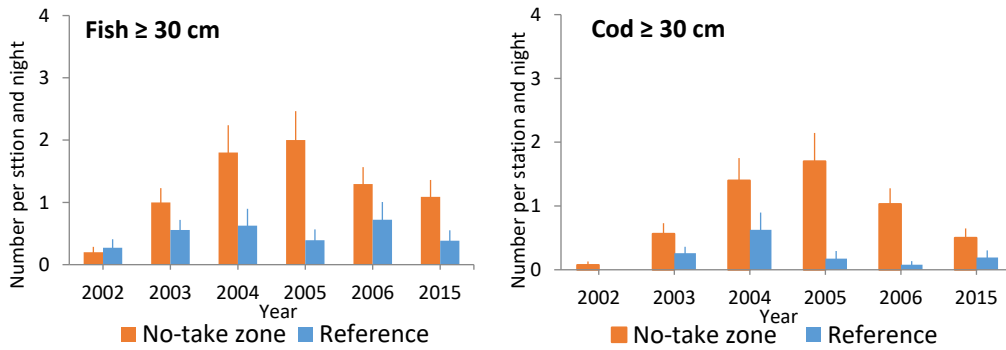
Figure 10. Catches in fyke nets (mean ± 95% confidence intervals) of large fish (> 30 cm, all species pooled), and of cod (> 30 cm) during 2002–2006 and 2015 in the no-take zones and the reference area.
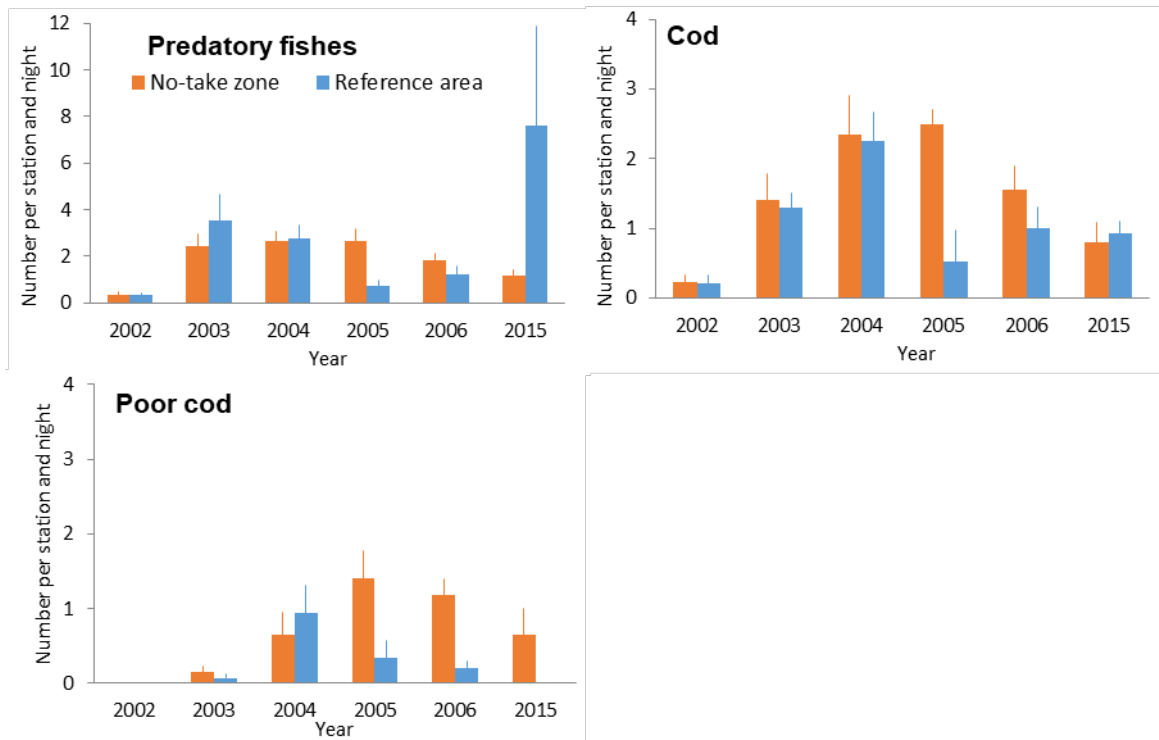
Figure 11. Catches of predatory fish in fyke nets (mean ± 95% confidence intervals) in the NTZs and in the reference area.

11.3.3.   Re-establishment of ecosystem function

Decrease in abundance of smaller crustaceans

There was a decrease in abundance of smaller decapod crustaceans in the NTZs, and an overall divergence in community composition between the two management regimes (Fig. 12, ANCOVA, Site * Year, F = 8.11, p < 0.01). Simultaneously, the abundance of edible crab increased in the reference area but not in the NTZs (ANCOVA, Site * Year, F = 3.01, p = 0.08).

To further investigate the development of the demersal fish and crustacean communities over time in the NTZs compared to the reference areas, a multivariate principal coordinate (PCO) analysis was conducted. The PCO revealed substantial changes in community composition of fish and crustaceans during the first years after the establishment of the artificial reefs and the NTZs in Tanneskär and Buskär (Fig. 13). However, over time the communities became similar again, and the community composition in 2015 is very similar to the results in 2006 in both areas (Fig. 13). The community composition in the NTZs was characterised by an increase in abundance of corkwing wrasse (*Symphodus melops*), rock cook (*Centrolabrus exoletus*), lobster and poor cod, whereas community composition in

the reference areas was characterised by high abundances of smaller crustaceans (Fig. 13).

To take certain aspects of environmental conditions that may influence the results of the study into consideration, temperature was measured during the fyke net fishing. The mean temperature varied among years, between 10.3 and 15.1 °C, but there was no difference in temperature between the no-take zones and the reference area (Fig. 14). Therefore, temperature was not likely to have an effect on the divergent community development in the areas.
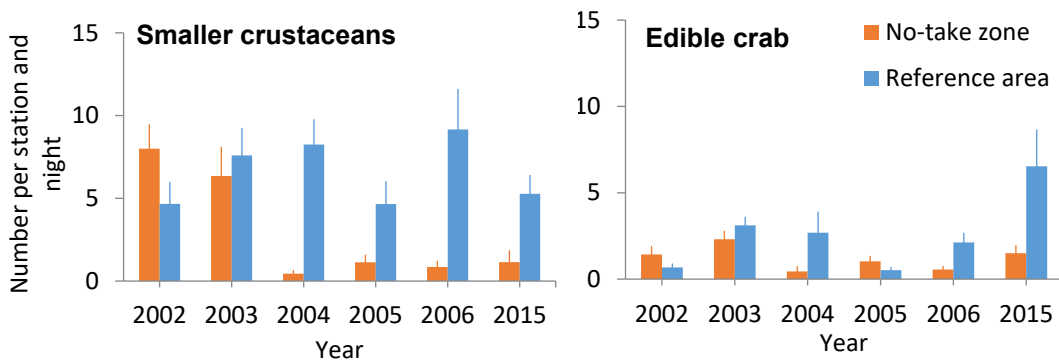
Figure 12. Catches of small crustaceans (mean ± 95% confidence interval) in the no-take zones and the reference area.
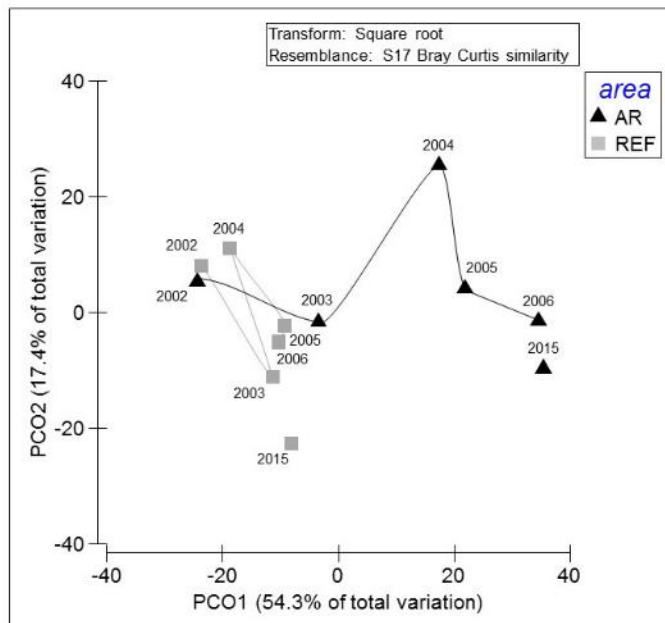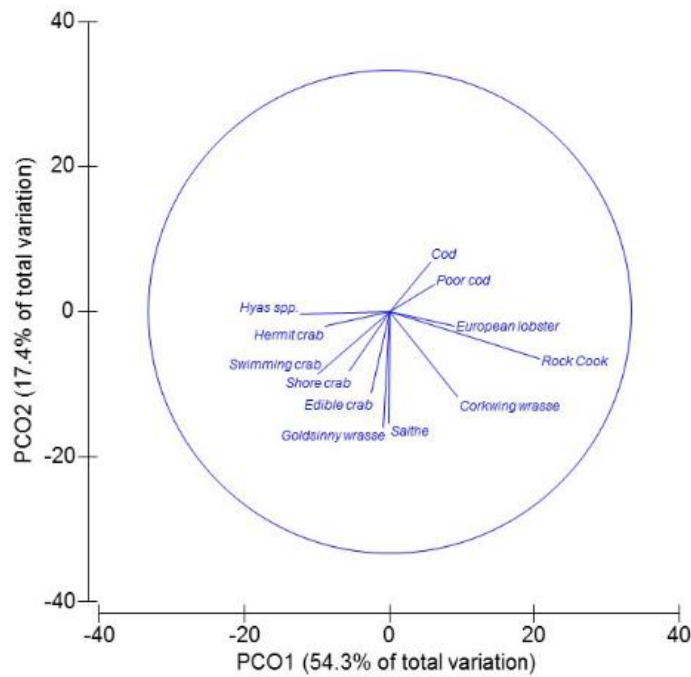
Figure 13. Multivariate PCO-analysis showing the development of the demersal fish and crustacean community composition in the no-take zones and the reference area over time. The community displays a clear development pattern over time in the no-take zones whereas the community in the reference area did not change substantially. The lower graph shows which species that contributed the most to the observed dissimilarities and in which directions. AR = no-take zone. The PCO-ordinations are taken from Kraufvelin et al. (2022).
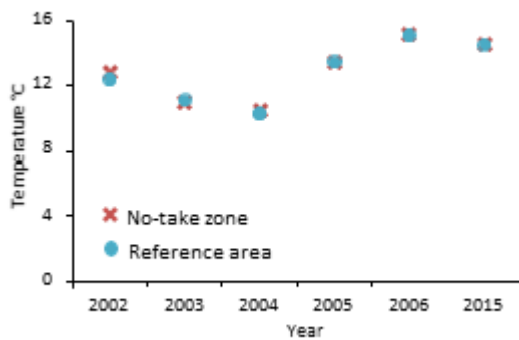
Figure 14. Mean temperature at the bottom for all stations in the no-take zones and the reference area. Temperature was measured each time the fyke nets were emptied.

11.4. Discussion

The NTZs in combination with a number of artificial reefs in Vinga in the Gothenburg archipelago have after 11 years of total closure contributed to several positive effects. The most obvious effects were observed for lobster, where an increase in both abundance and size occurred within a few years and continued to increase over time. There were also effects of the NTZs on small crustaceans and demersal fish (Table 2). The study shows that relatively NTZs, illustrated here by Tanneskär (1.2 km^2) and Buskär (3.2 km^2), can strengthen local populations of lobster, which in turn may contribute to restoring top-down control in local food webs.

In the NTZs in Vinga, there was an increase in population abundance of lobster. In the surrounding archipelago area where there is an active recreational fishery for lobster, abundances remained unchanged, indicating that the increases in lobster catches in the NTZs is an effect of the fishing ban. In a similar study of NTZs conducted at the Norwegian Skagerrak coast, the catch per unit effort (CPUE) of lobsters increased on average by 245% in the protected areas four years after establishment, whereas the increase in the reference area was 87% (Moland et al., 2013a). In the present study, a similar increase in CPUE was observed in the NTZs; 120% on the artificial reefs and 275% in the natural lobster habitats. In the Norwegian study, lobster length increased over time in the protected areas (Moland et al., 2013a), which were similar to the observations in the NTZs. The combination of increased lobster size and abundance in the Swedish study area suggests that the potential egg production per female has also increased, as this is proportional to female weight (Agnalt 2007).

In the first years (2004–2006) after the establishment of the NTZs, lobster abundance was higher on the artificial reefs than in the natural habitats. From 2008 and onwards, this difference disappeared, which implies that the increase in lobster abundance was mainly explained by the fishing ban rather than differences between artificial and natural habitats. A previous study in the Kåvra lobster reserve in the Brofjorden estuary on the Swedish Skagerrak coast shows a similar pattern (Moland et al., 2013b). For the species that were evaluated based on the fyke net fishing, it was not possible to disentangle the effects of habitat and management regime as only the artificial reef habitats were fished within the NTZs.

In the present study, the spillover effect, i.e. the migration of adult lobsters from the NTZ, which would benefit the adjacent fishery, was small, only 6.8% of recaptures were outside the NTZs. In the Kåvra lobster reserve (2.2 km^2) adult lobsters were also very stationary. Of a total of 4016 tagged lobsters, only 1.4% were caught outside the NTZ (Øresland & Ulmestrand, 2013). However, in both the present

study and in Kåvra (Øresland & Ulmestrand, 2013), the recaptures of tagged individuals inside the NTZs were based on scientific surveys, whereas the recaptures outside were dependent on reports from the public, which may lead to an underestimation of the emigration rates. Interestingly, some individuals migrated very long distances. One lobster that was tagged in the NTZ was recaptured two years later in Glommen, which is located 90 km away. Two other individuals were found in Lysekil, which is approximately 70 km from the tagging area. This could potentially also be a contributing factor to underestimations of emigration rates.

Among the many animals that display strong site fidelity, spiny lobster adults are usually more sedentary, whereas juveniles move over larger areas (Davis & Dodrill, 1980). In the present study, a higher mobility of juvenile lobsters that were attracted to the NTZs could potentially explain the increase of large lobster in Tanneskär and Buskär, although no preceding increase in juvenile lobsters was detected. Juvenile lobster are rarely caught in commercial and recreational fisheries in Sweden, because escape exits minimise the harvest of small individuals (< 80 mm). Consequently, it is possible that lobster migrations in the present study were underestimated, but further studies are needed to determine this.

In the study in Kåvra by Øresland & Ulmestrand (2013), it was also shown that lobster larvae were concentrated beneath the halocline (the salinity boundary layer), at approximately 16 m depth, and therefore retention of the pelagic larvae is high. The larval production in Kåvra was estimated to disperse over a total area of 16 km^2 (Øresland & Ulmestrand 2013). This indicates that even relatively small NTZs can function as local sources of lobster larvae, and contribute to a healthy lobster population in the near surrounding seascape. No increase in lobster catches (caused by spillover effects from the NTZs) in the reference areas of the lobster trap and fyke net surveys were observed. This lack of increase could potentially be explained by a high fishing pressure in the surrounding area, as the NTZs attracted a lot of interest from fishers. Another potential effect of lobster larvae dispersal from the NTZs could be mitigation of genetic erosion, which can arise from a size selective fishery, where only large individuals are targeted. This can potentially result in long-term declines in growth rate and smaller size at sexual maturity.

For large fish, no long-term effect of the NTZs was observed, with the exception of high catches of saithe in the reference site during 2015 at two stations. Saithe is a semi-pelagic schooling species, which has a high variation in densities among years, reflected in the results from the continuous national monitoring program using fyke nets (Ericson et al., 2016). Because saithe is not strongly associated to substrate type, and younger individuals primarily feed on zooplankton and small fishes (Pihl & Wennhage, 2002), it is not likely that this increase was related to the conditions in the NTZs.

For cod, which was the dominating species among the piscivores, there was no increase in abundances in the NTZs during the study period. On the other hand, there was a tendency towards increased densities of poor cod in the NTZs, compared to the reference area. The observed increase in numbers of large cod during the first years (2002–2006), was initially interpreted as a protection effect. However, a corresponding increase was observed in the national monitoring data from the coastal trawl surveys, where the abundance of large cod peaked in 2005, thereafter it decreased again. A strong year class of cod recruits was present in the coastal and estuarine areas of Skagerrak in 2003, but densities of both small and large cod were depleted again in 2006 (Havs- och vattenmyndigheten, 2012). The lack of a NTZ-effect upon densities of large predatory fishes aligns with the results from Havstensfjorden (chapter 14, this report). The limited size of the NTZ compared to the home range of cod could be an explanation for the lack of a detectable recovery (Moffitt et al., 2009; Abecasis et al., 2014).

The abundance of small crustaceans decreased within the NTZs in Tanneskär and Buskär, but not in the adjacent reference area. This decrease could be an effect of the increase in abundance of lobsters and poor cod, because small crustaceans are an important food resource for both these species (Baden et al., 1990; Armstrong, 1982). Previous studies have shown that mesopredators, such as smaller fishes and crabs, have increased along the Swedish west coast, probably as a consequence of decreased abundances of large predatory fishes, i.e. a release of top-down control. This agrees with previous observations of the importance of large predators for ecosystem structure and dynamics of the whole coastal food web (Eriksson et al., 2011; Baden et al., 2012). The present study indicates that not only large fish like cod, but also lobster, which has declined along the west coast (Sundelöf et al., 2013), has the potential to impact densities of small crustaceans.

The abundance of edible crab increased in the reference area but not in the NTZs. This pattern could partly be explained by the fact that edible crab generally avoid entering fishing gear where lobsters are present (Addison & Bannister, 1998). However, this was observed in lobster traps, whereas in the present fyke nets were used, which are much larger (six connected fyke nets per sampling station). Because the mean catch per station was rather low (approximately one lobster), avoidance behaviour should not have affected the catches of edible crab. This was further supported by the lack of a negative relationship between the two species within the same station. Therefore, a plausible explanation is that the decline in edible crab is due to species interactions, with increased numbers of lobster decreasing the abundance of the edible crab in the NTZs. This would add to the growing body of evidence that different species of lobster can have a large impact upon other species in its habitat (Boudreau and Worm, 2012; Guenther et al., 2012). Large crustaceans, such as crabs and lobsters are known to compete over suitable habitat such as

shelter, and to predate upon each other, potentially affecting the abundance of each other (Hoskin et al., 2011; Boudreau and Worm, 2012). In summary, the study exemplifies that NTZs can be quite efficient to support assemblages of fish and large crustaceans that have small home ranges and display high site fidelity, and that positive results can be seen within relatively short time periods (2–4 years). Here, the increased abundance of the commercially important lobster, particularly large individuals, indicates a direct relationship to decreased fishing pressure. The results, further, indicate that local restoration of large predator populations may also initiate regulatory functions in the food web, as seen by re-established top-down control in the NTZ. The study demonstrates how local measures can be used to alleviate the impact of human-induced physical and biological pressures related to overfishing and as such constitutes an example of the principles and potential benefits of marine conservation. However, the Vinga NTZ also serves as a demonstration of successful species conservation in an intensively utilized marine region, where many populations and habitats are under high pressure from human activities. Continuation of the Vinga NTZ and studies to follow up on potential long term ecosystem-level effects are needed to see to what extent the responses observed in this study continue, and if they can be manifested in the food web at larger spatial scales. Combining restoration efforts, such as the reconstruction of reefs with species and habitat protection, as in the case of the Vinga NTZ, can be important for creating refugia for large predatory fish and crustaceans as part of an ecosystem-based management. These measures are also important *per se* for facilitating scientific investigations of the importance of reef habitats in coastal environments, where large knowledge gaps occur for the North-eastern Atlantic and in the Baltic Sea.

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12. No-take zone for cod and flatfish in the Havstensfjord, Skagerrak

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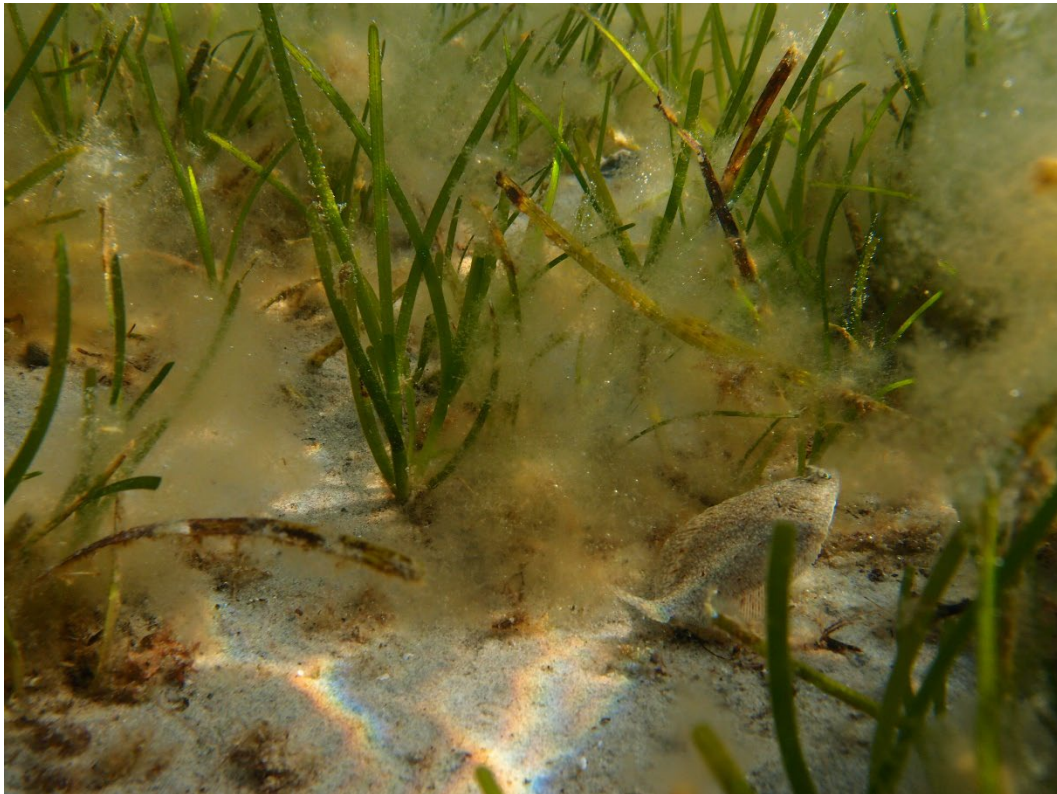
Photo: Ulf Bergström

Summary

The Havstensfjord is located between the island Orust and the Swedish mainland on the west coast. It was selected as a MPA (Marine Protected Area) based on the need of protection for local fish populations combined with the history of local management initiatives in the region. The Havstensfjord and surroundings have a history as a productive fishing area with substantial historical landings of commercial species such as cod, plaice, herring and sprat as far back as the 1960s. However, many of these commercially important species have since then decreased in density distribution, and size composition. The decline or disappearance of the coastal fish populations is an effect of an intensified fishing pressure during the last 50 to 60 years.

Based on the area´s potential for fish production, a no-take zone was established in the Havstensfjord in February 2010. The no-take zone (referred to as NTZ) was designated for protection of mainly turbot, cod and plaice and created conditions for a potential recovery of local fish populations. It includes two small NTZs, where there is a total fishing ban in the northern NTZ and a fishing ban except for handheld poles used from a shore in the southern NTZ. The buffer zone surrounding the NTZ only allow fishery with selective gear, such as creels/pots and hook and line, enabling gentle release of non-target species. Fishing of cod, haddock and pollock is prohibited. The total area of the NTZs and the buffer zone is 13 km^2 and 154 km^2, respectively. The MPA and NTZs has now been in place for 12 years. This report will evaluate the effect of the NTZ on cod, plaice and turbot in the sea area surrounding the NTZ and buffer zones.

Within the NTZ there has been an increase in the abundance of cod from 2013-2021, due to recruitment peaks during some years. However, these effects are not reflected in cod biomass. The recruitment peaks are also seen within the reference areas, which indicates the pattern is not clearly linked to the NTZ and buffer zone. There were low numbers of large cod ($\geq$ 40 cm), in the NTZ and buffer zone, with several years with zero catches. However, fish egg surveys show annual presence of cod eggs in the NTZ and buffer zone, including findings of the earliest development stages, which suggests spawning in the area. Parallel to this, large individuals of cod have been documented by Stereo-BRUVs (baited remote underwater stereo-video systems) in the NTZ but in very low numbers and mainly on rocky substrates where the use of the bottom trawl data is limited. This gives some support to the results from the egg survey by documenting the presence of large fish in the area. The large individuals are very important to protect in order to enable a future recovery of the species. Larger individuals produce higher quality and more viable eggs (Trippel et al., 1997) as well as higher sperm volume (Trippel and Morgan, 1994). Furthermore, large predatory fish are important for composition and function of food webs. Another purpose of the study has been to implement non-harmful assessment tools like the BRUVs, to continue monitoring of the target species in the area. However, Stereo-BRUV is still a novel method of monitoring predatory fish in ecosystems such as the Swedish fjords. In order to continue evaluation and development of the method it is recommended that data collection with stereo-BRUVs in the Havstensfjord continues as a supplement to other ongoing monitoring surveys.

The evaluation for plaice shows that there were no trends during the period 2010-2021 for abundance, biomass, size distribution or presence of large individuals ($\geq$ 24 cm). However, when the full time series of coastal survey data from 2002 to

2021 is considered, the biomass of plaice declined, along with large individuals (both biomass and abundance) and size distribution.

The results for turbot show no trend for abundance or biomass during the period 2010 to 2021. Turbot is a rare species along the coast of Bohuslän but is documented annually within the NTZ and buffer zone in low but consistent levels. Large individuals ($\geq$ 35 cm) and size distribution of turbot, show a negative trend during the period 2010 to 2021. No large turbot have been documented in the coastal survey within the NTZ and the buffer zone during the period 2019 to 2021. The results indicate a negative trend for the species the last 12 years in the NTZ and the buffer zone.

In summary after 12 years, the NTZ and buffer zones within the Havstensfjord have not contributed to a measurable recovery of demersal fish stocks. This does not necessarily mean that NTZs are ineffective as a management strategy. For heavily decimated stocks and species with long life cycles, an evaluation period of 12 years can be a too short time frame. The generation time for e.g. cod and plaice, two of the focal species in the area, are approximately 7 to 10 years and they reach sexual maturity after 2 to 4 years. To obtain a quick response from the implementation of a NTZ after a long period of overfishing, it requires high recruitment of the depleted species and a low fishing mortality. The combination of these two factors can explain why no measurable effect has been documented so far. It might require several generations for a recovery, if ever, when the stocks have been heavily decimated.

It is important to continue fish monitoring in the area to obtain spatial and temporal information about focal species, as well as to develop and implement non-harmful assessment tools to minimize the impact on fish. Furthermore, it is important to monitor parallel changes in ecosystem and environmental conditions so the necessary conditions for a recovery can be evaluated.

12.1. Background

Havstensfjorden is a marine area located between the island Orust and the Swedish mainland. The area was selected as one of the MPAs along the Swedish west coast due to the need to protect local fish populations combined with the history of local management initiatives in this region. Havstensfjorden in the group of 8-fjords, which is a composition of the following fjords: By, Havsten, Kalvö, Stig, Halse, Askerö, Hake and Älgö. The 8-fjords has been subject to a local co-management initiative since 1999 called *Projekt 8-fjordar* (Projekt 8-fjordar, 2005). During the autumn of 2004 the municipalities Uddevalla, Orust, Stenugnsund, Tjörn and

Kungälv – together with the Swedish anglers association (Sportfiskarna) and The Swedish Society for Nature Conservation (Naturskyddsföreningen), received government funding for the nature conservation project *Projekt 8-fjordar*. The purpose of the project is to work with measures to improve the marine environment and the status of the fish populations in the 8-fjords. The collaboration within the *Projekt 8-fjordar* was therefore a natural way forward for the Swedish Board of Fisheries (the Swedish Agency for Marine and Water Management, SwAM, since 2012), when introducing no-take zones along the Swedish west coast. During the project the collaboration between *Projekt 8-fjordar*, the Department for Aquatic resources at the Swedish University of Agricultural Sciences (SLU Aqua) and SwAM has continued.

12.1.1. Historical perspective from the area and its fjords

The focus area, 8-fjords, is located between the large islands Orust and Tjörn. It has historically been a very productive fishing area with a well-developed local commercial and supplementary fishing industry, targeting species such as cod and plaice. The catches of cod and plaice were estimated as 130 and 54 tonnes respectively in 1962 (Table 1). The largest catches of cod were caught in the area of Havstensfjorden (Hannerz 1970).

Table 1. Landings in tonnes of fish in 1962 from the marine area surrounding Tjörn and Orust, source: Hannerz 1970.

Species	Commercial fishing	Supplementary fishing	Subsistent fishing	Recreational fishing
Sprat	193.3	0	0	0
Cod	41.5	27.6	34.9	25.7
Plaice	23.8	9.5	17.5	3.2
Eel	21.3	3.6	3	1
Herring	14.7	2.1	0.5	0
Garfish	13.1	10.1	0.1	0.1
Flounder	4.9	3.7	7.2	4.3
Pollock	3.9	0.8	0.8	1.3
Sea trout	1.5	0.8	3	1
Mackerel	1.4	3	9.4	9.9
Turbot/Brill	0.8	0.3	0.8	0.6
Loster	0.6	0.4	1.3	0.1
Dab	0.5	0.7	0.6	0.2
Sole	0.2	0.3	0.6	0.2
Whiting	0	0.5	3.3	8.9

The cod stocks along the Swedish west coast have since the 1960s shown a dramatic decrease in density, distribution and size composition. Today there no longer exists a commercial cod fishery along the coast of Bohuslän, in contrast to the high abundance of fishes historically recorded in the area (Svedäng 2003; Svedäng & Bardon 2003). The decline of the cod populations was already noticeable during the 1970s (Svedäng et al. 2001, Cardinale et al. 2009a, 2009b, 2012, Bartelino et al. 2012). The population decline continued and by the turn of the millennium the last commercial fishery targeting cod was discontinued, when the final fishing grounds in the Gullmarfjord and the Koljö fjord ceased to be active (Svedäng et al 2004). Parallel to the depletion of the cod population along the coast of Bohuslän, a drastic deterioration also occurred along the coast of Halland, an area that historically harboured very rich fishing waters for cod (Svedäng & Bardon 2003).

However, cod spawning continues, even though monitoring and acoustic studies show that the populations are at exceptionally low levels (Svedäng 2003, Svedäng & Svensson 2003, Svedäng et al 2004, Sköld et al 2011, Andersson et al 2021). To estimate the stock structure before the decline, focus has been on identifying the historical cod fishery during spawning time (February-March). This fishing indicates where spawning of cod might have occurred in the fjords and in the coastal waters. Along the coast of Bohuslän spawning grounds for cod have been identified by interviewing former fishermen from e.g., Bro fjord, Koljö fjord, Havstensfjorden and Gullmar fjord (Svedäng et al. 2004).

12.1.2.   Regulations of fishery in historical/chronological order

In the wake of the collapse of the demersal fish stocks along the coast of Bohuslän, a number of management measures were implemented to reverse the negative effects of overfishing. One of the measures was shifting the trawl border from 2 nautical miles to 4 nautical miles from the base line in 2004. At the same time all fishing (including recreational fishing) inside the trawl border targeting cod, haddock and pollack during January to March was prohibited.

Several restrictions within the marine areas inside Tjörn and Orust have been introduced in order to protect fish stocks from non-selective and efficient fishing methods. In January 2004 a prohibition on purse-seine fishing was implemented in Havstensfjord, By fjord, Koljö fjord and Stig fjord. Furthermore, the regulations within the area were strengthened in January 2008, when limits for fishing with gillnets and hand-held gear were established. The length of gillnet allowed per fisherman was reduced and limited to fishing only in the depth interval from 0 to

10 m. Fishing with hand-held gear was restricted to a bag limit for cod, haddock and pollock. Overall, there were a number of local and regional fishing regulations put in place before the NTZ (no-take zone) was introduced.

12.1.3. Studies in connection with the introduction of the NTZ in the Havstensfjord

Before the introduction of the NTZ, the area was monitored with yearly bottom trawls from 2000 until 2008 by the Swedish Board of Fisheries (Fiskeriverket 2009). The monitoring found regular catches of large individuals of turbot, indicating the presence of a small but stationary population of turbot that existed in the area (Svedäng et al. 2004). Declining, and from a historical perspective, very low abundance of large individuals of fish in 2000-2008 confirmed the perception that vital populations of cod and plaice had disappeared.

The decimated populations of demersal fish led to the end of the commercial fishery targeting cod in the early 2000s (Svedäng et al. 2004). The negative development of the fishery is evident when comparing data from commercial- and recreational fisheries in the 8-fjords area from the 1960s (Hannerz 1970) or landings from 1930 until 1960 (Figure 1). The fishing district 6A (the fjord area east of Orust that includes the Havstensfjord) was important fishing area, but there was a declining trend in catches of cod (Fig. 1). On the contrary, the catches of cod generally increased during the period 1930 until 1960 along the coast of Bohuslän as opposed to the decreasing trend in the Havstensfjord. Plaice and turbot showed declining catches along the Bohuslän coast and in the Havstensfjord. Up until the 1940s there are some years with very high catches, which are not present in later years (Fig. 1). This pattern is indicative of overfishing (of plaice and turbot) in the Havstensfjord even before the 1960s.
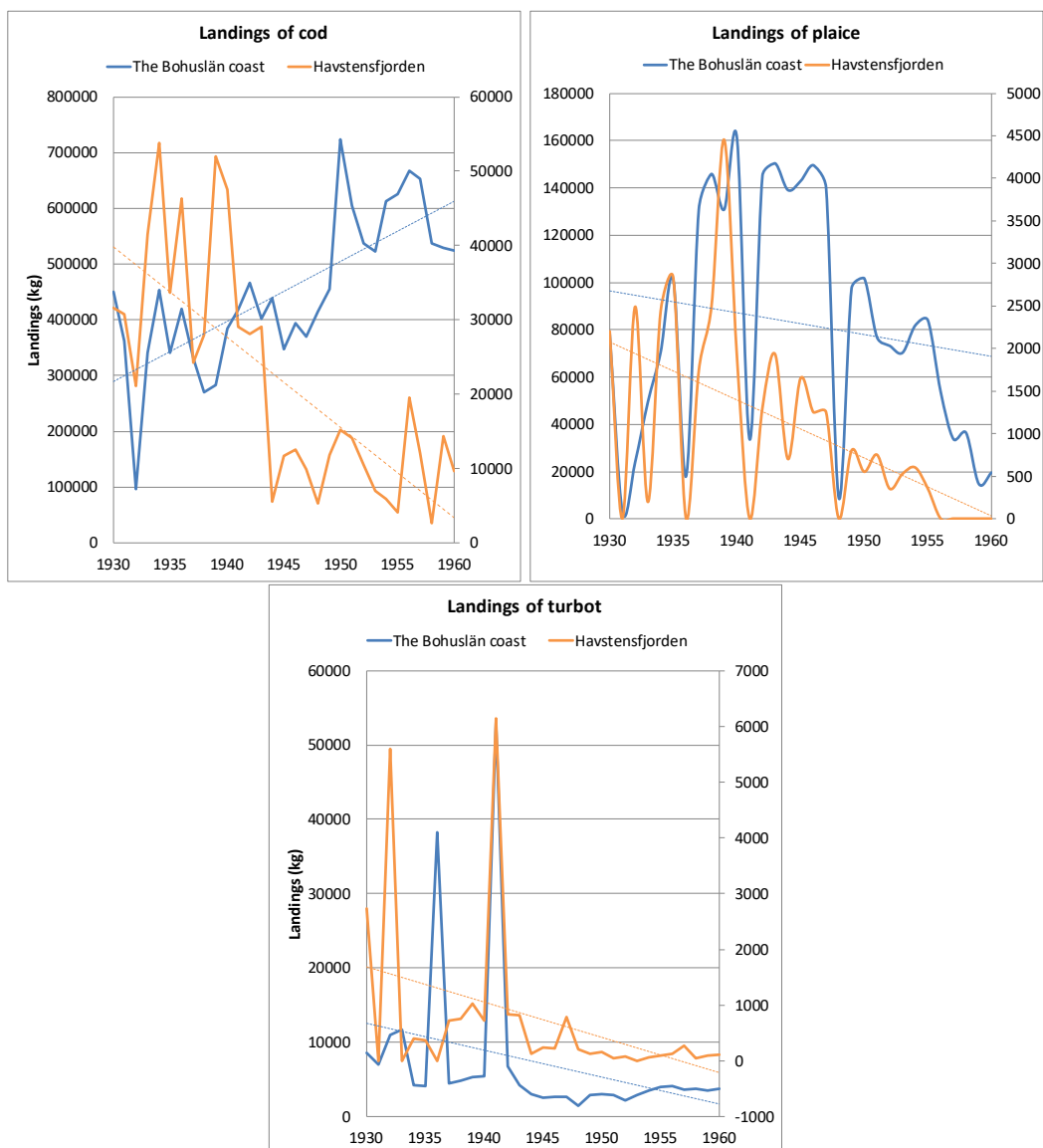
Figure 1. Presentation of fisheries landings (kg) for cod, plaice and turbot compiled by the fisheries superintendent for Gothenburg and Bohuslän for the coastal fishery on a yearly basis during 1930 to 1960. Landings are presented for district 6A, which represents Havstensfjorden (scale on the right, y-axis), and the Bohuslän coast (scale on the left, y-axis).

Prior to the establishment of the NTZ an extended study was done in the area to document abundance of predatory fish (e.g., cod) and the presence of spawning cod in Havstensfjorden. The study was conducted in a collaboration between *Projekt 8-fjordar* and SLU Aqua. The data was collected with creels/pots, bottom trawl and hook/line. In addition, an acoustic data collection was performed. The result from the bottom trawl survey could not confirm the presence of spawning cod individuals. The hydroacoustic surveys showed a low density of large fish individuals ($\geq$ 45 cm), and a non-random spatial distribution pattern of large fish (Figure 2a). However, with the creels/pots and hand held fishing poles spawning cod was caught. Together, these results indicate that cod probably reproduce and

spawn within the area (Figure 2b). The results from the studies were used to draw boundaries for the NTZ. The designated NTZ areas corresponded to historically important fishing grounds for cod. The presence of spawning cod was considered an important factor for the area selection, to enable a potential recolonization of these formerly important fishing areas.
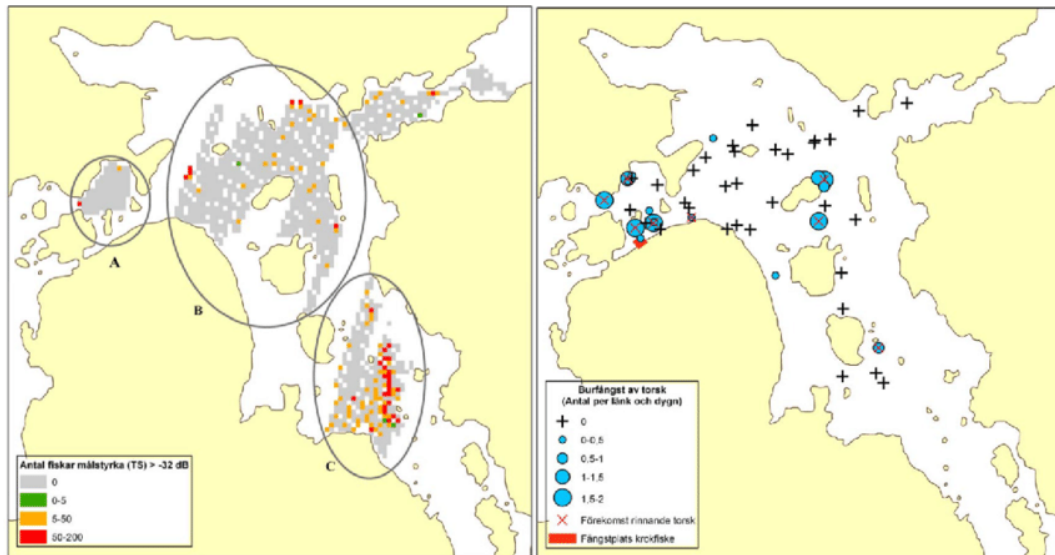
Figure 2a-b. Left figure (a) illustrates density of fish from the ecointegration, defined as number of fish with a target strength of > -32dB (≥ 45 cm total length) per hectare. Calculations per transect were made separately per area A, B & C. Left figure (b) catches of cod in creels (number of individuals per link). The size of the catch is proportional to the size of the blue circles. Red crosses represent presence and black crosses absence of spawning cod. The red zone (b) represents the single position where cod was caught with a fishing pole. Source: Fiskeriverket (2009) – Appendix 2: Pm Underlag för fiskevårdsåtgärder i fjordarna kring Tjörn & Orust.

12.1.4.   The establishment of the conservation area – cooperation with the initiative group for 8-fjordar

When the Swedish Board of Fisheries designated the MPA the steering group for 8-fjords worked with the premise that the MPA should not be too big or complex in order not to restrict the fishing opportunities for the general public (Fiskeriverket 2009). The steering group made the assessment that it is important that the new regulations had a strong support from the general public. For the regulations to be functional and respected, a design with one or a few small areas surrounded by buffer zones with less regulations was prioritized. During the process, the group behind 8-fjords had a suggestion for a designated MPA and regulations of fishery in a surrounding buffer zone. Based on these premises the Swedish Board of Fisheries proposed a NTZ with surrounding buffer zones. The proposal was discussed with different stakeholders such as the Swedish anglers association, subsistence fishery, coastal fishery and the Swedish Fishermen´s National

Association. Furthermore, Swedish Board of Fisheries held several consultation meetings during the autumn of 2009 in Stenungsund and Uddevalla, where all concerned stakeholders had the opportunity to discuss the background material and comment on the proposal from the working group behind 8-fjords. Based on the area´s fish production potential and the suggestion for a designated NTZ from *Projekt 8-fjordar*, a decision was taken to introduce a NTZ in a part of the Havstensfjord starting the first of February in 2010. The NTZ was designated for protection of mainly turbot, cod and plaice and aim to create conditions for the recovery of local populations. The MPA is made of two small no-take zones (NTZ) surrounded by a large buffer zone where landings of cod, haddock and pollock are prohibited. While there is a total fishing ban in the northern NTZ, handheld fishing poles used from ashore is allowed in the southern NTZ. Only selective gear, such as creels/pots, hook and line, which enabling gentle release of non-target species are allowed in the buffer zone (Figure 3). The total area of the NTZ and the buffer zones are 13 km^2 and 154 km^2, respectively. The regulations within the buffer zones resulted in that the existing ban on purse seining fishing also includes the Halse- and Askerö fjords outside Stenungsund.

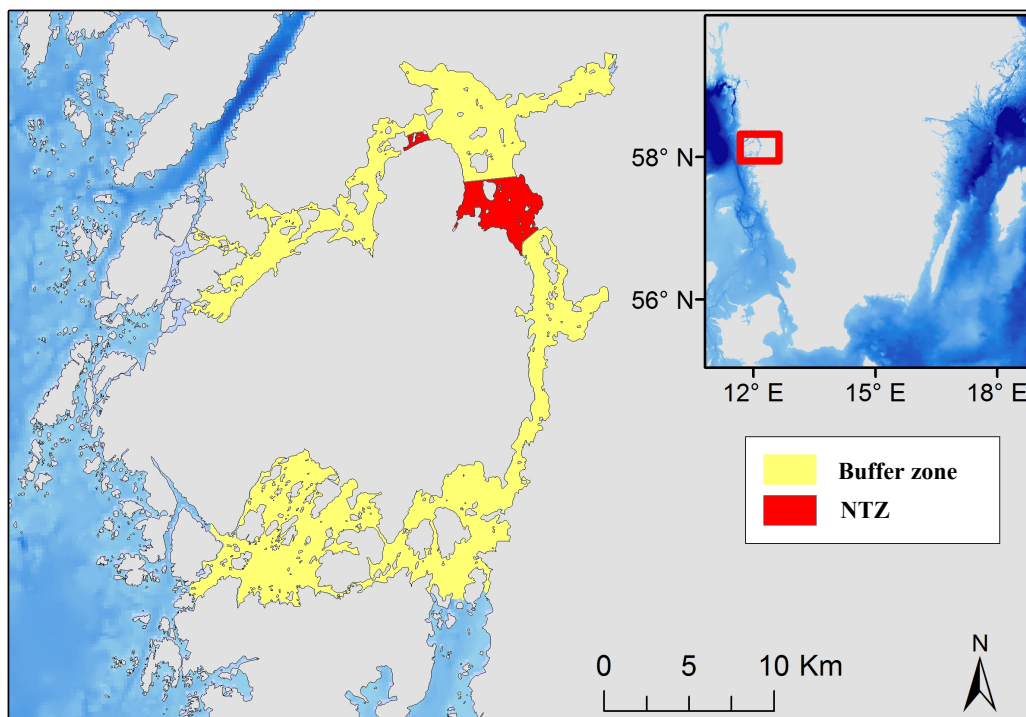*Figure 3. Illustration of the no-take zone in the Havstensfjord and its buffer zone. No-take zone (upper red area) = total fishing ban. No-take zone (lower red area): total fishing ban with the exception of handheld fishing poles from Orust and the mainland. Buffer zone (yellow area): fishing ban for cod, haddock and pollock. Fishing is only allowed with handheld fishing poles and pots/creels targeting shellfish/crustaceans.*

12.1.5.   Purpose of this report

The NTZ and buffer zone was introduced to the Havstensfjord in the beginning of 2010 and in 2016 the effects on the target species cod, plaice and turbot were evaluated (Svedäng et al. 2016). The results did not show a recovery for the focal species. Turbot had a low but stable abundance from 2010 to 2016. The abundance of large fish individuals was very low, especially for cod and plaice. However, results from sampling of cod eggs in the area indicated spawning by cod. Catch data gathered from local lobster fishermen showed occasional by-catches of large cod (≥40 cm) in the pots. These findings were considered important and a prerequisite for the recovery of the historical local cod stock in the area. This report focuses on the long-term effects of the NTZ from 2010 to 2021.

For the first evaluation in 2016 the monitoring data used had limitations in its ability to detect small scale changes over time for decimated stocks. However, adding more sampling effort with traditional fishing methods (e.g. trawl, gillnet and fyke net) would increase mortality and was undesirable. Recently, non-destructive methods such as BRUV (baited remote underwater stereo-video systems) have been used for data collection of fish assemblages, as a complement to the traditional monitoring being conducted in the area. This report will present and evaluate this data.

12.2. Methods

During the designation of the NTZ and buffer zone in the Havstensfjord, several objectives, indicators and target criteria (Goals, Objectives, Indicators, Success criteria) were defined for cod, plaice and turbot in the area (Table 2, in section 14.4 Results). Monitoring programmes were installed in order to survey if the objectives had been reached. Methods for monitoring were selected to prioritise non-destructive methods due to the severe depletion of fish stocks in the area.

12.2.1.   Monitoring of demersal fish

Data on catch in the NTZ, its buffer zone and reference areas come from the Swedish coastal survey (Andersson et al. 2021). The coastal survey has collected data on fish abundance along the Swedish west coast using bottom trawl since 2001. However, the distribution of the stations, the time of the year for sampling and the number of samplings each year has shifted over time, along with the aim of the survey. In early years the survey targeted larger commercial fish species, especially cod, sampling several times a year all along the west coast. Today's survey also aims to collect smaller species and juveniles along the northern archipelagic part of

the west coast, sampling only once a year in quarter three to avoid aggregations of spawning cod. With the new aim the mesh size of the trawl (the so-called *cod end*) shifted from 70 to 16 mm in 2013. Due to these changes, extra caution had to be made interpreting the results of the trend analysis and so the period post the establishment of the NTZ (2010-2021) was divided into two time periods 2010-2012 and 2013-2021 to deal with this, (see statistics section below).

The data used in the analysis were collected in the proximity of Marstrand and northwards to the Norwegian border in 2002-2012 (October) and in 2013-2021 (August-September). Only stations with two or more samples within each of the time frames 2002-2009, 2010-2015 and 2016-2021 were included to allow comparison of the abundance before and immediately after the NTZ was established as well as longer-term effects.

Since there is only one station within the NTZ and two in the buffer zone, the NTZ and buffer zone were combined in the analysis (NTZ + buffer zone) and compared to three reference areas (Figure 4). The first reference area constitutes the stations within the nearby Gullmar fjord (n=5). The rest of the coastal stations (between the coastline and one nautical mile outside the baseline, n=14) form the second area. The third area consists of the offshore stations (between one nautical mile outside the baseline and the outer border of the exclusive economic zone (EEZ) of Sweden, n=6).

Catch is standardized and presented as catch per km^2 trawled area. The trawled area is calculated from the wing spread of the trawl and the speed and duration of the haul. An average catch per species is used in the calculations when stations have been sampled several times the same year and quarter, but different dates. In the case of more than one sample per station the same day, only data from the first haul of the day is included. Weight is calculated from length using species specific length-weight relationships[4].

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[4] https://www.fishbase.in/manual/english/FishBaseThe_LENGTH_WEIGHT_Table.htm
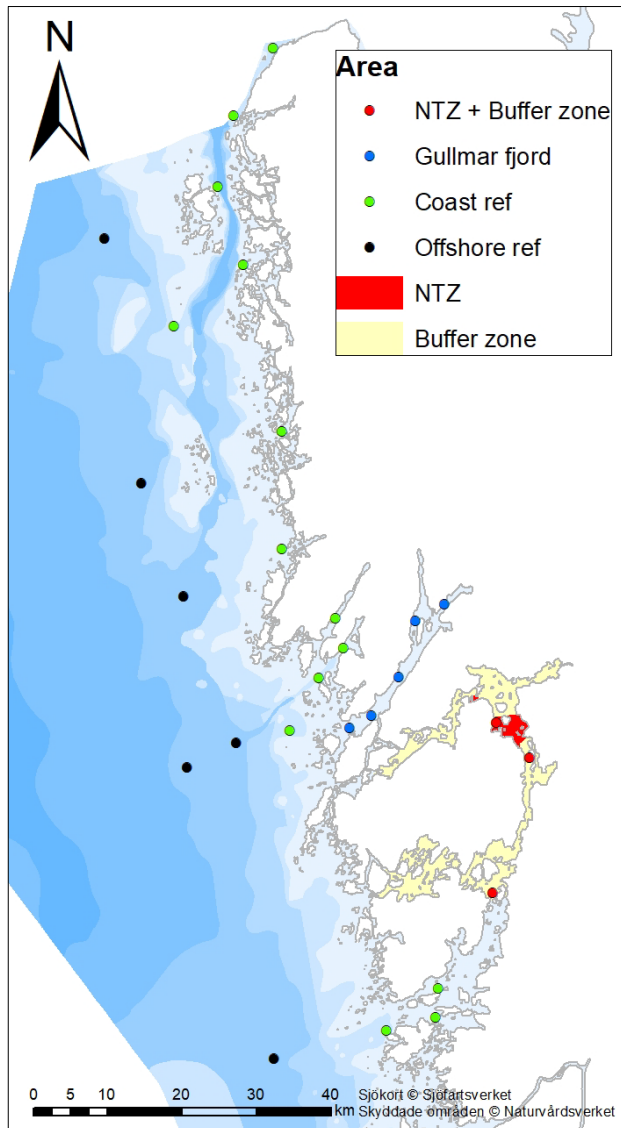
Figure 4. The sampling stations within the coastal survey used in the longer-term evaluation of the NTZ (red area) in the Havstensfjord and the buffer zone (yellow area), in the results defined as NTZ + buffer zone. The data from the coastal survey has been used to study the development of cod, plaice and turbot. Data is separated into four different areas: NTZ + buffer zone (n=3), and its reference areas, the Gullmar fjord (n=5), Coastal reference (n=14) and Offshore reference (n=6).

Statistics

Spearman´s rank correlation in R was used for trend analysis of the time series from the coastal survey (Petr Savicky 2014, R Core Team 2021), where the correlation coefficient rho (values between –1 and 1) shows the strength and direction of a significant correlation. Time series of number of length classes, biomass (kg per km^2) and abundance (individuals per km^2) of cod (total, <40 cm, ≥40 cm), plaice (total, <24 cm, ≥24 cm) and turbot (total, <35 cm, ≥35 cm) before (2002-2009) and after the establishment of the NTZ (2010-2021), as well as for the whole time series for the different areas were analysed separately. In order to take the changes in sampling season and mesh size of the trawl cod end from 70 mm to 16 mm in 2013

into consideration when analysing the data, separate analyses were also made for the periods 2010-2012 and 2013-2021 post the establishment of the NTZ were made.

12.2.2. Monitoring of spawning cod

In areas with very low abundances of cod, the fish may redistribute to be more associated to hard bottom substrates (rock- or reef areas) and/or in the water column (Righton et al. 2010). Traditionally demersal fish has being sampled with a bottom trawl. Bottom trawling is limited to soft bottom (if used over rocky- and heterogeneous bottoms the trawl run high risk of being shredded) and can therefore give an underestimation of the actual abundance of cod, when there are low densities (due to impaired catchability). Presence of spawning cod aggregations are best sampled during the spawning period (for North Sea Cod January to March). Studies of gonad maturity as an indicator of recent or coming spawning event can introduce a level of uncertainty as there is no way to ascertain if fish may actually participate in a local spawning event or if the fish was migrating to other spawning areas (reference). However, increased sampling during the spawning period using methods that cause high mortality runs the risk of further deteriorating any local spawning cod population. Sampling of fish eggs with a vertical ring net hauled through the water mass is a non-destructive sampling method to study the presence of spawning fish. The method is based on the principle that presence of newly fertilised cod eggs (or any other species) indicates spawning in the area. Eggs at early stages of development indicates that the eggs are only a few days old, and could not have travelled far from the actual spawning site. Eggs with early stages of development therefore give a good indication of spawning sites in the vicinity. However, it is difficult to identify fish eggs to species level using morphological characters. For instance gadoids fish eggs from cod, haddock, pollock and whiting are similar and display overlapping egg-sizes. In order to get an accurate classification of individual egg to species level genetic methods (DNA barcoding) can be used (Svensson et al. 2019).

During the last four years (2017-2021) of egg sampling 8-fjord area where the NTZs have been sampled on at least one occasion each year during the spawning season of cod (Figure 11, for years 2013-2014 see Svedäng et al. 2018).

Hydroacoustics is a non-destructive sampling method, which was used to study the presence of large (mature) cod in the NTZ and surrounding areas during the spawning period. Sampling was conducted annually in the Havstensfjord from 2009 -2015 in February until April and the NTZ and buffer zone were sampled at night four times each year. The sampling was conducted with a Simrad EK60 120 kHz split-beam sonar. The speed during sampling was on average 4 knots (2 m/s) and

position data was collected continuously. Hydroacoustic measurements were conducted during night time to minimize the risk of fish being too close to the bottom or forming dense shoals, which makes it difficult to separate individuals from the data. The data collection was done in a network of randomized cross sections. The analysis of distribution of large fish from the hydroacoustics was based upon Rose & Porter (1996), where target strength ≥ -33 dB is defined as the echo from a large cod (fish) ≥ 40 cm in total length. The results are presented as number of large individuals of cod ≥ 40 cm in total length per km^2.

12.2.3. Data collection with Stereo-BRUV

Body size and biomass distribution data are recognized as essential metrics for biodiversity conservation and fisheries management (Langlois et al. 2012). Remote video-based sampling methods are increasingly being adopted to collect this data due to: (a) their non-destructive nature, (b) ability to sample rare species over broad depth ranges, (c) provision of a permanent record that can be reviewed to reduce interobserver variability, (d) ability to collect concomitant data on habitat (e.g. epibenthic cover and substrate) and (e) provision of images for science communication (Langlois et al. 2020). Multiple remote systems can be deployed in the field consecutively to make efficient use of field time and enable spatially extensive sampling (Langlois et al. 2012). In addition, BRUV systems with stereo-video cameras (stereo-BRUVs) enable precise measurements of body size (Harvey et al. 2001). Studies have shown that stereo-BRUVs can provide comparable body size distribution data to fisheries-dependent methods such as trawls (Cappo et al. 2004), hook and line (Langlois et al. 2012) and trap fishing (Langlois et al. 2015). Despite being considered unsuitable for estimating density, stereo-BRUVs are considered to provide a cost-effective and statistically powerful method to detect spatiotemporal changes in the relative abundance (RA), length and biomass distribution of fish assemblages (Bornt et al. 2015; Harvey et al. 2013; Malcolm et al. 2015). Although stereo-BRUVs are commonly used globally, this method has so far rarely been used in the North East Atlantic (Unsworth et al. 2014). As far as we know, the study presented in this report represents the first attempt to use this method to monitor large predatory fish in low densities in protected fjord systems in Sweden, such as the population of cod in the Havstensfjord.

To test and develop the method under local conditions stereo-BRUV deployments were conducted throughout the northern part of the Havstensfjord, including the NTZ, during 2019 – 2020. Due to the exploratory and evaluating properties of the first deployments, sampling design (regarding spatial separation and deployment duration) was initially not fully standardised. From 2021, when the stereo-BRUVs were used to establish a baseline on relative abundance and body size structure of

cod on four selected sites for possible long-term monitoring, sampling design and method was set according to a standardized protocol defined in Langlois et al. (2020). Video analysis of the collected stereo-video was conducted in the software EventMeasure from SeaGis®. Number of individuals of cod was determined by size separation (minimum 5 cm difference in length) of all measured cod and by collection of MaxN (maximum number in one frame; Whitmarsh et al. 2017) for all deployments. Other species of fish and other larger marine organisms (such as harbour seal, cormorant and European lobster) were only noted. Habitat of the stations was determined by visual inspection of the video footage and classified in accordance to the Natura 2000 habitat system (Romaro 1996), into either reefs (rocky substratum, code 1170) or soft bottom (code 1000). Stereo-calibration of the BRUV systems was conducted with a 3D calibration cube and the software CAL from SeaGis® (https://www.seagis.com.au). Artificial light (Big Blue®, model AL2600XVP) was used on all deployments from 2021. Relative abundance of cod was calculated as the sum of counted individuals / by total number of deployments per habitat and year. Deployments where the visibility was continually less than 1 m (the distance to the bait) or when the BRUV tipped over on landing, such that the cameras pointed vertically towards the surface, were considered invalid and excluded from the analysis.

12.2.4. Information of bycatches of cod in the recreational lobster fishery

Projekt 8-fjordar has collected reports on catches in the recreational fishery for lobster, including bycatches of cod, in nine regions from the 8-fjords area since 2013 (Figure 5). Data from this reporting contributes to information on the presence of large cod in the area.

The fishery starts in late September and usually lasts until mid-November. How often the lobster pots were checked and emptied by the recreational fishermen varied, but typically, the interval between the occasions was between 1 and 4 days. Since 2014, all lobster pots (i.e. the buoys) have been counted in different sub-areas of the 8-fjords area, although not all areas have been surveyed every year (Sundelöf 2021).
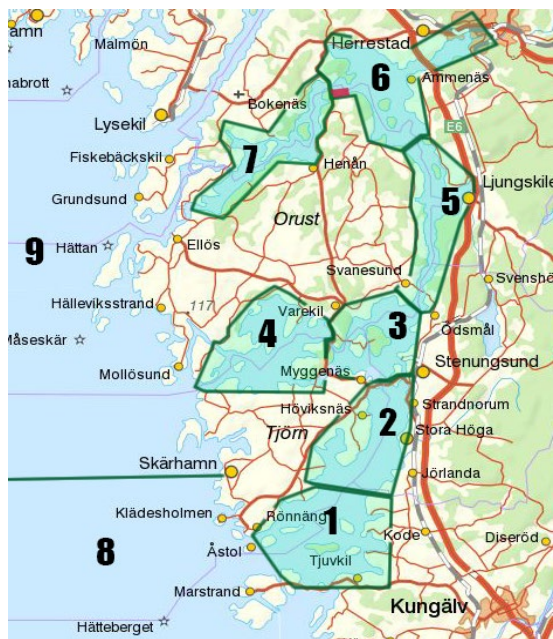
Figure 5. The different regions for reporting of catches in the recreational fishery for lobster in the 8-fjords area.

12.2.5.   Marine mammals and seabirds

The role of predators such as seals and cormorants was discussed in the evaluation in 2016 and that it could have the potential of impacting recovery of fish stocks negatively (Svedäng *et al.*, 2016). The ecological role of top predators and their impact on codfish and flatfish stocks in the area are unknown. However, due to a large interest from various local stakeholders, SLU, together with *Projekt 8-fjordar* has initiated projects on prey choice and abundance of harbour seals and great cormorants in the area. Some preliminary results from these ongoing projects are presented in this report

12.3. Results

Summary of the results from the evaluation in relation to goals and indicators for the NTZ in Havstensfjord, both from the first evaluation in 2016 and for the long-term evaluation in 2022, respectively is given in Table 2. The Table also contains references to figures and tables in this report for specific results.

Table 2. GOIS-table for the NTZ in Havstenfjord summarizing the results.

Goal	Sub-goals	Indicators	Success Criteria	Methods	Results 2016	Results 2022	Reference
Reconstruction of demersal and local fish stocks	Increasing population size	Population density of cod, plaice and turbot	Increasing number of individuals of fish and kg fish per km², respectively	Bottom trawl from the coastal survey	No increase in stock density of the three target species: cod, plaice and turbot during the conservation period. However, the average density of turbot in FFO indicates a relatively stable stock	No increase in stock density of the three target species: cod, plaice and turbot during the conservation period. However, number of cod individuals do show a positive trend due to a few peaks but not for kg (biomass). Both plaice and turbot show negative trends over time.	Figure 6a-f
	Improved spawning stock	Potential spawning biomass, measured as incresing trends of number of large cod (≥40 cm) and biomass of large cod, large plaice (≥24 cm) and large turbot (≥35 cm).	Potential spawning biomass increases over time	Hydroacoustics; Bottom trawl from the coastal survey	Analysis of acoustic data shows that the amount of large cod (≥40 cm) individuals within the NTZ has not recovered over time, on the contrary, a negative development is seen. Information from coastal survey confirms the result and does not show a positive development of spawning biomass for cod, neither for plaice or turbot. Interseting is that spawning biomass of turbot is highest within the NTZ and the buffer zone compared to the other areas.	The data from the coastal survey show no recovery for cod or plaice. The result show very low biomass of potentially mature individuals of cod and plaice in the NTZ after the stablishment in 2010. Regarding turbot, the NTZ generally has the highest biomass of large individuals in the time series, however, from 2019 and forward is is zero and has negative trend	Figure 7; Figure 8a-c
	Diversified size distribution	Size structure (number of large individuals and length distribution)	Increasing number of large individuals of cod (≥40 cm), plaice (≥24 cm) and turbot (≥35 cm). The number of length classes (cm) increases over time.	Bottom trawl from the coastal survey	Size distribution in the fish community for the target species do not show a recovery. However, the result for turbot have a more diverse size distribution in the NTZ and more large individuals compared to the other areas.	The width of the size distribution of cod is very limited in the NTZ, even so, there is a positive trend in increasing number of length classes over time in the NTZ. For plaice and turbot the number of length classes show a decrease over time. Number of large individuals for the three target species does not show a recovery. For both plaice and turbot there is a negative trend for number of	Figure 9 a-f
	Increased recruitment	Recruits in the adult habitat as a measure of juvenile production	Increased desities of young individuals of cod (<40 cm), plaice (<24 cm) and turbot (<35 cm)	Bottom trawl from the coastal survey	No increase of the average density for young individuals for the three target species. The average density of young turbot indicates a regular recruitment	A positive trend for cod but no trends for plaice or turbot within the NTZ. The average density of young turbot indicates a regular recruitment in the NTZ.	Figure 10a-c
	Spillover (Juveniles and adult individuals)	CPUE per age group	CPUE increase over time in adjacent areas	Hydroacoustics; Bottom trawl from the coastal survey	Have not been analyze since no reconstruction occurred in the NTZ	Not relevant since no reconstruction occurred in the NTZ	Not relevant
Population structure for cod in the 8-fjords area	Spawning of cod	Egg from cod	Documentation of eggs in early development stages	Vertical hauls with plankton nets, eggs identified through morphological characters combined with genetic methods	Documentation of cod eggs in early stages of development indicates local spawning areas for cod in the NTZ and the 8-fjords area	Annual documentation of cod eggs in early stages of development indicates local spawning areas for cod in the 8-fjords area	Figure 11; Figure 12
	Coastal population of cod	Genetic differentiation	Genetic differences between cod eggs collected in the North Sea and Kattegat compared to cod eggs collected along the coast of Bohuslän	Population genetics	The results indicates on a genetic differentiation of eggs collected along the coast of Bohuslän compared to eggs collected in the Nort Sea and the Kattegat, indicating a coastal population of cod	Recent large-scale genomic studies on coastal cod along the Skagerrak coast did not display any significant genetic differentiation from cod from North Sea or from Kattegat; suggesting that a mixture of cod of either North Sea or Kattegat origin today inhabit the coastal zone of the Swedish Skagerrak coast	Not relevant

12.3.1.   Reconstruction of demersal and local fish stocks

Density of fish using bottom trawl

Total (including fish of all sizes) cod abundance increased post the establishment of the NTZ during the period 2013-2021 both in the NTZ + buffer zone (p=0.03; rho=0.42) and in the coastal reference area (p<0.01, rho=0.27). The other reference areas showed no trends in abundance over the same period. In contrast a decrease in biomass could be seen in Gullmar fjord (p=0.03, rho=-0.36), while there were no trends in any of the other areas including the NTZ + buffer zone.

In the immediate period post the establishment of the NTZ (2010-2012) the only area showing any trend in cod abundance or biomass was the offshore area where both abundance and biomass increased (p<0.01, rho=0.79). Over the whole period post the establishment (2010-2021), cod abundance increased in all reference areas (Gullmar fjord: p=0.01, rho=0.35; Coast: p<0.01, rho=0.37; Offshore: p=0.01, rho=0.32), but not in the NTZ + buffer zone. No change in biomass could, however, be seen in any area indicating that more but smaller fish were caught. A likely result considering the decrease in mesh size 2013.

Prior to the establishment of the NTZ (2002-2009) both cod abundance (Gullmar fjord: p=0.01, rho=-0.42; Coast: p=0.02, rho=-0.23) and biomass (Gullmar fjord: p=0.01, rho=-0.47; Coast: p=0.03, rho=-0.22) decreased in Gullmar fjord and along the coast while no trend was seen in the NTZ + buffer zone or offshore. The only other significant trend of cod abundance in the NTZ + buffer zone was an increase over the whole period 2002-2021 (p=0.03, rho=0.29). However, this trend seems to be largely due to a few peaks of cod during the period 2013-2021 and it does not lead to a subsequent accumulation of biomass in the following years. Similarly, cod abundance also increased in all the reference areas (Gullmar fjord: 0.01, rho=0.29; Coast: p<0.01, rho=0.29, Offshore: p<0.01, rho=0.43) over this period without any change in biomass.

Total plaice biomass has decreased in the NTZ + buffer zone over the whole study period 2002-2021 (p<0.01, rho=-0.37), but no trend can be seen post the establishment of the NTZ within this area neither in abundance nor biomass. In fact, there are no other significant trends of total plaice abundance or biomass within the NTZ + buffer zone during any other period. The biomass has, however, decreased also in Gullmar fjord (p<0.01, rho=-0.36) and along the coast (p<0.01, rho=-0.29) since 2002. Also, the abundance decreased along the coast (p<0.01, rho=-0.19) over the same period.

Prior to the establishment of the NTZ (2002-2009) both abundance and biomass decreased in Gullmar fjord (p=0.01, rho=-0.45) and along the coast (Abundance: p<0.01, rho=-0.40; Biomass: p<0.01, rho=-0.36). The same goes for biomass in Gullmar fjord (p=0.04, rho=-0.29) and both abundance (p=0.04, rho=-0.16) and biomass (p<0.01, rho=-0.25) along the coast post the establishment (2010-2021). In contrast both abundance and biomass increased in Gullmar fjord (p=0.04, rho=0.55) during the period 2010-2012, while no trends could be seen in the other areas. At last, both abundance and biomass decreased in Gullmar fjord (Abundance: p<0.01, rho=-0.45; biomass: p<0.01, rho=-060) and along the coast (Abundance: p<0.01, rho=-0.39, Biomass: p<0.01, rho=-0.49) 2013-2021, the biomass also offshore (p=0.04, rho=-0.28).

Total turbot biomass has decreased since 2002 in the NTZ + buffer zone (p=0.02, rho=-0.31), but this was not reflected in the overall abundance. The only other significant trends of total turbot biomass and abundance is a decrease in both in Gullmar fjord since 2010 (p=0.04, rho=-0.29). No fish was, however, caught in Gullmar fjord during the period 2002-2009, nor along the coast 2010-2012. The presence of cod and plaice is very low in the NTZ + buffer zone, while the presence of turbot is higher within the NTZ + buffer zone throughout the time series compared to the other areas (Figures 6a-f). All trends can be seen in tables (1-2) in Appendix (1).
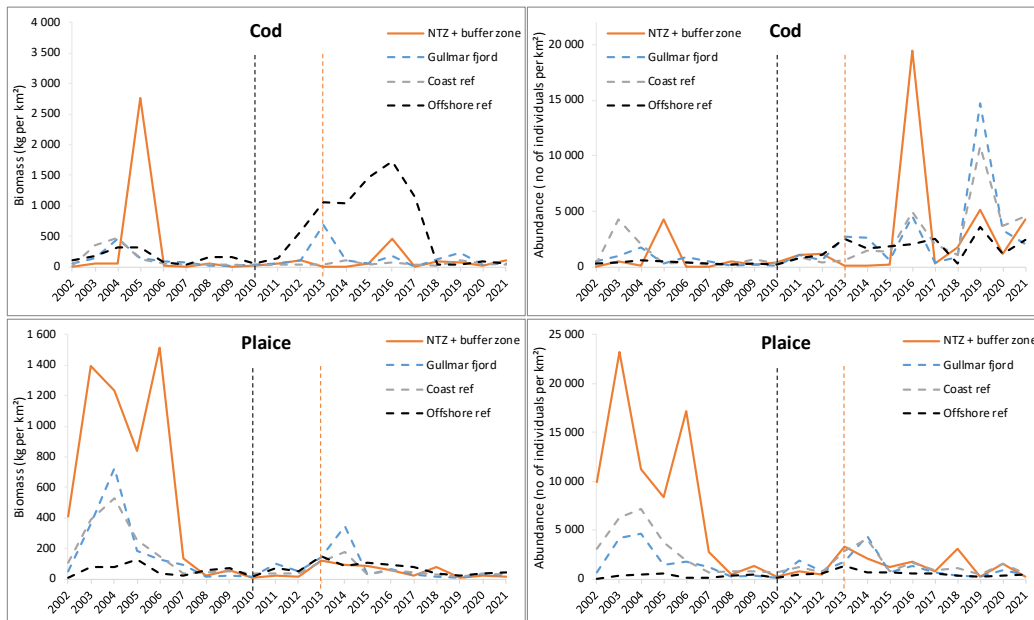
Figure 6a-f. Results from the coastal survey 2002-2021, data collected with bottom trawl and presented as a yearly average for the target species cod (top), plaice (middle) and turbot (lower). The results are presented as biomass (kilogram per km²) and abundance (individual fish per km²), respectively. Noticeable, dashed black horizontal line represents the time, 2010, of establishment of the NTZ and dashed orange horizontal line represents the time, 2013 of decrease in mesh size (70 to 16 mm) and change in survey period (October to August/September).

Spawning stock measured with hydroacoustics and bottom trawl

The presence of potentially spawning/mature cod (≥40 cm total length) based on hydroacoustics are presented in figure 7. Data collection was conducted during the cod spawning period in February - April 2010-2015 in the NTZ and Havstensfjord. The result within the NTZ show a negative trend over time ($p<0.05$) for the presence of large and, thereby, potentially spawning cod (≥40 cm) during the period 2010 until 2015.
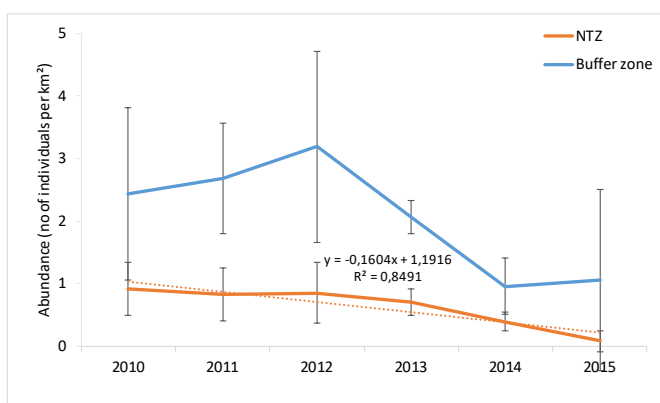
Figure 7. The number of large cod individuals (≥40 cm) from the hydroacoustic measurements as annual averages (±95% CI) within the NTZ and Havstensfjord buffer zone.

To further study large, potentially mature fish in the NTZ and the buffer zone, data on abundance and biomass from the coastal survey was analysed for the three focal species cod (≥40 cm), plaice (≥24 cm) and turbot (≥35 cm) during the same periods as in the previous section from 2002 until 2021, before (2002-2009) and after the NTZ was established (2010-2021).

There was no change in large cod abundance or biomass within the NTZ + buffer zone during the study period, although the data show a large amount of zero values for cod. In contrast both abundance and biomass of large cod decreased in the reference areas over the whole period 2002-2021 (Gullmar fjord: p=0.04, rho=-0.22; Coast (abundance): p<0.01, rho=-0.21; Coast (biomass): p<0.01, rho=-0.20 and Offshore (abundance): p=0.03, rho=-0.20, Offshore (biomass): p=0.02, rho=-0.21). Post the establishment of the NTZ (2010-2021) large cod continued to decrease both in abundance and biomass along the coast (p=0.02, rho=-0.19) and offshore (p<0.01, rho=-0.34), although a brief increase could be seen offshore during the period 2010-2012 (Abundance: p=0.01, rho=0.59; Biomass: p=0.04, rho=0.50).

Large plaice abundance (p<0.01, rho=-0.46) and biomass (p<0.01, rho=-0.47) declined over the whole period 2002-2021 in the NTZ + buffer zone. The same goes for all reference areas (Gullmar fjord: p<0.01, rho= -0.63; Coast: p<0.01, rho= -0.43; Offshore (abundance): p<0.05, rho=-0.19 and Offshore (biomass): p=0.04, rho= -0.20). No further trends of large plaice can be seen within the NTZ + buffer zone, however, in the Gullmar fjord the decline was significant also prior to the establishment of the NTZ during 2002-2009 (p<0.01, rho=-0.49) as well as post the establishment during the periods 2010-2021 (Abundance: p<0.01, rho=-0.55; Biomass: p<0.01, rho=-0.54) and 2013-2021 (p<0.01, rho=-0.63). Plaice abundance and biomass also declined along the coast post establishment 2010-2021 (Abundance: p<0.01, rho=-0.24; Biomass: p<0.01, rho=-0.25) and 2013-2021 (Abundance: p=0.01, rho=-0.25; Biomass: p<0.01, rho=-0.26). Offshore the decline was also significant since 2013 (Abundance: p=0.03, rho=-0.29; Biomass: p=0.04, rho=-0.28).

Large turbot abundance (p=0.01, rho=-0.41) and biomass (p=0.02, rho= -0.40) has declined since 2010. Meanwhile large turbot has increased in abundance and biomass offshore (p=0.04, rho=0.19) since 2002, mainly due to a few catches above zero. No other trends were significant. The NTZ + buffer zone generally has the highest biomass of large individuals, however, from 2019 and forward it is zero. During the period 2002-2009 no large turbot was caught offshore or in Gullmar fjord and between 2010 and 2021 none were caught along the coast.

Noteworthy is also that data show low biomass of potentially mature individuals of cod and plaice in the NTZ + buffer zone after the establishment in 2010. The results are similar for all the areas except for the offshore reference which shows the highest average value of biomass of potentially mature individuals for the period 2012 to 2017 during the time series. After 2017 the biomass of large cod is very low also for the offshore reference. No large cod were caught within the NTZ +

buffer zone between 2010 and 2012 (Figures 8a-f). All trends can be seen in tables (2) in Appendix (1).

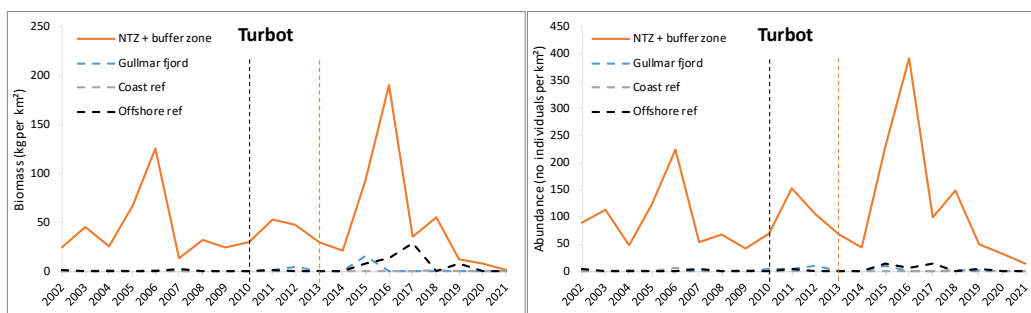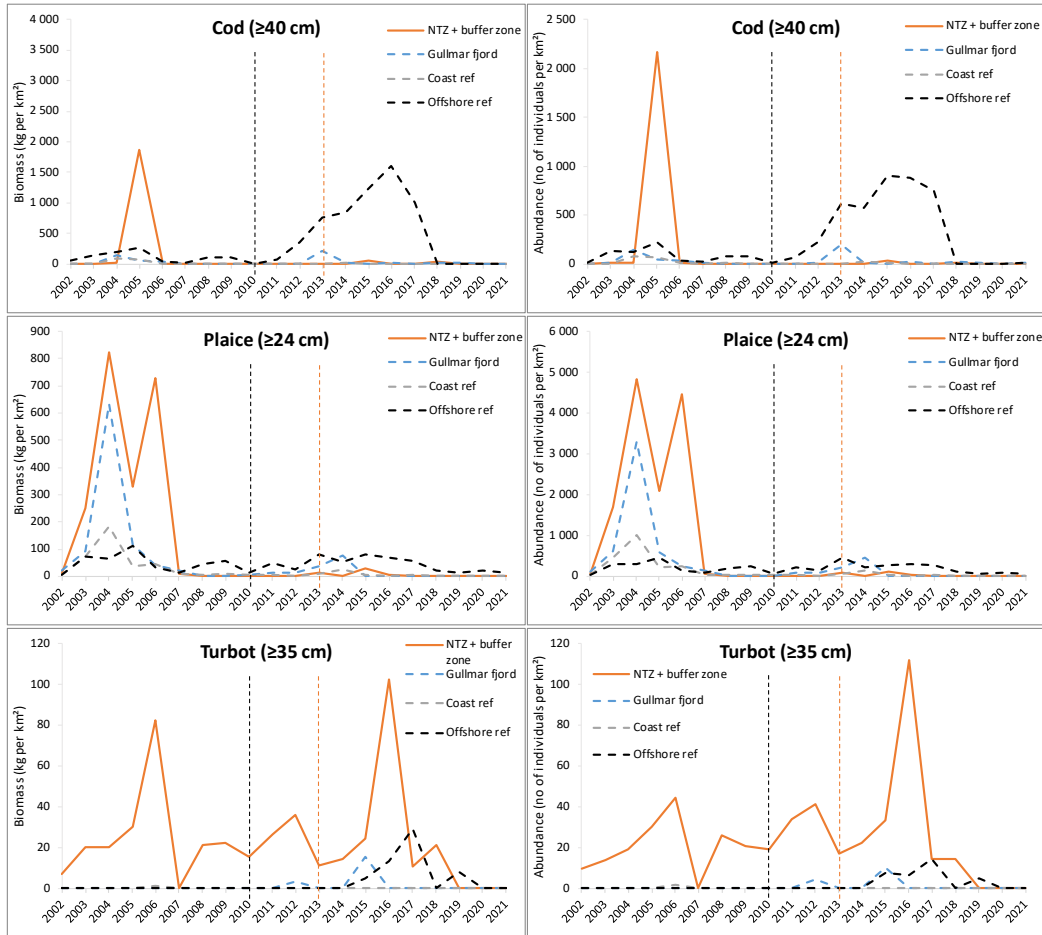*Figure 8a-f. Results from the coastal survey, data collected with bottom trawl and presented as an annual average for large, potentially mature, individuals of target species cod (top), plaice (middle) and turbot (lower). The results are presented as abundance: individuals per km^2 and biomass: kilogram fish per km^2. Noticeable, dashed black horizontal line represents the time, 2010, of establishment of the NTZ and dashed orange horizontal line represents the time, 2013, of decrease in mesh size (70 to 16 mm) and change in survey period (October to August/September).*

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Diversified size distribution

The width of the size distribution of cod is very limited within the NTZ + buffer zone and the number of large individuals is extremely low. Even so, the number of length classes of cod within the NTZ + buffer zone has increased since 2013 (p=0.04, rho=0.41) and an increase could also be seen between 2010-2012 offshore (p<0.01, rho=0.84). The only other significant change happened prior to the establishment of the NTZ (2002-2009) when the length classes decreased along the coast (p0.04, rho=-0.21).

The length classes of plaice, however, have decreased since 2002 in the NTZ + buffer zone (p=0.01, rho=-0.34) as well as in Gullmar fjord (p<0.01, rho=-0.36) and along the coast (p<0.01, rho=-0.38). In Gullmar fjord and along the coast the length classes of plaice also decreased during the periods 2002-2009 (Gullmar fjord: p=0.04, rho= -0.36; Coast: p=0.01, rho=-0.25), 2010-2021 (Gullmar fjord: p=0.01, rho=-0.38; Coast: p<0.01, rho=-0.36) as well as 2013-2021 (Gullmar fjord: p=0.01, rho=-0.41; Coast: p<0.01, rho=-0.40).

The number of length classes of turbot (p=0.03, rho=-0.29) within the NTZ + buffer zone has decreased since 2002. The decrease is significant also since 2010 (p=0.04, rho=-0.35). A decline in length classes can also be seen in Gullmar fjord during this period (p=0.03, rho=-0.30). No other trends were significant. Turbot has a greater size distribution in the NTZ compared to the other areas (Figure 9c). Large individuals of turbot are more common in the NTZ compared to the reference areas except for the period 2019-2021. In the other areas there are only occasional catches of turbot in the time series (Figures 9a-c). All trends can be seen in tables (1 and 3) in Appendix (1).
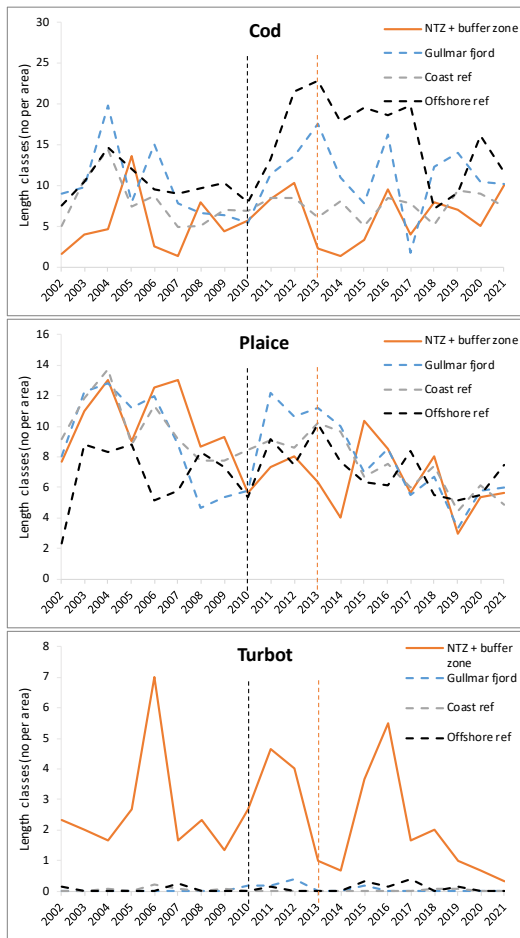
Figure 9a-c. Size distribution presented as annual average of number of length classes per area for the species cod (top), plaice (middle) and turbot (lower) in the NTZ + buffer zone and the reference areas. The data come from the coastal survey. Noticeable, dashed black horizontal line represents the time, 2010, of establishment of the NTZ and dashed orange horizontal line represents the time, 2013, of decrease in mesh size (70 to 16 mm) and change in survey period (October to August/September).

Recruitment

Although abundance of small cod (< 40 cm) increased in all areas over the entire period (2002-2021) only the NTZ + buffer zone (p=0.03, rho=0.43) and the coastal reference area (p<0.01, rho=0.28) showed an increase also after 2013 when the mesh size changed. No trends could be seen in the NTZ + buffer zone the first few years (2010-2012) after the establishment of the NTZ, while both abundance and biomass increased offshore (p<0.01, rho=0.75) and biomass also in the Gullmar fjord (p=0.02, rho=0.62). However, from 2013 and onwards the biomass of small cod only increased in the NTZ + buffer zone (p=0.03, rho=0.43). The data show a few peaks of young individuals of cod in the NTZ, e.g. 2016 and 2019. Similar patterns are observed in the reference areas as well and indicate that cod still recruit to the investigated areas.

There was no change in biomass or abundance of small plaice (<24 cm) in the NTZ + buffer zone. During the period 2013-2021 there was, however, a negative trend of plaice in all the reference areas except offshore (Abundance: Gullmar fjord: p=0.01, rho=-0.40; Coast: p<0.01, rho=-0.37; Biomass: Gullmar fjord: p<0.01, rho=-0.50; Coast: p<0.01, rho=-0.45). This was also true during 2002-2009 (Abundance: Gullmar fjord: p=0.03, rho=-0.38; Coast: p<0.01, rho=-0.37; Biomass: Gullmar fjord: p=0.04, rho=-0.37; Coast: p<0.01, rho=-0.34) The high numbers of young plaice recorded in the beginning of the time series have not been observed in recent times.

There was a decline in both abundance and biomass of small turbot (< 35 cm) in the Gullmar fjord between 2010-2021 (p=0.03, rho=-0.30). No other trends were significant in any areas including the NTZ + buffer zone. However, young individuals of turbot in the NTZ show the highest values throughout the time series compared to the other areas. In the other areas the presence of young individuals of turbot is very low, with only occasional occurrence in the dataset (Figure 10a-c). All trends can be seen in tables (1-3) in Appendix (1).
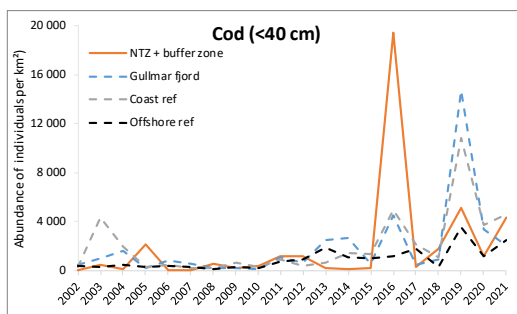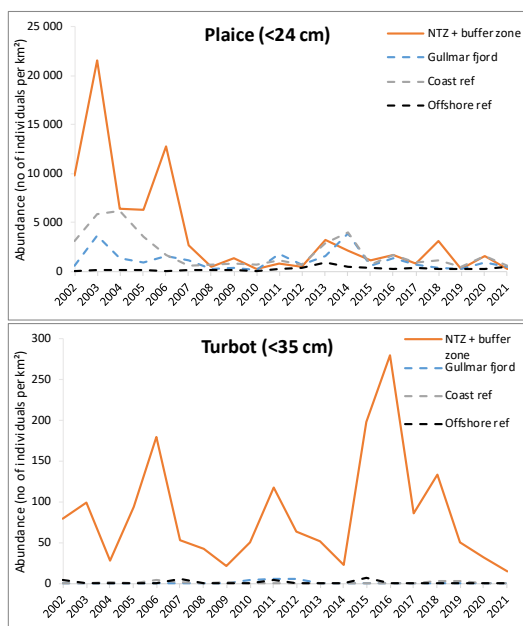
Figure 10a-c. Results from the coastal survey, data collected with bottom trawl and presented as an annual average of young individuals of target species cod <40 cm (top), plaice <24 cm (middle) and turbot <35 cm (lower). The results are presented as number of individuals per km². Noticeable, dashed black horizontal line represents the time, 2010, of establishment of the NTZ and dashed orange horizontal line represents the time, 2013, of decrease in mesh size (70 to 16 mm) and change in survey period (October to August/September).

Findings of cod eggs in the 8-fjordar area

In 2017 sampling of the 8-fjord area, where the NTZs are situated, stretched down to the island of Åstol just south of Tjörn, also the western parts of the Stigfjord were included in the sampling. In the later years, (2018-2021) sampling of the 8-fjord area was limited to the area north of the city of Stenungsund. Eggs from cod have been found in the 8-fjord area in all the years of sampling (2013-2014, 2017-2021), this also includes findings of eggs of the earliest development stage (stage 1a-1b). In general, compared to the rest of the area sampled during the egg survey, concentration of cod eggs in the 8-fjord area was lower than in the area stretching from Smögen to the Gullmar fjord (Figure 11). Variation of egg concentration between sites may be interpreted as an indication of specific spawning sites for cod. However, given that eggs are passively transported by ocean currents patterns of egg findings should not be over interpreted over small geographical scales. Seen over the whole sampled area findings of the earliest stages of cod eggs (1a-1b) have since 2017 displayed a steady decline ($p < 0.05$, rho = -1; Figure 12).

2017
egg m²
· 0
• 5
● 15
● 50

27/2-3/3

3/4-7/4

24/4-28/4

2018
egg m²
· 0
● 5
● 10

13/2-19/3

13/3

26/3

2019
egg m²
· 0
● 5
● 10

6/2-7/2

26/2

11/3-20/3
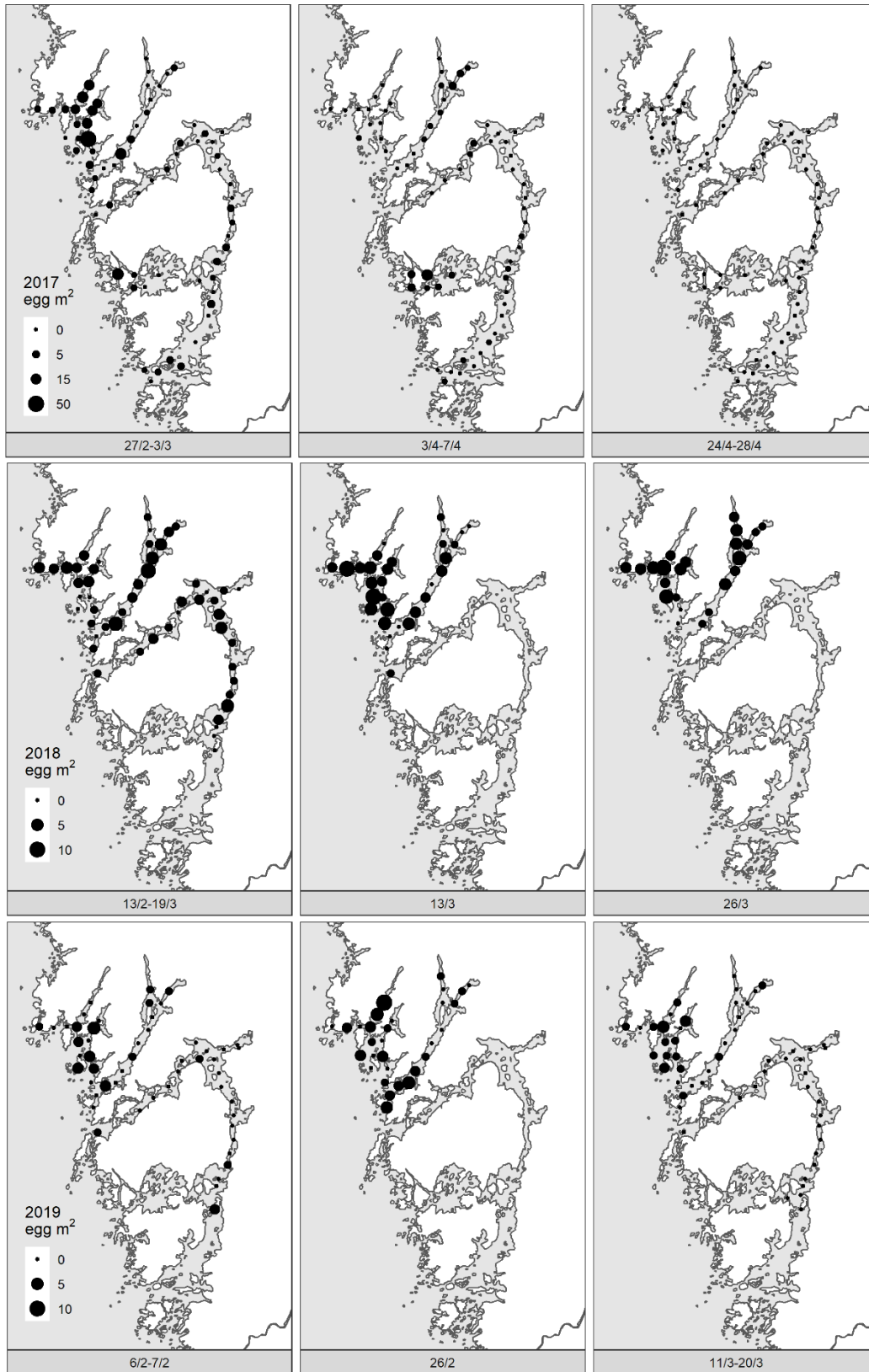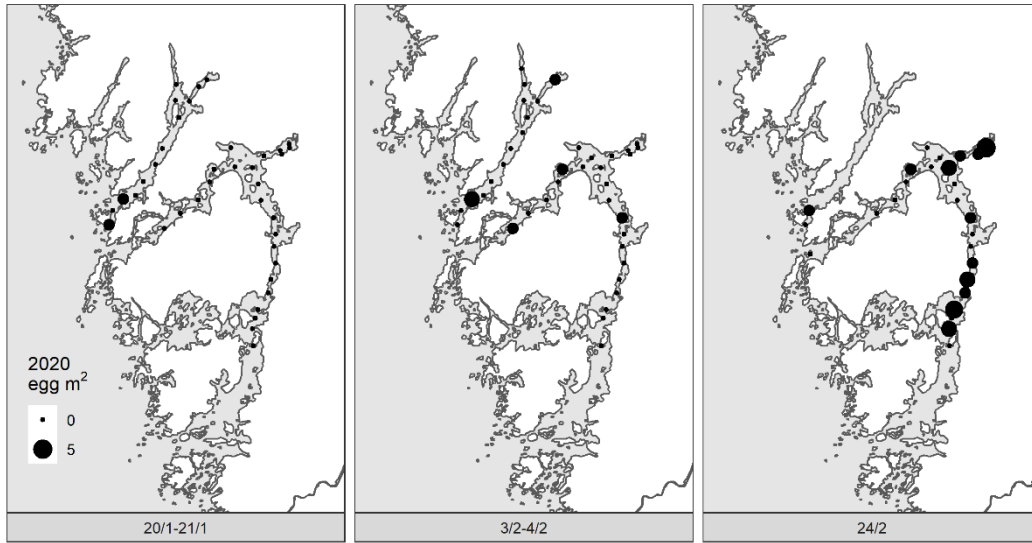
Figure 11. Number of cod eggs per m² sampled on three occasions during the winter/spring period in the years 2017, 2018, 2019 and 2020, along the Bohuslän coast. Eggs includes all the different development stages of cod eggs from newly fertilized (stage 1a) to just before hatching to larvae (stage 5). The symbol size correlates to number of eggs per m². Dates of sampling can be seen just below each map.
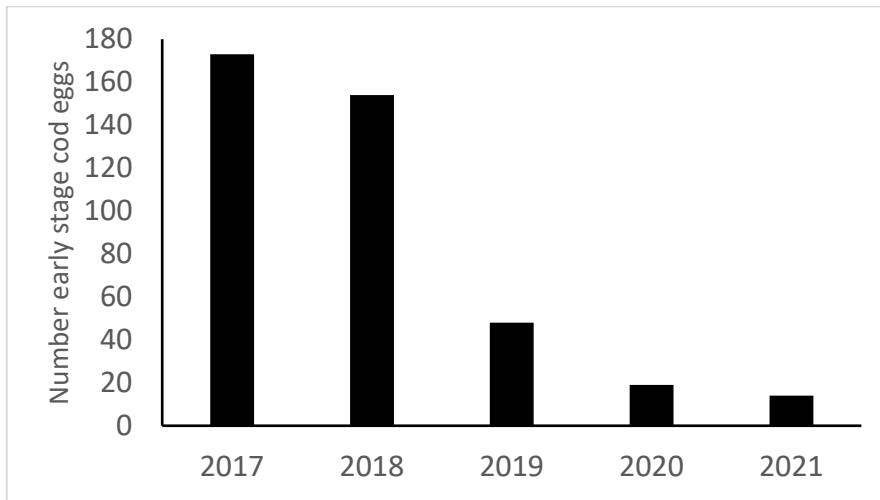
Figure 12. Total number of early stage (stage: 1a-1b) cod eggs found in all areas sampled along the Bohuslän coast 2017 to 2021.

Stereo-BRUV

The sampling conducted 2019 and 2020 resulted in 106 valid deployments, and 45 counted individuals of cod with measured body lengths from 6 to 48 cm (Table 3, Figure 13 and 14). Of the valid deployments, 58 were classified as stations with rocky substratum (1170), and 46 as stations with soft substratum (1000) (Table 3). The comparison between the habitats showed that relative abundance (RA) of cod was approximately 10 times higher on hard bottom substrate compared to soft

bottom substrate (Table 3). Spatial distribution of the cod show that abundance of larger sized individuals was highest in the north-western part of the sampled area in the Havstensfjord, in and around the north-western small NTZ (Figure 14). Number of cod on hard bottom substrate stations per depth interval indicate that the deeper stations (16 – 17 m, all located to the area around the north-western small NTZ) had higher abundance of cod than stations from shallower depths (Figure 16).

Table 3. Stereo-BRUV deployments in the Havstensfjord during September 2019, August 2020 and September 2021. Number of deployments on hard bottom (No 1170), number of deployments on soft bottom (No 1000), mean depth in meters (± SD) for deployments in each habitat, total number of observed cod (Tot No cod), length interval of the observed cod in cm (Length) and Relative Abundance (RA) of cod in each habitat are based on number of valid deployments (No valid).

	Sep-19	Aug-20	Sep-21
No deployments	32	84	36
No valid	29	77	36
No 1170	6	52	18
No 1000	21	25	18
Depth 1170	10.5 ± 0.73	10.6 ± 2.6	9.2 ± 2.0
Depth 1000	11.0 ± 0.80	10.8 ± 2.0	9.7 ± 1.0
Tot No cod	11	34	23
Length	8 - 12	6 – 48	8 – 27
RA 1170	1.33	0.62	1.33
RA 1000	0.14	0.08	0.00
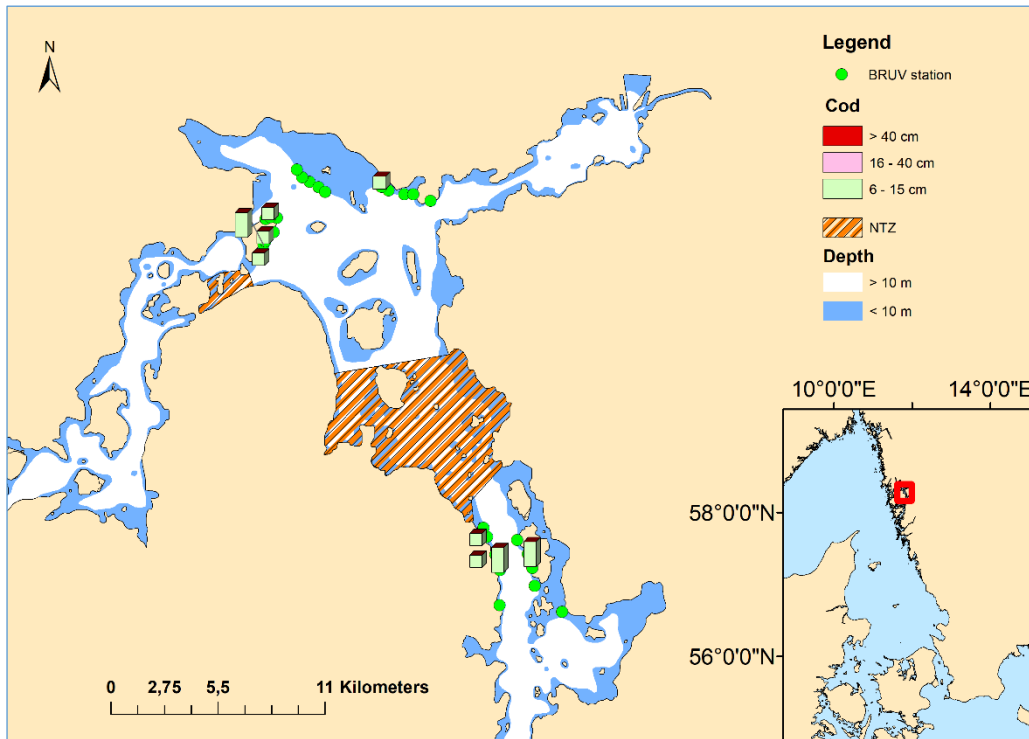
Figure 53. Relative number of observations of cod in size class 6-15 cm (green bars), 16-40 cm (purple bars) and > 40 cm (red bars) per valid stereo-BRUV station (green dot) in the Havstensfjord during sampling in 2019. Red striped areas represent NTZ surrounding areas are part of the buffer zone.
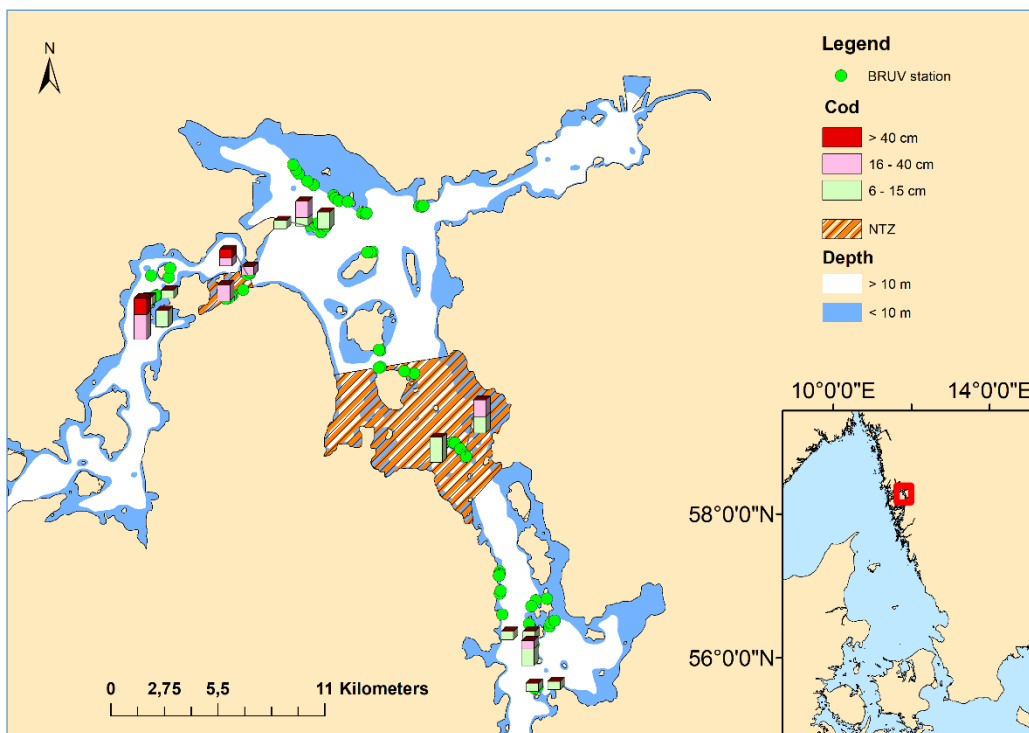
Figure 14. Relative number of observations of cod in size class 6-15 cm (green bars), 16-40 cm (purple bars) and > 40 cm (red bars) per valid stereo-BRUV station (green dot) in the Havstensfjord during sampling in 2020. Red striped areas represent NTZ surrounding areas are part of the buffer zone.

During 2021, 36 valid stereo-BRUV deployments were conducted in the Havstensfjord (9 deployments per reference site, Table 3, Figure 13). At the reference site with hard bottom substrate, located to the boarder of the north-western small NTZ (HB1, Figure 15), one single individual of cod with measured body length of 27 cm was observed. At the second hard bottom substrate reference site, located to the island "Kråkan", 22 individuals of cod with a measured body lengths from 8 – 13 cm were observed (HB2, Figure 3). No observations of cod were made on the reference sites with soft bottom substrate (SB1 and SB2, Figure 13). The combined count of individuals per habitat type resulted in a relative abundance (RA) comparable to the result of the exploratory sampling conducted during 2019 and 2020 (Table 3).
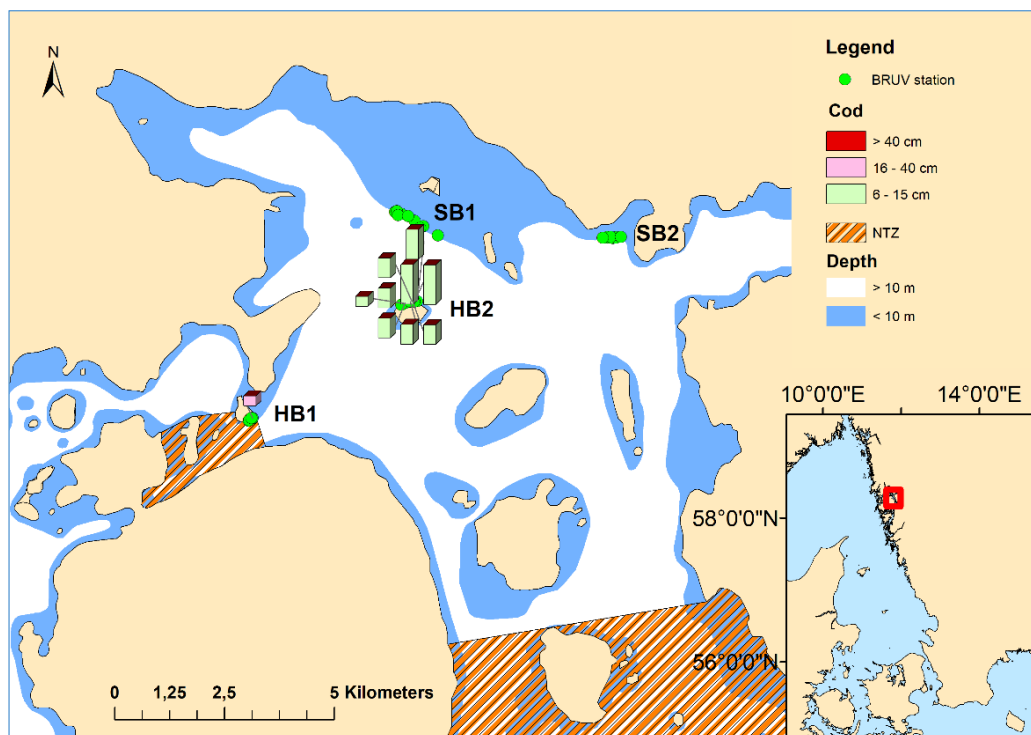
Figure 15. Relative number of observations of cod in size class 6-15 cm (green bars), 16-40 cm (purple bars) and > 40 cm (red bars) per valid stereo-BRUV station (green dot) in the Havstensfjord during sampling in 2021. HB1 and HB2 indicate the location of hard bottom substrate reference sites (code 1170), SB1 and SB2 indicate the location of soft bottom substrate reference sites (code 1000). Red striped areas represent NTZ.
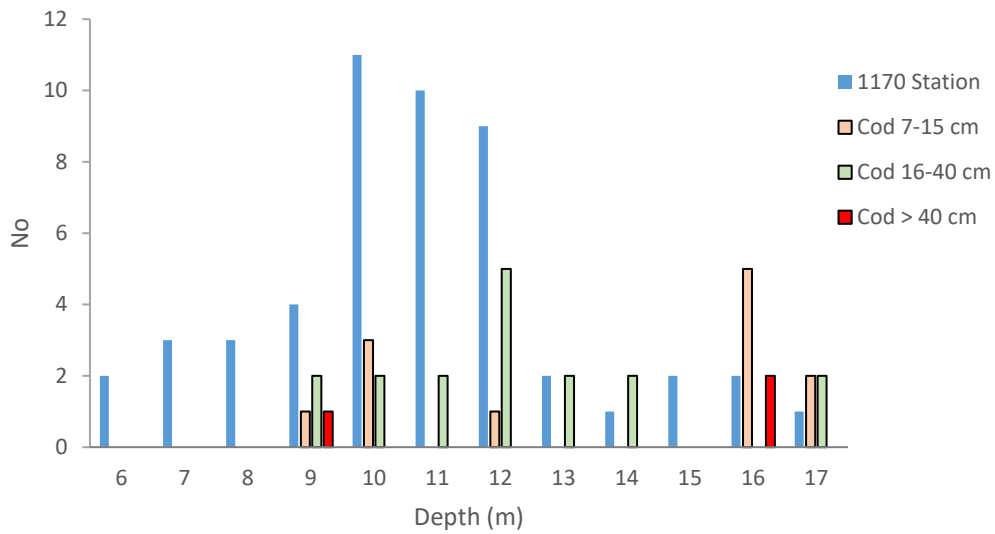
Figure 16. Number of valid stations and number of cod in size class 7 – 15 cm, 16 – 40 cm and > 40 cm at depth 5 – 17 cm on hard bottom substrate (1170) during sampling in 2020.

Bycatches of cod in the recreational lobster fishery

The number of individuals providing catch reports has varied between 8 (2013) and 3 (2018) and most of the reports regard the lobster fishery in region 5 and 6, i.e. including parts of the buffer zone surrounding the NTZ in the Havstensfjord (Fig. 5). The average bycatch of cod in the lobster fishery in region 5 and 6 between 2013 and 2021 has been 0.06 cod per pot and fishery occasion, with a marked higher bycatch rate in the first year (2013: 0.15 cod per pot, Figure 17). The average bycatch rate of cod in the total 8-fjords area was 0.05 per lobster pot between 2013 and 2021.
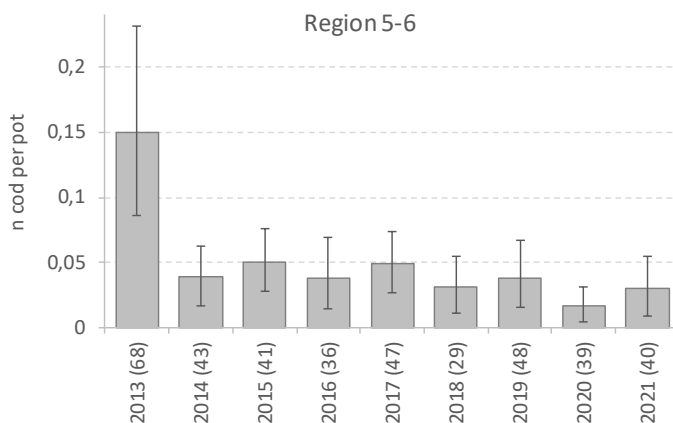
Figure 17. Mean number of bycaught cod individuals per lobster pot and fishery occasion in the recreational fishery for lobster in region 5 and 6 (Figure 5). Error bars show 95% confidence interval and numbers within brackets indicate number of lobster pots from which catch reports were collected.

The reported bycaught cod individuals were dominated by fish from 25 to 40 cm, but also contained larger fish individuals (Figure 18). The bycatch of cod contained both alive and dead fish, but the relative proportions are unknown.
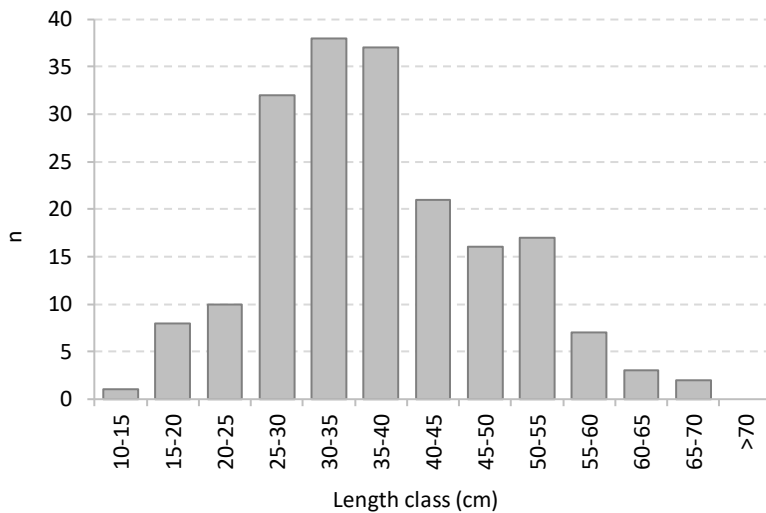
Figure 18. Length-frequency distribution of cod (n=192) reported as bycatch in the recreational fisher for lobster in the 8-fjords area 2013-2021.

On average, approximately 800 pots have been counted in the regions 5-6. Assuming a fishery season length of 50 days and 25 fishery occasions (every second day) and a bycatch of 0.06 cod per day, this implies that the lobster fishery in region 5-6 has a catching capacity of around 1200 cod per year. Extrapolating to the whole 8-fjords area (region 1-7), where, on average, 2800 lobster pots were counted each year during 2014-2021 and assuming a bycatch of 0.05 cod per day, this implies a catching capacity of 3500 cods per year. A mortality for pot-caught cod released in a tank of 9 % submerging and 79 % not submerging (40 % of total), shown by Humborstad et al. (2016) mean that the lobster fishery could have a potential of removing 1295 individuals of cod per year from the area.

12.3.2. Marine mammals and seabirds

Harbour seals (*Phoca vitulina*) and great cormorants (*Phalacrocorax carbo*) are the most abundant species of marine mammals and sea birds, respectively, occurring in the 8-fjords area. Harbour porpoises (*Phocoena phocoena*) are occasionally observed in the area, and seem to increase in numbers during later years, whereas a few grey seals (*Halichoerus grypus*) are regularly observed in the archipelago outside the area (www.artportalen.se).

Harbour seals

The 8-fjords area is generally not included in the national monitoring of harbour seals (Isaksson, 2003). In general, the occurrence of seals in the area seems to be very low during the summer (Alexandersson, 2011; Lundström et al. 2023a). The number of seals counted in the vicinity of the 8-fjords area, yet outside, between latitude 57.80 and 58.27, indicate that the population increased during the first decade of the 21st century, but levelled out during mid-2010s to around 2000 counted seals (Lundström et al. 2023a). Information is lacking about how seals migrate from the areas outside into the NTZ, and back, and to what extent the NTZ is used as feeding areas for harbour seals.

The number of harbour seals in the 8-fjords area and in the NTZ is unknown. However, in an ongoing project, the occurrence of harbour seals in a subarea of the 8-fjords area, overlapping the fishery-closure area (the Hake fjord-Havstensfjord), has been monitored on a monthly basis 2019-2021 (Lundström et al. 2023a). The abundance of seals varied both between years and throughout the year, with higher numbers during winter-early spring (n≈80 seals) and lower during late spring-autumn (n≈10 seals). These results are in line with the outcome of an earlier study that used game cameras to estimate the abundance of seals at specific haul outs between 2015 and 2017 (Carlsson, 2019). It is unknown whether it is the same seals that reside in the 8-fjords area over time, or if it is a continuous turnover of seals from the much more densely populated areas outside.

Great cormorant

Because of the lack of information on the recent development of the great cormorant population along the Skagerrak coast, all cormorant colonies off the coast were surveyed in 2020-2021 (Lundström et al. 2023b). In the 2021 survey, approximately 1300 nests in 7 colonies were counted in the vicinity of the 8-fjords area, ranging from the Nordre Älv river mouth to the Havstensfjord. All but one colony, one of the smallest with only 30 breeding pairs, are outside the NTZ (Lundström et al. 2023b). Information is lacking about how cormorants from colonies and night roosts outside the 8-fjords area use the area as feeding grounds or to what extent migrating cormorants from other areas use the 8-fjords area as feeding ground during temporary stop-over. In general, great cormorants are more abundant in the outer archipelago, both during summer and winter (Alexandersson, 2011, Nilsson and Haas, 2016).

In an ongoing project, the occurrence of great cormorants in a subarea of the 8-fjords area (Hakefjorden-Havstensfjord, i.e. the fishery-closure area) is monitored on a monthly basis (Lundström et al. 2023a). Results from 2019-2021 indicate that the abundance in the area increases after the breeding season with a culmination

during the autumn (n≈500 cormorants), and then decreases towards the winter (n≈100 cormorants) (Lundström et al. 2023b). The post-breeding increase of cormorants in the inner fjords is due to dispersal from the individual colonies, and the dispersal pattern varies in direction and distance over time. Additionally to this dispersal there might be migration of birds from further north which is partly dependent on the extent of freezing waterbodies.

12.4. Discussion

After 12 years the NTZ (no-take zone) and buffer zones within the Havstensfjord have not contributed to a measurable recovery of demersal fish stocks (Table 2). This does not necessarily mean that NTZs are ineffective as a management strategy. For heavily decimated stocks and species with long life cycles, an evaluation period of 6 to 12 years can be a short time frame.

The generation time for cod and plaice, two of the focal species in the area, are approximately around 7 to 10 years and they reach sexual maturity after 2 to 4 years. To obtain a quick response from the implementation of a NTZ after a period of overfishing, requires high recruitment of the focal species and a low fishing mortality. After a period of growth overfishing, where an intense fishing effort catches fish at young age and does not allow the fish to age, a rapid recovery might occur when introducing a NTZ, since it allows the young individuals to grow and offers protection from fishing. Recruitment overfishing occurs when adults have been reduced to a critical level, thus reducing the production of egg and fry/juveniles. A recovery of the spawning biomass may take several generations if it recovers at all. When the NTZ and buffer zone were introduced, stocks of cod had already collapsed, indicating recruitment overfishing. Ecosystem overfishing occurs when the fishery has impacted the food web and habitats to an extent that species cannot assume their original roles in the ecosystem once fishing ceases. Recruitment- and ecosystem overfishing have a strong impact and a potential recovery might take long time (Myers et al 1994). Increased knowledge about the ecosystem regarding habitats, prey, potential competitors and predators and these component´s development over time, can give information about the prerequisites for decimated fish stocks to regain its historical size and productivity. A recovery can therefore take decades instead of years, and it is then very important that changes in the ecosystem and environmental conditions are monitored in parallel, so conditions for a recovery can be evaluated. In a situation like this the management, and also the general public, need to be patient, and may consider further strategies that can improve the status of a threatened stock (Bryhn et al. 2022).

12.4.1.   Fish populations investigated within the coastal- and egg surveys and BRUVs

Plaice did not show a recovery in the NTZ but has declined since the monitoring started in 2002. Similar patterns were also observed in the reference areas. Plaice settle in shallow coastal bays including the 8-fjord area during spring where they spend their first summer as juveniles (Pihl et al. 2000). In the nursery grounds, the juveniles are sensitive to eutrophication induced macroalgal blooms, which may

negatively influence the growth and survival of plaice (Wennhage & Pihl 2007). During the first half of the 1990s there was a plaice spawning time fishery in a neighboring fjord (the Koljö fjord), where plaice caught were used as brood stock in a research fish culture (Wennhage 1999). This indicates that local spawning still occurred in the area during that time. Plaice in ICES management area IIIa has recently been divided into a North Sea stock including the Skagerrak and a stock for the Kattegat, the Belt Sea and the South West Baltic Sea (ICES 2012). Both stocks are presently considered as being fished at sustainable levels with historically high spawning stock biomass levels (ICES advice 2021). The decline in plaice juveniles and adults indicates that a similar decline as for cod has occurred in a species with large and sustainably fished offshore populations. In plaice, an earlier offshore migration of juveniles has been reported from other North Sea areas. Knowledge on plaice energetics and temperature preference indicate that this may be a response to climate change (Teal et al 2012, Van de Woolfshaar et al 2015). It should be noted though that the Eastern Skagerrak is considered to be an area with relatively low abundance of plaice recommended to be monitored separately (Ulrich et al 2013)

The chosen location of the NTZ and buffer zone may influence the potential contribution of this protection measure to the recovery of the target species. The area was mainly chosen based on information about the presence of cod and the historical fishery in the area with less consideration taken for plaice and turbot. The result from the egg surveys do not indicate that the NTZ today is a hot spot for spawning of cod, however, the earliest stages of eggs are found in the area indicating that spawning does occur. Given the extremely low abundance of large cod in the area, it is unclear if spatial distribution patterns can be used as a quantitative measure to assess the intensity of spawning or even the geographical location of spawning, since eggs are regularly transported by currents. In retrospect the results do not support that the selected area is favourable for a recovery of the stocks compared to surrounding areas. The intense historical fishery shows that the area has harboured large amounts of fish and, therefore, should have the potential to spark the recovery of target species when the NTZ was implemented. This was observed when the NTZ implemented along the Norwegian Skagerrak coast showed fast recovery of a local cod population (Moland et al. 2013).

Furthermore, it is possible that only a fraction of the natural area of distribution for local fish stocks are protected by the relatively small size of the NTZ. However, a study on the Norwegian Skagerrak coast has shown recovery of a decimated cod stock after only 4 years closure of a small NTZ ($\sim$1 km^2) (Moland et al. 2013). It is, however, unclear if local cod populations exist confined to such small areas, but show that a response in size or abundance can occur given limited movement of the

cod. In order for cod populations in small geographic locations to recover, it seems to be essential to have a local population usually consisting of a specific more adaptive ecotype (Wårøy Synnes 2020). If the target species is stationary, then fishing closures in smaller marine/fjord areas might result in increased population density and individual sizes. With both Kattegat and North Sea cod using the 8-fjords as nursery grounds it has presently not been possible to discern and verify a continued existence of local stationary cod.

Is there a coastal population of cod separate from offshore cod stocks?

Recent large-scale genomic studies on coastal cod along the Skagerrak coast (Norway and Sweden) included adult cod from 8-fjord area and the Gullmar fjord (Barth et al. 2017). Cod from these areas did not display any significant genetic differentiation from cod from North Sea or from Kattegat suggesting that a mixture of cod of either North Sea or Kattegat origin inhabit the coastal zone of the Swedish Skagerrak coast. The archipelagos along the Skagerrak coast are still considered important nursery areas for Kattegat and North Sea cod ecotypes (Knutsen et al 2018). Similar results have been found in a separate study using early-stage cod eggs sampled in 2013 and 2014 from the 8-fjord area and the Gullmar fjord suggesting that cod genetically resembling cod of North Sea and Kattegat origin use the coastal zone as spawning ground – albeit at low levels. (Svedäng et al. 2018; Cardinale et al. 2019; André et al. 2019). More recent genetics findings indicate that cod along the Swedish west coast do consists of two ecotypes, an "offshore" and a "coastal" (Havsutsikt 2/2022). Within the fjords of Bohuslän such as the area for 8-fjords, the proportions of coastal ecotype is very high. Interesting is that the coastal ecotype within the fjords in Bohuslän does not genetically differ from adult individuals from the Kattegat or Öresund. This means that the fish might have once been juveniles originated from these areas but we do not know for sure.

12.4.2.   Monitoring methods

In the previous evaluation in 2016, the development of demersal fish stocks, e.g. cod, plaice and turbot, were studied using existing monitoring data from the coastal survey, supplemented with hydroacoustics. The combination of a non-lethal method with monitoring was chosen in order to minimize or avoid damaging the already decimated stocks of demersal fish in the area further. The coastal survey should provide a good standard to detect a recovery of the fish stocks given its long time series (from 2002 with some older historical hauls) and large spatial cover along the coast. The 2016 evaluation, which was done six years after establishing the NTZ, clearly shows that the cod stock is still depleted, but is limited in its ability to show trends in cod abundance with the sampling effort used. The coastal survey

data is limited to soft bottom environments since heterogeneous areas with rock and boulders can damage the trawl. The hydroacoustics also exhibit difficulties in detecting fish that are in the close vicinity to complex structures in rocky bottoms typical for coastal areas. However, this method works well for fish in open water in the pelagic zone (Righton et al. 2010). The limitations of the different assessment methods might have reduced the strength of the previous evaluation in making conclusions about the recovery of the fish stocks. At low abundance, cod can appear spatially dispersed as it uses multiple habitats such as hard bottom substrate or the pelagic zone. Sampling at these low abundances and in some of these habitats is difficult with traditional assessment tools. The results from the hydroacoustics indicate extremely low, and decreasing, densities of large fish (presumed cod), while the coastal monitoring only show occasional catches of large individuals. Using these methods, it is therefore difficult to establish if fractions of a stock/population still exist in the area.

A recommendation from the 2016 evaluation was to apply non-harmful monitoring methods that could collect data in complex environments and in vulnerable habitats for monitoring of endangered species. With BRUVs (baited remote underwater stereo-video systems) as a complementary method to other traditionally obtained monitoring data it is possible to obtain information about species and relative abundance of large fish in the NTZ with an improved spatial and temporal resolution without causing mortality of fish. Additionally, the BRUVs can give insight into habitat complexity and composition. Moreover, supporting the results from the egg survey by documenting the presence of larger fish with non-harmful assessment tools increases the acceptance of the current restrictive fishing rules. The results from this evaluation indicate that stereo-BRUV is a valid method for monitoring predatory fish in protected fjord systems, such as the population of cod in the Havstensfjord. Although the relative abundance of cod was low, the consistent differences between habitats, and a spatial distribution where larger individuals partly match previous surveys (hydroacoustics), suggest that this method could be as good (or better) as invasive methods for collecting abundance and body size data of cod in the Havstensfjord. However, stereo-BRUV is still a novel method when it comes to monitoring of predatory fish in ecosystems such as the Swedish fjords. Bias associated with bait use have been discussed in various studies (e.g. Coghlan et al. 2017; Hardinge et al. 2013). Reduced visibility, which proved to be the main limiting factor contributing to invalid deployments during this study, could also severely restrict the usability of this method. Due to only three years of sampling, and the exploratory design of the first deployments, it is currently not possible to conclude if stereo-BRUVs could be used to detect spatiotemporal changes in the relative abundance, length and biomass distribution of cod in relation to the establishment of the NTZ in the Havstensfjord. In order to determine if stereo-

BRUV is a valid method to evaluate the effect of the NTZ, and similar management initiatives in this type of habitat, more data is needed. Therefore, we suggest that data collection with stereo-BRUVs in the Havstensfjord continues as a supplement to other ongoing monitoring. The design of a monitoring program with BRUVs in the Havstensfjord and surrounding areas will be produced and implemented during 2022 within the project.

Other sampling methods such as information on bycatch of cod in the recreational fishery for lobster provides valuable information both on occurrence and size distribution of cod and quantities of cod caught in the lobster fishery. The monitoring of the lobster fishery could be improved by engagement of more record keepers and expanding activity to more regions (i.e. also outside region 5-6 within 8-fjords). Tissue samples from dead large-sized bycaught cod could be collected for genetic analyses of stocks. The occurrence of large individuals in the complex reef habitats generally used in the lobster fishery makes up a valuable addition to the traditional monitoring where larger cod is now almost absent.

12.4.3. Marine mammals and seabirds

The impact of harbour seals on the fish stocks in the fishery-closure area is unknown. Results from studies of harbour seal prey choice in the Skagerrak show that various species of codfish (Gadidae), flatfish (Pleuronectiformes) and herring typically dominate the diet, but that the prey choice differs between areas, years and seasons (Härkönen, 1987, Härkönen, 1988, Härkönen and Heide-Jørgensen, 1991, Aspholm et al., 1995, Olsen and Bjørge, 1995, Sørlie et al., 2020). Preliminary results from an ongoing study in the 8-fjords area indicate that flatfish constitutes the bulk of the seal diet, followed by herring, shorthorn sculpin and whiting (K. Lundström, SLU). A result that, so far, do not indicate that harbour seals exclusively feeds from the focal species in the area. However, more information on prey choice, abundance and spatial dynamics of harbour seals in the 8-fjords area, together with data on composition of fish stocks and ecosystem interactions, is needed to be able to evaluate the ecological role of seals in the area.

A study on the prey choice of cormorants in the 8-fjords area exist, based on diet samples from 2001-2002 show that the diet was dominated by shorthorn sculpin, flatfish and black goby (Lunneryd and Alexandersson, 2005). Two more recent studies from the area, one from 2016 and one from 2019-2020, show that the prey choice varies markedly between years, areas and seasons (Ljunggren, 2017, Fleet, 2021). The most common prey species in these studies were various species of gobies and flatfish. The contribution of codfish to the diet was also significant, but the relative proportion of cod compared to other gadoid species is unknown. The

most recent study also found a different diet compared to earlier studies, with a larger contribution of freshwater species, such as perch and roach (Fleet, 2021). Even in the relatively small 8-fjords area, the diet composition varied a lot between different locations and over time, which might be an indication of migrating birds with different feeding preferences. More information on prey choice and abundance, together with large-scale migrations and fine-scale spatial dynamics, of great cormorants in the 8-fjords area is needed. In addition, data on composition of fish stocks and ecosystem interactions must be combined with cormorant ecological data to be able to evaluate the impact of cormorants on fish in the 8-fjords area.

The various ecological roles of harbour seals and great cormorants and their impact on fish stocks in the NTZ is not clear. Thus, no clear conclusions can be made to what extent these predators may influence a potential recovery of fish stocks, partly due to limited information on the feeding ecology and spatial dynamics of both seals and cormorants. Subsequent monitoring of the development of fish stocks in the NTZ would likely benefit from more ecosystem-based approach and integrate monitoring of harbour seals and great cormorants.

12.4.4.   Conclusions

In conclusion, after 12 years, the NTZ and buffer zones within the Havstensfjord no measurable recovery of demersal fish stocks of the focal species cod, plaice and turbot can be seen. The change of methodology within the used time series, the Swedish coastal survey, as well as the few replicates within NTZ and buffer zone made the analysis before and after the establishment more complicated to interpret. New methods such as BRUVs also indicates a very low number of large fish individuals and no measurable re-establishment of the focal species.

It is important to continue fish monitoring in the area to obtain spatial and temporal information about focal species, as well as to continue develop and implement non-harmful assessment tools to minimize the impact on fish. Furthermore, it is important to monitor parallel changes in ecosystem and environmental conditions so that favorable conditions for a recovery can be evaluated. The ongoing recreational fisheries in the area should also be examined in order to make sure it does not counteract a re-establishment of the focal species. Effects of other factors than fisheries on the fish populations within the area e.g. top predators such as harbour seals and great cormorants is currently under investigation and more information will be available about the ecological effects of these top predators. In all it seems that 12 years is too short a time frame to draw conclusions on the effect of the NTZ and buffer zone within the 8-fjords area. More time is required and more information about other inhibiting factors such as habitat loss, shifts in the

distribution of fish caused by climate change, or large-scale trophic changes in the ecosystem in the area have not been considered in this evaluation.

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12.5.1. Appendix

Table 1. Results of Spearman's rank correlation of fish abundance in the NTZ + buffer zone and the three reference areas during different time periods. Significant results (p<0.05) are shown in red, where the sign of the rho-value show the direction of the trend. Note, a decrease in mesh size took place in 2013. Data from the Swedish coastal survey

Area			NTZ + buffer zone		Gullmar fjord		Coast		Offshore	
Species	Fish size	Time period	p	rho	p	rho	p	rho	p	rho
Cod	Total	2002-2021	0.029	0.288	0.007	0.290	0.000	0.288	0.000	0.432
Cod	Total	2002-2009	0.761	0.067	0.015	-0.424	0.019	-0.234	0.458	-0.116
Cod	Total	2010-2021	0.077	0.303	0.010	0.351	0.000	0.372	0.006	0.322
Cod	Total	2010-2012	0.521	0.264	0.092	0.454	0.551	0.097	0.000	0.787
Cod	Total	2013-2021	0.033	0.421	0.750	0.053	0.003	0.274	0.501	0.094
Cod	< 40 cm	2002-2021	0.021	0.303	0.005	0.303	0.000	0.296	0.000	0.510
Cod	< 40 cm	2002-2009	0.642	0.102	0.022	-0.400	0.018	-0.235	0.440	-0.121
Cod	< 40 cm	2010-2021	0.077	0.303	0.009	0.356	0.000	0.374	0.001	0.400
Cod	< 40 cm	2010-2012	0.521	0.264	0.092	0.454	0.573	0.092	0.001	0.748
Cod	< 40 cm	2013-2021	0.031	0.426	0.691	0.066	0.003	0.276	0.097	0.230
Cod	>= 40 cm	2002-2021	0.091	-0.224	0.039	-0.223	0.001	-0.205	0.029	-0.205
Cod	>= 40 cm	2002-2009	0.410	-0.180	0.538	-0.111	0.695	0.039	0.947	-0.010
Cod	>= 40 cm	2010-2021	0.772	0.051	0.926	0.013	0.020	-0.185	0.004	-0.341
Cod	>= 40 cm	2010-2012	NA (no catch)	NA (no catch)	0.914	0.032	0.960	0.008	0.011	0.594
Cod	>= 40 cm	2013-2021	0.503	-0.137	0.629	-0.081	0.103	-0.151	0.075	-0.247
Plaice	Total	2002-2021	0.069	-0.24	0.124	-0.167	0.003	-0.187	0.143	0.138
Plaice	Total	2002-2009	0.361	-0.199	0.009	-0.453	0.000	-0.401	0.252	0.178
Plaice	Total	2010-2021	0.951	0.011	0.416	-0.114	0.044	-0.161	0.715	-0.044
Plaice	Total	2010-2012	0.521	0.264	0.037	0.548	0.825	0.036	0.065	0.446
Plaice	Total	2013-2021	0.245	-0.236	0.005	-0.452	0.000	-0.395	0.131	-0.210
Plaice	< 24 cm	2002-2021	0.185	-0.176	0.636	-0.052	0.023	-0.141	0.001	0.307
Plaice	< 24 cm	2002-2009	0.371	-0.195	0.028	-0.385	0.000	-0.372	0.253	0.178
Plaice	< 24 cm	2010-2021	0.995	0.001	0.658	-0.062	0.086	-0.137	0.460	0.089
Plaice	< 24 cm	2010-2012	0.521	0.264	0.107	0.435	0.824	0.036	0.183	0.329
Plaice	< 24 cm	2013-2021	0.254	-0.232	0.015	-0.395	0.000	-0.369	0.906	-0.017
Plaice	>= 24 cm	2002-2021	0.000	-0.463	0.000	-0.633	0.000	-0.431	0.046	-0.187
Plaice	>= 24 cm	2002-2009	0.081	-0.372	0.004	-0.491	0.098	-0.166	0.624	0.077
Plaice	>= 24 cm	2010-2021	0.891	-0.024	0.000	-0.545	0.003	-0.240	0.054	-0.230
Plaice	>= 24 cm	2010-2012	0.270	0.433	0.117	0.423	0.831	0.035	0.757	0.079
Plaice	>= 24 cm	2013-2021	0.169	-0.278	0.000	-0.631	0.006	-0.253	0.033	-0.294
Turbot	Total	2002-2021	0.069	-0.24	0.802	0.027	0.398	-0.053	0.444	0.072
Turbot	Total	2002-2009	0.328	-0.213	NA (no catch)	NA (no catch)	0.483	0.070	0.619	-0.078
Turbot	Total	2010-2021	0.139	-0.255	0.037	-0.288	0.225	0.097	0.772	0.035
Turbot	Total	2010-2012	0.437	0.316	0.383	0.243	NA (no catch)	NA (no catch)	1.000	0.000
Turbot	Total	2013-2021	0.526	-0.13	0.557	-0.098	0.431	0.073	0.827	-0.031
Turbot	< 35 cm	2002-2021	0.171	-0.182	0.865	-0.019	0.405	-0.052	0.323	-0.093
Turbot	< 35 cm	2002-2009	0.284	-0.233	NA (no catch)	NA (no catch)	0.486	0.070	0.619	-0.078
Turbot	< 35 cm	2010-2021	0.240	-0.204	0.028	-0.303	0.225	0.097	0.320	-0.119
Turbot	< 35 cm	2010-2012	0.613	0.211	0.851	0.054	NA (no catch)	NA (no catch)	1.000	0.000
Turbot	< 35 cm	2013-2021	0.582	-0.113	NA (no catch)	NA (no catch)	0.431	0.073	0.454	-0.105
Turbot	>= 35 cm	2002-2021	0.203	-0.170	0.565	0.063	0.314	-0.063	0.043	0.190
Turbot	>= 35 cm	2002-2009	0.576	0.123	NA (no catch)	NA (no catch)	0.904	0.012	NA (no catch)	NA (no catch)
Turbot	>= 35 cm	2010-2021	0.014	-0.413	0.556	-0.083	NA (no catch)	NA (no catch)	0.392	0.103
Turbot	>= 35 cm	2010-2012	0.613	0.214	0.234	0.327	NA (no catch)	NA (no catch)	NA (no catch)	NA (no catch)
Turbot	>= 35 cm	2013-2021	0.093	-0.336	0.557	-0.098	NA (no catch)	NA (no catch)	0.827	-0.031

Table 2. Results of Spearman's rank correlation of fish biomass in the NTZ + buffer zone and the three reference areas during different time periods. Significant results (p<0.05) are shown in red, where the sign of the rho-value show the direction of the trend. Note, a decrease in mesh size took place in 2013. Data from the Swedish coastal survey.

Area			NTZ + buffer zone		Gullmar fjord		Coast		Offshore	
Species	Fish size	Time period	p	rho	p	rho	p	rho	p	rho
Cod	Total	2002-2021	0.929	0.012	0.158	-0.154	0.311	-0.063	0.527	-0.060
Cod	Total	2002-2009	0.689	-0.088	**0.006**	-0.473	**0.027**	-0.220	0.433	-0.122
Cod	Total	2010-2021	0.456	0.130	0.799	-0.036	0.416	0.065	0.076	-0.212
Cod	Total	2010-2012	0.708	0.158	0.055	0.510	0.227	0.195	**0.000**	0.787
Cod	Total	2013-2021	0.083	0.347	**0.025**	-0.364	0.276	0.101	0.283	-0.150
Cod	< 40 cm	2002-2021	0.843	0.027	0.325	-0.107	0.518	-0.040	0.274	0.103
Cod	< 40 cm	2002-2009	0.848	-0.043	**0.002**	-0.523	**0.012**	-0.249	0.720	-0.056
Cod	< 40 cm	2010-2021	0.447	0.132	0.908	-0.016	0.405	0.067	0.229	-0.144
Cod	< 40 cm	2010-2012	0.708	0.158	**0.015**	0.624	0.321	0.160	**0.001**	0.748
Cod	< 40 cm	2013-2021	**0.030**	0.427	0.057	-0.312	0.248	0.108	0.194	-0.181
Cod	>= 40 cm	2002-2021	0.114	-0.210	**0.041**	-0.221	**0.001**	-0.205	**0.024**	-0.211
Cod	>= 40 cm	2002-2009	0.410	-0.180	0.669	-0.077	0.592	0.054	0.795	-0.041
Cod	>= 40 cm	2010-2021	0.752	0.055	0.951	0.009	**0.020**	-0.186	**0.004**	-0.337
Cod	>= 40 cm	2010-2012	NA (no catch)	NA (no catch)	0.914	0.032	0.939	0.012	**0.036**	0.501
Cod	>= 40 cm	2013-2021	0.535	-0.127	0.615	-0.084	0.103	-0.151	0.488	0.090
Plaice	Total	2002-2021	**0.005**	-0.367	**0.001**	-0.359	**0.000**	-0.292	0.807	-0.023
Plaice	Total	2002-2009	0.299	-0.226	**0.009**	-0.451	**0.000**	-0.361	0.413	0.128
Plaice	Total	2010-2021	0.999	0.000	**0.035**	-0.290	**0.002**	-0.251	0.260	-0.135
Plaice	Total	2010-2012	0.613	0.211	**0.037**	0.548	0.998	0.000	0.107	0.393
Plaice	Total	2013-2021	0.119	-0.314	**0.000**	-0.596	**0.000**	-0.487	**0.043**	-0.279
Plaice	< 24 cm	2002-2021	0.082	-0.231	0.169	-0.150	**0.001**	-0.198	**0.009**	0.244
Plaice	< 24 cm	2002-2009	0.432	-0.171	**0.036**	-0.367	**0.000**	-0.343	0.253	0.178
Plaice	< 24 cm	2010-2021	0.912	-0.019	0.329	-0.136	**0.015**	-0.193	0.651	0.055
Plaice	< 24 cm	2010-2012	0.613	0.211	0.078	0.472	0.755	0.051	0.149	0.355
Plaice	< 24 cm	2013-2021	0.124	-0.309	**0.002**	-0.500	**0.000**	-0.448	0.832	-0.030
Plaice	>= 24 cm	2002-2021	**0.000**	-0.468	**0.000**	-0.630	**0.000**	-0.429	**0.037**	-0.195
Plaice	>= 24 cm	2002-2009	0.092	-0.360	**0.005**	-0.486	0.137	-0.149	0.775	0.045
Plaice	>= 24 cm	2010-2021	0.882	-0.026	**0.000**	-0.540	**0.002**	-0.251	0.077	-0.211
Plaice	>= 24 cm	2010-2012	0.270	0.433	0.136	0.404	0.912	-0.018	0.640	0.118
Plaice	>= 24 cm	2013-2021	0.162	-0.282	**0.000**	-0.628	**0.005**	-0.259	**0.042**	-0.281
Turbot	Total	2002-2021	**0.019**	-0.307	0.801	0.028	0.393	-0.053	0.440	0.073
Turbot	Total	2002-2009	0.593	-0.117	NA (no catch)	NA (no catch)	0.486	0.070	0.675	-0.066
Turbot	Total	2010-2021	0.063	-0.319	**0.037**	-0.287	0.225	0.097	0.759	0.037
Turbot	Total	2010-2012	0.613	0.211	0.335	0.267	NA (no catch)	NA (no catch)	0.341	0.302
Turbot	Total	2013-2021	0.522	-0.131	0.557	-0.098	0.431	0.073	0.853	-0.026
Turbot	< 35 cm	2002-2021	0.104	-0.216	0.865	-0.019	0.400	-0.053	0.325	-0.093
Turbot	< 35 cm	2002-2009	0.299	-0.226	NA (no catch)	NA (no catch)	0.491	0.069	0.675	-0.066
Turbot	< 35 cm	2010-2021	0.141	-0.254	**0.028**	-0.303	0.225	0.097	0.320	-0.119
Turbot	< 35 cm	2010-2012	0.880	-0.053	0.851	0.054	NA (no catch)	NA (no catch)	1.000	0.000
Turbot	< 35 cm	2013-2021	0.577	-0.114	NA (no catch)	NA (no catch)	0.431	0.073	0.454	-0.105
Turbot	>= 35 cm	2002-2021	0.141	-0.195	0.565	0.063	0.314	-0.063	**0.041**	0.192
Turbot	>= 35 cm	2002-2009	0.604	0.114	NA (no catch)	NA (no catch)	0.904	0.012	NA (no catch)	NA (no catch)
Turbot	>= 35 cm	2010-2021	**0.018**	-0.399	0.556	-0.083	NA (no catch)	NA (no catch)	0.374	0.107
Turbot	>= 35 cm	2010-2012	0.810	0.107	0.234	0.327	NA (no catch)	NA (no catch)	NA (no catch)	NA (no catch)
Turbot	>= 35 cm	2013-2021	0.120	-0.313	0.557	-0.098	NA (no catch)	NA (no catch)	0.874	-0.022

Table 3. Results of Spearman's rank correlation of fish length classes in the NTZ + buffer zone and the three reference areas during different time periods. Significant results (p<0.05) are shown in red, where the sign of the rho-value show the direction of the trend. Note, a decrease in mesh size took place in 2013. Data from the Swedish coastal survey.

Area			NTZ + buffer zone		Gullmar fjord		Coast		Offshore	
Species	Fish size	Time period	p	rho	p	rho	p	rho	p	rho
Cod	Total	2002-2021	0.235	0.158	0.976	0.003	0.975	-0.002	0.183	0.125
Cod	Total	2002-2009	0.514	0.143	0.109	-0.285	**0.036**	-0.209	0.885	0.023
Cod	Total	2010-2021	0.528	0.110	0.975	-0.004	0.396	0.068	0.723	-0.043
Cod	Total	2010-2012	0.708	0.158	0.083	0.466	0.266	0.180	**0.000**	0.835
Cod	Total	2013-2021	**0.037**	0.412	0.219	-0.204	0.126	0.142	0.627	-0.068
Plaice	Total	2002-2021	**0.010**	-0.335	**0.001**	-0.358	**0.000**	-0.378	0.973	-0.003
Plaice	Total	2002-2009	0.935	-0.018	**0.039**	-0.363	**0.012**	-0.249	0.328	0.153
Plaice	Total	2010-2021	0.392	-0.149	**0.006**	-0.376	**0.000**	-0.363	0.271	-0.132
Plaice	Total	2010-2012	0.552	0.240	0.058	0.504	0.899	-0.021	0.329	0.244
Plaice	Total	2013-2021	0.374	-0.181	**0.012**	-0.405	**0.000**	-0.396	0.158	-0.196
Turbot	Total	2002-2021	**0.028**	-0.289	0.819	0.025	0.386	-0.054	0.481	0.067
Turbot	Total	2002-2009	0.597	-0.116	NA (no catch)	NA (no catch)	0.468	0.073	0.646	-0.072
Turbot	Total	2010-2021	**0.037**	-0.355	**0.032**	-0.296	0.224	0.097	0.796	0.031
Turbot	Total	2010-2012	0.493	0.269	0.509	0.185	NA (no catch)	NA (no catch)	1.000	0.000
Turbot	Total	2013-2021	0.560	-0.119	0.557	-0.098	0.429	0.074	0.845	-0.027

13. No-take zone for lobster and wrasses at Kåvra, Skagerrak

Diana Perry, Andreas Wikström, Håkan Wennhage, Andreas Sundelöf

Håkan Wennhage (above) and Mattias Sköld (below) measuring the European lobster, Homarus gammarus*, in the no-take-zone Kåvra. Photo credit: Mattias Sköld*

Summary

The no-take zone (NTZ) Kåvra, on the west coast of Sweden is one of the oldest marine closure areas in the country, having been closed since 1989. The area was initially closed in order to create a place where researchers could study European lobster biology, *Homarus gammarus*, in the absence of harvesting. There was a direct and steady increase in number of individuals and size in response to the closure, which has continued over the last three decades. This has resulted in approximately four times higher catch per unit effort (CPUE), significantly larger individuals, and 7-9 times higher reproductive potential within Kåvra NTZ compared to surrounding reference areas. Despite the positive response of the focal species *H. gammarus* to the NTZ, the fish and benthic communities have not responded in the same positive way. No closure effects were found for the fish community, as represented by the relatively stationary family of wrasses (Labridae). Additionally, the benthic edible crab species, *Cancer pagurus*, shows markedly lower numbers within the NTZ boundaries, compared to the two reference areas, potentially as a result of interspecific interactions with *H. gammarus*. In conclusion, after three decades of closure, Kåvra continues to show positive effects on the focal species *H. gammarus*, however no other species studied with the current evaluation show any effects due to the NTZ.

13.1. Background

Due to European lobster, *Homarus gammarus,* population decreases along the west coast of Sweden, with all-time lows observed from the 1950s-1975, a series of regulations were set in place. A moratorium on female catches began in 1985, followed by minimum carapace length restrictions in 1994, and a further ban on fishing gear type in 2003 (Sundelöf *et al.*, 2013). Both commercial and recreational lobster fishing take place in Sweden. However, recreational fishing catches are not included in official landing reports. This is despite recreational fishing being estimated to have accounted for 85-90 percent of all lobster fishing gear from 2007-2016, and calculated to approximately 75 percent of the landings from 2014 until 2016 (Havs- och Vattenmyndigheten, 2019). Therefore, further regulations were set in place in 2017 which reduced the number of fishing pots for commercial fishers to 40 and down to 6 pots for recreational fishers, as well as increasing the minimum carapace length landing size from 80 mm to 90 mm. Additionally, restrictions on fishing season were set in place limiting the days commercial and recreational fishers can be active. The fishing period, before the new regulations in 2018, was set to the first Monday after the 22nd of September and lasted until the

end of April. The new regulations shortened the fishing period to the end of November for recreational fishers and the end of December for commercial fishers.

As well as implementing the moratorium on berried females in the mid-1980s to increase reproductive output, a no-take zone (NTZ) was created on the west coast of Sweden, Kåvra, to increase the biological knowledge of the species. This NTZ is now classified as a long-term closure, as the area has restricted lobster fishing particularly, and all other forms of fishing besides hook-and-line since 1989 (Øresland and Ulmestrand, 2013). This location is unique within Sweden given the areas excellent natural rock habitats ideal for lobster, in contrast to other NTZs such as Vinga farther south on Sweden's west coast. Vinga, although it is also a long-term closure area that has been closed since 2002, has been partly created using artificial reefs, potentially confounding the two different management actions and not providing the same quality for understanding lobster biology within natural habitats. Kåvra was protected in order to create a place for researchers to study the development of the lobster population biology in the absence of harvesting (Moland, Ulmestrand, *et al.*, 2013). As such, it has been very successful in increasing size, density and abundance of lobster within the boundaries of the NTZ, however the effects on the fish community have not been as well documented until recently (Bergström *et al.*, 2016). The effects of the closure yielded a quick response from the lobster population, with a reduction in fishing mortality of 75 percent within the first four years of NTZ establishment (Bergström *et al.*, 2007). Additionally, there is evidence that the density of lobster within the NTZ has increased since closure, while the density in the surrounding unprotected areas showed a decline, according to Moland *et al.* (2013). This lack of spillover effect seen in the lobster populations surrounding the NTZ can be attributed to the extremely restricted movements of adult lobster, with mark-recapture studies in Kåvra showing that of the more than 4000 individuals tagged only 1.4 percent were recaptured farther than 1 km from the NTZs boundaries (Øresland and Ulmestrand, 2013). High adult residency rates have also been documented in European lobster in Norway (Huserbråten *et al.*, 2013). The increase in number of individuals, and the larger size of females within Kåvra has resulted in 3.5 times higher egg production rates compared to a similarly sized unprotected area (Bergström *et al.*, 2007). However, it is projected that larvae within Kåvra can only spread approximately 16 km^2 to surrounding areas because of a persistent retention of the water mass below 16 meters depth where the larvae are found (Øresland and Ulmestrand, 2013).

While the closure benefits have been clearly documented for the target species, the goal of the NTZ was not to document the effects of the reserve over time, and therefore control fishing has not been conducted outside the NTZ in reference areas,

until recently. Fishing using fyke nets and lobster pots was performed over the two year period 2017-18 within Kåvra NTZ, as well as two reference areas, where fishing is permitted, in order to determine if differences in size, diversity, and/or abundance could be found in the fish/crustacean assemblage.

13.2. Methods

The study was conducted in three small coastal areas in Bohuslän on the Swedish west coast outside Lysekil (See figure 1.). The NTZ Kåvra (58.33N; 11.63E) was compared with two adjacent reference areas, St Kornö (58.29N; 11.37E) and Långö (58.25N; 11.38E), both used as commercial fishing grounds by local fishermen targeting mainly lobsters and wrasses (Bourlat et al. 2021). In St Kornö and Långö there are also recreational fisheries targeting lobsters and fish such as mackerel, cod, flatfish, Ballan wrasse and sea trout.

The main area Kåvra is approximately a 2.1 km^2 large marine protected area with a depth interval ranging from 0 to 30 m where fishing for lobster has been banned since 1989, and where regulations only allow for hook and line fishery. Kåvra is situated nearshore and dominated by exposed and submerged bedrock islets separated by channels 10 to 25 m deep. The substratum of the channels largely consists of sediments (sand or mud). The NTZ is situated in an area of historically productive lobster fishing grounds.

St Kornö is a 2.5 km^2 large marine coastal area located 2.3 km south of the marine protected area Kåvra. Långö is a 2.3 km^2 marine coastal area located 6.7 km south of the protected area Kåvra. St Kornö and Långö are both used as reference areas in this study and are very similar to the NTZ Kåvra with regards to depth, substrate and topography.
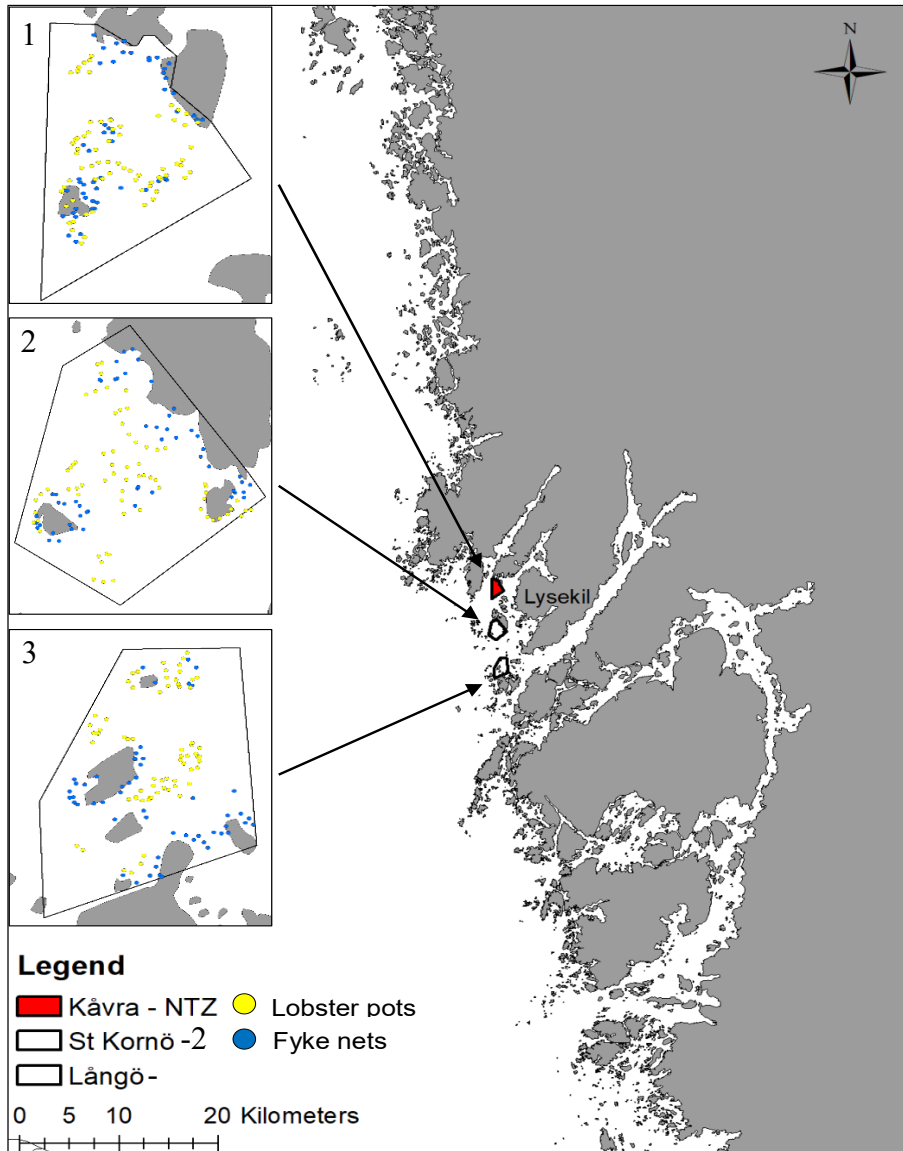
Figure 1. Map of the study area along the coast of Bohuslän, outside the city of Lysekil. Kåvra NTZ (top-1) marked in red has been closed for fishery for 30 years. The two areas St Kornö (middle-2) and Långö (below-3) are used as reference sights. The yellow circles represent the positions sampled with lobster pots and blue circles represent positions sampled with fyke nets. Sampling was conducted in August/September 2017 and 2018, respectively.

13.2.1.   Fishing gear

The fyke nets used in the study to document the fish and crustacean assemblage were the same as used by commercial wrasse fishermen in the area. The fyke nets were structured with a double-codend with 3 chambers and 7 hoops, the largest measuring 30 cm, with a 6 m leader, with a mesh size of 10 mm in the codend and 15 mm in the leader. In Kåvra 110 fyke net stations were sampled, 108 in Långö and 108 in St. Kornö. Sites for fyke net sampling were randomized within each area (St Kornö, Långö and Kåvra) across three different depth intervals, 0-6, 6-10 and 10-20 m (Havs- och Vattenmyndigheten, 2016). Average soak time for each station

was 23.6 hours, with a minimum of 18.3 and a maximum of 28.3 hours. Lobster pots- targeting European lobster, were 92 cm long, 45 cm wide, 40 cm high, structured with two entrances to the baited chamber (one on each side) with a diameter of 120 mm. All traps had a mesh size of 50 mm and were baited with mackerel or herring. The pots were set at depths from approx. 5-30 meters (4.6-29), total number of stations for both years 133 Kåvra, 132 Långö, 130 St Kornö. The average soak time for each station was 45.2 hours, with a minimum of 24 and a maximum of 96 hours.

All lobsters in the catch were recorded, including information on carapace length and sex. Fish caught in the fyke nets were identified and individually measured (length in cm). All other catches were identified and number of individuals per species were recorded. Catches were handled with care and released back in the water at the place of capture. Historical data from time of NTZ closure in 1989 to 2007 were used to calculate the lobster catch average size per sex (both fyke net and pot data were used historically).

13.2.2.   Statistical analysis

All statistical analyses were performed using STATISTICA 64 version 13. Data were checked for normal distribution prior to analysis using the Shapiro-Wilk test. Even after transformation the data did not meet the assumptions of the parametric analysis of variance (ANOVA), and therefore the non-parametic Kruskal-Wallis analysis of variance was performed on the pooled 2017-18 data. Catch per unit effort (CPUE) per 24 hours was calculated for *Homarus gammarus* as well as for the Labridae family. For *H. gammarus* the CPUE was standardized for soak time following the method proposed by Moland et al 2013, in order for the current CPUE to be comparable to the historical pot data collected in Kåvra from 1994 until 2007 (see figure 2A). For the previous period data from 1994-2007, hereafter called "early period", the reference data refers to the "reference fleet" which were data collected from volunteer fisherman catch diaries. Potential egg production is a proxy for reproductive potential for *H. gammarus* and was calculated using methods established by Ulmestrand (2003) and Sundelöf *et al.* (2015), where first the proportion berried (PB) females are calculated using carapace length (CL) where a=37.62, b=0.48, c=0.01 (equation 1), followed by the number of potential eggs using linear regression (equation 2) where x is the carapace length value per individual, and this information is subsequently multiplied by the number of caught individuals to determine reproductive potential per area (Kåvra, Långö, and St Kornö). Thus, reproductive potential accounts for both the total number of captured female individuals as well as the size per individual female for each area.

$$PB = 1\frac{1}{1+e^{(a-b*CL)}}/e^{(c*CL)} \qquad\qquad \text{Eq. 1}$$
$$y = 372.09x - 22598 \qquad\qquad \text{Eq. 2}$$

13.3. Results

13.3.1.   Lobster data

There were a total of 527 *H. gammarus* individuals caught during the two fishing seasons in 2017 and 2018, with 348 caught within the NTZ, and 98 and 81 from the reference areas Långö and St. Kornö, respectively. The catch per unit effort (CPUE) for *H. gammarus* in the NTZ Kåvra was an average of 0.33 in 2017 and 0.45 in 2018, while it was only 0.09 in the reference area Långö in 2017 and 0.17 in 2018. The second reference area had a CPUE that was lower at 0.11 in 2017 and 0.10 in 2018. These results are shown in relation to the CPUE of Kåvra and other fished areas from historical data published by Moland et al 2013 (See Figure 2a and b).
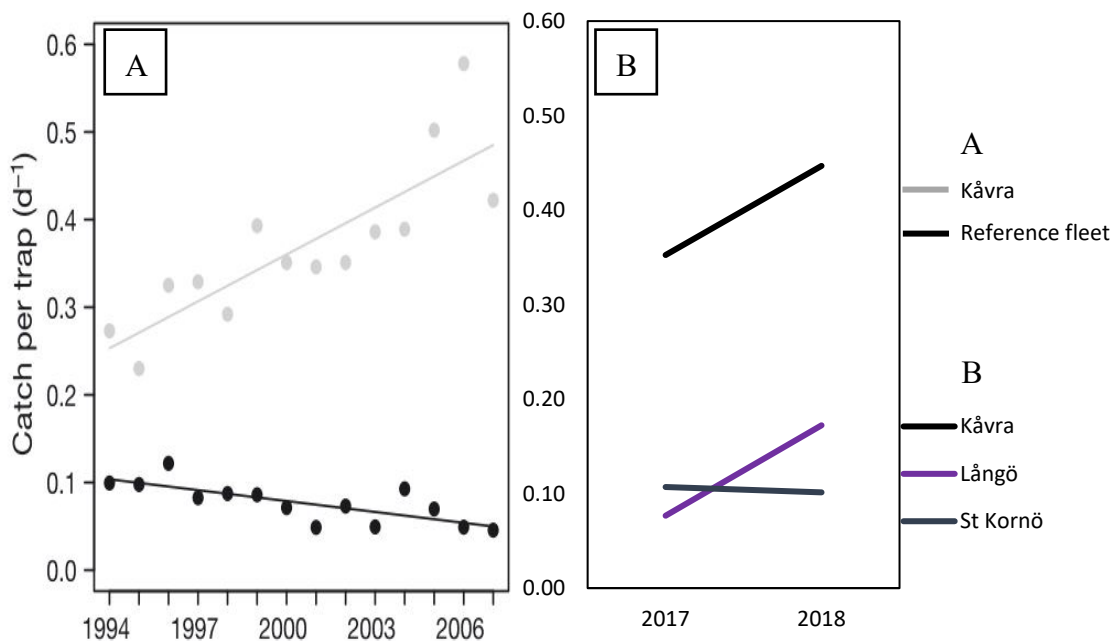
Figure 2: Homarus gammarus *catch per unit effort per lobster pot per day from 2017 and 2018 (recalculated using the Moland et al 2013 method) contrasted to the early period CPUE in Kåvra from Moland et al 2013. Historical data in image on the left (A) and recent data on the right (B). For the historical data in image (A) the reference fleet is shown in black while the CPUE for Kåvra is shown in gray*

When reviewing the average size per sex within Kåvra using data from a previous survey (1989-2007) and fishing in 2017-2018 there is a general trend in increasing

size for both males and females until the end of the early period, 2007, with the average size in 2007 being 97.5 mm for females and 104.4 mm for males, respectively. The current data shows a continued increase in the average size of both males and females from the end of the historical time frame in 2007 (Figure 3).
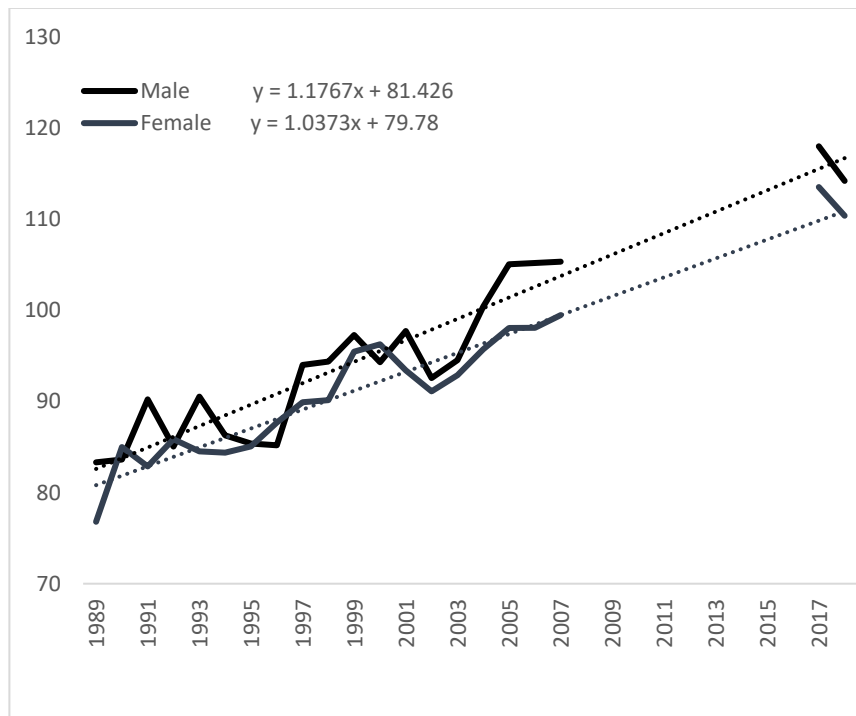

Figure 3: Average size, measured in millimeter carapace length, of male (black) and female (gold) Homarus gammarus *within Kåvra NTZ from time of closure in 1989 until 2018. Dashed black and gold lines denotes the linear model fitted to data for the years 1989-2007 and extrapolated to the years 2017-2018 when the NTZ was revisited. Linear regression equations shown for male and female average size over years.*

For the period 2017-2018 using both gear types, lobsters were larger in the NTZ (range 52-175 mm) than the two reference areas (57-117 mm in Långö and 60-117 in St. Kornö) (Figure 4). The maximum size, measured as carapace length, caught within the NTZ was 175 mm, while the two reference areas both had a maximum recorded carapace length of only 117 mm. Kruskal-Wallis test for comparison of lobster size between Kåvra and the reference areas shows a significant difference with larger individuals caught within the NTZ $H(2)=201.93$, $p<0.0001$. Additionally, the post-hoc test shows a significant difference of the NTZ compared to both the reference areas, $p<0.001$ for Långö and $p=0.004$ St Kornö, however lobsters in the two reference areas do not differ from one another in size.
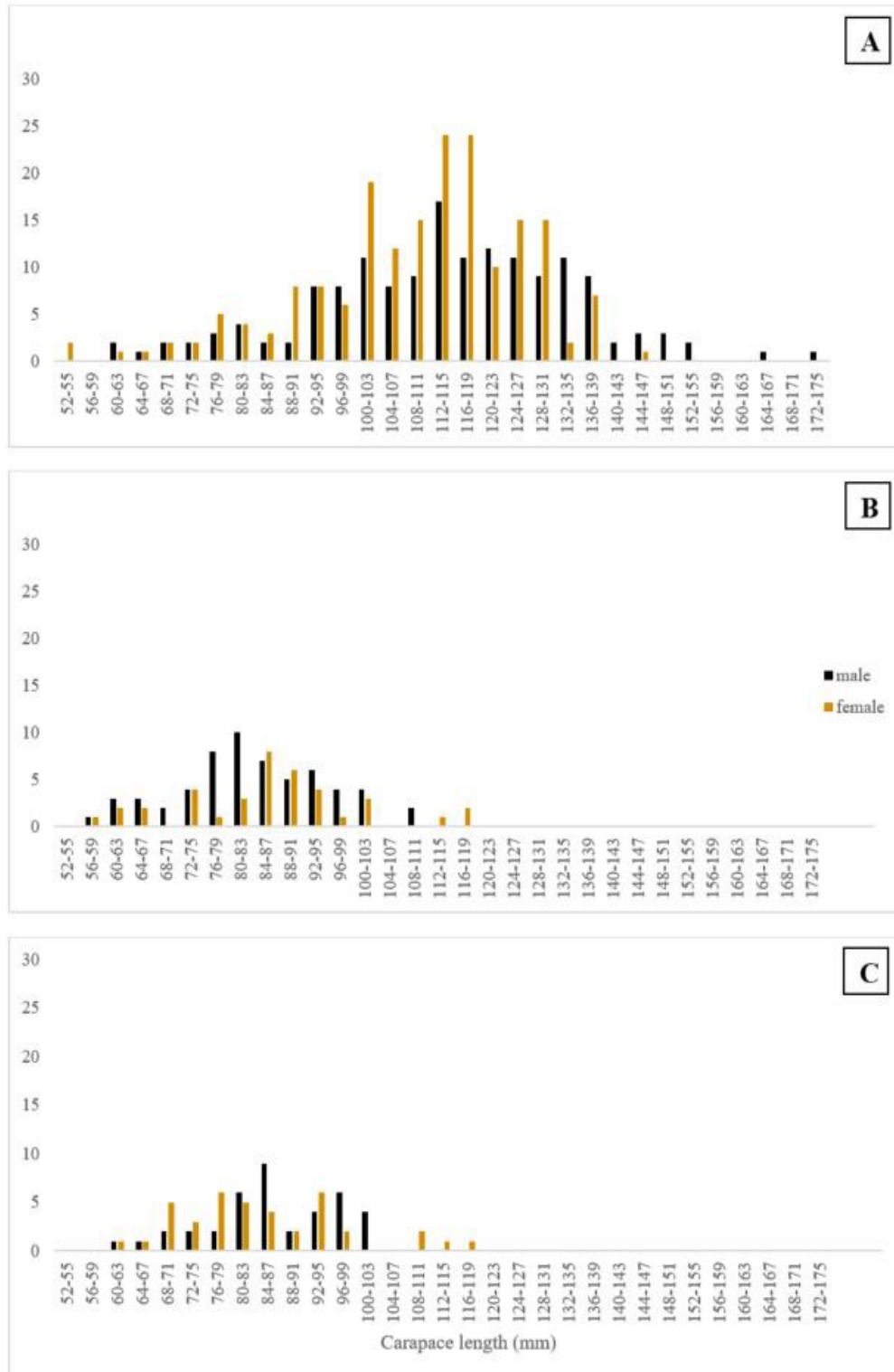
Figure 4: The total number of individuals per size distribution of Homarus gammarus for males (black) and females (gold) for the NTZ Kåvra (A) and two reference areas, Långö (B) and St. Kornö (C), respectively, fished from 2017-18.

The average reproductive potential of lobsters was highest in Kåvra, with an average of approximately 6100 (±1033 SD) eggs produced per mature female per year in 2017, while it was slightly lower at around 5700 (±1593 SD) in 2018. The

reference area Långö had a reproductive potential of approximately 3700 (± 2167 SD) eggs in 2017 and 3600 (± 1659 SD) in 2018, while St. Kornö had the lowest average egg production at nearly 3000 (± 2204 SD) in 2017 and approximately 2800 (± 2246 SD) in 2018. Consequently, the total reproductive potential by the sampled individuals per year was nearly seven times higher in Kåvra compared to Långö and almost nine times higher compared to the other reference area St. Kornö (Figure 5).
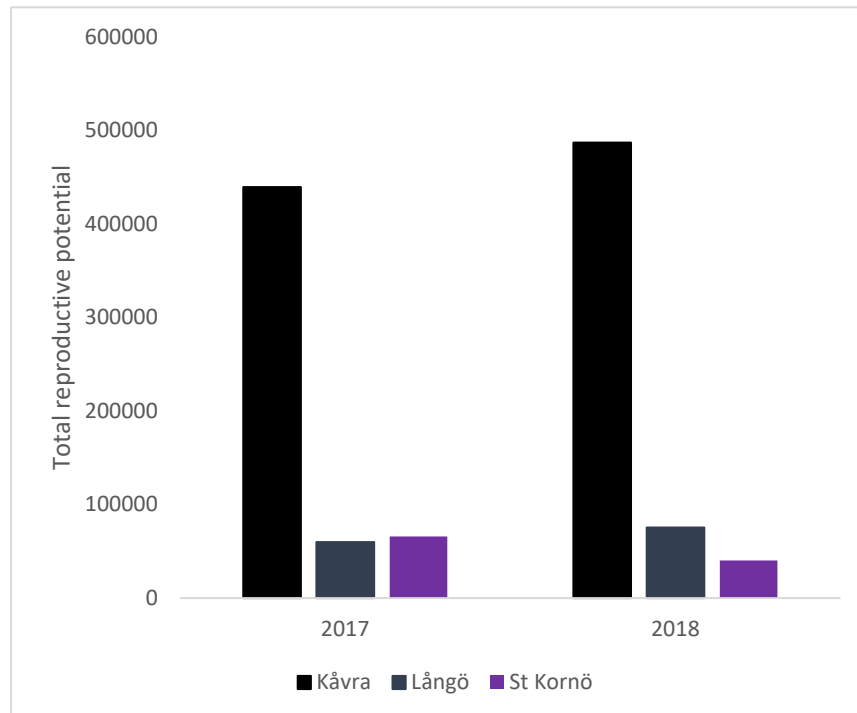

Figure 5: Total reproductive potential in total number of eggs produced per site per year 2017-18. Kåvra data are shown in black, Långö in gold, and St. Kornö in purple.

13.3.2. Ecosystem effects

Fish community

When evaluating the entire marine community within the NTZ and reference areas 22 families observed under the 2017-18 fishing period, of which 13 families were fish. For the NTZ 4459 individuals were caught from 19 different families, while for the two reference areas there were 4615 individuals from 19 families caught at Långö and 5081 individuals from 19 families at St Kornö.

The most abundant family caught in the NTZ, as well as the two reference areas, were wrasses (Labridae). Four wrasse species were caught within the NTZ, whereas both reference areas had five species. However, no difference between the total

number of wrasses caught between sites was found (Kruskal-Wallis H(2)=0.70, p=0.71). The CPUE for all Labridae species was generally higher in 2017 compared to 2018 for all sites (Figure 6). For all sites, the most abundant wrasse species was the corkwing wrasse, *Symphodus melops*.
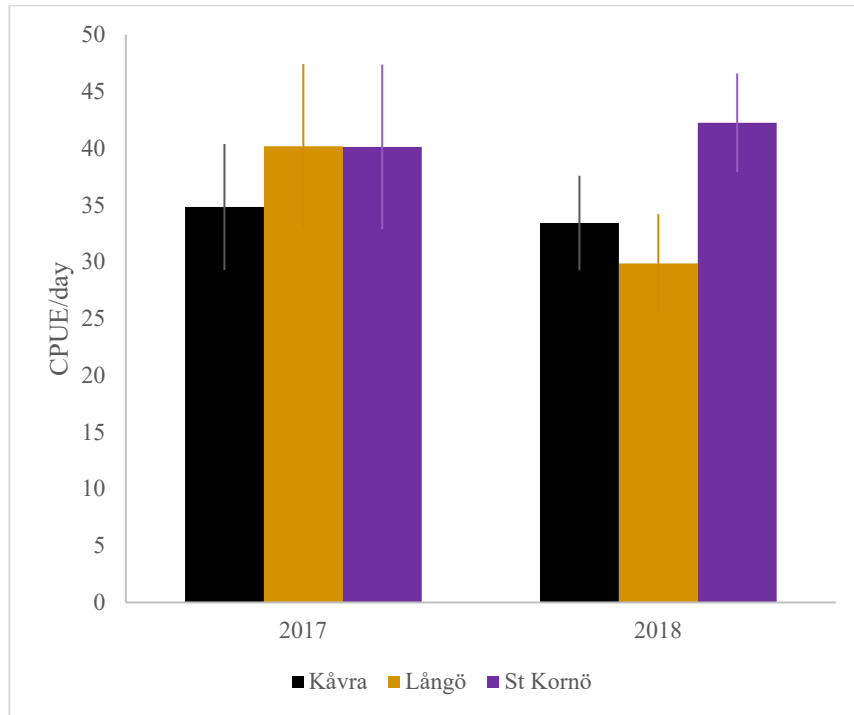

Figure 6: Catch per unit effort per day for all species within the Labridae family from 2017-18. The NTZ Kåvra data are shown in black, Långö in gold, and St. Kornö in purple with error bars showing standard deviation (SD).

The average size of fish within the Labridae family did not differ between the NTZ and reference areas H(2) = 1.433, p = 0.489 (Figure 7).
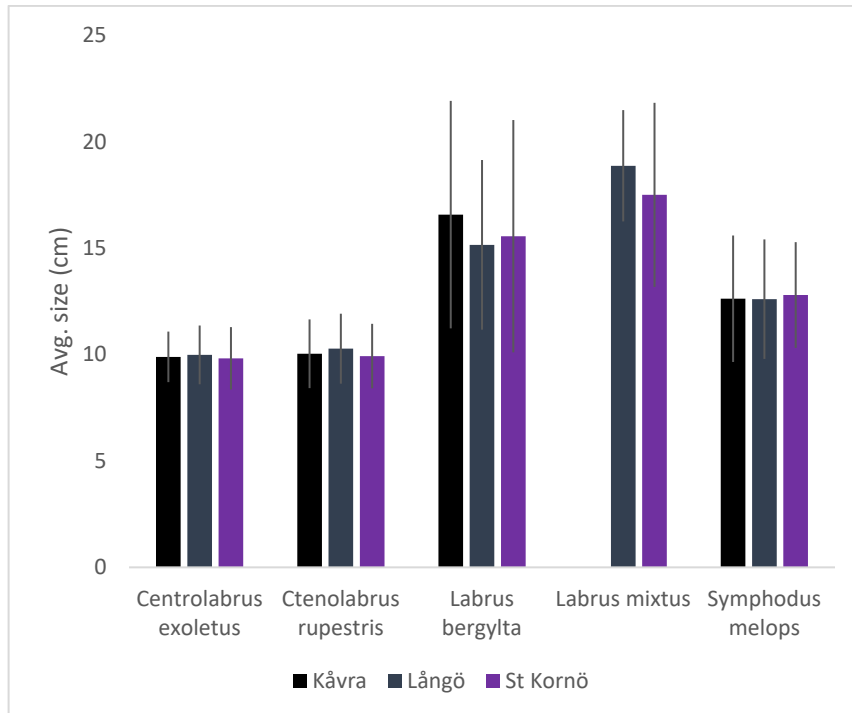
Figure 7: Labridae species average size (cm) per location. The NTZ Kåvra data are shown in black, Långö in gold, and St. Kornö in purple with error bars showing standard deviation (SD).

Benthic fauna

The benthic crab species, *Cancer pagurus*, was approximately 3-4 times more abundant in the reference areas outside the NTZ, with the numbers caught in St. Kornö highest for both years 2017-18 (Figure 8). The total number caught within the NTZ was 131 individuals compared to 410 in Långö and 500 in St. Kornö. The NTZ Kåvra had significantly less benthic crabs compared to the two reference areas ((Kruskal-Wallis H(2) = 28.89, p<0.001) p=0.0002 for Långö and p<0.0001 for St Kornö), however the two reference areas did not differ from one another.
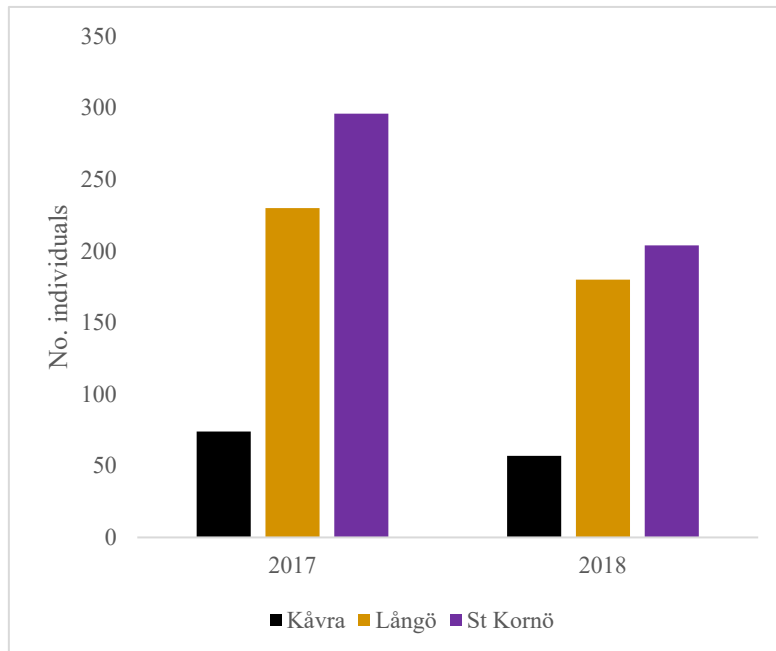
Figure 8: Total number of Cancer pagurus *individuals caught per year for the NTZ Kåvra (black) and the two reference areas Långö (gold) and St. Kornö (purple) in 2017-18 using fyke net and lobster pots.*

13.4. Discussion

The Kåvra NTZ was established in 1989 and since the closure of the area to fishing there has been a consistent and clear increase in number and size of lobster caught within the Kåvra NTZ (Moland, Ulmestrand, *et al.*, 2013; Øresland and Ulmestrand, 2013; Øresland, Oxby and Oxby, 2018). The current study also clearly illustrates that the fishing closure increased the abundance and average size of *Homarus gammarus* in Kåvra NTZ compared to the reference areas. The increased abundance also leads to a much higher CPUE in Kåvra. However, the current time frame does not show a continued increase in CPUE, calculated by numbers of individuals, since the values yielded from the historical sampling ending in 2007 (Moland, Ulmestrand, *et al.*, 2013), perhaps indicating that Kåvra has reached carrying capacity for *H. gammarus* after three decades of closure. The pots used for sampling lobster in Kåvra 2017 and 2018 are size-selective, and miss the catches of both the smallest individuals and the largest males, indicating that the maximum size of lobsters in the area could be higher (Øresland, Oxby and Oxby, 2018).

The rapid and continued response of the increased abundance and size of the European lobster, *H. gammarus*, within the established marine protected area (MPA) has been shown in Norway as well (Knutsen *et al.*, 2022). The effect of the Norwegian NTZs has even been shown to increase the size structure, thereby rebuilding phenotypic complexity, of the lobsters within the marine protected areas

providing protection from harvest selection (Fernández-Chacón *et al.*, 2020). This buffering against harvest selection is important given that harvesting of *H. gammarus* may change mating behavior, and could lead to fisheries-induced evolution towards smaller body size (Sørdalen *et al.*, 2018). There is evidence that there has been an ecological spillover effect of the Norwegian reserves with increased lobster biomass near reserve boundaries (Thorbjørnsen *et al.*, 2018), however, adult lobsters display restricted movement and high residency. Despite the limited adult dispersal, a high level of gene flow has been shown in Skagerrak, indicating the importance of larval drift (Huserbråten *et al.*, 2013). However, Øresland and Ulmestrand (2013) noted that there was high retention of the deep water mass around Kåvra, not expanding more than approximately 16 km^2, which they suggest would lead to a relatively localized outflow of larvae. While the flow of larvae may not extend as far from the Swedish NTZ Kåvra as it was proposed to do in Norway, the current study's two reference areas both fall within 16 km^2, a possible reason for the slight increase in CPUE from the end of the historical data sampling period in 2007 to what was sampled in 2017-18 (figure 2b). It should be noted, however, that while there has been a slight increase in CPUE in the surrounding reference areas, without genetic sampling it is not currently possible to determine whether this is a result of larval drift from the NTZ. It may also be a result of technological creep, a phenomenon where increased catches are observed due to the increased quality of gear over time, rather than a true increase in the lobster population (Kleiven *et al.*, 2022).

While the NTZ has been very effective for increasing the lobster population within the areas borders, this study did not find any broader effects on fish. Certain fish species found in the area, such as members of the Labridae family, are highly residential (Halvorsen *et al.*, 2021), and are targeted by the fishery for use in aquaculture in the region (Bourlat *et al.*, 2021). They could therefore be expected to show increases in abundance as a result of the protection of fishing, however the current study found no such effects. This is in contrast to results seen in Norway where, in fact, not only have the Norwegian NTZs been effective at increasing the size and abundance of lobsters in the area, the establishment of the reserves has also lead to increases in the abundance of other target fisheries populations such as cod and different species of wrasse (Moland, Olsen, *et al.*, 2013; Halvorsen *et al.*, 2017). The ineffectiveness of Kåvra at increasing the abundance of non-target species such as Atlantic cod, as was shown in Norway, may be related to the extremely low cod abundances found on the Swedish west coast and the critical state of the population (Andersson, Högvall and Larsson, 2021), making recovery difficult despite decades of different management strategies in the area. Within the NTZ Vinga also on the west coast of Sweden there was an initial increase in number and size of cod within

the borders of the closed area, however nearly a decade after closure no differences were found (Wikström *et al.*, 2016)

Interestingly, while Kåvra NTZ shows positive effects on the lobster population and no effects on the fish community, lower abundances of benthic brown crab species *Cancer pagurus*, are found in the NTZ compared to the reference areas Långö and St Kornö (figure 6). This lower abundance of *C. pagurus* has been noted in previous studies using dive surveys, and as the authors point out, these low numbers are present despite the protection from harvesting (Øresland, Oxby and Oxby, 2018). Therefore, it is likely that there are inter-specific interactions taking place between the brown crab and the European lobster.

While the Kåvra NTZ is a highly effective lobster protection area with clear long-term effects on the size, abundance, and reproductive potential of *H. gammarus* within the NTZ boundaries, there are no clear benefits to the fish community as a result of the decades of fishing exclusion. This may be a result of the NTZs small size given the mobility of the fish assemblage, even if some species and families such as the Labridae are highly resident, or an indication that these species are not depleted and have high abundances in the NTZ and reference areas. Even if small reserves may be limited in the extent of protection for mobile species (Turnbull *et al.*, 2018), there is still much evidence showing the effectiveness for certain target species, such as the lobster in Kåvra. Additionally, there has been much discussion and support for the idea of networks of small reserves as a valuable management strategy both for enhancing fisheries and for conservation benefits (Gaines *et al.*, 2010; Green, White and Kilarski, 2013; Fovargue, Bode and Armsworth, 2018; Berkström, Wennerström and Bergström, 2021; Halvorsen *et al.*, 2021). For networks to be successful, species movements and migrations, connectivity, gene flow, and larval dispersal should be considered and it is critical to have a clearly defined objective for the management area.

The Kåvra NTZ shows fast, strong and prolonged effects on the abundance and size structure of lobsters with a negative effect on brown crab, most likely due to competition between the species. Other effects on the species assemblage have not been observed. The use of this small NTZ furthers biological knowledge on the focal species, as well as provides baseline information on unfished populations, in addition to the effectiveness of NTZs as a conservation method. The use of small NTZs may have genetic effects which help the European lobster avoid fisheries induced evolution. However, to establish population wide effects and sustainable management of lobster in the region, and other targeted species, single small area NTZs alone are insufficient and warrant general restrictions in the fishery of the species.

13.5. References

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