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Abundances of breeding seabirds as indicators of the environmental state in the Baltic Sea – challenges and alternative approaches

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A B S T R A C T

Policy-relevant environmental indicators are needed to support effective management of marine environments. Here, we present common challenges in selecting a set of seabird indicator species, whose abundances are used as indicators, and propose potential solutions. We use examples from the Baltic Sea and emphasize the requirements of the EU's Marine Strategy Framework Directive (MSFD). Processes that confound indicators can be classified as 1) contradictory indirect anthropogenic effects, e.g. poor state of Baltic cod stocks increasing auk abundances, 2) drivers affecting seabird abundance but not the state of the sea, e.g. decreased availability of dumps for gulls, and 3) transient dynamics and species interactions, e.g. the steep population growth of Great Cormorant and White-tailed Sea Eagle, and the resulting predation impact of the latter on Common Eider. To assess the spatial scale of seabird population dynamics, we compared population trends of the Baltic Sea with the North Sea, and trends of the Finnish coast with inland lakes. On average, trends in the Baltic Sea and Finnish coast were more stable or positive compared to the North Sea and inland, respectively. Finally, we propose two phases of indicator formation: i) a filtering phase – in which unwanted variation is filtered out from the species' population trajectories – and ii) an indicator-generating phase – which aggregates relevant multi-species patterns into one or several indicators. We suggest statistical methodological approaches that cover both phases, separately or simultaneously. At best, seabird abundance indicators are useful summary statistics for monitoring joint effects of a wide range of anthropogenic pressures.

1. Introduction

In the field of conservation ecology, ecological indicators provide useful tools for simplifying the monitoring of multidimensional ecological patterns and processes, particularly those that are challenging to measure directly due to their complexity. In addition to properly describing the ecological processes of concern, a successful indicator enhances communication between involved partners and improves transparency in the management of a complex system (Durant et al., 2009). There is a growing need for policy-relevant indicators to support a knowledge-based approach to ecosystem-level management of marine environments and resources.

Seabirds, i.e. bird species breeding near the sea and using food resources from the sea, have frequently been selected as indicators of the marine environment around the world (e.g., Piatt et al., 2007). Their position at or near the apex of most aquatic food chains makes them ideal sentinel organisms for monitoring changes within marine ecosystems (Ogden et al., 2014). Variations in lower trophic levels or in the physico-chemical environment are likely reflected in seabird populations (Parsons et al., 2008). Furthermore, bird populations experience various human pressures, such as overexploitation of their food resources, pollution, and by-catch mortality due to fishing practices. They are internationally considered important and have high resonance with the public and policymakers (Parsons et al., 2008). At the same time, using seabirds as indicators of ecosystem change and health is associated with many problems and challenges (see critical review by Durant et al., 2009). For example, the selection of suitable indicator species is a challenging but important step in the formation of indicators, but the justifications vary a lot (Siddig et al., 2016).

In Europe, the European Union's (EU) Marine Strategy Framework Directive (MSFD), launched in 2008, establishes an obligation to include the abundance of seabirds as a component in monitoring the environmental status of European sea areas. Even before the MSFD, the abundance of seabirds was systematically used as an indicator in the North Sea (e.g., Parsons et al., 2008). In the Baltic Sea, abundance indicators of both wintering and breeding seabirds have been used for biodiversity

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assessments conducted by the Baltic Marine Protection Commission, HELCOM (HELCOM, 2018a). EU member states use these assessments for national reporting of the MSFD. The breeding seabird indicator is based on the abundances of 29 species, categorized into six groups based on their feeding strategies (grazing feeders, benthic feeders, pelagic feeders, surface feeders, and wading feeders) as required by the MSFD (Commission Decision 2017/848). The indicator 'Abundance of waterbirds in the breeding season' evaluates the status by comparing an abundance index during the assessment period with a modern baseline (1991–2000). The indicators signify good status when at least 75 % of the assessed species exhibit deviations of less than 30 % from the baseline (20 % for species laying only one egg per year) (HELCOM, 2018a). This threshold concept follows the same concept as the OSPAR Indicator 'Marine bird abundance' (ICES, 2008; OSPAR, 2012). In general, this approach offers a straightforward assessment of the state of seabird communities.

In the MSFD the definition of a good environmental status (GES) is closely linked to anthropogenic pressures listed in Annex III of the directive, all of which have direct effects on the marine region of interest. While climate change is acknowledged, it is not among the pressures of main interest. The abundance of seabirds should be monitored as one component of Descriptor 1, with GES criteria stating that "The population abundance of the species is not adversely affected due to anthropogenic pressures, such that its long-term viability is ensured". Therefore, monitoring and indicators should strongly focus on the environmental status of the sea while considering the pressures outlined in Annex III. The straightforward abundance indicators of seabirds, as used by HELCOM in the Baltic Sea (HELCOM, 2018a), primarily measure changes in biodiversity alone and the perception of the environmental status, as intended by the MSFD, is likely weakened by additional noise from various other non-targeted drivers, as acknowledged in a recent report (HELCOM, 2023).

In addition to the anthropogenic pressures on the sea that locally or regionally affect seabird populations – i.e., the above defined primary MSFD-targets of seabird indicators – a range of other extrinsic and intrinsic drivers, largely independent of the sea's current state, often underlie the dynamics of bird populations and may be regarded as confounding factors. Examples from the Baltic Sea are presented in Subsections 2.2–2.3 for anthropogenic extrinsic drivers, and in Subsection 2.4 for intrinsic drivers and species interactions. Anthropogenic extrinsic pressures that occur on very large spatial scales or outside the focal area, e.g., on the wintering grounds of birds breeding in the Baltic Sea, or in the arctic breeding grounds for birds wintering in the Baltic Sea, may not be of interest in seabird indicators. Further, climate change alters species' breeding and wintering distributions, and locally affects population trends regardless of the state of the sea (Lehikoinen et al., 2013). Species that have recently colonised the Baltic Sea or have recovered from a depression in population size may show strong and consistent population growth, even if the underlying reason for such trends was, for example, a history of intense persecution. Additionally, interspecific interactions between seabirds, such as predator–prey and competitive relationships, can be important ecological drivers of population dynamics. It remains debatable to what extent the effects of such species interactions should influence policy relevant indicators. However, acknowledging that most species' populations are affected by multiple drivers, it may be suboptimal to routinely disregard potential indicator species that are affected by non-targeted drivers.

In this article – using seabirds in the Baltic Sea as a study system – we aim to highlight challenges in using population abundances as indicators and discuss how those issues can be overcome. We review various drivers influencing the dynamics of seabird populations in the Baltic Sea. We classify these drivers into two categories, indicating whether they are processes or pressures that we would typically want to be reflected in seabird indicators, e.g. for MSFD targets, or whether they merely introduce bias or noise into the indicators. We also list species potentially affected by these drivers and present numerical examples based on seabird breeding abundance data from the Baltic Sea, mostly from the Finnish coast. To evaluate the suitability of different species as indicator species in the Baltic Sea, we compare i) temporal trends in the abundances of seabirds in the Baltic Sea with those of the North Sea, and ii) trends of archipelago birds along the Finnish coast with those of inland populations. Finally, we discuss principles and potentially useful statistical methods for constructing and assessing seabird population indicators, or for assessing single species population trends, considering the issues raised in this article. Under the MSFD, climate change is considered an ecosystem driver outside the scope of management. In line with this, we give less attention to climate change in this article.

2. Drivers of seabird populations

In the following Subsections (2.1–2.4), we present a set of drivers that influence seabird populations, categorize them and discuss their potential roles in seabird indicators. Subsection 2.1 offers a brief overview of typical anthropogenic pressures with (local) effects on seabird populations through the state of the sea, i.e., pressures on which the MSFD indicators should be focused. Subsection 2.2 presents an example of a fisheries-induced regime shift, which may or may not be considered a target for MSFD indicators, depending on the chosen baseline. Subsection 2.3 describes an example of population drivers not related to the state of the sea, and Subsection 2.4 outlines cases of intrinsic community drivers, such as transient population dynamics and interspecific interactions between focal community members. We suggest that these population trajectories may still be informative in the construction of seabird indicators, but some processes or trends need to be prefiltered or modelled out from the resulting indicator.

2.1. Pressures with direct effects on local bird populations

The Baltic Sea, a semi-enclosed sea area, is significantly impacted by various human activities, affecting both waterbirds (shorebirds, ducks, geese and swans, divers, and grebes) and distinctively marine birds (cormorants, skuas, gulls, terns, and auks). Incidental bird mortality in fishing gear is a phenomenon that has been recorded across all Baltic Sea countries. Žydelis et al. (2009) identified over 20 unique studies reporting bird bycatch in fishing nets in the Baltic Sea and suggested that at least 76,000 birds are being killed annually. It is, however, obvious that decreased abundance of diving birds (e.g., Skov et al., 2011; HELCOM, 2018a) and a Baltic Sea wide ban on the use of drift nets in salmon fishery, established in 2008, have mitigated the bycatch problem. According to Larsen et al. (2021), the average annual bird bycatch of the Danish commercial gillnet fleet in the western Baltic Sea during 2010–2016 was 3,249 bycatches.

Sea-duck hunting traditions, particularly strong in Denmark, Sweden, and Finland, inevitably cause hunting mortality, with the Common Eider (*Somateria mollissima*, hereafter Eider) facing the highest toll. During 2011–2016, the average annual bag of Eider in these Fennoscandian countries was around 38,000 (HELCOM). The sea-duck hunting bags have decreased during the last decades, which is generally due to game management in the respective countries in response to observed local population declines, or application of a precautionary principle when the population status has been unclear (Bregnballe et al., 2006; Noer et al., 2009).

Non-native mammalian predators, notably the American mink (*Mustela vison*) and raccoon dog (*Nyctereutes procyonoides*) can cause severe losses for birds, particularly in the outer archipelago and on small islands. Minks have been one of the primary factors responsible for the decline of Black-headed Gulls (*Chroicocephalus ridibundus*) in Latvia (e.g., Viksne and Janaus, 1993). Mink control has also led to increased breeding populations of a large range of seabird species in the Baltic Sea Archipelago (Nordström et al., 2003; Nordström and Korpimäki, 2004; Banks et al., 2008).

Hazardous substances, such as DDT, PCBs and mercury, nearly caused the disappearance of White-tailed Sea Eagle (*Haliaeetus albicilla*) populations from the Baltic Sea. Reproductive success improved following the ban of DDT and other pesticides in the early 1970s and returned to normal levels in the mid-1990s (Helander et al., 2002), leading to population recovery and increased densities in many areas. According to HELCOM (2018b), the Baltic Sea remained heavily impacted by hazardous substances during 2011–2016, but there were signs of improvement. Oil spills and oil pollution remain a threat to waterbirds, although the scale of this threat is poorly understood. Larsson and Tydén (2005) investigated oil residues on Long-tailed Ducks (*Clangula hyemalis*) caught in fishing gear on Hoburgs Bank in the Baltic Sea, revealing that 11.8 % of 998 analysed birds had oil on their feathers.

The increase in leisure boating and subsequent increase in disturbance has affected seabird populations (e.g., Mikola et al., 1994), although it has been shown that well-organized tourism can also protect bird colonies from harmful effects of the White-tailed Sea Eagle (Hentati-Sundberg et al., 2021). It is likely that the increasing construction of marine wind-energy will have an impact on certain seabird species in the future.

These pressures – bycatch, hunting, invasive species, pollutants, and disturbance – directly reduce seabird abundances. However, their variable detection in current MSFD indicators highlights the need for refined methods.

2.2. Example of indirect anthropogenic effects: baltic cod, sprat and auks

Cod (*Gadus morhua*) predation regulates sprat (*Sprattus sprattus*) and herring (*Clupea harengus*) – key prey for auks like Common Guillemots (*Uria aalge*) and Razorbills (*Alca torda*). Overfishing disrupted this balance, with cascading effects. In the Baltic Sea, sprat and herring are the primary target species for fisheries and also serve as key food sources for the Common Guillemot and Razorbill (e.g., Kadin et al., 2012; Lyngs, 2001). Sprat, due to its higher fat content, is a more valuable species in bird diets. Thus, the abundance of Common Guillemot and Razorbill, integral elements of the Baltic Sea marine ecosystem, should be essential indicators of the ecosystem's responses to fishery activities. Fishing has commonly been regarded as a potential pressure that reduces the available food for seabirds and thus restricts population abundance (e.g., Cury et al., 2011). However, severe overfishing in the 1980′s, combined with eutrophication and warming climate, caused a collapse of cod stocks in the Baltic Sea. This led to a regime shift (Möllmann et al., 2008; Lindegren et al., 2010) where abundance of sprat increased dramatically during the 1990s following the collapse of its main predator, cod. The annual commercial catches of sprat from the Baltic Sea rose from less than 100 Mkg in 1980–1990 to 300 Mkg in 2010–2022. Despite the increased catches, the abundance of sprat strongly increased at the beginning of the 1990s (Fig. 1).

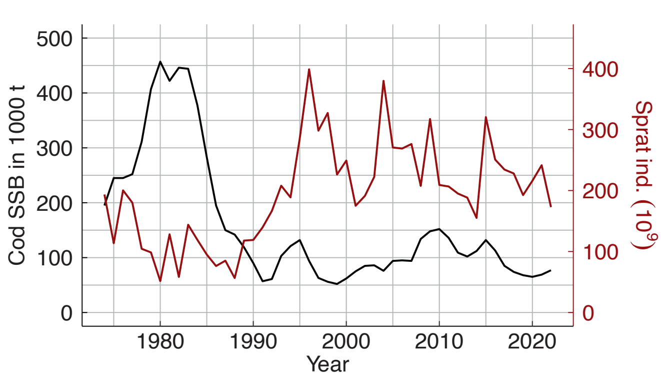

Fig. 1. Spawning stock biomass (SSB) of the eastern Baltic Sea cod stock and number of one-year old and older sprat in the Baltic Sea during 1974–2022 (ICES 2023a, b).

The island of Stora Karlsö, west of Gotland (57°17′ N, 17°58′ E) hosts the largest colonies of fish-eating seabirds in the Baltic Sea. It is estimated that 70 % of the Common Guillemots and 30 % of the Razorbills of the Baltic Sea are breeding there (Olsson and Hentati-Sundberg, 2017). The number of Common Guillemot and Razorbill breeding pairs at this breeding site has increased for 40 years, from roughly 6,000 and 1,000 pairs in the 1970s to 17,000 and 12,300 pairs in 2014–2016 (Olsson and Hentati-Sundberg, 2017). This growth has continued (HELCOM, 2023). Competition for breeding ledges in Stora Karlsö has probably led to increased dispersal of young Common Guillemots, which are forced to settle in other breeding sites (Olsson and Hentati-Sundberg, 2017). Satellite colonies exist in the northern Baltic Sea (e.g., in the Gulf of Finland and in the Stockholm archipelago) receiving regular replacements from the productive main colonies (Hario, 1982; Staav, 2009). Good availability of food, mostly sprat, has been an important reason for the growth of Common Guillemot and Razorbill populations in the Baltic Sea from the beginning of the 1990s (e.g., Österblom et al., 2006; Hjernquist and Hjernquist, 2010), possibly supported by decreasing concentrations of environmental contaminants and oil spills and a prohibition of driftnet fishing (Hentati-Sundberg and Olsson, 2016).

While the HELCOM-indicator "Abundance of waterbirds in the breeding season" of the Baltic Sea uses the period 1991–2000 as a reference period for 'good status' (HELCOM, 2023), the sprat abundance had already significantly increased during that period due to the regime shift. The cod fishery in the Baltic Sea is now heavily reduced to support the recovery of cod and to achieve a good status of the stocks, but there are no signs of recovery in the eastern cod stock (ICES, 2023a). There are valid reasons to assume that a potential recovery of cod, followed by a decrease in sprat abundance, would negatively impact Baltic Sea Guillemot and Razorbill populations. Model scenarios by Kadin et al. (2019) suggest this, and similar effects would likely be observed in the satellite colonies of Common Guillemot in the northern Baltic Sea, outside the main range of sprat.

From the perspective of bird indicators, achieving a good status for cod stocks would necessitate re-evaluating reference values or the entire structure of the indicators. In a scenario where cod stocks recover, it would be justified to filter out the effect of cod from the indicator, e.g., by detrending the auk trajectories or by including cod stock as a covariate in the model (see Section 4.). This cod-auks link serves as a good example of complexities and contradictions in simplified indicator approaches used in the Baltic Sea.

2.3. Example of drivers unrelated to the state of the sea: the case of Herring Gulls

"Rubbish" from dumps as well as fish waste have been important food sources for gulls, especially for the Herring Gull (*Larus argentatus*). Across its distribution area, these food sources constitute over half of their diet during both breeding and wintering seasons (Cramp and Simmons, 1983). According to Bergman (1939), Herring Gulls along the Finnish coast feed mainly on fish waste and rubbish, although some individuals specialize in consuming juvenile ducks. However, the implementation of two EU Directives as well as recent changes in coastal fishery have potentially reduced the availability of fish waste and other biowaste for gulls, especially the Herring Gull.

Directive 1999/31/EC (on the landfill of waste) set several goals, including development of waste treatment processes, increased recycling and recovery of waste and reduction in waste intended for landfills. Consequently, the amount of waste ending up in landfills has strongly decreased. In most EU countries, less than 1 % of municipal waste is nowadays disposed in landfills (Korhonen et al., 2018). This trend is also evident in Finland (Fig. 2).

Ämmässuo eco-industrial centre in Espoo, southern Finland (Fig. 3), is the largest waste treatment plant (formerly landfill) in Finland. It is located less than seven kilometres from the closest bay area. It is

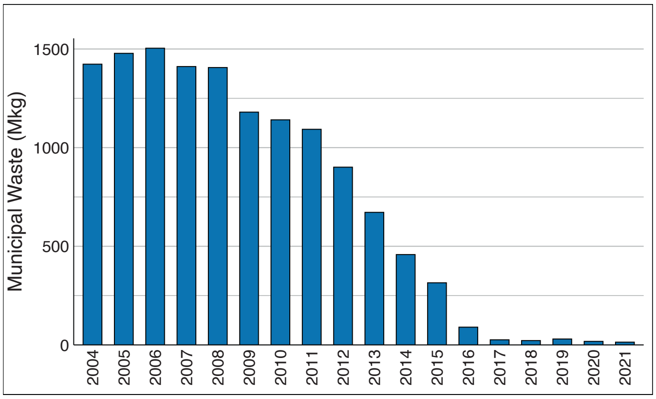

Fig. 2. The amount of municipal waste disposed to landfills in Finland during 2004–2021. Official Statistics of Finland.

operated by the Helsinki Region Environmental Services Authority (HSY) and they have exceptionally good statistics on waste treatment and also of the birds visiting the area. These statistics offer perfect illustration of the changes in former landfills and their likely link with the numbers of feeding birds pinpointing the potential impact of this driver to bird species included in indicators (Fig. 4).

In the Baltic Sea fisheries, approximately 10 % of the total commercial catch was discarded during 2000–2007, primarily including sprat, Baltic herring and Atlantic cod (Zeller et al., 2011). A majority of the discard has been ‘boat-based’ resulting from fishers’ actions. The estimates also included ‘underwater’ discards of herring and sprat caused by mortality of contacts with the deployed gear, and ‘ghost-fishing’ resulting from lost or abandoned gear (Zeller et al., 2011). The discarded bycatch consisting e.g. of juvenile fish, species of low interest or injured individuals has always formed a buffet for the ship-following gulls. Evidence suggests that discarding has earlier led to increases in some seabird populations, e.g., in the North Sea (e.g., Thompson, 2006). The scavenging of discards and offal has been a widespread phenomenon also in the Baltic Sea, with the Herring Gull as the dominating species (Garthe and Scherp, 2003).

The EU discard ban, or “the landing obligation”, was introduced in 2015 and fully implemented in 2019. This obligation requires fishermen to land all quota-regulated fish, including undersized fish. In the Baltic Sea, the landing obligation is applicable to species such as herring, sprat and cod. While this is effective on ‘boat-based’ discarding, it does not affect ‘underwater discard’. Several studies suggest that this obligation may create food shortage for scavenging birds and may have direct effects on certain seabird communities (e.g., Bicknell et al., 2013; Heath et al., 2014). While the exact consequences of this measure in Europe are still unknown (De la Cruz et al., 2023) and regional studies are not reported from the Baltic Sea, it is likely that the ban has led to reduced food availability for scavenging seabirds there as well.

Coastal small-scale fishery with gillnets has offered an almost year-round food source for gulls around the Baltic Sea. Bycatch and offal have often been fed to gulls which have been regarded as a local scavenging service. The gillnet fishery has, however, collapsed especially in the northern Baltic Sea, which is demonstrated by the trends in gillnet catches in the Finnish coast (Fig. 5). The increased abundance of seals (HELCOM, 2018c) has made the gillnet fishery difficult or almost impossible in many coastal areas and the profitability of small-scale fishery has decreased. This effect of collapse of gillnet fishery on scavenging seabirds has obviously been parallel with the effects of the landing obligation.

In summary, changes in landfill practices and various fishing practices may be primary drivers of gull population declines, influencing breeding bird indicators designed to monitor the ecological state of sea areas. While not aiming to explicitly confirm a direct causal relationship, we illustrate that all the mentioned changes in food supply coincide with the decrease in Herring Gull breeding populations in the western Gulf of Finland, where robust data are available (Fig. 6). We suggest that neither

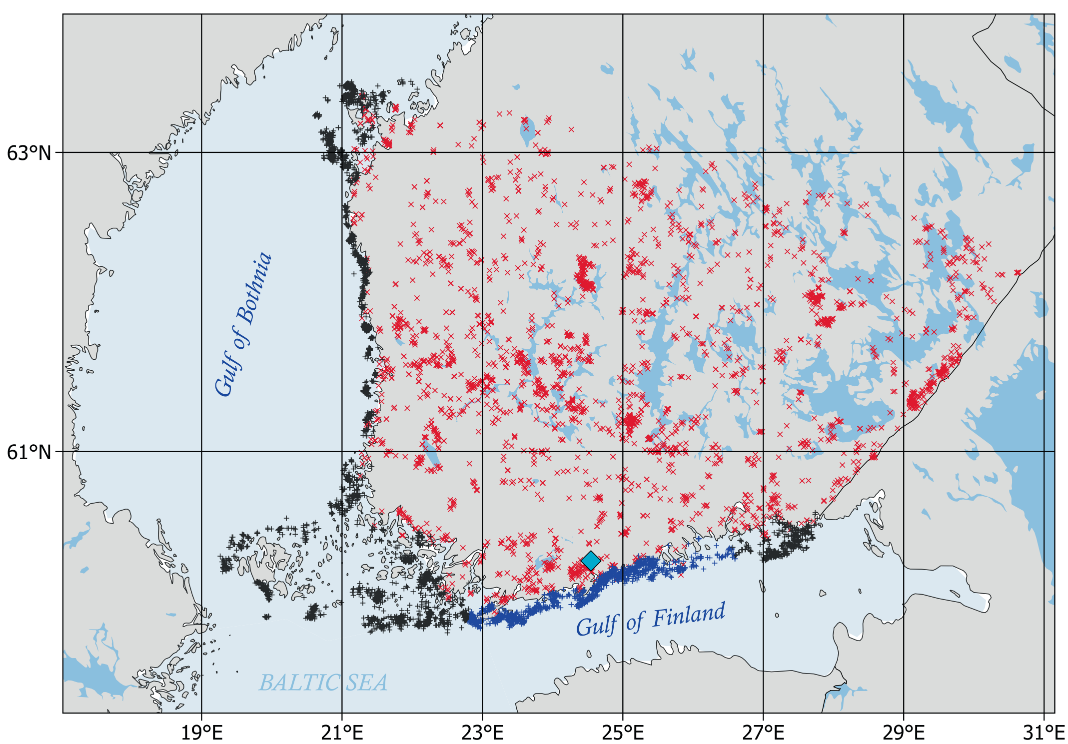

Fig. 3. Map showing bird monitoring sites used in the analyses. Red ’x’ symbols represent the monitoring sites for inland lake breeding waterfowl survey, black and blue crosses along the coastline are sites used for monitoring archipelago birds. The blue ones represent the subsample used in Herring gull trend analysis (Subsection 2.3). Light blue diamond shows the location of Ämmässuo landfill. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

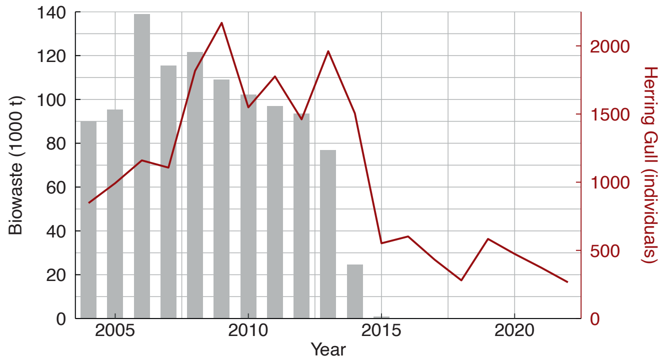

Fig. 4. The annual amount of disposed biowaste (1,000 tonnes) and average numbers of surveyed Herring Gulls (bird individuals) during middle of March to middle of July in the Ämmässuo landfill. During the years 2015–2022, the amount of biowaste decreased gradually from 790 tonnes to zero. The biowaste data are from HSY (waste management of Helsinki metropolitan area) and the gull data are from Tringa r.y. – the Helsinki region ornithological Society.

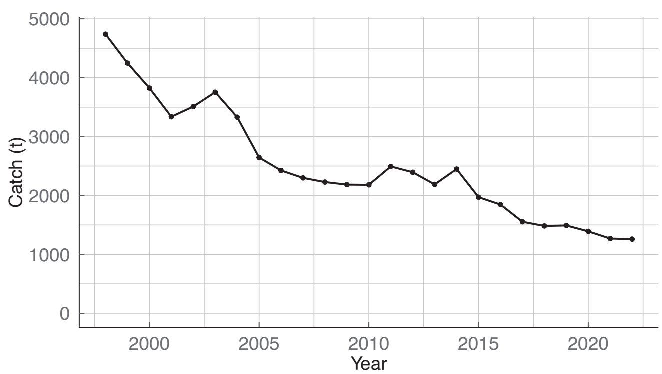

Fig. 5. The total catch (all species) in commercial gillnet fishery in the Finnish coast during 1998–2021.

of these processes – even if human-induced – are directly linked to the ecological state of the sea. They are not really what we aim to measure with the sea bird indicators.

2.4. Transient dynamics and species interactions

While certain population trends are undeniably driven by anthropogenic pressures directly related to the state of the sea, many

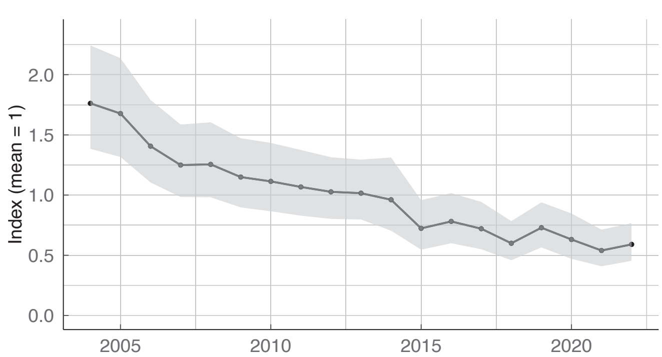

Fig. 6. The abundance of breeding Herring Gull (Larus argentatus) along the Finnish coast of the western Gulf of Finland (in the region of Uusimaa, see Fig. 3), shows a strong and consistent declining trend. The analysis is based on the Finnish archipelago breeding bird survey data (Below et al., 2019; Seimola et al., 2025).

population changes result from interspecific interactions or transient community dynamics. For example, when the population size of an effective predator changes, we logically expect to see top-down effects in the populations of its prey species, and potentially effects on its competitors as well. Similarly, population changes at lower trophic levels may have bottom-up effects on their predators, creating indirect effects higher in the food chain. Species that have recently established or show recovery from previously very low population numbers may display long periods of transient dynamics, such as substantial population growth before reaching carrying capacity.

Under the influences of climate change, many bird species are shifting their distribution ranges northward (Välimäki et al., 2016), potentially introducing new species to the community or making the entire community more similar to what it was previously further south. Transient dynamics or shifts in community composition may strongly impact the trend of a multispecies indicator, even when targeted pressures – or more generally the state of the sea *per se* – has remained constant.

It is important to acknowledge that nearly all processes in nature are to some extent influenced by humans. Therefore, classifying anything as purely "natural" is neither impossible nor desirable. Below in Subsections 2.4.1 and 2.4.2 we discuss examples that may be considered as typical ecological background processes which we recommend filtering out when forming an indicator of the state of the sea. In Section 4, we discuss analytical methods that can be applied to filter out these processes, reducing bias and noise while still being able to include relevant species into the indicator.

2.4.1. The steep population growth of Great Cormorant and White-tailed Sea Eagle

The Great Cormorant (*Phalacrocorax carbo*) population in the Baltic Sea has shown a strong increase since it re-established in the early 1990s. The piscivorous species was eradicated at the beginning of the 20^{th} century and cannot be classified as either an invasive species or a newcomer (Beike, 2014). In Finland, the first new breeding was recorded in 1996 and the population rapidly grew to ca. 17,000 breeding pairs in 2012 (Rusanen, 2014), reaching ca. 25,000 breeding pairs in 2015 (https://www.ymparisto.fi). After this, the Finnish population has shown slight decreases and fluctuations, suggesting that the population is approximately saturated. Cormorants are generalist predators, adapting their diet and feeding areas according to the availability of the prey species (Salmi et al., 2015). In the northern Baltic Sea, common prey species include, e.g., eelpout (*Zoarces viviparus*), roach (*Rutilus rutilus*), perch (*Perca fluviatilis*), sprat, and herring (Lehikoinen et al., 2011; Salmi et al., 2015; van Eerden et al., 2022). Eutrophication has probably promoted the population growth of coastal fish in the northern Baltic Sea, ensuring abundant food resources for the growing cormorant population. It is, however, likely that the lack of (peaceful) breeding sites has recently started to restrict the population growth, more so than the availability of food resources. Cormorants are not very well-liked among local fishermen and in addition to permissible obstruction of nesting in certain islands, there have been some illegal actions along the Finnish coast. However, van Eerden et al. (2022) suggested that the Great Cormorant would be a good indicator species for the state of the sea.

The White-tailed Sea Eagle population faced near-extinction in the 1950s and 1960s due to the adverse effects of pollutants in the Baltic Sea (Herrmann et al., 2011). Conservation efforts, including legal protection and a sustained supply of uncontaminated food, resulted in the recovery of this species (Fig. 7). Currently, White-tailed Sea Eagles are abundant with a saturated population along the Finnish coast. The species' vulnerability to pollutants accumulating at the top of the food pyramid makes it an attractive inclusion in indicators aiming to describe the state of the sea. As for the Great Cormorant, the steep population growth cannot solely be attributed to a continuously improving state of the environment. Instead, once the environmental conditions allowed

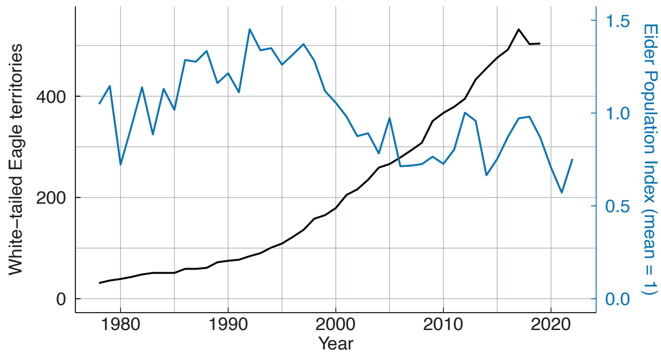

Fig. 7. The black line shows the number of occupied territories of the White-tailed Sea Eagle in Finland in 1978–2019. These data are reproduced from Stjernberg et al. (2016) for years 1990–2009, and from Högmander et al. (2020) for years 2010–2019. The blue line presents the population abundance trajectory of common eider (Somateria mollissima) along the coast of Southern Finland 1978–2022. There is a positive trend from the early 1980s to the mid 1990s, followed by a decline. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

population recovery, a long transient period was seen before the population saturation. This resurgence, however, has had significant repercussions for other nesting seabirds, notably the Eider, as detailed in the next Subsection (2.4.2).

Both the Great Cormorant and White-tailed Sea Eagle would be good candidate species to be included in an ecological indicator of seabirds. However, we suggest that incorporating their current population time-series in indicators assessing the state of the sea, would require modelling and filtering out the transient phase of strong population growth so that the trend itself does not affect the indicator. Naturally, if the indicator is initiated after the apparent saturation of these populations, the historical strong population growth is no longer an acute problem.

2.4.2. Baltic seabirds under the influence of the White-tailed Sea Eagle and Herring Gull

In this subsection, we describe examples of interspecific interactions, where seabird populations that may be suitable for indicators are likely affecting each other's population abundances. Hence, these changes are at least partially attributed to other factors than direct consequences of the Baltic Sea's state. The first example highlights how the resurgence of the White-tailed Sea Eagle has affected other seabird species, particularly the Eider, which has shown a rapid population decline in Finland during the last decades. The second, related example examines cascading effects of changes in Herring Gull population size and its foraging behaviour on the Eider, Lesser Black-backed ("Baltic") Gull (*Larus fuscus fuscus*) and other seabirds.

The estimated population trajectory of eiders showed an increase from the early 1980s to the mid-1990s, followed by a steep decline, which has become somewhat less steep after 2005. Many of the problems facing Eiders have been attributed to the population growth of White-tailed Sea Eagles (Fig. 7). Firstly, predation has led to higher female mortality rates and nesting disturbances among Eiders, naturally contributing to the population decline. The sex ratio is strongly male biased (with ca. 65 % males), and demographic modelling confirms that this pattern can be explained by the empirical sex-specific differences in adult survival (Ramula et al., 2018). Predation pressure, primarily driven by White-tailed Sea Eagles but also influenced by invasive alien predators American Minks and Raccoon Dogs, is more pronounced on open, exposed outer islets where Eiders historically bred abundantly (Hermansson et al., 2023; Jaatinen et al., 2022). Secondly, because of increased White-tailed Sea Eagle predation, Eiders have shifted their core nesting areas from the outer archipelago further in, to more sheltered and forested islands (Kurvinen et al., 2016). The habitat shift to the inner archipelago areas is likely driven by lower female mortality, better nesting success and higher recruitment of new breeders in the inner archipelago due to lower predation risk compared to the open outer islets (Öst et al., 2011; Ekroos et al., 2012). Moreover, human settlement in the inner archipelago with more summer cottages may provide protection against White-tailed Sea Eagle predation. Eiders have historically preferred inhabited islands that also have had a higher human population density, in outer archipelago as well. Such habitat shifts can potentially obscure the population trend included in an ecological indicator.

In Subsection 2.3 we described the processes behind the decline of Herring Gulls – another dominant species in the Baltic Sea archipelago. The Herring Gull has experienced a large change in food resources due to closure of rubbish dumps (Directive 1999/31/EC on the landfill of waste; Hario and Rintala, 2016), which has likely caused an immediate increase in predation pressure on nesting seabirds. Although the Herring Gull population is currently diminishing, it certainly has induced predation pressure on Eider ducklings and other seabirds, especially when its population was larger right after the implementation of the Waste Landfill directive. This phenomenon seems even more complicated: when the Baltic Herring Gull population was large and received substantial food resources from dumps and discards, their large breeding colonies offered protection for other breeding birds against White-tailed Sea Eagles. Consequently, the diminished Herring Gull population may no longer offer as much protection for other breeding birds, and simultaneously Herring Gulls consume more of their natural food, i.e., eggs and chicks of other birds, such as Eider and Lesser Black-backed Gull (Hario, 1994), the latter being an example of inter-guild predation (Polis and Holt, 1992).

The examples discussed here emphasise the importance of considering ecological interactions such as predation when using seabird populations as ecological indicators. The complex interplay between White-tailed Sea Eagle resurgence, Herring Gull population and food source changes, their predation, and habitat shifts demonstrates that a holistic understanding of population dynamics is necessary for a critical interpretation of ecological indicators. Achieving a "good state" for one species might lead to a "poor state" for another. In the estimation of multi-species indicators on seabirds, we suggest that modelling known interspecific interactions and isolating them from the resulting indicator (removing their effect from the outcome), would improve the usefulness and interpretability of the indicator.

3. Spatial comparison of population dynamical patterns

A good candidate species to be included in a local population abundance-based indicator, should typically show population dynamics primarily influenced by local pressures and drivers related to the state of the sea. Moreover, the spatial scale of the demographic processes would optimally align closely with the area that the indicator concerns and should neither be too large nor too fine-grained. This requirement is difficult to evaluate, but the degree of population synchrony between adjacent subpopulations can provide a cue. A distinct subpopulation actively driven by local pressures is likely to show its own uncorrelated dynamics or temporal trends that differ from neighbouring subpopulations. For example, a high degree of dispersal, effects of large-scale weather and climate change will alter dynamics over larger areas and induce synchronous dynamics. In migratory birds, synchronous population patterns can also be present in separate breeding subpopulations if they are subject to the same pressures on wintering grounds or along the migration route, such as extensive hunting during wintering or passage.

Interpreting spatial population synchrony between a focal (sub) population and neighbouring populations is complicated by the fact that the pressures themselves may be spatially autocorrelated, resulting in similar dynamics across large areas and multiple subpopulations. Additionally, neighbouring populations can be influenced by factors that are not relevant to the focal population, resulting in distinct dynamics. For example, a non-migratory population may show different dynamics compared to a migratory population, whose dynamics may be driven by pressures acting outside the breeding season.

Despite the outlined limitations, we assess the spatial scale of population dynamics, and hence, the species' potential to be included in indicators for the environmental state of the Baltic Sea. This assessment involves species-wise comparisons of medium-term temporal trends (ca. 20 years) in the following two Subsections (3.1 and 3.2):

i. Trends of seabirds breeding in the Baltic Sea, compared with the corresponding trends in the North Sea.
ii. Trends of waterbirds breeding in the archipelago along the southern–southwestern coast of Finland (Gulf of Finland, Archipelago Sea and Gulf of Bothnia), compared with the corresponding trends of the same species breeding at inland lakes in southern Finland (see study sites compared in Fig. 3).

The results of these comparisons provide a starting point for the discussion on potential drivers and the suitability of species as indicators. The rationale for comparing medium-term trends, of approximately 20 years, is motivated by the trade-off between sample size and the feasibility of describing approximative subpopulation dynamics with a simple log-linear (exponential) trend. In a short time-series, statistical uncertainty of the trend is considerable, while in a long time-series, the underlying trends are likely more complex, with several turning points. In medium-term trends, both problems are reduced, but can be present to some degree. The principles and methods for trend comparison are described in Supplement 1.

3.1. Comparison of population trends in the Baltic Sea and North Sea

Seabird populations in the Baltic Sea and North Sea regions represent a diverse array of species exhibiting a wide range of habitat preferences and feeding strategies. In addition, it is important to note that some species share common wintering areas, while others migrate to different regions during the winter months. These traits are essential for understanding the factors driving the population dynamics in both regions.

We compared the medium-term temporal trends spanning from 1991 to (2017–)2021 in breeding seabird populations between the Baltic Sea and North Sea (Greater North Sea, OSPAR region II). The trends were extracted from the most recent reports from both regions (HELCOM, 2023; OSPAR, 2022). The reference period for both regions was the average of 1991–2000. However, the assessment period differed slightly between the regions: for the Baltic region, the assessment period was the average of 2016–2021, while it ranged from 2012 to 2017 to 2014–2019 for the North Sea, depending on species (for detailed information, see HELCOM, 2023; OSPAR, 2022). We calculated the annual population trends (referred to as b_1 and b_2 in Supplement 1) as the natural logarithm of the population index during the assessment period divided by the number of years between the reference and assessment periods. The annual levels of population changes were used for species-wise comparisons (Fig. 8). There are 18 species that are reported in the Breeding Seabird Indicators in both regions. We classified species into five functional groups and determined, based on the Eurasian African Bird Migration Atlas (Spina et al., 2022) whether the birds from the two regions share the same wintering areas (Supplement 2).

We tested the mean difference in regional trends using a paired *t*-test. Additionally, we conducted a two-way ANOVA to determine whether the differences between regional trends (b_Δ; as defined in Supplement 1) could be explained by the factor variables *functional group* (levels: surface-feeding or pelagic) or *wintering area* (levels: shared or different). We here included only species from the largest functional groups, with known wintering areas for both regions, resulting in an analysis of 12 species. The reported coefficients of population change were not associated with any measures of uncertainty, so we used only their point estimates in our comparisons.

The trends of all 18 species included in both indicators were on average slightly more positive in the Baltic Sea than in the North Sea (paired *t*-test: $t = 2.33$, df $= 17$, $p = 0.033$). The population development was, on average, 2.0 % more positive in the Baltic Sea compared to the North Sea (Fig. 8). The two-way ANOVA revealed that neither functional group ($F_{1,\,9} = 0.497$, $p = 0.498$) nor wintering area ($F_{1,\,9} = 0.832$, $p = 0.385$) had a statistically significant effect on the observed differences in temporal trends. Nevertheless, there seem to be species-specific differences of relevant magnitude in e.g. Barnacle Goose (*Branta leucopsis*), Common Eider, Oystercatcher (*Haematopus ostralegus*), Common Guillemot (*Uria aalge*) and Great Cormorant.

Since the distinction between shared and separate wintering areas does not appear to account for the variations in regional trends, it is plausible that the processes underlying these population trends primarily occur within the breeding areas. On the other hand, some species that breed in the Baltic Sea area winter in the North Sea region, which

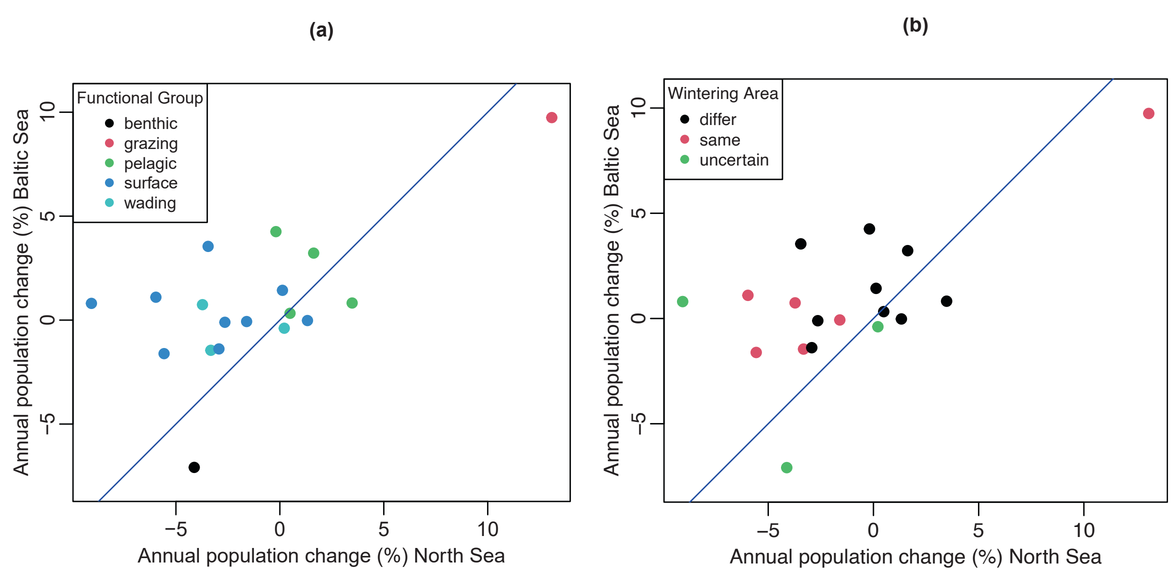

Fig. 8. Comparison of population trends of seabirds in the North Sea (x-axis) and the Baltic Sea (y-axis) among functional groups/feeding strategies (panel a) and similarity in wintering area (panel b). Trends that are similar in both areas would fall close to the blue line (y = x). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

may complicate direct comparison of the areas as the interplay between breeding and wintering grounds introduces uncertainty regarding the influence of breeding areas on the observed trends.

While two-way ANOVA revealed no effects of functional group in explaining the contrast between the two temporal trends, it is also evident that the number of species in this comparison was very small, and all functional groups were not represented in the comparison. Many benthic feeders common in the Baltic Sea (Velvet Scoter, Tufted Duck, Scaup), pelagic feeders (Goosander, Red-breasted Merganser and Great Crested Grebe), waders (Ruddy Turnstone and Dunlin) and grazers (Greylag Goose, Mute Swan) are not included in the North Sea indicator. Surface feeders, however, seem to reflect the processes present in breeding areas, e.g., the availability and abundance of small fish, and may be driving the main result that trends are more positive in the Baltic Sea (Fig. 8).

The North Sea is one of the areas experiencing a profound human impact on the ecosystem (Halpern et al., 2008) and is one of the most exploited fishing areas in the world (Daan et al., 2005). Seabird populations rely strongly on fish and are thus affected by the depletion of fish populations (Cury et al., 2011). Changes in fishery practices, including the EU discard ban, have probably altered feeding opportunities of seabird populations in both regions, especially for surface feeders. Pelagic feeders have possibly increased in the Baltic area due to the regime shift that has led to abundant sprat stock (see Subsection 2.2). The differing trends observed between regions may provide valuable insights, but should be interpreted with caution, especially because the drivers behind these trends are not explicitly known for most of the species investigated. In summary, the comparison between North Sea and Baltic Sea bird population trends does not provide straightforward recommendations to the selection process of indicator species, but the observed differences suggest that the spatial scale used for Baltic Sea indicators is informative.

3.2. Comparison of temporal trends in archipelago and inland subpopulations

In the Baltic Sea, several breeding bird species exclusively inhabit the archipelago but not inland lakes, some examples being the Eider, Razorbill, Common Guillemot, Black Guillemot (*Cepphus grylle*), Caspian Tern (*Hydroprogne caspia*) and Ruddy Turnstone. On the other hand, many common species breed both in the archipelago and in inland lakes, hence providing an opportunity for comparison of medium-term trends. Although many of these populations may be part of the same larger biological population, these species may show separate trends, offering valuable information on the proper spatial scale of indicators, and the potential suitability of these species as indicators of the state of the sea.

We were able to compare medium-term temporal population trends of six species of ducks and one grebe (Table 1) between the Finnish coastal archipelago from south of Vörå (approximately $< 63^\circ 30'$N; excluding Åland islands) and the southwestern parts of mainland Finland (Fig. 3). We used the Finnish archipelago breeding bird data and estimated the population indices of all the species according to statistical modelling principles described in Supplement 3 (also the data are described in Supplement 3). The time spans for the temporal trends were three generation lengths, also used e.g. by the IUCN and national Finnish red list (Hyvärinen et al., 2019), which for the pool of species analysed was on average 21.5 years. When choosing the length of the time-series, there is a trade-off between achieving sufficient statistical power (sample size is the number of years), and trying to capture the signal of ongoing processes, approximated locally with an exponential (log-linear) trend. The temporal population trends of seven species, estimated here for the archipelago, were compared with published temporal trends of the same species in inland lakes in the southern parts of Finland (Piha et al., 2023). The species-specific time spans used for the trend comparisons are matched. Trend comparisons were done according to the principles outlined in Supplement 1.

For four species, the temporal trends were estimated to be clearly more positive in the archipelago compared to inland lakes (Table 1), the difference corresponding to a population change of >100 % over three generations in the archipelago, under a scenario where the population trend is stable inland. These species included three ducks: Mallard (*Anas platyrhynchos*), Northern Shoveler (*Spatula clypeata*), Tufted Duck (*Aythya fuligula*); and one grebe: the Great Crested Grebe (*Podiceps cristatus*). For the rest of the species, i.e., the Common Goldeneye (*Bucephala clangula*) and the piscivore mergansers: Goosander (*Mergus merganser*) and Red-breasted Merganser (*Mergus serrator*), the differences were rather small, with 95 % confidence intervals including zero. Notably, in the Tufted Duck, population trends were clearly negative both in the archipelago and on the mainland, although the trend was considerably more positive in the archipelago.

For the species included in the comparison, the temporal trends in the archipelago were on average more positive than those of the mainland (paired t-test: $t = 3.18$, df $= 6$, p-value $= 0.019$). The average annual logarithmic growth rate in the archipelago was on average 0.0305 larger (95 % CI = 0.007, 0.054). To illustrate, if all the populations were on average stable on the mainland ($b_2 = 0$), this would imply on average a +93 % population growth in the archipelago over three generations (21.5 years).

The results suggest archipelago population trends may be influenced by different processes than those on the mainland, or by the same drivers operating at different intensity. While over-eutrophication poses a major threat to many species of waterfowl in general, in particular on the mainland (Pöysä et al., 2023; Holopainen et al., 2024), many of the analysed species may actually benefit from moderate eutrophication in shallow bays of the Baltic Sea by potentially enhancing food availability. The Tufted Duck is known to benefit from colonies of gulls and terns providing shelter against nest predators (Götmark, 1989; Väänänen, 2000), highlighting the potential impact of declines in gull and tern populations on some species. While we lack comparable information from inland lakes, differing temporal trends of gull and tern colonies at inland lakes and in the archipelago is another candidate explanation for this pattern.

Despite the unknown causes of differential trends, comparisons of trends in the archipelago and inland of Southern Finland clearly highlight that the drivers of population dynamics, or their effect sizes, differ between the two areas. Our results suggest that species such as Mallard,

Table 1
Comparison of temporal population trends of seven species in the Finnish archipelago and the mainland over three generations.

Scientific name	Time span legnth (years)	Time span (year)	Inland area	Archipelago	Difference in trends	Change/3 generations (%)
	3 gen.	Start–End	lnTrend (SE)	lnTrend (SE)	Estimate (SE)	Estimate (CI_{lo}, CI_{hi})
Anas platyrhynchos	19.8	2003–2022	0.0012 (0.0037)	0.0424 (0.0091)	0.0411 (0.0099)	123 (53, 225)
Spatula clypeata	19.5	2004–2022	–0.0499 (0.0070)	0.0169 (0.0172)	0.0668 (0.0185)	268 (81, 647)
Aythya fuligula	21.9	2001–2022	–0.0532 (0.0088)	–0.0159 (0.0047)	0.0373 (0.0099)	107 (42, 203)
Bucephala clangula	24.0	1999–2022	–0.0102 (0.0019)	0.0088 (0.0108)	0.0190 (0.0110)	45 (–5, 120)
Mergus merganser	21.9	2001–2022	0.0021 (0.0103)	0.0084 (0.0090)	0.0063 (0.0137)	13 (–33, 91)
Mergus serrator	21.9	2001–2022	0.0099 (0.0079)	0.0038 (0.0091)	–0.0060 (0.0120)	–11 (–44, 41)
Podiceps cristatus	21.3	2002–2022	–0.0181 (0.0042)	0.0311 (0.0196)	0.0492 (0.0200)	161 (21, 461)

Northern Shoveler, Tufted Duck and Great Crested Grebe are promising candidates for inclusion in seabird indicator. However we caution that some positive population responses in the archipelago may result from eutrophication, a process generally considered ecologically detrimental.

4. Statistical methods for developing indicators

A relevant initial step in developing a multi-species ecological indicator is to carefully select a set of suitable species with desirable traits for indicator species (Gregory et al., 2005; Fraixedas et al., 2020). Such traits are high sensitivity, rapid and straightforward numerical responses to the pressures of interest (as listed in Subsection 2.1), but otherwise stable dynamics with a low level of annual fluctuations. Good indicator species are also typically easily monitored and common enough, but not necessary occurring everywhere (Siddig et al., 2016; Fraixedas et al., 2020). Importantly, as discussed throughout this article, strong temporal trends and dynamical patterns which are unrelated to changes in the state of the focal sea area, are problematic as they can obscure the patterns of the resulting indicator.

Every species' population has its own dynamics, typically determined by a range of various known and unknown drivers. It may be possible to find reasons why most species may not serve as valid indicators, for example due to strong interspecific interactions, but drawing the limits is difficult. The approach presented here encourages liberal inclusion of many species by trying to control statistically for some of the most obvious patterns and undesired drivers. Examples discussed in this paper include changes in landfill practices and various fishing practices affecting gull populations (Subsection 2.3), the substantial transient population growth of Great Cormorant and White-tailed Sea Eagle (Subsection 2.4.1), and the predatory effects of White-tailed Sea Eagle and Herring Gull on other archipelago bird species (Subsection 2.4.2). Moreover, when selecting species for analyses, evidence of local and distinct population dynamics in the sea area of interest may provide an indication that the species' dynamics are representative of processes occurring in the focal sea area (Subsections 3.1 and 3.2).

In the strategy of analysis, we recognize two phases. First, the choice of species and possibly filtering out undesired variation from the single-species trajectory, is here referred to as the filtering phase (4.1). Secondly, the indicator is formed in the indicator-generating phase (4.2). Both phases can be executed separately, but ideally, they are merged into one hierarchical analysis (4.3), using models with multiple levels of organization, including separate species-specific population dynamics and common community level patterns. However, such models can be very complex and challenging to fit statistically.

4.1. The filtering phase

The purpose of the filtering phase is to generate a representative set of (possibly scaled) population indices for each species included in the analysis while statistically accounting for possible undesirable properties of the time-series. The inevitable step of selecting candidate species into the set of indicator species, can be regarded as an extreme part of the filtering phase – accepting either 0 % or 100 % of the variation of each species' population trajectory to affect the indicator – while some intermediate solution could better serve the purpose. As discussed in this paper, possible undesired properties of indicator species include transient dynamics, density dependent effects, cohort effects, the dynamics of interacting species, and, in brief, processes that are considered a normal or temporary part of the ecological system dynamics, rather than changes in the general quality of habitat or conditions. One way to think about the effects of habitat changes, or changes in the state of the environment, is to see it as alterations in the species' carrying capacities (see e.g., Ruokolainen et al., 2009).

For species showing strong temporal trends due to factors like recovery from a catastrophic event (such as White-tailed Sea Eagle), or recent (re)establishment in an area (e.g., Great Cormorant), various alternative filtering approaches can be considered to account for situations where the involved species' populations are temporally deviating from their carrying capacities. Examples of such approaches, all using log-scale abundance indices as the response variable, may involve:

1) Fitting a deterministic growth curve according to a density-dependent single-species population model, equivalent to an observation error growth model (e.g., logistic growth),
2) Fitting a linear autoregressive model on the log-scale, equivalent to a Gompertz-type density-dependent process error model, or.
3) Fitting a linear or non-linear temporal trend (e.g., with a GAM, or by including continuous variable 'year' as a covariate). However, notice that a pure log-linear trend corresponds to exponential growth, i.e., infinite carrying capacity, and is not recommended if the population is suspected to be saturated (getting close to its carrying capacity).

For each of the examples above, the model fit is filtered out as a separate step, leaving the residuals as the presumably informative outcome of the filtering phase (see Subsection 4.3 for further integrated methods). Approaches 1) and 3) are less likely to filter out rapid fluctuation in abundance but will allow them to contribute to the indicator. Presuming temporally autocorrelated residuals (e.g., using generalized least squares; GLS) may be recommended, not least to retain meaningful temporal structure in the residuals, i.e., the output for the next step. At simplest, these tasks may be done using linear regression, or log-link generalized linear models (GLM).

To account for interspecific interactions, the (optionally log-scale) population abundance of the interacting species from previous year can be included to explain the log-scale abundance of the focal species in the next year. This may be fitted as a multivariate regression type of community model for all species simultaneously (e.g., Ruokolainen et al., 2009). Abiotic drivers not desired to affect the indicator may also be included in a similar manner as covariates in the model. Examples of such interspecific interactions in the Baltic Sea include those between White-tailed Sea Eagle and most other species, as well as interactions between gulls and species that they predate upon or provide shelter for.

When raw data of a given species consist of multiple annual data points from different sampling sites, a time-series can be generated by treating "year" as a factor variable in the model (annual effects), accounting also for sampling site as another factor variable with a random or fixed effect on the intercept. This procedure can be combined with the filtering phase, by the inclusion of temporal covariates or effects (e.g., a trend) and annual random effects on the intercept (instead of fixed effects), which will represent the filtered output. In general, we encourage to filter time-series or multi-annual data whenever possible rather than skipping a good indicator species from the analysis. However, we encourage the analyst to be modestly restrictive in the filtering phase, to avoid removing meaningful variation. For example, allowing for autocorrelated residuals can be a good idea.

4.2. The indicator-generating phase

Once the population indices of a set of species have been extracted and, optionally, filtered from unwanted effects and properties (Subsection 4.1), a typical choice is to summarize the common dynamics of a set of species by calculating an unweighted geometric mean of their scaled population indices (Gregory et al., 2005). In this case, annual arithmetic means would be calculated from the scaled species-specific (log-scale) outputs of the filtering phase, and the results would be exponentiated to form a time-series of geometric means. Estimates of statistical uncertainty for the resulting indicator can be calculated using bootstrap or Monte Carlo methods (Soldaat et al., 2017), but it is also feasible to directly generate statistical estimates and associated uncertainties of an indicator compatible with the geometric mean estimator. One method involves multivariate regression with separate species' log-transformed time series as response variables, a separate intercept for every species and a factor variable 'year' as a fixed effect affecting all species similarly. The annual effects (exponentiated; possibly scaled to a baseline value), would be the indicator. This approach would simultaneously provide the annual estimated multi-species means and their uncertainties (Fraixedas et al., 2020).

Temporal trends and annual deviations in the indicator can be modelled simultaneously, by further developing the model described above into a multivariate mixed model (log-transformed species abundances as responses), where the trend is explicitly modelled as a fixed effect (function of time) and "year" as a factor variable is included as a random effect on the intercept – both affect all species equally. Some software, such as package "glmmTMB" in R (Brooks et al., 2017) allows for temporally autocorrelated random effects, which could be a desirable trait for this kind of model. Here, the annual indicator values would be the trend plus annual random effects, and that result exponentiated (possibly also scaled to a baseline value).

Dynamical factor analysis (DFA) is a modern approach for modelling common patterns and latent variables in time-series data and has been used in various contexts to produce indicators, in both econometrics and ecology (Zuur et al., 2003). DFA can be used to model classic state-indicators, but also to explicitly link indicators with known pressures of interest, resulting in pressure indicators. This approach presumes a dynamical model (typically a random walk) for the resulting indicators. Further, it differs from the unweighted geometric mean approach by estimating the role of the different species in forming the indicators (the factor loadings), and by typically forming a set of several indicators (dynamical factors) instead of one. Further, smoothed dynamical factor analysis is a tool particularly developed for detecting temporal trends in multivariate time-series (Ward et al., 2022). This is a good alternative for non-stationary ecological time-series where the assumption of a random walk may be too restrictive.

In the regular geometric mean model, each population index time-series has a weight of unity, leading to a joint index of the amount and diversity of birds. Regardless of the statistical framework chosen, it may be worth considering models where the impact of candidate species on the indicators may vary both in sign and magnitude, i.e., with species weights not fixed to unity. Some positive changes in avifauna may result from processes generally considered undesirable (see Subsections 2.2 on auks and the fish community, and Section 3.3.2 on the trends in the archipelago vs. inland). Importantly, taking this approach is not a statement that species with negative effects on an indicator (e.g., factor loadings) are harmful species *per se*, or should be managed in response to an undesirable development the indicator monitored.

4.3. Models jointly analysing all aspects in one step

The filtering phase (Section 4.1) and indicator-generating phase (Section 4.2) can involve complex statistical procedures and may thus be easier to execute properly in two separate steps. However, a more elegant and modern solution, potentially improving indicator estimation, involves combining both steps into one comprehensive analysis. Bayesian methods offer flexible frameworks to integrate complex hierarchical steps of analysis into one jointly estimated large model. Simultaneous species-specific filtering, modelling of between-species interactions and estimation of annual trajectories of community patterns is also possible to do in the framework of joint species distribution models (Tikhonov et al., 2020).

We here present two existing analytic approaches, which may be applied for this purpose, both resulting in a single indicator (in contrast to DFA; see Section 4.2). Both approaches are described for simplicity assuming that the input (response variables) consists of (unfiltered) time-series of species-specific abundance indices without replicate values, not raw field observations.

Firstly, building on the geometric mean indicator, the proposed ideas from Subsections 4.1 and 4.2 can be integrated into one multivariate linear model (regression or mixed effects model). In this model, each species' log-transformed population index serves as one response variable, with its own residual variance. Covariates, e.g., the abundances of interacting species last year, abundances of the focal species last year, or year of observation as a continuous variable, are included as predictors tailored separately for each species. Depending on how the model is built (the syntax), the covariates applied might need to be assigned for all species but can be set to zero for the species where a particular covariate is presumed to have no effect. Each species is set to have its own intercept. The fixed annual effects or temporal trends (and associated random annual effects) are presumed to affect all species equally, conforming with the idea of the geometric mean indicator. The exponentiated partial annual predictions for those annual effects, common for all species, serves as an estimate of the indicator time series. This approach is much like the multivariate linear (mixed) model described in Subsection 4.2, but with additional covariates.

Secondly, we propose the usage of multivariate state-space models, where the state-variables are the log-transformed abundances of each indicator species. The process model describes the stochastic dynamics, allowing the species to interact with themselves (autoregressively, accounting for density dependent effects) and with each other through carefully considered interactions. Extrinsic covariates may also affect the species in the model, allowing for more complex "filtering" options. Latent species abundance estimates are linked to observation data via a stochastic observation model, which may or may not allow for multiple replicates and missing values. So far, this setting is identical to the analyses in Forsblom et al. (2021), who modelled the dynamics of a benthic–pelagic invertebrate community. The extra component needed for forming an indicator is to introduce another (unobserved) state-variable into the process model, which is thought to affect all the indicator species (all species or a subset of the included species). Unlike the species abundances, this state-variable would not be linked to data at all in the observation model part (i.e., if data are required, missing values are assigned for all years) but would be determined by the common extrinsic shocks found in the other state-variables (species). The indicator would be presumed to have zero mean and a dynamical structure, such as a random walk. If the effect of the indicator is desired to describe common variation for all species (like in the geometric mean approach), its effects may be fixed to unity for all indicator species, and instead, its estimated variance will determine its effect. Alternatively, if the indicator is defined to have both zero mean and unit variance (being stationary, i.e., a random walk not allowed), it should be possible to estimate its effects on different species, e.g., allowing different sign of the effects for each species. Finally, if the purpose is to link the indicator to a specific pressure, it would be possible to explicitly link the latent indicator to observations of some pressure in the observation model.

5. Implications for indicator development

Our primary goal is to provide information that can enhance the development of the seabird abundance indicators of the Baltic Sea, e.g., to better meet the requirements of the MSFD. For this goal, it is essential to minimize the effects of noise and recognized unwanted effects by addressing them in, e.g., the selection of species used for indicators, or the modelling of indicators. Such effects include, e.g., transient dynamics, interspecific interactions, pressures acting outside the focal area of the indicator (e.g., on the wintering grounds or on land).

We underline that the selection of indicators and assessment methods of ecosystem health and effects of pressures should be done with care to maintain the credibility of ecological indicators and also because the results may be used to inform policy or management decisions (Parsons et al., 2008). Species that show distinct local dynamics specific to the sea area under consideration may be recommended as good candidates for indicator species.

One should not necessarily exclude species that show strong temporal trends due to recent population recovery, re-establishment, or known pressures acting on land, or somewhere outside the area of interest. Instead, we recommend considering whether that undesired variation can be modelled or filtered out, so that it does not affect the indicator. The same applies for interspecific interactions between the bird species included in the indicator. As a guideline, the indicator can be seen as a common fluctuation in the involved species' carrying capacities. This can be roughly modelled also for species whose abundances are not close to their carrying capacities, or whose abundances are modified through interactions with the other included species, or through those human activities which do not affect the state of the sea. A further alternative is to identify patterns and trends in the time-series, where the species contribute to the indicator(s) unequally, potentially with loadings of even opposite sign. We briefly discussed in Subsections 4.2 and 4.3 various statistical approaches for doing the types of analyses outlined above. Even for the current study system, testing and evaluation of methodological approaches is an issue of future research. We also recommend that, in the case of rapid indicator changes, a sensitivity analysis should be routinely conducted to assess which species and processes are mostly driving the result.

While indicators can be estimated accounting for the dynamics of a given bird community, processes such as climate change and habitat shifts (e.g., induced for many species by the predation pressure of White-tailed Sea Eagle), may change the whole community structure and the associated interactions. Evolving interspecific interactions are not considered in this paper but are possible to model, and further illustrates the potential complexity of the system.

Overall, we suggest that carefully constructed sea bird population indicators can be useful tools in conservation and management policy, but we warn against overinterpretation of the meaning or causal relationships behind such indicators. We claim that some human induced pressures affecting bird populations, such as eutrophication, overfishing, and disturbance of breeding islands, can be measured more directly and reliably than through their indirect effects on bird populations. Therefore, multi-species indicators, even when filtered for unwanted effects, should be viewed primarily as large-scale summarizing statistics used for monitoring, often affected simultaneously by multiple pressures that can be unknown.

CRediT authorship contribution statement

Andreas Lindén: Writing – review & editing, Writing – original draft, Methodology, Investigation, Formal analysis, Conceptualization. **Antti Lappalainen:** Writing – review & editing, Writing – original draft, Investigation, Conceptualization. **Markus Piha:** Writing – review & editing, Writing – original draft, Visualization, Investigation, Formal analysis, Conceptualization. **Tuomas Seimola:** Writing – review & editing, Writing – original draft, Investigation, Conceptualization.

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Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Andreas Linden reports financial support was provided by European Union. Antti Lappalainen, Markus Piha, Tuomas Seimola reports financial support was provided by European Union.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolind.2025.114033.

Data availability

The authors do not have permission to share data.

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Supplementary material

Supplement 1. A description of how the temporal trends of a given species was compared between two areas (in section 3).

Technical comparison of two exponential trends

For pair-wise comparison of the log-linear temporal trends of a given species from two different areas, we presume statistical independence of the trend estimates, which are based on two different data sets. The estimated trends are defined as the slopes of continuous variable “Year”, in a log-linear model, i.e., either a GLM with log-link or a linear regression model of log-transformed population indices. This value describes how much the natural logarithm of the population abundance changes on average in one year. The trend coefficients can also be approximated from the population level compared to a predefined reference period, with the same interpretation as the log-linear slope.

We denote these slopes as b_1 (for the focal population; here the Baltic Sea) and b_2 (for the population to be compared with), and their estimated standard errors as s_1 and s_2, respectively. Our primary interest lies in the differences between these trends, whose point estimates (b_Δ) and standard errors (s_Δ) can under the assumption of independence be derived as following:

$$b_\Delta = b_1 - b_2$$

$$s_\Delta = \sqrt{(s_1^2 + s_2^2)} \ .$$

Over a period of t years (here corresponding to three times the generation length), we express the difference between the trends in percentages of population growth as following:

$$\text{Growth \%} = 100 \times (\exp(t \times b_\Delta) - 1)$$

and its uncertainty can be expressed as a 95% confidence interval constructed as following:

$$\text{Growth \% CI low} = 100 \times (\exp(t \times (b_\Delta - 1.96 \times s_\Delta)) - 1)$$

$$\text{Growth \% CI high} = 100 \times (\exp(t \times (b_\Delta + 1.96 \times s_\Delta)) - 1) \ .$$

An intuitive way to think about the population growth relative to another population (b_{Δ}), when both populations show non-zero trends, is interpreting it as if the population to be compared with was stable (i.e., if $b_2 = 0$). Under that scenario, the growth describes how much the focal population would grow (or decline) in t years.

Supplement 2. Comparison of annual population trends of seabirds in the North Sea and the Baltic Sea. Species included in the Baltic Sea indicator, but not in the North Sea are: *Anser anser, Cygnus olor, Tadorna tadorna, Aythya fuligula, Aythya marila, Melanitta fusca, Mergus merganser, Mergus serrator, Podiceps cristatus, Arenaria interpres, Calidris alpina, Sterna caspia*.

Species scientific name	Wintering area	Functional group	North Sea, ann. trend	Baltic Sea, ann. trend	Difference in trend
Branta leucopsis	same	grazing	0.1308	0.0974	–0.0334
Somateria mollissima	uncertain	benthic	–0.0412	–0.0709	–0.0297
Haematopus ostralegus	same	wading	–0.0372	0.0074	0.0446
Recurvirostra avosetta	same	wading	–0.0332	–0.0145	0.0187
Charadrius hiaticula	uncertain	wading	0.0021	–0.0039	–0.0060
Larus canus	same	surface	–0.0557	–0.0161	0.0396
Larus marinus	differ	surface	–0.0293	–0.0139	0.0154
Larus argentatus	differ	surface	–0.0265	–0.0011	0.0254
Larus fuscus	differ	surface	0.0133	–0.0002	–0.0135
Sterna sandvicensis	differ	surface	0.0012	0.0143	0.0131
Sternula albifrons	same	surface	–0.0160	–0.0007	0.0153
Sterna hirundo	differ	surface	–0.0345	0.0355	0.0700
Sterna paradisaea	same	surface	–0.0596	0.0110	0.0706
Stercorarius parasiticus	uncertain	surface	–0.0907	0.0080	0.0987
Uria aalge	differ	pelagic	–0.0019	0.0426	0.0444
Alca torda	differ	pelagic	0.0163	0.0322	0.0160
Cepphus grylle	differ	pelagic	0.0049	0.0033	–0.0016
Phalacrocorax carbo	differ	pelagic	0.0348	0.0082	–0.0266

Supplement 3. A description of the Finnish archipelago breeding bird data, and the statistical methods applied for data analysis of these data for calculating population abundance indices.

Finnish archipelago bird data and statistical analysis

To investigate annual abundance changes in archipelago bird populations, we utilized the Finnish national nest count data on islands in the archipelago, where voluntary trained birdwatchers, typically ringers, have surveyed the number of nesting pairs on visited islands or islets (see Below et al., 2019; Seimola et al. 2025). The data are in practice point-based, with one pair of coordinates per site, which typically represents the point-of-gravity of the surveyed island.

The reported number of pairs did not include zeros, and the set of surveyed species was not consistent across the data, implying that it is challenging to infer the true zeros for all species from the information on other species. Therefore, in further analyses, we decided to exclude all zeros and analyse the data accordingly. Furthermore, we excluded data from islands where the focal species was reported from less than two years.

Annual species-specific population abundance indices were calculated using package glmmTMB (Brooks et al. 2017) in the programming environment R (R Core Team 2023). For each species separately – the non-zero count of pairs being the response variable – we fitted a generalized linear mixed model (GLMM) with: i) factor variable year as a fixed effect (separate intercepts for all years), ii) the natural logarithm of island area as a covariate, and iii) site ID (island or islet) as a random effect on the intercept. Further, we applied a logarithmic link function and zero-truncated negative binomial error distribution. If the estimated overdispersion parameter exceeded 20 (indicating lack of overdispersion), we refitted the model with the zero-truncated Conway-Maxwell Poisson error distribution, which accommodates also underdispersion, and used those results instead.

The abundance indices and their 95% confidence intervals (CI) were produced by first exponentiating the annual estimated intercepts (and their CIs), and then scaling, by dividing them with the grand mean of all exponentiated annual effects. The result is a time series of abundance indices with unit mean.

Supplementary material references

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