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COVER PICTURE
A common guillemot *Uria aalge* © Tone Kristin Reiertsen

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Abstract

Layton-Matthews K., Buckingham L., Critchley E.J., Nilsson A.L.K., Ollus VMS., Ballesteros M., Christensen-Dalsgaard S., Dehnhard N., Fauchald P., Hanssen F., Helberg M., Masden E., May R.F., Sandvik H., Tarroux A. & Reiertsen T.K. 2023. Development of a Cumulative Impact Assessment tool for birds in Norwegian Off-shore Waters: Trollvind OWF as a case study. NINA Report 2295. Norwegian Institute for Nature Research.

There is growing interest in the economic potential of marine areas for e.g., offshore renewables, fisheries, and shipping. Thus, the cumulative stress on marine ecosystems and the species inhabiting them is increasing. This is of particular concern for migratory birds and seabirds which are undergoing global declines. In the light of an expanding global market for offshore renewables, knowledge of their cumulative impacts combined with other human-derived pressures on marine populations is crucial. This is set against the backdrop of climate change and associated large-scale changes in our oceans. Achieving sustainable development, while mitigating the effects of climate change, requires effective tools to assess the cumulative impacts of anthropogenic stressors on ecosystems. Cumulative impact assessments for the marine environment are strongly influenced by an approach developed by Halpern et al. (2008).

In this report, we present and demonstrate a beta-version of a cumulative impact assessment tool for marine industrial pressures on seabirds, which is being developed through the Research Council of Norway (RCN) -funded MARCIS project. The goal of MARCIS is to assess the impacts of marine anthropogenic pressures on seabirds in the North-East Atlantic, which will both utilise and further develop the Halpern formula and provide a publicly open web-based tool that acts as a decision support for marine spatial planning. Equinor has been exploring the potential for constructing a floating offshore wind farm, called Trollvind, in the North Sea. This proposed floating wind farm is in the early planning stages of development and is proposed to be located around the Troll offshore oil and gas platforms, approximately 65 km west of Bergen. This report presents; 1) a summary of a scoping of existing data of birds using Norwegian offshore area, 2) an assessment of bird migration through the North Sea and the Trollvind area, and the likelihood that some bird groups will be more impacted by an offshore wind farm in this area, and 3) a demonstration of the tool under development (the MARCIS web application), where we assess the potential cumulative impact of the proposed Trollvind OWF development and ocean warming on two seabird populations, as a case study. This report has specific emphasis on the demonstration of the tool.

Results of the scoping study of birds using the Norwegian offshore areas indicated a below-medium to medium sensitivity of waterbirds to wind farms in the Trollvind area. However, there were strong seasonal differences showing above-medium values in summer in the eastern and north-eastern parts of the area. Our results also revealed that migratory bird groups differ in their type of risks of impact associated with the Trollvind development, where migrating raptors, gulls, waterfowl and owls were at greatest risk of collision, while migrating seabirds, waterbirds and waders had a higher risk of displacement and/or barrier effects. In the demo of the tool, we estimated the cumulative impact of two stressors (Trollvind OWF and ocean warming) on two study populations (kittiwake breeding at Ålesund colony and common guillemot from Sklinna colony). Both their non-breeding distribution and the cumulative impact of the two stressors was visualised in the demo of the MARCIS App. The impact of Trollvind OWF was negligible for both populations, while disturbance led to a small reduction in guillemots' body mass and consequently their survival rates. However, ocean warming had a larger population impact, given the high emissions scenario used to quantify impact weights, particularly for guillemots. However, it is important to keep in mind that this demonstration is a case study of only two populations and should be interpreted in the larger context of the results from the scoping study and bird migration studies. The extensive range of species using this area at different times of year are much larger, and this has implementations for a potentially broader impact of such offshore developments. This also highlights the need for both spatial explicit distribution data and demographic/population data to ensure an appropriate knowledge base of population impacts before any OWF developments are conducted.

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Sammendrag

Layton-Matthews K., Buckingham L., Critchley E.J., Nilsson A.L.K., Ollus V.M.S., Ballesteros M., Christensen-Dalsgaard, S., Dehnhard N., Fauchald P., Hanssen F., Helberg M., Masden E., May R.F., Sandvik H., Tarroux A. & Reiertsen T.K. 2023. Development of a Cumulative Impact Assessment tool for birds in Norwegian Off-shore Waters: Trollvind OWF as a case study. NINA Rapport 2295. Norsk institutt for naturforskning.

Presset på marine områder øker med økende økonomiske interesser for bruk av havområdene. Marine områder har blitt viktige arenaer for industriell utvikling som for eksempel havvind, petroleumsaktivitet, skipstrafikk og fiskeri. I lys av et ekspanderende globalt marked for havvind, er kunnskap om sumeffekter av både havvind, andre marine industriaktiviteter og klimaendringer avgjørende for å sikre en god sameksistens med sjøfugl og trekkfugler som benytter de samme havområdene. Økt utnyttelse av havarealene øker sumeffektene og stresset på marine økosystemer og artene som bor der, sett i lys av klima- og økosystem-endringene som skjer i marine økosystemer. Å oppnå bærekraftig utvikling, samtidig som effektene av klimaendringer reduseres, krever effektive verktøy for å vurdere sumeffektene av menneskeskapte stressfaktorer på økosystemene. Metoder for å estimere sumeffekter for havmiljøet er sterkt påvirket av en tilnærming utviklet av Halpern et al. (2008), som ser på summen av ulike stress-faktorer sin påvirkning på et miljø og miljøets sensitivitet til de ulike stress-faktorene.

I denne rapporten vil vi presentere og demonstrere en betaversjon av et verktøy som kan benyttes i marin arealplanlegging og som kvantifiserer effekter av marin industri aktivitet på sjøfugler. Dette verktøyet utvikles gjennom det forskningsråds-finansierte MARCIS-prosjektet. Målet med MARCIS er å vurdere virkningene av marin industriaktivitet og klimaendringer på sjøfugler i Nordøst-Atlanteren, og vil både benytte og videreutvikle Halpern-metoden. Verktøyet vil bli gjort tilgjengelig som et offentlig åpent nettbasert verktøy, og kan fungere som beslutningsstøtte for marin arealplanlegging.

Equinor har undersøkt potensialet for bygging av en flytende havvindpark, kalt Trollvind, i Nordsjøen. Denne flytende vindparken er i et tidlig planleggingsstadium og foreslås plassert rundt olje- og gassplattformene i Trollfeltet, ca. 65 km vest for Bergen. Denne rapporten presenterer: 1) en oppsummering av en scoping av eksisterende data om fugler som bruker i norsk offshore-område, 2) en vurdering av fugletrekk gjennom Nordsjøen og Trollvind-området, og sannsynligheten for at noen fuglegrupper blir mer påvirket av en havvindpark i dette området, og 3) en demonstrasjon av verktøyet som er under utvikling (MARCIS-webapplikasjonen), der vi vurderer de potensielle sumeffektene av den foreslåtte Trollvind OWF-utbyggingen og havoppvarmingen på to sjøfuglbestander, som et casestudie. Denne rapporten har lagt spesifikk vekt på demonstrasjonen av verktøyet.

Resultatene av dette studiet indikerte at trekkfugler knyttet til vann (eks. dykkere, lom osv) hadde en under middels til middels følsomhet for havvindparker i Trollvind-området. Det var imidlertid sterke sesongforskjeller som viste over middels verdier om sommeren i østlige og nordøstlige deler av området. Resultatene våre avdekket også at grupper av trekkfugl hadde ulik risiko for å bli påvirket av en havvind-utbygging knyttet Trollvind-området. Trekkende rovfugler, måker, gjess og ender, og ugler hadde størst risiko for kollisjon med turbiner, mens trekkende sjøfugler, andre fugler knyttet til vann og vadefugler hadde høyere risiko for å bli fordrevet fra området eller utsatt for barriereeffekter av havvind-installasjoner. Sum-effektene av to ulike stressfaktorer (Trollvind havvindpark og havoppvarming) ble estimert for to studiepopulasjoner (krykkje og lomvi fra hhv Ålesund og Sklinna) og visualisert i beta-versjonen av MARCIS-appen. Effekten av en potensiell havvind-installasjon i Trollvind området var ubetydelig for begge populasjoner, mens fordrivelse fra området førte til en liten reduksjon i lomviens kroppsmasse og dermed effekt på bestandens overlevelsesrate. Havoppvarmingen hadde imidlertid en større påvirkning på bestandene, og spesielt for lomvi. Det er imidlertid viktig å huske på at denne demonstrasjonen er en casestudie av bare to populasjoner og bør tolkes i en større kontekst i lys av resultatene fra scoping-studiet og fugletrekkstudiet. Begge disse viser det omfattende spekteret av arter som bruker dette området, og hvordan mengde og sammensetning av arter varierer til ulike tider av året. Effekten av en offshore havvind utbygging i Trollvind området har dermed en potensielt større effekt. Dette fremhever også behovet for gode data, både romlige

distribusjons-data og data på demografiske rater eller bestandstall for å sikre et godt nok kunn-skapsgrunnlag om effekter på populasjoner før eventuelle havvind-utbygginger tar til.

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Contents

Abbreviations

Table 1. *Overview of abbreviations, terms and definitions used in this report.*

Abbre-viation	Term	Definition	Relevant methods section
WSI	Wind farm sensitivity index	Total sensitivity of waterbirds to collision and displacement from OWFs	2.1
BBMM	Brownian bridge movement model	Estimates an animal's likely occurrence in an area based on individual observations.	2.2.1
LCIA	Life cycle impact assessment	Tool to assess potential environmental impacts throughout the lifetime of a stressor.	2.2.2
PDF	Potentially disappeared fractions of species	Measure of the potential loss of species richness in an area, due to a given impact.	2.2.2
ABM	Agent-based model	Simulation-based models to assess individual seabird sensitivity to spatially explicit stressors.	2.3.3
DEE	Daily energy expenditure	Energy used (calories) per individual per day	2.3.3
CMR	Capture-mark-resight	Resighting data of adult seabirds used to estimate survival rates and climate effects	2.3.4
PVA	Population viability analysis	Risk-based assessments of stressor impacts on population growth rates, used to quantify impact weights for the Halpern formula	2.3.5
CIA	Cumulative impact assessment	Approach to assess and map cumulative human impacts	2.3.7
OWF	Offshore Wind Farm	Cluster of installed wind turbines in an offshore area	

Foreword

This report is the result of an add-on project to a larger research project, MARCIS (Marine spatial planning and cumulative impact assessment of blue growth on seabirds), which is funded by the Research Council of Norway. MARCIS aims to calculate and visualize the cumulative impacts from human pressures on seabirds in the North-East Atlantic. The outcome of MARCIS will be to provide a publicly open, web-based tool that acts as a decision support for marine spatial planning of at-sea industrial developments. This add-on project has been funded by Equinor, one of the industrial partners in MARCIS. Equinor has been investigating the possibility of developing a proposed floating offshore wind farm, called Trollvind, in the North Sea. It is proposed to be located in the vicinity of the Troll offshore oil and gas platforms, approximately 65km west of Bergen. In this add-on project, we have had the possibility to test out and demonstrate the MARCIS app and how it can provide an assessment of the cumulative impacts of two pressures (offshore wind farms (OWFs) and ocean warming) on seabirds. Through this project we have also compiled existing data on birds using these offshore marine areas and the risk posed to different groups of birds by OWF developments. This is highly relevant knowledge about how migratory birds and seabirds use offshore areas in a broader context than only for the proposed Trollvind area.

The content of this report was developed in the period November 2022 – May 2023, hence prior to Equinor's decision to postpone further development of the Trollvind offshore wind initiative indefinitely (published 22 May 2023). Since the current document was in its final stages of editing and proof-reading, we have chosen not to change formulations such as "proposed Trollvind development". The results and discussion in this report are however still valid based on the early-phase design parameters that formed the basis for the Trollvind offshore wind farm project.

We would like to thank the two Norwegian monitoring programs SEAPOP and SEATRACK, for providing non-breeding distribution- and activity- data of seabirds and timeseries data on demography and abundance of seabirds breeding in Norway. Such large-scale monitoring programmes and data are crucial and highly needed to provide a knowledge basis when cumulative impact assessment analysis is conducted.

We would also like to thank Stavanger Ringmerkingssentral og Håvard Husebø for providing ring recovery data, Morten Helberg and James Wilson for colour ring data on gulls and oystercatchers respectively. We thank Equinor for sharing detailed data on turbine dimensions, number of turbines, locations etc. that were needed for parts of the analysis in the development of the tool and the results. Also, thanks to Equinor for the funding and the opportunity to boost the development of the MARCIS-tool, which will be highly valuable for continuous development and as a foundation for further co-design of the tool together with stakeholders.

02.06.2023   Tone Kristin Reiertsen (Project leader)

1 Introduction

1.1 Background

Anthropogenic pressure on oceans is increasing worldwide (Halpern et al. 2015). At the same time, the world is undergoing a climate and biodiversity crisis (Johnson et al. 2017, Pörtner et al. 2022). This is reflected in global population declines of many bird species, particularly migratory species and seabirds (Dias et al. 2019, Howard et al. 2020). The threats to marine birds are many, e.g., invasive species, bycatch, hunting, disturbance and climate change (Dias et al., 2019). There is therefore a pressing need to understand the cumulative effects of anthropogenic activities (Breitburg et al. 1998).

Cumulative impact assessments are an increasingly used approach to inform management of developments on reducing the negative consequences for wildlife (Canter and Kamath 1995). However, quantifying cumulative impacts is challenging as negative effects associated with human activities ('stressors') can interact and species responses can be context dependent (Folt et al. 1999). Cumulative Impact Assessments (CIAs) provide a framework for assessing and mapping cumulative human impacts and are designed to evaluate trade-offs between human use and protection of marine ecosystems, for example development of fisheries and prey availability for marine predators (Cury et al. 2011). CIAs in marine ecosystems are strongly influenced by the global CIA performed by Halpern et al. (2008), who introduced a new additive modelling approach. Since then, CIAs largely following this approach have been applied worldwide, which have provided a sound basis for ecosystem-focused marine spatial planning (Korpinen and Andersen 2016). The formula introduced by Halpern et al. 2008 (hereafter the Halpern approach) describe the cumulative impacts of multiple stressors as follows:

$$CIA(x,y) = \sum_{i-1}^{n} \sum_{j-1}^{m} P_i * E_j * \mu_{ij}, \qquad \text{(Equation 1)}$$

where spatial data on human pressures (P_i) and ecosystem components (E_j) are combined with the impact weight (μ_{ij}). of the ith ecosystem component to the jth anthropogenic stressor The final product, CIA(x,y), is the cumulative environmental impact across all anthropogenic stressors and ecosystem components at a given location. In an RCN funded collaborative project between scientists and marine industrial partners, the MARCIS project (nina.no/English/Ecosystems/Marine-ecosystems/MARCIS), the Halpern formula is used as the basis for the development of a CIA tool. This project aims to assess the cumulative impacts of marine anthropogenic pressures on seabirds and migratory birds in the North-East Atlantic and to act as a decision-support tool for marine spatial planning. The CIA tool will be built as a web application ('the MARCIS web application'), which is openly available for the public and stakeholders. It will further develop the flexibility of the equation 1 to include both discrete spatially-explicit data (e.g., presence-absence of OWFs) as well as continuous data (e.g., timeseries of climate change and harvesting) (Halpern and Fujita 2013). Further, the selection of pressures and ecosystem components can be adjusted depending on the research focus or questions asked by stakeholders. For example, it is possible to use at sea abundance data combined with data layers of different stressors e.g., fishery or shipping (Maxwell et al. 2013, Lieske et al. 2020) and then get quantified estimates through modelling approaches of impact weights for these pressures and the cumulative impacts they have on seabirds in specific areas. This report presents results from a demonstration of the web application, with a case study of two stressors: ocean warming and a proposed offshore wind development and will illustrate the use of the tool and how we further develop the Halpern approach. An advantage of the MARCIS approach is that the impact weights are based on quantitative values of μ_{ij}, rather than expert judgement, which was originally used in the Halpern approach, which can lead to biases (Hartley and French 2021). Although the use of quantitative values can improve the reliability of impact assessments, estimating quantitative impact weights can be challenging, especially for data-poor species. We overcome this challenge in MARCIS by using different statistical approaches, which utilise different data sources to estimate impact weights. For migratory birds, where data mainly consists of ringing recovery data, Brownian Bridge Movement Models (BBMM), Life Cycle (Impact) Assessments (LC(I)A) can be used to estimate the most likely occurrence of birds and their sensitivity to stressors in a given area. For seabirds, detailed data of year-round abundances and demographic rates (reproduction and

survival) exists, which is collected through national monitoring programmes like SEAPOP. Additionally, information of seabird distributions and key seasonal foraging areas in the North-East Atlantic (Merkel et al. 2021, Buckingham et al. 2022) are available through geolocation technology, collected via international programmes like SEATRACK. This gives the possibility to link populations' distributions at sea to data collected at breeding colonies in MARCIS.

In MARCIS we use a two-step modelling approach to quantify impact weights, first we estimate year-round stressor effects on demographic rates like survival and productivity, using two different modelling approaches 1) Agent-based models (ABMs) and 2) demographic models (e.g., capture-mark-resight, CMR analysis) dependent on whether effects from spatial explicit or timeseries data of the pressures are in focus and, secondly, we scale up these effects to measure population-level impacts, through using output of these first two approaches as input to population viability analysis (PVA). In the first step, we quantify birds' sensitivity to anthropogenic pressures as effects on survival rates or breeding success, using either ABMs or demographic models. The approach used is determined by the data available for a given stressor. For spatially resolved data like present–absence data of OWFs or bycatch rates, we need ABMs to provide estimates of additive mortality or mass change due to a stressor. With an ABM, we can simulate an environment where individual 'agents' move and behave to pre-defined rules that resamble real-world responses to stimuli. When we change the environment of that world, we can observe how the behaviour of our agents' changes and how these changes affect mortality rates and/or body mass. ABMs have been used to predict the impacts of OWFs on a variety of taxa, including bats (Ferreira et al. 2015), marine mammals (Nabe-Nielsen et al. 2018) and raptors (Masden 2010, Eichhorn et al. 2012). ABMs also enable us to assess the cumulative effects of spatial explicit multiple stressors (or multiple OWFs), which are difficult to measure using standard impact assessments (Maxwell et al. 2013, Goodale and Milman 2016, Horswill et al. 2022). Additionally, ABMs allow us to determine the timing of stressor interaction and impacts, which is valuable when considering mitigation efforts. In seabirds, ABMs have previously been used to predict the impacts of OWFs on populations during the breeding season (e.g., Freeman et al. 2014, Searle et al. 2018, Warwick-Evans et al. 2018, Pollock 2022), but not the non-breeding season, which is an understudied period of the annual cycle in seabirds (Marra et al. 2015), and which have a stronger focus in MARCIS. When data of anthropogenic pressures are available as timeseries (e.g., ocean warming, harvest intensity) demographic modelling can be used to directly estimate stressor effects on demographic rates (survival, productivity) and abundances. We use three types of demographic models to study stressor effects in MARCIS, based on different data sources. In the absence of any demographic data, we fit state-space models (Searle et al. 2023) to estimate stressor effects on abundance trends. When breeding success and capture mark-resight (CMR) data are available we can estimate stressor effects on demgraphic rates. Using CMR analysis, we can measure how much of the variation in annual survival is explained by annual variability in an environmental covariate within birds' non-breeding foraging areas. This approach has been used on several seabird species (Reiertsen et al. 2014, Guéry et al. 2019, Reiertsen et al. 2021). For instance, by linking annual variability and trends reflecting climate change (a stressor in MARCIS) from seabirds' autumn and wintering areas to annual survival, we can determine seabird populations' sensitivity to climate change and potentially identify areas and seasons where they are most vulnerable to its effects. Such knowledge is key to understand the combined effects of the impacts of industrial pressures in the context of ongoing climate change, under marine spatial planning.

In the second step of our modelling approach, we combine the results from the ABMs and demographic models to calculate stressor effects on population sizes using a PVA. PVAs provide risk-based assessments of stressor impacts on population growth rates, which are used as impact weights in the Halpern formula (μ_{ij}). A PVA provides a robust framework, based on demographic rates (survival and reproduction), to forecast future population sizes under either baseline conditions or under scenarios of change resulting from ocean warming or OWF developments (Maclean et al. 2007, Freeman et al. 2014). PVAs facilitate the predictive modelling of populations under alternative scenarios and hence can be used to evaluate the effectiveness or consequences of different management decisions. Thus, PVAs can be considered as a type of risk assessment of the long-term viability of animal populations and are considered as best practice for understanding population-level consequences of single or multiple stressors.

Rapid climate change is causing global population declines of seabirds and it has been identified as one of the top-three threats to seabirds (e.g., Dias et al. 2019). Climate change is impacting seabird populations in many ways, e.g. increased frequencies of winter storms causing direct mortality (Clairbaux et al. 2021) or increased ocean temperatures affecting prey distributions, abundances and composition (Fossheim et al. 2015). While renewable energy developments will be part of the solution to mitigate the climate crisis and meet growing energy needs, such developments are also often harmful to migratory birds (Furness et al. 2013, Goodale and Milman 2016), including seabirds (Masden et al. 2010). Increasing demand for green energy and, consequently, future offshore energy developments (e.g., offshore wind, wave and tidal installations) will increase the pressure on marine areas. Such offshore renewable installations can occupy important areas for foraging and migrating seabirds (Searle et al. 2022), and finding solutions for their co-existence remains a challenge. Therefore, ocean warming associated with climate change and offshore renewables both have the potential to impact marine bird populations (Searle et al. 2022). Understanding their cumulative impacts is fundamental to perform accurate assessments.

In the light of an expanding global market for offshore renewables, knowledge of the cumulative impacts on seabirds of OWF and other human-derived pressures such as climate change and fisheries is crucial. In Norway, wind energy now meets a substantial and increasing proportion of Norway's energy demand (9% in 2021, NVE 2023b). Although Norway currently has few operational OWFs, the Norwegian government recently announced plans to increase capacity of OWFs to 30GW by 2040 (Regjeringen 2022), which will require a rapid OWFdevelopment of Norwegian waters. In addition, with the development of 'floating' wind farms, it is now possible to build in deeper waters (Díaz et al. 2022), potentially leading to OWF development of areas of the Norwegian coast that were previously unsuitable e.g., the Barents Sea. In fact, the Norwegian government has recommended investigation of 20 new areas for potential OWF development, spread along the entire Norwegian coastline (NVE 2023a).

Seabirds spend much of their lives at sea, often only coming to land to breed, and depend solely on the marine environment for food. They are therefore closely connected to the marine environment, representing top predators in marine ecosystems. Development of offshore areas will likely impact pelagic seabirds in particular, being a creature inhabiting the environment both below (diving) and above ocean surface (flying), especially during the non-breeding season when seabirds are less constrained to remain close to breeding colonies and shows a wider offshore distribution. Seabirds breeding in temperate and polar environments also typically experience their highest levels of mortality during the non-breeding season (Harris et al. 2007, Acker et al. 2021), when they are energetically challenged by harsh environmental conditions such as low temperatures, increased frequency of storms, and reduced daylight hours for foraging. Therefore, knowledge of why and how migratory birds and seabirds uses the marine ocean throughout the year is important when planning for OWFs, in addition to how the different bird groups are affected by OWFs. Information on the distribution of migratory birds and seabirds key seasonal foraging areas at sea can also contribute to targeted risk management of developments like OWFs. For example, turbines may be switched off during high-risk periods.

Seabirds rely on highly productive marine areas to feed and are at risk from direct mortality through collisions with wind turbines and rotor blades. They can also be displaced from foraging areas when they avoid OWF structures, particularly rotating turbines (Dierschke et al. 2016). Such structures can also act as attractors, by providing resting or nesting sites which can, in turn, increase collision risk (Dierschke et al. 2016). Displacement has 'sub-lethal' effects, resulting in behavioural and energetic changes to individual seabirds. Sub-lethal effects include reduced access to foraging habitats thus potentially reducing energetic intake (Busch and Garthe 2016); 'barrier effects', where seabirds travel around a development area, increasing the energetic cost of a journey (Masden et al. 2009, 2010); and increased time spent in energetically costly 'escape behaviours' e.g., flying or diving (Fliessbach et al. 2019). Sub-lethal impacts of displacement on seabirds are more difficult to quantify than the lethal effects of collision (Drewitt and Langston 2006), but they are potentially important as they can impact future breeding success and survival (Searle et al. 2014, Lane et al. 2020). However, such structures can also enhance foraging opportunities. As many seabird species are migratory and can travel long distances, making use of different ocean areas throughout the year (Fauchald et al. 2021), individuals can encounter

several OWFs during their annual cycle, potentially leading to substantial cumulative effects. A key process linking sub-lethal effects of OWFs to demographic rates is the empirical relationship between body condition and survival, allowing us to predict how changes in individuals' condition affects their demographic rates and therefore population sizes. For instance, recent work from Daunt et al. (2020) identified positive mass-survival relations from several seabirds breeding in the UK.

Migratory birds, including waterfowl, waders and passerines, also regularly cross and utilise marine areas. Millions travel biannually between breeding and wintering grounds and often cross extensive barriers such as marine areas, deserts, and mountain ranges (Alerstam 1990). Novel anthropogenic stressors have the potential to affect birds on migration, by creating an additional barrier. Thus, with the development of offshore renewables, it is imperative to gain an overview of the potential overlap between this stressor and migratory trajectories, to determine how OWFs affect migratory birds (Masden 2010). Potentially, OWFs can represent a challenge for migratory birds (Newton 2010), being a barrier that infer extra costs that especially may impact small terrestrial songbirds, which rely on fat deposition during crossings and awaiting favourable weather conditions. Additionally, sudden changes in weather conditions during crossings (e.g., fog, heavy rains) are known to cause massive downfalls of migratory birds on islands, light-houses, ships and other offshore installations, because birds lose their orientation. This can po-tentially, increase collision risk as well, especially in a seasonal context. Some species move on a broad front, whereas others (e.g., raptors and small terrestrial birds) often hesitate to cross open seas and closely follow leading coastlines until crossing is unavoidable (Bildstein 2006, Santos et al. 2020). Migratory trajectories over open sea are hitherto not mapped in detail, alt-hough some species' trajectories are better known than others (Newton 2010, Shariatinajafabadi et al. 2014). Having an overview of the likely occurrence and impact of OWFs on migratory birds, and especially link this to time of the year is therefore also important for marine spatial planning. This knowledge enables the potential to avoid high-risk areas or seasons, or to implement oper-ational limitations in certain time of the year, which may lead to a better co-existence between birds at sea and human industrial activities.

1.2 Project aims

Equinor is exploring the potential to develop a floating offshore wind farm in the Trollvind area in the northern North Sea. Its' proposed location is around the Troll offshore oil and gas platform, approximately 65 km west of Bergen (*Figure 1*). The early phase design parameters are pre-sented in appendix 2, and installed capacity would be 1GW. This report includes:

1. Summary of a scoping exercise and compilation of existing data on birds using the Trollvind OWF area and their sensitivity to OWFs, which was already published in a previous report (Ollus et al. 2023).
2. Assessment of bird migration trajectories through the North Sea area and the potential im-pact of Trollvind OWF on these, based on modelling of ringing recovery data from Norway, UK, Belgium, the Netherlands, Germany and Denmark.
3. Demonstration of a light-version of the MARCIS web application and how it can be used in future marine spatial planning, through a case study of the proposed Trollvind OWF devel-opment. A demonstration version has been developed for two seabird populations (common guillemot and kittiwake), selected based on the initial scoping report, where we:
   - quantify overlap between seabird distributions and the location of Trollvind OWF,
   - estimate the individual sensitivities to the OWF as body mass change or additive mortality, by developing population-specific ABMs,
   - estimate lethal effects (mortality) from non-lethal effects (body mass change) through mass-survival relationships,
   - estimate effects of ocean warming in the non-breeding season on population adult survival
   - quantify population-level impacts of the two stressors using a PVA,
   - provide population- and stressor-specific impact weights visualised in the web appli-cation, for two stressors: Trollvind OWF and ocean warming,

- calculate and visualise spatially resolved population-specific sensitivities to Trollvind OWF and ocean warming in the MARCIS web application.
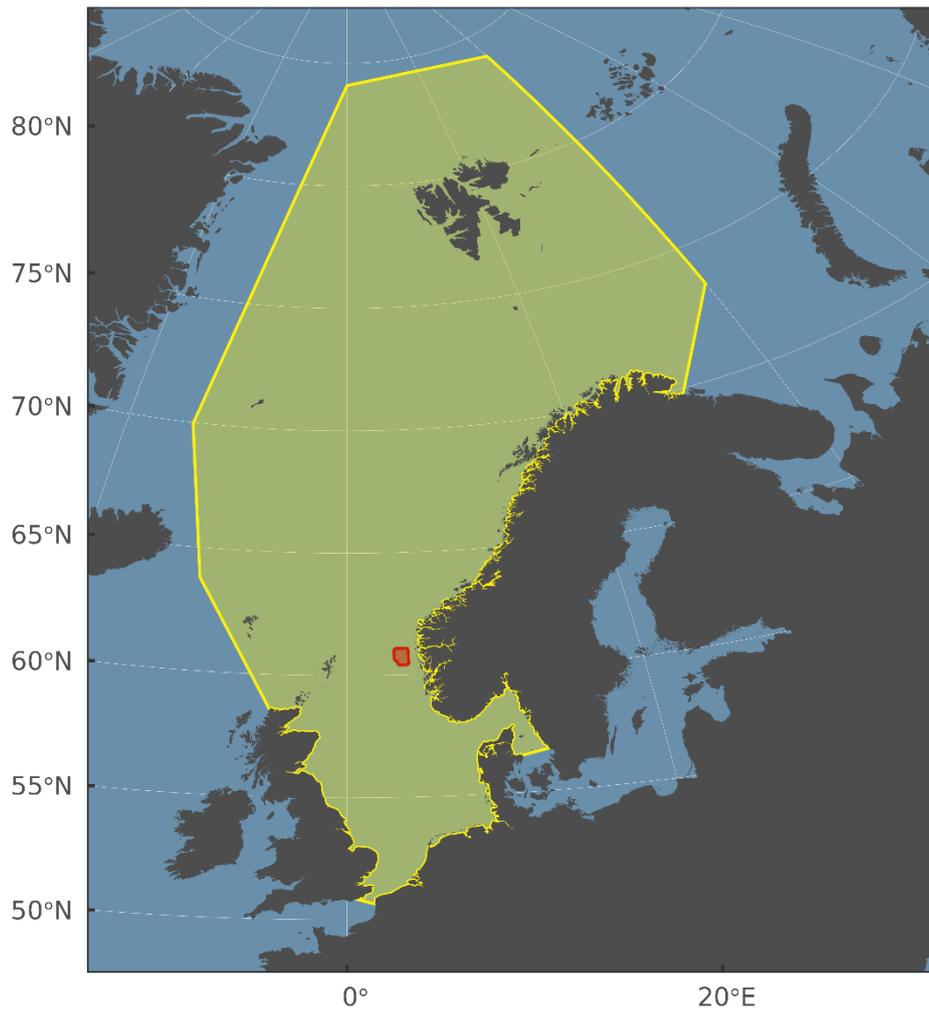
Figure 1. *Location of Trollvind area (including a 10km buffer zone, red) for the proposed OWF development within the greater MARCIS study area (yellow).*

2 Methods

2.1 Seasonal use and sensitivity of birds to Trollvind OWF

The Trollvind area was scoped for its importance for waterbirds in an earlier report that investigated which species use the area and assessed their sensitivity to wind farms (Ollus et al. 2023). This was done by generating species habitat preference models and species-specific sensitivity indices for 58 species of seabirds and other waterbirds and combining these into sensitivity models. These models allowed the assessment of the relative sensitivity of birds to wind farms in the Trollvind area compared to the Norwegian Exclusive Economic Zone at large. They do not quantify the vulnerability of the bird populations but allow the comparison of sensitivity between areas, seasons, and species.

2.2 Potential impacts of OWFs on migratory birds

To evaluate the effects of Trollvind OWF on migratory birds, we used ringing recovery data provided by the Norwegian Bird Ringing Centre at Stavanger Museum, Norway, and their collaborators within the EURING network. We used Norwegian ringing and recovery events crossing the North Sea basin, where at least one event occurred either in United Kingdom, Belgium, the Netherlands, Germany, or Denmark. Events occurring at longitudes > -17 and < 40, and latitudes > 45 were excluded. Time intervals longer than 60 days between ringing and recovery events were likewise excluded to reduce the likelihood of including recoveries from two migrations. This resulted in data from 123 species, which were then grouped into functional groups according to May et al (2021; *Appendix 1, Table A1.1*).

2.2.1 Estimating bird migration trajectories

The Brownian bridge movement model (BBMM) estimates an animal's likely occurrence in an area based on individual observations. The BBMM is based on a conditional random walk, taking into account the distance and time between observations (Horne et al. 2007). Thus, it can also be used to estimate migration trajectories (Horne et al. 2007, Palm et al. 2015). We used the BBMM to estimate the migratory trajectories by estimating an animal's probability of occurrence between the ringing and recovery events in a 2 x 2 km grid across the North Sea basin. For the functional group corvids, other non-passerines, owls, and raptors we did the BBMM analysis in one run. For computational reasons, we had to run the BBMMs for the other functional groups separately for each species, and then combine the migration maps of each species into joint maps for each functional group. We used the R package 'adehabitatHR' to model the BBMMs and exported the produced maps in tif-format using the R package 'raster'.

2.2.2 Estimating potential impacts of Trollvind OWF on migrating birds

Knowledge of migratory trajectories alone is insufficient to assess the impact of novel stressors on numerous species of migratory birds. Life cycle assessment (LCA) is a tool to assess potential environmental impacts throughout the lifetime of a stressor, for instance a technological installation. The LCA can then be joined by a life cycle impact assessment (LCIA), which models the potential impact per unit of a stressor, for instance the impact of collisions per kWh produced in an OWF (May et al. 2021). LCA and LCIA were originally developed for greenhouse gas emissions and energy accounting (Wang et al. 2019), but LCIA has since been developed also for assessing impacts of renewable energy developments on birds, through habitat loss, disturbance, collisions, and barrier effects (May et al. 2020, 2021). The LCIA uses the spatial distributions of species and has the advantage of allowing the assessment of multiple impact pathways simultaneously. Consequently, the joint combination of BBMM, LCA and LCIA allows us to assess the impact of the Trollvind OWF on migratory birds whose migratory trajectories overlap with Trollvind OWF.

A LCIA was applied to the migration maps to assess the potential impacts of collision, disturbance and barrier effects in the Trollvind OWF. The methods were adapted from (May et al. 2020, 2021), who developed LCIAs to evaluate impacts of onshore wind energy on bird species richness. The potentially disappeared fractions of species (PDF) – a measure of the potential loss of species richness in an area – for each impact was calculated for each functional group (May et al. 2021, *Appendix 1*). PDF values per functional group were calculated based on the

expected number and size of the turbines in the Trollvind OWF (*Appendix 2 Table A2.1*). As the exact siting of the turbines remains to be decided, random locations were selected for the LCIA and iterated 100 times to generate average PDF values from the Trollvind OWF. In addition, PDF values were calculated for every grid square (2 x 2km grid) in the wider Trollvind area based on the hypothetical turbine placement in each grid square. Maps of these results were produced to assess where cumulative PDF values combined for collision, disturbance and barrier effect were expected to be highest, both by functional group and for all migratory species combined. The following equations from (May et al. 2020, 2021) were used for calculating the PDF values:

Disturbance, D:

$$PDF(D)_{k,w} = \frac{S_k P_{k,i} \left(1 - \left(\frac{A_{org} - t_w * (\pi \times (D_k * d_{k,max})^2)}{A_{org}}\right)^z\right)}{\sum_i^I S_k P_{k,i}}$$

where $S_k \cdot P_{k,i}$ = number of species locally present at cell *i* within group *k*, A_{org} = 2 km^2, t_w = 1 turbine, D_k = disturbance factor within group *k* and $d_{k,max}$ = maximum flight initiation distance within group *k*.

Collision, C:

$$PDF(C)_{k,w} = \frac{S_k P_{k,i} \left(1 - \left(\frac{A_{org} - R_k * t_w \times (\pi * r_w^2)}{A_{org}}\right)^z\right)}{\sum_i^I S_k P_{k,i}}$$

where r_w = rotor blade length of turbine *w* and R_k = probability of annual per-turbine collision within group *k*.

Barrier, B:

$$PDF(B)_{k,w} = \frac{S_k C_{k,i} \left(1 - \left(\frac{A_{org} - (\pi * t_w * M_k * (D_k * d_{k,max})^2)}{A_{org}}\right)^z\right)}{\sum_i^I S_k C_{k,i}}$$

where $S_k \cdot C_{k,i}$ = number of species locally present at cell *i* within the 50% kernel for group *k* and M_k = migration cost within group *k*.

As this study included several species previously not included in wind energy LCAs, mainly seabirds, new data was collated for these species. There is currently little available data on turbine collision rates for seabird species, given the challenges of monitoring offshore wind farms. Collision rates were therefore estimated based on the species ranking within a collision vulnerability index, which were calculated using the Furness et al. (2013) methodology and updated to account for avoidance behaviour (Wade et al. 2016) and modelled estimates of time spent flying at turbine height (Johnston et al. 2014). For species with missing collision rates, the average value between the two species ranked above and below them in the collision vulnerability rankings was selected. Flight initiation distances were taken from May et al. (2021) and updated with new values for 14 species (*Appendix 1, Table A1.1*). New values for D_k and R_k per functional group were then calculated using this updated data. The low collision rates for seabirds resulted in some negative PDF values, which were instead set to zero. Migration distance and average mass values (the inputs for M_k) were collated for all new species. To calculate the barrier effect, it was assumed that migratory trajectories would lie within the 50% kernel of the migration maps. Any presence value outside of the 50% value were set to zero. Values within the kernel were then rescaled to between zero and one.

2.3 Demonstration of MARCIS web application

2.3.1 Study populations

In this section of the report we developed a demonstration of the MARCIS tool, which presents its use as a decision support tool for planning OWF development. We used the Trollvind area, and the potential development of an OWF, as a case-study to develop a demonstration version of the MARCIS web application, using geolocation and demographic data (*Table 2*) from two seabird populations and focusing on two stressors (OWF and ocean warming). Based on the initial scoping report for Trollvind OWF (Ollus et al. 2023), we selected two seabird populations (for which there was sufficient data) that were considered most at risk from the Trollvind development during the non-breeding season: common guillemots *Uria aalge* (hereafter 'guillemot') breeding on Sklinna and black-legged kittiwakes *Rissa tridactyla* (hereafter 'kittiwake') breeding in Ålesund. Geolocation and demographic data are collected annually at the colonies as part of the SEAPOP (seapop.no/en) and SEATRACK (www.seapop.no/en/seatrack) programs.

Table 2. Overview (analytical use, source of data and time series length) of the spatial data (geolocation) and demographic data, used for visualisation in the MARCIS App, and as inputs to the agent-based models (ABM, *Methods 2.3.3*), mass-survival analysis (mass, *Methods 2.3.4*), climate-survival analysis (climate, *Methods 2.3.5*) and population viability analysis (PVA, *Methods 2.3.6*).

Data type	Analysis	Period data available	
		Guillemot	**Kittiwake**
Geolocation	ABM Climate MARCIS App	2013-2021	2015-2021
Body mass and size	Mass	2008-2050	2011-2050
Adult survival	Climate PVA	2008-2021	2011-2021
Immature survival	PVA	Study period average	Study period average
Breeding success	PVA	Study period average	Study period average
Adult survival-breeding success correlation	PVA	Study period average	Study period average
Population size	PVA	2020	2020

2.3.2 Extraction and visualisation of bird densities and stressors data

Bird density maps for the two study populations were based on light-level geolocation data collected through the SEATRACK project. Geolocation-immersion loggers (hereafter 'geolocators' or GLS) were deployed on adult breeding birds, with the geolocator attached by cable tie to a colour ring on the bird's leg. Geolocators measured light and salt-water immersion. Raw light data were used to derive locations, following the methods described in Bråthen et al. (2021). Salt-water immersion data were used to estimate colony departure and arrival dates and time-in-activity budgets. Species- and colony-specific species distribution models based upon environmental variables at sea and information about the population sizes were fitted on filtered locations to derive monthly utilisation distributions following Fauchald et al. (2021). For each month, these maps provide the average number of birds that can be expected to be observed in a cell of the size 0.25° × 0.25° (approximately 20 × 20km in the study area).

Visualising ocean warming impacts in the MARCIS App required a data layer that was spatially explicit and ideally available at a monthly resolution, narrowing down the possibilities considerably. We selected sea surface temperature as it reflects ocean warming and has been shown to indirectly affect survival and productivity of seabirds (Sandvik et al. 2005). The sea surface temperature data was used to quantify the impact of ocean warming on seabirds (*Methods 2.3.5-2.3.6*). Forced historical data of SST were taken from the High Resolution Model

Intercomparison project (HighResMIP) for CMPI6 (Haarsma et al. 2016), which provides high spatial resolution ocean data (0.25° × 0.25°), where historical SST data were available from 1950-2014 and forecasted SST data were available from 2015-2050. Gridded monthly SSTs were downloaded for the MARCIS App study area (Figure 11), as well as any areas utilised by the seabirds outside of this MARCIS core area during the non-breeding season (*Figure 11*). Change in SST was visualised in the web application as the difference in monthly SST from year 2008 (start of guillemot demographic time series) to year 2050, per grid cell, at a resolution of 0.25°.

The Halpern framework ranges between 0 and 1 per raster cell (Halpern and Fujita 2013). We used the Min-Max approach (Han et al. 2011) to geographically rescale the monthly stressor and seabird distribution maps:

$$\text{Rescaled(x)} = \frac{\text{Pixel(x)} - \min}{\max - \min}.$$

The rescaled seabird density maps thus ranged in value from 0 (low bird density) to 1 (high bird density), independent of the underlying population size. The rescaled ocean warming stressor map had a value ranging from 1 (max influence) in pixels with maximum SST increase to 0 (lowest influence) in pixels with lowest SST increase. The rescaled OWF stressor map had a value ranging from 1 (max influence) at the OWF (represented by the centroid of the project area) to 0 (no influence) as a function of the inverse distance to the border of the project area.

2.3.3 Seabird sensitivity to Trollvind OWF

Long-term tracking data were used to develop an agent-based model (ABM) to predict the impacts of Trollvind OWF on the two focal seabird populations during the non-breeding season. We used data from geolocation-immersion loggers (GLS), which record distribution and activity. Although the tracking data that parameterised our model covered the annual cycle, we focused on the non-breeding season as geolocation-immersion loggers have much lower spatial resolution than e.g., GPS data and therefore are unsuited for displaying fine-scale foraging ranges during the breeding season. However, they are extremely useful for understanding broad-scale distribution patterns, including population-level migration during the non-breeding season (e.g., Merkel et al. 2021, Buckingham et al. 2022), which are essential when predicting the likelihood of interaction with a stressor. In addition, the salt-water immersion data that these loggers also record can be used to estimate behaviours and daily activity budgets, enabling us to calculate energy expenditure (Fayet et al. 2017, Burke and Montevecchi 2018). Incorporating energy expenditure enables us to also assess sub-lethal effects, as we can investigate how an energetic budget may change due to displacement effects from an OWF. Guillemots are sensitive to displacement effects (Peschko et al. 2020) but, due to their low flight height (Cook et al. 2012) and reduced time flying during the non-breeding season (Buckingham et al. 2023), they have low sensitivity to collision (Furness et al. 2013, Ollus et al. 2023). Although kittiwakes have shown weak avoidance of OWFs during the non-breeding season (Peschko et al. 2020), they are generally considered more sensitive to collision than displacement effects (Furness et al. 2013, Ollus et al. 2023). We therefore assessed guillemot sensitivity to displacement and kittiwake sensitivity to collision within our ABM.

We predicted interactions with Trollvind and assessed the lethal effects of collision in kittiwakes and the sub-lethal effects of displacement in guillemots. Using code from the stochLAB package (Caneco et al. 2022), we used the stochastic Collision Risk Model (McGregor et al. 2018) to assess the likelihood of kittiwakes colliding with Trollvind if interaction occurred. To assess displacement effects in guillemots, we incorporated activity data to estimate time-in-activity budgets. As different behaviours have different energetic costs, we used these activity budgets to calculate energy expenditure. This enabled us to estimate the impacts of sub-lethal effects of displacement from the Trollvind development area.

2.3.3.1 Data collection and processing

Daily utilisation distributions for each day of the non-breeding season were estimated by smoothing monthly utilisation distributions (see *Methods 2.3.2*). Time-in-activity budgets were estimated for each species using salt-water immersion data.

For guillemots, behaviours were allocated following Buckingham (2022). Time was allocated to the following behaviours:

- Active on the water (T_{Active}) if the geolocator was ≥ 92% wet, which included time foraging, pauses between foraging bouts, and swimming and preening behaviours.
- Flight (T_{Flight}) if the geolocator was completely dry for ≤ 30 minutes during the day (Dunn et al. 2020).
- Resting at the colony ($T_{Rest:\ at\ nest}$) if the geolocator was completely dry for > 30 minutes during the day or for the entire night (Sinclair et al. 2017, Dunn et al. 2020).
- Resting on the water ($T_{Rest:\ at\ sea}$) if the geolocator was between 0-92% wet, indicating that the bird had tucked the geolocator-equipped leg into the plumage whilst resting on the water.

As we could only measure leg-tucking occurrences of the geolocator-equipped leg, we accounted for this by doubling $T_{Rest:\ at\ sea}$ and reducing the T_{Active} accordingly.

As we were only interested in collision risk for kittiwakes, and therefore did not need to calculate energy expenditure, we only needed to extract the time spent in flight to calculate maximum distance travelled per day. Therefore, time was allocated to flight (T_{Flight}) if the geolocator was completely dry (McKnight et al. 2011). Remaining time was allocated to other (T_{Other}).

For each species, we extracted the mean and 95% confidence intervals for time spent in each behaviour for each day. Mean dates for leaving and arriving back at the breeding colony were calculated using the conductivity (wet-dry) data from the individual loggers. Briefly, individual time series of conductivity data were first aggregated per colony and per year, then smoothed using general additive models before running a Lavielle partitioning analysis (Barraquand and Benhamou 2008) to identify transitions between periods of varying conductivity levels, e.g. from mostly wet (assuming the birds were at sea, to mostly dry (assuming the birds were at their colony). The approach is described in detail in (Fauchald et al. 2019). We created a range of dates by adding 20 days before and after these means to include stochasticity.

2.3.3.2 Agent-based models

We created simulations of 1000 individual agents for each population.

For guillemots, the schematic modelling framework is shown in *Figure 2*. We assigned each simulated agent a day for leaving the breeding colony, by sampling a random date within the colony-specific mean ± 20 days. On this day, we assigned the location of the agent to the grid cell of the utilisation distribution that the breeding colony was located in. We simulated a daily activity budget by sampling a value of T_{Active}, T_{Flight}, and $T_{Rest:\ at\ nest}$ from within their 95% confidence range for that day, with the remaining time allocated to $T_{Rest:\ at\ sea}$. Using the simulated activity budget, we determined the maximum distance that the agent could have travelled in that day, using the maximum flight (19.1ms^{-1}, Pennycuick 1997) and swim speeds (1.15ms^{-1}, Merkel and Strøm Submitted). We extracted the grid cells within this range of the agent's current location and assigned the agent a new grid cell, using the probability of occurrence from the utilisation distribution for that day. We extracted the mean daily sea-surface temperature (SST) for the agent's location from satellite-derived data (0.25° resolution, Reynolds et al. 2007).

If at least part of the grid cell that the agent was newly located in overlapped with the Trollvind development zone, we allocated it to one of three categories a) the footprint + 3km; b) the 3-9km range; or c) > 9km from the footprint (i.e., outside of the impact zone). We used the area of overlap between the grid cell and each category to determine the probability of the agent being in each category. If the agent was allocated into a category that was within 9km of the wind farm, we selected a probability of displacement from within the ranges of the displacement values from Peschko et al. (2020): 63% (range: 47-74%) within the footprint + 3km, and 49% within 3-9km (no range provided, thus we selected an integer from 0-100). If the agent was displaced, we added an additional 30 minutes of flight to its daily activity budget, reducing $T_{Rest:\ at\ sea}$ accordingly. As the energetic impacts of displacement on guillemots are poorly understood, and may

include reduced access to food, reduced foraging time, and increased escape behaviours, this additional flight time represented an energetic cost of displacement.

We repeated this process for each day of the non-breeding season, assigning a new grid cell and activity budget each day. We assigned each agent a day for arriving back at the breeding colony, thus indicating the end of the non-breeding season, by sampling a random date within the colony-specific mean ± 20 days.

At the end of the simulations, we calculated daily energy expenditure (DEE) for each agent using the following equations (Elliott and Gaston 2014, Buckingham 2022, Patterson et al. 2022):

1) when the daily mean SST ≤ 14.3°C

$$DEE_{Guillemot} = 508T_{Flight} + (118 - 2.75 \times SST)T_{Active} + 33T_{Rest:at\ nest} + (72 - 2.75 \times SST)T_{Rest:\ at\ sea}$$

2) when the daily mean SST > 14.3°C

$$DEE_{Guillemot} = 508T_{Flight} + (118 - 2.75 \times SST)T_{Active} + 33 \times (T_{Rest:\ at\ nest} + T_{Rest:\ at\ sea})$$

We summed DEE for the full non-breeding season for each agent. We used a paired t-test to calculate if there was a significant difference between total non-breeding season energy expenditure with and without interaction with Trollvind. Finally, we converted this to the difference in mass at the start of the subsequent breeding season for each agent, with 0.072g lost per additional kJ (Dunn et al. 2022).
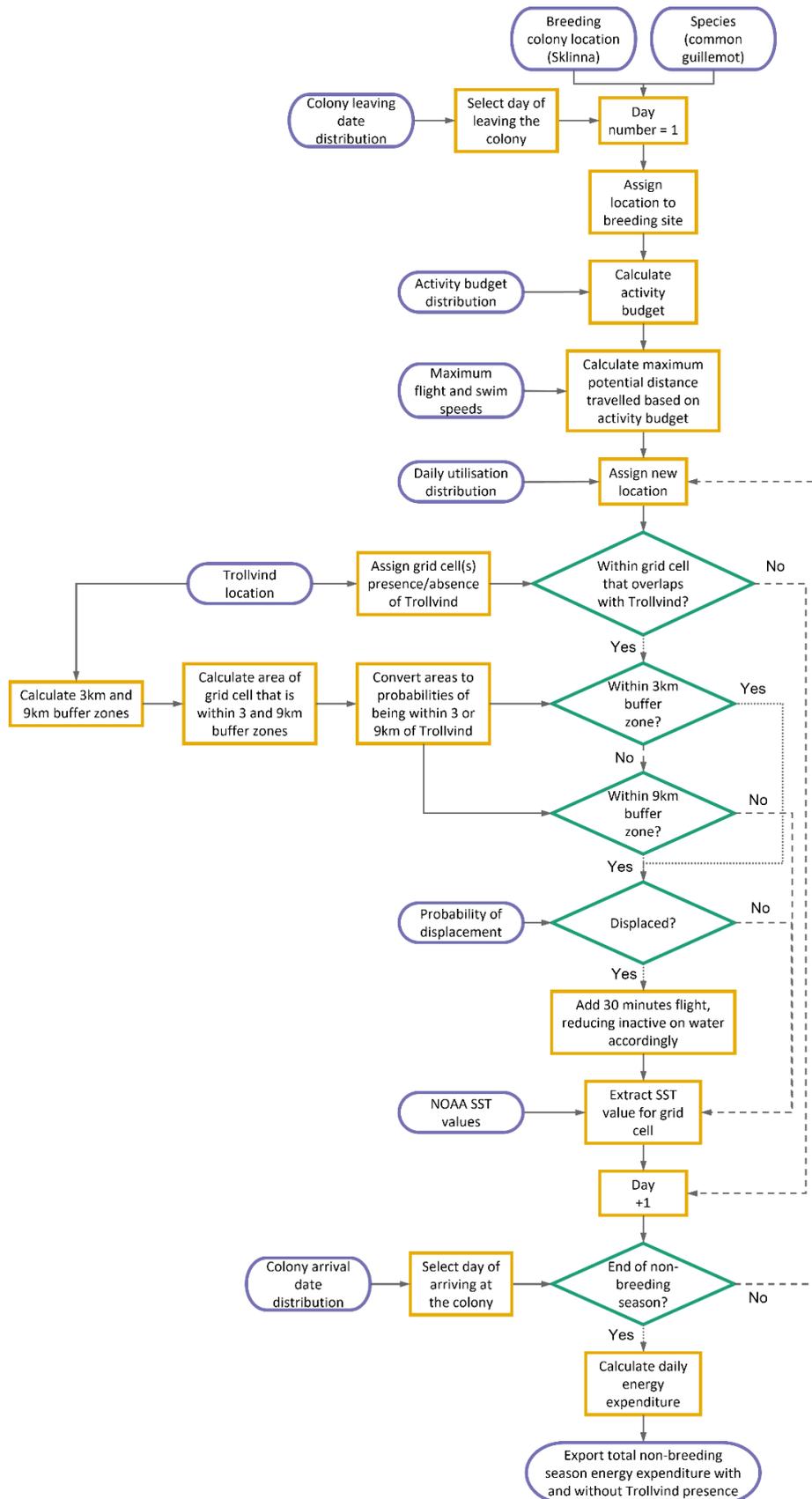
Figure 2. *Schematic of the agent-based model for guillemots. Purple oval boxes are input and output data, yellow square boxes are processes and green triangular boxes are questions.*

For kittiwakes, the schematic modelling framework is shown in *Figure 3*. We assigned each simulated agent a day for leaving the breeding colony, by sampling a random date within the colony-specific mean ± 20 days. On this day, we assigned the location of the agent to the grid cell of the utilisation distribution that the breeding colony was located in. We simulated a daily activity budget by sampling a value of T_{Flight} from within their 95% confidence range for that day, with the remaining time allocated to $T_{Other}$. Using the simulated activity budget, we determined the maximum distance that the agent could have travelled in that day, using the maximum flight speed (13.1ms-1, Pennycuick 1997) We extracted the grid cells within this range of the agent's current location and assigned the agent a new grid cell, using the probability of occurrence from the utilisation distribution for that day.

If the grid cell that the agent was newly assigned to overlapped with the Trollvind footprint, we used the area of overlap between the grid cell and the footprint to determine the probability of the agent being within the development footprint. If the agent was allocated into the footprint, we selected a probability of collision from a range of simulated values. We simulated 1000 values of the probability of collision from a single transit through the rotor swept area of a turbine using code extracted from the stochLAB R package (Caneco et al. 2022) to run the stochastic Collision Risk Model (Band 2012, Masden 2015, McGregor et al. 2018). We used the pre-determined flight height distribution (Johnston et al. 2014) and the avoidance rate for kittiwakes and gave an equal probability to upwind and downwind approaches. We included the following early phase design parameters for Trollvind, as communicated by Equinor on 14 December 2022 and updated on 16 January 2023: prospective wind farm footprint map; expected MW output of each turbine; number of blades on each turbine; turbine rotor radius; airgap (distance from the highest sea level to the bottom of the rotor area); maximum rotor blade width; rotation speed; and blade pitch (full details in *Appendix 2 Table A2.1*) and subsequently multiplied the probabilities by the number of turbines within the Trollvind development. If the agent collided, we assumed mortality and stopped the simulation. If there was no collision, we repeated this process for each day of the non-breeding season, assigning a new grid cell each day. We assigned each agent a day for arriving back at the breeding colony, thus indicating the end of the non-breeding season, by sampling a random date within the colony-specific mean ± 20 days.

At the end of the simulations, we extracted the number of collisions, and therefore mortality events, that occurred due to interaction with Trollvind.
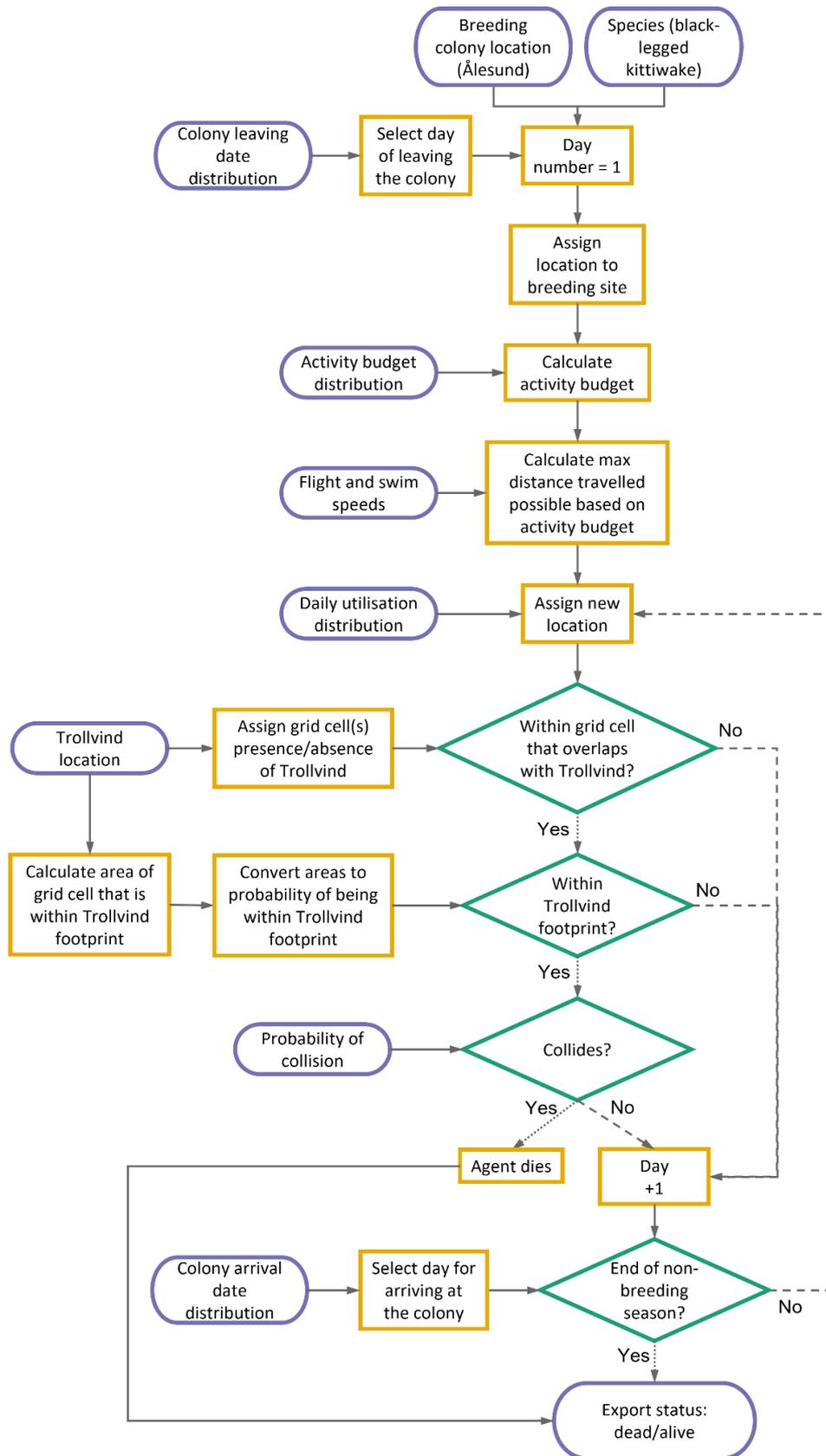
Figure 3. *Schematic of the agent-based model for kittiwake. Purple oval boxes are input and output data, yellow square boxes are processes and green triangular boxes are questions.*

2.3.4 Quantifying body mass-survival relationships

A key process linking sub-lethal effects of OWFs to changes in demographic rates (and thereby quantify population impacts) is the relationship between body condition and survival the year after. We quantified relationships between body mass at the end of the breeding season and adult survival from the end of the breeding season to the start of the following breeding season, using the methodology set out in Daunt et al. (2020). We developed joint statistical models for mass and size. The first component of the model was for mass and size (head and bill length) data, where each bird's mass at the end of the breeding season was estimated as a latent variable. The second component was a survival model for the capture-mark-resight (CMR) histories of each bird, which used the latent masses as a covariate, to estimate the probabilities of birds in the breeding population surviving from the end of one breeding season to the start of the next. A resighting model was fitted to account for annual variation in resighting rates and trap dependence. Separate models for mass and size were developed but with a correlation random effect (X_i and X_i') for each bird. An effect of sex was fitted to both models and sex allocation was estimated (assuming a Bernoulli distribution) to allow for missing sex information in the model ('Sex' being a variable which takes the value +0.5 for males and −0.5 for females). Only for the mass model (M) was a separate year-specific effect (Y_i) included as well as a linear effect (k) reflecting the trend in mass over the breeding period, in relation to day since the mean laying date per colony which was assumed to be constant across years ('Day'). The survival model (P) included effects of annual (Y) and bird-specific mass effects, imputed from the mass model. The resighting model (O) included an effect capturing trap dependence based on being observed the previous year. The model equations can be written as follows:

Mass model:

$$M_{ijk} = \mu_M + \alpha_{Msex}\,Sex_i + X_i + Y_j + \alpha_{Day,j}\,k + \varepsilon^M_{ijk}$$

Size model:

$$S_{ijk} = \mu_S + \alpha_{Ssex}\,Sex_i + X'_i + \varepsilon^S_{ijk}$$

Survival model:

$$logit(P_{ij}) = \mu_P + \beta_X\,X_i + \beta_Y\,Y_j + \beta_{Age}\,Age_{ij} + \varepsilon^P_j$$

Observation model:

$$logit(O_{ij}) = \mu_O + \gamma_O\,I(O_{ij-1} = 1) + \varepsilon^O_j$$

Where i = individual bird, j = year and k = day relative to laying date.

Models were fitted to mass, size and CMR data for multiple species and populations (see full results in *Appendix 3 Table A3.1*), which will be utilised in the full MARCIS project. Only results for guillemots from Sklinna were relevant for this report and therefore the results from this analysis are reported in the main text. Models were run in a Bayesian framework. Posterior distributions of parameters were obtained using Markov Chain Monte Carlo (MCMC) simulations implemented in jags (Plummer 2012) via the R package 'jagsUI' (Kellner 2015). Convergence of estimated parameters was assessed using the Gelmin-Rubin convergence statistic $\hat{R}$ for each stochastic node as modified by Brooks and Gelman (1998). Parameter estimates are summarised as posterior means with 95% credible intervals.

Model simplification was done by omitting parameters with $\hat{R} > 1.01$ (omitting the parameter with the largest $\hat{R}$ first) and parameters whose credible intervals included zero (omitting the parameter with the largest overlap first), unless the parameters were essential for the model. Essential parameters were μ_M, X_i, α_{Day}, μ_P, β_X and μ_O.

2.3.5 Seabird sensitivity to ocean warming

We measured the effect of annual variation in sea surface temperatures (SSTs) during the non-breeding season on adult survival. For this analysis, annual mean SSTs were calculated for the core foraging area in autumn (August-October) and winter (November-January). We took an

average of SST values from within the core foraging areas, based on the monthly core foraging distributions (see *Methods 2.3.2*). Individual encounter histories for each population were modelled with a Cormack-Jolly-Seber (CJS) framework. SST was included in survival models as a time-varying covariate. The model of survival (Φ) was $\Phi_{i,t} \sim \beta_1 SST_{aut,t} + \beta_2 SST_{win,t}$, for individual *i* in year *t*. Modelling was performed using the programme E-SURGE (Choquet et al. 2009b).

The goodness of fit (GOF) of the model to the data was assessed using the program U-CARE (Choquet et al. 2009a) to test for heterogeneity in the resighting probability (transient or trap-dependent effects). Transient effects refer to birds ringed (and captured) but never seen again, and trap-dependency effects refer to individuals that have a higher or lower probability of being re-sighted in the following years, and therefore being referred to as being trap-happy or trap-shy. Transience and trap-dependency are common issues in seabird CMR datasets and may have biological causes (e.g., high dispersal rates can lead to transience) or the topography of bird cliffs can mean individuals are easier to resight than others (a trap-happy effect). A goodness-of-fit test of the data showed that there were no transient effects for either kittiwakes from Ålesund ($N(0,1) = 0.47$ df = 10, $\chi^2 = 8.77$, p=0.55) or for guillemots from Sklinna ($N(0,1) = -1,98$, df = 12, $\chi^2 = 13.42$, p=0.43). However, there were trap-dependent effects for both kittiwakes ($N(0,1) = -5,07$, df = 9, $\chi^2 =23.74$, p=0.005), and guillemots ($N(0.1) = -7.93$, df=12, $\chi^2 = 79,14$, p = 1.51 x 10^{-11}). To account for trap dependence, we fitted a multi-state model with three states (Gimenez and Choquet 2010), including an unobservable state for non-resighted birds the previous year). An analysis of deviance was performed to test the ability each covariate has to describe significant variation in survival by using analysis of deviance tests comparing the covariate model with a constant and also fully time-dependent model (Lebreton et al. 2012). Model selection was performed using QAICc (Quasi-likelihood Akaike's Information Criterion corrected for small sample size and overdispersion) (Burnham 1998). The model with the lowest QAICc was considered the best. Models with scores of $\Delta QAICc \leq 2$ are strongly plausible, 4–7 less plausible and ≥ 10 are improbable.

2.3.6 Population viability analysis

We used a population viability analysis (PVA) to combine the results from the several approaches described above which quantified additional mortality due to 1) increased collisions and disturbance associated with the placement of Trollvind OWF and 2) increased sea surface temperatures in non-breeding areas under a high emissions scenario of global warming.

PVAs project the population size forward in time based on estimated demographic rates and impact scenarios (Boyce 1992). For each PVA, the baseline or impacted population was projected forward from 2021 - 2050 (i.e., 30 years), from an initial population size in year 2020. Initial population sizes were available as counts from monitoring data at each colony (*Table 2*). Age of first breeding was assumed to be 4 years for kittiwakes and 6 years for guillemots. Mean adult survival as estimated using the approach outlined in *Methods 2.3.5*. Mean productivity (fledged chicks per nest) over the study periods was estimated using generalised linear models. Immature survival was estimated using an integrated population model for both colonies (see Layton-Matthews et al. In review for methodological details). We used a stochastic PVA, where random changes in demographic parameters were included based on probability distributions defined from empirical data. Analyses were performed using the 'nepva' R package developed by (Searle et al. 2019). For kittiwakes from Ålesund (SEAPOP key site Runde), data on productivity (large chicks/nest from a sample of monitored nests) and CMR data were available from 2011-2021. For guillemots from Sklinna, CMR data were available from 2008-2020 and qualitative assessments of productivity were converted to quantitative values.

We ran PVAs for a baseline and impacted population of kittiwakes from Ålesund and guillemots from Sklinna and the two stressors, respectively: Trollvind OWF development and a high emissions ocean warming scenario (SST), to quantify their population-level impact. To assess the population impact of Trollvind OWF, additional mortality caused by the Trollvind OWF was implemented annual for the 'impacted' population, based on the results from *Methods 2.3.3-4*. Mortality was implemented each year from 2021 over the expected lifetime of Trollvind OWF, which was taken to be 25 years, as 25–30 years is a typical lifetime of an offshore wind farm (Rennesund 2020).

CMPI6-HighresMIP provides high resolution forecasts for oceanographic variables from 2014 until the year 2050, based on the high-emissions global warming scenario 'SSP 8.*5*', which is considered a worst-case, high emissions scenario. Median annual forecasts of SST were taken from CMPI6-HighresMIP model runs and were averaged for the two non-breeding seasons autumn and winter. To quantify the impact of ocean warming, through effects of SST on survival, we specified a 'baseline' population by assuming no trend in SST in the future (after year 2020), while the 'impacted' population was based on future annual SST based on the forecasts from CMPI6-HighresMIP.

The ratio of the baseline and impacted population growth rates gives a comparative and robust (i.e., insensitive to misspecification of parameters, see Jitlal et al. 2017) population-specific metric as the impact weight (μ) to the Halpern equation. An age class structured matrix model was parameterised for kittiwakes and guillemots. The predicted population trajectories through time were used as the population impact weight metric (μ) for each population (ecosystem component, *i*) and stressor (*j*) as the input to the Halpern equation. Following a review by Jitlal et al. (2017), we used the ratio of the median impacted population growth rates and median baseline population growth rates, averaged over the projection time period (2021-2050), as ratios are considered more robust metrics:

$$\mu_{ij} = 1 - \left(\frac{\text{median time averaged impacted population growth rate}}{\text{median time averaged baseline population growth rate}} \right)$$

We calculated the inversed ratio so that larger differences between the impacted and baseline population growth rates (i.e., a bigger impact) had a larger, positive impact weight. Values range from 0 to 1. In cases, where impacted populations have a population growth rate lower than the baseline (due to additional mortality) then values of μ are positive. Conversely, if impacted populations have higher median population growth rates than then baseline then values of μ are negative. We therefore took the absolute value of μ and distinguish positive and negative impacts visually in the App. All analyses were run in R (R Core Team 2022).

2.3.7 System architecture, user interface and functionality

The web application (*Figure 4*) was developed in the Google Earth Engine (GEE) Java API (Google Earth Engine 2023), and shared as a GEE web application (Google Earth Engine 2023), utilising the geospatial processing service powered by the Google Cloud Platform (Google Cloud 2023). The web application consumes all input data from the Google Cloud MARCIS asset (Google Cloud Assets 2023), which is synchronised with data from an internal file repository with the help of Google Colab (Google Colab 2023).
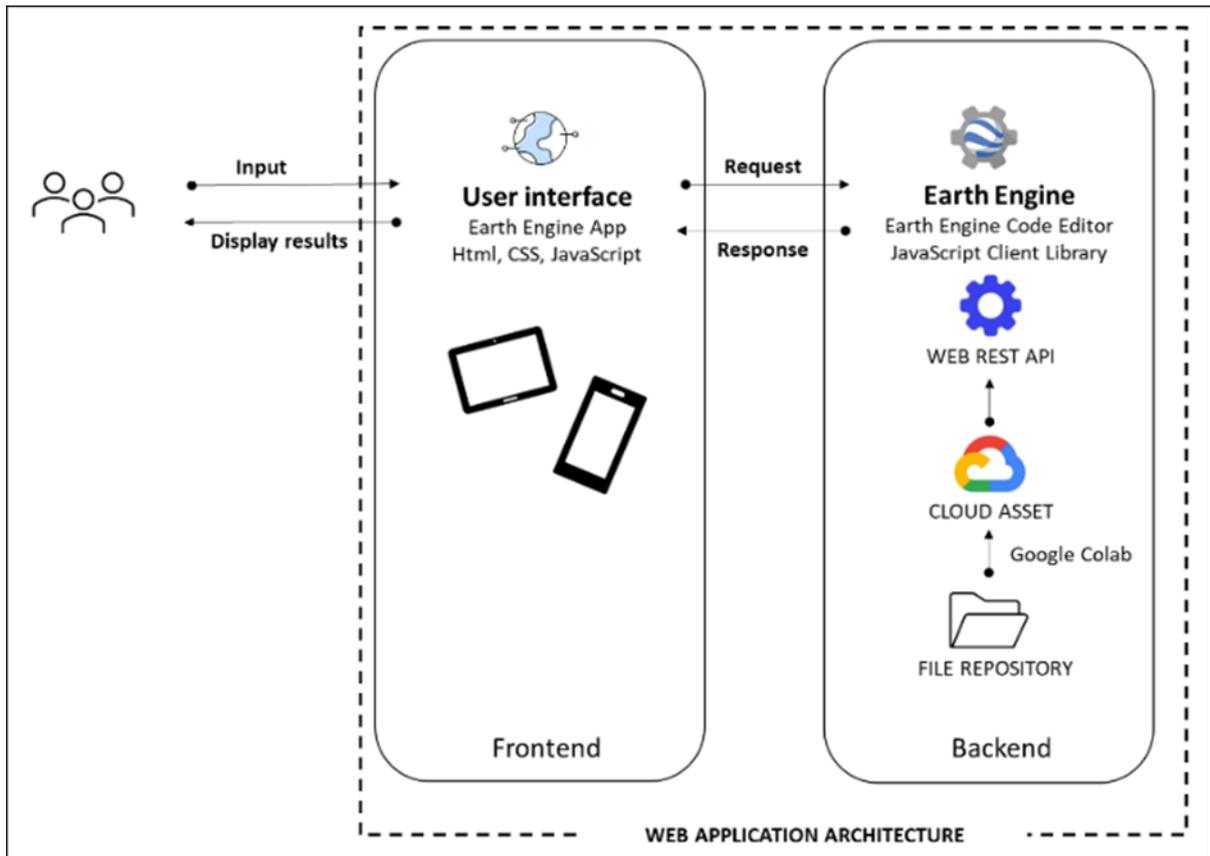
Figure 4. *Schematic diagram of system architecture for MARCIS web application.*

3 Results

3.1 Seasonal use and sensitivity of birds to Trollvind OWF

The highest total sensitivity found over the year is shown in *Figure 5*, i.e., each cell shows the highest seasonal sensitivity value of the combined sensitivity of all species of waterbirds present in the area (for a detailed description of the calculation of total sensitivity, Ollus et al. 2023).
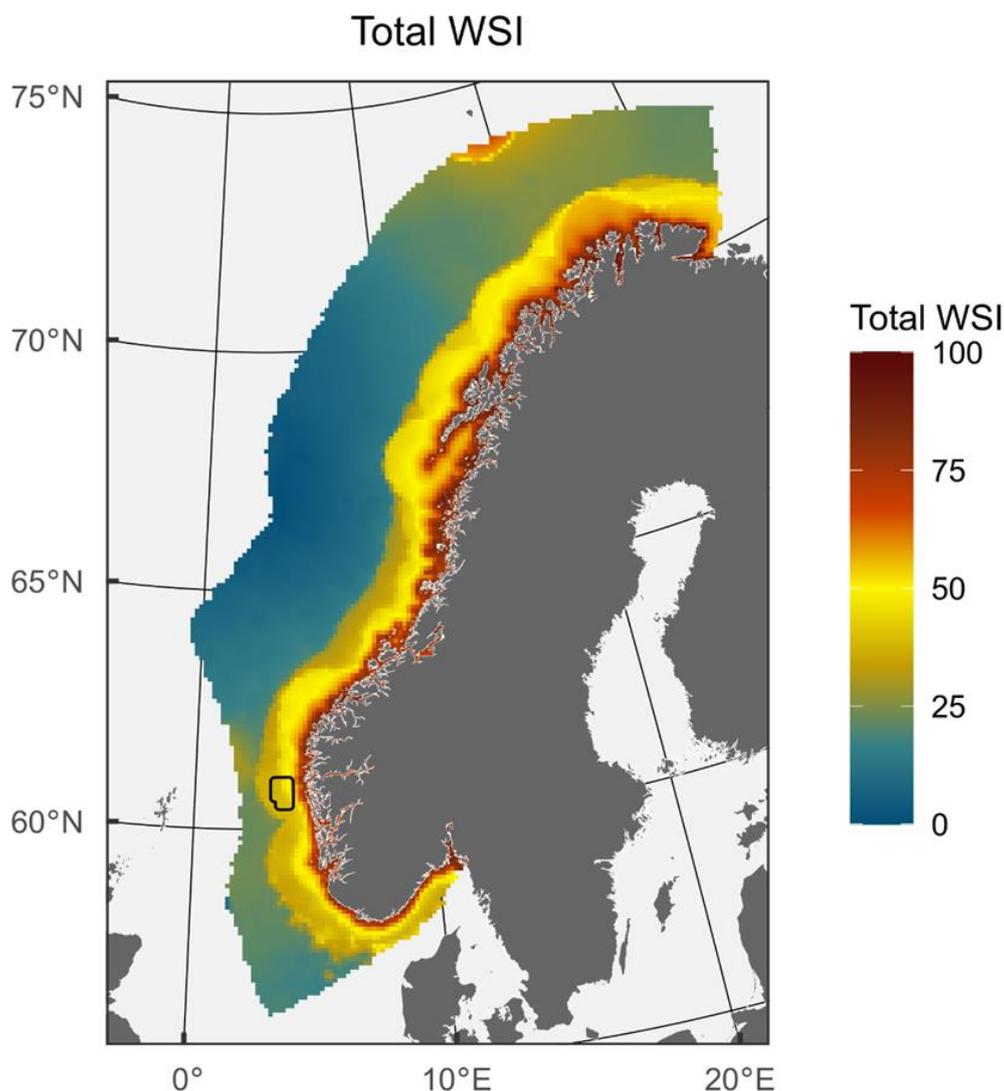
Figure 5. Total sensitivity of waterbirds to wind farms (total WSI) for the whole Norwegian Ex-clusive Economic Zone. The borders of Trollvind study area are marked in black. Each cell rep-resents the highest seasonal normalized and log-transformed sensitivity value. Figure from Ollus et al. (2023).

The models from the scoping study predicted that 49 out of 58 assessed species use the area during at least some part of the year. They further indicated below-medium to medium sensitivity of waterbirds to wind farms in the Trollvind area when species were viewed collectively. Highest seasonal sensitivity was found in summer, with the model showing above-medium values in the eastern and north-eastern parts of the area, i.e., the parts lying closest to the Norwegian coast. The number of species using the area was also shown to be highest in summer. Lowest sensi-tivity was found in winter, when below-medium sensitivity was found in the whole area.

The most sensitive species in the Trollvind area varied between seasons due to seasonal migration and changes in habitat use. In general, the Trollvind area is too far offshore to have a negative impact on sea ducks and waders. Sensitive species that also breed locally include coastal surface-feeding gulls and terns. These are lesser black-backed gull (*Larus fuscus*), common gull (*Larus canus*), herring gull (*Larus argentatus*), great black-backed gull (*Larus marinus*), and Arctic tern (*Sterna paradisaea*), which have all seen significant reductions in local populations during the last decades. Sensitive species that use the area as habitat but do not breed locally, include pelagic diving and surface-feeding seabird species. These are northern gannet (*Morus bassanus*), common guillemot (*Uria aalge*), black-legged kittiwake (*Rissa tridactyla*), northern fulmar (*Fulmarus glacialis*), European storm petrel (*Hydrobates pelagicus*), and razorbill (*Alca torda*). Great skua (*Stercorarius skua*) and brant goose (*Branta bernicla*) migrate through the area and are especially sensitive in autumn. Finally, divers that breed elsewhere but use the area as winter area were found to be sensitive. These were great northern diver (*Gavia immer*), red-throated diver (*Gavia stellata*), and red-necked grebe (*Podiceps grisegena*).

Kittiwakes are among the species that would be most sensitive to collision with wind turbines in the Trollvind area. This is because they are active flyers (also at night), flying at rotor height part of the time, and do not show much avoidance of wind turbines. They may even be attracted to offshore wind farms as breeding sites (Dierschke et al. 2016). Kittiwakes are most sensitive in summer, but they are present in the area and therefore sensitive to collision throughout the year. Conservation measures directed towards the Norwegian kittiwakes are of high importance because the species is listed as endangered on the Norwegian red list and because a large share of the European kittiwake population resides in Norway (Artsdatabanken 2021).

Guillemots are among the species that would be most sensitive to displacement by a wind farm in the Trollvind area. Its conservation status on the Norwegian red list is critically endangered and the fact that the species has a high survival rate among adults makes it highly sensitive to disturbance that can increase adult mortality and fecundity. Because the guillemot avoids anthropogenic constructions and activities, it is vulnerable to displacement by wind farms and therefore sensitive to habitat loss. Its high sensitivity and preference for the area makes it a particularly sensitive species to habitat loss in the Trollvind area in all seasons, especially in winter.

3.2 Potential impacts of OWFs on migratory birds

Following the LCIA, the groups with the highest disturbance and barrier PDF values, and there-fore estimated to be most impacted by disturbance and barrier effects from the Trollvind devel-opment, are migrating seabirds, waterfowl and raptors (see *Table 3* and *Figure 6*). The groups with the highest collision PDF values, and therefore most impacted by collisions, are migrating raptors, gulls, waterfowl, and owls. Migrating seabirds, waterbirds and waders have the lowest PDFs for collision, in contrast to their much higher susceptibility to the impacts of disturbance and barrier effects. The species most impacted across all three impact pathways (see cumulative rank in *Table 3*) are raptors and waterfowl, followed by gulls and seabirds. All migrating songbird groups ranked the lowest for combined impacts, although they do rank slightly higher for collision risks on their own. PDF values for all migration groups combined are highest for disturbance impacts, followed by barrier impacts and lowest for collision impacts (*Table 3*).

Mapping cumulative PDF values for all migration groups combined (*Figure 7*) highlights vari-ation in estimated impact across the area of, and surrounding, the Trollvind OWF. Higher cumu-lative PDF values are estimated for the south-west of the area, with slightly lower PDF values as you move north.

Table 3. Mean PDF values of estimated disturbance, collision and barrier impacts for Trollvind based on current turbine plans. Rankings go from 1-highest impact to 12-lowest impact. The cumulative rank is calculated as the ranked sum of the ranks for all three impact factors. (See Table A1.2 in Appendix 1 for lower and upper PDF values)

Migration Group	Disturbance		Collision		Barrier		Cumulative
	Mean	Rank	Mean	Rank	Mean	Rank	Rank
Corvids	7.35×10^{-5}	8	5.97×10^{-5}	6	3.34×10^{-5}	7	8
Gulls	1.34×10^{-4}	6	9.09×10^{-5}	2	9.62×10^{-5}	6	3
Herbivorous songbirds	6.03×10^{-6}	12	5.38×10^{-5}	8	1.70×10^{-6}	12	12
Insectivorous songbirds	6.36×10^{-6}	11	5.74×10^{-5}	7	4.49×10^{-6}	10	10
Non-passerines	2.87×10^{-5}	9	6.03×10^{-5}	5	1.91×10^{-5}	9	9
Owls	1.28×10^{-4}	7	7.25×10^{-5}	4	3.24×10^{-5}	8	5
Polyphagous songbirds	8.63×10^{-6}	10	5.24×10^{-5}	9	3.70×10^{-6}	11	11
Raptors	4.70×10^{-4}	3	9.75×10^{-5}	1	1.97×10^{-4}	3	1
Seabirds	2.37×10^{-3}	1	3.80×10^{-5}	12	5.87×10^{-4}	1	3
Waders	1.56×10^{-4}	5	4.66×10^{-5}	10	1.24×10^{-4}	4	5
Waterbirds	3.60×10^{-4}	4	4.11×10^{-5}	11	1.07×10^{-4}	5	7
Waterfowl	1.37×10^{-3}	2	7.94×10^{-5}	3	4.58×10^{-4}	2	1
Summed values	**5.11×10^{-3}**		**0.75×10^{-3}**		**1.66×10^{-3}**		
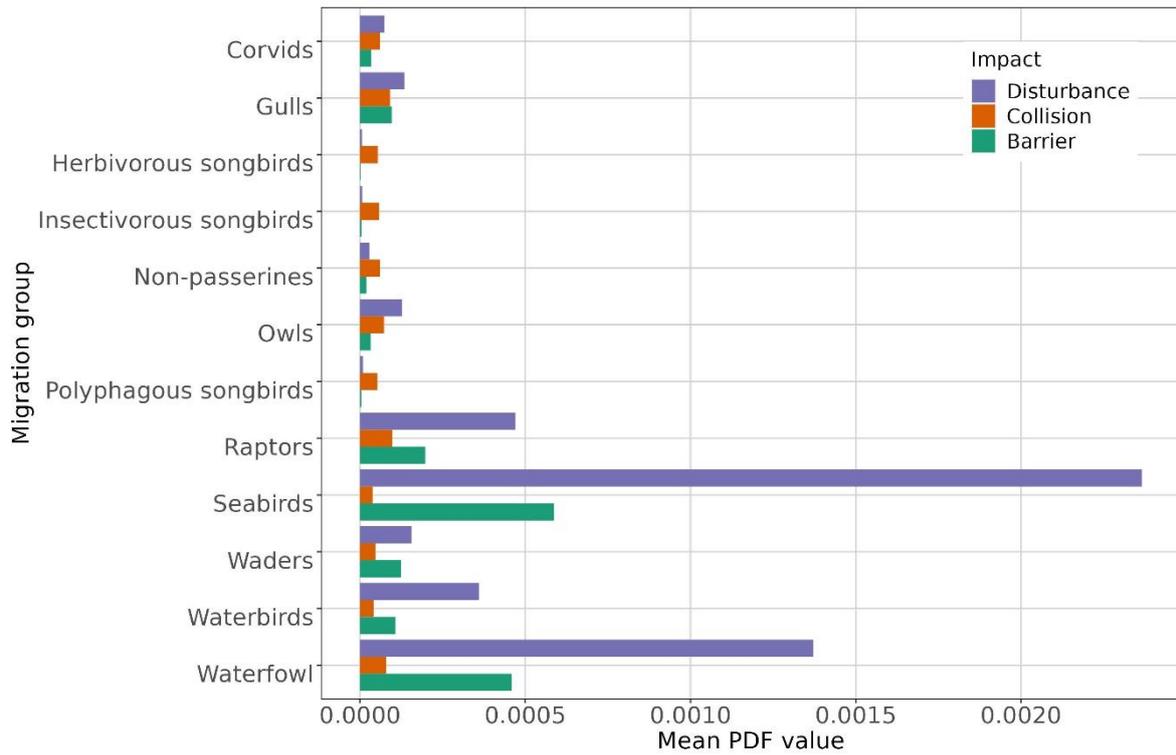
Figure 6. *Estimated impacts (mean potentially disappeared fractions of species values, PDF) due to disturbance, collision and barrier effects for each migration group in the Trollvind OWF. See Table A1.2 in Appendix 1 for upper and lower PDF values per group.*
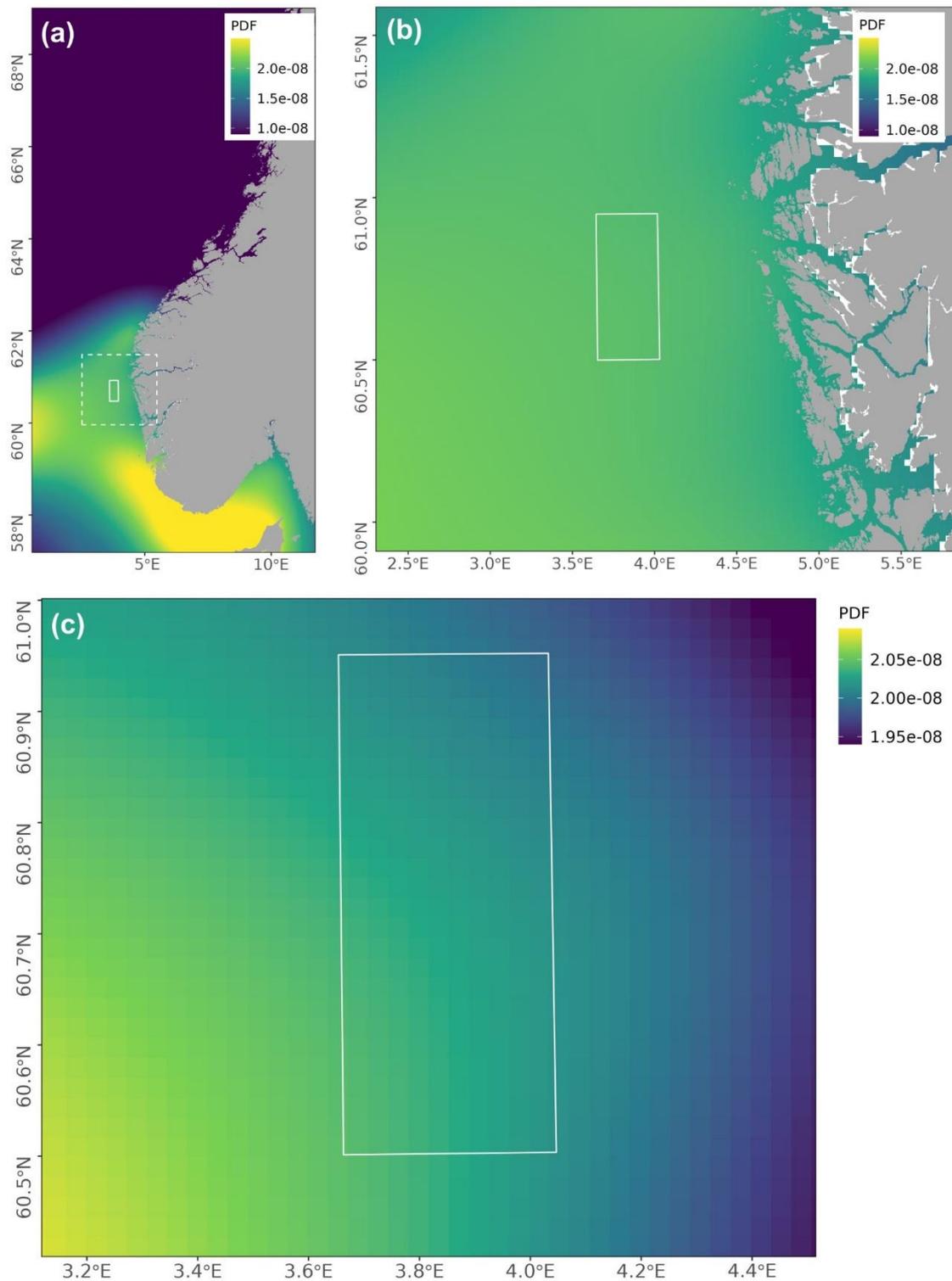
Figure 7. *Cumulative PDFs in a) Norwegian waters, b) the Troll area, and c) the Trollvind OWF for all impact pathways (disturbance, collision, and barrier effects) and migration groups combined. Values per 2x2km grid square are the estimated cumulative PDF values if one turbine was placed in each grid square. The white rectangle shows the boundaries of the Trollvind OWF and the dashed white lines in (a) show the extent covered by (b). Note that (c) is plotted on a different colour scale to (a) and (b) to show the variation in impact within the Trollvind OWF.*

3.3 Demonstration of MARCIS web application

A demonstration-version of the MARCIS App was developed for two seabird populations, guille-mots from the colony at Sklinna and kittiwakes from the Ålesund colony, which were selected based on the initial scoping report (Ollus et al. 2023, summarised in *Results section 3.1*) and based on availability of demographic and spatial data.

3.3.1 Seabird sensitivity to Trollvind OWF

For guillemots breeding at Sklinna, 7.1% of the agents overlapped with the Trollvind develop-ment area on at least one day during the non-breeding season (range of days of overlap per agent: 0-11). This resulted in displacement effects in 6.1% of the population (based on displacement probability inferred from Peschko et al. 2020) with effects occurring between 1-8 times per agent. Total non-breeding season energy expenditure across all agents was signifi-cantly different with the Trollvind OWF (mean = 555.22 MJ, SD = 8.35) and without (mean = 555.19 MJ, SD = 8.39; paired t-test: $t(999) = -6.50$, $p = < 0.001$), however the mean difference between the groups was extremely small (0.0324MJ lower with Trollvind; *Figure 8*). Across all individuals, end of non-breeding season mass was 2.33g lower with the Trollvind OWF develop-ment (SD = 11.4). Of the 6.1% of the population that experienced displacement effects, end of non-breeding season mass ranged from 16 to 131g lower than without the Trollvind OWF devel-opment, depending on the number of displacement events per individual (*Figure 9*).
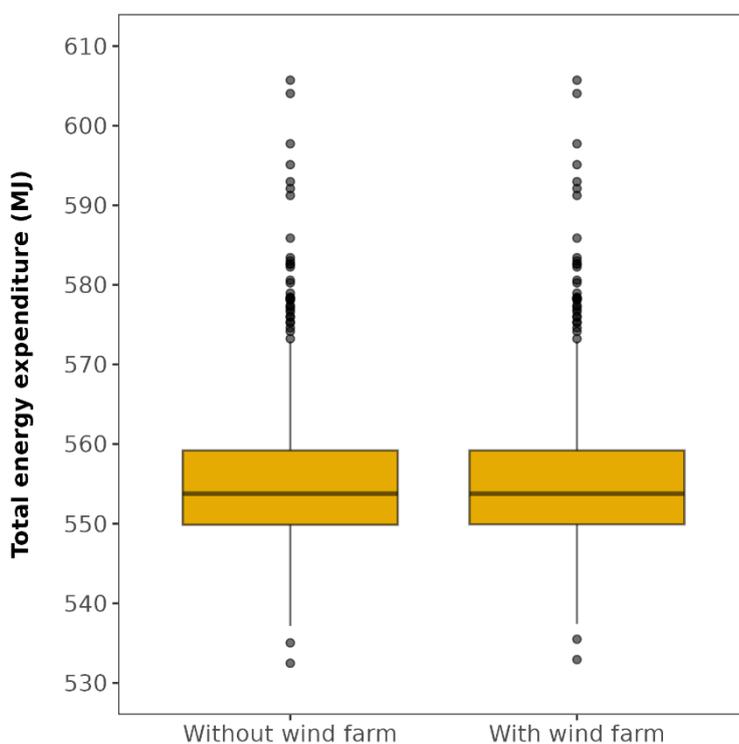
Figure 8. Total non-breeding season energy expenditure for each guillemot agent, without and with the Trollvind OWF.
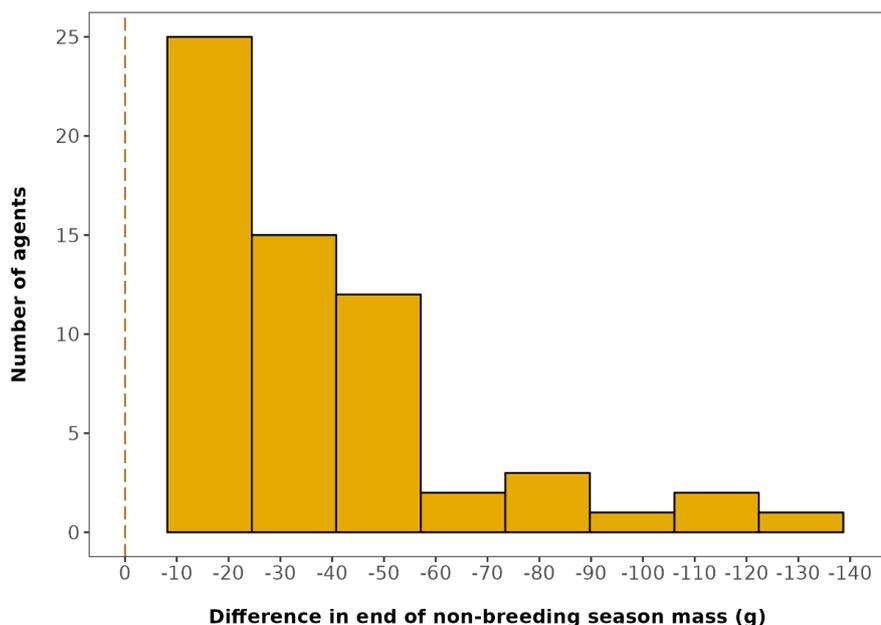
*Figure 9.* *The difference in end of non-breeding season mass (g) of guillemots with Trollvind presence for agents that experienced displacement effects. The dark yellow dashed line indicates zero.*

For kittiwakes breeding in Ålesund, 22.7% of agents overlapped with the Trollvind footprint on at least one day during the non-breeding season (18% once, 4% twice and 0.7% three times). However, the stochastic collision risk model predicted a very low risk of collision (mean = 0.0000224; SD = 0.00000185). Therefore, despite the relatively high overlap with the wind farm footprint we observed a very low rate of collision (< 0.001% of agents), which resulted in a negligible change in non-breeding season mortality.

3.3.2 Body mass-survival relationships

The model estimating body mass, size and survival as a function of body mass was fitted to mark-resight and morphometric data from several study populations and results (shown in *Appendix 3 Table A3.1*). Here, we report the results specifically for guillemots breeding at Sklinna. $\hat{R}$ values were ≤ 1.001 for all parameters indicating that convergence was achieved. The estimated effect of body mass on adult survival was 0.002 (95% credible intervals: -0.001, 0.006), indicating a positive effect of body mass at the end of the breeding season on adult survival the following year.

The estimated regression coefficient describing the body mass-survival relationship was used to estimate the change in survival resulting from a 2.33g decrease in body mass for guillemots due to Trollvind OWF (*Results 3.3.1*). Predicted adult survival as a function of body mass is shown in *Figure 10*. Mean survival, at the average body mass value for the end of the breeding season (918.35 g) was estimated to be 0.9187 (0.9047,0.9308). Survival decreased to 0.9184 (0.9042,0.9307) given a 2.33 g decline in body mass, reflecting a 0.024% reduction in survival. This was implemented as an annual reduction in survival in the PVA for the first 25 years of the PVA (2021-2045, average lifetime of an OWF), to estimate the impact of Trollvind OWF on the guillemot population.
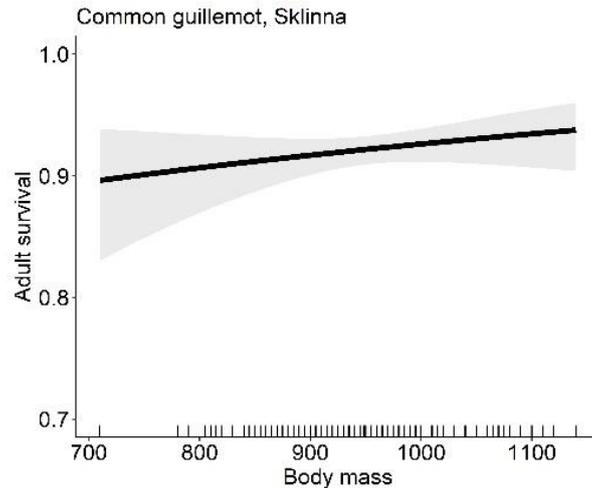
Figure 10. *Adult survival predicted as a function of body mass for guillemots from Sklinna. Rug shows measured body mass values of individuals. This relationship was used to quantify the change in survival over the non-breeding season caused by a reduction in body mass.*

3.3.3 Seabird sensitivity to ocean warming

Annual, spatially averaged sea surface temperature (SST) was calculated for autumn and winter periods in the specific seabird populations core distributions (*Figure 11a-b*), to include as covariates in the survival models. For guillemots, the core foraging area in autumn was in the southern Barents Sea (*Figure 11a*). The core wintering area was in the Norwegian Sea, close to the breeding colony at Sklinna (*Figure 11b*). There was an increasing temporal trend in SST over the period with survival data (2008-2022) in autumn (trend = 0.10, SE = 0.05, p-value = 0.085) and a weaker trend in winter (0.02, 0.03, p-value = 0.47) foraging areas. However, both trends were strongly significant by 2050 (*Figure 11c*). For black-legged kittiwakes breeding at Ålesund, the core autumn foraging area chosen was east of Greenland (*Figure 11b*) and the core wintering area was in the Grand Banks region south of Greenland (*Figure 11b*). There was a non-significant increasing temporal trend in autumn SST over the study period 2011-2021 (trend = 0.08, SE = 0.06, p-value = 0.20) and a weak negative trend in the winter foraging area (trend = -0.03, SE = 0.04, p-value = 0.50). However, by winter SST has a strong positive trend by 2050 (*Figure 11d*).

Common guillemots - Sklinna　　　　**Kittiwakes - Alesund**
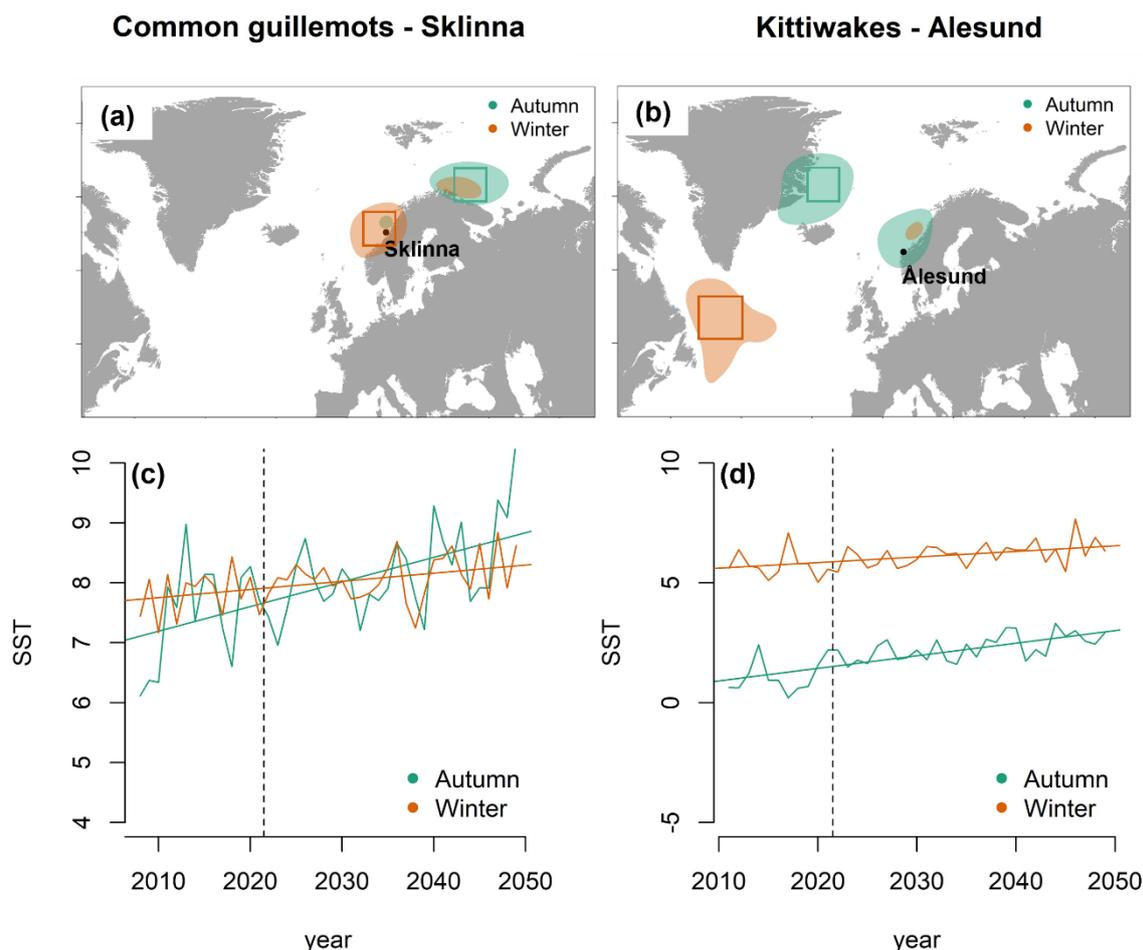
*Figure 11. Non-breeding distributions, represented as 50% kernel contours of birds from each population in autumn (green, August−October) and winter (orange, November−January) for guillemots from Sklinna (**a**) and kittiwakes from Ålesund (**b**). Points show locations of the breeding colony and boxes show approximate area where SST values were extracted from. Annual mean sea surface temperature (SST) values extracted from a focal area within the autumn and winter core foraging areas (lower panels) for guillemots breeding in Sklinna (**c**) and kittiwakes breeding in Ålesund (**d**).*

For guillemots, autumn and winter SST showed a moderate correlation (0.35, year 2008-2021), so models including SST_{aut} and SST_{win} were run as separate, single covariate models. Annual SST fluctuations in the autumn core area had a tendency for a negative effect on adult survival and explained 4% of the annual variation in adult survival (*Table 4*). For the Ålesund kittiwake population, the correlation between SST in autumn and winter foraging areas was low (-0.001) and so a model with both covariates were included as candidate model. In both seasons, SST had a negative effect on survival rates, where autumn SST had a stronger effect. Annual fluctuations in SST in the autumn and winter core foraging areas explained 20% of the annual variation in adult survival (*Table 4*).

Table 4. *Estimated effects of autumn SST and winter SST over the study periods on adult survival rates for the two study species/populations. For guillemots, there was a correlation between autumn and winter SST and so only the slope and R^2 for best single-covariate models are shown.*

Study population	Study period	SST_{aut} slope (95% CIs)	SST_{win} slope (95% CIs)	Model R^2
Kittiwakes, Ålesund	2011-2021	-2.84 (-6.73, 1.05)	-0.05 (-4.70, 4.61)	20%
Guillemots, Sklinna	2008-2021	-0.97 (-3.51, 1.58)		4%

Based on the estimated coefficients for both species (*Table 4*), predicted adult survival decreased as a function of autumn SSTs for guillemots (*Figure 12a*), and as a function of both autumn and winter SSTs for kittiwakes (*Figure 12b,c*) until year 2050 (the final year of the population viability analysis). Predicted survival rates were used in the PVA to quantify the impact weight of ocean warming based on the forecasted SST values following a high emissions global warming pathway (SSP8.5). For guillemots, survival in year 2020 was estimated to be 0.912 (0.897, 0.925) and in 2050 to be 0.872 (0.767, 0.934), i.e., a 4.4% decrease in survival. For black-legged kittiwakes, survival in year 2020 was 0.829 (0.750, 0.887) and 0.799 (0.598, 0.914) in 2050, i.e., a 3.6% decrease in survival over the projection period. The annual, additional mortality due to ocean warming was calculated as the difference in survival from year 2021 to 2050, divided be the number of projection years (30).
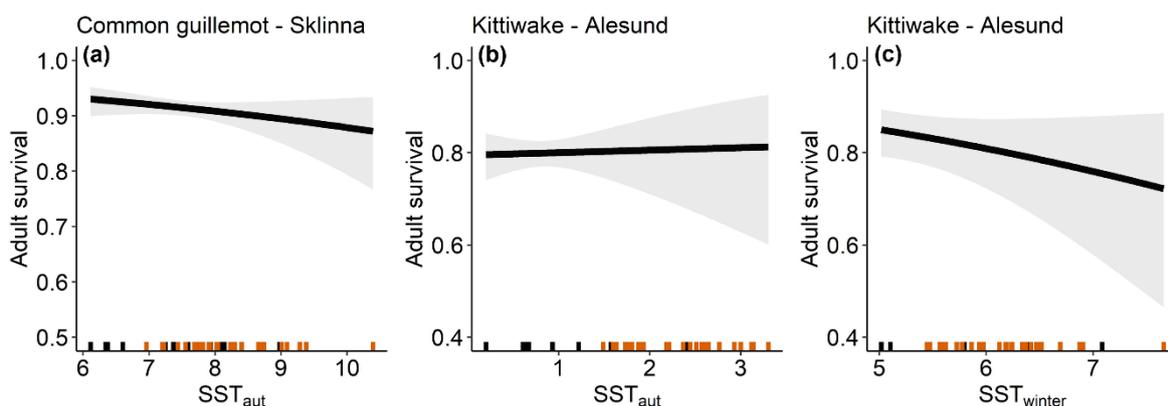

Fig. 12. Predicted annual adult survival rates, for guillemots (a) and kittiwakes (b,c), based on SST values from first year of demographic data until year 2050, based on the modelled regression coefficients using historic and forecasted SST data. Rugs show actual values of SST, black values represent SST for the period with mark-recapture data (2008-2021) and red values represent SST for the period after (2021-2050).

3.3.4 Population impacts of ocean warming and Trollvind OWF

Four population viability analyses (PVAs) were run, two per study population: guillemots (Sklinna) and kittiwakes (Ålesund), and two per stressor: Trollvind OWF and ocean warming. Additional mortality associated with ocean warming in autumn and winter foraging areas (summarised in *Appendix 3 Table A3.2*), was implemented for the 'impacted population' in the PVA over the projection period (2021-2050) and compared with the baseline (non-impacted) population trajectory. Additional mortality associated with Trollvind OWF in autumn and winter foraging areas, was implemented for the 'impacted population' in the PVA over the expected lifetime of the Trollvind OWF (2021 – 2045).

The impact metrics (summarised in *Appendix 3 Table A3.2*) for Trollvind OWF and ocean warming for the MARCIS tool demonstration, were calculated by comparing the baseline population and impacted populations from the PVA simulations, as one minus the ratio of the median time-averaged population growth rate from 2021-2050. The difference in the impacted population from the baseline population is due to the reduction in adult survival as a result of 1) Trollvind OWF and 2) ocean warming. Since there were no estimated collisions of kittiwakes due to Trollvind OWF and therefore no additional mortality, the impacted and baseline populations for Trollvind OWF were similar but not equal to zero, due to uncertainty introduced through the PVA simulations (*Figure 13a*) and therefore the impact weight was negligible. For guillemots, the small reduction body mass caused by the Trollvind OWF, which translated into a reduction in survival, led to the impacted population having a marginally lower trajectory (*Figure 13c*) and thus a small impact weight. Projected ocean warming, following a high emissions scenario, caused an increase in mortality in both populations. Due to the larger reduction for guillemots, through effects

of autumn ocean warming, this led to a larger difference between the impacted and baseline populations, compared to kittiwakes and thus a larger impact weight (*Figure 13c,d*).
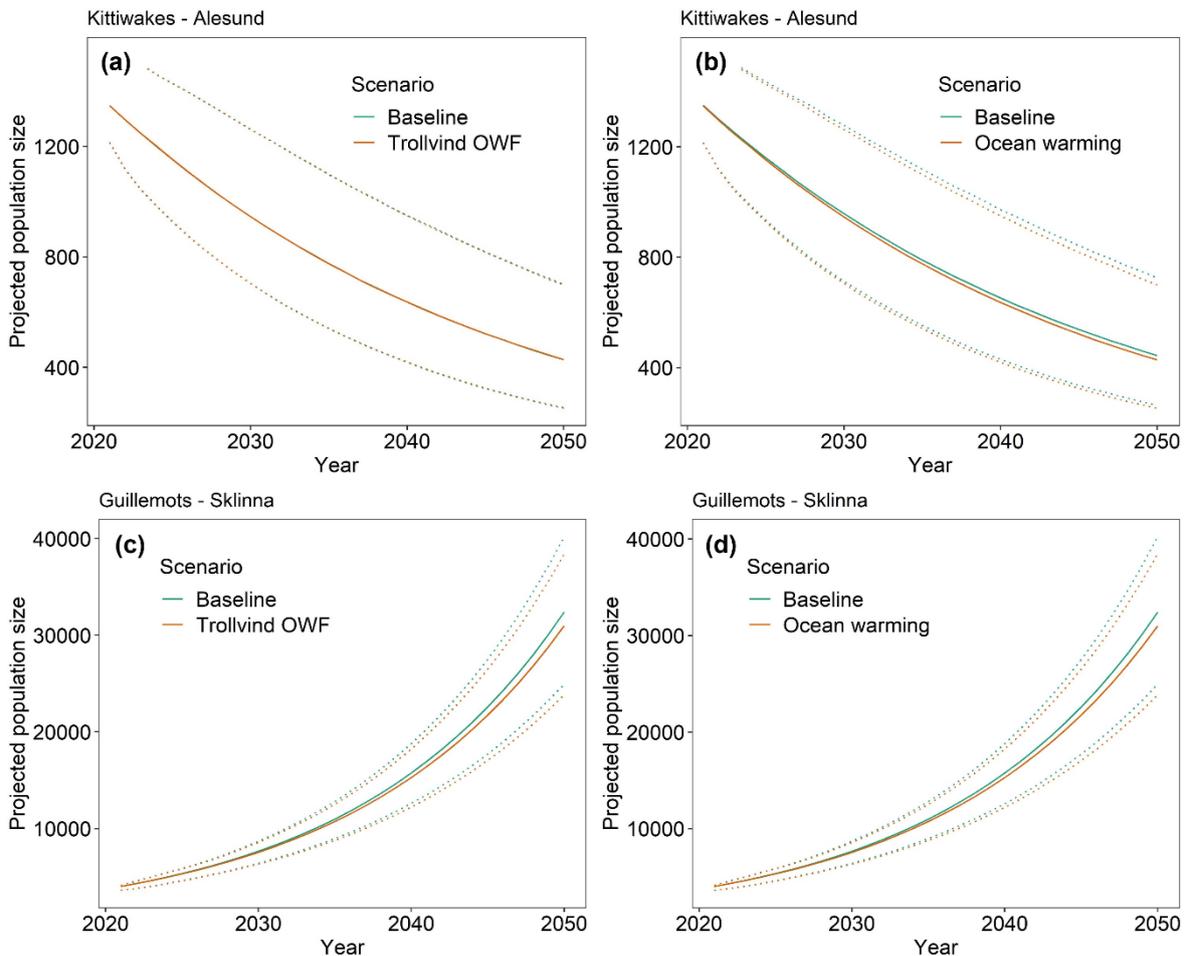
Figure 13. Projected population sizes for baseline and impacted populations, for kittiwakes breeding at Ålesund colony (a,b) and guillemots breeding at Sklinna (c,d), for the two stressors: Trollvind OWF and ocean warming. The first year of the population projection is 2021 and it runs until 2050, where mortality due to Trollvind OWF was implemented each year until 2045 and mortality due to ocean warming was implemented each year until 2050. NB: lines both projections in (a) overlap since there was no mortality associated with Trollvind OWF for kittiwakes.

3.3.5 Demonstration of the MARCIS tool

The beta-version of the web application helps to calculate and visualise both individual and cumulative impacts of Trollvind OWF and ocean warming within the Trollvind project area (*Figure 14*). First the web application prompt the user to select species (guillemot or kittiwake), colony (Sklinna or Ålesund (Runde)), and month before clicking the "Run assessment" button". The corresponding results include individual/cumulative impacts of ocean warming and the Trollvind OWF (based on the Halpern formula) and zonal statistics indicating minimum, mean, and maximum impact levels. All maps can be turned on and off from the map layer list. When the user wants to do a new assessment, he or she can click the "Reset panel setting"-button and continue from the beginning.
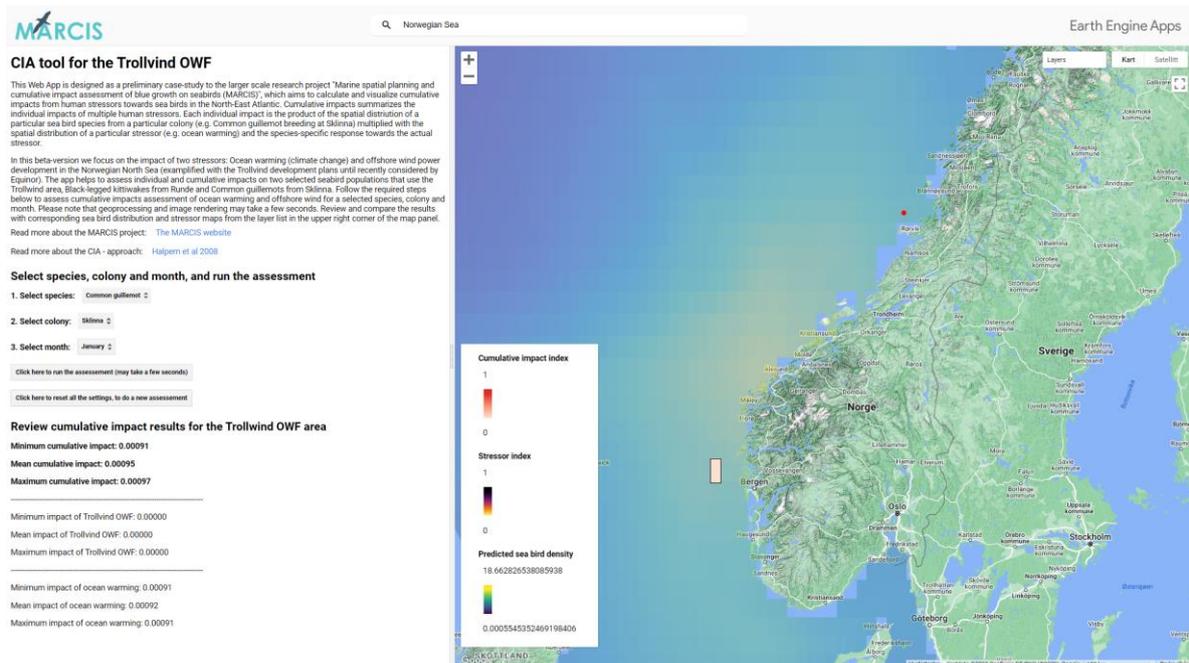
Figure 14. An image from the MARCIS web-app showing the beta-version of the MARCIS web-app user interface.

The user can inspect seabird distribution maps for a selected species, colony, and month (*Figure 15* and *16*), stressor maps (*Figure 17* and *18*), individual impacts and cumulative impact maps (*Figure 19, 20, 21, 22* and *23*).
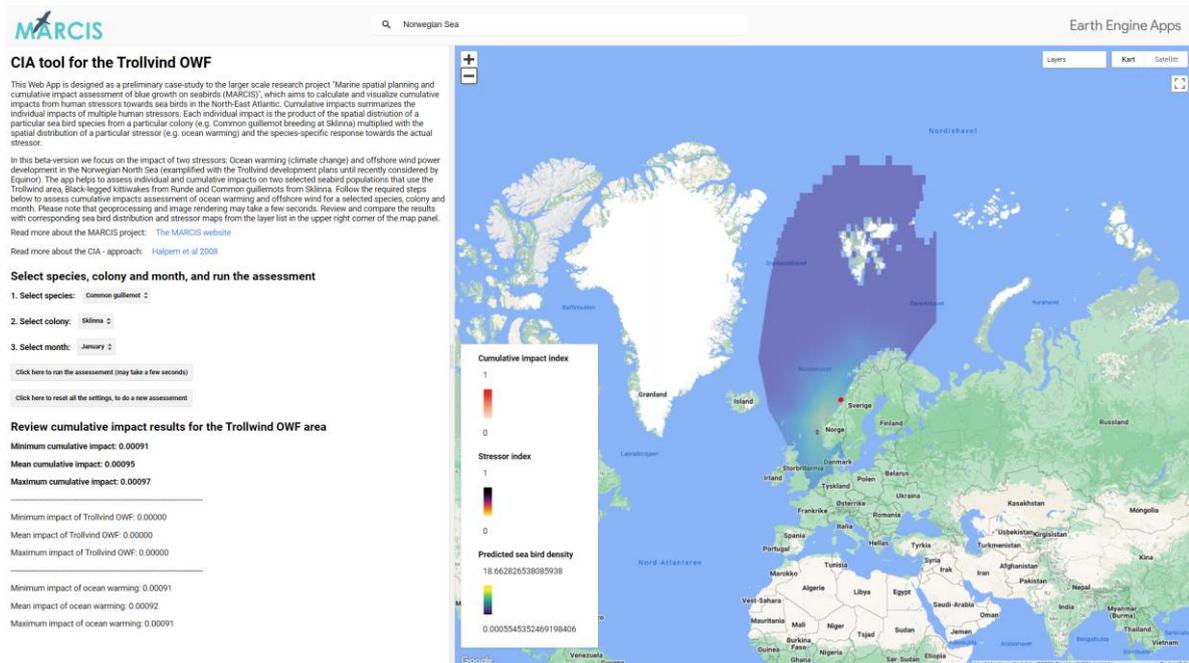
Figure 15. An image from the MARCIS web-app showing the distribution of guillemots from Sklinna in January.
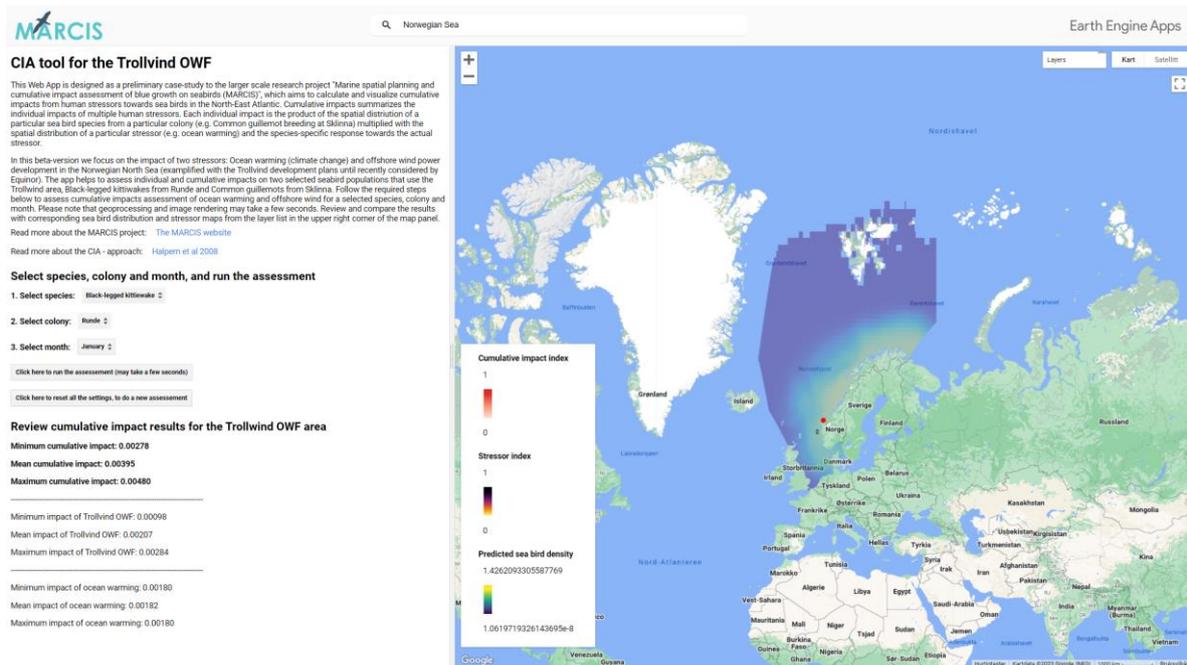
Figure 16. *An image from the MARCIS web-app showing the distribution of kittiwakes from Åle-sund (Runde) in January.*
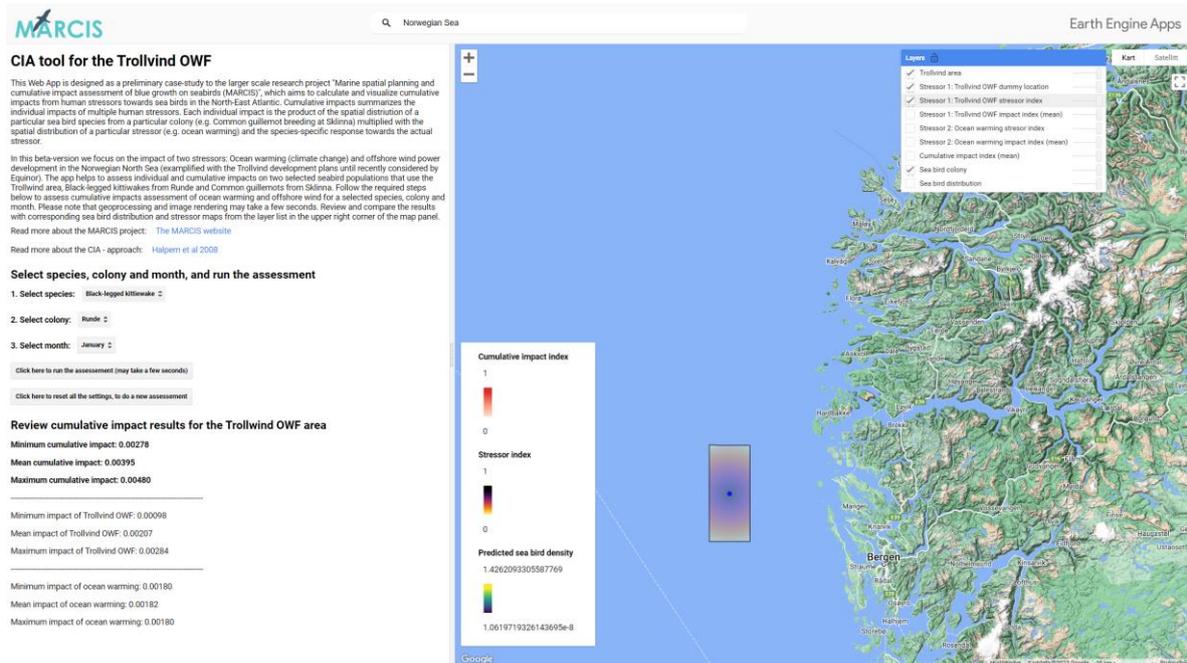
Figure 17. *Stressor 1: An image from the MARCIS web-app showing the Trollvind offshore wind power plant (dummy location marked with a blue point symbol) and the potential stressor intensity (ranging from 1 at the OWF location and decreasing to 0 at the border of the Trollvind area).*
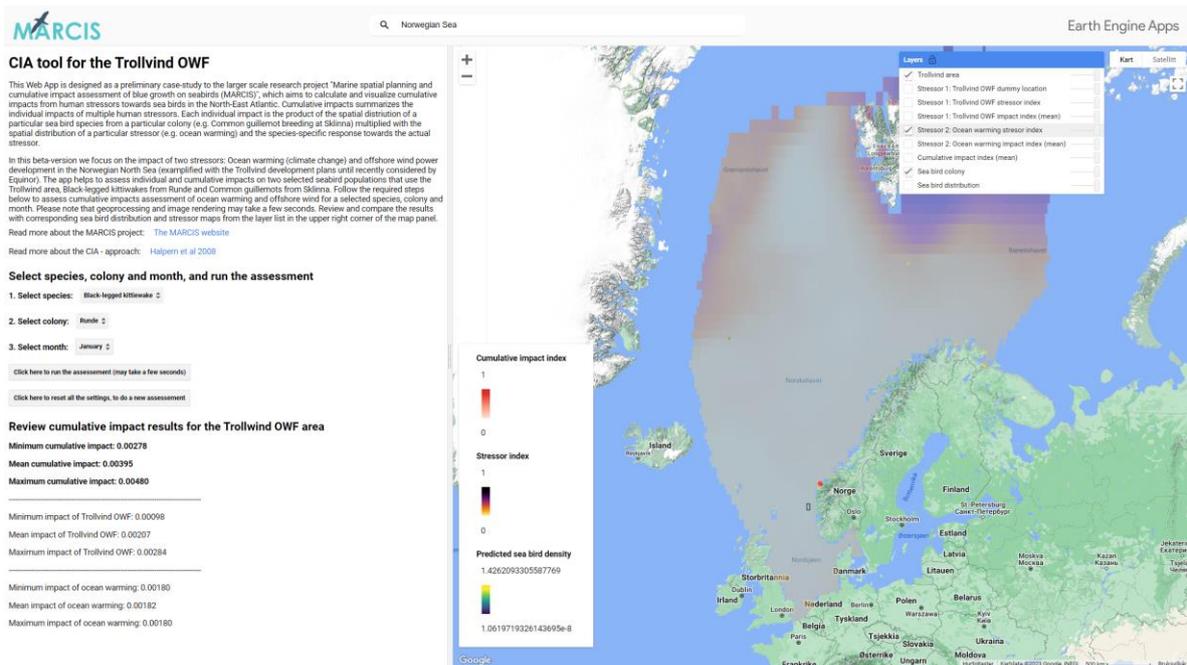
Figure 18. *An image from the MARCIS web-app showing the stressor 2: ocean warming in January. The stressor intensity ranges from 0 (low ocean warming) to 1 (high ocean warming).*
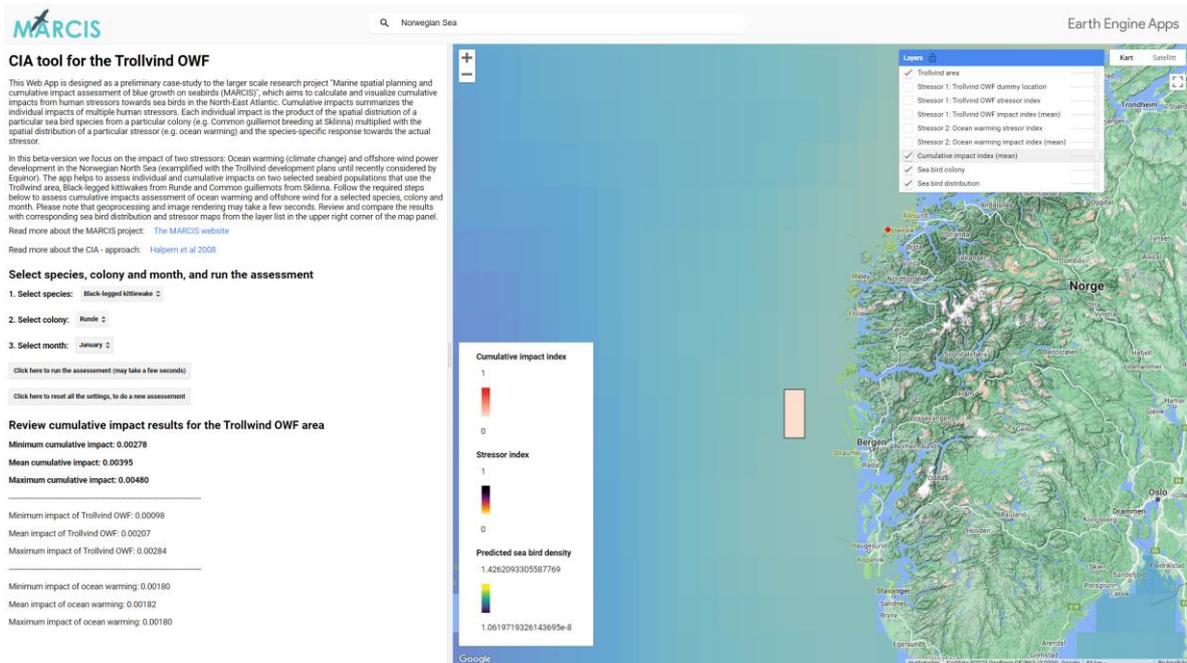
Figure 19. *An image from the MARCIS web-app showing the mean cumulative impact on kittiwakes from ocean warming and the Trollvind OWF in January.*
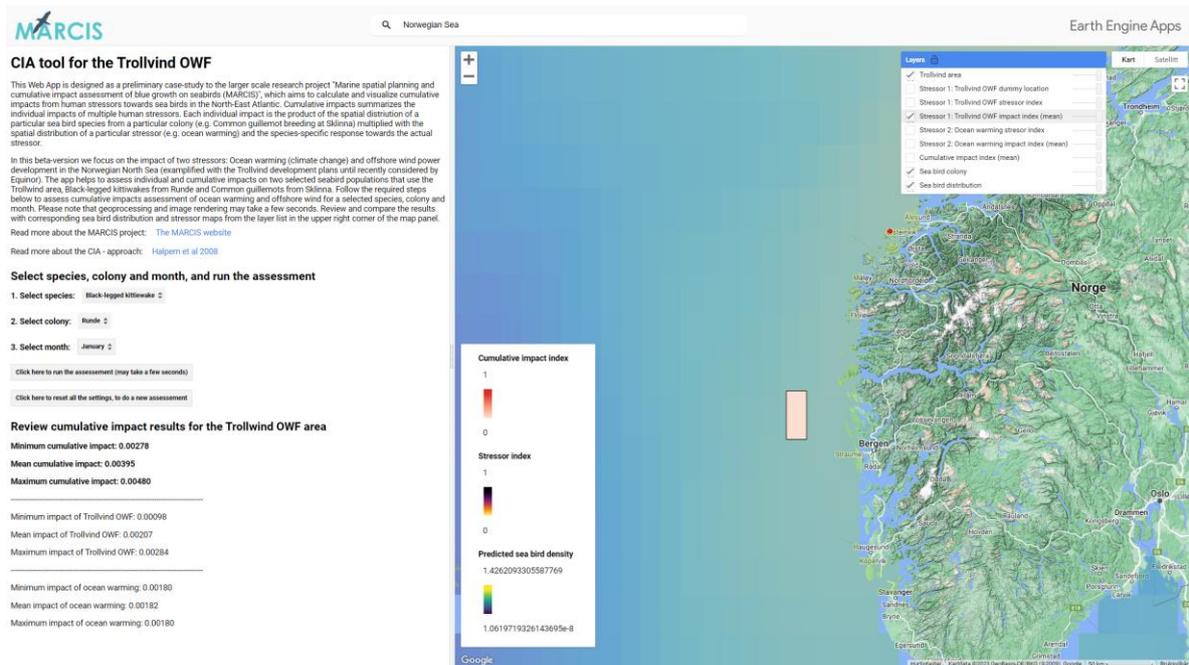
Figure 20. *An image from the MARCIS web-app showing the mean impact on kittiwakes from Trollvind OWF in January.*
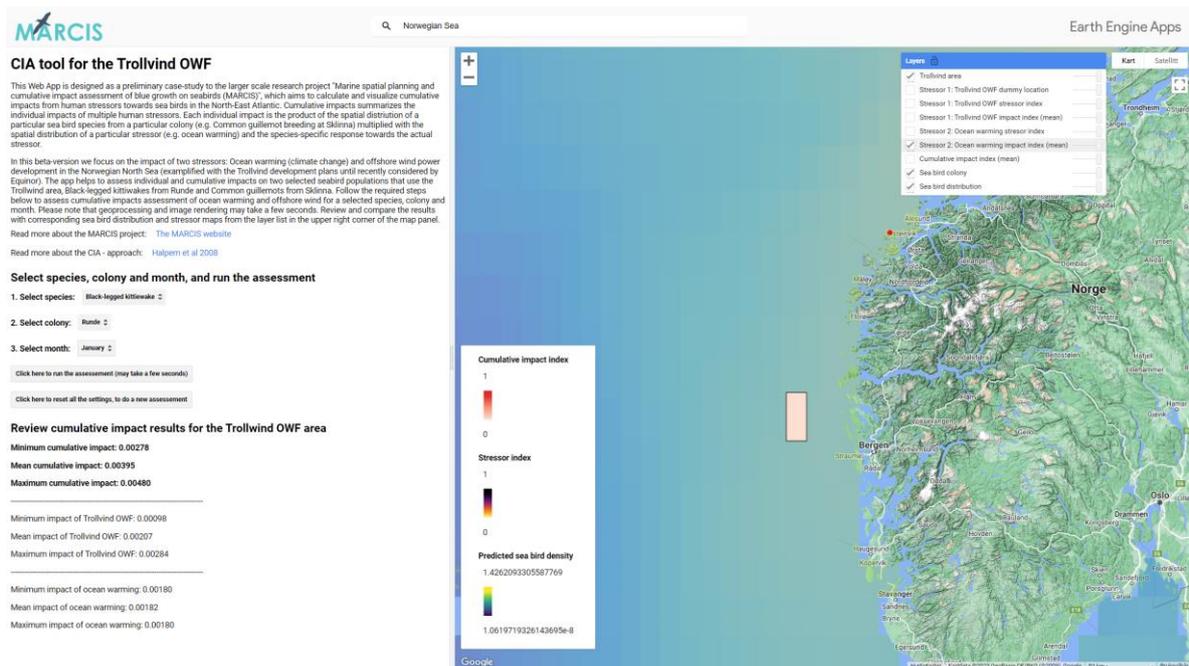
Figure 21. *An image from the MARCIS web-app showing the mean impact on kittiwakes from ocean warming in January.*
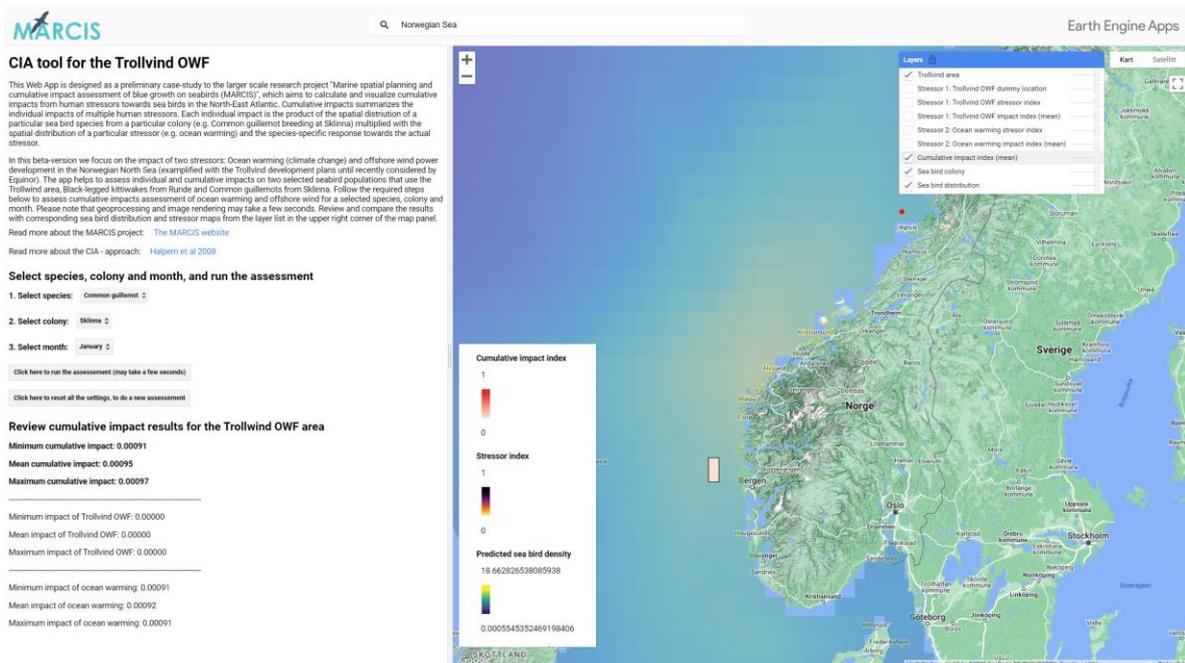
Figure 22. An image from the MARCIS web-app showing the mean cumulative impact on guillemots from ocean warming and the Trollvind OWF in January.
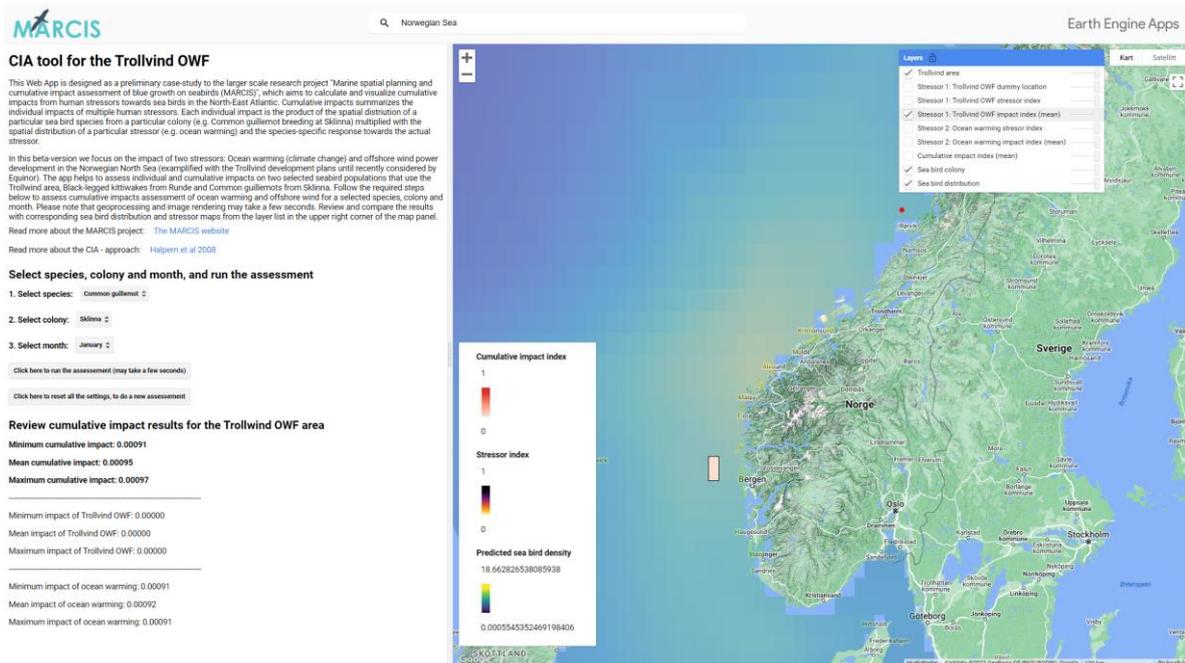
Figure 23 An image from the MARCIS web-app showing the mean impact on kittiwakes from Trollvind OWF in January.
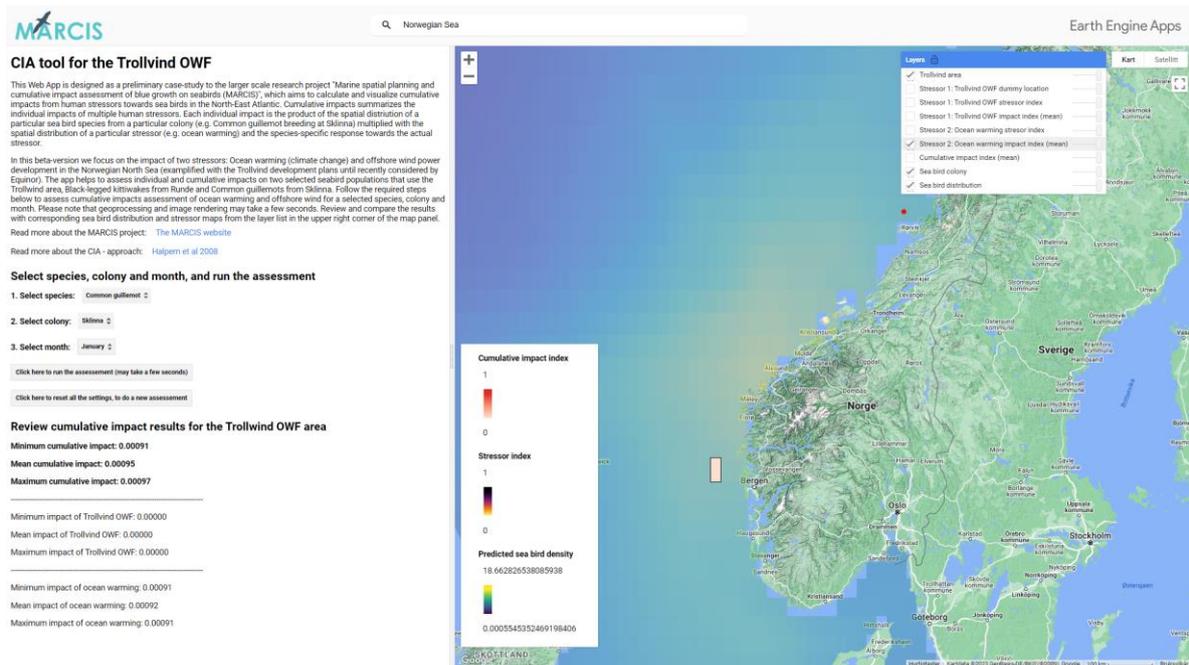
Figure 24. *An image from the MARCIS web-app showing the mean impact on kittiwakes from ocean warming in January.*

4 Discussion

The results summarised in this report provide an overview of which migratory bird species utilise the area proposed for the OWF development 'Trollvind' and to what extent they are at risk from this potential development. Such knowledge is essential to assess birds' vulnerability in this pre-development phase as well as assessing what additional information is needed for future assessments. We present the results from a demonstration of the MARCIS web application, which was applied to a case study of two populations considered at risk from the proposed OWF, a kittiwake population and a guillemot population. In Norway, these species are listed as endangered and critically endangered, respectively (Artsdatabanken 2021). The results provide a demonstration of how this tool could be used to inform marine spatial planning. Overall, results from the initial scoping report showed a below-medium to medium sensitivity of waterbirds to wind farms in the Trollvind area. However, there were strong seasonal differences showing above-medium values in summer in eastern and north-eastern parts of the area. Results from the analysis of the likely occurrence of migratory birds in the Trollvind OWF area and their vulnerability to OWFs revealed that bird groups differ in terms of the risk of the development. Migrating raptors, gulls, waterfowls and owls were at greatest risk of collision, while migrating seabirds, waterbirds and waders had a higher risk of displacement and/or barrier effects. We then estimated the cumulative impact of two stressors (Trollvind OWF and ocean warming) on two study populations (one black-legged kittiwake population and one common guillemot population) based on the the Halpern equation, which was visualised in the demonstration of the MARCIS App. The impact of Trollvind OWF during the non-breeding season was negligible for both populations, while ocean warming in non-breeding areas had a larger population impact, particularly for guillemots. However, it is important to keep in mind that this demonstration is a case study of only two populations and should be interpreted in the larger context of the results from the scoping study and bird migration studies, which reflects the extensive range of species using this area at different times of year, and thus the potentially broader impact of the development.

4.1 Seasonal use and sensitivity of birds to the Trollvind OWF

In the scoping study (Ollus et al. 2023), a sensitivity index based on best available data and knowledge was used to assess the relative sensitivity of 58 species of seabirds and other waterbirds to wind farms in the Trollvind area in relation to other areas in the Norwegian Exclusive Economic Zone (EEZ). The index does not quantify the vulnerability of bird populations but is a relative measure that can be used to compare sensitivity among areas, seasons, and species. For the EEZ at large, bird sensitivity was found to be highest along the coast and decrease offshore. Furthermore, in offshore areas, continental seas (e.g., Barents Sea and North Sea) showed higher bird sensitivity than deep-sea areas in the Norwegian Sea. Compared to the rest of the EEZ, total bird sensitivity in the Trollvind area ranged from below-medium to medium. In general, waterbirds are most sensitive in summer and in the eastern and north-eastern parts of the Trollvind area, though there is spatial and temporal variation in sensitivity between species.

The Trollvind area is used by several bird species; of the 58 species considered in the scoping study, 49 were predicted to use the area in one or more season. Because of species' migration habits and differences in area-use during the breeding and non-breeding seasons, seasonal variation in species-specific distribution and sensitivity to wind farms within the Trollvind is important for several species. Sensitive species include both locally breeding species and species that breed elsewhere and use the area for migration or as a non-breeding habitat (e.g., for overwintering). Sensitive species that breed locally include coastal surface-feeding gulls and terns whose local populations have been heavily reduced during the last decades, with poor prey availability likely being the main cause (Fauchald et al. 2015). Additional stressors causing increased adult mortality could hamper the restoration of these populations (i.e., due to cumulative effects). Sensitive species that use the area but do not breed locally include pelagic diving and surface-feeding seabird species, such as the endangered black-legged kittiwake and the critically endangered guillemot that have been studied more closely for this report. Also, divers that breed elsewhere but use the Trollvind area, as winter area were found to be sensitive to wind farms.

Limitations in available data and knowledge may affect the results from the scoping study. The species distribution models used to calculate habitat preference, and ultimately for sensitivity estimation in the EEZ, include some uncertainty arising from the uneven spatial and temporal coverage of observation effort, however we statistically corrected for this uncertainty following the approach of(see Ollus et al. 2023). Furthermore, the knowledge available on the size and vulnerability of most bird populations is limited, particularly knowledge of their sensitivity to wind farms. These factors directly affect species sensitivity indices, and changes in them may have large effects on our predictions of bird sensitivity. Due to limited data, we were not able to model the distribution of all species that potentially use the study area. Furthermore, for some species we were not able to predict the distribution during every season, either due to a seasonally low abundance in the area, or due to a seasonal lack of observation effort. It is likely that this limitation would significantly affect our result, as species for which there are few observations are typically rare. However, some of these rare species could potentially be sensitive to wind farms, suggesting that the results could change if these species were included.

4.2 Potential impacts of OWFs on migratory birds

The most obvious consequence of offshore wind developments is bird collisions, representing a source of direct mortality (Drewitt and Langston 2006). Most studies report relatively low levels of collision mortality, primarily focused on large birds found by carcass surveys in post-construction studies (Drewitt and Langston 2006, Falkdalen et al. 2013, Stokke et al. 2020). Most studies of wind power induced mortality have been conducted onshore, mainly because there is hitherto no standardised method of registering collisions offshore, and most offshore estimates therefore originate from collision risk models, as is the case for seabirds (Masden and Cook 2016). Based on the LCIA, the functional groups with the highest collision PDF values and therefore expected to be most affected by collisions were, raptors, gulls, waterfowl and owls. There has been much concern about raptors in relation to onshore wind energy development, in Norway particularly on Smøla wind farm in the coast of central Norway (Stokke et al. 2020), but also worldwide (Drewitt and Langston 2006, Kuvlesky et al. 2007). Raptors are particularly vulnerable to anthropogenic stressors due to their longevity, low reproductive rates, and preference for thermal soaring during foraging trips as well on migration. Despite limited possibilities for thermal soaring across open sea, raptors are also prone to collisions offshore during migratory crossings. Waterfowl is a functional group with migratory trajectories spanning the entire Norwegian coast, connecting northerly breeding grounds with wintering areas on the British Isles, the Netherlands, Belgium, Germany and Denmark. In contrast to waterfowl passing on migration, gulls are notorious for their attraction to anthropogenic offshore structures for roosting, which naturally increases the risk of collision mortality.

Disturbance and barrier effects can affect birds to in the long-term, i.e., affecting future survival and reproduction on a population level. Disturbance might lead to migratory land birds avoiding the area, entirely or partly, whereas barrier effects induce migrants to fly around or take height and fly over the wind farm with subsequent extra energetic flight costs (Masden et al. 2009). As for collisions, raptors and waterfowl, together with seabirds, were the groups with the highest disturbance and barrier PDF values and therefore expected to be impacted most by disturbance and barrier effects. Whereas migrating seabirds, waterbirds and waders had the lowest values for collisions, they were considerably more sensitive to disturbance and barrier effects. The songbirds were the least affected among the compared bird groups. In contrast to breeding (sea)birds encountering a neighbouring wind farm, habituation to OWFs during migratory crossings seems less likely. Particularly for short-lived, terrestrial species which encountering them twice a year, given they survive until the return migration. Seabirds and waterfowl are known to avoid and adjust their flight trajectories in response to OWFs, with subsequent increased energetic flight costs, at least to some degree (Petersen et al. 2006, Masden et al. 2009, Pettersson 2011). For all groups combined, disturbance and barrier effects resulted in the highest PDF values and expected impacts whereas collision PDF values were substantially lower – in line with the findings of May et al. 2021 for onshore wind energy developments in Norway. In summary, the functional groups of migratory birds that had the highest cumulative PDF values and are likely to be impacted the most by Trollvind OWF are raptors, waterfowl, gulls and seabirds compared to the other migratory groups assessed here. These groups were affected the

most by all three investigated impacts: collision, disturbance, and barrier effects. Their trajectories overlapped considerably with Trollvind OWF, and they are known to be affected by collisions, disturbance as well as barrier effects. However, the basis for the present study is ringing events and the subsequent recoveries of ringed birds, which are associated with accompanied uncertainties and potential biases. The data are dependent on the locations where birds are ringed and recovered, which are not evenly spread across the investigated countries. The bulk of ringing, and many of the recoveries, are done at bird observatories and by other active bird ringing sites. Bird observatories have the advantage of being located at sites where many migrants aggregate before and after sea crossings. However, this is dependent on migratory season: some sites are used more during autumn than spring, and vice versa. Moreover, aggregations can be weather dependent, where migrants in good weather conditions use other migratory trajectories, which might not be apparent from the data where sites with less ringing activity naturally are underrepresented. Thus, the data is dependent on the location of ringing and recovery, and ringing is strongly biased towards bird observatories and other sites with high ringing activity. However, recovery locations are less biased, because recoveries also stem from observations of dead ringed birds by the public, typically birds killed by cats, or killed in collisions with windows and cars. Although there are hundreds of thousands of ringing recoveries between the selected countries, the data becomes considerably restricted when enforcing the 60-day limit on the time interval between ringing and recovery events (to exclude recoveries encompassing two migratory seasons). The results presented here should be used as a relative indicator of the variation in impact between migratory groups and not an exact measure of the number of birds that will be impacted. It should also be noted that the location of the areas of highest impact from the LCAs differs to the areas of highest sensitivity found in the scoping study. The highest PDF values for the LCAs occur in the southwest corner of Trollvind OWF, whereas the highest sensitivity values in the scoping study occur in the northeast corner of Trollvind OWF. These results are to be expected as the underlying distributions and subsequent sensitivities from the scoping study were highest closest to the coast, whereas the migration data used for the LCA is dependent on ringing locations which are not evenly spread along the coast and contain the bias mentioned above.

PDF values are calculated based on the early phase design parameters of the proposed development that were communicated by Equinor to NINA on 14 December 2022 and updated on 16 January 2023 (*Appendix 2 Table A2.1*). If the location or any other parameters of this development change, our findings will no longer be valid. Despite the mentioned biases, the data still offers a unique insight into the migratory trajectories and the vulnerability of different species to the development of an OWF, here exemplified by the Trollvind OWF.

4.3 Scaling up OWF effects from individual sensitivity to population impacts

To demonstrate the use of the MARCIS tool for marine spatial planning, we have studied the potential consequences of an OWF for one population of kittiwakes and one population of guillemots which forage in the area during at least part of the non-breeding season. We found a negligible impact of the Trollvind OWF on non-breeding season survival for both study populations of guillemots from Sklinna and black-legged kittiwakes breeding in Ålesund. For kittiwakes from Ålesund, across all ABM simulations, there were no collisions and therefore no estimated additional mortality. For guillemots from Sklinna, ABM simulations estimated a small reduction in body mass due to Trollvind OWF. Since body mass had a small, positive effect on adult survival rates of guillemots, this translated into a correspondingly small, negative effect of Trollvind OWF on survival rates. Although population growth rates are highly sensitive to even small changes in adult survival rates (Gaillard et al. 2000), the impact of Trollvind OWF corresponded to an annual reduction of only 0.024% in survival. As a result, this had a very low population impact. Consequently, it is unlikely that the development of Trollvind OWF will have a significant impact on the size of the study populations. Nevertheless, there is a need to quantify such impacts on all seabird populations to determine the overall impact e.g., at a national scale.

It is important to note that the ABM framework has not considered the cumulative effects of the proposed development alongside other OWFs in the area, or alongside other stressors, such as fisheries bycatch and competition, or marine pollution. To fully assess the likely impact of this

development on the study populations, a study of cumulative effects of all stressors currently impacting seabirds across their full annual cycle should be undertaken, as is planned in the full MARCIS project. In addition, the ABM contained several assumptions. Firstly, we used displacement values taken from a study of guillemots in the southern North Sea (Peschko et al. 2020). To our knowledge, this is the most recent peer-reviewed study that provides estimates of displacement impacts in guillemots. Despite this, the study was not ideally suited for our needs as it took place during spring, thus did not fully encompass the non-breeding season. Guillemots may respond differently to OWFs at different times of the annual cycle, and have different responses based on the location of the OWF and other nearby stressors, the populations involved, and the energetic challenges currently impacting that population. Therefore, to improve our understanding of how seabirds respond to displacement within Norwegian waters, and thus improve the accuracy of future impact assessments, we recommend undertaking pre- and post-construction surveys within the OWF footprint and nearby areas throughout the annual cycle through analyses that are similar to Peschko et al. (2020). Alternatively, or in addition to these surveys, fine-scale tracking of seabirds from different populations throughout the annual cycle before and after construction would improve our understanding of the likely effects of displacement for different populations. Additionally, exactly how displacement affects the behaviour and energetics of an individual guillemot is unknown, as displacement effects include reduced access to preferred foraging habitats, increased competition for remaining food sources, reduced time spent foraging due to increased time spent in 'escape' behaviours, and increased travel time and costs due to barrier effects. Within the ABM, we assumed an energetic cost of displacement in guillemots equivalent to an additional 30 minutes of flight per day, which was a conservative estimate based on the high energetic cost of flight in guillemots (Elliott et al. 2013). Our ABM would therefore be improved considerably by an increased understanding of how behaviour changes when an individual guillemot is displaced from an OWF, and how this impacts energy expenditure. Finally, our findings from the ABM are based on the early phase design parameters of the proposed development that were communicated by Equinor to NINA on 14 December 2022 and updated on 16 January 2023 (*Appendix 2 Table A2.1*). If the location or any other parameters of this development change, our findings will no longer be valid.

4.4 Cumulative effects of an OWF and ocean warming

Cumulative Impact Assessments can be used to assess the cumulative effects of multiple stressors on marine ecosystems and to evaluate trade-offs between industrial development and conservation (Foley et al. 2010). Climate change is causing dramatic rises in ocean temperatures (Cheng et al. 2020), which alters the distributions and abundances of seabirds' prey (e.g., Frederiksen et al. 2006, Sydeman et al. 2015). This is considered one of the major causes of decline in seabird populations breeding in Norway (Fauchald et al. 2015) and globally (Dias et al. 2019). While offshore renewables will likely play an important role in reducing global carbon emissions and, hopefully, limit global warming, it is imperative to perform cumulative impact assessments assess the potential risks associated with climate change together with planned renewable developments on marine ecosystems.

Here, we provide a cumulative impact assessment for the planned Trollvind OWF and ocean warming during the non-breeding season. Projected ocean warming reduced survival of both species in this study, supporting findings from previous studies (e.g., Sandvik et al. 2005). Ocean warming trends were generally stronger in autumn than winter, with a substantial increase by 2050 (the end of the PVA projection). Warming trends were based on a high emissions scenario (SSP 8.5), often referred to as a worst-case scenario, which is considered the likely scenario if there is a lack of policy to reduce greenhouse gas emissions (Hausfather and Peters 2020). This was reflected in the larger impact weights for ocean warming compared to Trollvind OWF.

Although impact weights were small overall, it is important to highlight that we only investigated guillemot and kittiwake sensitivity to the Trollvind OWF development and to ocean warming during the non-breeding season. There are likely additional impacts of these stressors during the breeding season on survival but also potentially on breeding success, which was not included in this analysis due to the focus on the non-breeding season. However, while winter in particular appears to be a bottleneck for survival, there is evidence that ocean warming affects the breeding success of seabirds (Sydeman et al. 2012). Due to limitations of the current PVA framework,

while the predicted change in survival from 2021 to 2050, based on forecasted ocean temperatures, was implemented in the PVA as a linear increase from 2021 to 2050. In future version of the PVA framework, we aim to develop a more realistic approach including annually predicted survival estimates.

4.5 Future directions: the MARCIS App

The goal of the MARCIS App is to provide a decision-support tool, which gives a scientifically grounded knowledge base of cumulative impact of marine industrial pressures for how to improve the coexistence between seabirds and marine industries, with a strong seasonal focus. The aim is to provide transparent and openly available information to aid decision making surrounding the planning of marine developments like OWFs and to contribute to targeted recommendations ensuring successful co-existence with marine birds. While this demonstration of the MARCIS web application does not provide a comprehensive assessment of the potential impact of Trollvind OWF development on all seabird population using the area, it does provide a showcase of its potential use as a marine spatial planning tool. As discussed above, a challenge of this approach to CIAs is that it is highly data and analytically demanding since we provide quantified impacts of stressors on populations rather than relying on expert judgement. While this is a major analytical advancement in assessing cumulative impacts, the data requirements to parameterise these models mean it can only be applied to extensively monitored species. This highlights the need for large-scale cross-species monitoring, particularly for species groups that are likely to be strongly impacted by such marine developments, to perform much-needed CIAs given the current focus on developing marine ecosystems.

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6 Appendix
6.1 Appendix 1

Table A1.1. The species composition of the functional groups used in the Brownian Bridge Movement Models (BBMM) and subsequent Life Cycle Impact Assessments (LCIA) to elucidate the migratory trajectories of birds based on Norwegian, British, Belgian, Dutch, German and Danish ringing events and recoveries, including the sample sizes (N = number of ringed and recovered individuals).

Group	Latin name	Common name	N
Herbivorous songbirds	*Anthus pratensis*	Meadow pipit	105
	Anthus petrosus	Water pipit	100
	Anthus trivialis	Tree pipit	9
	Carduelis cannabina	Linnet	9
	Carduelis chloris	Greenfinch	481
	Carduelis flammea	Common redpoll	59
	Carduelis flavirostris	Twite	36
	Carduelis hornemanni	Arctic redpoll	2
	Carduelis spinus	Eurasian siskin	238
	Carpodacus erythrinus	Common rosefinch	2
	Coccothraustes coccothraustes	Hawfinch	5
	Emberiza citrinella	Yellowhammer	3
	Emberiza schoeniclus	Comon reed bunting	69
	Fringilla coelebs	Chaffinch	99
	Fringilla montifringilla	Brambling	155
	Loxia curvirostra	Red crossbill	6
	Motacilla alba	White wagtail	5
	Motacilla cinerea	Grey wagtail	2
	Prunella modularis	Dunnock	158
Insectivorous songbirds	*Acrocephalus schoenobaenus*	Sedge warbler	452
	Acrocephalus scirpaceus	Common reed warbler	218
	Delichon urbica	House martin	2
	Hirundo rustica	Barn swallow	46
	*Parus caeruleus**	Blue tit	38
	Parus major	Great tit	8
	Phylloscopus collybita	Chiffchaff	65
	Phylloscopus inornatus	Yellow-browed warbler	2
	Phylloscopus trochilus	Willow warbler	196
	Riparia riparia	Sand martin	8
	Sylvia atricapilla	Blackcap	235
	Sylvia borin	Garden warbler	92
	Sylvia communis	Common whitethroat	20
	Sylvia curruca	Lesser whitethroat	3
Polyphagous songbirds	*Bombycilla garrulus*	Bohemian waxwing	33
	Certhia familiaris	Eurasian treecreeper	4

	Cinclus cinclus	White-throated dipper	27
	Erithacus rubecula	European robin	255
	Ficedula hypoleuca	Pied flycatcher	36
	Luscinia svecica	Bluethroat	2
	Muscicapa striata	Spotted flycatcher	6
	Oenanthe oenanthe	Northern wheatear	11
	Phoenicurus phoenicurus	Common redstart	13
	Regulus regulus	Goldcrest	215
	Saxicola rubetra	Whinchat	7
	Sturnus vulgaris	European starling	64
	Troglodytes troglodytes	Eurasian wren	17
	Turdus iliacus	Redwing	39
	Turdus merula	European blackbird	241
	Turdus philomelos	Songthrush	21
	Turdus pilaris	Fieldfare	17
Corvids	*Corvus corone*	Hooded crow	1
	Lanius collurio	Red-backed shrike	3
	Lanius excubitor	Great grey shrike	2
Other non-passerines	*Apus apus*	Common swift	1
	Caprimulgus europaeus	European nightjar	1
	Cuculus canorus	Common cuckoo	2
	Jynx torquilla	Eurasian wryneck	6
	Streptopelia decaocto	Collared dove	1
Owls	*Asio flammeus*	Short-eared owl	1
	Asio otus	Long-eared owl	1
Raptors	*Accipiter nisus*	Eurasian sparrowhawk	27
	Buteo buteo	Common buzzard	2
	Buteo lagopus	Rough-legged buzzard	2
	Falco columbarius	Merlin	5
	Falco peregrinus	Peregrine falcon	5
	Falco tinnunculus	Common kestrel	7
	Milvus milvus	Red kite	2
	Pandion haliaetus	Osprey	8
Gulls	*Larus ridibundus*	Black-headed gull	599
	Larus argentatus	Herring gull	680
	Larus canus	Common gull	309
	Larus fuscus	Lesser black-backed gull	714
	Larus hyperboreus*	Glaucous gull	3
	Larus marinus*	Great black-backed gull	1802
	Rissa tridactyla*	Black-legged kittiwake	16
	Sterna hirundo	Common tern	97
	Sterna paradisaea	Arctic tern	9
Seabirds	*Alca torda**	Razorbill	2
	Hydrobates pelagicus*	European storm petrel	324
	Phalacrocorax carbo	Cormorant	35
	Stercorarius skua*	Great skua	4

	Sula bassana*	Northern Gannet	8
	Uria aalge*	Common guillemot	37
Waders	*Actitis hypoleucos*	Common sandpiper	16
	Arenaria interpres	Ruddy turnstone	47
	Calidris alba	Sanderling	10
	Calidris alpina	Dunlin	349
	Calidris ferruginea	Curlew sandpiper	34
	Calidris maritima	Purple sandpiper	67
	Calidris minuta	Little stint	74
	Charadrius hiaticula	Common ringed plover	132
	Charadrius morinellus*	Eurasian dotterel	2
	Gallinago gallinago	Common snipe	9
	Haematopus ostralegus	Eurasian oystercatcher	118
	Limosa lapponica	Bar-tailed godwit	89
	Limosa limosa	Black-tailed godwit	3
	Lymnocryptes minimus	Jack snipe	3
	Numenius arquata	Eurasian curlew	5
	Philomachus pugnax*	Ruff	37
	Pluvialis apricaria	European golden plover	3
	Pluvialis squatarola	Grey plover	3
	Scolopax rusticola	Eurasian woodcock	7
	Tringa glareola	Wood sandpiper	5
	Tringa nebularia	Common greenshank	5
	Tringa ochropus	Green sandpiper	8
	Tringa totanus	Common redshank	42
	Vanellus vanellus	Northern lappwing	3
Waterfowl	*Anas crecca*	Eurasian teal	11
	Anas platyrhynchos	Mallard	2
	Anser anser	Greylag goose	138
	Anser brachyrhynchus	Pink-footed goose	274
	Anser erythropus	Lesser white-fronted goose	2
	Anser fabalis	Bean goose	16
	Aythya fuligula	Tufted duck	2
	Branta bernicla	Brant goose	4
	Branta canadensis	Canada goose	7
	Branta leucopsis	Barnacle goose	91
	Cygnus cygnus	Whooper swan	15
	Cygnus olor	Mute swan	49
Waterbirds	*Ardea cinerea*	Grey heron	18
	Fulica atra	Eurasian coot	4
	Gallinula chloropus	Common moorhen	3
	Grus grus	Common crane	108

*indicates inclusion of a new species to the functional group.

Table A1.2. Summed lower, mean and upper PDF values of estimated disturbance, collision and barrier impacts for Trollvind based on current turbine plans. Rankings go from 1-highest impact to 12-lowest impact, according to mean PDF value. The cumulative rank is calculated as the ranked sum of the ranks for all three impact factors.

Group	Disturbance				Collision			
	Lower	Mean	Upper	Rank	Lower	Mean	Upper	Rank
Corvids	1.11×10^{-5}	7.35×10^{-5}	1.80×10^{-4}	8	3.82×10^{-5}	5.97×10^{-5}	8.09×10^{-5}	6
Gulls	6.26×10^{-5}	1.34×10^{-4}	2.04×10^{-4}	6	6.40×10^{-5}	9.09×10^{-5}	1.17×10^{-4}	2
Herbivorous songbirds	2.69×10^{-6}	6.03×10^{-6}	1.01×10^{-5}	12	2.97×10^{-5}	5.38×10^{-5}	7.75×10^{-5}	8
Insectivorous songbirds	2.33×10^{-6}	6.36×10^{-6}	1.34×10^{-5}	11	3.16×10^{-5}	5.74×10^{-5}	8.26×10^{-5}	7
Non-passerines	3.05×10^{-6}	2.87×10^{-5}	6.16×10^{-5}	9	1.18×10^{-5}	6.03×10^{-5}	1.07×10^{-4}	5
Owls	1.11×10^{-5}	1.28×10^{-4}	2.74×10^{-4}	7	7.06×10^{-5}	7.25×10^{-5}	7.45×10^{-5}	4
Polyphagous songbirds	3.77×10^{-6}	8.63×10^{-6}	1.35×10^{-5}	10	2.36×10^{-5}	5.24×10^{-5}	8.04×10^{-5}	9
Raptors	5.92×10^{-5}	4.70×10^{-4}	1.07×10^{-3}	3	2.92×10^{-5}	9.75×10^{-5}	1.62×10^{-4}	1
Seabirds	6.32×10^{-5}	2.37×10^{-3}	7.33×10^{-3}	1	0.00×10^{-5}	3.80×10^{-5}	8.09×10^{-5}	12
Waders	1.89×10^{-5}	1.56×10^{-4}	4.18×10^{-4}	5	1.87×10^{-5}	4.66×10^{-5}	7.39×10^{-5}	10
Waterbirds	4.74×10^{-5}	3.60×10^{-4}	6.30×10^{-4}	4	2.50×10^{-5}	4.11×10^{-5}	5.70×10^{-5}	11
Waterfowl	5.95×10^{-8}	1.37×10^{-3}	5.82×10^{-3}	2	4.77×10^{-5}	7.94×10^{-5}	1.10×10^{-4}	3

Group	Barrier				Cumulative
	Lower	Mean	Upper	Rank	Rank
Corvids	5.03×10^{-6}	3.34×10^{-5}	8.19×10^{-5}	7	8
Gulls	4.50×10^{-5}	9.62×10^{-5}	1.47×10^{-4}	6	3
Herbivorous songbirds	7.60×10^{-7}	1.70×10^{-6}	2.85×10^{-6}	12	12
Insectivorous songbirds	1.65×10^{-6}	4.49×10^{-6}	9.44×10^{-6}	10	10
Non-passerines	2.03×10^{-6}	1.91×10^{-5}	4.09×10^{-5}	9	9
Owls	2.82×10^{-6}	3.24×10^{-5}	6.97×10^{-5}	8	5
Polyphagous songbirds	1.62×10^{-6}	3.70×10^{-6}	5.80×10^{-6}	11	11
Raptors	2.49×10^{-5}	1.97×10^{-4}	4.46×10^{-4}	3	1
Seabirds	1.60×10^{-5}	5.87×10^{-4}	1.76×10^{-3}	1	3
Waders	1.50×10^{-5}	1.24×10^{-4}	3.32×10^{-4}	4	5
Waterbirds	1.41×10^{-5}	1.07×10^{-4}	1.87×10^{-4}	5	7
Waterfowl	2.01×10^{-8}	4.58×10^{-4}	1.89×10^{-3}	2	1

6.2 Appendix 2

Table A2.1. *The following early phase design parameters for the Trollvind offshore wind farm were used in the stochastic Collision Risk Model, as communicated from Equinor on 14 December 2022 and updated on 16 January 2023.*

Variable	Value
Number of turbines	50
Output of each turbine	20 MW
Number of blades on each turbine	3
Turbine rotor radius	135m
Airgap	23-24m
Maximum rotor blade width	7.5m
Rotation speed	6.6 rpm
Blade pitch	5.8°

6.3 Appendix 3

Table A3.1 *Results from linked mass-survival models (see Methods 2.3.4). Results show the estimated mean body mass at the end of the breeding season (μ_M) and the effect of mean body mass on adult survival during the following non-breeding season (β_X). Overall effects of body mass were low with 95% credible intervals overlapping zero.*

Species	Colony	Mean mass (μ_M)	Mass effect (β_X)
Common guillemot	Sklinna	943.87 (941.92, 945.83)	-0.002 (-0.001, 0.006)
Black-legged kittiwake	Sklinna	378.16 (376.19, 380.11)	0.000 (0.000, 0.000)
Black-legged kittiwake	Svalbard, Isfjorden	386.17 (384.29, 388.09)	-0.001 (-0.01, 0.009)
Brünnich's guillemot	Svalbard, Isfjorden	989.94 (987.97, 991.89)	-0.002 (-0.006, 0.001)
Brünnich's guillemot	Svalbard, Kongsfjorden	990.80 (988.82, 992.79)	0.001 (-0.004, 0.006)
Little auk	Svalbard, Kongsfjorden	161.50 (159.94, 163.11)	-0.001 (-0.011, 0.009)
Common Guillemot	Hornøya	1041.83 (1039.91, 1043.76)	0.001 (-0.034, 0.037)

Table A3.2. *Summary additional mortality effects and population-level impact weights associated with the two stressors (Trollvind OWF and ocean warming), following the results in Results sections 3.3.1-3.3.3. Impact weights were calculated as the difference in population growth between baseline and impacted populations, based on 100,000 population simulations. Estimates are shown as means (95% confidence intervals). Estimates are multiplied by 10^4.*

Study population	Additional mortality		Impact weight	
	Trollvind OWF	Ocean warming	Trollvind OWF	Ocean warming
Kittiwakes-Ålesund	0.0 (0.0, 0.0)	10.4 (-9.2, 52.2)	-1.9 (-11.8, 6.1)	154.8 (155.2, 158.3)
Guillemots-Sklinna	2.2 (0.0, 4.5)	13.8 (-3.0, 44.9)	44.9 (42.5, 45.3)	301.3 (294.0, 306.0)

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