Environmental Impact Assessment Review 90 (2021) 106635

Contents lists available at ScienceDirect

Environmental Impact Assessment Review

journal homepage: www.elsevier.com/locate/eiar

Life-cycle impacts of wind energy development on bird diversity in Norway

Roel May [a,*], Craig R. Jackson [a], Heleen Middel [b], Bård G. Stokke [a], Francesca Verones [b]

[a] Norwegian Institute for Nature Research (NINA), P.O. Box 5685 Torgarden, NO-7485 Trondheim, Norway
[b] Industrial Ecology Programme, Department of Energy and Process Engineering, Norwegian University of Science and Technology (NTNU), Høgskoleringen 5, 7491 Trondheim, Norway

ARTICLE INFO

Keywords:
Potentially disappeared fraction of species (PDF)
Habitat loss
Disturbance
Collision
Barrier
Life cycle assessment (LCA)
Levelized cost of energy (LCOE)

ABSTRACT

While wind energy remains a preferred source of renewable energy, understanding the full spectrum of impacts are vital to balance climate-related benefits against their costs to biodiversity. Environmental impact assessments often fail to assess cumulative effects at larger spatial scales. In this respect, life cycle assessments are better suited, but have to date mainly focused on greenhouse gas emissions and energy accounting. Here, we adapt a recent global life-cycle impact assessment (LCA) methodology to evaluate collision, disturbance and habitat loss impacts of onshore wind energy development on bird species richness in Norway. The advantage of a local model for Norway is that it enables employing species distribution models to more accurately estimate the potential distribution area of species. This facilitates more realistic site- and species-specific assessments of potential impacts within a local scale but excludes habitat ranges outside Norway. Furthermore, a new characterization factor was developed for potential barrier effects. Larger onshore wind-power plants overall had greater site-specific potentially disappeared fractions (PDF) of species, while smaller plants were less efficiently located with greater impacts per GWh. Overall, Norwegian wind-power plants were sited least efficiently (PDF/GWh) regarding indirect habitat loss (2.186×10^{-9}) and disturbance (1.219×10^{-9}), followed by direct habitat loss (0.932×10^{-9}), and finally collisions (0.040×10^{-9}) and barriers (0.310×10^{-9}). Vulnerability differed among bird groups with seabirds, raptors and waterfowl emerging as the most impacted groups (e.g. 5.143×10^{-9}, 3.409×10^{-9} and 3.139×10^{-9} PDF/GWh for disturbance, respectively); highlighting the sympatric distribution of their habitats and the majority of Norway's onshore wind-power plants. Current practice has not succeeded in avoiding sites with higher impacts for birds, fuelling conflicts surrounding environmental concerns of onshore wind energy development in Norway. Operative LCA models can help decision-makers assessing localized life-cycle environmental impacts to support environmental-friendly wind energy production in specific regions.

1. Introduction

Development of renewable energy has increased worldwide to address climate change concerns (UNFCCC, 2016). In order to secure energy supply and adhere to international calls and agreements to reduce greenhouse gas emissions – as stipulated in the IPCC report, Paris Agreement, and the UN Sustainable Development Goals (SDGs) – the Norwegian government aims to expand the electricity supply system with renewable energy technologies (OED, 2016). The global potential for wind-power generation (Lu et al., 2009) is regarded as an important renewable energy source (IEA, 2019a, 2019b). However as the IPCC Special Report on Renewable Energy (IPCC, 2011) stressed: "*environmental and social issues will affect wind energy deployment opportunities*". While wind energy contributes to reductions in greenhouse gas emissions, it also affects biodiversity negatively. This necessitates balancing trade-offs between climate change mitigation and environmental protection (Köppel et al., 2014; May et al., 2017).

The construction and operation of wind-power plants impacts wildlife, especially birds, through habitat alterations, reduced habitat utilization due to disturbance, collision mortality, and barriers to movement (Laranjeiro et al., 2018; Marques et al., 2014; Schuster et al., 2015). The likelihood and magnitude of those impacts are site- and species-specific (May et al., 2017). Still, with the fast rate of development, it will be of the utmost importance to minimize the environmental costs per kWh generated from wind energy. Wind-power plants should therefore be sited at larger and accessible sites with good wind conditions, and with acceptable levels of environmental impact (Allison et al., 2014; Evans et al., 2009; Warren and Birnie, 2009).

* Corresponding author.
E-mail address: roel.may@nina.no (R. May).

https://doi.org/10.1016/j.eiar.2021.106635
Received 7 December 2020; Received in revised form 24 June 2021; Accepted 28 June 2021
Available online 2 July 2021

Norway has a considerable potential for onshore wind-power generation (Duffy et al., 2020; Ryberg et al., 2019), which is regarded as an important renewable energy source both onshore and offshore (OED, 2020). Although comprehensive consenting procedures are employed to balance onshore wind energy production against potential environmental impacts, the rapid rate of development increases the potential for conflicts (Rygg, 2012; Solli, 2010). Lack of generic knowledge on the spatial distribution of potential environmental risks, uncertainty in the expected impacts, and the extent to which these can be mitigated are important causes of conflict in wind energy proposals, leading to significant and costly delays or refusal of consent (Hastik et al., 2015; May et al., 2017). This results in subjective assessments of environmental risks, driven largely by people's perceptions rather than empirical evidence (McLachlan, 2009; Stoutenborough and Vedlitz, 2016). Social concerns on consenting procedures and the perceived environmental risks require a combined effort to find solutions to improve procedural acceptance (Toke et al., 2008) and mitigate the adverse impacts on nature (Gartman et al., 2016a, 2016b; Marques et al., 2014; May et al., 2015). The centralized consenting system in Norway promotes an efficient decision-making process (Thygesen and Agarwal, 2014), but hampers legitimacy of the process to address local environmental concerns (Liljenfeldt, 2014). The sectoral regulatory authority responsible for the consenting process have given the licensing body "*considerable room for decisional discretion*" (Inderberg et al., 2019). This has led to a situation lacking transparency in the grounds for consenting outcomes, hampering comparability among projects and addressing local concerns (Inderberg et al., 2019). Concerns are often related to local impacts on the environment such as visual impacts on the landscape or impacts on wildlife populations, particularly birds (Blindheim, 2015; Inderberg et al., 2019; Rygg, 2012; Solli, 2010; Thygesen and Agarwal, 2014). Environmental impact assessment (EIA) is the key regulatory mechanism for informing consenting authorities on the potential for environmental impacts of proposed developments (Thygesen and Agarwal, 2014). Still, EIA practice has received much negative reputation in Norway with regard to the quality of assessments (Inderberg et al., 2019; Solli, 2010) and their of lack impact on consenting decisions (Thygesen and Agarwal, 2014). Currently, the consenting process has been amended with stronger consideration for environment and landscape values, stricter requirements for EIAs and regional assessments of cumulative effects (OED, 2020). A more evidence-based approach enhances sound decision-making processes, particularly when based on "best available science" efforts (Weber et al., 2019).

While EIA can evaluate site-specific environmental impacts, assessing the cumulative environmental effects throughout a project's life cycle and comparing impacts across alternative sites, strategies or technical options for development requires other mechanisms (Hoffman, 2017; Masden et al., 2010a). Life cycle assessment (LCA) is an appropriate tool to systematically assess potential environmental impacts occurring throughout a technological system's life cycle at different stages and at different locations (Verones et al., 2017). An LCA starts with goal definition and scoping to establish the context of the assessment and to identify the boundaries and environmental effects to be reviewed for the assessment (e.g. assessing the biodiversity impacts of 1 GWh of electricity production). The resources used and emissions released throughout the life cycle of a product or process identified in a life cycle inventory are thereafter joined with a life cycle impact assessment (LCIA) model to quantify the potential impact per unit of stressor (e.g. collision impacts per kWh produced at a certain location). This enables the consideration of multiple impacts simultaneously and in a spatially explicit manner (e.g., climate change and ecosystem consequences of wind-energy production), thereby enabling the identification of potential trade-offs between strategies for renewable energy development (Hellweg and Mila i Canals, 2014; Laranjeiro et al., 2018).

LCA, and especially LCIA, has developed rapidly over the last decades (Verones et al., 2017; Woods et al., 2017), however, wind energy LCAs so far only include environmental impacts from greenhouse gas emissions and energy accounting (Al-Behadili and El-Osta, 2015; Ali Alsaleh and Sattler, 2019; Arvesen and Hertwich, 2012; Gomaa et al., 2019; Wang et al., 2019) due to a lack of LCIA models. Although wind energy performs well regarding greenhouse gas emissions, impacts on the natural environment should also be included in LCA (May et al., 2012). Regarding the development of onshore wind energy, the construction and operational phase are considered to pose highest impact on biodiversity through habitat loss, disturbance, collisions and barrier effects (Laranjeiro et al., 2018), and these are therefore the life cycle phases that our work focuses on. Recently, May et al. (2020a) developed a global LCIA model quantifying the first three impact pathways of onshore wind energy development on bird biodiversity.

Most LCIA methods that deal with impacts of land use on species richness resort to species-area relationships (SAR), which relate area loss to an exponential loss in species richness (e.g. Chaudhary et al., 2015; de Baan et al., 2013a). Since these methods aim to take the entire world into account, they typically utilize global range maps of species presence (e.g. IUCN, WWF, BirdLife), which assume equal probability of presence across a species' entire distribution (Hurlbert and Jetz, 2007). However, impacts of onshore wind energy development, as well as wind resources, are site-specific as they directly depend on species' habitat requirements and presence. Assuming equal probability of presence across large areas can therefore lead to an inaccurate assessment of impacts. We argue, therefore, that this aspect should be included in LCIA models by estimating species' probability of presence in a spatially differentiated way using species distribution models (SDMs) (de Baan et al., 2013a; Laranjeiro et al., 2018; Maia de Souza et al., 2015). SDMs explicitly account for habitat suitability and the spatial variability thereof. The configuration of suitable habitat patches affects species' capacity to move across landscapes and therefore connectivity between these habitats (Mitchell et al., 2015). SDMs therefore additionally facilitate the assessment of connectivity-related impacts of development using circuit theory (McRae et al., 2008).

Consenting processes should in principle holistically balance economic and environmental interests and identify potential trade-offs between strategies for renewable energy development. This requires that site-specific (environmental) costs and (economic) benefits can be quantified in a standardized manner and can be offset against each other. The objective of this paper is, by focusing on onshore wind energy generation, to present the procedure to offset the levelized cost of energy (LCOE) function as a proxy for economic interests (Duffy et al., 2020) against the four main LCA impacts during the construction and operational phase on bird diversity. This will be done by adapting the methodology developed by May et al. (2020a) using an SDM approach specifically for Norway. The three impacts pathways developed by May et al. (2020a) will be complemented by developing a characterization factor for barrier effects of onshore wind energy development. This approach will thereafter be showcased for Norway to assess to which extent potential bird impacts have been taken into account regarding the siting of operational onshore wind-power plants.

2. Methodology

The methodology to quantify impacts of wind-power plants on bird diversity are based on generic SAR models, which are widely used in LCA (Chaudhary et al., 2015; de Baan et al., 2013a; de Baan et al., 2013b). LCIA usually quantifies species richness loss as the potentially disappeared fractions of species (PDF) (Verones et al., 2017; Woods et al., 2017). SAR-derived PDFs are here considered to be a measure of the *potential* loss of species richness due to wind-energy development; not the actual number of species lost. The main impact pathways of onshore wind energy on birds are (1) habitat alterations, (2) disturbance, (3) collisions, and (4) barrier effects. May et al. (2020a) developed a methodology for the first three impact pathways, which will be adapted for Norway in this article. In addition, we develop a methodology to assess barrier effects. LCIA typically utilizes range maps of species presence, which assume equal probability of presence across the entire distribution of a species. However, habitat loss, displacement, collision mortality and barrier effects are directly dependent on the habitat requirements and therefore abundance of a species at any given site. We therefore used SDMs to estimate the spatially explicit likelihood of occurrence of species across Norway. The assessment was done separately for 13 groups of birds (Table S1). Groups consisted of taxonomically (by (sub)order or parvorder) and functionally similar species, as well as one miscellaneous group consisting of various non-passerines, which were grouped together (Apodidae, Caprimulgidae, Columbidae, Cuculidae, Picidae, Upupidae) as they were only represented by few species each.

2.1. Mapping species occurrence

Species-specific SDMs were constructed with MaxEnt (Phillips et al., 2017), using presence-only data and a set of ecologically relevant environmental variables. MaxEnt is well suited given that it only requires presence data, which is available for most species while absence data often is unreliable and rarely available. Presence data was downloaded in batches from the Global Biodiversity Information Facility (GBIF) between 21 and 26 November 2019. Data was restricted to records collected in Norway between 2010 and 2019. Seasonal migration was not specifically considered in the analyses, consequently all records throughout the year were included. Species observations included in such databases are inherently prone to geographic sampling biases (Dennis and Thomas, 2000). For example, survey effort is greater closer to towns, cities and along road networks. A variety of bias correction methods have been proposed when using such data in SDMs (Fourcade et al., 2014). We applied a systematic record sampling approach which outperforms many other approaches in such applications (Fourcade et al., 2014). To do this we created 1 km^2 grid cells across the entire map area, the resolution of which corresponded to that of the selected environmental predictor variables (details below). The centroid of each cell in which a species had been recorded at least once was extracted and subsequently used as a presence record in the MaxEnt models. Due to the large quantities of data available for most species, half the occurrence data was used for model training and half for model testing. Species with less than 500 GBIF records within Norway, which usually translated to far fewer centroid locations, were not modelled. A total of nine environmental predictor variables were included in the models: annual mean temperature (°C), temperature seasonality (S.D.*100), annual precipitation (mm) and precipitation seasonality (C.V.) downloaded from WorldClim 2.0 (1 km^2 resolution; values based on data from 1970 to 2000) (Fick and Hijmans, 2017); digital elevation model (Norwegian Mapping Authority); land cover, latitude and distance to sea (AR50, Norwegian Mapping Authority); and terrain ruggedness index (Norwegian Water Resources and Energy Directorate (NVE)). Species-specific SDMs were thereafter aggregated for each of the 13 groups of birds. The resulting group-wise SDM maps give spatially-explicit scores proportional to the probability of occurrence $P_{k,i}$ of number of species S_k of functional group k at a 1×1 km^2 resolution i.

The use of species distribution models, rather than broadscale range maps, more accurately reflect habitat suitability and therefore a given species' potential distribution. Although we only used data from 2010 to 2019, we did not include any information on infrastructure that may reduce habitat suitability. By doing so, the models are suitable for species-area relationship analyses, which calculate the potential number of species present in a specific location prior to human intervention. Furthermore, the 1 km^2 mapping resolution restricts fine-scale impacts of human interventions. All Norwegian wind-power plants are located in open habitats, thereby negating the need for extensive habitat transformations (i.e. deforestation).

2.2. Life cycle assessment for the main impact pathways of onshore wind energy

The PDF's for habitat loss, disturbance and collision mortality were calculated following May et al. (2020a) (see also Table 1). We calculated the locally present number of species $S_k \cdot P_{k,i}$ as the summed species-specific MaxEnt probabilities of presence across all pixels (1×1 km^2) i in Norway for all 13 groups of birds k. PDF's were calculated per turbine for each functional group individually, and thereafter aggregated per wind-power plant. Turbine data were downloaded from NVE (http://nedlasting.nve.no/gis/, downloaded 17.10.2019), and updated to cover all currently operational wind-power plants in Norway (N = 39). The slope of the species-area relationship in logarithmic scale (z) was taken to be 0.21 (Storch et al., 2012). After construction, the PDF for habitat loss was approximated by the direct and indirect (e.g. roads) area requirements per MW capacity a_{EP} of respectively 0.3 ha/MW and 0.7 ha/MW (Denholm et al., 2009) at wind turbine w with electrical power EP_w relative to the total area available to the species within bird group k at site i ($A_{org} = 1$ km^2). The PDF for disturbance is measured as the proportion of species displaced from the influence area at wind turbine w, predicted using the relative integral of the sigmoid function relating the proportion of species displaced over distance (D_k). This displacement factor was derived from minimum and maximum flight initiation distances for each of the bird groups as described in May et al. (2020a). Generically, the PDF for collision was quantified as the reduction of the species at risk to species surviving collision. The collision probability (R_k) was approximated using the species-specific collision rates (per turbine/year) averaged (± SD) for each bird group (Thaxter et al., 2017). The values used for both displacement and collision are given in Table S2.

In addition to these previously developed impact pathways, we also developed a separate approach to quantify barrier effects (eq. 1). Barriers to movement cause increased distances travelled and thereby energy expenditure. Barrier effects therefore assess a different pathway than disturbance; while the latter assesses displacement from the impacted area given the local habitat suitability, the first assesses fitness costs due to increased energy expenditure given the connectivity of the impacted area relative to the surrounding landscape. Within the scope of this study we assume barrier effects to increase with distance travelled and species-specific morphology (Masden et al., 2010b), which may have the largest effect for migratory birds (Somveille et al., 2018). The SDM giving the probability of presence for bird group k can be used to predict connectivity across the landscape, using the Circuitscape software (version 4.0.5). Circuitscape predicts connectivity in heterogeneous landscapes by applying algorithms from electronic circuit theory (McRae et al., 2008). Here, habitat suitability (i.e. where species prefer to move) funnels the relative conductance to movement which affects current flows across the landscape. The relative flow of movement from such analysis ($C_{k,i} \in [0,1]$) can be used to quantify barrier effects (i.e. reduced probability of connectivity) of wind turbines by substituting $P_{k,i}$ by $C_{k,i}$ in the general PDF formula (Table 1, eq. 1). Here, barrier effects are dependent on the relative conductance to movement across the landscape, the location of a wind turbine relative to this and the distance ($d_{k,max}$) and proportion (D_k) over which a species group k is expected to be affected. Loss of such sites due to the construction of wind turbines, is assumed to have a proportional effect on energy expenditure (Masden et al., 2010b; Masden et al., 2009). The energetic impact (M_k) this has on birds depends on their energy requirement for seasonal migration (α_k) and the migration distance (l_k) (Somveille et al., 2018). Somveille et al. (2018) showed that migration cost, estimated in arbitrary (i.e. relative) units of energy use, is contingent on body mass (m, in grams) and distance travelled: $M_k = 2 \bullet \alpha_k \bullet l_k$ (the factor 2 is to account for cost of both spring and autumn migration), with energy requirement as $\alpha \approx 6.5 \bullet 10^{-5} \bullet m^{-0.01}$ per km travelled per season (averaged over species within each group k). We calculated this proxy for the additional relative annual energy cost within the affected area using the information on

Table 1

Equations for calculating the potential disappeared fraction (PDF) of species from May et al. (2020a), and how the parameters were defined for Norway. PDF's were calculated per turbine, and thereafter summed for each wind-power plant.

Habitat loss (H)	Disturbance (D)	Collision (C)
$PDF(H)_{k,w} = \frac{S_k \bullet P_{k,i} \bullet \left(1 - \left(\frac{A_{org} - a_{EP} \bullet EP_w}{A_{org}}\right)^z\right)}{\sum_i^I S_k \bullet P_{k,i}}$	$PDF(D)_{k,w} = \frac{S_k \bullet P_{k,i} \bullet \left(1 - \left(\frac{A_{org} - t_w \bullet \left(\pi \bullet \left(D_k \bullet d_{k,max}\right)^2\right)}{A_{org}}\right)^z\right)}{\sum_i^I S_k \bullet P_{k,i}}$	$PDF(C)_{k,w} = \frac{S_k \bullet P_{k,i} \bullet \left(1 - \left(\frac{A_{org} - R_k \bullet t_w \bullet \left(\pi \bullet r_w^2\right)}{A_{org}}\right)^z\right)}{\sum_i^I S \bullet P_{k,i}}$
$S_k \cdot P_{k,i}$ = number of species locally present at cell i within group k $A_{org} = 1$ km^2 $a_{EP} = 0.3$ or 0.7 ha/MW EP_w = output (MW) of turbine w	$t_w = 1$ turbine D_k = disturbance factor within group k $d_{k,\max}$ = maximum flight initiation distance within group k	r_w = rotor blade length of turbine w R_k = probability of annual per-turbine collision within group k

body mass and migration distance obtained from Vincze et al. (2019). Using this equation, species-specific energy efficiency factors were averaged by species group k.

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

The characterization factors for each of the impact pathways were estimated by aggregating PDF's across functional groups and/or wind-power plants and dividing the cumulative PDF's by the annual energy production (E_w in GWh) per turbine w or wind-power plant.

2.3. Balancing economic and environmental interests of onshore wind energy

The extent with which wind energy development has considered environmental concerns, using the LCA impacts as proxy, regarding the siting of currently operational wind-power plants was tested in the following manner. First, the centroid of each wind-power plant was moved to a random location inside Norway, and the wind turbine locations were recalculated accordingly. The LCA impacts were recalculated for these adjusted wind-power plant sites; this process was iterated 100 times. Using this dataset of random (0) and used (1) sites with associated four LCA impacts, while controlling for LCOE values at each wind-power plant site, were thereafter used to assess the relative contribution on siting using a discrete choice model. LCOE values across Norway were obtained from NVE (Weir, 2018). Thereafter, we compared used versus random sites using a generalized linear mixed effects model with a Poisson distribution. The model was constructed using the glmer function in the lme4 library (Bates et al., 2015), and included an offset term for LCOE (log-transformed) as well as a random effect for each wind-power plant. Second, we quantified the LCA impacts across Norway at a 1-km resolution based on an average wind turbine (3 MW, 50 m rotor blade length), and thereafter extracted these per-turbine impacts for all operational turbines in Norway. The optimum break-points in the statistical distribution for LCOE and summed LCA impacts were calculated using Jenks natural breaks optimization with four breaks (i.e. three groups) using the getJenksBreaks function of the BAMMtools package (Rabosky et al., 2014). Thereafter the LCOE and the summed LCA impacts across all four impact pathways were scored into good (1), intermediate (2) and bad (3) sites for wind energy development. These scores were thereafter combined into equal zones based on their summed scores into good sites (<4), intermediate sites (4) and bad sites (>4). Apart from mapping species distributions (MaxEnt) and connectivity (Circuitscape), all analyses were performed in RStudio Version 1.2.5019 and R 3.6.3 (R Core Team, 2020).

3. Results

The SDM, aggregated per group of birds, performed well (training AUC: 0.814, test AUC: 0.811, entropy: 8.605). The polyphagous songbirds and seabirds had the lowest and highest performance, respectively (Table S3). The overall performance declined with number of species within each group of birds (Pearson's product moment correlation of −0.454 and − 0.466 for the training and test AUC, respectively). Overall elevation and annual mean temperature had the highest contribution to the models, which was the most important of these two varied however by group. Distance to sea, latitude and land cover also had important contributions for some of the groups (see Table S3). Seasonality and precipitation had overall limited contributions. Fig. 1 shows the aggregated species richness (A) and connectivity (B) map for Norway across bird groups. Group-specific species richness and connectivity maps can be found in the S.I. These maps formed the basis for calculating the LCA impacts of wind-power plants for the four impact pathways.

Overall, wind-power plants in Norway were expected to lead to an annual PDF of 0.703×10^{-5} and 1.650×10^{-5} due to direct and indirect habitat loss, respectively. Across bird groups, an annual PDF of 0.920×10^{-5} (0.082×10^{-5}–2.650×10^{-5}) was estimated to occur due to disturbance and 0.030×10^{-5} (0.013×10^{-5}–0.046×10^{-5}) due to collisions at Norwegian wind-power plants. In addition, they caused barriers to movement leading to an annual PDF of 0.234×10^{-5} (0.022×10^{-5}–0.659×10^{-5}). Smøla wind-power plant (68 wind turbines, 150.4 MW and 356 GWh) had the largest effect on the chronological cumulative increase in PDF in 2002 (Fig. 2 – left-hand panel), followed by Bjerkreim (55 wind turbines, 231 MW and 14 GWh) and Storheia

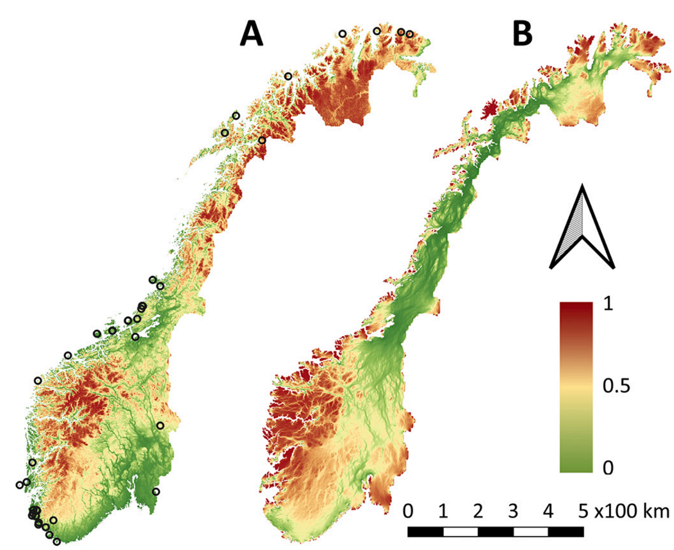

Fig. 1. Relative bird diversity (A) and connectivity (B) in Norway based on species-specific MaxEnt distribution models and averaged across 13 bird groups. Relative bird diversity was obtained by dividing pixel (1 km^2) values by the maximum number of species (286). The open circles indicate the centroids of all included wind-power plants.

wind-power plant (80 wind turbines, 288 MW and 973 GWh) both in 2019. The increase in 2012 was due to five sites being developed that year (S.I.). For barrier impacts, a sharp increase in the cumulative PDF occurred in 2001 with the construction of Mehuken wind-power plant (Fig. 3). Contrary to the other impacts, cumulative barrier PDFs kept increasing nearly linearly over time. When taking the annual energy production at each wind-power plant into account, Bjerkreim wind-power plant, developed in 2019, was least efficient (Fig. 2 – right-hand panel; largest increase), followed by Rye Vind wind-power plant (1 wind turbine, 0.2 MW and 0.3 GWh). Smøla wind-power plant was by far the most efficient for barrier impacts (Fig. 3). Overall, Norwegian wind-power plants were estimated to be least efficient regarding their impact on species diversity (i.e. PDF / GWh) foremost due to indirect habitat loss (2.186×10^{-9}) and disturbance (1.219×10^{-9} [0.109×10^{-9}–3.513×10^{-9}]), followed by direct habitat loss (0.932×10^{-9}), and finally collisions (0.040×10^{-9} [0.017×10^{-9}–0.061×10^{-9}]) and barriers (0.310×10^{-9} [0.029×10^{-9}–0.874×10^{-9}]). Smøla and Bjerkreim wind-power plants have overall had the largest impact on birds in Norway (Fig. 4). For habitat loss, disturbance and collision, impacts increased significantly with turbine capacity and number of turbines (Table 2). However, larger wind-power plants with more powerful turbines had significantly reduced impacts, as shown by the negative interaction terms. Contrary to this, more powerful turbines – especially within larger wind-power plants (interaction term) – had a small but significant positive effect on barrier impacts.

Onshore wind-power plants differed in their efficiency (measured as PDF/GWh) of siting for the different taxonomic bird groups. Seabirds were the most affected group; foremost due to their susceptibility to habitat loss, disturbance and barrier effects (Table 3). In second place, raptors were mostly affected by collisions, disturbance and barrier effects. Thereafter, waterfowl were mostly affected through disturbance and barrier effects, and gulls through collisions. Gallinaceous birds, owls and songbirds (herbivorous and polyphagous) were overall least affected by wind-power plants in Norway. Raptors, seabirds, waders and waterfowl had much higher estimated PDFs for disturbance and barrier effects compared to other bird groups. For habitat loss and collisions, between-group differences were less pronounced.

Comparing LCA impacts of operational wind-power plants to random sites, while controlling for LCOE, indicated that siting caused significant disturbance impact ($z = 4.365$, $P < 0.001$) while habitat loss was the only impact that tended to be avoided ($z = -0.985$, $P = 0.325$) (Fig. 5). Neither collision ($z = 0.651$, $P = 0.515$) nor barrier impacts ($z = 0.046$, $P = 0.963$) had any effect on siting. Disturbance impacts at sited wind-power plants were most prominent for coastal species (seabirds, waders, gulls, waterbirds, waterfowl) (S.I. Table S4). The country-wide assessment showed that while most of Norway (86.2%) has a relatively low LCOE, half of the country (49.7%) had low cumulative LCA impact (Fig. 6). Sites with intermediate or high LCOE were respectively located in inland valleys (12.0%) and high-alpine areas (1.8%). Sites with intermediate (40.2%) and high impact (10.1%) were more widely distributed in inland regions and along the coast, respectively. After zoning, most of Norway was suitable for wind energy (81.9%), while coastal areas (15.2%) would require mitigation and inland valleys (2.9%) should be avoided. The operational wind-power plants in Norway were sited at locations with relatively low LCOE but varied along a range of impact levels (Fig. 7). Six wind-power plants (15%) were sited at locations with intermediate to high impacts.

4. Discussion

In Norway, most of the onshore wind-power plants are situated along or close to the coastline. Our results have shown that seabirds, raptors,

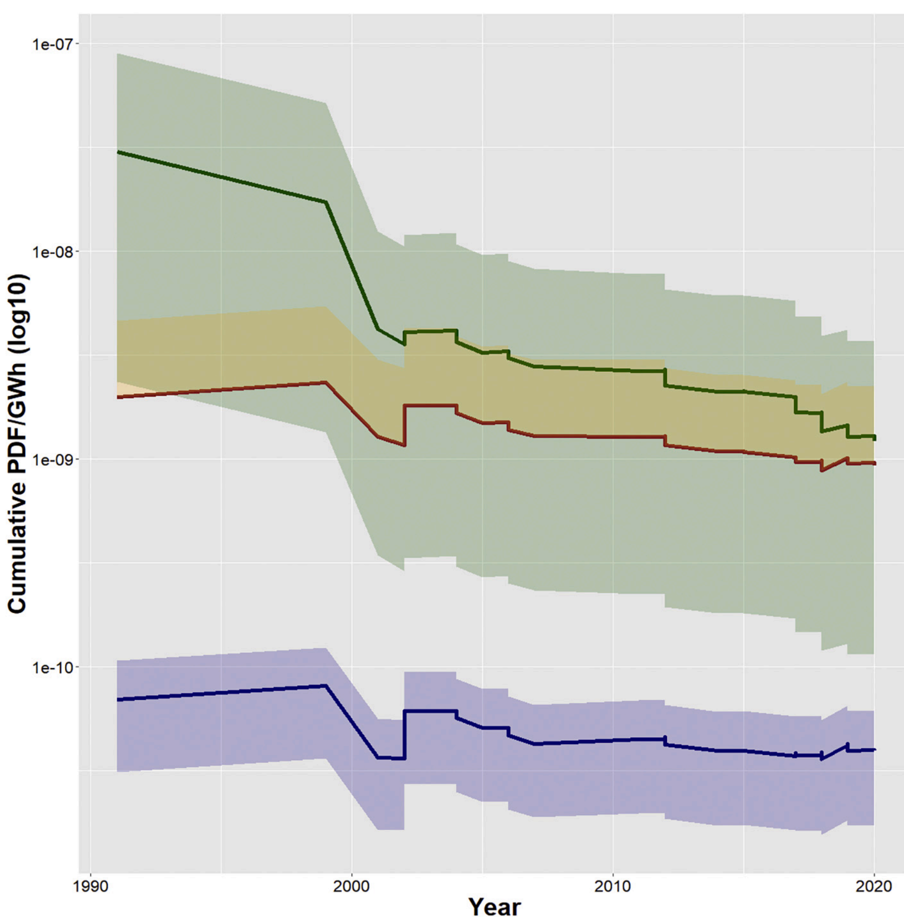

Fig. 2. Potentially disappeared fraction (PDF) of species for the three main impact pathways of operational wind-power plants on birds in Norway, both absolutely (upper panel) and relative to the annual energy production (lower panel). Direct and indirect habitat loss are given, respectively, in brown and orange shades. Disturbance in green shades (± SD), and collisions in blue shades (± SD). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

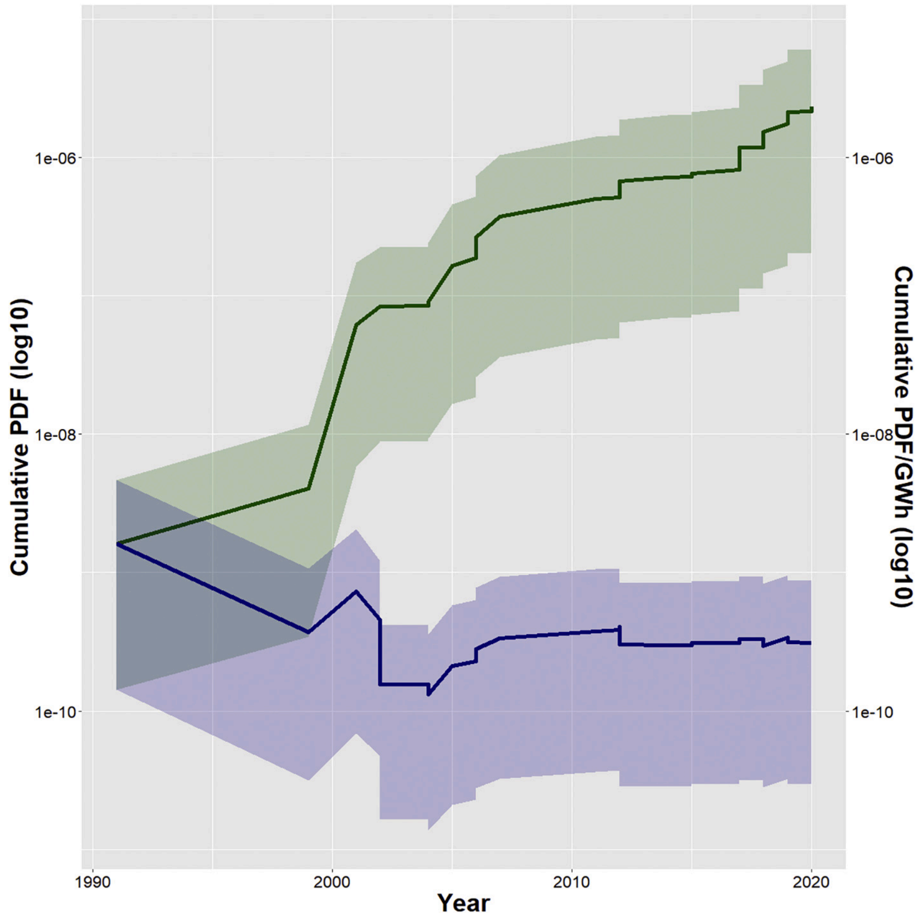

Fig. 3. Potentially disappeared fraction (PDF) of species for the barrier impact pathway of operational wind-power plants on birds in Norway (± SD), both absolutely (green shades, left y-axis) and relative to the annual energy production (blue shades, right y-axis). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

gulls and waterfowl are among the most susceptible species and are therefore negatively affected by coastal wind-power plants. This gives cause for concern, since these bird groups are overrepresented in the Norwegian Red List for species (Kålås et al., 2015; Stokke et al., 2021). To reduce (or avoid additional) negative impacts, caution should be taken in siting of future developments so as to leave especially important areas of such bird groups undeveloped. Since seabirds are among the groups that are especially vulnerable to wind power developments along the coast, this should also have implications for siting of offshore wind-power plants. In addition, many wind-power plants are located in areas of high relative bird diversity, such as in the south-western and central coastal regions of Norway. That, however, means that the wind-power plants were overall least efficiently sited for raptors and coastal birds including seabirds, waterfowl and gulls (Table 3), and best for gallinaceous birds, owls and songbirds (herbivorous and polyphagous). Even though our analyses show that gallinaceous birds are less affected by power plant developments in Norway compared to other groups, there is undoubtedly spatial variation in how such species are prone to negative effects of such developments. At Smøla wind-power plant, willow ptarmigan (*Lagopus lagopus*) and white-tailed eagle (*Haliaeetus albicilla*) are the highest-ranking species when it comes to risk of colliding with wind turbines (May et al., 2020b; Stokke et al., 2020). Still, in general siting of wind-power plants was preferred in areas with low levelized cost of energy but higher risk of disturbance. Apparently, the current practice has not succeeded in avoiding sites with higher impacts for birds (Inderberg et al., 2019; Solli, 2010; Thygesen and Agarwal, 2014). This has in part fuelled conflicts surrounding environmental concerns of wind energy development in Norway (Blindheim, 2015; Inderberg et al., 2019; Rygg, 2012; Solli, 2010; Thygesen and Agarwal, 2014). Here, we need to clarify that our approach of zoning, based on the statistical distribution of cost and impact values across Norway, did not in any way consider societal thresholds of what is acceptable. It was rather used to elucidate how estimation of impacts can be used to balance costs and impacts for siting of wind energy. In general, while the largest wind-power plants caused the largest overall impacts, smaller wind-power plants were least efficiently sited, i.e. they had the largest impacts per GWh. While turbine capacity and number of turbines in isolation increased impacts, the current development of larger wind-power plants with more powerful turbines may be expected to lead to reduced impacts. This effect has earlier also been shown for collision rates (Thaxter et al., 2017), but has now also been quantified for disturbance and habitat loss.

The model performance of the species distribution models showed considerable variation, with the polyphagous songbirds and seabirds performing worst and best, respectively (S.I. Table S3). The seabirds group consists of nine species, all having a strong affinity for coastal areas or open water. This is more likely to lead to better models. Polyphagous songbirds, on the other hand, are represented by 20 species that have very different distributions, habitat preferences and varied in the number of species-specific occurrence records (Phillips and Dudík, 2008). Thus, there is far greater variability, with the environmental predictor variables not necessarily being as good for some of these species.

The LCIA approach employed here also allows the weighing up of various impact pathways. Wind energy developments are often publicly controversial, with the impacts resulting from avian collisions being

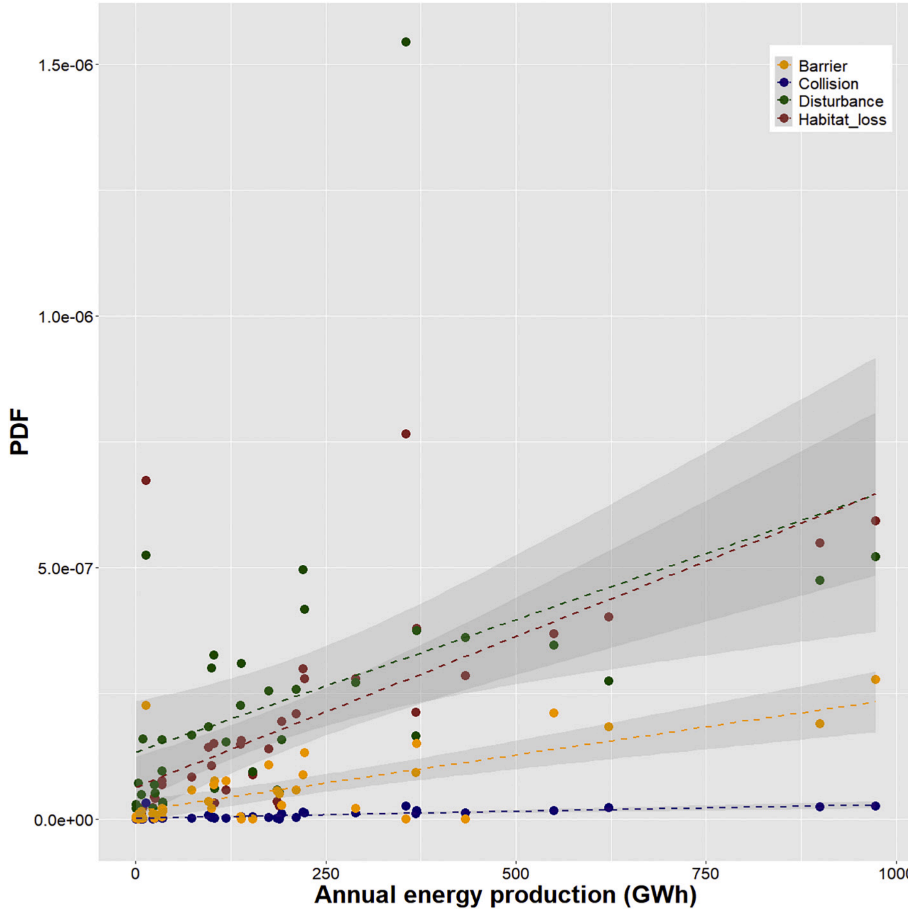

Fig. 4. Potentially disappeared fraction (PDF) of bird species per annual energy produced (GWh) for the four impact pathways for all operational wind-power plants in Norway.

Table 2
Linear regression model outcomes of the effect of wind-power plant characteristics on the (log-transformed) potentially disappeared fraction of species for the four main impact pathways of operational wind-power plants on birds in Norway. The values signify the effect size using the F statistic. The sign indicates the direction of the effect. One, two or three asterisks indicate $P < 0.05$, $P < 0.01$ and $P < 0.001$, respectively.

Covariate	Impact pathway			
	Habitat loss	Disturbance	Collision	Barrier
Turbine capacity (MW)	94.511***	9.857**	125.329***	5.465*
Number of turbines	66.151***	53.949***	77.239***	0.772
Interactive effect	−11.723**	−7.447*	−10.259**	6.466*
adjusted R^2	0.817	0.642	0.847	0.2034

particularly obvious to local residents, and may therefore attract much media attention (May, 2019; Rygg, 2012; Solli, 2010). Our results however reveal that in most instances collisions result in lower impacts than habitat loss, disturbance and barrier effects. However, while collisions result in direct mortality, the other impact pathways affect species indirectly through reduced fitness due to stress responses, increased energy expenditure or changed foraging behaviour (May et al., 2019). These less obvious impacts, rarely considered in the media, may therefore be even more important to consider (Raiter et al., 2014). Hence, energy developments may cause a local decline in some species without them necessarily becoming locally extinct. Lower population sizes in one area, however, may lead to less emigration to other areas (i.e., sink) and hence also influence the occurrence over larger areas. Predicting the impact of development requires monitoring occurrence of susceptible species at planned and operational wind farm sites, as well as (control) sites not subject to development. Balancing wind energy development with bird diversity conservation should thereby be evaluated based on the cumulative effects of all wind energy developments on a larger geographical scale (May et al., 2019).

The LCA approach, which the assessment was based on, has both strengths and weaknesses (cf. May et al., 2020a). Its strength lies in that it allows for standardized comparisons across sites or regions at various spatial scales to assess trade-offs. On the other hand, the methodology has its inherent uncertainties. These relate to the level of accuracy in capturing the true distribution and (migratory) movement of species, turbine-specific footprints and species-specific responses (i.e. disturbance distances, energetic requirements and collision rates) to estimate the factual impact. These should therefore be judged as a relative indicator for impact. As long as one is aware of the assumptions and limitations of the methodology, the proposed procedure to balance LCOE against LCA impacts will provide valuable decision-support information to identify trade-offs, compare alternative sites or trends in cumulative development and minimize impacts on biodiversity (May et al., 2020a). As the main impacts (mortality, avoidance/attraction, habitat alterations) are expected to be similar, LCA-based metrics developed for onshore wind energy can be adapted for other species (e.g. bats) and/or offshore wind energy development as well.

The resulting impacts were ~ 3 orders of magnitude higher than for the global, spatially differentiated model (see May et al., 2020a), which is reporting global impacts on species losses (i.e. extinction across the whole world, including the spatial component). The difference between regional/local and global losses is that regional impacts are only looking at the loss of a species in a certain region (e.g. Norway), while global

Table 3
Average (range) LCA impacts for the four main impact pathways of operational wind-power plants on birds in Norway relative to the annual energy production (PDF/GWh). Ranking was based on all four impact pathways (i.e. rank of summed ranks across impact).

Group	PDF / GWh (x 10^{-9})				Rank
	Habitat loss	Disturbance	Collisions	Barrier	
Corvids	0.933–2.189	0.253 (0.027–0.634)	0.052 (0.027–0.076)	0.047 (0.005–0.118)	5
Gulls	0.990–2.322	0.442 (0.173–0.718)	0.062 (0.038–0.086)	0.153 (0.060–0.249)	3
Herbivorous songbirds	0.955–2.240	0.035 (0.019–0.053)	0.035 (0.015–0.055)	0.007 (0.004–0.011)	12
Insectivorous songbirds	1.032–2.419	0.036 (0.013–0.093)	0.044 (0.022–0.065)	0.013 (0.005–0.034)	8.5
Gallinaceaous birds	0.766–1.797	0.210 (0.050–0.369)	0.020 (0.010–0.030)	0.021 (0.005–0.038)	13
Non-passerines	1.088–2.552	0.143 (0.038–0.280)	0.037 (0.006–0.067)	0.029 (0.008–0.057)	6
Owls	0.781–1.833	0.525 (0.045–1.131)	0.024 (0.007–0.041)	0.081 (0.007–0.174)	11
Polyphagous songbirds	0.976–2.289	0.048 (0.019–0.082)	0.038 (0.015–0.061)	0.014 (0.005–0.023)	10
Raptors	0.899–2.108	3.409 (0.437–8.120)	0.070 (0.024–0.113)	1.143 (0.147–2.693)	2
Seabirds	1.122–2.631	5.143 (0.413–15.251)	0.052 (0.023–0.081)	0.591 (0.048–1.711)	1
Waders	0.813–1.907	0.818 (0.099–2.166)	0.024 (0.013–0.034)	0.522 (0.063–1.380)	8.5
Waterbirds	0.864–2.027	1.644 (0.003–5.816)	0.025 (0.017–0.032)	0.491 (0.001–1.720)	7
Waterfowl	0.899–2.109	3.139 (0.083–10.950)	0.032 (0.006–0.058)	0.921 (0.024–3.156)	4

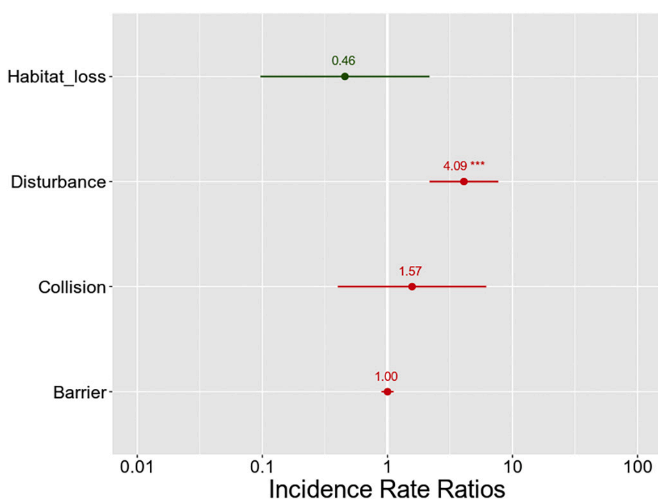

Fig. 5. Incidence rate ratios (IRR) from a discrete choice model comparing Potentially disappeared fraction (PDF) of bird species for the four impact pathways at operational wind-power plant sites to random sites in Norway. IRR below and above 1 indicate respectively lower (green) and higher (red) impact relative to random. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

losses model the global extinction of that same species, including the spatial component. Taking the global potential habitat of a species into account leads to a smaller impact, because one wind-power plant will have a smaller effect and still a larger habitat area will be available on a larger scale. On a local level, however, the loss of a certain area might mean already a substantial loss of the available habitat area and thus a substantial impact to the local population of a species. It makes sense that local impacts are higher, since species can become locally extinct more rapidly than they become extinct on a global scale. This shows the important difference between local/regional and global impacts. Kuipers et al. (2019) assessed the potential consequences of regional species loss for global species richness and proposed a conversion factor called global extinction probabilities which for birds inhabiting Scandinavian ecoregions equates to circa 3.5×10^{-4}. This likely explains most of the global to regional differences in impact magnitudes. Other potential differences might to lesser extent be explained by the finer spatial resolution and the modelling approach for mapping species occurrences. For these reasons, it is important always to evaluate impacts relative to the scope of the assessment performed.

In LCIA models, global approaches are generally preferred since they will be applicable in different parts of the world and the results will be comparable. However, as pointed out in the beginning of this article, global models often resort to the inclusion of global range maps with the assumption of equal probability of presence and thus may lead to inaccurate results. Although applied to Norway, the presented approach can theoretically be applied to any region in the world given that all input data exist. The data we used on species records and environmental variables are available from WorldCim2.0 and GBIF on a global scale. Other input variables like DEM, data for wind turbines and land cover were specific for Norway and would thus need to be available in the model region in question, to successfully apply the presented approach. A particular advantage of more regional models, like the one developed for Norway here, is that these can take more detailed data into account and derive models with a greater level of detail (e.g. 1 km^2). On a global level, such a level of detail would be, however, very difficult to handle. For local information and siting information, for comparing the impacts of birds among the different wind-energy generation sites and potential future sites, such a local approach is therefore of great value and can almost be seen as a hybrid between an LCIA and an EIA approach (Loiseau et al., 2013). An additional consideration when selecting regional-scale map areas is the extent and movement of birds across national borders, for instance during seasonal migration. Future refinements of the work presented here may look to include entire Fennoscandia instead, and specifically map seasonal migration corridors. Although larger, it would still be feasible with regards to data availability and computational modelling requirements. When considering Norway on its own, for example, the country's narrow, elongated geographical shape results in models excluding potential movement and connectivity along and through its extensive boundary with Sweden. The bird populations that span such international borders are the same and move according to local habitat and climatic conditions.

5. Conclusion and policy implications

5.1. Implications of the outcomes

Norway is committed to significant developments of onshore (and offshore) wind energy to facilitate the transition to a low-emission society by 2050. However, as wind energy deployment increases and larger wind-power plants are considered, the pressure on vulnerable bird populations become more acute (Beston et al., 2016). Bird declines and extinctions may in turn affect the ecosystem services they provide, which are critical to the health of many ecosystems and to human well-being (Whelan et al., 2015). Our methodological procedure for balancing trade-offs between wind energy development and its implications for bird diversity conservation, indicated that the operational wind-power plants along the coast of Norway caused significant impacts especially on seabirds and raptors. Current siting practice indicated that wind-power plants are constructed in the economically least costly sites, with a partly avoidance of habitat loss impacts. Technological upscaling

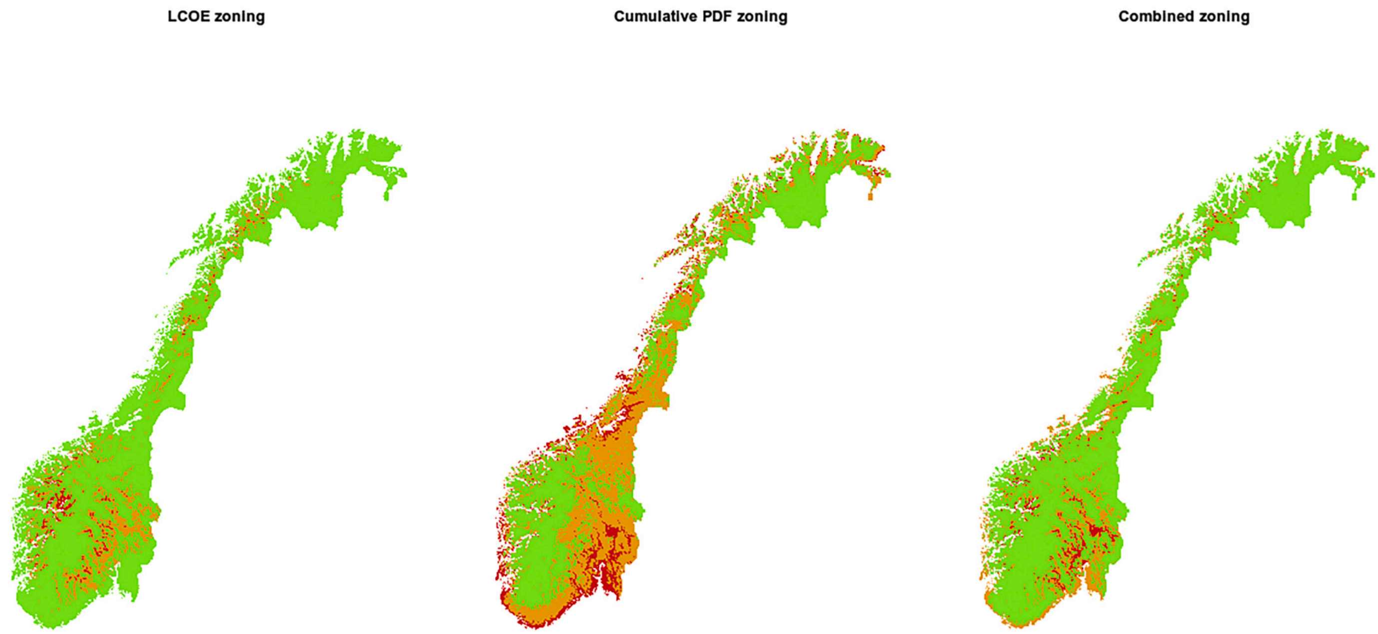

Fig. 6. Zoning maps for Norway wind energy development based on Jenks natural breakpoint thresholds (4 breaks; three groups: green, orange, red) for the Levelized Cost of Energy (LCOE) and per-turbine (3 MW, 50 m rotor blade length) cumulative Potentially Disappeared Fraction of species (PDF) across four impact pathways (habitat loss, disturbance, collision and barrier) as well as the combined zoning identifying areas suitable for development (green), requiring site-specific mitigation (orange) or to avoid (red). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

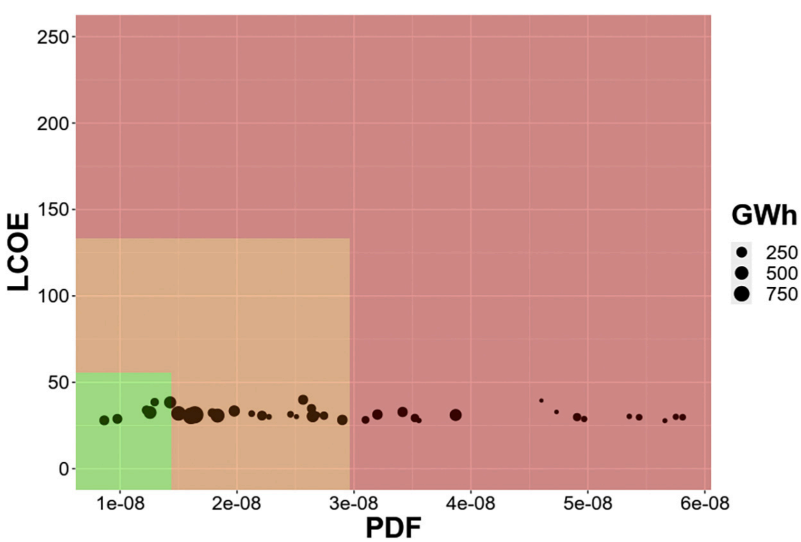

Fig. 7. Zoning for mitigation of wind energy development in Norway. Zoning is based on Jenks natural breakpoint thresholds (4 breaks) for Levelized Cost of Energy (LCOE) and per-turbine (3 MW, 50 m rotor blade length) cumulative Potentially Disappeared Fraction of species (PDF) across four impact pathways (habitat loss, disturbance, collision, barrier). Coloured backgrounds indicated regions suitable for development (green), requiring site-specific mitigation (orange) and to avoid (red). Dots indicate the per-turbine LCOE and cumulative PDF for each wind-power plant in Norway, sized after their total annual energy production (GWh). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

however seems to have resulted in lower impacts overall. There is an increasing demand for such knowledge, from consenting and environmental authorities, non-governmental organizations, the public, and wind energy developers as those will reduce the need for reactive and costly post-construction measures hampering performance and negatively affecting financial security (EWEA, 2010; May, 2019). The strength of such an approach lies in the consideration of multiple impacts to compare various alternative development options. It does however not allow the quantification of local changes in occurrences but can direct pre- and post-construction monitoring efforts at sites with acceptable impact levels, as well as cumulative effect assessments to ensure bird diversity conservation. Such a strategy will help to minimize trade-offs found in the UN SDGs and contribute to reaching more sustainable decisions (Helling, 2017).

5.2. Policy recommendations

Our procedure for zoning employing spatially explicit LCIA models can help decision-makers in picking the sites with the least impact on biodiversity, in conjunction with a complete EIA and an LCA (Jeswani et al., 2010). This may in turn contribute to improved consenting processes and reduced conflicts surrounding environmental concerns of wind energy development in Norway. Still, the lack of influence of the environmental authorities (Norwegian Environment Agency) in the consenting process for energy projects (Norwegian Water Resources and Energy Directorate) hampers holistic decision-making (Inderberg et al., 2019; Thygesen and Agarwal, 2014). Operative tools to assess life-cycle environmental impacts can however support environmental-friendly and publicly supported wind energy production. This will directly and significantly benefit technological performance: more wind energy projects will be realized with reduced environmental, and societal, impact per GWh. Effective tools that are able to identify any unintended avian conflicts already during the planning phase, allows for the development of wind-power at environmentally benign sites improving the utilization of wind resources at specific sites without increasing potential conflict levels (May, 2017). LCA may be promising tool to increase the performance of environmental impact assessments, especially at the strategic and early planning phase (Tukker, 2000; Wu and Ma, 2018).

Declaration of Competing Interest

None.

Acknowledgments

This work was executed as part of the project "*Towards sustainable renewable energy production: Developing a Life Cycle Impact Assessment framework for biodiversity impacts (SURE)*", funded by the Research Council of Norway (grant: 244109). We would like to thank Koen Kuipers, Martin Dorber and Tiago Laranjeiro for discussions during the development of the methodology.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.eiar.2021.106635.

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