



## Global life-cycle impacts of onshore wind-power plants on bird richness

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## ABSTRACT

Wind energy development continues to expand rapidly to meet international climate targets. However, these developments impact biodiversity and associated sustainability targets negatively. Environmental impact assessments (EIA) predominately focus on site-specific impacts; failing to account for cumulative impacts at larger spatial scales. Life cycle assessments (LCA) can take such a holistic view, however models quantifying the main impact pathways on biodiversity are still lacking. To address this gap, we present a methodology to quantify habitat loss, disturbance and collision impacts of onshore wind power plants on bird biodiversity globally. By overlaying species ranges with wind-power plant locations, species-area relationships were used to calculate the potentially disappeared fraction of species (PDF) for the three impact pathways. Results revealed habitat-specific bird richness values, at order level, which were greatest in tropical and subtropical regions. Temporal changes in PDF were related to operationalizing wind-power plants. Despite similar PDF curves throughout the time period, disturbance and habitat loss had a greater effect compared to collisions. Annual energy production values strongly affected overall impacts for all impact pathways, which in turn differed between continents. When controlling for continent, bird order rather than country more strongly influenced variation in the pathway-specific PDF per GWh. Our approach represents the first step towards incorporating the impacts of wind power production in strategic environmental planning and the LCA framework. Failing to account for these negates an assessment of the trade-offs between biodiversity and energy production, and therefore the balancing of global sustainability goals.

## 1. Introduction

Climate change concerns (UNFCCC 2016) and the need to reduce greenhouse gas (GHG) emissions have boosted the innovation, development and application of renewable energy sources worldwide. The global potential for wind energy generation is enormous (Lu et al. 2009) and regarded to be one of the most promising renewable energy sources. Globally, the wind energy capacity currently accounts for almost 4% of the electricity generation (2017: 497 GW onshore wind and 18 GW offshore wind; IEA 2019c). Onshore wind energy is expected to expand by over 300 GW in the coming five years (2019–2024; IEA, 2019a). This represents a forecasted 9% annual increase up to 2024 following the current policy scenario. However, reaching the Paris Agreement's objective to hold the rise in global temperatures to “well below 2°C ... and pursuing efforts to limit [it] to 1.5°C” by 2030, as well as related Sustainable Development Goals (SDG3 – Good Health and Well-being, SDG7 – Affordable and Clean Energy, SDG13 – Climate Action), requires an

annual increase in onshore wind energy generation of 12%. This represents a yearly net capacity expansion from 47 GW in 2018 to 108 GW in 2030 (IEA, 2019b). At the same time, the IPCC Special Report on Renewable Energy (IPCC 2011) stressed that “environmental and social issues will affect wind energy deployment opportunities”. For example, the development of renewable energy adds to the pervasive and continuous anthropogenic land-use change causing unprecedented declines in biodiversity (Pimm et al. 2014; Rounsevell et al. 2018).

While renewable energy production contributes to reducing GHG emissions, its impacts on biodiversity are thus cause for concern (Köppel et al. 2014; May et al., 2017). Environmental concerns regarding onshore wind energy development have focused primarily on birds and bats, assessing in particular the effect of habitat alterations and disturbance within the wind energy footprint, and collisions with wind turbines (Schuster et al. 2015; May et al., 2017; Laranjeiro et al., 2018). However, current knowledge is mostly restricted to the assessment of effects at single wind-power plants due to limitations in data availability and

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site-specific study design. It is consequently difficult to assess how these effects upscale to population level impacts or what cumulative effect multiple wind-power plants have on species and biodiversity at larger (e.g. continental) scales (May et al., 2017, 2019).

Balancing ambitious renewable energy targets and biodiversity conservation requires foremost holistic spatial planning approaches to ensure the lowest possible environmental costs per kWh (Warren et al. 2005). A central element to include in strategic planning of renewable energy technologies will be the quantitative and spatially explicit mapping of both (scaled) climatic and environmental costs/benefits for sustainable development alternatives (Evans et al. 2009; Allison et al. 2014). This requires the scoping of established criteria for assessments, as well as setting thresholds for acceptance using equal metrics. This will enable stakeholders to select among the options for balancing climatic benefits versus environmental costs on the same premises. Therefore, we look to tools that can perform product-related assessments across multiple impact categories, where wind-energy specific impacts on biodiversity can be added.

Environmental impact assessments (EIA) inform policy makers during the decision-making process and are important to minimize the potential environmental impacts of industrial developments. However, although EIAs can evaluate potential positive and negative environmental impacts of a planned project, they are less suited to address global and regional environmental effects spanning a project's life cycle (Chang et al. 2013; Hoffman 2017). Life cycle assessment (LCA), on the other hand, is a standardized method to evaluate various environmental impacts of e.g. climate change and ecosystem consequences of electricity production in a holistic manner (Verones et al. 2017), thereby identifying potential trade-offs between development strategies (Hellweg and Mila i Canals, 2014). LCA characterizes the technological system and assesses the environmental impacts occurring throughout a system's life cycle, considering various impact pathways simultaneously, and often in a spatially differentiated manner. This makes LCA an appropriate tool for assessing the total environmental consequences of e.g. technological systems, comprising various stages that affect ecosystems in different ways and at different locations (Verones et al. 2017).

LCA has developed rapidly over recent decades, incorporating more and more complex environmental impacts such as effects of land use on biodiversity (de Baan et al., 2013a,b; Chaudhary et al. 2015; Woods et al. 2017; Kuipers et al. 2019). Still, assessments of environmental impacts from wind energy mostly focus on greenhouse gas emissions and energy accounting (Arvesen and Hertwich 2012; Al-Behadili and El-Osta 2015; Alsaleh and Sattler 2019; Gomaa et al. 2019; Wang et al. 2019). As yet, LCAs are not able to assess the actual impacts on the natural environment as part of the total life-cycle load of renewable energy, necessitating further methodological improvements applicable to wind energy development (May et al., 2012). While wind energy performs favourably regarding GHG emissions, the mere assessment of emissions alone does not constitute a complete LCA, which is necessary for a thorough and complete decision-making process both on a strategic (e.g. for cumulative effects) and operational (e.g. siting) level. However, currently no life cycle impact assessment (LCIA) models exist that can quantify the main impact pathways of wind energy development on biodiversity. To address this methodological gap, we present for the first time a LCIA methodology aimed to quantify the habitat loss, disturbance and collision impacts of onshore wind-power plants on bird biodiversity, and then apply this approach on a global level.

## 2. Materials and methods

### 2.1. Species-area relationship

The quantification of the impacts of wind-power plants on birds at the order level is based on the generic concept of a species-area relationship (SAR), which is widely used in LCA (de Baan et al., 2013a.; de Baan et al., 2013b; Chaudhary et al., 2015). SARs predict the effect that area loss has

on species richness. Species richness loss in LCIA is normally quantified in units of potentially disappeared fraction of species (PDF), which is a measure of the potential loss of species richness due to a given stressor (Eq. (1) and Eq. (2)). Different types of SARs are in use in LCA, such as the classical SAR, matrix-calibrated SAR or countryside SAR. The latter two account for habitat heterogeneity and the fact that species may survive in human-modified landscapes as well. We use the classical SAR for our purposes here, and instead introduce a probability of presence including a habitat-specific relative diversity (see section 2.2).

$$\frac{S_{new}}{S_{org}} = \left[ \frac{A_{new}}{A_{org}} \right]^z \Leftrightarrow S_{lost} = S_{org} - S_{org} \cdot \left[ \frac{A_{new}}{A_{org}} \right]^z \quad (1)$$

$$PDF = \frac{S_{lost}}{S_{org}} = \frac{S_{org} \cdot \left( 1 - \left[ \frac{A_{org} - A_{lost}}{A_{org}} \right]^z \right)}{S_{org}} \quad (2)$$

The number of species remaining (Snew) and lost (Slost) at a given site depend on the effect proportional area conversion of the original habitat area (Aorg) to the remaining habitat area (Anew) has on the original number of species (Sorg). z represents the slope of the species-area relationship on a logarithmic scale. We used the continental-scale z-values from Storch et al. (2012), where we differentiated between lower and upper slopes to account for restricted and wide geographic ranges, respectively (nested quadrat design; area threshold at 3 × 106 km<sup>2</sup>). Species richness does not necessarily need to be distributed uniformly across the landscape, however, and therefore the number of species present at any given site may vary from location to location. Hence, we take a localized approach where Sorg at a given site i (i ∈ [1, I]) depends on the probability of presence (Pi) for all species S across their combined range: Sorg = SnPi (Eq. (3)).

$$PDF_i = \frac{SnP_i n \left( 1 - \left( \frac{A_{org,i} - A_{lost,i}}{A_{org,i}} \right)^z \right)}{\sum_{i=1}^I SnP_i} \quad (3)$$

### 2.2. Mapping species occurrence

The spatially-explicit global assessment was based on BirdLife range maps (status: November 21, 2019) (BirdLife International, 2018) indicating (binary) presence of species s within bird order k. As some bird orders consist of a limited number of species, or lack necessary data (see further below), we performed the analyses on 18 bird orders (Table 1). The probability of presence Pk,i at any given site i was defined as the proportional species richness of the local number of species Sk,i to the total number of species Sk within bird order k across their combined continental range (Eq. (4)) and including a factor accounting for habitat-specific relative diversity hk,l (Eq. (6)). The number of species (Sk,i) within bird order k present at any given site i across its combined range is however unlikely to be uniform (and binary) due to species' migratory status and relative habitat preferences (Chaudhary et al., 2015).

$$P_{k,i} = \frac{S_{k,i}}{S_k} n h_{k,i} \quad (4)$$

We accounted for migratory species that are only present half of the year in their breeding or wintering areas, respectively, by using a weighted sum for the number of local species over their migratory status (with ms as: resident = 0, migratory = 1) (Eq. (5)). Migratory status was determined by the presence of separate seasonal ranges for each species s (s ∈ [1, Sk]).

$$S_{k,i} = \sum_{s=1}^{S_k} s_{k,i} n 2^{-m_s} \quad (5)$$

Furthermore, to account for habitat-specific relative diversity (i.e.

bird order-specific habitat preferences), we multiplied the proportional species richness with a land-cover based relative species diversity factor ( $h_{k,l}$ , see eq. 6). Land cover information was obtained from the 300-m global European Space Agency Climate CI Land cover map of 2015 (version 2.0.7; ESA 2017); reclassified into 18 categories (Supplementary Information, Table S1) and resampled to a  $10 \times 10$  km resolution using the majority algorithm. The relative diversity factor  $h_{k,l}$  was defined as the ratio of the effective number of species ( $ENS_{k,l}$ ) per land cover type  $l$  to the maximum number of species  $Sk$ .  $ENS_{k,l}$  was based on the exponential of the Shannon diversity index for true alpha diversity (Eq. (6)).

$$h_{k,l} = \frac{ENS_{k,l}}{S_k} = \frac{e^{-\sum_{s=1}^{S_k} p_{s,l} \ln(p_{s,l})}}{S_k} \quad (6)$$

Here,  $p_{s,l}$  equals the proportion of a species' range covered by land cover type  $l$ . In principle different orders of the Hill numbers can be used to calculate  $ENS_{k,l}$ , such as zero-order for richness (not considering relative abundances) and second-order for the inverse Gini-Simpson index (higher weight to common/abundant species) (Jost 2006). The rationale for using the first-order Shannon diversity index was that it is frequency-weighted, without favouring rare or common species. The resulting global relative bird richness maps give the probability of presence of bird orders at a  $10 \times 10$  km resolution.

### 2.3. Mapping of wind power plants

The following PDF's for habitat loss, disturbance and collision mortality are calculated using  $P_{k,i}$  at the centroid of each wind-power plant. The coordinates (latitude, longitude), commission year, number of turbines, total power (kW) and turbine rotor diameter (m) of all recorded onshore wind-power plants globally ( $N = 23,068$ , Table S2) were obtained from TheWindPower.net (status: October 22, 2019). The database contains in addition the records for over 800 offshore projects, which were outside the scope of this study. Only onshore wind-power plants that are either approved, under construction or in operation were included in the analyses. As the rotor diameter was unknown for 21.8% of all wind-power plants, we estimated those by regressing known rotor diameters against per turbine power output after log-log-transformation using linear regression (intercept:  $0.280 \pm 0.008$  SD, slope:  $0.529 \pm 0.001$  SD;  $F = 213,455$ ,  $P < 0.001$ ,  $R^2 = 0.922$ ).

### 2.4. PDF for habitat loss (H)

After construction of a wind-power plant  $w$ , habitat is lost surrounding each turbine base. This affects species within the transformed area approximated by the direct and indirect area requirements per MW capacity  $aEP$  of respectively  $0.3$  ha/MW and  $0.7$  ha/MW (Denholm et al., 2009) within wind-power plant  $w$  with electrical power  $EP_w$  relative to the total area available to the species within bird order  $k$  at site  $i$  ( $A_{org} = 100$  km<sup>2</sup>, i.e. spatial resolution of  $10 \times 10$  km) (Eq. (7)). We applied both direct and indirect values for  $aEP$  to assess impacts ranges for our PDF estimations.

$$PDF(H)_{k,w} = \frac{S_k n P_{k,i} n \left( 1 - \left( \frac{A_{org,i} - aEP n EP_w}{A_{org,i}} \right)^z \right)}{\sum_i S_k n P_{k,i}} \quad (7)$$

### 2.5. PDF for disturbance (D)

Although the immediate area surrounding the turbines is lost as habitat, it may still be used by birds for crossing this non-habitat patch to distinct portions of their habitat. The disturbance effect is therefore additive to habitat loss. Contrary to habitat loss, disturbance has a relative effect whereby a proportion of the species are lost from the area. The disturbance effect is measured as the proportion of species displaced from the influence area at wind-power plant  $w$ , predicted using the

relative integral of the sigmoid function relating the proportion of species displaced over distance ( $D_k$ ) (Eq. (8)). This function was derived from flight initiation distances (FID) for the species within bird order  $k$ , with  $dk_{min}$ ,  $\bar{d}_k$  and  $dk_{max}$  representing the minimum, mean ( $\pm$ SD) and maximum species-specific FIDs. The maximum FID ( $dk_{max}$ ) delineates the extent of the influence area surrounding each turbine for all turbines within a wind-power plant ( $tw$ ). FIDs were obtained from the literature. If more than one value was found per species, the largest FID was taken (Table S3). Such data was not available for all species within each bird order (or lacking completely for some orders). Therefore, the available species-specific FIDs were assumed to be representative of the bird order in question.

$$PDF(D)_{k,w} = \frac{S_k n P_{k,i} n \left( 1 - \left( \frac{A_{org,i} - t_w n \left( \frac{\pi n (D_k n d_{k,max})^2}{A_{org,i}} \right)^z \right)}{\sum_i S_k n P_{k,i}} \right)}{\sum_i S_k n P_{k,i}} \quad (8)$$

with  $D_k = \int_{d=0}^{dk_{max}} \frac{1 - \frac{1}{(1 + \beta(d - \bar{d}_k)^{\alpha})}}{(dk_{max} - dk_{min})} dd$  and  $\beta = \frac{\log\left(\frac{2 - \alpha}{\alpha}\right)}{(dk_{min} - \bar{d})}$  and  $\alpha = 0.1$  as level for the confidence interval. We derived both mean as well as lower and upper ( $\pm$ SD) values for  $D_k$  to assess uncertainty in our PDF estimations.

### 2.6. PDF for collision mortality (C)

Collision affects the presence of individuals within a bird order within the area of influence surrounding all turbines at wind-power plant  $w$ . Generically, the PDF for collision is quantified as the reduction of the species at risk due to collision. The number of species at risk are those that are utilizing the influence area surrounding each turbine delineated by rotor swept zone (with  $rw$  as rotor blade length, Eq. (9)).

$$\frac{S_{at\ risk}}{S_{org}} = \frac{S_k n P_{k,i} n \left( 1 - \left( \frac{A_{org,i} - t_w n \left( \frac{\pi r_w^2}{A_{org,i}} \right)^z \right)}{S_k n P_{k,i}} \right)}{S_k n P_{k,i}} \quad (9)$$

However, not all species at risk will be lost due to collision. Collisions affect both the local abundance of individuals present at one location ( $n$ ), but also cumulatively the total abundance of individuals across the landscape ( $N$ ). Here we assume that abundance is relative to the relative use of area:  $\frac{A_i}{\sum_{i=1}^N A_i} \sim \frac{n}{N}$  (Boyce and McDonald 1999). The probability of collision hereby reduces the area of influence surrounding each turbine (Eq. (10)). The collision probability ( $R_k$ ) is approximated using the species-specific collision rates (per turbine/year) averaged ( $\pm$ SD) for each bird order (Thaxter et al., 2017).  $R_k$  is calculated by transforming the Poisson-distributed collision rates (ratek) to the probability for obtaining at least one collision occurring per year (of the local population present):  $R_k = 1 - e^{-ratek}$ . We derived both mean as well as lower and upper ( $\pm$ SD) values for  $R_k$  to assess uncertainty in our PDF estimations.

$$PDF(C)_{k,w} = \frac{S_k n P_{k,i} n \left( 1 - \left( \frac{A_{org,i} - R_k n t_w n \left( \frac{\pi r_w^2}{A_{org,i}} \right)^z \right)}{\sum_i S_k n P_{k,i}} \right)}{\sum_i S_k n P_{k,i}} \quad (10)$$

### 2.7. Aggregation to characterization factors and statistical analyses

The characterization factors (CF) for each of the impact pathways ( $X$ ) can be derived by aggregating PDF's across bird orders ( $k \in [1, K]$ ) and dividing the cumulative PDF's by the annual energy production ( $E_w$  in GWh) per wind-power plant  $w$ . Annual energy production was estimated based on the country-wise renewable electricity capacity and generation statistics accessed from IRENA's Query Tool (status: August 28, 2020) (IRENA 2020). Annual production was estimated by multiplying energy capacity (MW) by the average full load hours per year (2134 h, 95% C.I.: 1952–2316). Global (or regional) estimates for each of the impact pathways or the summed impacts are thereafter derived by summing CFs

across wind-power plants ( $w \in [1, W]$ ).

$$CF(X)_w = \frac{\sum_{k=1}^K PDF(X)_{k,w}}{E_w} \quad (11)$$

We statistically assessed to which extent the variation in PDF was affected by annual energy production, country and order. First, we used multivariate analysis of variance to estimate how variation in PDF impacts (log-transformed) aggregated by country and order could be explained by either production (log-transformed) and continent. Thereafter, we used linear mixed-effects models to further assess the effect of country and order on the biodiversity impact per unit of stressor (PDF/GWh) in each of the three main impact pathways by using PDF impacts (log-transformed) as response variable, annual energy production as offset (log-transformed) and including both country ISO code and bird order as explanatory factors, while controlling for random effects of continent. These models were built using the lmer function of the lme4 library (Bates et al. 2015). All mapping, calculations and analyses were executed within the R 3.6.1 environment (R Core Team, 2019) using RStudio version 1.2.1335. All scripts, input as well as output files can be found in the Supplemental Information data files.

### 3. Results

Based on the BirdLife range maps, order-specific species richness maps were made incorporating affinity to land cover as well as migratory status. Fig. 1 shows the aggregated global species richness across bird orders, highlighting the overall larger number of bird orders in tropical and subtropical regions. On average, the highest affinity was found for broadleaved tree cover, (mosaic of) cropland, shrubland and grassland (Table S4). Affinities were lowest for generally poor-quality environments, such as permanent snow and ice, lichens and mosses, but also needle-leaved tree cover. Order-specific species richness maps are found in the Supplemental Information data files. These maps formed the basis for calculating the characterization factors of wind-power plants.

Over the years, the PDF increased with the increasing rate of wind-power development (Fig. 2, left-hand panel). Especially from the onset of the development until 1986, the rate of change in impacts was steep. From 1995 and onwards a new, albeit more gradual increase occurred which over time has slowly but steadily levelled out. After the initial years, the biodiversity impact per unit of stressor (PDF/GWh) in developing wind energy has gradually decreased until 2000 (Fig. 2, right-hand panel); only the biodiversity impact per unit of stressor pertaining to

disturbance started to level off from 2010 and onwards. The biodiversity impact per unit of stressor was lowest, with the highest impact (PDF) per GWh, in 1985–1986 when several large wind-power plants consisting of many smaller turbines were developed (e.g. Green Ridge Power, USA (1984, 1384 0.1 MW wind turbines, 293 GW h), Solano County, USA (1985, 600 0.1 MW wind turbines, 120 GW h), Muppandal, India (1986, 3000 0.5 MW wind turbines, 3000 GW h)). From 1995 and onwards, the biodiversity impact per unit of stressor improved concurrent with the development of more powerful wind turbines.

Despite a qualitatively rather similar shape of the cumulative PDF curve for habitat loss, disturbance and collisions throughout the time period (Fig. 2), the effects of disturbance and habitat loss have been larger than that for collisions throughout. Of the three impact pathways, disturbance had wider variability intervals compared to collision and habitat loss.

Annual energy production strongly affected the overall impacts for all impact pathways (Pillai trace = 0.713, approximated  $F = 1620.83$ ), while the impact pathways were much affected by their continental location (Pillai trace = 0.043, approximated  $F = 7.11$ ). While controlling for continent, further assessment revealed that the variation in the pathway-specific PDF per GWh (using an offset term) was more affected by bird order than by the country where wind-power plants were located. The effect size of the explanatory value for bird order and country (indicated by the  $F$  statistic) was apparent for all three impact pathways: habitat loss (28.804 versus 8.131), disturbance (46.200 versus 8.509) and collision (29.109 versus 7.885). The country-specific differences in the biodiversity impact per unit of stressor (Fig. 3, cf. Table S2) can in part be related to their development history as well as overall bird diversity (cf. Fig. 1). The four top-most impacted bird orders overall (Anseriformes, Accipitriformes, Falconiformes and Charadriiformes) all had disturbance impacts exceeding impacts due to habitat loss (Fig. 4). In all other bird orders, impacts of habitat loss exceeded effects of disturbance. Impacts of collisions were in general markedly lower than impacts from habitat loss and disturbance. Focusing on collision impacts, the orders Accipitriformes, Falconiformes, Charadriiformes, Podicipediformes and Ciconiiformes were most affected.

### 4. Discussion

The development of large-scale infrastructure typically has diverse consequences, many of which are not immediately apparent at a given project site (e.g. functional habitat loss). Furthermore, cumulative effects

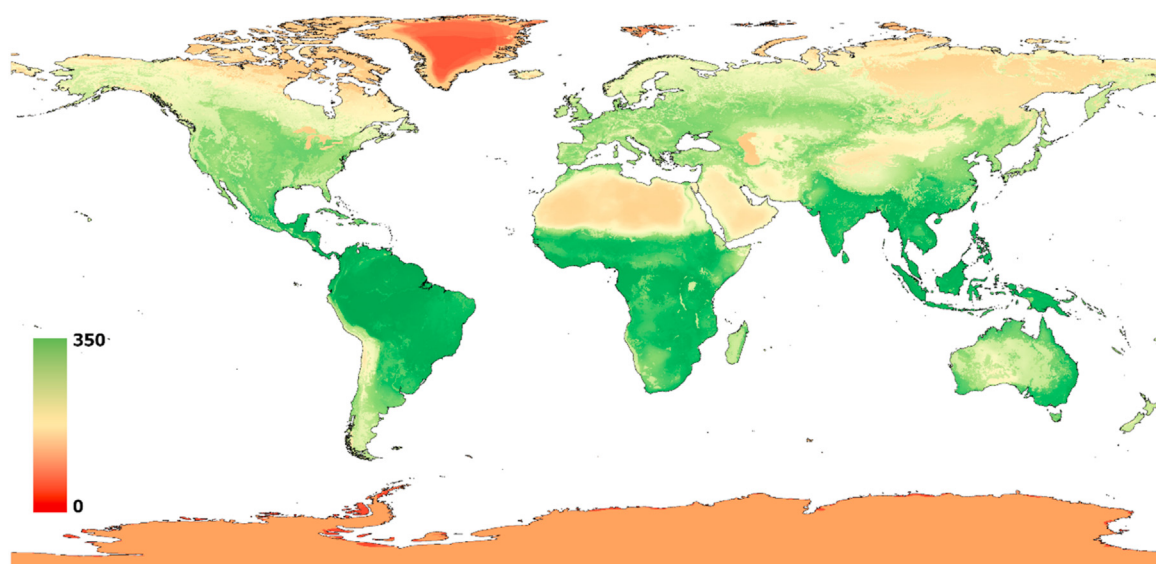
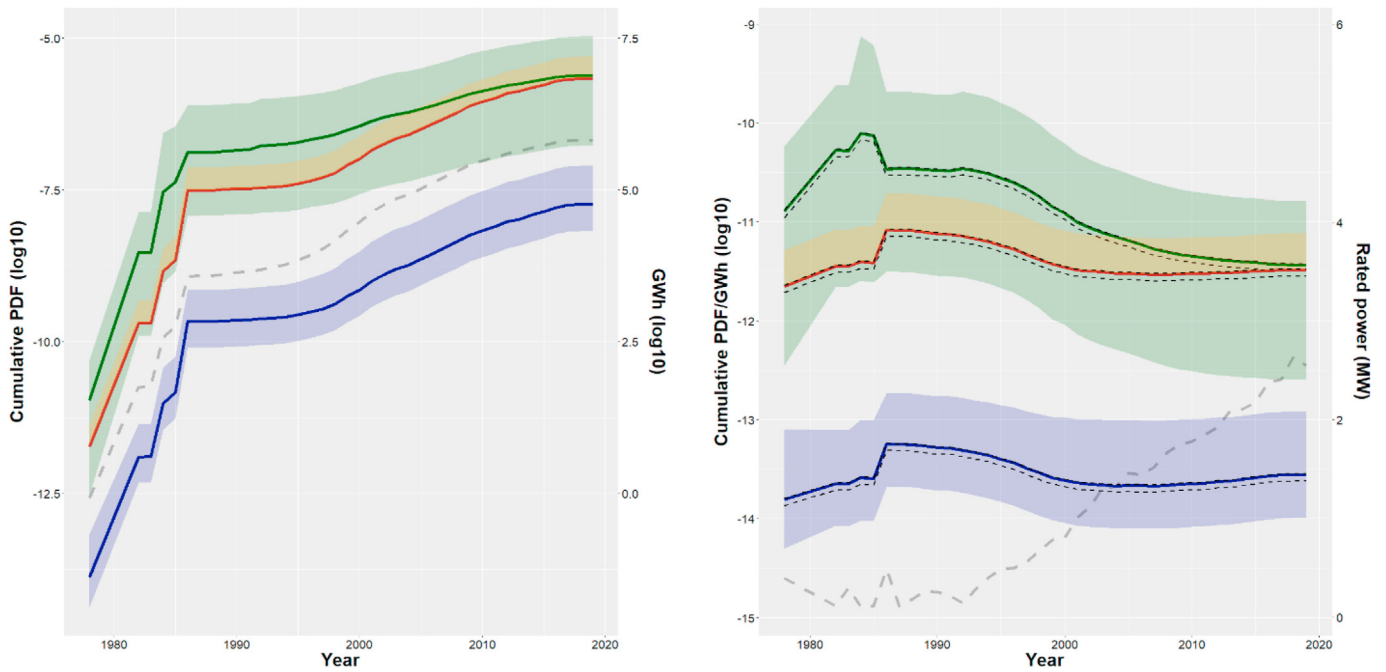
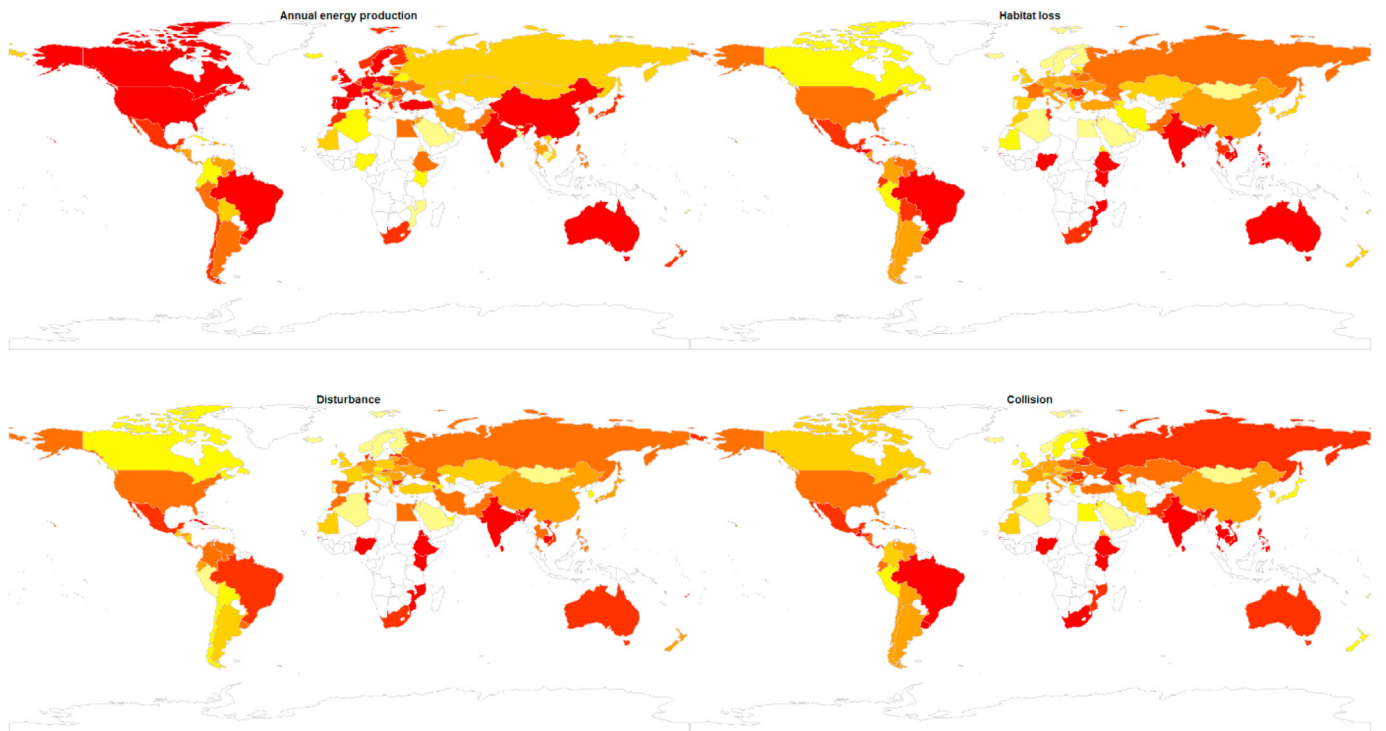


Fig. 1. Global bird richness as derived from BirdLife range maps.



**Fig. 2.** Absolute (left-hand panel) and relative (right-hand panel) cumulative Potentially Disappeared Fraction (PDF) of bird richness per order for the three main impact pathways of wind energy production: habitat loss (orange), disturbance (green) and collision (blue). The dotted grey line indicates, respectively, the annual energy production (GWh) and average rated power (MW) for the left- and right-hand panel. Shaded areas indicate the uncertainty bands of the estimates. Dotted lines indicate the uncertainty bands (95% C.I.) for the annual energy production. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)



**Fig. 3.** Country-wise annual energy production (top-left), and Potentially Disappeared Fractions of bird orders for the three main impact pathways of wind energy production: habitat loss (top-right), disturbance (bottom-left) and collision (bottom-right). Colours indicate quartiles from low (yellow) to high (red). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

from similar developments at different geographic locations have population-level effects and are therefore important to consider. Despite the rapid proliferation of wind power plants across the globe, a lack of suitable methodology has to date prevented a more holistic assessment of

impacts on bird populations. We have focused mainly on the impacts occurring during the operational phase of wind energy deployment. While habitat loss first occurs during the construction phase, it will have lasting effects over the entire lifespan of a wind-power plant. Still, during

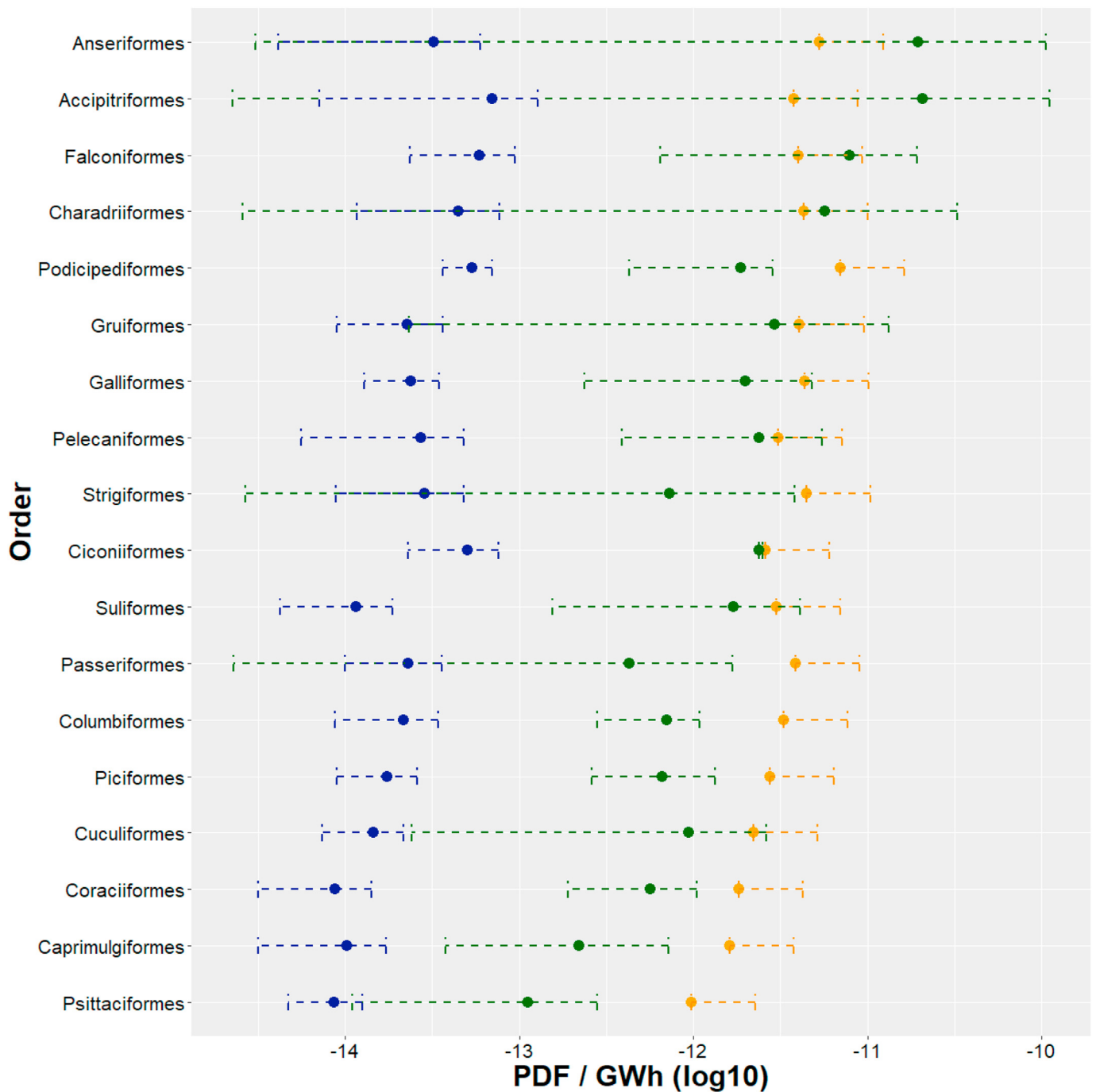


Fig. 4. Potentially Disappeared Fractions of bird richness per order per GWh for the three main impact pathways of wind energy production: habitat loss (orange), disturbance (green) and collisions (blue). Bird orders are decreasingly ordered by their total summed impact from the top down. Whiskers indicate the variability range in the impact across species. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

the construction phase specific and shorter-term effects may occur, such as disturbance due to increased human activity and traffic, construction noise and vibrations (Wszolek et al. 2014).

Bridging this methodological gap for the first time, is not without uncertainties. We used static maps of potential bird distribution and a land cover map from 2015. In addition, we used all land-based wind park information available to us (up to October 22, 2019), 2628 new wind onshore power plants (out of a total of more than 29'000) have been established, most of which (1579) are in Europe (793 of them in Germany), followed by Asia (512) and the Americas (505). We might have missed the associated land cover change for the affinity factor *h* for these recent establishments. However, for example in Germany, less than 40%

of the new wind-power plants are sited at new locations. Thus, we consider the uncertainty introduced with the difference between 2015 and 2019 to be minor. Another potential source of uncertainty is the per-MW footprint of turbines derived from Denholm et al. (2009). Given the fast rate of technological development of wind turbines, footprints will likely have increased in line with increased hub height and rotor diameter. Still, whether per-MW footprints have remained stable since 2009 is unclear. As long as there is no verified information on the factual footprint for modern turbines, we feel the used approach will represent the current best practice. The assessment of variability in annual energy production and full load hours per year is beyond the scope of this study. The electricity output per turbine depends not only on the wind regime, but also

on wind turbines specifics (rated power, swept area, power curve) and operational variability (Hoogwijk et al. 2004; Ciupăgeanu et al. 2019). We have therefore limited inclusion of uncertainty regarding annual energy production to the variance in country-wise full load hours per year.

We found that overall, the characterization factors vary over four orders of magnitude (Fig. 4). Impacts per GWh of electricity produced are per order smallest for collision. Habitat loss is for most orders the largest impact category, albeit the difference to disturbance can be small. These are the impacts per GWh of electricity produced, as commonly reported for LCIA models. However, Fig. 3 shows that the impacts are not distributed in the same way across countries. The five largest wind energy producers are China, USA, Germany, India and Brazil (GWEC 2019). Interestingly, none of the top-three producers (China, USA, Germany) are among those with the highest impacts. Impacts are instead dominated by India and Brazil, which is related to them having a larger bird diversity (Fig. 1). Species richness, but also presence of migratory birds, is larger and thus the likelihood of species being affected is greater.

Among the five orders with the highest overall impact there are three (Anseriformes, Charadriiformes and Podicipediformes) that are linked to wetland systems and marine intertidal habitats globally, while two (Accipitriformes and Falconiformes) have their habitat predominantly in forests, open areas and shrubland (cf. Table S5). Overall, those bird orders with a narrower niche breadth (cf. Table 1), typically with a higher occurrence in (sub)tropic regions, enable those species to avoid overlap with wind energy development. Thus, in countries such as Brazil not only is a large species diversity present, but also the home of some of the more vulnerable species. In assessments like LCA, it is important to not only investigate the impacts on species richness, but also to have a look at the vulnerability of the present species to impacts (Kuipers et al. 2019), in order to minimize impacts. Our approach allowed us to separate impacts per bird order each with their specific vulnerabilities for habitat loss, disturbance and collisions. Species, also within a specific bird order, may vary in their vulnerability to wind turbines. In our approach the range of species-specific vulnerabilities within each order was accounted for by considering not only the average impact, but also calculating the lower and upper variability bands. Within-order variability was highest for the disturbance impact pathway. A standardized LCA metric (i.e. PDF) further allows directly comparing impacts across pathways.

Every year hundreds of thousands of birds are killed through collisions

**Table 1**

Bird orders included in the assessment of life-cycle impacts of onshore wind energy on global bird richness, as derived from BirdLife range maps. Levin's niche breadth was calculated from the bird order-specific habitat diversity factors (Table S4). The disturbance effect was calculated as  $D_k d_{k,max}$  per bird order  $k$ . The collision probability ( $R_k$ ) was calculated as  $1 - \exp(-rate_k)$ . For more details refer to the main text and Supplemental Information (Table S5).

Bird order	Number of species	Levin's niche breadth	Disturbance effect	Collision probability
Accipitriformes	251	0.744	90.2	0.072
Anseriformes	176	0.870	75.3	0.023
Caprimulgiformes	597	0.575	14.3	0.023
Charadriiformes	383	0.861	44.7	0.040
Ciconiiformes	20	0.765	36.1	0.077
Columbiformes	366	0.571	18.0	0.026
Coraciiformes	188	0.604	21.2	0.019
Cuculiformes	151	0.648	24.7	0.025
Falconiformes	66	0.767	55.0	0.058
Galliformes	308	0.661	26.3	0.021
Gruiformes	192	0.716	33.2	0.022
Passeriformes	6646	0.637	13.0	0.023
Pelecaniformes	115	0.734	34.0	0.033
Piciformes	485	0.625	19.0	0.024
Podicipediformes	23	0.827	20.1	0.030
Psittaciformes	415	0.543	13.7	0.033
Strigiformes	241	0.674	15.6	0.025
Suliformes	54	0.780	29.2	0.015

with wind power plants in the USA alone (Loss et al. 2015). In addition come impacts from the other impact pathways presented here, plus impacts from various anthropogenic causes, such as collisions with power lines, which amount to several million birds annually (Loss et al. 2015). However, as apparent from the output of the present study, habitat loss and disturbance stand out as more important contributors to PDF for all bird orders represented here. Interestingly, the bird orders most likely to collide with wind turbines are largely similar to those likely to collide with power lines (see e.g. Avery 1978; Bevanger 1998). Large soaring birds represented in orders Accipitriformes and Ciconiiformes were among the most likely to be influenced by collisions. Such species are known to have elevated collision rates (cf. Table 1) due to their attraction to areas with thermal or orographic updrafts enabling soaring flight (Barrios and Rodriguez 2004; Santos et al. 2017). Falconiformes and gull species (suborder Lari) within the order Charadriiformes also use soaring and hovering flight. In addition, many species within the order Charadriiformes perform aerial display flights at rotor swept height (e.g. suborders Scolopaci, Charadrii); many waders are in addition migratory. The high collision impacts for Podicipediformes are likely the result of a relatively high overlap of their distribution with operational wind-power plants. While their collision probability is not especially high in itself (nor their distribution effect), both their impacts for collision and habitat loss are relatively high. Disturbance impacts were highest for raptors (Accipitriformes, Falconiformes) and water birds (Anseriformes, Charadriiformes). These bird orders had the highest maximum FID (cf. Table 1) and for Falconiformes also a relatively high proportion that was disturbed across species within the order. Studies have shown that FIDs increase with body size as well as age at first reproduction (Blumstein 2006), and also with wing morphology and migratory status (Fernández-Juricic et al. 2006). The four most affected bird orders all share some of these characteristics.

## 5. Conclusions

In LCA studies the aim is to assess the overall impacts, over the entire life cycle of a product, e.g. a wind power plant or a device using energy. However, these assessments are often biased due to a lack of impact pathway coverage. Despite all their benefits, renewable energy systems, such as wind (Laranjeiro et al., 2018) and hydropower (Dorber et al., 2018; Dorber et al. 2019) may contribute to impacts on biodiversity. The UN Sustainable Development Goals strive to both achieve clean energy (SDG 7) and protecting life on land (SDG 15 on terrestrial ecosystems). However, trade-offs exist between these two and multiple other SDGs. LCA can play an important role in highlighting where such trade-offs exist, to allow the stakeholders to decide how they want to minimize the overall impacts on a global level. Due to a lack of models, a comprehensive assessment of the impacts of electricity production, including the impact of transmission and distribution grids (Biasotto and Kindel 2018), has up to now not been possible in LCA. This methodology is the first step towards including the impacts of wind power production in strategic environmental planning. With methodological improvements like this, existing life cycle impact assessment methods like ReciPe (Huijbregts et al. 2016), Impact World+ (Bulle et al. 2019) and LC-IMPACT (Verones et al. 2020) can be complemented. Therefore, impacts of future renewable energy expansion scenarios can be assessed in a more complete and comparative manner and thus ultimately contribute to choosing the scenario with the lowest overall environmental impacts, by taking not only climate change, but also considerations about biodiversity loss into account. This in turn can inform decision-makers with strategic planning or priority setting and evaluate consequences of renewable energy policy implementation, to achieve a more sustainable wind energy development.

## Declaration of competing interest

The authors declare that they have no known competing financial

interests or personal relationships that could have appeared to influence the work reported in this paper.

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## Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.indic.2020.100080>.

## Data archiving statement

Data files and analyses codes are made available via the Mendeley Data Repository, at <https://doi.org/10.17632/vv5dhvnh3d.2>.

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