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LCIA OF IMPACTS ON HUMAN HEALTH AND ECOSYSTEMS

Impacts of onshore wind energy production on birds and bats: recommendations for future life cycle impact assessment developments

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Abstract

Purpose Models for quantifying impacts on biodiversity from renewable energy technologies are lacking within life cycle impact assessment (LCIA). We aim to provide an overview of the effects of wind energy on birds and bats, with a focus on quantitative methods. Furthermore, we investigate and provide the necessary background for how these can be integrated into new developments of LCIA models in future.

Methods We reviewed available literature summarizing the effects of wind energy developments on birds and bats. We provide an overview of available quantitative assessment methods that have been employed outside of the LCIA framework to model the different impacts of wind energy developments on wildlife. Combining the acquired knowledge on impact pathways and associated quantitative methods, we propose possibilities for future approaches for a wind energy impact assessment methodology for LCIA.

Results and discussion Wind energy production has impacts on terrestrial biodiversity through three main pathways: collision, disturbance, and habitat alterations. Birds and bats are consistently considered the most affected taxonomic groups, with different responses to the before-mentioned impact pathways. Outside of the LCIA framework, current quantitative impact assessment prediction models include collision risk models, species distribution models, individual-based models, and population modeling approaches. Developed indices allow scaling of species-specific vulnerability to mortality, disturbance, and/or habitat alterations.

Conclusions Although insight into the causes behind collision risk, disturbance, and habitat alterations for bats and birds is still limited, the current knowledge base enables the development of a robust assessment tool. Modeling the impacts of habitat alterations, disturbance, and collisions within an LCIA framework is most appropriate using species distribution models as those enable the estimation of species' occurrences across a region. Although local-scale developments may be more readily feasible, further up-scaling to global coverage is recommended to allow comparison across regions and technologies, and to assess cumulative impacts.

Keywords Collision · Disturbance · Habitat alteration · LCIA · Quantitative models · Wind turbine

1 Introduction

In an attempt to halt climate change, wind energy has emerged as a promising alternative to fossil fuels, with an annual average growth rate of 24.3% from 1990 to 2014 (IEA 2016). In 2013, it represented 2.5% of the global electricity supply, and it is expected to grow to between 15 and 18% by 2050 (International Energy Agency 2013). However, research has shown that both onshore and offshore wind farms can harm wildlife directly and indirectly (e.g., Edenhofer et al. 2012; Rydell et al. 2012; Schuster et al. 2015). For onshore wind energy, research highlights that bats and birds are particularly vulnerable to collision, disturbance, and habitat alterations during the construction and operational stages. Even if these detrimental effects may be relatively low compared to other energy sources (Sovacool 2013), the cumulative impacts of projected wind farms may affect significantly more vulnerable populations (Carrete et al. 2009; Masden et al. 2010a;

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Schaub 2012). Wind power impacts might serve as an additional impact to existing environmental threats, thereby critically contributing to increased impacts upon specific species and populations. Different wind energy impact assessment approaches exist; however, these are all site, species, or impact specific and a globally applicable tool is still lacking.

Life cycle assessment (LCA) is an environmental impact assessment tool, which is widely used to evaluate and compare the environmental performance of products or services through their whole life cycle using different impact categories, such as climate change, ecotoxicity, or land use (Hauschild and Huijbregts 2015). LCA typically focuses on greenhouse gas emissions (Evans et al. 2009) but has been used to evaluate and compare environmental impacts associated with different energy production systems. Martínez et al. (2009) performed a LCA of a multi-megawatt wind turbine, analyzing the manufacturing, use, disposal, and transport stages throughout several impact categories (e.g., global warming carcinogens, acidification). Manufacturing of the components was found to be the largest contributor to wind turbine impacts, and supported by Arvesen and Hertwich (2012). However, none of these studies account for impacts on biodiversity due to insufficient or lacking impact assessment models. Including biodiversity will likely increase the contribution of the construction and operational stages of a wind farm to its overall impacts, although the magnitude of it is unknown. Even with recent developments in incorporating biodiversity-related impacts in LCA (e.g., Azevedo et al. 2013; Chaudhary et al. 2015; Verones et al. 2017b; Cosme et al. 2017), current life cycle impact assessment (LCIA) models do not incorporate wind energy impacts on biodiversity.

To address the lack of biodiversity impacts from renewable energy production in LCIA, this review aims to assess and summarize the existing knowledge base and its applicability for the future development of LCIA models covering the impacts of wind energy on biodiversity. Future LCIA models should consider the varying degrees of vulnerability for different species groups to each impact type. Focusing on onshore wind energy, we provide an overview of the main impact pathways affecting two major and particularly vulnerable taxonomic groups, bats and birds. We highlight the most relevant state mechanisms and conditional variables that should be considered in the development of an impact assessment model. Although other authors have qualitatively reviewed this topic before, a summary of quantitative methods and a link to LCIA are still lacking. Therefore, we present the most commonly used environmental impact assessment tools in the wind energy sector, as well as recent developments in these. Finally, we explore how these can be used as a basis to develop future LCIA models and provide recommendations for the next steps in the direction of these model developments.

2 Methods

Several authors have comprehensively reviewed the effects of wind energy on biodiversity from an ecological point of view (Drewitt and Langston 2006; Kunz et al. 2007b; Rydell et al. 2012; Langston 2013; Marques et al. 2014; Dai et al. 2015; Wang et al. 2015; Schuster et al. 2015). These served as a gateway to a more refined search within the subsections covered in each article (e.g., articles focusing on one species or group of species, or on a particular impact pathway). Despite the availability of several reviews, there was only one article focusing on quantitative models, and this concerned avian collision risk models (Masden and Cook 2016).

We searched for available peer-reviewed and "gray" literature on the topic of impacts of wind energy on wildlife published up until the date of final submission. Using mainly Google Scholar and Oria, we began by using key terms including, but not limited to, "wind energy," "wind power," "biodiversity," "LCA," "impacts," "assessment," "birds," "bats," "collision," "displacement," "disturbance," "avoidance," "habitat loss," and "habitat alterations." For an overview of available quantitative models, we mainly used Google Scholar to conduct our search, using key terms such as "collision risk," "model," "quantifying," "quantitative," "habitat loss," "avian," "displacement," "bat," "species distribution," and "wind energy." When searching for LCA-related methodologies, we also included the key terms "LCA," "LCIA," "Life Cycle Assessment," and "Life Cycle Impact Assessment" in addition to the previous terms. We went through each article's reference list in search of other potentially relevant studies. The most highly cited literature was taken as a basis for understanding the topic. Mendeley and Elsevier also proved to be valuable sources of knowledge by linking previous searches to related articles and providing recommendations on relevant articles. "Gray" literature was also considered in this review and consisted mainly of technical reports from highly credited institutions or companies. These were included because of the reports' high number of citations or applicability to this review. Some articles were excluded from this review, as they were already well described in other reviews and would not contribute any additional content to this article. We also excluded articles describing non-predictive quantitative methods, i.e., those that would not contribute to the development of LCIA models. In total, we reviewed 136 articles.

3 Effects of wind energy development on biodiversity

The first step to adequately quantify impacts, outside and within the LCA framework, is to understand the effects of wind energy on biodiversity at a species level and how these may reflect impacts at a population level (May et al. 2017). Collision, disturbance, as well as habitat loss and change have emerged as the main effects from both on- and offshore wind power on birds (Drewitt and Langston 2006). For bats, Brinkmann (2006) stated that collision is likely the main cause of impacts. Schuster et al. (2015) consolidated literature on effects from wind power on birds and bats, with a focus on both taxa. We note that disturbance and displacement are two similar terms that may be used interchangeably in wind energy impact assessment literature and should therefore be clarified. As defined by Furness et al. (2013), disturbance relates to the added expenditure of resources by animals to avoid a wind farm and associated activity. Displacement refers to the reduced number of animals occurring in the wind farm area and its immediate vicinity. We adhere to this terminology throughout this article.

3.1 Collision

Collision risk, or the probability of mortality due to collision of individuals intersecting with a wind turbine, occurs during the operational life cycle stage of a wind farm. Species that do not generally exercise avoidance behavior toward human-made structures, specifically wind turbines, are at risk of colliding with turbine blades or the monopoles (Kunz et al. 2007a). Cook et al. (2014), and later May (2015), described three main types of bird avoidance behavior, according to the scale of its occurrence. Two of these, meso- and micro-avoidance, take place inside the wind farm space and therefore directly affect collision risk. Meso-avoidance is described by May (2015) as the process by which birds evade the wind turbines by anticipating or reacting to their presence. However, the longer it takes the bird to do this (i.e., the closer it gets to the wind turbine before it responds to the obstacle), the more likely it is to collide. At this point, birds may still narrowly escape the turbine structure, which the author classifies as a micro-scale avoidance. The bird may also avoid the wind farm altogether (macro avoidance), in which case it will either lead to no response (if the avoidance does not alter the birds' habitat use) or displacement through disturbance.

Various factors affect the collision risk for birds and bats and have been observed to be site, species, and turbine specific (Drewitt and Langston 2006; Marques et al. 2014; Hein and Schirmacher 2016). Some studies show that wind turbine collisions only account for a considerably small percentage of total bird mortality (Erickson et al. 2005; Calvert et al. 2013; Sovacool 2013). This may appear as an argument to reduce efforts to mitigate impacts of wind energy development on wildlife. However, the authors agree that fatalities from wind energy come in addition to other sources of mortality. In other words, not only the main source of a species' mortality should be assessed (while ignoring other causes), as even smaller additions to a population's mortality rate can have severe consequences, especially to species with slow life-history traits (i.e., long lifespans, few offspring, and late maturity), such as raptors or bats.

3.2 Disturbance

Displacement can be considered as reduced flight activity within the wind farm area due to a functional loss in habitat (May 2015). This is true for not only resident species but also migratory species through loss of stopover sites. It may also lead to increased energy expenditure when individuals need to alter their flight path to avoid wind farms (also known as "barrier effect"), which may potentially have consequences on population health if numerous wind farms need to be avoided (Masden et al. 2009, 2010b). The extent and severity of disturbance and consequent displacement is dependent on site and species characteristics (Drewitt and Langston 2006), and some authors consider displacement to be potentially more threatening than collision for birds (Kuvlesky et al. 2007). Pearce-Higgins et al. (2012) show how the construction stage of wind farms may have a greater displacement impact on bird populations than the operational stage. Nevertheless, indirect impacts of wind energy production remain greatly understudied, making their quantification very challenging (May 2015). Bird displacement from wind farms has been shown to translate into functional habitat loss (Pedersen and Poulsen 1991; Larsen and Madsen 2000; Pearce-Higgins et al. 2008, 2009; Garvin et al. 2011; Petersen et al. 2011; May et al. 2013). However, some species may return to their original habitat with time, habituating to wind farm presence (Madsen and Boertmann 2008). Masden et al. (2009) evaluated this deviation and concluded that although avoidance of a single wind farm may be negligible in terms of energy cost, there may be a harmful cumulative effect over the avoidance of several wind farms.

Bats, on the other hand, appear to either be undisturbed by wind turbines and, in some cases, even attracted to them, which thereby can increase the number of collisions (Rydell et al. 2012). Kunz et al. (2007b) present several hypotheses that may explain bat attraction to turbines. Most are related to a potential attraction to insects drawn to the wind turbines or associated altered landscape, which is also supported by other authors (Brinkmann 2006; Rydell et al. 2010a). Another hypothesis presented by Kunz et al. (2007b) is that tree-roosting bats are attracted to the turbines that they perceive as potential roosts. This is further described in the work of Cryan et al. (2014), as well as other observed bat behaviors around wind turbines in an experimental setting. Nevertheless, Rydell et al. (2012) noted that indirect effects of wind energy on bats are relatively small and possibly most relevant for birds.

3.3 Habitat alterations

Construction of wind turbines, like any infrastructure development, alters habitats at and surrounding the construction sites. However, the extent of this effect may vary depending on the original setting. For instance, habitat alteration effects may be more pertinent in, e.g., forested and/or pristine wilderness areas as opposed to multiple-use landscapes with preexisting anthropogenic influences. Specialist species, i.e., species with a narrow range of suitable habitats (high habitat specificity), are more vulnerable (Swihart et al. 2003; Munday 2004; de Baan et al. 2013) and therefore potentially suffer a higher impact than more wide-ranging and generalist species.

Apart from the direct loss of habitat for certain species immediately surrounding the turbines, the tall turbine structures may be mistaken for natural structures such as trees, which, as described in the previous section, may attract certain species and lead to increased collision risk (i.e., an ecological trap; May 2015). In addition, roads and power lines associated with the wind farm may cause habitat fragmentation, which can be particularly damaging in previously unaltered areas (Rydell et al. 2012). Although these alterations can reduce habitat suitability for some species, the altered environment may be more favorable for other species (Hötker et al. 2006). In turn, increased densities of benefiting species may attract predators, such as bats or birds of prey, which may end up suffering higher collision rates while hunting. Smallwood et al. (2007), for instance, showed how increased densities of ground squirrels near the base of wind turbines attracted burrowing owls closer to the blades, consequently increasing collision risk.

3.4 Conditions influencing effects of wind farms on wildlife

3.4.1 Species-specific conditions

Bat behavior toward wind farms and turbines can be explained using a guild concept. Denzinger and Schnitzler (2013) group different bat species based on their use of echolocation, foraging habitats, and foraging modes, as well as sensory and motor adaptations. They identify three main guild types, namely open space, edge space, and narrow space, which forage at different distances from background structures (such as wind turbines) and may be more or less apt to avoid them. The authors conclude that the foraging and echolocation behaviors of species within a given guild are so similar that a small number of species or observations can be used as proxy for the whole guild with high certainty.

Birds' sensory capabilities, as well as behavior, may play a significant role in their response to a wind farm or turbine (e.g., Marques et al. 2014; May et al. 2015). Moreover, bird morphology appears to be a determinant parameter for collision risk (e.g., Bevanger 1994; Janss 2000; Herrera-Alsina et al. 2013). Rayner (1988) grouped flying birds according to their size, aspect ratio, and wing loading, relating these to different flight behaviors. The mechanisms behind bird (and bat) flight, and how this in turn reflects in their flight behavior, are further described by Norberg (2006).

3.4.2 Environmental conditions

Topographical features influence bat and bird activity. Migrating bats use linear aspects of the landscape for navigation and movement, such as river valleys, tree rows, or forest edges (e.g., Ahlén et al. 2009; Furmankiewicz and Kucharska 2009), which could increase collision rates with wind turbines placed in the proximity of such features (Rydell et al. 2010b). Similarly, Johnson et al. (2004) determined a negative correlation between bat activity and distance to woodlands. This is particularly important for the conservation of tree roosting bats, which may mistake wind turbines as potential roosting or mating sites (Cryan et al. 2008), as these activities typically take place in tall trees (Cryan et al. 2014). Certain birds, such as raptors, are also known to utilize landscape features enhancing thermal or orographic lift, such as ridgelines or slopes, in order to save energy, making their passages predictable to a certain extent (Duerr et al. 2012). An analysis by Hötker et al. (2006) on collision risk factors showed that habitat type has a significant influence on bird casualty rates, particularly mountain ridges and wetlands.

Bird and bat behavior varies seasonally, particularly in terms of habitat use and flight activity, and consequentially collision risk. The highest bat fatality rates due to collision are observed during late summer and autumn, during which bat activity is typically at its peak (due to, among other factors, migration periods) (e.g., Brinkmann 2006; Rydell et al. 2010a, b; Baerwald and Barclay 2011). May et al. (2010) and May et al.(2011) determined that the white-tailed eagle (*Haliaeetus albicilla*) had considerably higher flight activity in the spring, as well as more fatal collisions with wind turbines. Barrios and Rodríguez (2004) also noted a seasonal variation in the flight frequency of vultures in wind farms, with higher counts, but also variance, during the winter–autumn period. These findings are supported by Smallwood et al. (2009), who evaluated different bird species flying in wind farms at the Altamont Pass Wind Resource Area, USA. Relatively large seasonal variations in bird numbers are associated with migratory behavior, although some of these also coincide with post-breeding periods, when the number of young and inexperienced birds increases (Drewitt and Langston 2008).

Meteorological conditions, particularly wind speed and direction as well as temperature, are essential in determining the probability of negative effects (e.g., by creating orographic and thermal updrafts) influencing the flight behavior and activity of different species (Richardson 1998; Langston 2013; May et al. 2015). In particular, wind, fog, and rain have a direct impact on birds' maneuverability, flight height, and sensory perception (Langston and Pullan 2003; Arnett et al. 2007). Furthermore, temperature (Arnett et al. 2006) and low wind speeds are positively correlated with bat activity near wind turbines and therefore a useful parameter in determining the areas of highest collision risk (e.g., Rydell et al. 2010a, b; Baerwald and Barclay 2011; Cryan et al. 2014). Brinkmann et al. (2006) report that operating wind turbines only at wind speeds above 5.5 m/s can be an effective measure to reduce bat collision rates with wind turbines. This was also tested and confirmed by Baerwald et al. (2009), at the same start-up speed, with only marginal costs from the decreased electricity production. Similarly, Barrios and Rodríguez (2004) show that wind speed affects bird collision risk of raptors, with the highest being at wind speeds between 4.6 and 8.5 m/s, which is consistent with the observations of Smallwood et al. (2009). However, some species are able to fly at speeds considerably higher than these observed limits (Winter 1999), which needs to be taken into consideration when planning such mitigation strategies.

3.4.3 Technological conditions

Finally, type, size, and number of wind turbines, as well as layout of wind farms are considered by some authors to be relevant aspects in determining avian and bat collision risk. Smallwood and Thelander (2004) identified tower size, blade tip speed, and wind farm layout to be the most important factors contributing to golden eagle (*Aquila chrysaetos*) mortality at the Altamont Pass Wind Resource Area. Barclay et al. (2007), on the other hand, reported that turbine height had a significant effect on bats, but not birds, while rotor blade length had no effect on bird or bat fatality rates. de Lucas et al. (2008) found taller turbines to be linked to a higher number of fatalities, although they could not conclude on the effect of the wind farm layout. Hötker et al. (2006) drew opposing conclusions, determining a statistically insignificant effect of turbine hub height on collision rates. Nevertheless, Hötker et al. (2006) recommended that wind farms be arranged with turbine arrays parallel to the main flight direction to decrease the risk of collision. Rotor speed has also been identified as a determinant collision risk factor by model developers (e.g., Tucker 1996a), such that more rotations per minute imply a higher chance of a bird or bat colliding if it traverses the rotor swept area. This makes turbine designs of inherent slower blade rotation (e.g., vertical axis wind turbine) potentially less deadly to birds and bats (Islam et al. 2013; Santangeli and Katzner 2015). Furthermore, designs that can cause a lower degree of motion smear of the blades may potentially be more detectable by avian species (Hodos 2003).

4 Impact assessment modeling approaches

Integrating wind energy impacts on biodiversity in LCIA depends not only on knowledge on the impacts but also on how these can be assessed using currently available models. Therefore, and given the current lack of a literature review on the matter, we compiled different predictive modeling approaches used in assessing collision, disturbance, and habitat alterations on bird and bat species. We grouped these models by type of method used, noting that each type may cover more than one impact. Table 1 summarizes our findings and provides an overview on the inputs required for each model type to cover the relevant conditions as described in the previous section. All model types are further detailed in the following paragraphs. At the end of this section, Table 2 summarizes a critical comparison between the different model types, showing the different advantages and disadvantages of each model type for inclusion in LCIA.

4.1 Collision risk models (CRMs)

Masden and Cook (2016) recently reviewed available avian collision risk models. Tucker (1996b) presented the first of these models, calculating collision risk as a ratio between the time spent flying by a bird through the rotor swept area over the time taken by one single rotation of the rotor blades. Similarly, Band et al. (2007) developed a model for onshore wind turbines which associates the risk of collision with the probability of the bird occupying the same space as the turbine blade during its flight through the rotor swept area. This model was then expanded to take into account the variable distribution of birds with height within the rotor swept area (Masden and Cook 2016). Other models have been developed (e.g., Podolsky 2008; Holmstrom et al. 2011; Eichhorn et al. 2012), but in general these take a similar approach to Tucker (1996b) and Band et al. (2007). Bird size, flight characteristics, as well as rotor blade length and speed are typical inputs in these types of models and are combined with the expected number of birds flying within rotor swept height. In another approach, Korner-Nievergelt et al. (2013) used a combination of carcass searches and animal density indices in a mixed model to determine collision rates, yielding results "at least as precise as conventional estimates" from carcass search data. New et al. (2015) developed a predictive CRM based on the assumption of a relationship between pre-construction avian exposure and subsequent fatalities. Among other differences, this model distinguishes itself for the direct inclusion of uncertainty, as well as considering the entire turbine height when calculating the total hazardous volume of a wind turbine. This means that birds in this model are considered to be able to collide when flying under the rotor area, as opposed to most CRMs which only consider rotor blade length. Chamberlain et al. (2006) assessed the effects

Table 1 Summary of reviewed quantitative models, by type, for the three main impact pathways of wind energy on biodiversity (the rightmost column summarizes relevant parameters used in the development of each model type)

	Collision	Disturbance	Habitat alterations	Relevant parameters
Birds				
Collision risk models	Tucker (1996); Bolker et al. (2006); Desholm (2006); Podolsky (2008); Holmstrom et al. (2011); Band (2012); Eichhorn et al. (2012); Calvert et al. (2013); Korner-Nievergelt et al. (2013); Bolker et al. (2014); New et al. (2015)	–	–	Bird avoidance behavior Bird flight mechanics Bird morphology Flight activity Flight height Migration patterns Turbine characteristics Wind conditions Wind farm layout
Index-based models	Garthe and Hüppop (2004); Furness et al. (2013); Diffendorfer et al. (2015); Beston et al. (2016); Busch and Garthe (2016); Warwick-Evans et al. (2017)	Garthe and Hüppop (2004); Furness et al. (2013); Diffendorfer et al. (2015); Beston et al. (2016); Busch and Garthe (2016); Warwick-Evans et al. (2017)	Garthe and Hüppop (2004); Furness et al. (2013); Diffendorfer et al. (2015); Beston et al. (2016); Warwick-Evans et al. (2017)	Index scores on: Bird maneuverability Disturbance sensitivity Fatalities per year Flight activity Flight height Habitat specialization Life-history traits Population dynamics Threat level and conservation status
Species distribution models	Bright et al. (2008); Pearce-Higgins et al. (2008); Masden (2010); Masden et al. (2012); Liechti et al. (2013); Pocewicz et al. (2013); Mcnew et al. (2014); Miller et al. (2014); Reid et al. (2015); Bastos et al. (2016); BirdLife International (2017)	Bright et al. (2008); Pearce-Higgins et al. (2008); Liechti et al. (2013); May et al. (2013); Pocewicz et al. (2013); Mcnew et al. (2014); Miller et al. (2014); BirdLife International (2017)	Bright et al. (2008); Pearce-Higgins et al. (2008); Mcnew et al. (2014); Miller et al. (2014); Reid et al. (2015); BirdLife International (2017)	Body mass Distribution data Flight behavior Flight distance Flight energy expenditure Flight height Land cover Season Topography Wind farm locations
Individual-based models	Masden (2010); Eichhorn et al. (2012); Schaub (2012)	Masden (2010)	Masden (2010)	Collision and avoidance probabilities Flight activity Flight distance Flight height Flight through rotor probability Habitat quality Rotor blade length Wind turbine height
Population models	Hötker et al. (2006); Carrete et al. (2009); García-Ripollés and López-López	–	–	Age at first reproduction Carrying capacity

Table 1 (continued)

	Collision	Disturbance	Habitat alterations	Relevant parameters
	(2011); Dahl (2014); Rushworth and Krüger (2014); Erickson et al. (2015); Sanz-Aguilar et al. (2015); Grünkorn et al. (2016)			Fecundity Mortality rates Number of turbines Population size
Bats				
Species distribution models	Roscioni et al. (2013); Santos et al. (2013); Hayes et al. (2015)	–	Roscioni et al. (2013)	Presence data Land cover Precipitation Season Temperature Topography Wind farm locations
Individual-based models	Ferreira et al. (2015)	–	–	Altitude Bat behavior Collision data Land cover Number of turbines Rotor blade length Temperature Wind speed
Population models	Rydell et al. (2012); Erickson et al. (2015)	–	–	Fecundity Number of turbines Survival rates
Index-based models	Diffendorfer et al. (2015)	–	Diffendorfer et al. (2015)	Fatalities per year Habitat specialization Life-history traits Population dynamics Threat level and conservation status

Table 2 Critical comparison between different types of models, summarizing advantages and disadvantages of each

	Advantages	Disadvantages
Collision risk models	Direct estimation of collision fatalities Can take into consideration wind turbine characteristics	Requires knowledge on fatalities, which is not often available Lack of research on species–turbine interaction leads to inaccuracy in these models
Index-based models	Smaller data requirements Easier quantification than more complex models Score-based results are easier to communicate to stakeholders	Provides a ranking of impacts rather than an estimation or direct measurement May be hard to implement in LCIA due to not relating to a loss in species richness
Species distribution models	Allow the prediction of species occurrence Makes habitat loss and disturbance easy to quantify Can be combined with other models to determine collision effects and impacts Spatially explicit	Needs to be combined with other models in order to quantify impacts Hard to implement at a global scale High variability within the study species group increases the complexity of the model Time consuming when many species are being studied
Individual-based models	Able to simulate the movement of individuals of a species Could potentially give the most accurate estimations of impacts	Far too complex for a high number of species across large study areas Data intensive Requires good knowledge on each species' behavior around wind turbines
Population models	Transition from effect to impact Allows an easier integration into LCIA when species loss is quantified	Species demographic data at a large scale may not be obtainable Requires fatality data and/or knowledge of the effect of habitat loss on the populations

of estimating and using avoidance rates in the development of a collision risk model, based on the original Band model (Band et al. 2007). Fatality rates derived from estimated avoidance rates may be used for comparative purposes, but the authors underline the urgent need for more specific and empirical avoidance rate studies. Lastly, Calvert et al. (2013) estimated avian mortality due to different sources in Canada. The authors developed a stochastic simulation model and compared the impacts of mortality at different life stages of different species, as well as across different mortality sources, also at a population level.

4.2 Species distribution models (SDMs)

Species distribution models are used to estimate the probability of occurrence of a species in a given location and, together with posterior effect modeling, the likelihood of a negative effect. One interesting application of SDMs is seen in a recent study by Santos et al. (2013), who applied a maximum entropy model (MaxEnt; Phillips et al. 2006), using presence-only data to determine the collision risk associated with wind farms of four different bat species in Portugal. Given a small number of occurrences and a given set of environmental conditions, MaxEnt can be used to identify regions where a species is likely to be present (Pearson et al. 2007) and therefore delineate areas of higher conflict probability. Roscioni et al. (2014) also applied the MaxEnt approach, but rather to determine the impacts of wind energy developments on habitat connectivity for bats. Rebelo and Jones (2010) compared this approach with the ecological niche factor analysis (ENFA) (Hirzel et al. 2002), a similar model which also uses presence-only data, for modeling the potential distribution of a bat species in Portugal. The authors conclude that the differences between the two models make ENFA more appropriate for determining a species' potential distribution, while MaxEnt is better suited for determining a species' realized distribution. Hayes et al. (2015) created seasonally dynamic SDMs to study the impacts on migratory hoary bats (*Lasiurus cinereus*). Apart from MaxEnt, the authors used four other SDM approaches to model the species' distribution. Bastos et al. (2016) assessed the local impacts of wind energy on skylark (*Alauda arvensis*) populations in Portugal via an index derived from a SDM, showing how this combined framework can be used for predictive impact assessments. Elith et al. (2006) summarizes and compares other modeling methods used in predicting species' distributions from occurrence data.

Bright et al. (2008) present a bird sensitivity map of 16 protected species in Scotland, in which species distribution data were buffered and rated taking into account foraging ranges, collision risk, and susceptibility to disturbance. The SDM was then overlapped with a map of existing or planned wind farm locations in order to provide a proportion of affected bird species by these developments. Similarly, Reid et al. (2015) modeled the movements of bearded vultures (*Gypaetus barbatus*) in southern Africa in terms of habitat use. Other behavior-inclusive SDMs focus on migratory species. Pocewicz et al. (2013) mapped important migratory areas for birds in Wyoming, USA, including stopover habitats. The authors combined different geographical features (such as ridges, streams, and likely thermal updraft locations), which directly correlate to increased activity of migratory bird species. Similarly, Liechti et al. (2013) developed a model enabling the determination of areas with predictably high concentration of migratory bird species in Switzerland, which translate into a higher collision risk. Also, with a focus on soaring birds, BirdLife International (2017) developed a sensitivity mapping tool for migratory soaring birds in the Middle East. If migratory paths are known or predictable, siting new wind farms outside thereof could potentially decrease collisions and displacement effects on those species. These and other applications of species distribution models are further analyzed by Guisan and Thuiller (2005). May et al. (2013) evaluated habitat utilization and displacement of white-tailed eagles using Resource Utilization Functions (RUFs), which correlate a species' space use to its resource utilization. Other studies have also used RUFs to assess potential negative effects on birds from wind energy developments (Mcnew et al. 2014; Miller et al. 2014).

Two models have been developed to quantify the spatial implications of "barrier effects". Masden et al. (2012) details models used to determine birds' movement in response to wind farms based on bird movement data collected after wind farm construction. Masden et al. (2010b) modeled the energy cost of avoidance by several seabirds due to offshore wind farm placement, using the modeling software developed by Pennycuick (2008). The study concluded that the additional energy costs of avoiding the wind farm may be insignificant for some species, but a species-specific approach should be taken when assessing the impacts of wind farms on seabirds.

4.3 Individual-based models (IBMs)

Several individual-based models (IBMs) have been developed to assess potential impacts on avian species. IBMs allow researchers to simulate interactions of individuals with the surrounding environment, as well as their adaptations to environmental changes. Grimm et al. (2006) further describe the concepts behind this tool, potential applications, and provide a protocol for further developments, named ODD ("Overview," "Design concepts," and "Details"). Eichhorn et al. (2012) followed this protocol in their collision risk model of red kites (*Milvus milvus*). Three entities were used in this model: a landscape grid (based on habitat characteristics of West Saxony, Germany), a red kite, and a wind turbine. The bird entity is essentially based on its behavior and flight characteristics, as well as probability of collision (based on the Band model) and avoidance. For the wind turbine, position, hub height, and rotor blade length were used as inputs. Schaub (2012) also based his model on the red kite species, although not following the same protocol, but nevertheless modeling the effect of a varying number and layout of wind turbines on the population dynamics of the species. Ferreira et al. (2015) also followed the protocol proposed by Grimm et al. (2006) for estimating bat mortality risk at wind farms. As with the model produced by Eichhorn et al. (2012), three entities were selected, referring to landscape, the bat, and the wind turbines. Land cover and altitude of the landscape were included in the first entity, taking into consideration the use thereof by bats for foraging and/or roosting. Wind speed, temperature, and species behavior determined the inputs of the bats' entity. As for the turbines, the authors included blade length as a variable, but not height. Masden (2010) developed an IBM following the ODD protocol to evaluate the effect of technological changes in collision mortality and habitat-related productivity in hen harriers (*Circus cyaneus*). From her results, the author concludes that the impacts of wind turbines on hen harriers depended not only on the number of turbines but also their location, suggesting the need for knowledge on a species' ecology in wind energy development planning. A recent work by Warwick-Evans et al. (2017) shows the use of the ODD protocol to study the effect of wind turbines on body mass, mortality rate, and breeding success of Northern gannets (*Morus bassanus*). The authors state that this is the most complex and comprehensive model of its kind yet and has the potential to be adapted for other seabird populations and types of impacts from altered spatial environments.

4.4 Population models

Widely used in ecology, population viability analyses (PVA) estimates the probability of a population or species becoming extinct in a given period of time and is based on a number of case-dependent variables together with demographic parameters (Beissinger and McCullough 2002). Multiple studies have used the program VORTEX (Lacy and Pollak 2014), an IBM used for PVA, to simulate the effects of avian mortality from wind farms on population dynamics of different species (Hötker et al. 2006; Carrete et al. 2009; García-Ripollés and López-López 2011; Rushworth and Krüger 2014). This type of modeling is mainly based on demographic parameters (e.g., mortality rates, population size, age at first reproduction), although some environmental variables such as carrying capacity can be incorporated. Sanz-Aguilar et al. (2015) designed a PVA without using VORTEX, using instead linear regression and R-based scripts to determine stochastic population growth. Nevertheless, their model is based on demographic parameters. Erickson et al. (2015), using branching process models, delivered a predictive model for the probability of extinction of four representative species: two bats and two birds. Although branching process models are in essence individual-based models, this output is characteristic of PVAs and is based on population dynamics. Rydell et al. (2012) presented a simple, deterministic population model based on population size, survival rates, fecundity, and number of turbines. Mortality from wind turbines is a simple subtractive factor in the equation, dependent only on the annual mortality at each turbine and the number of turbines. Bellebaum et al. (2013) estimated mortality thresholds for red kites in Germany using a potential biological removal (PBR) model. They affirm that PBR models are needed to enable more precise estimations of thresholds for the added mortality from wind energy developments. Dahl (2014) used a different approach and presented an age-structured matrix-based population model for the white-tailed eagle in Smøla, Norway. This model focused on the demographic parameters of the studied population, including not only survival rates but also reproductive success. In a report by Grünkorn et al. (2016), matrix and elasticity models were used to identify consequences of bird mortality at a population level, for three raptor species, taking into account age-specific mortality and reproduction rates. Lastly, Cook and Robinson (2017) present a framework for assessing wind energy impacts at a population level using Leslie matrix models. These models consider a generic seabird species with characteristics derived from the literature. Of note is the evaluation of decision criteria previously summarized by Green et al. (2016). The authors highlight the need for transparency when it comes to the use of demographic values of populations. However, it would be very difficult, if not impossible at the moment, to obtain demographic data for a large number of species at scales relevant to LCIA.

4.5 Index-based models

Data scarcity can be a constraint when modeling ecological processes, especially at higher scales when many different species are involved. To overcome this obstacle, index-based models can potentially be used as proxies, delivering score-based outputs on effects rather than, for instance, a number of individuals affected. Data requirements are lower and often based on what is known of a species in terms of, e.g., behavior, morphology, and habitat use. Garthe and Hüppop (2004) developed a vulnerability index for species affected by offshore wind power farms, with a focus on German seas, based on different seabird characteristics as well as their conservation status. More recently, Furness et al. (2013) constructed similar indexes for collision and displacement impacts on Scottish marine birds. Although somewhat simplistic in its nature, these types of sensitivity indexes can be used to identify important impact sources, as well as map areas of higher risk, even when experimental data is not widely available. Using the indexes from these publications, Busch and Garthe (2016) developed a novel method for assessing displacement combining a matrix of potential displacement and mortality levels of seabirds from offshore wind farms with a PBR model (Wade 1998). One of the methodologies that perhaps encompasses most impacts of wind energy on bats and birds was designed by Diffendorfer et al. (2015). The methodology prioritizes species based on previously gathered data, combining each species' conservation status, as well as its relative risks from collision fatalities and habitat modification. The consequent impacts at a population level are then evaluated with the methodology's demographic and PBR models. The authors followed up on this work, this time focusing on prioritizing bird taxonomic orders according to their impact risk indexes (Beston et al. 2016).

5 On modeling biodiversity impacts from wind energy production in LCIA

The integration of wind energy impacts on biodiversity in LCIA should include all three aforementioned impact pathways: collision, disturbance, and habitat alterations. Figure 1 illustrates how the impact pathways can conceptually be integrated into a logical assessment flow (conditions–state–effect–impact) and the potential contribution of the different prediction models to quantify these. We propose that separate characterization factors should be developed for the three impact pathways and both birds and bats. All bat and bird species should be grouped into guilds or functional groups depending on their morphology and behavior in order to cover as many species as possible without requiring all relevant information for every individual species (which may not be available). However, a final impact score should include all the impacts on all species groups together, expressed in common LCIA units such as potentially disappeared fraction of species (PDF) as recommended by the UNEP-SETAC Life Cycle Initiative (Verones et al. 2017a). Verones et al. (2015) propose four different options to aggregate land and water use impact scores into a single score: equal weight for species, equal weight for taxa, and two options with special consideration of species' vulnerability. Similar approaches could be used to combine impact scores for bats and birds, over the main impact pathways, into one score compatible with current LCIA methodologies. These options are particularly relevant when deciding if and which bird and bat groups should be attributed higher impact score due to higher vulnerability.

The three impact pathways generally affect a species' probability of occurrence at a specific site. Whereas habitat alterations may lead to the absence of a species at a site, displacement and collision reduce the number of individuals and thereby indirectly the probability of occurrence. Spatial estimation of species probability of occurrence can be done using SDMs. Harte et al. (2009) presents an approach on species–area relationships that estimates the number of species in

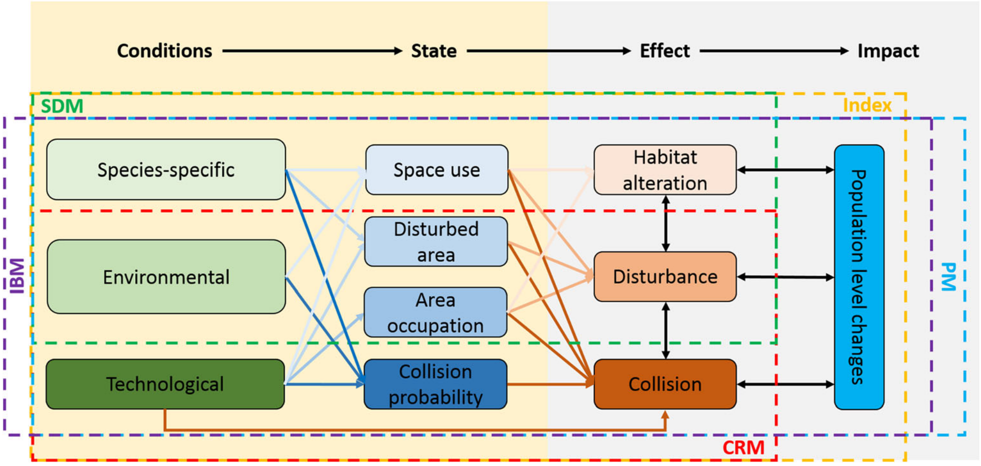

Fig. 1 Integrating wind energy impacts in LCIA. The gray background represents the processes that are modeled at a LCIA level. The yellow background represents data necessary for those processes. Some of the "State" processes are found at a Life Cycle Inventory level. PM, population models; Index, index-based models; CRM, collision risk models; SDM, species distribution models; IBM, individual-based models. Conditions relate to the inputs used for the model: species specific (e.g., physiological, cognitive, sensory, behavioral), environmental (e.g., topography, vegetation, season, wind speed, wind direction, temperature), and technological (e.g., turbine size, configuration)

a certain area through correlation of species richness and probability of occurrence. With such estimates, and knowing at which sites wind turbines are located, GIS tools can be used to quantify effects from wind energy developments in a spatially explicit manner. Estimating an altered probability of occurrence due to the expected effect, e.g., using respectively flight initiation distances (Blumstein 2006) and collision risk models (e.g., Tucker 1996a; Band 2007), the expected loss of occurrence at a site can be determined. MaxEnt, for instance, is a SDM that derives a score in each map cell proportional to the probability of occurrence of a species. Summing scores across species renders insight into the species richness at a site, allowing the calculation of regional and potentially global PDFs. An impact score can then be derived by applying species–area relationship models (SARs), which are already used in LCIA. Unlike classical SARs, which consider all biodiversity to be lost when habitat is changed, countryside SARs (Pereira and Daily 2009) factor in habitat suitability for a given species. This habitat suitability factor is analogous to the proposed use of MaxEnt scores. In addition, estimating a species distribution rather than directly using binary presence–absence range map is an improvement in terms of ecological significance.

Only in cases where population size and species distribution are known (either empirically or through estimation) can the number of affected individuals in each cell be determined. With such data, other approaches such as PVAs and IBMs also become feasible for developing (regional) LCIA models. Furthermore, if a relation between the area (or number of individuals) lost and probability of extinction is known, one can potentially quantify results directly in terms of PDF and therefore easily integrate the results in LCIA. However, to our knowledge, such relations are not known, and population data is scarce for a large number of species. As a generic approach for inclusion within the LCIA framework, such models are therefore deemed less appropriate. Although IBMs would yield the most detail, they are in general too complex and data intensive to be able to cover a large number of species and spatial distribution. Nevertheless, future research can be done to further develop or adapt CRMs or index-based models in order to obtain a descriptive result of a fraction of species lost, or another justifiable unit in LCIA.

It is important to note that the three identified impact pathways are hierarchical. Displacement of individuals only occurs outside the area of habitat alteration. Only individuals which were not displaced face the risk of collision with turbines. This hierarchy should be taken into account to avoid double counting. However, species are known to respond behaviorally to these risks through avoidance, reducing the risk of an impact to occur (May 2015). Attraction of bats, or birds, toward wind turbines may on the contrary lead to increased occurrence and thereby a higher risk of collision. Such pertinent avoidance and attraction effects should therefore also be taken into account.

Furthermore, it is necessary to consider that different species or populations may be more vulnerable to an impact than others. Understanding a species' or species group's behavior and population dynamics is key to adequately integrating vulnerability at an impact level. (Verones et al. 2013) added a vulnerability score to their LCIA characterization factors for biodiversity impacts from water consumption. The authors developed this score from species geographical distribution ranges together with IUCN threat levels. More variables could be added in order to adapt this method to other types of impacts on biodiversity, such as those from onshore wind energy on bats and birds. It is also important to keep the spatial scale that the methodologies are developed for in mind. Characterization factors developed for a certain region may not be applicable in another due to differences in species composition, vulnerability, as well as technical and environmental characteristics. Furthermore, data may not be available for every region in the same quantity or quality, which therefore adds uncertainty to methodologies developed at a global scale. In addition, scaling up or down (i.e., going from a local to a global spatial scale, or vice versa) must take into consideration that species composition, as well as environmental variables, may change in the process (Wessman 1992).

Irrespective of the approach used to quantify the impacts in question, various types of data are required (Table 1). Several existing databases cover some of these information needs (e.g., species data, turbine characteristics and locations, environmental data), while other types of data may require the use of allometric relationships (e.g., bird wing loading from body mass). Empirical species-related data at a global level can be obtained from BirdLife International (2018) on birds, while IUCN (2017) provides data on many other species groups, including threat status and range maps. For occurrence data, GBIF (2017) provides an open access database with location data for more than 1.6 million species. In addition, Wilman et al. (2014) compiled a great amount of data on animal diet and mass for all extant bird and mammal species, which can potentially be used to estimate important morphological parameters such as wing loading and aspect ratio using allometric relationships (Norberg 2006). Lack of species data can also potentially be coped with by using better-known species, with similar characteristics, as proxies for a larger group (Denzinger and Schnitzler 2013). Such data can be used to, for instance, rank species according to characteristics that render them more vulnerable to the different impacts of wind energy developments. Environmental data, such as wind speed and topography, may be required to estimate a species' potential distribution, especially when using SDM software such as MaxEnt. Temperature and wind speed data can be acquired from databases such as the NASA Langley Research Center Atmospheric Science Data Center Surface meteorological and Solar Energy (SSE) web portal (NASA 2016), among others. The U.S. Geological Survey (2016) provides remote sensing data, including digital terrain models. Technological data may be available through direct contact with the operating company or local datasets. Remote sensing databases such as the CORINE Land Cover (Heymann et al. 2000) can provide information for present land cover types, which can also aid in the prediction of a species' preferred habitat. Knowledge on a species' flight initiation distance allows the determination of the extent of area disturbed for that species, although no database currently exists to provide these distances for a large number of bird species (but see Blumstein 2006). Lastly, although many of these databases provide relatively generic data, local datasets may also exist with higher resolution or more accurate data (e.g., in Norway—Artsdatabanken 2017; Kartverket 2017; NVE 2017) to complement larger databases.

6 Conclusions and recommendations

Current literature on the impacts of wind energy on biodiversity directed the focus of this article on two main research gaps: a lack of a review on predictive quantitative methods on the topic and a lack of attempts to develop a methodology for LCIA to address these types of impacts. This is a first effort to provide the necessary background knowledge for the development of said LCIA methodology, in terms of the effects and impacts of wind energy on birds and bats and how these are modeled outside of LCA. Based on the results in this study, we can now start to develop LCIA models for assessing impacts of onshore wind power on birds and bats.

Collision, displacement, and habitat alterations have been identified as the main impacts of wind energy on wildlife. According to current research, birds and bats are the most susceptible species groups to these effects for onshore wind turbines. As their responses to wind energy developments are considerably different, models should be developed separately for each of the two species groups. In addition, assessment of these species should take into consideration that within the two taxonomic groups there is considerable behavioral and morphological variation, especially among bird species.

Existing predictive models for the three main impact pathways show that quantitative estimations can be performed. GIS tools and remote sensing have proven invaluable in spatially differentiating areas of variable risk. More specifically, SDMs are widely used for determining areas of higher probability of conflict with biodiversity. This type of modeling has proven especially important in collision risk modeling, given the existing scarcity of data usually required by the more complex CRMs. However, an application of SDMs at a global scale for estimating wind energy impacts on biodiversity is still lacking. Index-based models offer a clear, simplistic approach to not only scale impacts according to the species' sensitivity but to include certain aspects that are often excluded from assessments, particularly those related to a species vulnerability (e.g., life-history traits, behavior).

Inclusion of the three main pathways for impacts of wind energy on biodiversity in LCIA requires adaptation of these quantitative methods to the methodologies used in the LCA framework. In other words, results must be compatible with those of other ecosystem-related impact categories, which should be communicated in units of PDF (Verones et al. 2017a). As an example, in order for a number of fatalities to be integrated, knowledge of a total number of individuals would be needed, so that a percentage loss of each species is obtained. This integration must be spatially explicit, with the support of GIS tools, given the variability between regions or countries in terms of ecosystem composition and wind energy technology. We suggest local characterization factors be constructed first, as data requirements should be lower and more accessible. Once a working model is in place, it should then be followed by an attempt of upscaling to a global level, taking into consideration data and technological constraints of upscaling models. In either case, we point out that modeling habitat alterations, together with or followed by disturbance, is more readily feasible compared to collision. Modeling the first two impact pathways relies strongly on available GIS tools and remote sensing data, as well as knowledge of each species group's general behavior toward wind turbines. SDMs show promise in their ability to tackle this set of impacts and can be combined with currently used SARs in order to directly obtain characterization factors in units of PDF, as described before. Vulnerability should be introduced at this point for instance by means of indexes in order to weight species according to how strongly they are affected.

The proposed LCIA development is not only a step towards more comprehensive impact assessments in LCA but also outside of it. Most of the reviewed quantitative methods focused on only one or two of the three main impact pathways and at relatively small scales. Also, many studies are based on small samples or on few species that are not representative for all birds or bats (Sovacool 2013). This underlines the importance of grouping species after, e.g., morphological similarities and creating archetypes for environmental conditions when data for all species and conditions is not available. Furthermore, there is still a lack of impact quantification relative to the energy produced by each turbine or wind farm. This hinders the possibility of an adequate comparison between wind energy production and other types of energy production, as well as between wind farms with variable production efficiencies. In the future, LCA has the potential to cover all these gaps, as well as integrate impacts on biodiversity from other energy sources.

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