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Evaluating anthropogenic landscape alterations as wildlife hazards, with wind farms as an example

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ABSTRACT

Anthropogenic alterations to landscape are indicators of potential compromise of that landscape's ecology. We describe how alterations can be assessed as 'hazards' to wildlife through a sequence of three steps: diagnosing the means by which the hazard acts on individual organisms at risk; estimating the fitness cost of the hazard to those individuals and the rate at which that cost occurs; and translating that cost rate into a demographic cost by identifying the relevant demographically-closed population. We exploit the conservation-oriented literature on wind farms to illustrate this conceptual scheme. For wind farms, the third component has received less attention than the first two, which suggests it is the most challenging of the three components. A wind farm provides an example of a 'spatially localized hazard', i.e., a discrete alteration of landscape hazardous to some population but of which there are some individuals that do not interact directly with the hazard themselves but nevertheless suffer a reduction in fitness in terms of their contribution to the next generation. Spatially localized hazards are identified via the third component of the scheme and are of particular conservation concern as, by their nature, their depredations on wildlife may be underestimated without an appropriate population-level estimation of the demographic cost of the hazard.

1. Introduction

Anthropogenic alterations of landscape may serve as an indicator of ecological compromise to habitats, though not in a straightforward manner. Determining the consequences for wildlife of such landscape transformation is an important component of conservation science. The implications of habitat loss and fragmentation have received considerable attention (Henle et al., 2004) and contributed to important theoretical developments such as spatially structured population modelling (Hanski, 2002). Less overtly disruptive landscape transformations hazardous to wildlife may be diffuse and widespread to the point of ambience, notably pollutants such as the historic worldwide occurrence of DDT in avian food webs (Hickey, 1969). Discrete alterations of landscape may also pose risks to wildlife. Notably, structures threaten collisions, particularly of birds (Drewitt and Langston, 2008; Loss et al., 2014a), and clusters or networks increase that risk. Networks of power lines pose the additional risk of electrocution (Drewitt and Langston, 2008; Loss et al., 2014b), while networks of roads and fences (Benítez-López et al., 2010) also constrain movement such as migrations and dispersion (Gadd, 2011).

Evaluating the danger posed to wildlife by landscape alteration can be a subtle conservation challenge. A discrete, but common, landscape alteration may be treated as an ambient risk when evaluating the implications for a regionally defined 'population' of interest. For example, counts of the annual mortalities of birds (or of a particular species) resulting from collisions with buildings or power lines might be converted into an additive component of per capita mortality rate based on estimates of the number of birds (of the species of interest) within some, politically or geographically defined, region. While such computations, and comparisons between them, may aid conservation (Loss, 2016), they have a poor biological basis if the 'population' is not a demographically closed population or if the risk posed by the landscape alteration is not the same for all individuals in the 'population'.

Our objective in this essay is to outline a conceptual scheme with a biological basis for evaluating the risk posed to wildlife by landscape alterations, focusing on discrete transformations of landscape. Hereafter, an alteration of landscape that is identified as posing such a risk will be called a 'hazard'. The scheme consists of three components: (1) identifying those features of a landscape alteration that act as the agents of risk and their modes of action on individuals interacting with the landscape alteration, which provides the foundation for identifying a landscape alteration as a 'hazard' and is essential for possible

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mitigation; (2) quantifying the reduction in fitness to those individuals directly interacting with the hazard, and estimating a rate of occurrence, which may take the form of annual counts of hazard-induced fatalities; (3) converting the rate obtained in (2) into a demographically meaningful per capita vital rate, which involves identifying the relevant population that pays the cost. The meaning of 'direct interaction' in (2) will be a byproduct of (1). The third component addresses the issue raised in the previous paragraph of measuring the impact of a hazard by, say, an annual mortality count, which is demographically ambiguous. We propose that the relevant population for estimating demographic cost is the smallest, demographically closed population containing the individuals interacting directly with the hazard, which we shall call the 'hazard's demographic population'. A further ingredient of the third component is determining whether the risk and fitness costs vary across individuals in this population, which, if so, adds to the subtlety of (3). We believe that organizing the evaluation of landscape alterations as spatial hazards into this scheme facilitates this important conservation task by clearly separating aspects of the problem into components that are conceptually distinct. The scheme will thereby aid planning, implementation, and assessment of such an evaluation.

As a byproduct, we formalize a notion of a particular kind of discrete alteration of landscape as follows. By a spatially localized hazard (SLH), we shall mean a hazard for which there are individuals in the hazard's demographic population that never interact directly with the hazard (never enter the spatial extent of the hazard) yet suffer a reduction in individual fitness indirectly (as measured by their contribution to future generations). The third component is essential for the identification of an SLH and alerting scientists/managers to the existence of individuals that are a less visible yet integral part of assessing the overall demographic cost of the hazard.

We emphasize that whether a particular hazard is best treated as ambient or discrete will depend on the spatial scale relevant to the organisms of interest. For an isolated (perhaps sessile) population of organisms existing within the spatial extent of the hazard, the hazard may be best treated as ambient, while for another, more widely distributed, and perhaps more mobile, organism, the same hazard may be discrete and even an SLH, though not necessarily so. Thus, the identification of a hazard as discrete, and more specifically an SLH, depends on both the hazard itself and the organisms interacting with it. The third component of the conceptual scheme underscores that the nature of the hazard itself and the population at risk might only become apparent through analysis, rather than being a priori evident.

Our formulation of the SLH concept was motivated by a study of the impact of the wind farm in the Altamont Pass Wind Resource Area (APWRA), California, on golden eagles *Aquila chrysaetos* (Hunt et al., 2017). As wind farms are an increasingly common component of renewable energy production and result in considerable deaths of birds and bats (Kunz et al., 2007a,b; Drewitt and Langston, 2008; Pagel et al., 2013; Smith and Dwyer, 2016), we will illustrate the conceptual scheme by exploiting the literature on wind farms, which will suggest that the third component of the scheme has received much less attention than the first two, no doubt due to its difficulty. Of course any potential hazard, such as the APWRA, and species interacting with it will have their own peculiarities and require analyses specific to that instance. It is for this reason we emphasize here a conceptual scheme. Nevertheless, we shall use the golden eagle study to illustrate how the scheme can be tailored to a specific application.

For wind farm literature, we first conducted a search using the terms 'spatial hazard', and 'wind farm' in the search engine Web of Science. To be abreast of ongoing publications, we examined emails notifying Table of Contents for the following journals: *Biological Conservation*, *Conservation Biology*, and *Journal of Applied Ecology*, and, less formally, perused *Journal of Raptor Research* and *Condor*. We then used a recursive procedure of examining the literature cited in each publication we found. We do not claim our literature is exhaustive, but we believe that it is at least representative through 2017 of the published literature on wind farms as it relates to our paper. Our review of this literature is integrative in that we cite publications within the context of our conceptual scheme. As noted above, we formulated our conceptual scheme based on our experience studying a golden eagle population in and around a wind farm in California and prior to conducting our literature search and review. We were prepared to modify our scheme if the literature indicated it was inadequate but did not find that to be the case.

2. Identifying the agents of a landscape hazard and their mode of action

For the remainder of this essay, we have in mind a discrete alteration of the landscape that is considered to pose a risk to some organism, some individuals of which interact with the potential hazard. By the 'agents' of the hazard, we mean identifiable interactions between hazard and individual expected to result in fitness costs to that individual, while 'mode' of interaction refers to features that govern the interaction.

The primary hazard agent posed by wind farms (other than habitat loss, Villegas-Patraca et al., 2012; Smith and Dwyer, 2016) appears to be fatal collisions of birds and bats with moving blades (Kunz et al., 2007a,b; Drewitt and Langston, 2008; Smith and Dwyer, 2016), though avian collisions with static components of wind farms (Drewitt and Langston, 2008) and internal soft-tissue damage in bats from air decompression (Kunz et al., 2007b) also occur.

While coincidental factors such as weather may contribute to wind farm deaths (Arnett et al., 2008), especially for migrating and flocking species, the study of why and how collisions between birds and bats with moving rotors occur has most usefully focused on intrinsic features of the wind farm and the species concerned. Intrinsic features of wind farms (Arnett et al., 2008, 2011; de Lucas et al., 2008; Horn et al., 2008; Kikuchi, 2008) include geographical location relative to existing flight paths, e.g., migratory routes, the landscape features on which the wind farm is imposed, the spatial layout of the wind farm itself, characteristics of the turbines such as height, length and speed of the blades, wind-farm lighting, and natural attractants within the farm such as prey (ground prey for raptors; flying insects for insectivorous bats). Intrinsic features of the vulnerable species (Kunz et al., 2007a,b; Kikuchi, 2008; Smallwood et al., 2009) include flight behaviour (e.g., flocking versus solitary; hunting/foraging flight, Horn et al., 2008; territorial display; species-specific flight characteristics, Hoover and Morrison, 2005, De Lucas et al., 2008; time of day) and morphology affecting flight (body weight and size; wing length, loading and aspect ratio; tail length; De Lucas et al., 2008); vision (binocular versus peripheral vision acuity, Martin, 2012); seasonal behaviour (migratory behaviour, Hüppop et al., 2006; Arnett et al., 2008); differences in life stage (e.g., floater versus breeder; subadult versus adult, Hunt et al., 2017); and the natural distribution and abundance of populations in relation to wind farm location and their environs (de Lucas et al., 2008). Nevertheless, wind-turbine-induced deaths of migratory tree bats were not adequately explained by such factors indicating that wind-farm-induced deaths are not yet understood for all organisms at risk (Jameson and Willis, 2014; Cryan et al., 2014).

Identification of the hazard agent and study of its mode of action can lead to modelling of the agent and its action, or at least to hypotheses and predictions, e.g., of collision risk in wind farms (reviewed by Marques et al., 2014; see also: Kikuchi, 2008; Noguera et al., 2010; Eichhorn et al., 2012; Ferrer et al., 2012; New et al., 2015) that may result in mitigation (Drewitt and Langston, 2008; Arnett et al., 2013; Marques et al., 2014; Martin et al., 2017; May et al., 2017). This modelling has focused primarily on the probability of single incidents, either of a single individual colliding with a turbine or of a flock colliding with a wind farm, but Wiens et al. (2017) employed individual-based population modelling in a spatially explicit context to assess the consequences of increasing wind farm development in southern California and propose pre-emptive mitigation.

The hazard agent for golden eagles at the APWRA is turbine strike, typically killing an eagle outright (Smallwood and Thelander, 2008). Interestingly, turbine-strike deaths are restricted primarily to subadults and floaters (adults without a territory and thus non breeders); territory holders rarely enter that wind farm (Hunt et al., 2017). Ground squirrels, possibly in greater numbers than in surrounding environs, probably lure non-territorial life stages into the wind farm (Hunt and Watson, 2016). The immunity of juveniles to turbine-induced death may be due to their inexperience at hunting such prey and thus not exposing themselves to that risk. In the APWRA, replacement of smaller, faster-rotating, densely packed turbines by taller, slower-rotating, more judiciously located turbines appears to be reducing eagle mortality (Smallwood and Karas, 2009), which is consistent with the hypothesis that golden eagles are vulnerable while 'contour hunting' for ground prey (Watson, 1997).

Data on fatalities together with variables that may characterize mortality risk (e.g., spatial distribution of fatalities together with eco-geographical variables, Santos et al., 2013) offer another approach to modeling and predicting risk, which for hazards that are deliberate alterations of landscape offer the possibility of pre-emptive reduction of mortality risk. This approach raises the issue of quantifying for example the annual rate of mortalities associated with a hazard.

3. Quantifying the fitness costs to individuals directly interacting with the hazard and their rate

Direct interaction of an individual bird or bat with a wind farm involves suffering damage due to collision or from pressure changes near turbines. While there is some evidence for non-fatal damage, the primary focus in the literature is on fatal collisions with turbines, which was also the direct interaction experienced by golden eagles flying within the APWRA. The fitness cost of direct interaction with the wind farm is therefore typically an increased risk of death for those entering the space of the wind farm. This simple fitness cost therefore focuses attention on estimating a rate at which it occurs, but in other cases the fitness cost suffered by an individual may be more subtle.

There have been numerous estimations of the number of fatal collisions with turbines at wind farms (reviewed by: Kunz et al., 2007a,b; Drewitt and Langston, 2008; Loss et al., 2013; Pagel et al., 2013). Typically, estimation of the number of deaths must also estimate the probability of detecting a carcass in a surveying area (Piorkowski and O'Connell, 2010; Huso, 2011), which will depend on a number of factors, most notably effort. For estimating wind-farm-induced fatalities, effort will be a function of coverage of the wind farm facility and frequency of searches. Partial coverage requires extrapolation. Frequency of searches affects probability of detection due to carcass loss from decay and scavengers, requiring further extrapolation. Factors affecting persistence and identifiability of a carcass will also make detection less than perfect. These challenges to counting wind-farm-induced deaths have been the subject of field studies, the results of which have been incorporated into various field counts and estimators of the annual rate of such fatalities (Smallwood and Thelander, 2008; Korner-Nievergelt et al., 2011; Bernardino et al., 2013; Péron et al., 2013; Huso et al., 2015).

Ferrer et al. (2012) combined counts of the number of wind-farm-induced fatalities with counts of exposure to the risk of turbine collisions to estimate risk for those individuals entering a wind farm. Another possible approach to assessing the impact of a hazard is to model the distribution of the population as a function of landscape/habitat features, including the hazard itself, though population suppression near a hazard may be ambiguous as to whether it is the result of displacement or mortality (Pearce-Higgins et al., 2009).

Fatalities resulting from collisions with turbines are often reported as annual rates per wind farm, or number of deaths per turbine or GWh. The last measure is considered useful for comparing the 'cost' to a taxon of different kinds of energy production at, for example, a national level, e.g., wind farms versus other forms of generating electricity (Sovacool, 2012), though related endangered, rare, and common species may be lumped together in a single taxon, thereby obscuring conservation concerns. At current levels of wind farm deployment, quoted figures typically indicate that the cost of wind farms, measured in this manner, is considerably lower than other forms of electricity production, as are raw counts of fatal collisions at wind farms compared to collisions with more widely distributed static obstacles imposed by humans on the landscape, such as buildings, towers, power lines, and fences (Sovacool, 2012). A count such as the number of deaths per GWh, however, is of dubious value in a cost-benefit analysis of an alteration to the landscape since it does not clearly identify demographically meaningful populations to which the costs refer. Another limitation is that such counts ignore the life history of the species of interest (Loss, 2016). The fatalities may impact some stages, ages, or sexes more than others and thereby have greater or lesser population consequences than unstructured fatality counts suggest.

Thus, one requires estimates of the number of hazard-induced fatalities, taking into account the probability of detection, and expressed in a relevant form (number per year or turbine) that also respects the life history of the species of concern. For example, Hunt et al. (2017) found that wind-farm induced fatalities of golden eagles at the APWRA were almost exclusively of subadults and floaters. Translating these results into an evaluation of the demographic cost suffered by the hazard's demographic population constitutes the third, and vital, ingredient of our scheme.

4. Evaluating the demographic cost of a hazard

We found little wind-farm literature addressing this aspect, though the distinction between the annual estimates of fatalities and a population-level estimation of cost has been noted, and sometimes stressed (B. Kendall in Schwartz 2004:59–60; Carette et al., 2009; Loss et al., 2012; Loss, 2016). Green et al. (2016) stressed that identification of the source populations for sea birds killed in offshore wind farms is lacking and thereby calls into question the demographic meaning of kill rates. Diffendorfer et al. (2017) provided a methodology for converting annual wind-turbine kills of birds or bats into national annual mortality rates. Such a scheme can raise an alarm if a species is threatened at the national level by wind farms but may overlook threats to the demographic population of a specific wind farm or regional network of wind farms. Genetic profiling and stable isotopes may provide information about the geographic origin of individuals of highly mobile species killed by a hazard (Kunz et al., 2007a, Katzner et al., 2016), and thus at least identify it as an SLH. Presumably the dearth of literature on this component reflects the difficulty in acquiring the relevant data to place annual hazard-induced mortality rates, say, in a population context (Frick et al., 2017).

We use the study of Hunt et al. (2017) of golden eagle fatalities in the APWRA to illustrate what might be involved in evaluating the demographic cost of a hazard. Hunt et al. (2017) employed known-fate analysis of a radio-tagged sample to obtain mortality rates per life stage for this sample, both with turbine-induced deaths censored and uncensored. As noted above, turbine-induced deaths occurred almost exclusively to subadults and floaters, providing important life history structure to these fatalities. Published estimates of turbine-induced deaths of golden eagles in the APWRA during the period of Hunt et al.,'s study were of about 55 – 65 per year (Smallwood and Karas, 2009). The annual death count and life-stage mortality rates only provide a partial picture for the demographic impact of the APWRA on its demographic population of golden eagles, however. While breeders rarely directly interact with the wind farm, they suffer an indirect fitness cost.

Since floaters provide replacements for breeders (subadults become breeders usually only by first transitioning to the floater life stage), i.e., fill territory vacancies arsing from breeder mortality, the breeding life stage is indirectly affected by the death of offspring in the form of floaters and subadults in the wind farm. Hunt et al. (2017) found that of 58 territories in 2000, mostly located within 30 kms of the wind farm boundary, all were occupied in 2005 and all but two in 2013 (one vacancy could be explained by circumstantial factors). Moreover, no trend in the proportion of subadults as territory occupants was detected that would have suggested a lack of floaters to fill territory vacancies. This evidence might lull one into thinking the territories near the wind farm can support the extra mortality imposed by the wind farm.

The number of territories sourcing both territorial and non-territorial golden eagles in and around the wind farm is unknown; i.e., the demographic population of the APWRA is still unknown. To circumvent this problem, Hunt et al. (2017, Appendix S4) used their computed vital rates, with wind-farm induced deaths censored, as follows. Since the average age of a golden eagle killed by collision with a turbine in this study was estimated at 40 months from fledging, this demographic data was employed, together with the study's estimate of annual breeding-pair fecundity, to compute the number of territorial pairs required to be self sustaining and otherwise produce a specified number N of offspring that survive to the age of 40 months. If N is the number of eagles killed by turbines each year, one obtains an estimate of the number of territorial pairs that just sustain themselves and the wind-farm induced mortality. The number of territorial pairs satisfying these requirements is linear in N so the result can be quoted as 3.93 territorial pairs per wind-farm induced death. The annual wind-farm induced fatality rate of 55 – 65 results in an estimate of 216 – 255 territorial pairs. The estimated total number of territorial pairs in the Diablo Range study area (5560 km^2, versus the 1500 km^2 core study area containing the monitored territories and wind farm) in 2014–2015 was 280 (95% CI = 256–305 pairs; Wiens et al., 2015).

Thus, the preceding minimum estimate of the demographic population of golden eagles for the APWRA is a substantial proportion of all the territories in the Diablo Range study area, but possibly extends beyond this range. This calculation, therefore, provides a measure of the demographic cost of the APWRA to golden eagles. It also indicates the range over which a study would have to be conducted to further pin down the demographic population itself and so the resources that would be necessary for such a study. The computation also identifies the APWRA as an SLH for golden eagles, a conceptual advance in the biological understanding of the interaction between golden eagles and this wind farm by extending the focus beyond those eagles directly interacting with the wind farm. The modelling conducted in Hunt et al. (2017) and outlined above can guide future studies of the APWRA and its impact on golden eagles but also highlights the need to address this third component of our conceptual scheme for evaluating the hazardous prospects of other wind farms and SLHs more generally.

5. Discussion

Hunt et al.'s (2017) study of the interaction between golden eagles and the APWRA and our review of the wind-farm literature led us to identify three conceptual components for evaluating wind farms as hazards to wildlife, golden eagles in particular. We found that an extensive literature exists addressing the first two components but that the third component, while it has been noted and sometimes stressed, is less often given substantive treatment in the literature. This lack constitutes an important shortcoming in the evaluation of wind farms as hazards to wildlife but no doubt requires substantial research effort.

Our conceptual scheme, diagrammed in Fig. 1, should be useful in the broader context of evaluating any anthropogenic alteration of the landscape as a hazard to wildlife. In particular, the third component of our scheme is vital to identifying SLHs and their more subtle effects on individuals not interacting directly with the hazard itself.

Some hazards, such as the long term effects of toxic spills and nuclear accidents, may act not by killing outright but by reducing the fitness of individuals interacting with the hazard, by reducing the probability of survival and/or fecundity, e.g., of birds near Chernobyl

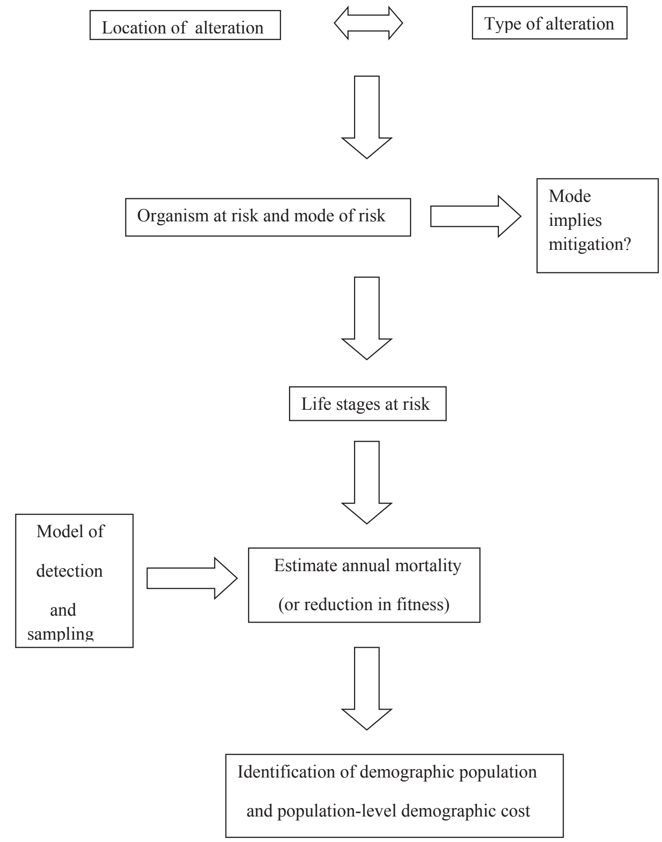

Fig. 1. Evaluating a discrete landscape alteration as a potential hazard to wildlife.

and Fukushima (Møller et al., 2012a,b; Mosseau and Møller, 2013). The modes whereby a toxin reduces fitness will depend on both the toxin and the species and may be cryptic, e.g., only some passerine species apparently show reduced nestling body condition due to lead poisoning (Roux and Marra, 2007) while other effects may occur secondarily through food-web transfer (Rattner et al., 2014). In such cases, the first and second components of our scheme are likely to be more challenging than for wind farms and golden eagles. The effect of a hazard may taper with distance (Powell et al., 2017), e.g., as toxicity declines from the focal point of a contaminant, but there may also be a fitness cost paid by individuals that are not exposed to the contaminant, i.e., the hazard may act as an SLH. Only the third component of our scheme can identify this possibility.

Point-source pollution, a single identifiable source of pollution from which pollutants are discharged, may or may not amount to an SLH. This observation highlights that the characterization of a hazard as an SLH depends on the spatial extent of the hazard and the spatial distribution of the individuals affected, including those affected only indirectly. A point-source pollution hazard may be ambient for an isolated population, e.g., of sessile organisms, whose spatial extent is covered by the pollution, yet an SLH for another, more spatially extensive, population of mobile individuals, some of whom never interact with the pollution directly yet pay a fitness cost. Thus, our notion of an SLH is not identical to that of point-source pollution, which narrowly focuses on the spatial properties of its source rather than the spatial scale of its interaction with populations of organisms.

As another example, showing the flexibility of the concept of SLH, domestic cats concentrated in urban areas surrounded by rural countryside may be viewed as a discrete anthropogenic alteration of the landscape. The importance of identifying the impacted bird populations (component three), and not just the number of birds killed (component two), has been recognized in assessing the predation cost of domestic cats on birds (e.g., Baker et al., 2005; van Heezik et al., 2010) and whether their urban concentrations act as SLHs.

The conceptual scheme we have outlined for evaluating landscape alterations as potential hazards does not dictate methodology for any component. Appropriate methods will depend on the landscape alteration and the species potentially at risk. Such specificity is recognized, for example, in the US Fish and Wildlife Service (2013) guidance for management of golden eagles *Aquila chrysaetos* at risk from wind energy projects. The identification of SLHs emphasized by the conceptual scheme, however, importantly broadens the scale of conservation action from that of personnel such as local managers focused on the question of whether a proposed, or existing, wind farm adversely impacts the perceived local 'population' to those managers with more regional responsibilities in a manner that would integrate conservation planning over more demographically relevant spatial scales.

The literature regarding any specific landscape alteration, such as wind farms, may nevertheless provide useful guidance and methods when suitably adapted to other circumstances. The killing of a bird or bat by a turbine sounds simple but the wind farm literature indicates the range of features of wind farms, organism, and context that may be needed to explain this mechanism of mortality, and offers methods for detection, and counting, of victims, and estimation of probabilities of suffering this fate, which may serve in other contexts. Our method for estimating a minimum size of the demographic population of golden eagles for the APWRA, though it reflects the life history of golden eagles, generalizes to a method of estimating how many reproductive 'units' of the organism in question are required to sustain themselves and the annual rate of hazard-induced mortality. How this computation is carried out will depend on the life history of the particular organism but nevertheless offers an approach for addressing the third component of the conceptual scheme and provides a preliminary step towards identifying the actual demographic population itself. Retroactive investigation of the impacts of existing landscape alterations on wildlife and prospective studies of proposed alterations will benefit from considering all three aspects of the conceptual scheme of Fig. 1.

Schematic for assessing the impact of a discrete landscape alteration. The first step includes several interactive parts. One must first identify the landscape alteration and the organisms potentially at risk. These two objectives interact because the nature of the structure (e.g., wind farm) informs one as to the organisms at risk (viz., birds and bats rather than terrestrial animals and plants) while the populations present indicate the relevance of the potential hazard to specific species of conservation concern. With this prerequisite accomplished, the first step involves identifying those features of the hazard and of a specific organism at risk that characterize how the hazard acts on the individuals of that species. Not only do these features aid in evaluating the hazard risk but they may also lead to remedial actions. Next it is vital to plan an estimation procedure of mortality (or more generally fitness reduction) that accounts for life history of the species, detection of mortality, and an explicit sampling protocol. The resulting estimates must then be translated into relevant population-level demographic costs.

6. Conclusion

We have outlined a conceptual scheme for evaluating anthropogenic alterations to landscapes as hazards to wildlife and illustrated the scheme with the literature on wind farms. While the first two components will be familiar to those who have studied such hazards, we believe the third component has received less attention and consequently may obscure the demographic cost of such a hazard by failing to identify the demographically, rather than geographically, defined population paying that cost. We provided an example that illustrates some of the issues in obtaining such a demographic assessment. Our scheme led us to identify a particular kind of hazard, which we call a spatially localized hazard, for which some individuals share the demographic cost even though they do not directly interact with the hazard, which further complicates a true assessment of the demographic cost of the hazard.

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Conflict of Interest

None.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at https://doi.org/10.1016/j.ecolind.2018.06.061.

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