Assessment, mitigation and monitoring of onshore wind turbine collision impacts on wildlife

A systematic review of the international peer-reviewed literature, and its relevance to the Victorian context

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Glossary

BACI: Before-After-Control-Impact, an experimental design which allows one to distinguish changes caused by a treatment (such as the implementation of a mitigation) from environmental factors and random variation that may have occurred coincidentally while the experiment was taking place. In this study design, key response variables are measured both 'Before' and 'After' the treatment is implemented (and often during as well), and sites or individuals are stratified such that some are exposed to the treatment ('Impact') and some are not ('Control').

Bayesian: A field of statistics where probability is expressed as a 'degree of belief' in an event, which can also incorporate prior knowledge such as existing empirical evidence or expert knowledge.

Buffer: The distances that regulations or guidelines specify wind turbines should be set back from key habitat features. In the context of the mitigation hierarchy (see below) buffers are an avoidance measure, but within the literature are also discussed as mitigations. We have included them in the mitigation section in this review.

Cumulative impacts: Biodiversity impacts that accrue over time and/or over space, and are greater than the assessment of a single facility or point in time would suggest. This may refer to some combination of: i) impacts that accrue as turbines are added to the landscape; ii) impacts that accrue as turbines continue to operate through time; iii) the overall combined effects of multiple types of wind energy-related impacts, such as displacement, loss of foraging resources, and mortality; or iv) the overall combined effects of multiple types of threatening processes, such as collisions, disease, and habitat loss to urban development.

Curtailment: Broadly used to describe times when turbines blades are not rotating and producing energy when they otherwise would be, because of settings that have been put in place. 'Low wind speed curtailment' specifically refers to a mitigation measure predominantly used at night-time for microbats, where the minimum wind speed at which turbines start rotating and generating electricity is increased to reduce collisions.

Cut-in speed: The wind speed at which turbines become operational and start feeding power back into the grid, and typically also the speed at which blades start rotating (when feathering is in place).

Data synthesis: Summarising or analysing data that have been collected in a standardised way, typically acquired from a central source or sources such as government authorities.

Echolocation: A form of sonar used by microbats to hunt and navigate. Sound waves are produced by the animal, and when these hit an object or a prey item they bounce back and are processed by the bat. These sounds are ultrasonic (high frequency), so most are inaudible to humans. While microbats primarily use echolocation for navigation, most also have good eyesight.

Feathering: When turbine blades are tilted parallel to the wind to stop or slow the turbine blades below cut-in speed, to prevent freewheeling of the blades when no energy is being produced.

Flying-fox: A large bat species from the genus, *Pteropus*, which feeds on pollen, nectar and fruits, and navigates predominantly by eyesight. Two flying-fox species are resident in Victoria (the Grey-headed Flying-fox, *Pteropus poliocephalus*, and the Little Red Flying-fox, *Pteropus scapulatus*).

Hub: See nacelle.

Microbat: Bats that are typically small (e.g. in Victoria, 3–48 g), echolocate and eat insects. Microbats are also sometimes referred to as 'insectivorous bats' or 'echolocating bats', which is useful in the Victorian context to distinguish them from the ecologically and morphologically distinct flying-foxes (see above).

Mitigation hierarchy: Sequential steps that are followed to reduce impacts to biodiversity associated with developments. For wind energy, these steps are 'avoid' (planning/siting), 'minimise' (mitigation actions), and 'compensate/offset' (to address residual impacts that cannot be avoided or minimised.

MET mast: A meteorological mast or tower. These are present at planned or constructed wind energy facilities, and are equipped with instruments to measure and record wind speed, direction, temperature, and air pressure. Acoustic recorders for microbats (bat detectors) are sometimes attached to MET masts to survey activity 'at height'.

Mortality: Also referred to as 'fatality' (used interchangeably here), the number of birds or bats killed through turbine collisions. Mortality or fatality rates are the number of birds or bats killed over a set time frame or spatial area (e.g. bats killed per year, or birds killed per turbine).

Nacelle: The turbine nacelle or 'hub' houses the turbine components at the rotor/centre of the turbine blades.

Parrot: Terrestrial birds of the order, Psittaciformes, with a strong curved bill that feed on nuts, seeds, fruits and buds. This group includes parrots, cockatoos, and galahs.

Passerine: A diverse group of perching birds of the order, Passeriformes, including the songbirds. This includes many small and medium-sized terrestrial Victorian birds such as magpies, larks, ravens, honeyeaters, and fantails.

Population sink: An area or subpopulation where the mortality rate exceeds the birth rate, and which can only persist through immigration from other source populations.

Raptor: Carnivorous birds that actively hunt or scavenge, and often take advantage of thermal updrafts during soaring flights, including eagles, hawks, kestrels, falcons, buzzards, kites and harriers. Here, we have included vultures as raptors because of the shared traits (hooked beak, large body size, and flight, migratory and feeding behaviours) that make them comparably vulnerable to turbine collisions as other raptor species. However, New World vultures are evolutionarily more closely related to storks.

Rotor-swept area (RSA): The circular area that turbine blades move through as they rotate. Birds and bats flying in this area are at risk of collisions. Also sometimes referred to as the 'Rotor-swept zone' (RSZ).

Volant: Used to describe an animal that is capable of flight, typically used to refer to birds, bats, and insects.

Yaw: The compass direction that a turbine is pointing. The yaw system controls the extent to which the turbine nacelle is pointing either into or away from the prevailing wind.

Summary

Context:

Renewable energy sources are critical for reducing greenhouse gas emissions, and the momentum of wind energy development in Victoria is now increasing to meet renewable energy targets. However, these developments also have direct, indirect and cumulative impacts on biodiversity, and fatalities of birds and bats due to collisions with wind turbines are of particular concern. Given the concerns about managing these impacts, and calls from the energy sector for guidance documents to improve clarity and consistency, a robust evidence base is required for decision making.

Onshore wind energy has been used in parts of the world for many decades, and some of the questions relevant to Victoria in relation to wind energy and biodiversity have already been well-studied elsewhere. Guidelines, policies and practices developed for Victoria can draw on this large body of evidence, acknowledging that there may be important differences within the Victorian context that need to be considered. While there are many existing guidelines, handbooks, and protocols from elsewhere in the world, it is sometimes difficult to determine what evidence underpins the various recommendations contained within these, and how they apply to Victorian species and landscapes. Our objective here was to systematically review the primary literature associated with wind energy research undertaken globally. From this, we sought to provide a comprehensive but accessible summary of both what is known and the key knowledge gaps as they apply to Victoria, and to make the link between the trends that we highlight and the associated evidence clear.

Aims and scope:

Specifically, the aims of this review were to:

- systematically review the available published peer-reviewed scientific literature on wildlife collisions at onshore wind energy facilities
- provide a transparent summary of evidence and knowledge gaps associated with wildlife impacts, assessment of risk, survey protocols, mitigation measures, and monitoring
- present the findings in the context of wind energy developments in Victoria, and consider how applicable this evidence is to the landscapes and species within the state.

To ensure that the review remained tractable, the scope was limited to evidence as it relates to onshore wind energy and impacts on birds and bats. Impacts of offshore wind energy developments were out of scope. While we also reviewed evidence related to impacts such as displacement, avoidance and attraction, our primary focus was on direct collisions, given their immediate nature and the consequential impacts on wildlife populations. Only scientific, peer-reviewed literature published in English were within scope for this review. We acknowledge that there are papers published in other languages on this topic, and that by only reviewing material written in English we have likely introduced some geographic bias. We did not include grey literature, as this not peer-reviewed, and we also excluded guideline and policy documents, as these relate to policy decisions and will be informed not only by scientific evidence but also social, economic and political factors.

Methods:

We conducted a systematic search of the peer-reviewed scientific literature, limiting findings to journal papers and book chapters that had been published in the English language from 2009 to January 2024 (when the search was carried out). A further 13 papers were added to the database over the course of the reviewing process, which we identified through email journal alerts, or were cited by papers that were reviewed. The titles and abstracts of the items of research were then screened for relevance, and removed from the inventory if they were deemed to be outside of scope or were duplicates. The resulting 565 items were reviewed in full, including tabulating the focus of the study, methods, geographic region, and the study species, design and duration.

Results:

The literature identified through the systematic search protocol was highly biased towards specific geographic regions and associated species. Over a third (35%) of studies were based on data collected in the USA, and the top ten species that were most frequently studied (four raptors and six microbats) made up

29% of the literature. Several case study areas repeatedly appeared in the literature – namely the Altamont Pass Wind Resource Area in the USA, the Strait of Gibraltar/southern Spain, and Smøla vindpark in Norway – and were the source of much of the available evidence. The former two of these cover large areas located in the migratory paths of many species, and are where a very large number of fatalities have been recorded. These biases need to be considered when interpreting the findings discussed below, particularly within the Victorian context. The 565 papers yielded through the search were categorised according to whether they focussed on impacts (320 papers), pre-construction assessments (252 papers), mitigation measures (119 papers) or post-construction monitoring (90 papers, noting some papers fell under several themes). These themes were then used to structure our review.

The impacts of wind energy on birds and bats

Based on the reviewed material, it is clear that the most immediate and direct impact of wind energy on bird and bat species are fatalities from collisions with turbines, but these are difficult to quantify. Published fatality rate estimates collected in other parts of the world range from 0.4–13 birds and 3–44 bats/megawatt/year, and <1–21 birds and 4–70 bats/turbine/year (noting that capacity or wattage varies substantially between turbine models). As of September 2024, there were 1,703 turbines that were operational or under construction in Victoria, with a collective capacity of 5,075 MW.

Collision fatalities can affect not only local populations in the short-term, but also have wider-reaching impacts, and it is unclear how to quantify and account for cumulative impacts on populations. Evidence from international studies using biomarkers indicates that for some species, a substantial proportion of individuals killed at turbines are not from the local population, but may have travelled tens or even hundreds of kilometres. As new immigrants arrive to replace individuals that have been killed, they then themselves become exposed to an increased risk of mortality, creating a population 'sink' effect. Wind turbines also have a long temporal footprint, and can continue to remove individuals from a population as long as they are operational (i.e. potentially several decades). Evidence for these phenomena (cumulative impacts and population sinks) was found in some studies, but not others, emphasising the importance of the specifics of focal species and landscape context.

The construction and operation of turbines can also influence how species are distributed in space and use habitats through the effects of attraction, habitat displacement and avoidance. While the evidence is mixed for both bats and birds depending on species and region, it does seem to suggest that some microbats might be attracted to turbines (12/33 cases identified), while for birds, raptors in particular may be able to avoid them (18/28 cases).

Pre-construction planning and assessment

Much of the published literature that related to the pre-construction phase highlighted key steps and best practice in assessment and planning processes to minimise wildlife impacts. This included desktop studies that used strategic planning approaches and tools to identify priority areas for development and conservation, and to assess trade-offs among objectives related to energy production, reduced collisions, and logistical constraints. Others developed processes to determine, at the regional or national level, what species might be most at risk from the impacts of turbine collisions. At the individual facility scale, the number, design and micro-siting of turbines can influence the magnitude of biodiversity impacts.

The evidence we reviewed indicated that larger turbines were associated with higher mortalities, but once the amount of energy generated is corrected for, mortality rates appeared to be relatively consistent per unit of energy generated. Whether this pattern will hold true for newer-generation, very large turbines (>6 MW) is unclear. However, it is likely that larger turbines will have a greater impact on some species (i.e. those that typically fly higher above the ground) and less on others, depending on flight behaviour. There were very few consistent patterns regarding the relationship between turbine siting and fatalities. However, consistent recommendations included avoiding critical breeding habitats, and ridges and areas of orographic uplift because they are frequently used by soaring raptors.

Acoustic surveys are frequently conducted for microbats during the pre-construction phase to investigate relative activity. However, there are persistent concerns about how effective these are in detecting species depending on environmental conditions and call characteristics, and whether deploying acoustic recorders at greater heights (i.e. closer to where the turbine blades are) can help reduce these detection issues. Bird utilisation surveys are typically used to determine occupancy, quantify flight activity behaviours, and to identify important habitat features and times of risk. However, for both birds and bats the type, timing and frequency of surveys can strongly influence conclusions, and synthesis studies to date indicate that pre-construction assessments perform very poorly at predicting post-construction mortality at individual facilities. Increasingly, radar and telemetry technologies, such as GPS tracking, are being used to assess species movements and inform wind energy risk analyses. Substantial post-processing is needed to reliably interpret the data from these approaches because of measurement errors.

Data collected from field observations, telemetry, and synthesised from elsewhere can be used to build Collision Risk Models (CRMs) and Population Viability Analyses (PVAs) that aim to predict how populations of species at risk will be impacted by wind farm collisions. However, empirical data relevant to both the species and location of interest are rarely available to parameterise these complex models. This means that predictions should be interpreted with caution, the underlying assumptions and uncertainty carefully considered, and the collection of fit-for-purpose data prioritised whenever possible.

Mitigation options and effectiveness

Several mitigation measures have been assessed for effectiveness in reducing bat or bird mortality rates. The most well-studied mitigation, which was also found to be consistently effective in reducing fatalities, was night-time low windspeed curtailment for microbats. This involves increasing night-time ‘cut-in’ wind speed (the speed at which turbines become operational) to avoid low wind speed periods when bats are most active and relatively little energy is generated. Every experimental curtailment study identified in our review found that there was a statistically significant reduction in bat mortality relative to a control where curtailment was not in place.

Acoustic deterrents have also been trialled for microbats with mixed success, indicating effectiveness for some species and in some years, while not in others. Potential limitations of acoustic deterrents highlighted in the literature include the limited distances over which sounds can be broadcast (i.e. limiting coverage of the rotor swept area), and concern that species may be attracted by the deterrent sounds or become acclimatised to them over time. With regards to visual deterrents, marking one turbine blade black was found to be effective in significantly reducing raptor mortality at a single wind farm in a single study in Norway, but no other published studies are available to indicate whether this finding will hold true in other contexts.

Proprietary systems for triggering real-time turbine shutdowns for birds, using automated identification of species from images, have also been experimentally tested at a small number of wind energy facilities. While these trials have indicated that these systems can be effective in reducing raptor mortalities, there are persistent issues with ‘false positives’ that trigger shutdowns for non-target species. While these issues remain, the costs and benefits of these mitigation systems should be carefully weighed up with respect to the ecological setting and the target species.

Finally, ‘buffers’ (minimum distance requirements between turbines and habitat features) are sometimes prescribed as avoidance or mitigation measures for both birds and bats. While studies have used observational, acoustic, and telemetry approaches to make recommendations about what buffer distances should be, none have assessed the relationship between the implementation of buffers and subsequent reduced mortality, which makes it difficult to assess their effectiveness.

Post-construction monitoring

The literature we reviewed demonstrated that fatality estimates from post-construction monitoring (i.e. inference about the number of birds and bats killed at a facility over a given time frame, incorporating bias corrections) can be strongly influenced by survey effort, or the frequency, duration, and area of searches. Studies indicated that having shorter intervals (e.g. weekly) between searches substantially increased mortality estimates, and that carcasses of new species continued to be detected in searches several years after construction.

Trained detection dogs consistently detected significantly more carcasses (on average 69–96% of carcasses placed in trials) than human searchers (who on average detected 9–65%), and performed better particularly when carcasses were small or were in lower-visibility environments. Ballistic models were often used to inform the size of the area searched underneath each turbine, and to estimate how many carcasses were likely to have fallen outside of this area, and would have been missed. However, the underlying assumptions of these models (e.g. assuming no turbulence despite the high-wind environments) are not always realistic.

How long a carcass persists on the ground (and is therefore available to be detected) once a collision has occurred varies greatly between taxa. For example, carcasses of raptors often persist in the landscape for hundreds of days, whereas carcases of small bats and birds often only remain for 2–3 days before being scavenged. Rare and threatened species may only collide with turbines infrequently, so are also more likely to go undetected in mortality surveys. Yet, when few individuals of these species remain, any fatalities will have a disproportionately large impact on the population as a whole.

Conclusions:

This systematic review of the scientific literature has indicated that turbine collisions have the potential to cause substantial, ongoing, and far-reaching impacts on bird and bat populations. Much of what is known about these impacts, as well as associated assessments, mitigations and monitoring, is based on a subset of very well-studied species in temperate systems. There are parallels and similarities between these and

Victoria, giving some certainty that strong and consistent results found elsewhere will hold here. Nonetheless, there is still a great need for more species- and context-specific knowledge for Victoria, and we identify 52 knowledge gaps (with some relating to even the most basic ecological information) that need to be addressed before some of the approaches and assumptions from the international literature can be applied here with any confidence. In particular, some unique Victorian species at risk have no comparable well-studied 'surrogates' in the available wind energy literature, such a flying-foxes and small migratory parrots.

Bats are disproportionately impacted at wind energy facilities, often experiencing higher fatality rates than birds, and microbats can be readily missed in carcass searches and are quickly scavenged. There is evidence that some microbat species can be attracted to turbines, and pre-construction assessments based on acoustic surveys poorly predict post-construction fatality risks, underlining the importance of rigorous mortality monitoring to determine impacts and inform mitigations. Night-time low windspeed curtailment is a highly effective and well-studied mitigation option for microbats; however, little is known about how to assess and mitigate risks to flying-foxes.

Some bird groups of interest in Victoria, such as passerines and parrots, are poorly represented in the wind energy literature. Nonetheless, it appears that raptors are the bird group most impacted by collisions. These species seem to have the potential to learn to avoid turbines, are less likely to be missed in mortality searches, and some potentially promising mitigations have been trialled elsewhere for this group (e.g. automated detection systems, increased blade visibility) but require further experimental testing in Victoria. For both birds and bats, an issue for rare and threatened species is that 'absence of evidence' (i.e. failing to detect a species) in mortality searches should not be interpreted as 'evidence of absence' (i.e. assuming no collisions have occurred).

Finally, as noted by other authors, the identification of trends in the wind energy literature would be greatly facilitated by increased transparency (e.g. improved information about turbine operation periods during mitigation trials), and use of consistent mortality survey techniques and reporting of results. As it is difficult to predict impacts from pre-constructions assessments alone, it may be prudent to adopt an adaptive approach to regulation, monitoring and mitigation, that can accommodate and account for improved information as data continue to be collected and published.

1 Introduction and aims

Victoria has ambitious legislated energy targets, aiming to produce 95% of the state's energy through renewable sources by 2035. Wind energy will be key in meeting these targets; however, it produced just 21% of Victoria's electricity in 2023 (DEECA 2025). As a result, there will be an accelerated development of wind energy in the coming decade, with more facilities required to meet these targets. As potential offshore wind energy is still in the early stages of development, there will be a requirement to continue increasing Victoria's onshore wind energy capacity.

While it is well-known that renewable energy sources are critical for reducing greenhouse gas emissions, there are also direct and indirect impacts on biodiversity through associated resource extraction for construction, land clearing to accommodate large-scale infrastructure, the installation of transmission lines, and disturbance, displacement, and collisions to wildlife caused by wind turbines (Bennun et al. 2021). Some of these impacts, such as habitat loss, are common to other types of development and land use change more generally, and are already addressed by well-established policies and practices and an associated supporting body of research. However, other impacts, and specifically collision fatalities, are more unique to wind energy so require focussed consideration.

There are already processes in place that seek to address the risk of collisions on bats and birds in Victoria for onshore wind energy, including pre-construction impact assessments under state and national legislation, permit requirements to develop site-specific Bat and Avifauna (bird) Management Plans (BAM Plans), and monitoring post-construction impacts through mortality surveys. Initial steps have been taken to analyse some of the post-construction mortality data (see Section 4.1) and to determine 'Species of Concern' that are at highest population-level risk of being impacted by collisions. However, the increased momentum of development, concerns about managing the impacts on biodiversity (including cumulative impacts), and calls from the energy sector for guidance documents to improve clarity and consistency, all require a robust evidence base for decision making.

Onshore wind energy has been used in parts of the world for many decades, and some of the questions most pertinent to Victoria in relation to wind energy and biodiversity have already been well-studied in different contexts. Moving forward, the guidelines, policies and practices developed in Victoria can draw on this large body of evidence, acknowledging that there may be important differences in the Victorian context that we need to account for. While there are many existing guidelines, handbooks, and protocols from elsewhere in the world, it is sometimes difficult to determine what evidence underpins the various recommendations contained within these.

Therefore, the objective of this study was to systematically review the primary wind energy research undertaken globally, to provide a comprehensive but accessible summary of both what is known and the key knowledge gaps as they apply to Victoria, and to ensure that it is clear which findings relate to which evidence.

Specifically, the aims were to:

- systematically review the available peer-reviewed scientific literature on wildlife collisions at onshore wind energy facilities
- provide a transparent summary of evidence and knowledge gaps associated with the impacts, assessment of risk, survey protocols, mitigation measures, and monitoring
- present this in the context of wind energy developments in Victoria, and consider how applicable this evidence is to our landscapes and species.

To ensure that the review remained tractable, the scope was limited to evidence as it related to onshore, as opposed to offshore wind energy developments, and impacts on volant (flying) vertebrate species, i.e. birds and bats. While we also reviewed evidence related to impacts such as displacement, avoidance and attraction, our primary focus was on direct collisions, given their immediate nature and the consequential impacts on wildlife populations, which is a priority concern internationally (Green et al. 2022). Because the aim here was to provide a summary of the evidence base from the scientific literature, we did not specifically review guideline documents from elsewhere in the world that relate to how surveys are conducted and what mitigations are required. These are policy decisions, and will be informed not only by scientific evidence but also social, economic and political factors. However, we have provided below some examples of international policy and guideline documents.

Key guideline or policy documents

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2 Methods

We conducted a systematic search of the peer-reviewed scientific literature, limiting our findings to journal papers and book chapters that had been published in the English language. We acknowledge that there are papers published in other languages on this topic, and that by only reviewing material written in English we have likely introduced some geographic bias.

For the first step, we used the advanced search function in two scientific databases, Web of Science (https://www.webofscience.com/wos/woscc/advanced-search) and Scopus (https://www.scopus.com/search/form.uri?display=advanced), by applying the search strings shown in Table S1. These search strings were intended to identify those pieces of evidence that were most relevant to the scope and aims of our study, and our search was limited to the title, abstract and keywords. We also limited our search of these databases from 1 January 2010 to 31 January 2024, when the search was carried out. Nonetheless, some items from 2009 also appeared in our results (perhaps because of the way the papers had been indexed) and we chose to retain these.

We also searched the Tethys Knowledge Base (https://tethys.pnnl.gov/knowledge-base), which is an international database specifically intended to act as a central point for information related to wind and marine energy technology. We searched the Wind Energy Content page, limiting the results to journal articles published where the 'receptor' was birds or bats, the 'technology' was land-based wind, and the 'stressor' was collisions, avoidance, displacement or attraction. For this search, we set the minimum year as 2009 because Web of Science and Scopus had returned 2009 search results unintentionally, and we wanted this search to be comparable. Also, we set the maximum year as 2023, because we could not limit our results to just January 2024.

Collectively, these searches yielded 2,270 results (Figure 1), which were loaded into the Covidence (https://www.covidence.org/) systematic review platform. A further 13 papers were added to the database over the course of the reviewing process, which we identified through email journal alerts, or reference lists within the other papers we were reading. Using the Covidence platform, we identified 993 duplicates, leaving 1,290 unique pieces of literature.

The titles and abstracts of the 1,290 items of research were then screened for relevance, and removed from the pool if they were deemed to be outside of scope. Some of the recurring reasons for this were that the research focussed on: offshore wind energy; species that were ground-dwelling; small turbines installed on homes or businesses, as opposed to full-sized commercial wind turbines; turbine design to increase efficiency; social acceptability and values; other aspects of biology or ecology with only a passing reference to wind energy; on environmental law; and the impacts of wildlife collisions on turbine blades (as opposed to impacts in the other direction). We removed 725 items through this initial screening process.

For the remaining 565 items, we then did a full review, which involved reading the title, abstract, and methods, and for those papers that we determined were highly relevant and of interest, we also read the introduction, results and discussion. During this process, we extracted information relating to the topical focus of the study, the methods used, the geographic region from where data were collected or collated, the study species (for which we extracted the individual common and scientific names if there were six or fewer), the duration, information about study design and contrasts (e.g. before-after-control-impact [BACI], or just control-impact), and which mitigations were tested (if any).

We have structured our review to cover the key topics and themes that emerged during this process. In each section, we highlight key references and studies that we identified as being the most robust and relevant to our aims, highlighting in particular existing reviews and meta-analyses. In some places we distinguish between data syntheses and meta-analyses. For our purposes, 'data syntheses' typically involve data (that have been collected in a standardised way) being acquired from a central source such as a government authority, and then summarised. The term 'meta-analysis' is used where the authors have sought to analyse data from multiple sources in a very specific way and applied strict inclusion criteria, and have sourced either point estimates and uncertainty, or the original raw data, to then re-analyse themselves.

For each of the topics covered in our review, we provide an indicator of the number of related papers.

Very large number	Large number	Moderate number	Small number	Very small number
>100 papers	51–100 papers	21–50 papers	6–20 papers	1–5 papers

In cases where there are at least six papers (i.e. at least 'small number'), we also indicate the extent of bias to a specific taxon:

High bat bias	Low bat bias	No strong bias	Low bird bias	High bird bias
>90% papers about bats	>70% papers about bats	50–70% papers about bats or birds	>70% papers about birds	>90% papers about birds

We also indicate the extent of bias based on country or region:

Very low geographic bias	Low geographic bias	Moderate geographic bias	High geographic bias	Very high geographic bias
0–30% papers focus on any one country or region	31–50% papers focus on any one country or region	51–70% papers focus on any one country or region	71–90% papers focus on any one country or region	>90% papers focus on any one country or region

For each topic, we provide a short summary of what is known, what the key knowledge and research gaps are, and what this means for Victoria.

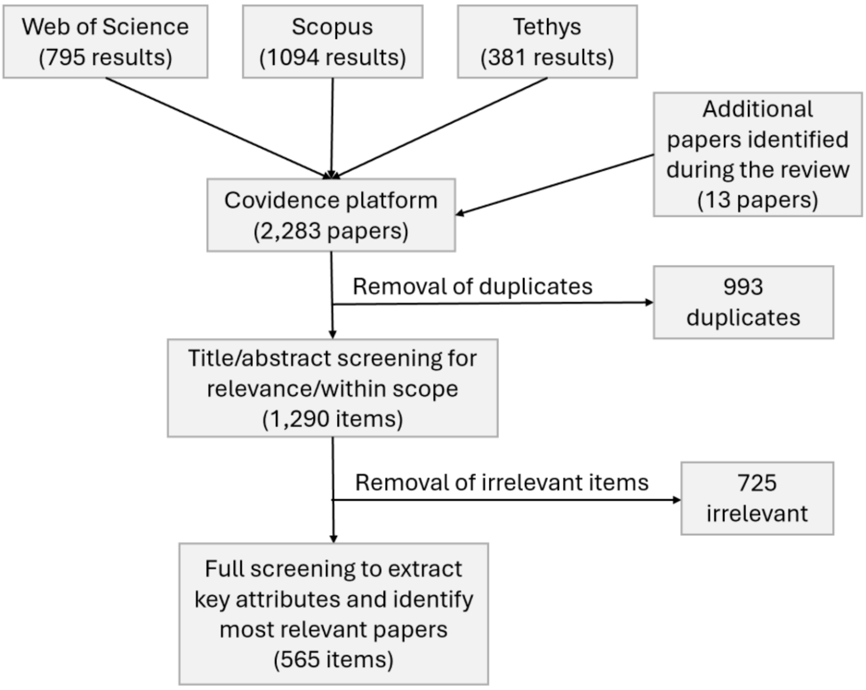

Figure 1. Systematic literature review protocol followed, illustrating the number of research items (predominantly scientific journal articles) that were reviewed during each stage.

References have been listed in three ways:

- At the end of each section, the most informative references relating to that section are listed ('key references').
- In some cases, 'additional references' are provided at the end of sections where they have been referred to in the text, but were not included in the systematic search (e.g. selected unpublished reports, published papers pre-2009, papers not specifically on wind energy facilities). These 'additional references' are not included in the reference list at the end of the document.
- The 565 research papers identified during the systematic search of the literature are all listed at the end of this document.

3 Patterns in the literature and applicability to the Victorian context

3.1 Geographic and taxonomic bias

All literature reviews should be interpreted in the context of biases in the evidence base, and in the case of the peer-reviewed literature related to wind energy, these biases were sometimes very pronounced.

The most apparent of these was the number of studies (196, or 35%) that focused on the United States of America (USA, Figure 2). Other countries that were the focus of a large amount of research were Germany (48 studies), Canada (47 studies), Spain (39 studies), and the United Kingdom (29). This reflects a broad trend commonly identified in literature reviews for any topic in the biological sciences, that research is biased towards North America and Europe, i.e. western nations with larger economies.

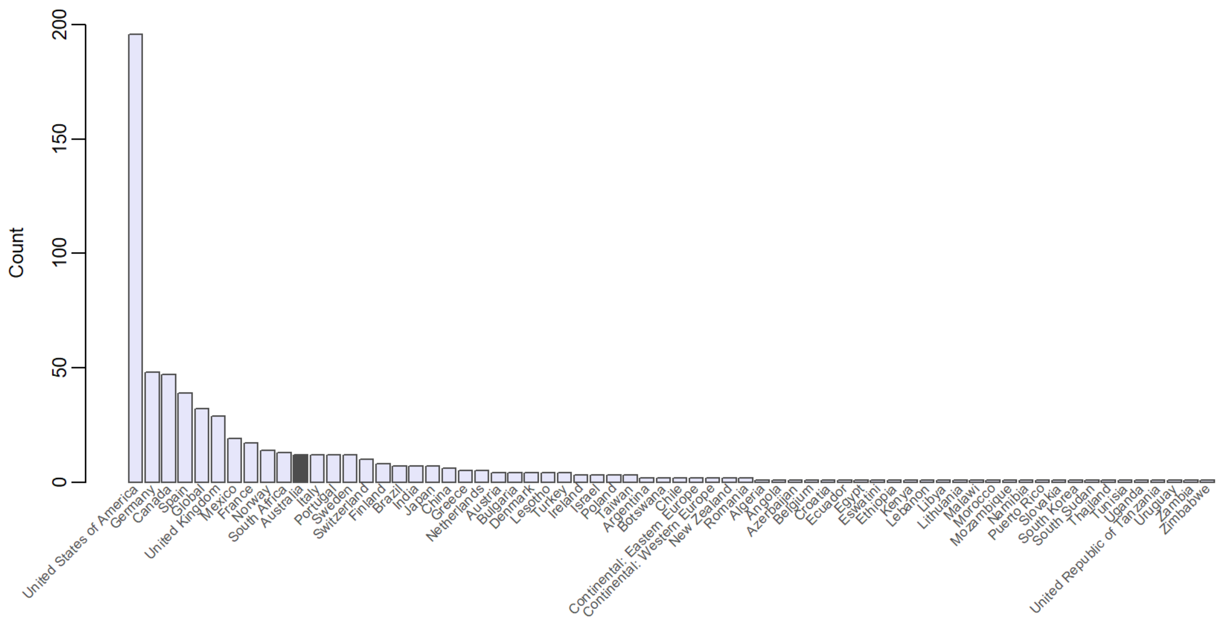

Figure 2. Number of studies focussed on different countries and regions, with Australia highlighted in dark grey.

This bias in the countries that produced the most peer-reviewed research was also reflected in the species that were focussed on (Table 1; Figure 3; and Table S2). In general, there were more studies focussed on birds (391 papers) than bats (234 papers), and a subset of these studies looked at both taxonomic groups (61 papers). Twenty-nine percent (29%) of the literature was focused on the top 10 most-studied species, comprising four raptor species (three eagles and one vulture) and six microbats. Note that throughout this report we do not provide the scientific names of species from studies included in the systematic search, as these are listed in Table S2. However, we do provide the scientific names of additional Victorian species upon first mention.

Part of the reason for the taxonomic bias was that there are regions or specific facilities that have been the focus of much research to date, and case studies that repeatedly appeared included the Altamont Pass Wind Resource Area in California (aka the APWRA, USA), the Smøla vindpark in Norway, and onshore facilities either side of the Strait of Gibraltar, which connects northern Morocco and southern Spain (Figure 3). These are all areas for which there was concern that facilities were having a pronounced impact on target species. For the APWRA, it's migratory tree-roosting bats (especially Hoary Bats), and Bald and Golden Eagles, for Smøla it's White-tailed Eagles, and for the Strait of Gibraltar it's Griffon Vultures and Black Kites, because it forms an important migratory passage.

Table 1. Individual species that were most frequently the focus of studies in the international wind energy literature (i.e. which were the focus of five or more studies).

'Count' indicates the number of papers that focussed on that species. Note that 251 papers were classified as 'Multiple' species, as was often the case where authors were reporting on the findings from carcass searches. A full list of studied species (including their scientific names) is provided in Table S2.

Common name	Scientific name	Guild	Study countries	Count
Hoary Bat	*Lasiurus cinereus*	Microbat	Brazil, Canada, Mexico, USA	40
Golden Eagle	*Aquila chrysaetos*	Raptor	Canada, Finland, France, Norway, Spain, Sweden, UK, USA	40
Silver-haired Bat	*Lasionycteris noctivagans*	Microbat	Canada, Mexico, USA	28
Eastern Red Bat	*Lasiurus borealis*	Microbat	Canada, Mexico, USA	26
White-tailed Eagle	*Haliaeetus albicilla*	Raptor	Finland, Germany, Netherlands, Norway, Sweden, Switzerland	19
Griffon Vulture	*Gyps fulvus*	Raptor	France, Germany, Italy, Portugal, Spain, Switzerland, UK	16
Bald Eagle	*Haliaeetus leucocephalus*	Raptor	Canada, USA	14
Common Pipistrelle	*Pipistrellus pipistrellus*	Microbat	France, Germany, Italy, Portugal, UK	12
Little Brown Bat	*Myotis lucifugus*	Microbat	Canada, USA	12
Big Brown Bat	*Eptesicus fuscus*	Microbat	Canada, USA	11
Common Noctule	*Nyctalus noctule*	Microbat	Germany, Sweden, Italy, UK	9
Raptors	(Multiple species)	Raptor	Global, India, Spain, USA	9
Red Kite	*Milvus milvus*	Raptor	Germany, Sweden, UK	9
Leisler's Bat	*Nyctalus leisleri*	Microbat	Germany, Italy, Portugal, UK	6
Nathusius' Pipistrelle	*Pipistrellus nathusii*	Microbat	France, Germany, UK	5
Red-tailed Hawk	*Buteo jamaicensis*	Raptor	Canada, USA	5
Skylark	*Alauda arvensis*	Passerine	France, Portugal	5

3.2 Relevance to Victorian species and ecosystems

Given the biases outlined above, it is important to consider the extent to which findings based on international species and systems are relevant, and can be applied or extrapolated to the Victorian context.

Victoria spans latitudes ranging from approximately -34°S to -39°S, and incorporates ecosystems including coastal and alpine areas, rainforests and sclerophyll forests, temperate woodlands and grasslands, and arid mallee, amongst others. These latitudes and systems match reasonably well with the areas that have been the focus of the much of the research globally (Figure 3), which have generally been in temperate zones with Mediterranean climates. Of course, given the suite of species that they consider, and differences in the types of facilities and associated turbine models that have been studied (which in older studies are typically smaller models with lower capacity than those operating in Victoria), one should exercise caution when making any comparisons. However, it is not unreasonable to expect that the general trends observed in these areas should also apply to Victoria.

In Victoria, the species that appear to be the most heavily impacted by collisions include tree- and cave-roosting microbats, flying-foxes, small and large raptors, and small and medium-sized passerines (Moloney et al. 2019). There are some key general differences between these species and those studies internationally that are worth noting here, particularly with regards to bats. Many microbat species in the USA and Europe use both hibernation and migration as strategies to survive periods when temperatures fall well below what Victorian bats experience. This means that there are specific time periods when they will be most active in an area and most vulnerable to collisions, i.e. when they are passing through during migration and/or outside of the winter hibernation period. Therefore, mitigation strategies such as curtailment can be highly targeted to those vulnerable periods. While Victorian microbats are most active during the warmer months, and they will become torpid (inactive) for several days at a time during winter, they still feed intermittently even during the coldest months. In addition, it has been suggested for some time that the White-striped Free-tailed Bat (*Austronomus australis*), in Australia's eastern states might undergo seasonal migratory movements, but to date no empirical evidence has conclusively demonstrated this. Southern Bent-wing Bats (*Miniopterus orianae bassanii*) also undertake both seasonal movements at particular times of year and frequently move between roosting caves throughout the year (van Harten et al. 2022). No other microbat species in Victoria is suspected of seasonal movements. Nevertheless, analysis of mortality data collected from Victorian wind farms does suggest that there is a heightened risk period over the late austral summer-autumn when a high proportion of microbat collisions occur (Moloney et al. 2019, Symbolix Pty Ltd 2020). Therefore, there are parallels with the fatality pattern seen in areas such as North America, Europe, and South Africa, where fatalities tend to peak during the autumn and spring periods (Aronson 2022, Llyod et al. 2023).

Victoria is also home to two species of flying-fox (the Grey-headed Flying-fox, *Pteropus poliocephalus*, and the Little Red Flying-fox, *Pteropus scapulatus*), both of which have been recorded colliding with turbines. Flying-foxes (genus *Pteropus*) are very different to microbats in that they feed primarily on pollen, nectar and fruits, they do not echolocate, they are much larger, and in Australia they are known to be highly mobile, i.e. nomadic or migratory. GPS tracking studies have shown that individual Little Red and Grey-headed Flying-foxes travel thousands of kilometres annually (>2,500 km and >6,000 km, respectively), tracking the availability of food resources (Welbergen et al. 2020). Flying-foxes occur throughout South Asia, Southeast Asia, Australia, East Africa, and some oceanic islands in the Indian and Pacific Oceans; areas that were generally poorly represented in the wind energy literature. We identified only one passing reference to flying-foxes in the reviewed papers (Pande et al. 2013), which mentioned a flying-fox in India that was electrocuted by transmission lines at a wind energy facility. Consequently, the available international literature provides little-to-no guidance on how to assess potential risks, or what effective mitigation measures may be, with regards to flying-foxes.

The relevance and applicability of the bird literature to the Victorian context may be more varied than for bats. For example, internationally there is a broad consensus on the risks that turbines pose to raptors (e.g. Estellés-Domingo and López-López 2024), which aligns with patterns of mortality observed in Victoria. However, some bird groups that are likely to be at risk in Victoria and across Australia more broadly (Lumsden et al. 2019, Reid et al. 2023) have no comparable well-studied 'surrogates' internationally. Perhaps the most notable of these groups is the small migratory parrots, which includes the Orange-bellied Parrot (*Neophema chrysogaster*), Swift Parrot (*Lathamus discolor*), and Blue-winged Parrot (*Neophema chrysostoma*). All three of these species are listed as nationally threatened under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act) and undertake regular seasonal migrations across the Bass Strait between Tasmania and the Australian mainland (Menkhorst et al. 2017), putting them at risk from collisions with both onshore and offshore turbines (Reid et al. 2023). Orange-bellied Parrots also frequently migrate at night (DNRET 2024), meaning they may be less able to detect turbines from visual cues. Given their small body size (ranging from 41–75 g), these three species are less likely to be detected in post-construction fatality monitoring (see Section 7), though Blue-winged Parrot carcasses have been recorded underneath turbines in Tasmania and Victoria (Hull et al. 2013, Symbolix Pty Ltd 2020).

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Symbolix Pty Ltd. (2020) *Post construction bird and bat monitoring at wind farms in Victoria, Public report v1.0*. Melbourne VIC.

Welbergen, JA, Meade, J, Field, HE, Edson, D, McMichael, L, Shoo, LP, Praszczalek, J, Smith, C, Martin, JM, (2020) Extreme mobility of the world's largest flying mammals creates key challenges for management and conservation. *BMC Biology* **18**, 1-13.

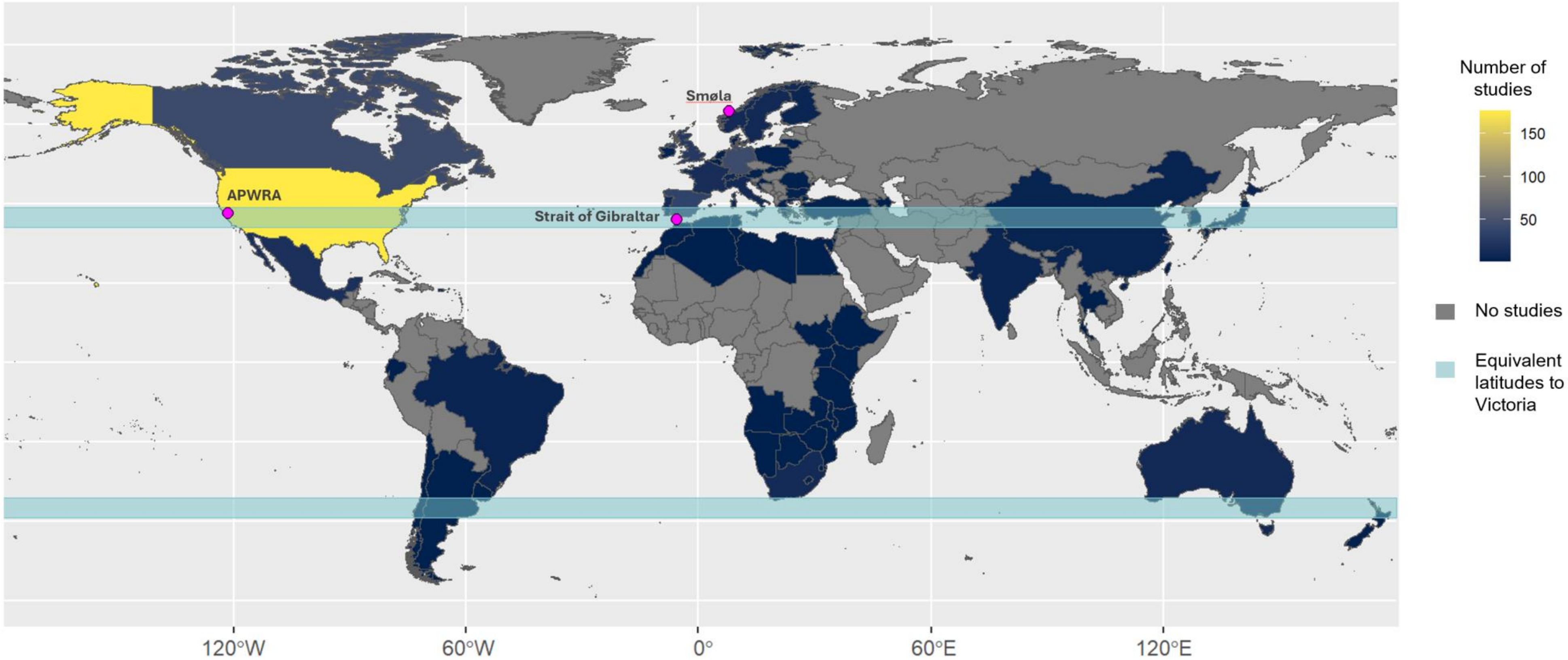

Figure 3. Map showing the number of studies published in English, focussed on each country (global studies have been excluded from this map), highlighting the latitudes equivalent to those that span Victoria (-34° to -39° in the south, and 34° to 39 in the north) in pale blue, and the locations of specific facilities or locations that have been the focus of much of the research to date as pink dots (APWRA: Altamont Pass Wind Resource Area, Smøla: Smøla vindpark).

4 The impacts of wind energy on birds and bats

4.1 Direct mortality associated with collisions with turbines

Very large number of papers (226)	No taxonomic bias	Low geographic bias (USA 37%)

One of the most pronounced impacts of wind energy operation on wildlife, and the primary focus of this review, occurs when birds and bats collide with turbine blades in-flight and are immediately killed, or mortally injured. Birds and flying-foxes have also been recorded colliding with transmission lines (Pande et al. 2013; Winder et al. 2014; Lin 2017) and turbine towers (Zeiler and Grünschachner-Berger 2009; Stokke et al. 2020), even when the blades are stationary. While barotrauma (damage to the lungs caused by changes in pressure) was once considered a likely additional cause of mortality for bats at wind energy facilities, various ecological, veterinary and simulation studies have concluded that if/when this occurs, it is of secondary importance to impact trauma associated with physical collisions (Capparella et al. 2012; Rollins et al. 2012; Lawson et al. 2020). Irrespective of how they are killed, in all cases it is considered to be due to the turbines, and so the precise cause of death is immaterial.

For context, fatalities estimates are typically presented as rates, either the number of individuals killed per turbine per year, or the number of individuals killed per megawatt (MW) per year. Individual turbines can vary greatly in their rated capacity or wattage (MW), with older models rated at <0.1 MW (see Section 5.4.1), and new models exceeding 6 MW. Hence, fatality rates/MW are a more standardised (and perhaps reliable) way of presenting these data, while rates/turbine are unlikely to be comparable between studies.

Estimates of fatalities/MW/year, or fatalities/turbine/year, have been published globally (Table 2). For birds, these estimates range from less than one, to 21 fatalities/turbine/year (North America and Belgium respectively), and 0.4–13 fatalities/MW/years (based on data from Japan and Mexico respectively). For bats, these figures are 4–70 fatalities/turbine/year (Mexico and Germany, respectively), and 3–44 fatalities/MW/year (South Africa and Mexico, respectively). While these values may seem extreme, they are broadly within the range of the best available fatality estimates for Victoria (Moloney et al. 2019, Symbolix Pty Ltd. 2020, published in the grey literature but included in Table 2 as a basis for comparison), although these Victorian estimates appear to be at the lower end of the scale. The exception to this is the German study (Voigt et al. 2022), which estimated that 70 bats were killed/turbine/year – a figure that appears to be substantially higher than estimates from elsewhere.

As of September 2024, there were 1,703 turbines that were operational or under construction in Victoria, with a collective capacity of 5,075 MW (Victorian Department of Energy, Environment and Climate Action data). If we take the ranges of published international fatality rates from Table 2 (specifically the individuals/MW/year estimates) and extrapolate these to Victoria's wind energy capacity, that translates to 2,030–65,982 birds and 15,227–223,322 bats per annum. If we do the same extrapolation based on the range of estimates (which are individuals/turbine/year) from the Victorian reports (i.e. Moloney et al. 2019, based on two wind energy facilities, and Symbolix Pty Ltd. 2020, based on 10 wind energy facilities) that translates to 2,384–38,999 birds and 2,725–41,553 bats per annum, based on turbine-level estimates. Extrapolations like these should be considered broadly indicative only, and there is clearly a large amount of variation and uncertainty in these values. However, they do indicate that somewhere in the order of several thousand to tens of thousands of birds, and several thousand to tens of thousands of bats, are likely killed as a result of collisions with wind turbines in Victoria each year. These figures will increase as the number of turbines increases, as those currently under construction become operational and those going through the approval process are commissioned.

Data syntheses and meta-analyses from North America have also attempted to estimate total fatalities at the country or continental scale (Table 2), made possible because post-construction surveys and systematic reporting have been carried out for many years. Three major published studies have attempted to estimate how many birds were killed in the USA annually due to collisions with turbines, based on 2012 turbine numbers (Loss et al. 2013; Smallwood 2013; Erickson et al. 2014). Johnson et al. (2016) provided a comparison of these three studies, discussing differences in scope, data that were included, analytical assumptions and potential biases. Nonetheless, despite the inherent differences they note that all three studies come to a similar conclusion; that approximately 250,000–500,000 birds were killed by wind farms in the USA annually, at a time when the national capacity was approximately 51,000 megawatts (MW). Another data synthesis conducted around this time, focussed on Canada (Zimmerling et al. 2013) estimated that 23,000 birds would be killed each year by wind turbines, when national capacity was around 5,000 MW.

Estimated total fatalities for bats are even greater. Hayes (2013) conducted a data synthesis to estimate how many bats are killed per year by wind turbines in the USA, and estimated the figure to be 600,000. Zimmerling and Francis (2016) also used a data synthesis approach to estimated fatalities of bats across Canada, and derived an annual estimate of 47,700, predicting that that figure would increase to 166,000 over the following 15 years based on planned increases in capacity. While comparable data syntheses are not available for Europe, Voigt et al. (2015) do state that "*Presumably, more than 250,000 bats are killed annually due to interactions with German wind turbines, and total losses may account for more than two million killed bats over the past 10 years, if mitigation measures were not practiced.*" In a systematic review, O'Shea et al. (2016) identified that since 2000, collisions with wind turbines have been the single greatest cause of bat multiple mortality events (i.e. where more than 10 bats are killed at a single location within a year) globally. They tabulated 281 collision-related multiple mortality events for 41 species.

Key references

Cabrera-Cruz, SA, Cervantes-Pasqualli, J, Franquesa-Soler, M, Muñoz-Jiménez, O, Rodríguez-Aguilar, G, Villegas-Patraca, R (2020) Estimates of aerial vertebrate mortality at wind farms in a bird migration corridor and bat diversity hotspot. *Global Ecology and Conservation* **22**, e00966.

Johnson, DH, Loss, SR, Smallwood, KS, Erickson, WP (2016) Avian fatalities at wind energy facilities in North America: a comparison of recent approaches. *Human-Wildlife Interactions* **10**, 7-18.

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Symbolix Pty Ltd. (2020) *Post construction bird and bat monitoring at wind farms in Victoria, Public report v1.0*. Melbourne VIC.

Table 2. Fatalities: Estimates of the number of individual birds and bats killed due to collisions annually, either per megawatt (MW), per turbine, or at a national scale, rounded to the nearest whole number.

We only present studies where estimates have been corrected to account for searcher efficiency, search area and carcass persistence. Reports marked with * are not from the peer-reviewed published literature and were sourced directly rather than via our database search, but have been included to provide a comparison of the best available estimates from Victoria with the rest of the world. Ranges are shown for studies where fatalities were estimated between years (indicated by 'y') or facilities ('f', i.e. individual wind farms), and for different regions or biomes ('r') or guilds ('g'), or based on statistical confidence intervals ('CI').

Study	Taxon	Country or region	Fatalities/ MW/year	Fatalities/ turbine/ year	Fatalities/ year	Approx. national capacity (at time of study)
Aronson (2022)	Bats	South Africa	3			
Bolívar-Cimé et al. (2016)	Bats	Mexico		4–20 [y]		
Cabrera-Cruz et al. (2020)	Bats	Mexico	20–44 [f]			
Cabrera-Cruz et al. (2023)	Bats	Mexico	4–15 [f]			
Hayes (2013)	Bats	USA	13		680,000	51,000 MW
Mantoiu et al. (2020)	Bats	Romania	14.2	30		
Smallwood (2013)	Bats	USA			651,000 – 888,000 [f]	51,000 MW
Voigt et al. (2022a)	Bats	Germany	39	70		
Weaver et al. (2020b)	Bats	USA	16			
Zimmerling and Francis (2016)	Bats	Canada		16	47,000	5,000 MW
Moloney et al. (2019)*	Bats	Victoria, Australia		1.4–22.9 [f]		
Symbolix Pty Ltd. (2020)*	Bats	Victoria, Australia		7–10.8 [f]		
Arikan and Turan (2017)	Birds	Turkey		2		
Bull et al. (2013)	Birds	New Zealand		5–6 [y]		
Cabrera-Cruz et al. (2020)	Birds	Mexico	9–13 [f]			
Erickson et al. (2014)	Birds	USA and Canada	2–3 [r]		134,000 – 230,000 [r]	63,000 MW
Everaert (2014)	Birds	Belgium		21		
Kerlinger et al. (2010)	Birds	USA and Canada		<1–7 [f]		
Kitano and Shiraki (2013)	Birds	Japan	0.4–2 [g]			
Loss et al. (2013)	Birds	USA	4	5	140,000 – 328,000 [CI]	56,000 MW
Perold et al. (2020)	Birds	South Africa	2	5		
Smallwood (2013)	Birds	USA			531,000 – 573,000 [f]	51,000 MW
Zimmerling et al. (2013)	Birds	Canada		8	23,000	5,000 MW
Moloney et al. (2019)*	Birds	Victoria, Australia		1.6–24.4 [f]		
Symbolix Pty Ltd. (2020)*	Birds	Victoria, Australia		3.4–6.7 [f]		

4.1.1 Morphological and ecological traits of species frequently impacted by collisions

Identifying patterns in the types of species that most frequently collide with turbines internationally could help predict which species in less-studied regions may also be most at risk. However, it is difficult to determine whether the groups that most frequently feature in the literature are in fact the most vulnerable, or if the observed patterns are a product of the geographic biases highlighted above.

Bats

Studies from temperate areas of North America often emphasise the large number of fatalities of tree-roosting and migratory microbats such as Hoary Bats, Silver-haired Bats, and Eastern Red Bats (e.g. Cryan 2011; Zimmerling and Francis 2016; Davy et al. 2021). Studies focussed on southern USA (Texas) and Latin America also note that a high proportion of fatalities (>70% of all detected carcasses) were Mexican Free-tailed Bats, which is another migratory species (though roosts in caves, Barros et al. 2015; do Amaral et al. 2020; Weaver et al. 2020a). Likewise, some of the most impacted species in western Europe also appear to be migratory, such as the Common Noctule or Nathusius' Pipistrelle (e.g. Voigt et al. 2015). Based on these, it is easy to get the impression that migration is a key risk factor in collisions for bats, but this may be because some of the most well-studied areas align with major migration routes (e.g. Germany, Voigt et al. 2015, Altamont Pass Wind Resource Area, Smallwood and Bell 2020b), and these patterns do not necessarily hold elsewhere. In a review of studies to date that had been conducted in South Africa, Aronson (2022) found that resident species were in fact more prevalent in carcass searches than migratory species. This was also the case in Bolívar-Cimé et al. (2016) whose study focussed on Mexico.

Therefore, the potential lack of migratory movements by Victorian bats (discussed in Section 3.2 above) does not necessarily make them any less vulnerable to collision risks. Instead, factors such as the speed and height at which species fly are likely to play a more important role. In their review of wind energy impacts, mitigation measures and policy related to bats, Voigt et al. (2024) highlight open-space and edge-space foragers from the families Vespertilionidae, Molossidae, Mormoopidae and Emballonuridae, and fruit-eating Pteropodids (which includes flying-foxes) as being at highest risk of collisions. All these families (except for the Mormoopidae) occur in Victoria, and constitute the majority of the state's bat species. It is unclear why microbats fail to detect and avoid turbines, but in their laboratory study Long et al. (2010b) found that the physical properties of turbine blades make returned echolocation pulses difficult for bats to perceive and respond to. They suggest that turbines with a greater number of- or wider blades would be more perceptible to bats, and that species with longer echolocation pulses should also have an advantage.

Birds

Most birds are diurnal and so are able to use visual cues to a greater extent than microbats. Biases in the wind energy literature on birds are guild-based, in that many individual studies as well as reviews emphasise the vulnerability of raptors to collisions. This includes both very large raptors, such as the Griffon Vulture (mass ~6–10 kg, distributed across southern Eurasia and northern Africa), and smaller birds of prey, such as the American Kestrel (mass ~120 g, distributed across the Americas). This is because raptors typically soar in open areas when searching for prey. They also take advantage of orographic features such as ridges and areas with thermal updraughts (Arnett et al. 2016), which also provide wind profiles suitable for the production of wind energy. Both Estellés-Domingo and López-López (2024), and Watson et al. (2018) provide global reviews of the impacts of wind energy developments on raptors specifically, and discuss reasons why this group is so frequently recorded in carcass surveys.

Marques et al. (2014) provide a more general overview of the types of traits that make birds more or less vulnerable to collisions, and note that behaviour (e.g. flight type, foraging strategies) is the risk factor most commonly-cited in the literature. However, collision risk also appears to be associated with a range of other interrelating factors including landscape features, flight paths, avoidance behaviour, turbine features, phenology, abundance, wind farm configuration, sensorial perception, morphological features, food availability, weather, blade visibility and turbine lighting (Marques et al. 2014). In their global analysis of the relationship between species traits and collision risk, Thaxter et al. (2017) found that bird species that use anthropogenic habitats (i.e. farmland and urban areas) or grasslands appeared to experience greater collision rates. They also note that frugivores experienced lower collision rates compared to other feeding guilds, though this broadly corresponds to areas where turbines tend to be installed. In addition to raptors, studies from the Americas and South Africa indicate that passerines or 'perching' birds (Erickson et al. 2014; Perold et al. 2020; Agudelo et al. 2021) are known to frequently collide with turbines.

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4.2 Population-level, broadscale, and cumulative impacts

Moderate number of papers (34 papers)	No taxonomic bias	Low geographic bias (Europe 50%)

Wind energy facilities have the potential to have wide-reaching and long-term impacts, beyond just the annual turbine- or site-level collision fatalities detailed above. These can manifest in different ways.

First, because the species that we are focussing on here are volant, and in many cases highly mobile, there can be a constant or seasonal arrival of new individuals to a facility. While these new individuals can help compensate for any losses caused by collisions, if they too are killed then a facility may act as a 'population sink', and cause regional-level declines.

Impacts may also be 'cumulative' (or additive), that is, fatalities continue to accrue and through time as long as the turbines are operational (currently expected to be ~30 years in Victoria), and also expand spatially, as new wind energy facilities are added to the landscape. Wind energy impacts may also be cumulative in that they add to and interact with other threatened process already causing species declines; for example, the effects of wind collisions, plus a disease, plus habitat loss from urban development may be unsustainable for a species population. Sometimes the combined effects of multiple turbines, or multiple threats, cause more extreme population declines than would be expected from their individual impacts (i.e. are greater than the sum of their parts), and these are referred to as 'synergistic' effects.

Understanding the impacts of wind energy collisions alone, or in addition to other threatening processes at the entire population scale is difficult. Ideally, for each species being impacted, monitoring data would be collected over several years, and there would also be a good understanding of key population parameters (fecundity and survival rates), population structure, and immigration and emigration rates. Unfortunately, this information is lacking for most Victorian species, as is the case for many species internationally. However, researchers have taken advantage of long-term monitoring data, molecular tools and techniques, and demographic modelling approaches to attempt to address at least part of the question of what the regional-scale, cumulative, or synergistic impacts of wind energy collision fatalities may be.

4.2.1 Populations sinks: stable isotope analysis of carcass origin

One approach to understand the spatial scale of wind energy impacts, that has been adopted in North America and Europe is to salvage carcasses found under turbines, then analyse stable isotopes in fur or feather keratin to determine the individual's likely region or area of origin. The ratios of stable isotopes such as hydrogen, nitrogen, and carbon in the environment vary spatially with latitude, elevation, and climatic conditions (Baerwald et al. 2014). Consequently, the known ratios of isotopes can be mapped as an 'isoscape', and matched to keratin samples, which will reflect the area where an individual foraged and grew, and assimilated those environmental isotopes into their feathers or fur. Based on the results, individuals can then be classified as being of local or non-local origin. While it is difficult to specifically define what 'local' or 'non-local' are (and this is typically not done in the literature), individuals that are of non-local origin could be resident species experiencing a population sink phenomenon, or could be migratory.

We identified nine studies in our review that involved the analysis of stable isotopes, to determine the likely origin of individuals found underneath turbines during carcass searches (Table 3). Of these, seven studies from across Germany, Canada, Romania and the USA focussed on bats (six species). The proportion of sampled individuals that were predicted to be of non-local origin ranged from 1% (Hoary Bats in the Appalachian Mountains, USA, Pylant et al. 2016) to 94% (Common Noctule in Romania, Mantoiu et al. 2020). Non-local individuals salvaged from facilities in Germany were predicted to have travelled from Scandinavia, Poland, Ukraine, Baltic countries, and Belarus, while Common Noctules collected in Romania were predicted to have travelled from Belarus and Russia (i.e. hundreds to ~2,000 kilometres away). In North America, non-local bats were predicted to have travelled from areas further north, with Baerwald et al. (2014) noting that some may have travelled hundreds of kilometres.

A further two studies from the USA used similar approaches for birds (Table 3). Katzner et al. (2017) analysed Golden Eagle carcasses from the APWRA, conducting both stable isotope and population genetic analyses (mitochondrial DNA, microsatellites, and SNPs [single nucleotide polymorphism]), and paired these with a demographic model. Based on these multiple lines of evidence, they concluded that at least 25% of the Golden Eagle population at these facilities was made up of non-locals, that it was in a state of decline, and that only supplementation via continental-scale immigration was allowing it to stay apparently stable. Consequently, the fatalities at the APWRA were having demographic impacts across the species' range. Vander Zanden et al. (2024) also use stable isotopes to assess source populations for Golden Eagles and ten other bird species salvaged from the APWRA (Table 3), and estimated that for these species 0% (White-tailed Kites and Mourning Doves) to 81% (Barn Owl) of individuals were non-local.

Movement plays a key role in the potential for wind energy facilities to act as population sinks. While the extreme intercontinental distances traversed by some Australian shorebirds via routes such as the East Asian–Australasian Flyway are well-recognised, within-continent migratory behaviours are not as well understood. A recent study made use of weather radar data and found that Australia's east coast has a similar bird migration system to North America, with peaks in spring and summer (Shi et al. 2024). However, they also note that movements in Australia were comparatively more variable year to year, that diurnal movements were relatively more common than nocturnal movements, and that they were also more dispersed (i.e. a smaller proportion of birds migrated over a longer time frame and flew in different directions).

4.2.2 Cumulative impacts: changes in occupancy and abundance over time

Long-term monitoring data are required to detect whether there are changes in occupancy or abundance over time due to the effects of collisions at an individual facility, or wind energy developments across a region more broadly. These data can take the form of capture or detection records at the population scale, or changes in the rate of carcasses found at facilities. In general, capture and detection data are preferable because apparent declines are less likely to be confounded by potential avoidance/displacement effects (see Section 4.3), and there are also issues associated with imperfect detection in carcass searches (see Section 7.1).

Davy et al. (2021) analysed seven years of bat carcass count data (corrected for searcher efficiency and carcass persistence) from 48 facilities across southern Ontario in Canada as a proxy for 'captures', and detected a 65–91% decline in estimated fatalities of four species (Hoary Bats, Silver-haired Bat, Big Brown Bat and Eastern Red Bat) over this time period. All four species had previously been considered common, with stable populations. Again, the extent to which these facilities could be causing population declines more broadly will be dependent on whether the immigration of non-locals is compensating for these declines (see Section 4.2.1 above). In contrast to this, another Canadian study, this time focussed on a single site in southwest Saskatchewan that had been monitored for 20 years (not at a wind energy facility), failed to detect any change in the number of Hoary Bat and Silver-haired Bat live captures during that time. This suggests that populations away from wind energy facilities were remaining stable (Green et al. 2020).

Long-term monitoring studies have also been conducted on birds; over nine years, Gómez-Catasús et al. (2020) monitored the abundance of 14 Dupont's Lark (a small passerine) populations in central Spain, including six that were in the vicinity of a wind energy facility. They showed that the populations close to the facilities declined at a rate that was four times faster (21% annually on average) than those farther away from turbines, and that these declines resulted in an overall negative trend for the regional population. Farfán et al. (2017b) also monitored the abundance of birds at a single wind energy facility in the south of Spain over an 11-year period, and found that while there was evidence that activity of raptor populations (representing 11 species) declined and then recovered during that time, non-raptor populations (30 passerine species, and 8 other non-passerines) did not recover. The authors note that collisions appeared to be rare at this facility, so they suggest that these patterns are due to initial avoidance and displacement, then subsequent habituation by the raptors (see Section 4.3).

4.2.3 Interactions and synergisms with other threats

As highlighted above, it is important to consider the additive and synergistic effects of multiple stressors and threatening processes that may be acting on a species, as opposed to just the impacts of collisions in isolation, to determine whether in combination they could push populations past a tipping point. For example, Rodhouse et al. (2019) used eight years of bat capture and acoustic data collected from 190 sites across Oregon and Washington state in the USA to develop occupancy models, and to assess the combined impact of the arrival of white-nose syndrome (WNS; a fungal disease killing millions of bats in North America) and expansion of wind energy on Hoary and Little Brown Bat populations. While the Hoary Bat population appeared to have experienced a ~25% decline in that time, the Little Brown Bat population had not, possibly because WNS was not yet impacting on this species regionally and the combined effects of both threats were not in place. This aligns with the work of Erickson et al. (2016) who used a full-annual-cycle population model to demonstrate that WNS impacts and turbine mortalities combined would have a synergistic effect on populations of the endangered Indiana Bat, that were greater than would be expected individually. This is because turbines caused local extinctions of some subpopulations and disrupted metapopulation dynamics, while WNS had a suppressive effect on the species population as a whole.

Martínez-Abraín et al. (2012) also demonstrated that additional, seemingly unrelated threatening processes can influence whether species populations can persist despite impacts caused by collisions, in this case focussing on Griffon Vultures in eastern Spain. A bovine disease epidemic in 2001 lead to the closure of vulture feeding stations, where cattle carcasses had historically been disposed of and made available to scavengers. Without access to these stations, vultures had to travel farther to find food, while at the same time there was a rapid expansion of wind energy facilities in the region, so vultures began frequently colliding with turbines. Using four years of mark-recapture data, in combination with long-term colony counts, Martínez-Abraín et al. (2012) were able to show that both of these drivers in tandem resulted in a 25% decline in the number of breeding pairs, and also impacted survival and fecundity. However, once the feeding stations were re-opened and the turbines associated with the largest number of fatalities were shut down, the population recovered.

4.2.4 Modelling current population-level impacts

Demographic models can take various forms, and are used to help understand how species populations change through time. For sexually reproducing vertebrates such as birds and bats, these models consider a species' population structure (the number of individuals, what age and sex classes they are made up of, whether they are reproductive or not) and vital rates (fecundity and survival rates of each sex and age class). Spatially explicit models can also account for movement distances and behaviours that dictate rates of immigration into—and emigration out of—subpopulations, the distribution of habitat within an area of interest, and how this influences the carrying capacity (the maximum number of individuals that an area can sustain) of each subpopulation (Akçakaya 2004).

When demographic models are used to predict how a species' population will change into the future, and whether or not it will remain viable or sustainable over a set time period, they are typically called 'Population Viability Analyses' (PVAs). PVAs often account for management actions, such as the reintroduction of individuals to a population, or threats, such as (in the case of turbines) the number of individuals that are lost from the population at each time step due to collision fatalities. For this reason, PVAs are sometimes used in the pre-construction phase to assess the risk of a facility or facilities to species of interest (see Section 5.6.2). Here, we focus on cases where real-world data collected during post-construction mortality surveys are incorporated into demographic models and PVAs to help understand what impact wind energy collisions have already had on species populations, and whether the species will persist into the future if those impacts continue.

Bellebaum et al. (2013) synthesised post-construction mortality data for Red Kites from 69 facilities across north-east Germany, and estimated that ~3% of the breeding population was being killed by collisions each year. Based on a potential biological removal model (PBR, which have significant limitations, see Section 5.6.2), they predicted that the population could sustain losses of up to 4%, and expressed concern that that threshold would be crossed as wind energy capacity across the region increased. Durietz et al. (2023) used a similar approach for another medium-sized raptor, the Lesser Kestrel in France. They also estimated annual turbine-related mortality rates of approximately 3% of the population, and using a matrix population model, predicted that the population could sustain fatalities up to a rate of 11%. Schippers et al. (2020) used a matrix population modelling approach for seven bird species across Europe representing different body sizes and guilds (e.g. Common Starling, Black-tailed Godwit, White-tailed Eagle) to demonstrate that even apparently small increases in mortality rates can have substantial implications for species' population viability. They found that over 10 years, a 1% increase in mortality rates resulted in a 2–24% reduction in population size (depending on the species), and that these reductions increased to 9–77% if mortality rates increased to 5%, with a high level of variability between species.

It is well-recognised that populations of longer-lived species, that produce fewer young each year (such as larger raptors and microbats), are not as capable of sustaining population numbers with increased mortality rates. Using individual-based demographic models for species with different life histories (cave and tree-dwelling microbats, a passerine and a raptor), Erickson et al. (2015) showed that longer-lived species were at greater at risk of extinction from wind turbine-induced mortalities. They found that populations of these species could appear stable until a threshold was reached, after which even small increases in collision rates caused rapid increases in extinction risk. The study of Carrete et al. (2009) provides a clear example of this; the authors were able to implement a full PVA (using the program VORTEX) for Egyptian Vultures in Spain, informed by eight years of intensive territory surveys and carcass searches at wind farms. They estimated that mortality rates of territorial individuals living close to wind energy facilities were 1.5% higher than those living farther away. While this may appear to be a relatively low increase in risk, because the species is long-lived and the population was already in decline, the added mortality associated with collisions significantly increased the species probability of extinction in the region.

Frick et al. (2017) also assessed increased extinction risk caused by turbine fatalities, this time for the Hoary Bat in the USA, using a combination of expert elicitation, estimated mortalities from post-construction monitoring and population projection models. They found that the species could undergo up to a 90% population decline over a 50-year time frame (assuming an initial population size of 2.5 million individuals). A subsequent study by Friedenberg and Frick (2021) that used the same model(s) indicated that implementing low-wind speed curtailment mitigation measures (see Section 6.1.1) slowed future declines and reduced extinction risk. However, the authors emphasise that there was a great deal of uncertainty associated with these predictions, and that better estimates of population size in particular were needed.

Demographic models and PVAs are the best way to assess how wind energy developments could increase the extinction risks for threatened or even common species, as well as how effective different siting options and mitigation measures may be at reducing this risk. However, the construction of these models is far from trivial, as exemplified by the case study of the Hoary Bat highlighted above (Frick et al. 2017; Friedenberg and Frick 2021). This species was the equal most well-studied in our review (40 papers, Table S2), is broadly distributed and common, and occupies a region that has both a monitoring program in place (the North American Bat Monitoring Program, NABat 2024), and where systematic post-construction surveys have been conducted for decades. Nonetheless, the authors highlighted that precise baseline population estimates were still needed before confident predictions could be made. This is certainly the case for many Victorian bird and bat species, where comparatively much less is known about life history parameters, metapopulation structure and size, movement behaviours, population trends and the rate of mortalities caused by collisions with turbines.

Migratory species present unique challenges when attempting to predict impacts across an entire species range, and this applies to both bats and birds. Indeed, in their paper presenting an online app to run demographic simulations to assess population-level impacts of fatalities for European birds, Chambert et al. (2023b) caution that "…*currently, there is no easy solution to this issue and it remains very difficult to assess the demographic consequences of fatalities occurring along migratory pathways*". As noted above in Section 4.2.1, our understanding of the movement and migratory patterns of birds in Victoria is still relatively limited, while it is arguably negligible for bats (with the exception of flying-foxes), with much still to be learnt to inform risk assessments.

Knowledge gaps for Victoria

1. There are no broad-scale, comprehensive long-term monitoring programs for bats or birds (equivalent to the North American Bat Monitoring Program) in Victoria, to provide baseline population data for the detection of changes over time due to the impacts of wind energy facilities or other threats, including cumulative impacts.
2. Basic empirical information about population size and structure, and vital rates (fecundity and survival rates) is not available for most species that are impacted by turbines in Victoria. This prevents the development of reliable demographic models (e.g. PVAs), and limits our ability to predict both broad-scale impacts and also the potential effectiveness, at a population level, of different siting options and mitigation measures.
3. There is little information on movement or migratory patterns of most species of microbats and birds in Victoria.
4. No studies to date have developed terrestrial isoscapes for Australia, so it is not possible to use stable isotope analyses to identify the origin location of birds and bats that have collided with turbines.

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Table 3. Biomarker (isotopic, trace element, and genetic) studies assessing broad-scale implications of turbine collisions for species.

No.: sample size (number of individuals), % local: percent of the individuals sampled that are predicted to have come from the local population, PMD: predicted minimum distance from the carcass location to the likely area of origin of individuals identified as non-local, NEHIK: non-exchangeable hydrogen in keratin, SDM: species distribution model, C: carbon, N: nitrogen, H: hydrogen. Rows are ordered so that comparable studies and species are grouped together (European bats, then North American bats, then North American birds).

Study	Country	Taxon	Species	No.	% local	Predicted origin of non-locals and other findings	Methods
Lehnert et al. (2014)	Germany	Bats	Common Noctule	136	76%	Baltic countries (Lithuania, Latvia, Estonia), Belarus and Russia	NEHIK
Mantoiu et al. (2020)	Romania	Bats	Common Noctule	40	6–16%	Ukraine, Belarus and western Russia	NEHIK
Voigt et al. (2012)	Germany	Bats	Common Noctule	14	Not estimated	Poland, Scandinavia, Baltic States or Belarus	NEHIK
			Common Pipistrelle	16		Western Europe	
			Leisler's Bat	7		Baltic countries or Belarus	
			Nathusius' Pipistrelle	10		Estonia or Russia	
Voigt et al. (2016)	Germany	Bats	Common Noctule	38	Not estimated	Sweden, Poland, Lithuania, Latvia, Belarus	Stable C, N, and NEHIK
			Common Pipistrelle	11		Germany (i.e. predominantly locals)	
			Nathusius' Pipistrelle	19		Fennoscandinavia, Baltic countries, Belarus, Russia	
Kruszynski et al. (2022)	Germany	Bats	Nathusius' Pipistrelle	119	93%	Northeastern areas in Europe, such as Russia and Finland	NEHIK
Wieringa et al. (2023)	USA, Canada	Bats	Eastern Red Bat	96	Not estimated	Cell with highest probability 655 km from known origin	Trace elements, stable H, SDM
			Hoary Bat	134		Cell with highest probability 1485 km from known origin	
			Silver-haired Bat	49		Model results have low accuracy	
Murtaugh et al. (2019)	USA	Bats	Eastern Red Bat	35	29%	Eastern US (Iowa, Illinois, and Ohio down to northern Florida) and Canada (bats salvaged in Illinois)	Stable H
Pylant et al. (2016)	USA	Bats	Eastern Red Bat	144	43%	Summered to the south or west. No population genetic structure, large effective population size.	NEHIK, population genetic analyses
			Hoary Bat	246	99%	Summered to the north. No population genetic structure, relatively small effective population size.	

Study	Country	Taxon	Species	No.	% local	Predicted origin of non-locals and other findings	Methods
Baerwald et al. (2014)	Canada	Bats	Hoary Bat	176	Not estimated	Originated from farther north (carcasses collected from southern Alberta), potentially hundreds of kilometres away	Stable C, N, and H, microsatellites
			Silver-haired Bat	119		Most individual migrants originating from farther north, potentially hundreds of kilometres away	
Katzner et al. (2017)	USA	Birds	Golden Eagle	67	<75%	California, Nevada, Idaho, Colorado, Arizona, New Mexico, Wyoming, and Oregon most likely sources of eagles (carcasses collected from California). APWRA acting as a population sink.	Stable H, SNP genotyping
Vander Zanden et al. (2024)	USA	Birds	Golden Eagle	76	63%	PMD of non-locals 262 ± 346 km	NEHIK
			American Kestrel	42	67%	PMD of non-locals 101 ± 24 km	
			Barn Owl	54	19%	PMD of non-locals 177 ± 168 km	
			Burrowing Owl	37	30%	PMD of non-locals 167 ± 210 km	
			Great Horned Owl	43	60%	PMD of non-locals 133 ± 115 km	
			Horned Lark	43	77%	PMD of non-locals 97 ± 27 km	
			Mourning Dove	6	100%	All individuals predicted to be locals	
			Red-tailed Hawk	86	37%	PMD of non-locals 192 ± 203 km	
			Western Meadowlark	15	40%	PMD of non-locals 179 ± 92 km	
			Wilson’s Warbler	5	40%	PMD of non-locals 286 ± 193 km	
			White-tailed Kite	3	100%	All individuals predicted to be locals	

4.3 Attraction, avoidance and displacement

Large number of papers (79)	Low bird bias (78%)	Low geographic bias (Europe 46%)

The construction and operation of wind energy facilities can impact on both the number of individuals in a species' population (as detailed above, see Section 4.2), as well as its home range or the spatial footprint of the habitat that it uses, and it is difficult to disentangle the effects of these two phenomena.

As a hypothetical example to illustrate this: once turbines are constructed and operating, and separate from any impacts due to collisions, there may be (Scenario A) an increase in local resources such as insects, so an individual microbat may be attracted to the area and may not have to travel as far to forage, resulting in the area covered by its home range contracting. With more resources and smaller individual home ranges, the area around the wind energy facility could accommodate more individual bats, resulting in an overall local population increase. Conversely, construction and operation might result in (Scenario B) habitat destruction, causing a local population reduction and forcing remaining individuals to occupy a smaller area. Based on field studies (such as acoustic monitoring) it can be difficult to ascertain whether local increases in activity are a result of Scenario A or Scenario B. Likewise, in cases where fewer individuals are observed post-construction, it can be difficult to determine whether this is because some have been killed by collisions on-site, or if they have been displaced and individuals that were part of the local population are now persisting elsewhere.

In this section, we report the findings according to whether there has been apparent attraction, avoidance/displacement (we do not attempt to distinguish the two), no significant effect, or a 'recovery' (i.e. initial avoidance or displacement but then improvement over time) as reported by the authors of the studies. However, we acknowledge that these effects are complex and may be the result of multiple processes occurring simultaneously (Figure 4).

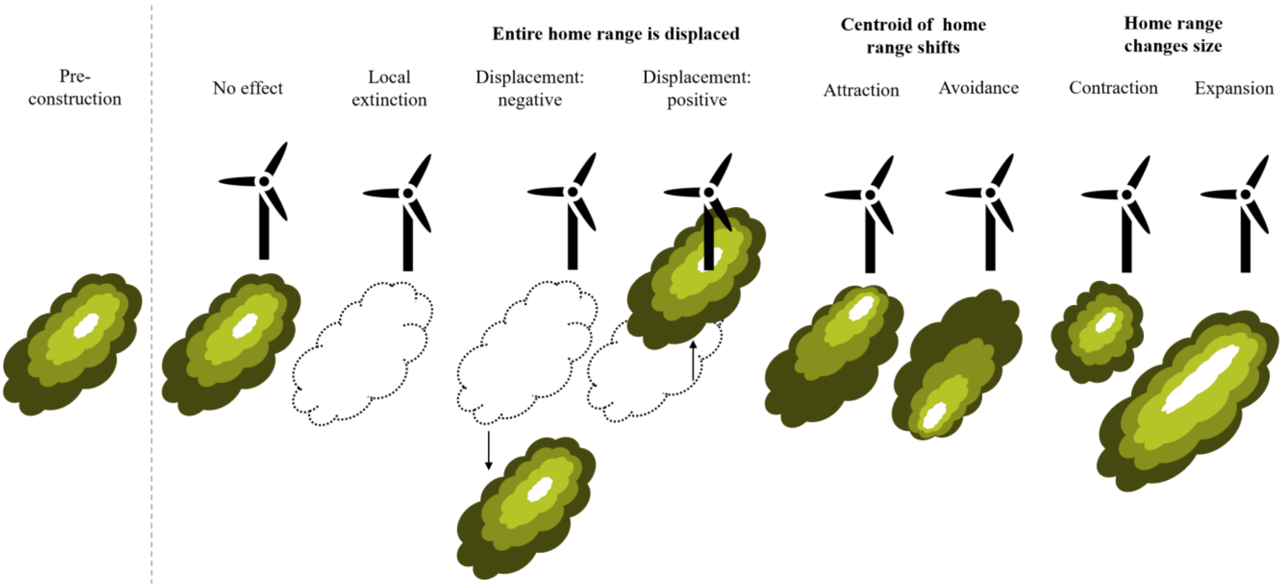

Figure 4. Conceptual classification of changes to bird or bat home ranges and habitat use caused by the construction of a wind turbine.

Adapted from Winder et al. (2014), with the home range shown as a patch, and higher levels of activity indicated by lighter colours. Turbine icon: Iconathon, CC0.

While attraction and avoidance/displacement effects are non-fatal, and therefore may appear out of scope for this review (given our focus on collisions), they are important to consider here because they have the potential to directly influence actual and predicted collision fatality risks. Pre-construction risk assessments are, by definition, conducted before turbines become operational. Consequently, if the presence of the turbine structure or the motion of the blades does draw some species in and repel others, the risks of collisions will be underestimated and overestimated, respectively. Avoidance rates are a key parameter in collision risk models used for birds (CRMs, see Section 5.6.1), and are both difficult to estimate and have a strong influence on predictions of collision rates (Smales et al. 2013). Therefore, an accurate understanding of these behavioural responses is important to the risk assessment process.

Because of the potential for both fatality-related and avoidance-related impacts, some jurisdictions adopt policies and guidelines recommending that turbines be set back a certain distance from key habitat features, a practice known as ‘buffering’ (see Section 6.2). While the motivation for several of the attraction and avoidance studies reviewed in this section was the recommendation of buffer distances, we do not include these here, and instead discuss them below in ‘Mitigation options and effectiveness’ (Section 6.2).

4.3.1 Attraction, avoidance and displacement of bats

We identified 22 papers from across the USA, Canada and Europe that empirically assessed the behavioural effect of operating wind turbines on bats (Table 4), using a range of metrics (e.g. acoustic activity, movement patterns, carcass counts) and contrasts (e.g. distance to turbines, turbine densities, turbine operation). Sometimes the same study identified different effect types for different species (e.g. some were attracted while others were displaced) or investigated different response types (e.g. activity, and also movements). For the purpose of the summaries presented here, a ‘case’ is a unique effect and response type combination found for at least one species per paper. Where the same effect/response combination was recorded for multiple species in the same paper it was counted as only one ‘case’, to avoid any biases introduced by the field or analytical methods used, or by the landscape context specific to that study. The trade-off here is that this approach may introduce bias towards unusual results; for example, if in a study many species were displaced, while only one was attracted, then it would only count as one case each way. However, based on our reading of these papers, we do not believe that this potential bias had a substantial influence on the patterns that emerged.

In total, there were 12 cases where the response for bats was characterised as attraction, 12 where it was avoidance/displacement, and nine where there was no significant difference between treatment groups or contrasts (Figure 5a). Interestingly, this result differs from a recent systematic review of displacement effects of wind power published by Tolvanen et al. (2023), in which the authors concluded that displacement of bats occurred in 72% of the 29 cases they reviewed (from 9 studies), and that on average this occurred up to 1 km. The differing conclusions drawn by our review and that of Tolvanen et al. (2023) seem to be the result of differences in study scope (they focussed solely on displacement, and attraction was lumped as ‘no significant difference’ if distance was not analysed) and inclusion criteria (we limited our search to the period 2010–2024, and excluded studies of small, non-commercial turbines because these are not common in Victoria).

Bats are typically classified into guilds according to their morphology, echolocation characteristics and feeding behaviours. For example (and as a broad generalisation), fast-flying species with longer, narrower wings tend to have louder, lower-frequency calls and forage in more open areas. The use of guilds is often informative when deducing patterns in species responses to their environment and threats. However, our review did not reveal patterns in the types of bat species and guilds that tended to be attracted to (or conversely, displaced by) turbines, and in some cases a species classified as ‘attracted’ in one study was an ‘avoider’ in another (e.g. Common Pipistrelle, Table 4).

Acknowledging this fairly even split between response types, and the lack of clear patterns in the traits that may predispose a species to being attracted to- or displaced by- turbines, there was however, compelling evidence that some species were in fact attracted to operational turbines. Several studies used thermal cameras to directly observe the behaviours of bats around turbines (e.g. Cryan et al. 2014; Goldberg et al. 2021) and recorded what they describe as ‘investigative’ and ‘approach’ behaviours. Others observed that bat activity increased when turbines became operational (Smallwood and Bell 2020a), or that there was a correlation between bat activity and recorded fatalities during operational periods, but not when the turbines were not operating (Peterson et al. 2021).

Bat attraction to operational turbines has long been postulated as a potential cause for the high fatality rates observed for some species (e.g. Cryan et al. 2009). Hypotheses for why species may be attracted include noise and heat, that bats perceive the towers as providing roosting or mating opportunities, and that aggregations of insects around towers attract foraging bats (Smallwood and Bell 2020a; Guest et al. 2022). The lack of clear patterns found here emphasises the critical role of species- and context-specific information when considering how bats interact with turbines.

Table 4. Summary of studies that have presented evidence for an effect of attraction, avoidance, or displacement of bats from turbines.

Each row represents a single case, and they are sorted according to the effect found, and then by study, with attraction first (12 cases), then avoidance/displacement (12 cases), and then no significant difference (nine cases). Where the same study investigated multiple response types or found multiple types of effects, it is listed once for each response × effect type combination, with the corresponding species detailed.

Study	Methods	Country	Setting	Response	Contrasts	Effect found	Species or guilds	Detail
Ellerbrok et al. (2022)	Acoustics	Germany	Woodland or forest	Activity	Distance to turbines	Attraction	Open-space foragers (three genera)	Higher activity close to turbines in late summer
Ferri et al. (2016)	Acoustics	Italy	Grassland	Activity	One- versus 3-blade turbines	Attraction	Common Pipistrelle	Higher activity after repowering to the 3-blade turbines
Jameson and Willis (2014)	Acoustics	Canada	Woodlots, farmland	Activity	Sites with/ without towers (turbine proxy)	Attraction	Silver-haired, Hoary, and other low-frequency bats	Higher activity at communication towers during migration, exceeded that at open fields and matched that at woodlots
Leroux et al. (2022)	Acoustics	France	Farmland	Activity	Sites with/ without turbines	Attraction	Short-range echolocators (three genera)	Higher activity at intermediate distances from hedgerows (43–100 m) under turbines compared to areas without turbines
Leroux et al. (2023)	Acoustics	France	Unspecified	Activity	Turbine density	Attraction	Western Barbastelle and Nathusius' Pipistrelle	Higher activity with higher turbine density in low-wind conditions
Leroux et al. (2023)	Acoustics	France	Unspecified	Activity	Distance to turbines	Attraction	Western Barbastelle and Kuhl's Pipistrelle	Higher activity closer to turbines in low-wind conditions
Richardson et al. (2021)	Acoustics	UK	Unspecified	Activity	Sites with/ without turbines	Attraction	Common Pipistrelle	Higher activity at sites with turbines than without
Peterson et al. (2021)	Acoustics, carcass searches	USA	Alpine or ridges	Activity Fatalities	Temporal, Operational	Attraction	Not specified (up to 14 species)	Fatalities more strongly predicted by activity during operational, versus non-operational periods
Roeleke et al. (2016)	GPS tracking	Germany	Farmland	Resource selection	Distance to turbines	Attraction	Common Noctule	Three females used areas closer to turbines than expected based on random samples
Cryan et al. (2014)	Thermal camera	USA	Farmland	Behaviours	Wind and blade speed	Attraction	Eastern Red, Hoary and Silver-haired Bats	Actively approached close (<50 m) to turbine blades when they were stationary or slow
Goldenberg et al. (2021)	Thermal camera	USA	Farmland	Behaviours	Temporal (single turbine)	Attraction	Not identified	High proportion of observed behaviours 'approaches', with peaks in summer/ autumn

Study	Methods	Country	Setting	Response	Contrasts	Effect found	Species or guilds	Detail
Smallwood and Bell (2020a)	Thermal camera	USA	Grassland	Passage rates	Operational	Attraction	Not identified	Passage rates significantly declined when turbines were shutdown compared to controls that remained operational
Barré et al. (2018)	Acoustics	France	Farmland	Activity	Distance to turbines	Avoidance/ displacement	Eight species or complexes	Lower activity in hedgerows when closer to turbines
Ellerbrok et al. (2022)	Acoustics	Germany	Woodland or forest	Activity	Distance to turbines	Avoidance/ displacement	Narrow-space foragers (two genera)	Lower activity closer to turbines over distances of several hundred metres
Ellerbrok et al. (2024)	Acoustics	Germany	Woodland or forest	Activity	Distance to turbines Operational	Avoidance/ displacement	Narrow-space foragers	Lower activity at high wind speeds close to turbines only when operational
Ferri et al. (2016)	Acoustics	Italy	Grassland	Activity	One- versus 3-blade turbines	Avoidance/ displacement	Geoffroy's Bat	Lower activity after repowering to the 3-blade turbines
Gaultier et al. (2023)	Acoustics	Finland	Woodland or forest	Activity	Distance to turbines	Avoidance/ displacement	Northern Bat and *Myotis* sp.	Lower activity within 600–800 m of a turbine
Leroux et al. (2022)	Acoustics	France	Farmland	Activity	Sites with/ without turbines	Avoidance/ displacement	Common Pipistrelle, short- (three genera) and long-range echolocator (three species)	Lower activity in areas close to hedgerows when also under turbines
Leroux et al. (2023)	Acoustics	France	Unclear	Activity	Turbine density	Avoidance/ displacement	Noctule bats	Lower activity with higher turbine density
Millon et al. (2018)	Acoustics	New Caledonia	Alpine or ridges	Activity	Sites with/ without turbines	Avoidance/ displacement	*Chalinolobus* (wattled bats) and *Miniopterus* (bent-winged bats)	Lower activity (10 and 20 times, respectively) at sites with turbines compared to those without turbines
Millon et al. (2015)	Acoustics, radio-tracking	France	Farmland	Activity	Sites with/ without turbines	Avoidance/ displacement	Three species complexes (comprising 14 individual species)	Lower activity in crops under wind turbines than in crops without wind turbines
Reusch et al. (2022)	GPS tracking	Germany	Coastal	Movements	Distance to turbines	Avoidance/ displacement	Common Noctule	Flight movements avoided turbines

Study	Methods	Country	Setting	Response	Contrasts	Effect found	Species or guilds	Detail
Reusch et al. (2023)	GPS tracking	Germany	Woodland or forest	Movements	Distance to turbines	Avoidance/ displacement	Common Noctule	Avoided turbines over the scale of several kms when they were >500 m from a roost
Roeleke et al. (2016)	GPS tracking	Germany	Farmland	Resource selection	Distance to turbines	Avoidance/ displacement	Common Noctule	Five males used areas further from turbines than expected based on comparison to random samples
Barré et al. (2018)	Acoustics	France	Farmland	Activity	Distance to turbines	No significant difference	Six species or complexes	No significant effect of distance to turbines on activity
Budenz et al. (2017)	Acoustics	Germany	Woodland or forest	Activity	Height (lattice towers as a proxy for turbines)	No significant difference	Western Barbastelle	No evidence of exploratory flights
Ellerbrok et al. (2022)	Acoustics	Germany	Woodland or forest	Activity	Distance to turbines	No significant difference	Edge-space foragers (two genera)	No significant effect of distance to turbines on activity
Ellerbrok et al. (2024)	Acoustics	Germany	Woodland or forest	Activity	Distance to turbines Operational	No significant difference	Open-space (three genera) and edge-space foragers	No significant effect of distance to turbines on activity
Ferri et al. (2016)	Acoustics	Italy	Grassland	Activity	One- versus 3-blade turbines	No significant difference	Seven species or complexes	No significant effect of number of blades on activity
Leroux et al. (2023)	Acoustics	France	Unclear	Activity	Turbine density	No significant difference	Four species or complexes	No significant effect of turbine density on activity
Reimer et al. (2018)	Acoustics	Canada	Farmland, Grassland or prairie	Activity	Height Structure type (turbine versus tower)	No significant difference	Silver-haired and Hoary Bats	No significant effect of height or structure type on foraging rates
Richardson et al. (2021)	Acoustics	UK	Unclear	Activity	Sites with/ without turbines	No significant difference	Soprano Pipistrelle	No significant effect of presence of turbines on activity
Segers and Broders (2014)	Acoustics, radio-tracking	Canada	Woodland or forest	Activity	Sites with/ without turbines	No significant difference	Little Brown Bats and Long-eared Myotis	No significant effect of presence of turbines on activity

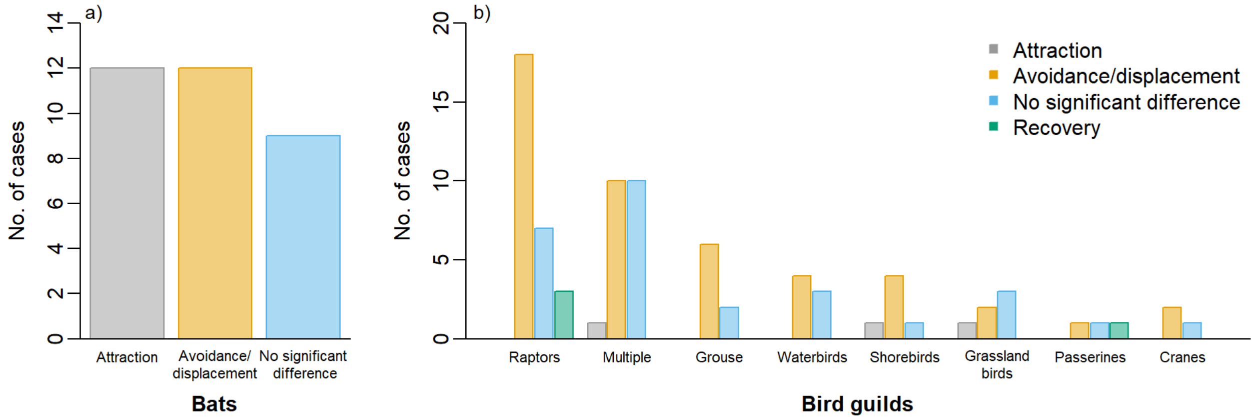

Figure 5. Number of cases where evidence of attraction, avoidance/displacement, and recovery were found, or there was no significant difference for a) all species of bats and b) birds of different guilds. Note that a single paper may have found an effect for multiple species, but these are only represented as one 'case' to avoid any biases introduced by particular field or analytical methods. A 'recovery' effect refers to studies where species appear to have been displaced following construction, but then local abundance or activity return to previously-observed levels.

4.3.2 Attraction, avoidance and displacement of birds

We reviewed 74 papers that assessed attraction and avoidance for birds, and from these identified three cases (or 4%) of attraction, 47 cases (57%) of avoidance, 28 cases (34%) where there was no significant difference, and 4 cases of recovery (5%), where a species had been recorded being displaced but then appeared to have 'bounced back' and recovered (Table 5; Figure 5b; see Section 4.3.1 above where explanation of 'cases' is provided).

In contrast to the findings from the studies of bats, there were some clear patterns that emerged from our review for birds. Namely, there seemed to be consistent evidence that raptors, waterbirds and grouse can avoid operational wind turbines, or there was no significant difference between treatment groups or contrasts, possibly because their primary means of navigation is visual and therefore they able to perceive and avoid turbines more readily than bats. One reason that raptors may still frequently collide with turbines despite their use of visual cues is that they look down while hunting and scavenging, so may not detect a turbine in front of them as readily (Martin et al. 2012).

The patterns we identified are broadly consistent with those found in three other reviews of bird behavioural responses to turbines. The Tolvanen et al. (2023) study discussed above in relation to bats found that 63% of the 116 cases they reviewed (from 66 papers) indicated there was evidence of displacement; cranes (during migration) and owls were displaced on average up to 5 km, while for waterbirds, shorebirds, raptors, and passerines this distance was up to 500 m. In another general bird displacement review, Marques et al. (2021) found displacement in 41% of experimental trials (from 71 papers), and mean displacement distances of 116 m for waterbirds, 248 m for passerines, 474 m for raptors, 2.5 km for shorebirds and 4.5 km for grouse. A general review of wind energy impacts specifically related to raptors was also recently published by Estellés-Domingo and López-López (in press), and they found that 75% of papers reported avoidance or displacement behaviours for that group.

Avoidance and displacement responses can take many forms, but can be grouped according to the scale at which they occur (following May 2015, and Estellés-Domingo and López-López in press):

- macro-avoidance (territory abandonment, or changing flight paths generally used to avoid a facility altogether)
- meso-avoidance (flying higher or lower on average compared to controls in response to the presence of turbines to avoid the rotor-swept area, or navigating between turbines)
- micro-avoidance (last-minute changes in flight directions and altitudes).

While we did not categorise studies at this resolution, Estellés-Domingo and López-López (in press) found that micro-scale avoidances were the most well-studied for raptors. May (2015) also outlines the likely ecological and evolutionary drivers that influence avoidance behaviours for birds.

Knowledge gaps for Victoria

5. No published research has assessed attraction or avoidance effects of turbines on Australian microbats – ideally, this would be achieved experimentally by comparing passage rates at operational versus non-operational turbines, but comparing pre- and post-construction activity levels could also provide limited insight.
6. As attraction to turbines has been reported considerably more frequently in bats than birds, studies aimed at understanding the prevalence of attraction in bats should be a high priority, as should studies to determine the underlying traits or mechanisms associated with attraction.
7. Only one published study (Hull and Muir 2013) has experimentally tested avoidance or attraction of Australian birds to turbines, and this focussed on two eagle species at two facilities in Tasmania. Further information is needed for a broader range of bird taxa.

Key references

Estellés-Domingo, I, López-López, P (in press) Effects of wind farms on raptors: A systematic review of the current knowledge and the potential solutions to mitigate negative impacts. *Animal Conservation*.

Guest, EE, Stamps, BF, Durish, ND, Hale, AM, Hein, CD, Morton, BP, Weaver, SP, Fritts, SR (2022) An updated review of hypotheses regarding bat attraction to wind turbines. *Animals* **12**, 343.

Marques, AT, Batalha, H, Bernardino, J (2021) Bird displacement by wind turbines: assessing current knowledge and recommendations for future studies. *Birds* **2**, 460-475.

May, RF (2015) A unifying framework for the underlying mechanisms of avian avoidance of wind turbines. *Biological Conservation* **190**, 179-187.

Tolvanen, A, Routavaara, H, Jokikokko, M, Rana, P (2023) How far are birds, bats, and terrestrial mammals displaced from onshore wind power development? – a systematic review. *Biological Conservation* **288**, 110382.

Table 5. Summary of studies looking at evidence for the effect of attraction or avoidance/displacement of birds from turbines.

Each unique combination of bird guild, response variable (and methods used to assess this), types of contrasts and effect detected (no significant difference, avoidance/displacement, attraction, and recovery) is presented. A 'recovery' effect refers to studies where species appear to have been displaced following construction, but then local abundance or activity return to previously observed levels. The total number of cases are presented for each guild, with a 'case' being a study finding an effect for one or more species for a given response. Where a single study detected multiple effect types (e.g. some species were attracted, while others were displaced) or quantified different types of responses (e.g. investigated both movements and abundance), it has been presented as multiple cases. Abbreviations used for methods are – DObs: Direct observation (direct counts or behavioural assessments), Telem: Telemetry (radiotracking, and satellite or GPS tracking). Abbreviations used for contrasts are – B/A: Before-after construction, W/WO: Sites with, versus those without, turbines, Dist: Distance to turbines, OP: Sites where turbines were operational vs non-operational. Rows are ordered such that guilds with the most cases appear first.

Response (methods used)	Contrasts	No significant difference	Avoidance/displacement	Attraction	Recovery
Overall total (all guilds)		**28**	**47**	**3**	**4**
Raptors (28 cases)		**7**	**18**	**0**	**3**
Abundance (DObs)	B/A	Hernández-Pliego et al. (2015)	Dohm et al. (2019) Garvin et al. (2011)		Dohm et al. (2019) Farfán et al. (2023)
Density (DObs)	B/A		Hernández-Pliego et al. (2015)		
Flight height (DObs)	B/A	Garvin et al. (2011)	Campedelli et al. (2014)		
Movements (DObs)	B/A		Campedelli et al. (2014) Hull & Muir (2013)		
Movements (DObs)	W/WO	Dahl et al. (2013)			
Movements (Radar)	B/A		Cabrera-Cruz and Villegas-Patraca (2016)		
Movements (Telem)	B/A		Fielding et al. (2021) Fielding et al. (2022)		
Movements (Telem)	Dist		Santos et al. (2022)		
Occupancy (Acoustics)	B/A		Husby and Pearson (2022)		
Passage rates (DObs)	B/A	Nishibayashi et al. (2022)			Farfán et al. (2023)
Resource selection (DObs)	B/A	Nishibayashi et al. (2022)	Johnston et al. (2014)		
Resource selection (Telem)	B/A		Fielding et al. (2023)		
Resource selection (Telem)	Dist	May et al. (2013)	Marques et al. (2020) Santos et al. (2021)		
Resource selection (Telem)	W/WO	Balotari-Chiebao et al. (2018)	Fielding et al. (2023) May et al. (2013) Schaub et al. (2020)		

Response (methods used)	Contrasts	No significant difference	Avoidance/displacement	Attraction	Recovery
Multiple guilds (21 cases)		**10**	**10**	**1**	**0**
Abundance (DObs)	B/A	Farfán et al. (2009)	Farfán et al. (2009) Lehnardt et al. (2024)		
Abundance (DObs)	Dist	Rehling et al. (2023)	Cheng et al. (2021) Rosin et al. (2016)	Cheng et al. (2021)	
Community composition (Acoustics)	W/WO	Raynor et al. (2017)			
Community composition (DObs)	Dist	Rehling et al. (2023)	Rosin et al. (2016)		
Flight height (DObs)	B/A		Therkildsen et al. (2021)		
Flight height (DObs)	Dist	Pearce-Higgins et al. (2009)			
Occupancy (DObs)	Dist	Pearce-Higgins et al. (2009) Rehling et al. (2023)	Pearce-Higgins et al. (2009)		
Passage rates (DObs)	B/A	Farfán et al. (2009) Pande et al. (2013)	Farfán et al. (2009) Therkildsen et al. (2021)		
Passage rates (DObs)	OP	Smallwood and Bell (2020a)			
Passage rates (Radar)	W/WO		Villegas-Patraca et al. (2014)		
Grouse (8 cases)		**2**	**6**	**0**	**0**
Abundance (DObs)	B/A	Zwart et al. (2015)	González et al. (2016)		
Density (DObs)	B/A	Coppes et al. (2020b)			
Occupancy (DObs)	Dist		Coppes et al. (2020b)		
Resource selection (DObs)	B/A		Zwart et al. (2015)		
Resource selection (Telem)	B/A		Winder et al. (2014b)		
Resource selection (Telem)	Dist		Taubmann et al. (2021) Winder et al. (2014b)		

Response (methods used)	Contrasts	No significant difference	Avoidance/displacement	Attraction	Recovery
Waterbirds (7 cases)		**3**	**4**	**0**	**0**
Abundance (DObs)	B/A	Zehtindjiev et al. (2017)			
Abundance (DObs)	Dist		Fijn et al. (2012)		
Density (DObs)	W/WO	Loesch et al. (2013)	Loesch et al. (2013)		
Density (Telem)	Dist		Zhao et al. (2020)		
Flight height (Telem)	W/WO		Zhao et al. (2022)		
Passage rates (DObs)	B/A	Zehtindjiev et al. (2017)			
Shorebirds (6 cases)		**1**	**4**	**1**	**0**
Flight height (DObs)	B/A		Bai et al. (2021)		
Flight height (DObs)	W/WO		Bai et al. (2021)	Bai et al. (2021)	
Movements (DObs)	B/A		Bai et al. (2021)		
Movements (DObs)	W/WO		Bai et al. (2021)		
Occupancy (DObs)	W/WO	Niemuth et al. (2013)			
Grassland birds (6 cases)		**3**	**2**	**1**	**0**
Density (DObs)	Dist	Hale et al. (2014), Shaffer and Buhl (2016)	Shaffer and Buhl (2016)	Shaffer and Buhl (2016)	
Occupancy (DObs)	Dist	Stevens et al. (2013)	Stevens et al. (2013)		
Passerines (3 cases)		**1**	**1**	**0**	**1**
Flight height (Radar)	B/A		d'Entremont et al. (2017)		
Occupancy (DObs)	B/A				Lemaître and Lamarre (2020)
Occupancy (DObs)	Dist	Lemaître and Lamarre (2020)			
Cranes (3 cases)		**1**	**2**	**0**	**0**
Resource selection (Telem)	B/A	Pearse et al. (2016)			
Resource selection (Telem)	Dist		Ellis et al. (2022) Pearse et al. (2021)		

5 Pre-construction planning and assessment

5.1 Environmental Impact Assessment

Small number of papers (13 papers)	High bird bias (92%)	Moderate geographic bias (USA 54%)

Environmental Impact Assessment (EIA) is the term broadly applied to the set of tools and processes that are used to predict the overall positive or negative effects of renewable energy developments before planning consent is given. Each jurisdiction has its own EIA framework and guidelines and these vary substantially globally. However, there are some common elements that we highlight here.

Katzner et al. (2016) provide a review of the EIA approach for wind energy recommended by the US Fish and Wildlife Service, which comprises three steps:

- Preliminary desktop assessments of mapped habitats, and records of species that occur in the area surrounding the proposed developments – if the site appears to be high risk (high value for wildlife), the area may be deemed a 'no go zone' and the proposal abandoned
- Site evaluation, consisting of ground-truthing of the available habitats and habitat features, and identification of roosts and breeding sites
- Quantitative assessment, which primarily includes acoustic surveys of bats and utilisation surveys (both point counts and targeted) for birds, detailed surveys of species behaviours and flight heights using tracking and other technology, and the development of collision risk, resource selection, or population models.

Katzner et al. (2016) note that an ongoing limitation of these processes is that very few studies are conducted using a before-after-control-impact approach which limits evaluation, and there is also little consideration of cumulative impacts.

Chang et al. (2013) conducted a more fine-scale assessment of common EIA elements by reviewing 23 different wind energy siting guidelines developed by states in USA. They broke these into preliminary assessment, pre-construction, and post-construction phase components, and note that while some aspects such as assessment of nearby habitats were frequently included, others such as nocturnal migratory bird surveys were less common (Table 6). They also found that only 46% of the 49 EIAs that they reviewed included some form of bat surveys, compared to 82% that involved at least one bird survey, despite the high recorded rates of fatalities for bats.

Finally, in light of the rapidly developing wind energy industry, Seaton and Barea (2013) reviewed the available literature to develop a risk assessment framework for the New Zealand Falcon, the country's only endemic bird of prey, and a species for which there was relatively little known about its behaviour around turbines and vulnerability to collisions. Their framework entails a nine-stage approach, with the first seven stages occurring during the pre-assessment and construction phases: 1) assessing whether the proposal is within the species' range; 2) a habitat assessment; 3) targeted falcon surveys multiple times over at least two breeding seasons; 4) monitoring breeding success and determining how much time adults and fledglings spend in the RSA; 5) a quantitative assessment of collision risk; 6) determining avoidance, remediation, and mitigation measures; 7) nest surveys (within 1 km) during construction; 8) monitoring nests and the survival of adults and fledglings over multiple years; and 9) reviewing mitigation and biodiversity offsets.

The development of EIAs is complex and will necessarily be regionally specific, informed by target species, landscape context, what existing data, knowledge, tools and models are available, priorities and trade-offs with other values, and various policies and legislative instruments that apply to that area and how they are linked. Nonetheless, identifying elements that are commonly incorporated into these frameworks can help to highlight what might be considered best practice and should be considered, and we have used these to structure the pre-construction section of our review (see Section 6).

Key references

Chang, T, Nielsen, E, Auberle, W, Solop, FI (2013) A quantitative method to analyze the quality of EIA information in wind development and avian/bat assessments. *Environmental Impact Assessment Review* **38**, 142-150.

Katzner, T, Bennett, V, Miller, T, Duerr, A, Braham, M, Hale, A (2016) Wind energy development: methods for assessing risks to birds and bats pre-construction. *Human-Wildlife Interactions* **10**, 42-52.

Seaton, R, Barea, LP (2013) The New Zealand falcon and wind farms: a risk assessment framework. *New Zealand Journal of Zoology* **40**, 16-27.

Table 6. Categories and sub-categories commonly featured in USA state wind energy siting guidelines for Environmental Impact Assessments, as reviewed by Chang et al. (2013).

The authors scored sub-categories from 1–7, with higher values assigned to those that appeared most frequently in the documents (shown in brackets after the sub-category). Note that in some jurisdictions, post-construction monitoring and mitigations are part of a separate process and were not included in this review.

1. Preliminary site screening	2. Preconstruction avian and bat surveying	3. Post-construction protection plan
1.1 Consultation	**2.1 Consultation for protocol**	**3.1 Monitoring**
1.2 Law review	**2.2 General timing**	3.1.1 Mortality survey (4)
1.2.1 Federal wildlife laws (1)	**2.3 Diurnal bird survey methodology**	**3.2 Mitigation**
1.2.2 Local laws (1)	2.3.1 Area searches (2)	3.2.1 Habitat restoration and compensation (2)
1.3 Meteorological (MET) tower attributes	2.3.2 General bird use counts (2)	
1.3.1 MET tower characteristics (3)	2.3.3 Large bird use counts (1)	
1.3.2 Weather/mortality monitoring (1)	2.3.4 Small bird use counts (1)	
1.3.3 Guy wire diverter installation (1)	2.3.5 Breeding bird surveys (1)	
1.4 Development layout and design aspects	2.3.6 Migration counts (2)	
1.4.1 Buffer zones for habitat (3)	2.3.7 Mist netting (1)	
1.4.2 Proximity to protected regions (3)	2.3.8 Nocturnal migratory bird surveys (1)	
1.4.3 Turbine layout (3)	2.3.9 Raptor nest searches (3)	
1.4.4 Turbine characteristics (4)	2.3.10 Winter bird counts (1)	
1.5 Local species reviews	**2.4 Nocturnal bat survey methodology**	
1.5.1 Proximity to known habitat or corridors (7)	2.4.1 Acoustic detection (3)	
1.5.2 Threatened and endangered species identification (6)	2.4.2 Mist netting (1)	
1.5.3 Database search (3)	2.4.3 Radio and radar detection (3)	
1.6 Topographic survey of development site	2.4.4 Roost surveys (2)	
1.6.1 Proximity to hydrologic feature (5)	2.4.5 Visual monitoring (1)	
1.6.2 Vegetation community and structure review (3)	**2.5 Weather monitoring**	
1.6.3 Contour and ridgeline assessment (2)	2.5.1 General metrics (1)	
1.6.4 GIS model (2)		
1.6.5 Aerial photography (3)		

5.2 Strategic planning

Moderate number of papers (41 papers)	Low bird bias (83%)	Low geographic bias (Europe 49%)

Strategic or regional planning approaches can be used to assess the potential costs and benefits of wind energy developments at broader scales than just individual facilities, be it across individual regions (e.g. Obermeyer et al. 2011; Xing and Wang 2021), countries and continents (e.g. Değirmenci et al. 2018; Eichhorn et al. 2019), or even the global scale (e.g. Santangeli et al. 2018). These approaches are spatially-explicit and can account not only for biodiversity values but also economic considerations such as development costs, existing infrastructure and competing land uses, social preferences, and predicted potential for energy generation. Planning tools take advantage of mathematical approaches (optimisations) and heuristic algorithms to assess trade-offs in these considerations, and produce solutions (typically maps of suggested areas zoned for different land uses) that meet biodiversity or other targets in the cheapest way possible, or try to maximise gains within a set budget (Moilanen et al. 2009). In these planning problems, 'costs' may be financial, but could also be the predicted number of fatalities of species of interest, while 'benefits' may be the amount of habitat protected, or the amount of energy generated (e.g. Eichhorn and Drechsler 2010; Boggie et al. 2023).

Ideally, strategic planning should take place prior to any development occurring, to ensure flexibility in potential spatial solutions, though this is not always possible. Nonetheless, at any stage planning tools can help decision-makers assess and visualise complex trade-offs, and provide quantitative support to consideration of how alternative solutions or scenarios will perform against a set of objectives.

Planning tools and approaches typically require spatial data representing the costs, benefits, weightings, and constraints (e.g. existing protected or urban areas) that the decision maker wants to consider as inputs. From a biodiversity perspective, when considering bird and bat species vulnerable to collisions, these data may take the form of species distribution or habitat models (e.g. Santos et al. 2013; Wieringa et al. 2021), movement data collected through telemetry (e.g. Li et al. 2020; Gauld et al. 2022), occupancy records of locations where species have been observed (e.g. Tapia et al. 2009; Squires et al. 2020). Wind energy planning exercises can also take advantage of the significant datasets that are accrued through citizen science initiatives; for example, Newson et al. (2017) took advantage of the Southern Scotland Bat Survey to assess spatial overlap between the distribution of three high risk species and existing or approved wind facilities, while Ruiz-Gutierrez et al. (2021) drew on over 9,000 records of Bald Eagle nests collected in the United States through eBird to model exposure risk. Post-construction fatality monitoring data from existing facilities in a focal area can also be used to predict where areas of high collision risk may be (e.g. Bose et al. 2020a). These risk predictions can, in turn, be used as inputs for spatial planning of additional facilities.

Within the literature a range of approaches have been adopted in strategic planning for wind energy. In regions that are relatively data-poor, the only option may be hot-spot or scoring type analyses, where maps of biodiversity features (such as species habitat) are overlaid and the areas with the highest total biodiversity value are prioritised for protection (e.g. Bernard et al. 2014). However, this does not allow for assessment of trade-offs. Multi-criteria Analyses are also frequently applied in the literature, and as the name suggests these do allows the user to consider a range of criteria in addition to biodiversity, such as energy potential (Ajanaku et al. 2022), amenity impacts (Xing and Wang 2021) and terrain suitability (Değirmenci et al. 2018). If biodiversity is a key objective, then planning software designed specifically for conservation applications such as Zonation or Marxan (Moilanen et al. 2009) may be best suited, because these can also account for considerations such as habitat connectivity and the retention of important migratory corridors, or help to ensure that a certain proportion of a species' distribution remains protected from development (see Santangeli et al. 2018; Balotari-Chiebao et al. 2023 and Boggie et al. 2023 for examples).

Ultimately, the best approach to use in a given situation will depend on a combination of the objectives of the planning exercise, the available data and expertise, policy priorities, and the scale of the area being considered. One of the challenges in assessing the success of the various planning case studies presented in the literature is that it is unclear how many have been adopted and applied, and, subsequently evaluated and refined. As wind energy continues to expand, hopefully there will be greater focus on real-world applications, and, in turn, discussion of realised benefits, constraints, and lessons.

Key references

Balotari-Chiebao, F, Santangeli, A, Piirainen, S, Byholm, P (2023) Wind energy expansion and birds: identifying priority areas for impact avoidance at a national level. *Biological Conservation* **277**, 109851.

Additional references

Moilanen, A, Wilson, K, Possingham, H, (2009) Spatial conservation prioritization: quantitative methods and computational tools. Oxford University Press, Oxford.

5.3 Identifying species at risk

Small number of papers (14 papers)	Low bird bias (86%)	Low geographic bias (Europe 43%)

When considering potential development or expansion of wind energy facilities across a region, risk assessment-type approaches can be used to identify species that may be most impacted by collisions. The resulting assessments and species lists can be used in strategic planning (see Section 5.2), should inform the design of both pre- (Section 5) and post-construction monitoring programs (Section 7), and can also help identify potentially suitable mitigation actions (Section 6).

A first step is to identify species that occupy the area of interest. This can be achieved by compiling lists based on occupancy records, range maps, and predictions from distribution models (e.g. Balotari-Chiebao et al. 2021; Reid et al. 2023). At this step it is also important to consider differing habitat requirements; for example, some species may be present in an area in high numbers but only during part of the year, as is the case during migrations, or may use different types of habitats for roosting, breeding, and foraging (Beston et al. 2016). In some cases, information about the types of species that collide with turbines most frequently in the area of interest or in similar systems (from post-construction fatality monitoring) can also be used to inform assessments (see Section 4.1.1 for discussion of some of these patterns).

Though risk assessment processes take many forms with varying degrees of complexity, they typically consider 'risk' to be a function of both the likelihood, and the consequence of an event occurring (Lumsden et al. 2019). When considering wind energy, this means that for each bird and bat species occurring in a region of interest, one must consider how likely each is to collide with a turbine, and what the consequences of that would be for the species' population. We identified five examples in the literature where assessments had been conducted to identify bird and bat species most at risk from collisions with wind turbines (Table 7). In Victoria, a transparent, repeatable approach has also been developed to identify Species of Concern for wind energy (Lumsden et al. 2019), so this is also included in Table 7.

The criteria used to assess likelihood of collisions varied between studies, but typically captured some measure of flight height (and whether this overlapped with the relevant rotor-swept area), wing morphology (as a proxy for manoeuvrability, and in turn the species' ability to avoid turbines), and the habitats they use (and whether these intersected with the places where turbines would be/are typically constructed). In areas where wind energy facilities are already present, data from post-construction fatality monitoring can also be used to inform likelihood criteria. For example, Beston et al. (2016) included a 'Proportion of Fatalities due to Turbines (FT)' measure, which they calculated as , where n is the number of individuals killed by turbines annually, s is the adult survival rate, and N is the total population size. Consequence criteria were also varied between studies, but captured measures of the size of the species population and how concentrated it was in particular areas, breeding traits such as fecundity and age at first reproduction, and all studies incorporated the species' conservation status (at the state, national or international level).

The studies we identified were essentially desktop assessments drawing on information and knowledge from experts, existing datasets and databases, the primary scientific literature and reference material to score each species against likelihood and consequence criteria. However, it may be necessary to also collect fit-for-purpose field data in areas where these are lacking or inadequate. For example, in their assessment of the potential impacts of offshore wind development on Australian bird species (which also included inshore areas, hence its inclusion in our review), Reid et al. (2023) emphasised that empirical data were lacking for the flight heights of many of the species they were assessing. Given this is likely to be the case in many areas, if possible, assessments should also account for and incorporate measures of uncertainty, and consider adopting precautionary principals (i.e. consider a species to be high risk until more precise estimates or data become available).

Knowledge gaps for Victoria

8. Victoria's Species of Concern list has been developed using an expert elicitation process. For most species, there is a lack of empirical data for criteria used in this assessment, e.g. flight heights.
9. Further information is needed on wind energy-related mortality rates, and drivers of mortality in general, for each threatened species, to enable this to be incorporated into the Species of Concern criteria.

Key references

Beston, JA, Diffendorfer, JE, Loss, SR, Johnson, DH (2016) Prioritizing avian species for their risk of population-level consequences from wind energy development. *PLoS ONE* **11**, e0150813.

Reid, K, Baker, GB, Woehler, EJ (2023) An ecological risk assessment for the impacts of offshore wind farms on birds in Australia. *Austral Ecology* **48**, 418-439.

Additional references

Lumsden, LF, Moloney, P, and Smales, I (2019) Developing a science-based approach to defining key species of birds and bats of concern for wind farm developments in Victoria. Arthur Rylah Institute for Environmental Research Technical Report Series No. 301. Department of Environment, Land, Water and Planning, Heidelberg, Victoria.

Table 7. Studies that developed wind energy risk assessment processes to identify bird and bat species most likely to be impacted.
Parameters listed in square brackets '[ ]' form part of the formulae used to calculate indices (Indirect Risk Index, Fatality Risk Index and Fatalities due to Turbines).

Study	Country	Method	No. species assessed	Likelihood factors	Consequence factors
Balotari-Chiebao et al. (2021)	Finland	Data synthesis Literature review	214 birds	Indirect Risk Index [percent of a species' population living near turbines, number of habitat types used]	Fatality Risk Index [percent of a species' population living near wind turbines, maternity, fecundity, age at first reproduction] Conservation status (national list)
Beston et al. (2016)	USA	Data synthesis Literature review	428 birds	Fatalities due to Turbines Index [number of individuals killed by turbines annually, adult survival rate, total population size] Indirect Risk Index [percent of a species' population living near turbines, number of habitat types used]	Fatality Risk Index [percent of a species population living near wind turbines, maternity, fecundity, age at first reproduction] Conservation status (percentage of states where listed)
Lumsden et al. (2019)	Australia (Victoria)	Expert assessment	159 birds 7 bats	Flight height Habitat preferences	Population size Population concentration Population resilience (dispersal, fecundity, generation time) Conservation status (state list)
Morkune et al. (2020)	Lithuania	Data synthesis Expert assessment	69 birds 17 bats	N/A	Sensitivity to effects of wind power during the breeding period Population concentration Conservation status (national and IUCN list)
Noguera et al. (2010)	Spain	Literature review Data synthesis	9 raptors	Flight height Flight type Wing loading Aspect ratio Seasonality	Population size Breeding capacity (clutch size) Conservation status (Birdlife International)
Reid et al. (2023)	Australia (offshore and inshore)	Literature review Expert assessment	273 birds	Flight height Wing loading Habitat specialisation	Generation time Conservation status (Action Plan for Australian Birds)

5.4 Turbine design and siting considerations

Moderate number of papers (44 papers)	No taxonomic bias	Low geographic bias (Europe 45%)

Ideally, the specifics about how many turbines will be installed, what make and model they will be, and where they will be located within the footprint of a wind energy facility (siting) will be known *a priori* and factored into pre-construction survey design and risk assessments. However, it is worth noting that this is not always the case; these details often change throughout the planning and approvals process, and the pre-construction field surveys can inform the micrositing of the turbines.

5.4.1 Turbine design

As wind energy technology has progressed, turbines have become larger; they are taller, the rotor-swept area (RSA) is larger, and each individual turbine has the capacity to produce more power (Figure 6). For example, Anderson et al. (2022) highlighted that from 2006–2019, turbines in Ontario (Canada) increased 42% in size, from 120 to 170 m maximum blade tip height. This has created concern that the installation of larger turbines will come with increased fatality rates, as the RSAs will intersect with a greater proportion of bird and bat flight paths, and it will become harder for individuals to avoid collision.

There is substantial evidence that larger turbines are in fact associated with higher fatality rates for both birds and bats. We identified 13 studies from across North America and Europe that empirically assessed the relationship between turbine size (based on metrics including maximum blade tip height, RSA, rated capacity, ground clearance and hub height), and estimated fatalities, by collecting or synthesising data from across 5–59 individual facilities each (Table 8). Of these, 10 found a significant positive relationship between at least one measure of turbine size and fatalities, though there was no obvious pattern as to which metric was most consistently informative. Hub height was the most frequently tested parameter, and significant relationships were found in 3/7 studies for bats and 1/3 studies for birds. Rated capacity and rotor diameter were tested in five studies each, with each being found to be significant in two studies. The two parameters relating to the extremes of the blade tips (maximum blade tip height and ground clearance) were the least frequently assessed, and were included in a maximum of two studies for each taxon. Hence, it is very difficult to draw conclusions about the specific aspects of turbine design that might have the greatest influence on fatalities, particularly given that all of these measures tend to be correlated. For example, higher-capacity turbines will tend to have greater hub heights, maximum blade tips heights, and ground clearance, and also larger rotor diameters than lower-rated turbines (see Figure 6).

There are some important issues to bear in mind when considering these findings. First, small, old-model turbines installed in countries such as the USA were linked to very high fatality rates of some raptors (Smallwood and Karas 2009) because they presented perching opportunities. This subsequently led to efforts to repower facilities where they were installed, which essentially involved replacing older turbines with newer, increased-capacity ones as a mitigation measure. Second, if larger, higher-capacity turbines are installed then fewer individual turbines will be needed to generate the same amount of power. If these larger turbines are sited in a way that minimises collisions (see Section 5.4.2), then overall fatality rates could be reduced.

A meta-analysis conducted by Thaxter et al. (2017) demonstrates this latter point. The authors built a model relating species collision rates to traits and site-specific information, including turbine capacity. They then used the model to generate predictions of fatalities across a hypothetical 10 MW wind energy facility, where that power could be generated by many smaller turbines or fewer larger ones. While their model did indicate that larger turbines were associated with increased collision rates, their predictions showed that scenarios with a larger number of small turbines resulted in overall higher predicted mortality rates. There was an exception to this finding for bats, where scenarios with the largest turbines (2.5 MW) did in fact seem to result in a small increase in bat fatalities relative to the amount of energy generated, and this should be borne in mind considering that turbines currently being installed across Victoria are typically over 6 MW in capacity. The relative impact of changes in turbine height and RSA will ultimately depend on the flight behaviour of the bird and bat fauna where turbines are being installed; species that fly higher will be worse-off, while those that fly lower may benefit if minimum blade tip heights increase.

Finally, as highlighted in Section 5.5.4, it is not the rated or 'nameplate' (manufacturer's rating) capacity of a turbine that influences fatalities *per se*, but the amount of energy that is generated based on the periods that it is operational, and there can be many factors that influence whether or not a turbine is operational. Huso et al. (2021) found that once they had standardised recorded fatality rates by the amount of energy produced for each turbine (essentially hours of operation multiplied by capacity), that fatality rates of the smallest 108 kW turbines in their study were on a par with the largest (2.5 MW), so fatalities were relatively constant per unit energy produced. For this reason, transparency around operational information is critical to the analysis and interpretation of post-construction monitoring data.

Knowledge gaps for Victoria

10. Most (although not all) of the turbines currently being installed in Victoria are much larger (over 6 MW) than models that have been incorporated into existing international reviews and syntheses (0.1–4.2 MW). The potential trade-off between energy generation and fatality rates for Victorian species, in light of their flight heights and behaviours, warrant careful assessment and analysis.
11. In Victoria, most of the earlier-constructed turbines, which are smaller and have a lower RSA, are no longer being monitored for collisions. When post-construction mortality monitoring was conducted for these facilities, surveys were not of today's standards, so our understanding of the ongoing mortality rates at these facilities is unknown.

Key references

Anderson, AM, Jardine, CB, Zimmerling, JR, Baerwald, EF, Davy, CM (2022) Effects of turbine height and cut-in speed on bat and swallow fatalities at wind energy facilities. *FACETS* **7**, 1281-1297.

Huso, M, Conkling, T, Dalthorp, D, Davis, M, Smith, H, Fesnock, A, Katzner, T (2021) Relative energy production determines effect of repowering on wildlife mortality at wind energy facilities. *Journal of Applied Ecology* **58**, 1284-1290.

Thaxter, CB, Buchanan, GM, Carr, J, Butchart, SHM, Newbold, T, Green, RE, Tobias, JA, Foden, WB, O'Brien, S, Pearce-Higgins, JW (2017) Bird and bat species' global vulnerability to collision mortality at wind farms revealed through a trait-based assessment. *Proceedings of the Royal Society B-Biological Sciences* **284**, 20170829.

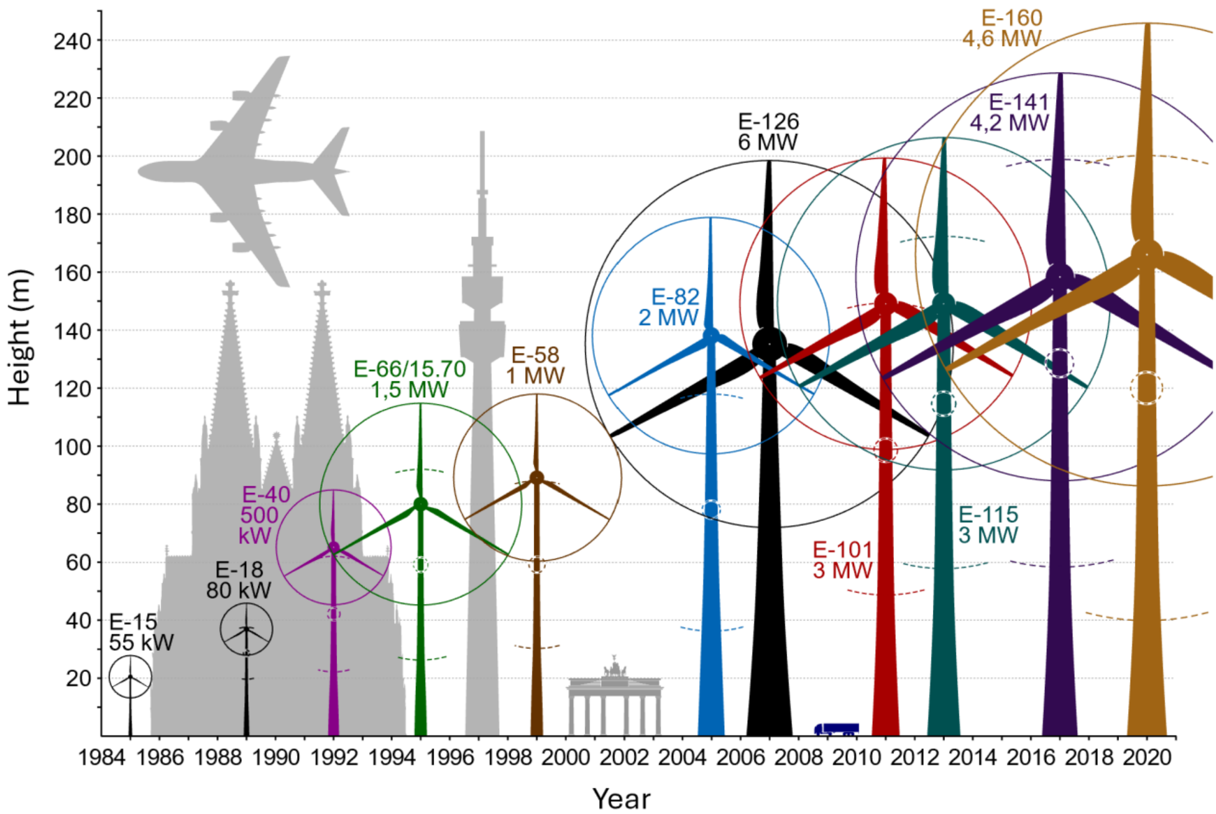

Figure 6. Comparison of the different heights and powers of Enercon model turbines, and the year they were introduced, against familiar German landmarks.

Landmarks are (left to right): Airbus A380, Cologne Cathedral, Florianturm, Brandenburg Gate, and a truck with a 40′ISO-Container. Blades can be mounted on towers of different heights; the tallest variant is shown in solid and the shortest as dashed lines. Image: Jahobr, CC0, via Wikimedia Commons.

Table 8. Summary of studies assessing the impact of turbine design on the number of recorded bat and bird fatalities.

Turbine details – NR: Not reported, μ: mean or average, MW: megawatt. Study type – FS: Field study, LR: Literature review, DS: Data synthesis. The other parameters tested were not significant in predicting fatalities, over and above the parameters that were influential. Studies marked with an asterisk (*) are repeated in the table (i.e. appear once for bats, and once for birds), and turbine details marked with a question mark ('?') indicates that there is uncertainty about the value reported because of unclear reporting. Cells containing 'NA' indicates that no parameters from the study are suited to that column (i.e. either did, or did not predict fatalities). Studies are sorted by taxon, and then reference.

Study	Turbine details	Taxon	Species	Country/ region	Study type	No. facilities	Parameters that predicted fatalities	Other tested parameters that did not predict fatalities
Anderson et al. (2022)*	21–110 m hub height 15–93 m rotor diameter 0.33–3 MW	Bats	Five species	Canada	DS	59	Max blade height (MBH)	NA
Baerwald and Barclay (2009)	50–84 m hub height 47–80 m rotor diameter NR MW	Bats	Hoary Bat, Silver-haired Bat	Canada	FS	8	Hub height	NA
Davy et al. (2021)	21–110 m hub height 15–93 m rotor diameter 0.33–3 MW	Bats	Four species	Canada	DS	48	NA	Hub height
Garvin et al. (2024)*	55–110 m hub height 61–136 m rotor diameter 1–3.6 MW	Bats	Hoary Bat	North America	DS	44	Ground clearance	Hub height Max blade height (MBH) Rotor diameter Rated capacity of turbine (MW)
Georgiakakis et al. (2012)	44–60 m hub height 52–90 m rotor diameter 0.8–2 MW	Bats	Eight species combined	Greece	FS	9	Hub height	Rotor diameter
Huso et al. (2021)*	33–80 m hub height 19–93 m rotor diameter 0.1–2.5 MW	Bats	All species combined	USA	FS	5	Energy generated	Rated capacity of turbine (MW)
Moustakas et al. (2023)	μ 60 m hub height μ 54 m rotor diameter 0.8–4.2 MW	Bats	All species combined	Greece	DS	9	Rated capacity of turbine (MW)	Hub height Rotor diameter
Rydell et al. (2010)	24–98 m hub height 60–90 m rotor diameter (?) NR MW	Bats	All species combined	Europe	LR	37	Hub height Rotor diameter	Ground clearance
Thompson et al. (2017)	NR hub height NR rotor diameter NR MW	Bats	All species combined	USA, Canada	LR	40	NA	Hub height

Study	Turbine details	Taxon	Species	Country/ region	Study type	No. facilities	Parameters that predicted fatalities	Other tested parameters that did not predict fatalities
Anderson et al. (2022)*	21–110 m hub height 15–93 m rotor diameter 0.33–3 MW	Birds	Four swallow species	Canada	DS	59	Max blade height (MBH)	NA
Carrete et al. (2012)	NR hub height (categorical?) NR rotor diameter NR MW	Birds	Griffon Vulture	Spain	DS	34	NA	Turbine height
Garvin et al. (2024)	55–110 m hub height 61–136 m rotor diameter 1–3.6 MW	Birds	Horned Lark, Red-tailed Hawk	North America	DS	44	Rotor diameter Ground clearance Rated capacity of turbine (MW)	Hub height Max blade height (MBH)
Huso et al. (2021)*	33–80 m hub height 19–93 m rotor diameter 0.1–2.5 MW	Birds	All species combined	USA	FS	5	Energy generated	Rated capacity of turbine (MW)
Loss et al. (2013)	36–80 m hub height NR rotor diameter 0.18–3 MW	Birds	All species combined	USA	LR	58	Hub height	NA

5.4.2 Turbine siting

Decisions about where turbines are sited within a facility are informed by a range of technical, logistical, and policy-related considerations, but they also have the potential to influence the magnitude of biodiversity impacts. Siting factors that might make a turbine more or less likely to be associated with collisions include the composition of the surrounding landscape, such as the proportion covered by woody vegetation, distance to important habitat features such as nesting and foraging sites, caves, waterways and refugia, topography, wind patterns, and elevation (Table 9). Because of the recognised relationship between habitat features and the flight activity (and in turn collision risk) for certain species, 'buffers' (or minimum distances between where turbines are located and species habitat) are sometimes recommended as an avoidance or mitigation measure (discussed in 6.2 below).

As noted by Masden and Cook (2016), collision rates observed at any one turbine will be a unique function of the specifics of the site, the species, and the turbine itself (see Section 5.4.1) and this may explain why we identified relatively little consistency in the siting factors that had the strongest influence on observed fatalities in the studies that we reviewed (Table 9). An exception to this is that, as noted elsewhere, in the absence of thermal currents, raptors are reliant on orographic uplift to gain altitude, so turbines sited on ridges in particular create collision hazards (reviewed in Estellés-Domingo and López-López in press). Others have also cautioned about this for bats, which may rely on ridges for navigation or foraging (Roeleke et al. 2018).

There is also evidence that micrositing decisions about the placement of individual turbines can have a strong influence on the impact of an entire facility, and that in some cases one or two turbines can account for a large proportion of the fatalities recorded. For example, in their study of Griffon Vultures in southern Spain, de Lucas et al. (2012) found that there were highly significant differences in mortality rates recorded between individual wind turbines (296 in total). The authors suggest that local conditions, such as small-scale topographical features and wind patterns were responsible for this. This meant that by selectively stopping 10% of the most problematic turbines, they were able to reduce fatality rates by 55%. Heuck et al. (2019) also looked at broad-scale mortality patterns, this time for the White-tailed Eagle in northern Germany. They found that while turbine density was an important predictor of fatalities that acted synergistically with habitat suitability, such that a disproportionately high number of fatalities occurred in areas where there were high turbine densities and also high predicted habitat suitability (based on a species distribution model).

Siting decisions for individual turbines are also important for bats. Georgiakakis et al. (2012) found that 15% of the turbines in their study in Greece were responsible for 51% of bat fatalities, while Jameson and Willis (2012) found that turbines at the north-west of a facility they studied in Canada were associated with the majority of recorded Hoary Bat and Eastern Red Bat fatalities. However, it remains unclear whether curtailing or decommissioning particularly problematic turbines in general (for both birds and bats) can effectively reduce mortalities, or if doing so will simply displace the impacts to other turbines.

Knowledge gaps for Victoria

12. Further information is required on the habitat and landscape features, turbine characteristics, turbine layout configuration and climatic conditions that most accurately predict fatality rates for bird and bat Species of Concern. This information would be valuable to inform siting decisions.
13. Little is known about whether mitigation measures applied at individual turbines or groups of turbines associated with the highest fatality rates can effectively reduce overall fatalities, or if the impacts will simply be displaced to elsewhere in the facility.

Key references

de Lucas, M, Ferrer, M, Bechard, MJ, Muñoz, AR (2012) Griffon Vulture mortality at wind farms in southern Spain: distribution of fatalities and active mitigation measures. *Biological Conservation* **147**, 184-189.

Heuck, C, Herrmann, C, Levers, C, Leitao, PJ, Krone, O, Brandl, R, Albrecht, J (2019) Wind turbines in high quality habitat cause disproportionate increases in collision mortality of the White-tailed Eagle. *Biological Conservation* **236**, 44-51.

Thompson, M, Beston, JA, Etterson, M, Diffendorfer, JE, Loss, SR (2017) Factors associated with bat mortality at wind energy facilities in the United States. *Biological Conservation* **215**, 241-245.

Table 9. Summary of studies assessing the impact of turbine siting on the number of recorded bat and bird fatalities.

Study type – FS: Field study, LR: Literature review, DS: Data synthesis. Parameters – LCC: Land cover composition. Cells containing 'NA' indicates that no parameters from the study are suited to that column (i.e. either did, or did not predict fatalities). Asterisk (*) indicates separate rows for bats/birds from one study. Studies are sorted by taxon, and then reference.

Study	Taxon	Species	Country/ region	Study type	No. facilities	Parameters that predicted fatalities	Other tested parameters that did not predict fatalities
Baerwald and Barclay (2011)	Bats	Hoary Bat, Silver-haired Bat	Canada	FS	1	Position in site (north)	Location in row
Bennett and Hale (2018)	Bats	Six species	USA	FS	1	NA	Distance to land use
Bolívar-Cimé et al. (2016)	Bats	21 species	Mexico	FS	1	LCC (secondary vegetation) LCC (fields: inverse relationship)	NA
Davy et al. (2021)	Bats	Four species	Canada	DS	48	LCC (woodlots) Elevation	Distance to topographic features
do Amaral et al. (2020)	Bats	Six species	Brazil	FS	1	Distance to urban areas (inverse relationship)	LCC
Georgiakakis et al. (2012)	Bats	All species combined	Greece	FS	9	Elevation	NA
Moustakas et al. (2023)	Bats	All species combined	Greece	DS	9	Distance from water Aspect Slope LCC (natural areas)	Altitude or elevation
Piorkowski and O'Connell (2010)*	Bats	Seven species	USA	FS	1	Topography composition (ravines)	NA
Rydell et al. (2010)	Bats	All species combined	Europe	LR	37	NA	LCC Altitude or elevation
Thompson et al. (2017)	Bats	All species combined	USA, Canada	LR	40	LCC (grasslands: inverse relationship)	Altitude or elevation Region
Bose et al. (2018)	Birds	Five groups: buntings, larks, raptors, pigeons, crows	Germany	DS	69	Distance to habitat features (forests, fields, and water)	NA
Bose et al. (2020b)	Birds	Common buzzard	Germany	DS	69	Distance to habitat features (watercourses and grasslands)	NA
Carrete et al. (2012)	Birds	Griffon Vulture	Spain	DS	34	Distance to colony/roost Aggregations at colony/roost	Altitude or elevation
Loss et al. (2013)	Birds	All species combined	USA	LR	58	Region (California, then the East, West, then Plains regions)	NA
Piorkowski and O'Connell (2010)*	Birds	Seven species	USA	FS	1	NA	Land cover composition Topography composition

5.5 Field assessments

Very large number of papers (135)	Low bird bias (75%)	Low geographic bias (Europe 41%)

There is a large and well-established body of literature on field survey techniques used for birds and bats in general, and it is not our intention here to provide either a comprehensive overview of this, or specific guidance on which approaches to use and when. Instead, we provide a summary of techniques used at proposed wind energy facilities during pre-construction risk assessments, and specifically refer to studies that comment on or test the efficacy of different approaches.

5.5.1 Field surveys for bats

Surveys using ultrasonic bat detectors (acoustic recorders) are commonly conducted as part of pre-construction environmental assessments. Many microbat species produce unique echolocation calls that can be used to identify calls to the species or genus level, or to allow calls to be assigned to a 'species complex' (groups of species with similar call characteristics). Acoustic surveys do not allow for estimation of the abundance of microbats, as the number of individuals producing the calls cannot be determined, but instead provide a measure of relative activity.

Most acoustic surveys conducted during pre-construction assessments are undertaken at ground level, but by mounting bat detectors at various heights, vertical activity profiles can be generated for different species or species groups (Collins et al. 2009; Roemer et al. 2017). For example, Wellig et al. (2018) compared windspeed data with the relative activity level of microbat species recorded at different heights, up to 65 m. While they found that there were differences in these vertical activity profiles between species, there was also an overall drop in bat activity with increasing wind speed such that there was a low (<5%) probability of activity at windspeeds above 5.4 m/s. A limitation in these types of studies is that only a proportion of the RSA is monitored (e.g. up to 30–65 m). Roemer et al. (2017) found a correlation between bat activity recorded at meteorological (MET) masts at heights of 20–45 m, and apparent species fatality risk; however, this was assessed through a 'collision susceptibility index' derived from the broader literature rather than actual observed fatalities at the sites.

Otherwise, our review did not find strong evidence for the effectiveness of pre-construction acoustic surveys in predicting bat collision risk (see Section 5.5.4), and for this reason some authors have questioned their value, given the considerable time and expense involved (Lintott et al. 2016). However, several studies have shown that *post*-construction acoustic activity recorded at turbines is a strong predictor of bat fatality rates (Peterson et al. 2021, Behr et al. 2023), which is useful for informing multi-factor curtailment approaches and algorithms (see Section 6.1.2). Voigt et al. (2022) found that both the spatial distribution of bats within the RSA, and the coverage of the RSA by acoustic detectors will influence the accuracy of predicted fatalities, and should be accounted for. For example, they showed that a concentration of bat passes at the edge or at the top of the RSA caused an underestimation of bat activity, but this effect decreased with increasing coverage by acoustic monitoring devices.

Overall, there are several limitations that influence the detection of bat calls by acoustic monitoring devices, which may subsequently influence predictions of collision risk based on bat call data. This includes environmental noise (and particularly wind, given the environments that the devices are being deployed in), geometric and atmospheric attenuation of calls (meaning that the sound pressure of pulses reduces as they move away from the bat), the directionality and sensitivity of microphones, and coverage of the RSA by the detectors (Voigt et al. 2021). The calls produced by bats that echolocate at higher frequencies (e.g. 40 kHz or higher) attenuate faster than bats that call at lower frequencies (e.g. 20 kHz), so can only be detected over shorter distances and are more likely to be drowned out by ambient noise from wind, rain and insects (Voigt et al. 2021). Consequently, acoustic data will underrepresent true bat activity on site. Because of these factors, and the limited amount of airspace that a detector can effectively survey (typically only approximately 5–50 m from the microphone), substantial survey effort during appropriate periods (i.e. warm weather, and/or during migration) may be required to have confidence in conclusions regarding species occupancy and relative activity (Richardson et al. 2019).

Gration (2011) proposed that radar could be used in pre-construction bat surveys in Victoria, as it is capable of clearly recording bat flight paths as well as vertical profiles (see also Section 5.5.3). A limitation of this method is the difficulty in distinguishing between species. However, Gration (2011) suggests that wingbeat (frequency and amplitude), speed and size could be used to differentiate between species in radar observations, or by using it in conjunction with acoustic detectors to identify species echolocation calls. Other alternatives to acoustic surveys for monitoring bat activity at proposed or operational wind farms include thermal and infrared cameras. Huzzen et al. (2020) deployed thermal and infrared cameras to survey bat behaviour at turbines. They found that while thermal cameras captured approximately 34% more flying individuals than the infrared, the use of infrared enabled better identification of specific behaviours and interactions with the turbines. Therefore, they recommended using both technologies together for surveying bat activity. Both camera types were best positioned 2 m from the turbine tower (facing upwards). Radar, thermal and infrared cameras may also be helpful for surveying the activity of flying-foxes. Acoustic surveys are not used for flying-foxes, because they do not echolocate, and although they are vocal when roosting or foraging in trees they do not vocalise while in flight which is when they are at risk from turbine collisions.

An important consideration when relating international literature to the Victorian context for acoustic surveys, whether for the pre- or post-construction phase, is that many Victorian microbats have overlapping echolocation calls. This means that it is not always possible to reliably identify calls to the species level, depending on the approach used for identification, the availability of locally-sourced reference calls, as well as the study region. For example, calls of the Critically Endangered Southern Bent-winged Bat (*Miniopterus orianae bassanii*) in south-west Victoria can overlap in their shape and frequency with calls of the Chocolate Wattled Bat (*Chalinolobus morio*), Little Forest Bat (*Vespadelus vulturnus*) and Southern Forest Bat *(V. regulus*). As a result, it is often difficult to distinguish the calls of these species, and therefore not all Southern Bent-wing Bat calls can be confidently identified. Given the limitations of acoustic surveys, precautionary approaches to risk assessments are prudent, and validation of acoustic monitoring through post-construction activity and mortality monitoring on-site is important for obtaining an accurate understanding of impacts.

Knowledge gaps for Victoria

14. There is little information available in Victoria on how bat activity levels at individual facilities differ between the pre- and post-construction periods.
15. Further information is required on the extent and type of pre-construction survey effort required to fully understand bat activity across a site, and how this varies seasonally, between years, between species, and at various heights above the ground.
16. It is unclear whether acoustic data recorded at heights (i.e. on MET masts) provides suitable data for pre-construction assessments for Victorian species.
17. Further research is required (including using new AI approaches), to improve the accuracy of echolocation call identification of Victorian bats, especially for those species with overlapping call characteristics.

Key references

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Voigt, CC, Russo, D, Runkel, V, Goerlitz, HR (2021) Limitations of acoustic monitoring at wind turbines to evaluate fatality risk of bats. *Mammal Review* **51**, 559-570.

5.5.2 Utilisation surveys for birds

Pre-construction bird utilisation surveys (sometimes abbreviated to BUS) generally fall into two main categories: point-count surveys and targeted surveys (Katzner et al. 2016). Point count surveys are used to identify species using the site, and to estimate flight heights, and the number and density of flights (flux) to inform risk assessments (including collision risk models if they are being used, see Section 5.6.1). Targeted surveys for specific species can include recording occupancy and flight path mapping, and are used to identify important habitat features (e.g. nest trees, roost sites, flock aggregations), and times of risk (such as migration periods). Data collected in bird utilisation surveys are also used to inform turbine siting to avoid and mitigate impacts.

Most of the papers in our review that evaluated pre-construction bird surveys focussed on point count surveys, and survey technique for raptors (and particularly Bald and Golden Eagles), which are often the focus of EIAs because of the high collision rates recorded across Europe and the USA (Katzner et al. 2016; Conkling et al. 2022). For example, a raptor-specific review of studies from across 321 facilities and 12 different countries found that point counts were the most commonly-used survey method; however, other approaches including behavioural observations, migration surveys, line transects, nest searches, sensitive species surveys, and even prey surveys were also used, and there was a lack of standardisation in general (Conkling et al. 2022). The authors also noted that just 27% of projects appeared to have undertaken monitoring over both the pre- and post-construction phases.

A range of temporal, spatial, and species-specific factors can influence the accuracy of surveys for estimating bird use at proposed wind energy sites. The timing of surveys can have a strong influence on inference about risk, because the seasonal timing selected specifically for some species may not be appropriate for capturing site use by other species at risk (Katzner et al. 2016). Skipper et al. (2017) found that detection of Golden Eagles (based on point counts) varied annually, suggesting that using survey data from a single year to estimate risk is likely to be unreliable because of the variation in eagle use, distribution, and activity patterns. Overall, increased spatial and temporal sampling reduced the rate of failed detections for the species (i.e. not detecting it when it was present), and bias for these surveys. To demonstrate this, Sur et al. (2018) used telemetry data from Golden Eagles and simulated point counts to understand the relationship between detections of eagles from point counts and actual flight activity. They found that random sampling (compared to systematic or stratified) and increasing spatial coverage across the site decreased sampling error (i.e. over- or under-estimation of activity). Likewise, for Bald and Golden Eagles, simulated surveys from a monitoring dataset from 1990–2014 across 22 sites indicated that full-day counts, undertaken weekly during the peak migration period, would be most effective at estimating eagle site use (Chabot and Slater 2018). In contrast, four-hour counts conducted on a weekly basis were ineffective at estimating eagle migration passage rates.

Some authors have also investigated and compared alternatives to visual surveys. Becker et al. (2020) compared results from traditional visual observations with an avian radar system for recording the activity of Cape Vultures at a proposed wind farm site in South Africa. They found that there was increasing inaccuracy of visual judgements of human observers with increasing flight height and distance from the observer, with disparities between visual and radar observations ranging up to 1,059 m. Ruiz-Gutierrez et al. (2021) validated the use of citizen science (eBird) data for Bald Eagles for defining low-risk areas for wind energy developments. They found that the year-round, weekly estimates of relative abundance, mapped at the 50th quantile of relative abundance values, captured 91–100% of high-use locations, nests and midwinter roosts in areas of exposure risk. This research ultimately led to a policy decision by the US Fish and Wildlife Service to use eBird data to assess collision risk as part of the permitting process.

Finally, Largey et al. (2021) compared four empirical measurement methods that can be used to improve the estimation of bird flight parameters to understand collision risk: radar, telemetry, ornithodolite (binoculars with an inbuilt laser rangefinder, inclinometer and digital magnetic compass) and LiDAR (a remote sensing method using lasers). Based on their evaluation, they propose a framework for collecting flight height data as part of a general EIA process using these four methods, depending on the target species and focal landscape. They acknowledge that while these methods may not be the most cost-effective option, use of the overall framework would improve accuracy and provide more reliable data.

While point counts offer a repeatable approach that can be standardised (e.g. through the adoption of guidelines), additional targeted surveys will likely be required for a detailed understanding of activity and habitat features for specific species at risk. Overall, evidence suggests that bird flight heights may be more accurately estimated using sensor technologies (such radar, LiDAR, and ornithodolites) rather than human observation only, and that increased survey effort and temporal and spatial coverage will improve the accuracy of activity estimates.

Knowledge gaps for Victoria

18. It is unclear what the most appropriate survey methods are for Victorian bird Species of Concern (e.g. swifts, migratory shorebirds, cranes, bitterns, geese, and migratory passerines and parrots), given their varying breeding, non-breeding, foraging and movement strategies that differ from well-studied raptors. Point-count surveys and targeted surveys don’t typically involve collecting data on flights for these species, so impacts and collision risk cannot be reliably predicted. Highly mobile and dispersive species are particularly poorly understood.
19. The level of survey effort required, and the most appropriate time of the year or day to accurately estimate activity for Victorian bird species of concern is unclear.

Key references

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Largey, N, Cook, ASCP, Thaxter, CB, McCluskie, A, Stokke, BG, Wilson, B, Masden, EA (2021) Methods to quantify avian airspace use in relation to wind energy development. *Ibis* **163**, 747-764.

5.5.3 Radar and telemetry to assess flight heights, migration and movement routes

In ecology, 'telemetry' is a generic term used to describe devices and technology such as GPS and satellite transmitters, and VHF (radiotracking) and microchip (PIT) tags, that are used to track animal movements. These devices are attached to individuals using glue, collars, leg bands or harnesses, or they are injected subcutaneously. To collect this data, animals have to be actively tracked (VHF) or recaptured (to download the data off GPS devices or to read PIT tags), or the device may be capable of remotely transmitting information via satellites, phone networks or PIT tag scanners. For GPS, phone network, and satellite transmitters the researcher decides how often the device records a location or takes a 'fix'. If fixes are taken more frequently then higher-resolution data are obtained, but the device will record over a relatively shorter time frame because of trade-offs with data storage capabilities and battery life.

Radar is another technology that is sometimes used as a source for movement information. Radars use radio waves to detect objects, and in the case of wind energy there are two types of systems that may be employed, with different applications: long-range networks, and mobile or on-site installations (Hüppo et al. 2019). Large-scale long-range radar networks (both marine and terrestrial) have been established in many areas globally to collect meteorological data and track the movements of marine vessels or aircraft. Researchers have taken advantage of this existing infrastructure and associated data processing workflows to detect flying animal migratory movements, and also to assess the timing and direction of departures and arrivals for large aggregations of birds or bats. For example, in Australia 15 major flying-fox roosts along the east coast are currently monitored using weather radar, which allows for estimation of the number of bats in the roost and also the timing of their departure each night (Welbergen and Meade 2025). Recent work using weather radar has also identified migratory patterns of Australian bird species (Shi et al. 2024). One key limitation of this approach is that that data coverage is limited to areas within 200 km of a radar, and in Australia these are biased towards more densely populated coastal areas (BOM 2025). In contrast, smaller mobile radars can be installed on-site in areas where wind energy facilities are planned or have already been constructed, and allow researchers to track the movement of individual animals or small groups (e.g. Jenkins et al. 2018). However, data from both long-range and on-site radar typically need to be paired with other methods (e.g. thermal or infrared cameras, or bat detectors) to confirm the species identity of observed targets (Hüppop et al. 2019).

The information and data obtained using telemetry and radar technologies can be highly valuable in both assessing the risk posed by wind energy facilities, and also in informing mitigation measures. The majority of the papers that we identified in our review reporting on movement studies (57/84) used a form of GPS telemetry (Table 10). There have been rapid technological developments in this field and progressive miniaturisation, allowing tracking of a wider range of species, including small migratory passerines and shorebirds. Nonetheless, there are still limits around the weight of devices that can be attached to individuals (relative to their body mass), and concerns about the potential for increased wind drag and reduce flight efficiency (Soulsbury et al. 2020). Consequently, there remains a lower mass limit for the species that telemetry can feasibly be used for. For this reason, it is not surprising that 52 of the 57 GPS papers that we reviewed focussed on birds, and typically large-bodied species (e.g. eagles, vultures, cranes, storks, grouse and ducks), rather than small birds or microbats. Presently, the smallest GPS units weigh around 1 g, and they are typically paired with a VHF tag so they can be located once they are shed by an animal. This means that the lower mass limit of an animal that they can be attached to is around 15 g for the short-term (i.e. equivalent to 10% of its body weight) or 30 g for longer-term studies (5% of its body weight). For context, some microbat species that collide frequently with turbines in Victoria include Gould's Wattled Bats (*Chalinolobus gouldii*, 10–20 g), various Forest bat species (*Vespadelus* sp., 3–8 g), and Southern Free-tailed Bats (*Ozimops planiceps*, 6–13 g), while some of the small passerines that collide with turbines in Victoria are the Australian Pipit (*Anthus australis*, 20–30 g), Striated Pardalote (*Pardalotus striatus*, 9–15 g), and Grey Fantail (*Rhipidura albiscapa*, 7–10 g).

There are three broad types of applications relevant to pre-construction risk assessment for wind energy impacts that have been described and tested in the literature using telemetry and radar technologies (Table 10). The first of these applications focuses on large-scale migratory movements, which have a seasonal, generally predictable and cyclical pattern. For example, Ellis et al. (2022) used six years of GPS tracking data on migratory movements of 56 Whooping Cranes across the USA Great Plains to make recommendations about suitable siting locations for future wind energy projects. These sites aimed to avoid areas predicted to become increasingly important for the cranes under drought conditions. Gauld et al. (2021) combined tracking data from 27 species susceptible to collision in Europe and North Africa (including Mallards, geese, raptors, owls, Common Cranes, and Eurasian Spoonbills) to assess sensitivity and vulnerability to turbine and transmission line collisions. Knowledge of patterns in the daily and seasonal timing of migratory movements can also help to ensure that timed mitigations, such as seasonal turbine curtailment (see Section 6.1), are tailored to minimise both losses of energy generation, and fatalities. For example, Abbott et al. (2011) used ten years of weather radar data to model temporal peaks in the arrival of migratory birds at stopover sites in the Gulf of Mexico, and suggested that these could inform temporary turbine shutdown periods.

Table 10. Cases where radar and telemetry methods have been used to assess different movement behaviours of microbats and birds in relation to wind energy facilities.

There are no studies on flying-foxes where movements have been studied specifically in relation to wind energy facilities. Note that individual papers (of which there were 84 in total) could use the same technology to address multiple behaviour types, so are tallied as multiple cases.

Taxon	Method	Migration timing and direction	Local and regional movements	Flight height
Microbats	GPS		5	2
	PIT tags		1	
	Radar		1	
	Radiotracking		2	
Birds	GPS	15	44	19
	Radar	8	8	7
	Radiotracking	2	1	

Telemetry and radar are also frequently used to assess bird and bat movements at local and regional scales, to address questions about the types of habitat features they are associated with (e.g. Veltheim et al. 2019; Li et al. 2020), and understand how local environmental conditions affect soaring suitability, and in turn, collision vulnerability for raptors (Hanssen et al. 2020; Schmuecker et al. 2020; Scacco et al. 2023). While these studies can be useful in the pre-construction phase to inform turbine siting at proposed sites, telemetry studies conducted in areas where turbines have already been installed can also help to improve knowledge about how target species interact with turbines and quantify avoidance, collision risk and attraction responses (e.g. Watson et al. 2018a; Fielding et al. 2021; see Section 4.3).

The final set of applications relate to the estimation of collision risk, and specifically involve studying bird and bat flight behaviours to quantify the range of heights at which they fly, as well as the amount of time spent at RSA heights. For example, Jenkins et al. (2018) used mobile radar to assess how frequently Great White Pelicans at a proposed wind energy site in South Africa performed 'high risk flights', while Cohen et al. (2022) used weather radar network data to assess the vertical distribution of birds moving through the USA Great Lakes district migration corridor. As noted above, however, GPS and satellite transmitters were the most frequently-used technologies for assessing flight heights and behaviours in the peer-reviewed literature (although a review by Largey et al. (2021) suggests that this occurs less frequently in the environmental consultancy sector when assessing collision risk for birds).

While GPS technology has greatly enhanced our ability to study the movements of birds and bats, and is continually improving, the flight height estimates yielded by these devices require careful consideration in the study design stage and need post-processing for reliable inference and interpretation. It is well recognised that GPS tags provide less accurate readings vertically than they do horizontally, and Péron et al. (2017) note that there are four potential sources of error associated with each point estimate of flight height from GPS transmitter data: i) error in the latitude/longitude estimate; ii) error in the digital elevation model from which height above ground is estimated; iii) error in ground elevation interpolation; and iv) error in the GPS-based measure of flight height above the ellipsoid (the Earth). Each of these sources of error have the potential to influence inference about how high a species flies, and therefore the amount of time it spends at RSA heights. In the Péron et al. (2017) example, they note that, based on the raw and uncorrected GPS data they collated from 19 devices attached to three species of raptor, 36% of flights were classified as being 'underground' (i.e. estimated altitude was <0 m above ground level). Lato et al. (2022) tested the accuracy of three different types of telemetry devices by attaching them to drones, and conducted a series of stationary, horizontal and vertical movement trials to assess sources of error. They found that the device that used barometric pressure sensors to estimate altitude, as opposed to the other two that relied on GPS triangulation, provided more accurate estimates, and that altitudes tended to be over-estimated during horizontal flights (by up to 40 m by some devices), and that amplitude (the difference between the high point and low point) was underestimated during vertical flights. Using 10 GPS transmitters from three manufacturers, Schaub et al. (2023) found that increasing frequency of in-flight fixes can increase vertical accuracy, and therefore provide more confidence in height estimates.

Given these observations and concerns regarding the accuracy of flight heights derived from telemetry devices, transmitters and GPS acquisition frequency should be selected and programmed to provide the highest possible accuracy for flight heights if these data are to be used to estimate collision risk. All height estimates being used to inform collision risk should be post-processed to remove errors and accompanied by a statement outlining likely sources of error and the potential influence on the assessment of collision risk. Whenever possible, independent height estimate validation trials (such as those conducted by Lato et al. 2022) should be conducted, and the accompanying data presented.

Knowledge gaps for Victoria

20. Very few radar or telemetry studies globally have been conducted that assess flight heights of microbats, and precise height information is lacking for all Victorian species.
21. There is the potential to make better use of existing weather radar networks in pre-construction assessment for bird and bat species that form large aggregations, particularly in coastal locations where radar coverage is more complete.
22. On-site radars have not been tested in Victoria for collecting information on the activity, numbers, flight heights and timing of bird or bat movements.
23. Information about flight heights, migration and movement routes for many Species of Concern is still lacking, and there is great potential for GPS tracking to address these knowledge gaps at a regional and state-level.

Key references

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Welbergen, JA, Meade, J (2025) *The Flying-Fox Radar Monitor*. The Lab of Animal Ecology, Western Sydney University, Richmond. https://www.animalecologylab.org/flying-fox-radar-monitor.html (accessed 24 January 2025).

5.5.4 Relationship between on-site activity and post-construction mortality

There are substantial errors and uncertainties associated with the field methods used to assess risk for birds and bats (as outlined in Section's 5.5.1 – 5.5.3), and because of this, the efficacy of these assessments in predicting post-construction fatalities has been questioned (Arnett and May 2016; Katzner et al. 2016).

We identified five papers for bats and seven papers for birds (Table 11) that assessed the relationship between on-site activity, and post-construction mortalities. However, risk assessments take place in the pre-construction phase, before the presence and operation of the turbines can influence bird and bat activity through attraction and avoidance effects (see Section 4.3). Therefore, only those studies that present true pre- versus post-construction comparisons (and ideally include both impact and control sites following a BACI design) can help us address questions around efficacy. We identified two large-scale data synthesis studies, one for bats and one for birds, that did this. Solick et al. (2020) collated pre-construction activity and post-construction mortality data for bats from 49 facilities from across the USA and Canada, that included studies where detectors were deployed at ground level (0–10 m) or were raised (30–50 m). Even after accounting for these differences in detector heights, as well as season, and splitting the analysis into echolocation guilds, they were unable to find any relationship between pre-construction activity and post-construction mortality rates. They also found that, at four facilities where bat activity was monitored during both the pre- and post-construction phases, bat activity increased overall in the post-construction phase. Similarly, Lintott et al. (2016) found that across 46 wind farms in the United Kingdom, EIAs (of which acoustic surveys are a key component) failed to accurately predict risk to bats. The authors of these studies suggest that this may be because bat activity is altered post-construction due the attractive effects of turbines to bats (see Section 4.3).

The other large-scale data synthesis was the study of Ferrer et al. (2012), which involved risk assessments for birds (utilisation surveys, or birds counted per hour) at 53 potential wind facility sites, and then mortality monitoring at the 20 sites where construction was authorised to proceed. Once again, they were unable to find any correlation between the indices developed in the pre-construction phase (a 'Relative Risk Collision Index' and 'Breeding Birds Relative Risk Index') and the observed mortalities. However, they also acknowledged that this may have been because construction did not proceed at the highest-risk sites.

The only study we reviewed that unconditionally (i.e. where it was not dependent on species, or data cleaning decisions) found a relationship between the pre-construction risk assessments and post-construction mortalities was that of Smales et al. (2013), which focused on Wedge-tailed Eagles and White-bellied Sea-Eagles at two facilities in Tasmania. While this paper is nominally about the collision risk modelling (CRM) approach, they demonstrated that collision estimates from the model (0.1–2.7 individuals per year, depending on assumed avoidance rates) matched up well with the mean annual number of individuals found during carcass searches.

All other studies that we reviewed either found no relationship, or only found relationships in specific conditions, even when activity and mortality surveys were only conducted in the post-construction/ operational periods (Table 11). One reason for the lack of correlation between pre-construction activity and post-construction mortalities may be attraction and avoidance effects (see Section 4.3). That is, once turbines become operational, they change the activity and collision risk of species on the site because they are repelled through avoidance or drawn in through attraction (see Section 4.3). Another reason suggested by Huso et al. (2021) is that many pre-construction assessments (and indeed scientific studies) fail to account for the periods when turbines are actually going to be operational, which is when most fatalities occur. Turbine operational periods are dictated by a combination of factors including planning agreements and permits, maintenance requirements, mitigation measures that may be in place that limit operational periods, weather conditions throughout the year, and energy demands (e.g. Victorian turbines are more likely to be operational at night during summer, because solar facilities provide power during the day). In the Huso et al. (2021) study, those factors combined meant that turbines produced a maximum of 25% of their potential energy output in a given year, i.e. were only operation for a quarter of the time at most. This highlights the importance of ensuring that pre-construction surveys take place at the same time of year, and even at the same times of day/night, as turbines will be operational.

There have been several other reasons proposed as to why conventional pre-construction assessments may be inadequate at predicting risk, and in turn, suggestions for how these surveys can be improved. Because small-scale patterns in wind currents and local siting factors can strongly influence flight behaviours (Huso et al. 2021, see Section 5.4.2), Ferrer et al. (2012) recommend that bird surveys should be conducted at the point where turbines are going to be constructed, rather than across the site more generally. Based on their study of vultures, Carrete et al. (2012) suggest that the spatial distribution of species aggregations (i.e. colonies) will be a better proxy for risk than standard point-counts, particularly for territorial species. Smallwood et al. (2009) emphasise the important influence of species-specific flight behaviours and visual acuity, and emphasise that assessments should be made on a species by species basis rather than aggregated into generic utilisation rates.

For bats, there has been some suggestion that deploying detectors at height could improve the accuracy of assessments; Roemer et al. (2017) found that the bat species they most frequently detected at height at 23 wind masts (not turbines) across France and Belgium were also the species that were most frequently found in carcass searches. Peterson et al. (2021) also deployed detectors at height (this time mounted to turbine nacelles) and found a strong relationship between bat activity and fatalities, though importantly this was once turbines were operational. Smallwood and Bell (2020b) also found a relationship between bat passage rates and fatalities, but in this case they recorded bats the night immediately prior (observed through a thermal camera) to the carcass searches being conducted. Collectively, this suggests that while assessments undertaken in the pre-construction phase can inform siting decisions to avoid potential high-risk locations and inform mitigation measures, impacts to bats are not likely to be well understood until the post-construction phase when turbines are operational, highlighting the importance of post-construction assessments and mortality surveys.

Knowledge gaps for Victoria

24. There have been no quantitative studies in Victoria on either birds or bats comparing either pre- or post-construction activity levels and operational phase mortality, and factors influencing this.
25. There is no information in Victoria about how post-construction activity levels of bats and birds change over time in relation to environmental conditions and population dynamics, and how these influence mortality estimates over longer timeframes.

Key references

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Table 11. Summary of studies assessing the relationship between on-site activity (either pre- or post-construction), and post-construction mortalities.

Study	Taxon	Country	No. facilities	Activity measure	Activity period	Pre-post correlation?	Summary
Bennett and Hale (2018)	Bats	USA	1	Resource map, Acoustic surveys	Only post-construction	No	Neither the resource availability model nor species activity predicted mortalities
Bolívar-Cimé et al. *(*2016)	Bats	Mexico	1	Acoustic surveys Trapping	Only post-construction	No	Capture and detection rates for species did not correlate with mortalities
Peterson et al. (2021)	Bats	USA	2	Acoustic surveys (at the nacelle)	Only post-construction	Yes – only when operational	Bat passes at the nacelle when turbines operational explained almost 80% of the variation in mortalities, but activity occurring during non-operating periods did not predict mortality rates during subsequent operating periods
Smallwood and Bell (2020b)	Bats	USA	1	Passage rates (thermal)	Only post-construction	Yes – night prior	Turbines where fresh carcasses were found in next-morning searches had on average four-times higher passage rates the night immediately prior
Solick et al. (2020)	Bats	USA, Canada	46	Acoustic surveys	Pre-construction	No	Activity rates did not predict bat fatalities, even after accounting for detector height, species call frequency, and season
Arikan and Turan (2017)	Birds	Turkey	3	Utilisation surveys (point counts)	Only post-construction	Yes – but inverse	Negative relationship between the flight frequency of birds within 0–500 m of wind turbines and bird mortality
Carrete et al. (2012)	Birds	Spain	34	Counts at roost and breeding sites	Pre-construction	Yes – but aggregations	Mortality of Griffon Vultures at turbines increased when they were located in areas with large species aggregations, but this would not have been detected with standard point counts
Ferrer et al. (2012)	Birds	Spain	20	Utilisation surveys (point counts)	Pre-construction	No	No relationship between mortality and risk indices developed from pre-construction assessments. Non-significant relationships between vultures/hour, and kestrels/hour, and mortality.
Hull et al. (2013)	Birds	Australia	3	Utilisation surveys (point counts)	Pre-construction	No	Carcass searches and field surveys for birds at two facilities in Tasmania, presence on-site a poor indicator of collisions
Kitano and Shiraki (2013)	Birds	Japan	3	Utilisation surveys (point counts)	Only post-construction	Yes – but some data omitted	Utilisation rates explained most of the variation in mortality rates among species and among locations, but only species detected in both utilisation and carcass searches included in analysis
Smales et al. (2013)	Birds	Australia	2	Utilisation surveys (point counts)	Pre-construction	Yes	Pre-construction CRM estimates compared well with average annual number of found carcasses of Wedge-tailed Eagles and White-bellied Sea-Eagles
Smallwood et al. (2009b)	Birds	USA	28 (plots)	Utilisation surveys (point counts)	Only post-construction	Species-specific	Fatality rates increased with rates of flights near RSA for large raptors, and perching and close flights for small non-raptors

5.6 Predicting impacts on species at risk

Moderate number of papers (40 papers)	Low bird bias (88%)	Low geographic bias (Europe 42%)

Once broader-scale desktop assessments have been conducted (see Sections 5.2, 5.3), and fit-for-purpose field data have been collected (see Section 5.5), quantitative analyses can be used to try and predict how a proposed facility will individually or cumulatively impact on species of interest. Laranjeiro et al. (2018) reviewed quantitative approaches used in wind energy EIAs, including i) collision risk models, ii) individual-based models, iii) population modelling approaches, iv) index-based models, and v) species distribution models (SDMs). They provide an overview of the input parameters required for each type of model, the types of impacts that they are typically applied to (collisions, disturbance, and habitat alterations), and they also discuss the relative advantages and disadvantages of each type of approach. Because SDMs are typically used both during the regional planning phase and to determine what species might be impacted prior to field surveys being conducted, and index-based models are covered in our species at risk discussion (see Section 5.3), here we focus on demographic (individual and population-based) and collision risk models.

5.6.1 Collision risk models (CRMs)

Collision risk models (CRMs) are often developed for diurnal bird species during pre-construction assessments, as a means of predicting the rate or number of collisions that may occur (or the probability of collision) once a facility becomes operational. CRMs are most frequently used for raptors (12/17 case studies we identified, or 70%) but we also found examples of CRMs that were developed for shorebirds such as gulls (Everaert 2014; Masden et al. 2021), waterbirds (Sugimoto and Matsuda 2011), and in one case, for 27 different bird species in a proposed development area in the Western Ghats, India (Pande et al. 2013). CRMs are not typically used for bats because they rely on direct observations of flight behaviours and heights, and ideally also rates of turbine avoidance, and these are not readily observable for nocturnal species (though Smales et al. 2013 note that an unpublished CRM has been used for the Pacific Flying-fox in Fiji).

There are now a range of different CRMs that are used in pre-construction assessments, and most have been developed from, or extend on, a model first described by Tucker (1996), which estimates the probability of collision when an individual bird flies through the RSA of a single turbine. The key input parameters for this early model relate to the dimensions and speed of both the bird species of interest, and also the turbine blades at the proposed site. However, there are additional data requirements for more recently-developed models, which have become increasingly complex and flexible to better capture real-world situations.

Masden and Cook (2016) provide a review of 10 different CRMs that appear in the scientific and grey literature, including details of their developments, associated assumptions and iterative improvements. They note that these models vary with regards to whether they: i) incorporate avoidance behaviours; ii) model risks at individual or across multiple turbines; iii) account for the static parts of the turbine (i.e. the tower) as contributing to collisions; iv) model individual birds or entire populations; v) assume wind speed and direction are constant or can be incorporated into the model; vi) allow for birds to approach turbines from oblique angles; and vii) are stochastic, and can account for uncertainty and allow for variation in key input parameters.

There are three key models that appear to be used most frequently both academically and in practice, based on descriptions available in the literature (Table 12), and all of which estimate the number of birds that will collide with turbines across an entire proposed facility (in contrast to the Tucker modelled probability of a single bird at a single turbine). In the UK, the Band model is commonly applied, and was originally developed for offshore developments as a direct extension of the Tucker model. In the USA, the US Fish & Wildlife Service (USFWS) model is used specifically for Golden and Bald Eagles within their adaptive management framework (discussed in Section 6.6). This model was designed to be continuously updated and improved, to incorporate new information from carcass searches (e.g. Bay et al. 2016; New et al. 2021). In Victoria, the Biosis model, which has been described and published (Smales et al. 2013), is often presented in pre-construction Bat and Avifauna Management (BAM) Plans, though other models are also used. The Biosis model allows for multidirectional flights and the assignment of different avoidance rates, but it cannot account for differences in wind speed and direction. Also, while the Biosis model can be modified to accommodate stochasticity (uncertainty), unlike the USFWS model it is by default deterministic. Uncertainty is an inherent property of ecological systems and critical for accurately assessing risk, in terms of both the magnitude but also the probability of an event (i.e. a collision) occurring (Battisti et al. 2020).

Table 12. Comparison of three collision risk models frequently used in the scientific and grey literature – adapted directly from Masden et al. (2016).

Note that all three of these models produce estimates of the number of birds that will collide with all turbines across a facility, so are population-level estimates of risk and incorporate measures of avoidance, but do not account for differences in wind speed or direction.

Model	Frequently used in	Stationary components	Oblique angles of approach	Stochastic
Band model	UK	No	No	No
Biosis model	Australia	Yes	Yes	No
USFWS model	USA	Not specified	No	Yes

CRMs estimate values that, in the real world, are a product of complex behaviours and physics, so are not without their critics. A key concern is that the predictions yielded by CRMs are highly sensitive to input parameters such as assumed avoidance rates (Chamberlain et al. 2006), but empirical data representing interactions with, and behaviours around, turbines for species of interest are rarely collected (Smales et al. 2013). Some have attempted to modify the CRM framework to make it less reliant on these avoidance rates; Kleyheeg-Hartman et al. (2018) present an alternative 'Flux Collision Model' that can incorporate empirical collision data from existing wind energy facilities to inform predictions for planned facilities. However, as yet, this approach does not appear to have been widely adopted. Douglas et al. (2012) also demonstrate that CRMs can be sensitive to the number of hours over which vantage point surveys are conducted; in their study of White-tailed Eagles at Smøla they found that variability in predicted collision rates only reached an asymptote after 62 hours. Very few CRMs are validated with post-construction monitoring data to assess how well they predict collisions (discussed in Section 5.5.4, though see Smales et al. 2013).

5.6.2 Demographic models and population viability analyses (PVAs)

As discussed above in Section 4.2.4, demographic models can be used to estimate the extent to which existing wind energy facilities have already impacted on bird and bat populations at broad and regional scales. However, these models are also used in pre-construction EIAs, to support assessments of whether or not a facility will have significant biodiversity impacts.

One approach that is commonly adopted for wind energy EIAs is the Potential Biological Removal (PBR) framework, which was originally developed to help determine sustainable limits of megafauna bycatch (whales, dolphins, seals etc.) in marine fisheries. Under the US *Marine Mammal Protection Act 1972*, the PBR is a mortality limit or *"the maximum number of animals, not including natural mortalities, that may be removed from a marine mammal stock while allowing that stock to reach or maintain its optimal sustainable population."*

In the context of wind energy developments, PBR models are used to attempt to identify levels of acceptable extra mortality from collisions, or 'harvest', that populations can tolerate. While the calculation of a harvest quota involves a fairly simple equation, it does require the user to define a 'recovery factor', which is not a biological property of the population being studied, like (for example) a birth rate would be. Instead, the recovery factor is a parameter that should be fined-tuned based on an explicit, pre-defined conservation objective, i.e. a target minimum population size that needs to be maintained even when there is substantial uncertainty. The fine-tuning should occur through an evaluation process, where the user runs a series of population simulations to ensure that the conservation objective can be met in the long-term when the PBR rule is implemented (Chambert et al. 2024).

Chambert et al. (2024) outline four key reasons why the PBR approach is ill-suited to EIAs for wind energy developments, namely:

- There is a scope mismatch between EIAs, which focus only on a single facility, and PBRs, which are designed to focus on an entire population of interest and should account for all threats causing non-natural fatalities (not just collisions from one facility)
- Because EIAs typically have a single-facility focus, a conservation objective is rarely articulated for the population of interest
- The PBR framework is typically implemented in EIAs without population simulations, which are required to ensure that an appropriate recovery factor is being used
- PBR implicitly assumes density dependence, that is, the population growth rate will increase and compensate for individuals lost to 'harvesting', but this is not always evident in bird populations.

Because of this, Chambert et al. (2024) advocate for the use of PVAs instead when assessing the potential impacts of a facility or facilities on a species of interest. This is supported by Schippers et al. (2020), who tested population simulations for seven European bird species, applying a 'harvest' according to the number allowed under a PBR with a range of recovery factor values. They show that continually removing individuals from the population according to the PBR could jeopardise population persistence, and that outcomes were highly sensitive to recovery factor values. Even when the recovery factor was set to a low value, there was a 5% decrease in the population growth rate on average. Diffendorfer et al. (2021) also tested both PVA and PBR approaches when predicting changes in 14 raptor species populations with projected increases in wind energy capacity, and found that conclusions from the two approaches differed, with the PBR producing more pessimistic predictions.

Both Schippers et al. (2020) and Diffendorfer et al. (2021) are examples where PVAs take the form of age-structured Leslie matrix models. For these models, individuals of different ages are assigned different vital rates (survival and fecundity), and as the vital rates are simulated over time and individuals die, are born, or transition to the next age class, the population can grow or shrink. Ideally, the range of values used to represent vital rates in these models should be based on the population of interest (from long-term monitoring data), or at least the species of interest. However, because these data are rarely available, model inputs are instead often based on closely-related species, or ranges of values observed in similar taxa. To model the impact of threats, such as increased mortality associated with turbine collisions, vital rates can be modified and the trajectories of 'impacted' versus 'control' populations can be compared. Again, these changes to vital rates should ideally be based on real-world estimates of how collisions impact on the species being modelled, in the types of locations being modelled. There are a range of metrics that can be extracted from PVAs to help quantify and articulate the differences in modelled outcomes, and Cook and Robinson (2017) present a framework with a set of decision criteria (Acceptable Biological Change, Decline Probability Difference, and Counterfactual of Impacted and Unimpacted Populations) to inform comparisons.

PVAs can be deterministic (so yield the same result every time they are run), or stochastic, taking into account the variation in conditions that occur between years (environmental stochasticity) or between individuals in a population (demographic stochasticity) in the real world. Stochastic models will yield a different result every time they are run to reflect this variation, so simulations are typically run thousands of times, and the mean and variation around the results are presented. Models can also be density independent, meaning that vital rates are constant irrespective of the number of individuals in the population, or density-dependent, so vital rates will change depending on how close a population is to carrying capacity. Another level of complexity that can be added is that models can be spatially explicit, that is, they account for differences in habitat quality and patch size across the landscape, and this informs carrying capacity, density dependence, and also rates of immigration and emigration between patches. An alternative approach to matrix-based models are agent- or individual-based models, where rather than modelling the population as groups of individuals in defined age classes, individuals are dispersed across a landscape and interact with threats (such as turbines) and other habitat features based on their adaptations. We do not detail these approaches here, though see Schaub (2012) and Ferreira et al. (2015) for bird and bat examples, respectively.

As this brief description highlights, demographic models are complex, data hungry, and can be sensitive to model inputs and assumptions. While there are examples in the literature where these models have been used to predict impacts on well-studied species for which there is ample mortality, monitoring, and tracking data (e.g. Bastos et al. 2016; Frick et al. 2017; New et al. 2021), in reality for most species impacted by collisions in Victoria, this baseline information is lacking. The complexity of the models can also make it difficult for less-experienced users to build and/or assess them, though in some cases fit-for-purpose tools have been developed to improve accessibility. For example, Chambert et al. (2023) created the 'EolPop' RShiny app, which allows users to run demographic simulations for various European bird species and to assess population-level impacts of collision fatalities. Likewise, in response to concerns about the potential cumulative effects of both the white-nose syndrome fungus and turbine collisions on bat populations in North America, scientists at the United States Geological Survey (USGS) have developed 'BatTool' (Wiens et al. 2023) for scenario-based matrix population models. An additional, important benefit of these platforms is that because inputs are somewhat standardised, and the underlying models are clearly and transparently documented, outputs and predictions are directly comparable and easier to interpret.

Knowledge gaps for Victoria

26. Different CRMs are applied and presented by different proponents, and the extent to which this influences predicted collision risks is unclear. There has not been any quantitative comparison of estimates yielded by the different CRMs.
27. Empirical data on avoidance rates are needed for all Victorian bird Species of Concern, in order to ensure more accurate estimates of risk are produced by CRMs.

28. Validation of CRMs is needed to understand how well they predict mortalities, and what parameters may be driving differences between predicted and actual mortality.

29. It is unknown whether a reliable CRM could be developed for flying-foxes based on observations at dusk and through the use of GPS tracking or on-site radar (see Section 5.5.3). Nothing is known about potential avoidance rates for flying-foxes.

2. (Knowledge gap 2 repeated from Section 4.2.4): Basic empirical information about population size and structure, and vital rates (fecundity and survival rates) are not available for most species that are impacted by turbines in Victoria. This prevents the development of precise and robust PVAs, and limits our ability to predict both broad-scale impacts and also the potential effectiveness, at a population level, of different siting options and mitigation measures.

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6 Mitigation options and effectiveness

6.1 Curtailment

Moderate number of papers (28 papers)	Low bat bias (79%)	Low geographic bias (USA 50%)

'Curtailment' is a generic term used to describe times when energy production is purposely reduced at a facility, and various types of curtailment that involve stopping turbine blades from spinning to reduce collisions with wildlife are used as mitigation measures. Curtailment was the most well-studied mitigation in our review, and was the most consistently effective with regards to there being a statistically significant reduction in estimated mortality relative to a control.

6.1.1 Low wind speed curtailment

The most well-researched form of curtailment is 'low wind speed' or curtailment, which is predominantly intended to reduce the impact of collisions on insectivorous microbats.

When wind speeds are low, there is not enough force to spin turbine blades at a sufficient rate to generate electricity. For this reason, manufacturers' settings specify a wind 'cut-in' speed, below which turbines are not operational and do not feed power back into the grid. This speed is typically set at around 2.0–4.0 metres per second (m/s), with 3.0–3.5 m/s typically being used in Victoria. Operators may also choose to 'feather' the blades below this speed, which involves rotating them 90° (parallel to the wind) so they do not freely spin ('free-wheel').

Many microbat species reduce their flight activity during high wind. Therefore, increasing night-time cut-in wind speeds to levels when there are typically fewer bats in flight, and while there is still relatively little energy being generated, can substantially reduce the number of fatalities while also minimising losses of revenue. Insectivorous microbats are also most active during the warmer months, particularly in temperate zones, meaning that periods of night-time curtailment can be somewhat constrained and targeted to those seasons, and to times between dusk and dawn. We found that in all studies reviewed where the authors had either experimentally tested cut-in speeds above the default settings, or had synthesised data from across multiple curtailment trials, there was a significant reduction in microbat fatalities (Table 13).

It is important to note that birds and bats collide both with turbines that are generating electricity (i.e. where blades are rotating because winds exceed the cut-in speed), and those that are free-wheeling (i.e. where winds are below cut-in speed, but blades are rotating because they have not been feathered), and in many studies it is unclear if and when feathering is occurring. This can make disentangling the effects of different cut-in speeds, versus feathering, very difficult to tease apart. Only one study that we are aware of (Baerwald et al. 2009) explicitly compared the effect of having turbines free-wheeling, versus feathering below the manufacturer's cut-in speeds, versus low wind speed curtailment (5.5 m/s) with feathering. The authors found that feathering alone reduced bat fatalities by 50%, though the manufacturer's cut-in speeds were already set quite high (4.0 m/s).

The question of what cut-in speed to use for low wind speed curtailment is going to be highly species and context specific. Microbat species with long and narrow wings (i.e. species with a high aspect ratio), such as the White-striped Free-tailed Bat, tend to fly high and fast and forage in open spaces away from clutter and vegetation. Species such as these are capable of flying and foraging at higher wind speeds than those with shorter and broader wings (i.e. species with a low aspect ratio). Consequently, higher cut-in speeds will likely be required to substantially reduce mortality for these open-area adapted species. These morphological factors will interact with landscape and ecological context, such as whether a facility is close to a roost or is in a preferred foraging area. For this reason, there is not going to be any one optimal cut-in speed that universally balances the trade-off between reduced fatalities and revenue. Nonetheless, the only Australian, peer-reviewed, low wind speed curtailment experiment (Bennett et al. 2022) found that even when cut-in speeds were increased from 3.0 to 4.5 m/s (a modest increase relative to international standards), the number of White-striped Free-tailed Bat carcasses found during comparable monitoring reduced from 18 to six, which aligns closely with studies from North America. For example, in a meta-analysis of data synthesised from across the USA and Canada, Adams et al. (2021) found that increases in cut-in speeds of 2.0 m/s or greater relative to a control significantly reduced fatalities. A subsequent North America meta-analysis (Whitby et al. 2024) found that for every 1 m/s increase in curtailment cut-in speed, there was a 33% reduction in overall bat fatalities, and that cut-in speeds of 5 m/s reduced fatalities for all bats by 62%, and fatalities of individual species by 48–61%.

It is unclear whether low wind speed curtailment could be as effective as a mitigation measure for birds or flying-foxes, and it appears to be rarely tested for this purpose. Larger-bodied raptors (which are typically the focus of bird collision mitigations) and flying-foxes likely remain active at higher wind speeds, to the point where curtailment would likely become practically and economically unfeasible. Individual species of smaller-bodied birds also collide with turbines more rarely than individual species of microbats, making it statistically challenging to demonstrate effectiveness. We found only one study that looked at the effect of cut-in speeds on bird fatalities (Anderson et al. 2022), and it reported that there was no significant difference in swallow species' fatalities across the spectrum of default manufacturer's cut-in speeds used in Canada (2.0–4.0 m/s). However, the authors acknowledge that their findings may have been confounded by the assumption that the manufacturers' cut-in speeds were actually being used for operations, and the fact that it was unclear whether or not turbines were being feathered. Noting this, it is possible that flying-foxes and nocturnal birds (including those that migrate at night, such as the Orange-bellied Parrot) may experience indirect benefits from night-time low wind speed curtailment being implemented for microbats.

Only a few papers report on the impacts of low wind speed curtailment on energy generation and revenue, and those that do use a variety of metrics, spanning different time periods (the trial period, versus a season, versus a year), making it difficult to compare findings. Bennett et al. (2022) stated that their Victorian 4.5 m/s curtailment trial for a four-month period resulted in a 0.16% decrease in annual power generation, though they do not state what the percent loss was for the experimental period itself. In a simulation study exploring the impacts of different curtailment strategies on annual energy production across the USA, Maclaurin et al. (2022) predicted that increasing cut-in speeds to 6.0 m/s from July-October (also four months) would result in a 1.3% reduction in annual energy production at the national scale, though the impacts varied greatly between regions. Arnett et al. (2011) predicted that for the 23 turbines curtailed during their 75-day experiment (approximately 2.5 months), increasing cut-in speeds to 6.5 m/s resulted in an 11% loss of energy output for that period, or a 1% loss of total annual output. They also predicted that if cut-in speeds had been set at 5.0 m/s instead, these figures would have been a 3% loss for the study period, and an 0.3% loss for the whole year.

Table 13. Curtailment: Findings relating to the effectiveness of increasing night-time turbine cut-in speeds to reduce bat fatalities, based on *in-situ* experiments and data syntheses (as opposed to models or simulations).

Rows are arranged from lowest to highest curtailment treatment cut-in speeds. We have indicated whether or not turbine blades were feathered below cut-in speeds for the manufacturer's settings, and curtailment treatments ([F]: feathered, [N]: not feathered, [?]: unclear whether feathering was in place), and where ranges are provided, we have indicated whether these are a result of variation between years ('[y]') or species ('[s]').

Study	Study type	No. sites	Region	Manufacturer (control) cut-in speed (m/s)	Curtailment treatment cut-in speed (m/s)	Treatment increase in cut-in speed (m/s)	Reduction in bat fatalities with curtailment treatment	Reduction in bat fatalities with difference (Δ) in cut-in speed (m/s)
Adams et al. (2021)	Meta-analysis	17	USA and Canada	3.0–5.0 [?]	4.0–7.0 [?]	1.0–3.5	63% overall (cf control)	Δ 1.0: 52% Δ 2.0: 70% Δ 3.0: 72%
Bennett et al. (2022)	Experiment	1	Australia	3.0 [?]	4.5 [?]	1.5	54%	NA
Anderson et al. (2022)	Data synthesis	59	Canada	2.0–4.0 [?]	5.0 [?]	1.0–3.0	33%	NA
Good et al. (2022)	Experiment	1	USA	3.0 [?]	5.0 [F]	2.0	42.5%	NA
Whitby et al. (2024)	Meta-analysis	8	USA and Canada	3.0–4.0 [?]	5.0 [F]	1.0–2.0	62%	Δ 1.0: 33%
Baerwald et al. (2009)	Experiment	1	Canada	4.0 [N, F]	5.5 [F]	1.5	60%	NA
Davy et al. (2021)	Data synthesis	48	Canada	3.5 [?]	5.5 [F]	2.0	59–81% [s]	NA
Arnett et al. (2011)	Experiment	1	USA	3.5 [F]	5.0–6.5 [F]	2.0–3.0	72–82% [y] (no difference between 5.0 and 6.5 treatments)	NA
Rnjak et al. (2023)	Experiment	1	Croatia	Unclear	5.0–6.5 [F]	Unclear	78%	NA

6.1.2 Multi-factor curtailment

There are also other, modified forms of curtailment where additional factors other than just wind speed (and season/time of day) dictate whether turbines are operational. We have reported on the use of automated detection systems to trigger turbine shutdowns below (see Section 6.3) so will not cover those approaches here.

Temperate microbat species, such as those found in Victoria, are most active during periods when flying insect prey availability is greatest, and this is typically at warmer temperatures when rainfall is low (Martin et al. 2017). Consequently, adding conditions that must be met before night-time curtailment is triggered, such as temperature, rainfall, or detected bat activity, can reduce the curtailment period and subsequent losses of revenue, while continuing to minimise collision impacts (this is sometimes also referred to as 'smart' curtailment).

We found two examples where these multi-factor curtailment approaches were experimentally tested to assess reductions in bat fatalities. Firstly, a study by Martin et al. (2017) compared control turbines (manufacturer's cut-in speed of 4.0 m/s), with a 'wind only' treatment (cut-in speeds of 6.0 m/s), and a 'wind + temperature' night-time curtailment treatment where turbines were shut down when both wind speeds were less than 6.0 m/s and temperatures exceeded 9.5°C. The interpretation of results from this study are complicated by the fact that temperatures rarely dropped below 9.5°C during the periods when high fatalities were expected, so the difference in fatalities between the 'wind only' and 'wind + temperature' treatment was not analysed. Nonetheless, they concluded that during those periods when temperatures did tend to drop below 9.5°C (late spring and early autumn), that including this as a condition decreased energy losses by 18%. Between them, the two night-time curtailment treatments reduced bat fatalities by 62% on average.

Second, two papers focus on the same set of field data collected at a facility in Wisconsin (USA) to test the 'TIMR' (proprietary) system, which triggers night-time curtailment when winds speeds are low (≤8.0 m/s) and any bat calls are identified in the 10 minutes prior. Hayes et al. (2019) compared 10 turbines curtailed using the TIMR system with 10 controls (manufacturer's cut-in speed set at 3.5 m/s), and concluded that this form of multi-factor night-time curtailment led to an 84.5% reduction in fatalities and reduced curtailment time by 48% relative to standard blanket night-time curtailment. However, in a later study, Rabie et al. (2022) re-analysed the same data and concluded that the original estimated effectiveness was inflated because of the analytical approach used (discussed below in Section 7.1.4). They suggested that the figure was more likely a 75% reduction in fatalities 'at most'. Rabie et al. (2022) also included data from a further 10 turbines where blanket night-time curtailment (with cut-in speed set at 4.5 m/s) had been applied. In this case, they concluded that the blanket night-time curtailment treatment led to only a 47% reduction in fatalities because of the substantial difference in cut-in speeds.

Other studies have explored the potential effectiveness of multi-factor curtailment approaches through simulation. Barré et al. (2023) used four years of post-construction bat activity data from wind energy facilities across France to build a predictive model of activity based on landscape, turbine, and environmental variables. They then tested the use of this model to inform multi-factor curtailment and the effect that it would have on energy production, compared to standard night-time low wind speed curtailment. They concluded that the multi-factor curtailment approach was in fact more efficient and would result in 7–31% fewer bats being exposed to rotating turbine blades compared to low wind speed curtailment, depending on the species guild and temperature threshold used. The Maclaurin et al. (2022) energy production simulation study mentioned above with regards to blanket night-time curtailment also explored multi-factor curtailment scenarios. In this case, they assessed a range of wind speed thresholds but also added the condition that temperatures had to exceed 10°C, and there had to have been <1 mm rain per hour, before curtailment was triggered. They found that the difference in energy production between blanket versus multi-factor approaches increased as the wind speed threshold increased, and there was greater financial benefit to adding temperature and rainfall conditions if the wind speed threshold was also set higher.

6.1.3 Seasonal and manual curtailment

Because some species may only be active in an area for a specific time period, for example during migration, it is possible to assign periods when turbines are curtailed to avoid fatalities. While there were relatively few examples of this in the literature, Peterson et al. (2021) found that bat fatalities could be predicted from activity levels recorded at the nacelle during operation, and also activity itself could consistently be predicted among turbines and years. They proposed that seasonal night-time curtailment during high-activity periods would be a more cost-effective approach than year-round night-time low wind speed curtailment. In another US study, Smallwood and Bell (2020) assessed the effectiveness of seasonal curtailment implemented at the Altamont Pass Wind Resource Area during the autumn migration period, in response to concerns about the large number of Golden Eagles being killed. They found that while it appeared that all-day curtailment (i.e. both daytime and night-time) substantially reduced fatalities for bats, it was ineffective for birds when all species were assessed together. They suggested that while moving blades may pose the greatest risk for some species such as eagles, kestrels, and flycatchers, for many others the structures themselves and stationary blades may pose more of a hazard. This is because birds used the towers and hollow blades for roosting and perching, and motionless blades (that are not creating noise) are harder for nocturnal species to detect, so curtailment alone does not reduce fatalities. Another reason why curtailment may appear to be ineffective for birds could be that the smaller-bodied species that are likely to be less active in higher wind speeds (as bats are) are killed in relatively small numbers, making it more difficult to detect an effect. By contrast, larger-bodied raptors that collide with turbines more frequently will still fly during higher wind speeds.

A more labour-intensive, but nonetheless effective approach is the ‘selective stopping’ or ‘Turbine Shutdown System’, as described by de Lucas et al. (2012) and Ferrer et al. (2022). This was implemented at 20 wind energy facilities across Cadiz province in southern Spain from 2008–2020 in response to the large number of soaring birds (specifically Griffon Vultures) being killed by collisions with turbines. During this time, regulations required that 8–17 trained observers be distributed over the area covered by these facilities every day of the year, from dawn until dusk. When an observer noticed that a soaring bird was on a trajectory that would potentially result in a collision with a turbine, they would call the control office and order that it be shut down and not re-started until a second call was made. Ferrer et al. (2022) analysed 15 years of monitoring data from these programs and estimated that implementation of the protocol reduced fatalities of all soaring birds by 62% and Griffon Vultures by 93%, and that counts of this species increased seven-fold over the period, while an energy production loss of only 0.51% was incurred.

In all cases here, when discussing effectiveness of curtailment strategies (low wind speed, multi-factor, or seasonal), we have presented values of the average percent reduction in fatalities across all species, for the sake of consistency. However, as highlighted above, effectiveness will differ between species depending on how high they fly and at what wind speeds, the seasonality of their activity patterns, whether they are migratory or resident etc. All of these factors need to be considered when the objective of a mitigation strategy is to reduce mortality for specific target species.

Knowledge gaps for Victoria

30. Only one cut-in speed for night-time low wind speed curtailment has been experimentally tested in Victoria (Bennett et al. 2022), and the bulk of the evidence identified in our review on the effectiveness of curtailment came from North America. Given the morphological and ecological differences in bats between regions, it is difficult to draw conclusions about what cut-in speed would optimise reductions in Victorian bat fatalities while minimising loss of revenue. Therefore, while we can be confident that low wind speed curtailment will reduce bat fatalities, empirical evidence is needed to inform decisions about cut-in speeds, as well as the seasonal, yearly and site-level variation in effectiveness.
31. The relationship between the wind speeds at which Victorian threatened microbat species fly and other environmental conditions (i.e. night-time temperatures) needs to be better understood to inform effective multi-factor night-time curtailment strategies.
32. Setting turbines to be ‘feathered’ when they are not operational (when wind speeds are not exceeding manufacturer’s cut-in speeds, or as part of low wind speed curtailment) has the potential to reduce collision fatalities. However, it is inconsistently and poorly documented whether or not these settings are in place, and little is known about the effectiveness of feathering in reducing mortalities, so further investigation is warranted.
33. There is no published information on the wind speeds at which flying-foxes are active, so it is difficult to determine whether low wind speed night-time curtailment could be an effective mitigation measure for this group as well.
34. There was little consistent published information on how curtailment strategies impacted energy generation and revenue depending on the wind speed, season, and duration parameters.
35. Further information is required on the effectiveness of curtailing individual turbines that cause the greatest number of mortalities, compared to curtailing a larger number of turbines across a wind energy facility (see the discussion regarding turbine siting in Section 5.4.2).

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6.2 Buffers

Moderate number of papers (35 papers)	Low bird bias (74%)	Moderate geographic bias (Europe 63%)

The term ‘buffer’ is used to describe the distance that turbines must be set back from key habitat features and resources, such as nesting sites. For example, under the US Bald and Golden Eagle Protection Act (1940), regulations for general permits (revised February 2024) stipulate that turbines cannot be constructed within 2 miles (3.2 km) of a Golden Eagle nest, or within 660 feet (201 m) of a Bald Eagle nest. The EUROBAT guidelines for wind farm projects recommend that there should be a minimum distance of 200 m between turbines and important bat habitats such as woodlands, hedgerows, and waterbodies (Rodrigues et al. 2015). Knowledge on appropriate buffer distances can inform both the initial siting of wind farms, as well as the location of turbines within the wind farm footprint. These distance recommendations are often made with multiple objectives in mind, including the reduction of fatalities caused by collisions with turbines, which is the focus of this review. However, other objectives may include ensuring that nesting and breeding habitats are not disturbed so as to not cause local population declines, or that the individual or combined effects of increased noise, changes in habitat structure, and avoidance behaviours do not lead to an overall change in community composition. Teff-Seker et al. (2022) provide a review of policies and regulations related to turbine zoning and buffering for California, Germany, and Israel, with a specific focus on how these address noise impacts on wildlife.

The methods used to assess how effective a mitigation measure is should be informed by the objective; in this case, the question is how effective a given buffer distance is in reducing fatalities caused by collisions. The most direct means of assessing this would be to analyse the estimated number of fatalities at turbines (from carcass searches) that are varying distances from habitat features, but we found few examples of this in the literature (though see Section 5.4.2). Instead, studies more frequently used indirect measures for the risk of collisions, such as telemetry (GPS or satellite tracking devices) to assess movements around turbines, or through field observational studies and acoustic recorders to explore changes in activity patterns with distance from turbines or the habitat features. As discussed in Section 4.3.1 (‘Attraction, avoidance and displacement of bats’) and also Section 5.5.4 (‘Relationship between on-site activity and post-construction mortality’), there is evidence to suggest that, particularly for bats, pre-construction activity patterns may not be a good indicator of post-construction collision risk, in part at least because of the attractive effect of turbines for some species.

Across both the bird and bat literature, findings were mixed and it is difficult to ascertain patterns, particularly given they relate to diverse species in different landscape contexts, and are focussed on distances from different types of habitat features (Table 14). For bats, recommendations based on carcass searches or acoustic surveys, all conducted in Europe, were that turbines should be set back at least 100–600 m from hedgerows, forests and roosts. While these distances are likely adequate for species that tend to forage close to vegetation, open-area adapted species are less constrained in their movements and may require buffer distances so large that they are impractical (Apoznanski et al. 2018) and even then may be ineffective. For these species, other mitigations should be considered.

The literature for birds also covered various guilds and was based on carcass searches, telemetry, field observations and surveys. For large raptor species such as White-tailed Eagles, Golden Eagles, Buzzards and Red Kites, recommendations were that buffers of at least 1–3.2 km should be applied around nesting sites (Balotari-Chiebao et al. 2016; Salomon et al. 2020; Watson et al. 2014). The exception to this was the Cape Vulture, where a colony buffer of 50 km was recommended, given it is highly vulnerable to collisions (Venter et al. 2019). A range of waterbirds, shorebirds and passerines were also studied, though recommendations of buffer distances for these are less explicitly prescribed or they are highly varied. Li et al. (2020) stated that turbines should be set back 800 m – 1.3 km from a dyke near important wetland habitats along the East Asian-Australasian flyway, while Veltheim et al. (2019) recommended buffers of at least 1.6–2 km from the centre point of wetland night roosts to avoid impacts on Brolga breeding success in Victoria. It was estimated that Dupont's Lark, a small, threatened passerine in Spanish steppe habitats, required 4.5 km buffers around populations to avoid further declines (Gómez-Catasús et al. 2018), while Howell et al. (2020) predicted that shorebirds in Canada would require setbacks of 2–14 km to gain enough height to clear turbines 165 m tall.

Knowledge gaps for Victoria

36. Globally, there have been few empirical studies assessing the relationship between the distance of turbines to key habitats, nest/roost sites, or waterbodies, and fatalities of bird and bat species. This research is needed to make recommendations about buffer distances.
37. For bird and bat Species of Concern, it is unclear what the relationship is between distance from key habitat features, and tendency to fly at heights within the RSA of turbines. This information would be very useful to inform buffer sizes.

Key references

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Additional references

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Table 14. Buffers: Studies assessing the effect of turbine distance to habitat features on fatalities or indirect measures of risk of fatalities.

Study	Taxon	Country	Method(s)	Focus	Recommendation/finding
Apoznanski et al. (2018)	Bats	Sweden	Carcass searches, Acoustics, Telemetry	Fatalities	Barbastelle Bats move very long distances but are not active around turbines, no recorded fatalities, very large buffers (2–7 km) are not justified
Moustakas et al. (2023)	Bats	Greece	Carcass search	Fatalities	Raw counts indicate that turbines in areas where the surrounding 5 km radius was >50% natural areas (trees and water) killed significantly more bats
Barré et al. (2018)	Bats	France	Acoustics	Activity or abundance	Gleaning and fast-flying bats were more active in hedgerows that were at least 1 km from turbines, therefore the EUROBATS 200 m buffer is likely inadequate
Gaultier et al. (2023)	Bats	Finland	Acoustics	Activity or abundance	Increased activity and occupancy at least 600 m from turbines
Leroux et al. (2022)	Bats	France	Acoustics	Activity or abundance	Turbines close (<43 m) to hedgerows deter bats, while they can increase activity and collision risk for some guilds when sited at intermediate distances (44–100 m). Consequently, they should be sited at least 100 m away from key habitat features
Reusch et al. (2023)	Bats	Germany	Telemetry	Activity or abundance	Bats most active at turbines <500 m from tree cavity roosts. Recommends a buffer distance of at least 500 m
Bose et al. (2020b)	Birds	Germany	Carcass search	Fatalities	Buzzard collisions greatest when turbines were <1 km from watercourses, <750 m from grassland, 750 m – 1750 km from open spaces, and <1.5 km from bushland edges
Lin (2017)	Birds	Taiwan	Carcass searches	Fatalities	Both passerines and waterbirds appear to collide more with turbines located close to fresh waterbodies
Balotari-Chiebao et al. (2016a)	Birds	Finland	Telemetry, Predictive model	Collision risk	Predict that collision risk is high for fledging White-tailed Eagles when turbines are constructed close to nesting sites, evidence supports the 2 km buffer prosed by the WWF White-tailed Eagle Working Group
Howell et al. (2020)	Birds	Canada	Telemetry, Predictive model	Collision risk	From estimated ground speed and published climb rates, predicted that shorebirds need a setback of 2–14 km to clear turbines 165 m tall after take-off

Study	Taxon	Country	Method(s)	Focus	Recommendation/finding
Murgatroyd et al. (2021)	Birds	South Africa	Telemetry, predictive model	Collision risk	For Verreaux's Eagle, advocate for the use of predictive collision risk model incorporating multiple factors (incl. nests, slope, elevation) to determine turbine placement, as opposed to circular buffers of nest sites
Salomon et al. (2020)	Birds	Germany	Simulation	Collision risk	Ecological-economic simulations indicate the buffers for Red Kite nests should be at least 1 km
Fernández-Bellon et al. (2019)	Birds	Ireland	Field observations	Activity or abundance	The density of forest species is lower within 100 m of turbines, no significant effect of distance on open-area species
Gómez-Catasús et al. (2018)	Birds	Spain	Field observations	Activity or abundance	Dupont's Larks need buffers of 4.5 km for their populations to remain unaffected by turbines
Hatchett et al. (2013)	Birds	United States of America	Field observations	Activity or abundance	Density and nesting success of Dickcissel (a small grassland bird) was unaffected by distance to turbines
Pearce-Higgins et al. (2009)	Birds	United Kingdom	Field observations	Activity or abundance	Reduced occupancy of various bird species (incl. Buzzard, Golden Plover, Curlew) within 500 m of turbines, but no significant effects of turbine proximity on the probability of raptors flying ‘at risk height’
Li et al. (2020)	Birds	China	Telemetry, field observations	Habitat use	A buffer zone of 800 m – 1.3 km needs to be established inland from a dyke adjoining migratory waterbird habitats at Chongming
Veltheim et al. (2019)	Birds	Australia	Telemetry	Habitat use	A buffer of 1.6–2 km from the centre point of wetland night roosts used by Brolgas is recommended to avoid impacts on breeding success in Victoria
Venter et al. (2019)	Birds	Botswana, Lesotho, South Africa	Telemetry	Habitat use	A buffer of 50 km should be applied around colonies of the Cape Vulture
Watson et al. (2014)	Birds	USA	Telemetry	Habitat use	Recommends 12.8 km buffers around Golden Eagle nests, or 9.6 km if areas of slopes and ridges are also included. The integrity of the core 3.2 km home range around the nest must be protected.

6.3 Automated detection systems

Moderate number of papers (22 papers)	Low bird bias (86%)	Low geographic bias (Europe 50%)

Various systems have been developed to automatically detect and identify birds and bats when they are approaching or are within wind energy facilities, to trigger subsequent mitigation actions such as turbine shutdowns. Here, we focus primarily on Automatic Detection Systems (ADS) that use machine learning or AI algorithms for image- and video-based species recognition. Acoustic surveys and associated ADS for bats are discussed above in the pre-construction context (Section 5.5.1), and also below in reference to their being incorporated into smart curtailment (Section 6.1.2), so they will not be covered here.

ADS is a rapidly-developing field, and Principato et al. (2023) conducted a systematic literature review to understand the state of machine-learning based bird detection research. They structured their review according to the 'reactive agent' framework, which essentially outlines the steps involved in applying machine learning algorithms to real-world problems. This framework includes: a) the sensors used (e.g. radar, acoustics, imagery); b) the format and amount of input data, including the data that are used to train and validate the model; c) pre-processing steps, such as data cleaning, transformations or conversions (e.g. audio to spectrograms); d) the model used (e.g. convoluted neural network, support vector machine, random forest); e) the output data and model performance (e.g. sensitivity and specificity); and f) resulting actions and outcomes – in this case, whether or not a mitigation action is triggered. While Principato et al. (2023) note that many of the studies included in their review reported high accuracy in identifying target species (typically around 80–90%), they also flag the need for both widespread testing in real-world settings, and the use of standardised performance assessment approaches.

Interestingly, the Principato et al. (2023) review does not include literature related to the IdentiFlight® proprietary ADS for detection and on-demand shutdown, which was the most well-studied system in our review. IdentiFlight® was originally developed in 2012 to address the specific issue of eagle collisions with wind turbines in the USA, but is now used globally, including in Tasmania. It consists of a network of tower-mounted cameras that feed imagery back to a data processing station, which then uses machine learning algorithms (convoluted neural networks) for real-time identification and detection. The identification of target species within a pre-specified distance from a turbine then automatically triggers shutdowns without the need for human intervention (McClure et al. 2021).

At a wind energy facility in Wyoming, McClure et al. (2018) ran field trials to compare the efficacy of IdentiFlight® to experienced human observers in detecting birds (and specifically eagles). They found that the IdentiFlight® system detected 96% of the bird flights detected by human observers, and that it detected 562% more individual birds. The system's false negative rate (i.e. not detecting an individual when it was present) for eagles was 6%, and the false positive rate (i.e. mistakenly saying individuals were present) was 28%. A subsequent BACI study conducted by McClure et al. (2021) compared eagle fatalities at the same facility, where turbine curtailment was triggered by the IdentiFlight® system, with eagle fatalities from a neighbouring facility 15 km away, where turbines were not curtailed. They found that fatalities declined at the treatment site by 63%, and increased at the control site by 113%, such that there was an overall reduction of eagle fatalities by 82%. These results sound substantial, but it is worth noting that both the assumptions underlying the study design (only one treatment site in one year) and also the analytical approach (ignoring annual variation, decisions about data inclusion, assignment of 'before and 'after' period) have been questioned (Huso et al. 2023). While the authors of the original study have responded to these criticisms (McClure et al. 2023), the controversy nonetheless highlights the need for robust and transparent testing of any new mitigation approaches in a variety of settings.

Two further studies have experimentally tested the IdentiFlight® system. Duerr et al. (2023) conducted a study at a Californian wind farm where the system was installed, and was recording bird detections, but not yet triggering on-demand curtailment. They assessed detections made over the span of a year, broken into five periods corresponding to a series of modifications that were made to both the identification algorithms and equipment. They found that while the system correctly identified 77% of eagles, there was overall a 13% false negative rate and a 20% false positive rate. The authors also found, that had curtailment been in place, it would have been triggered six times as frequently by false positives (non-eagles, and especially ravens), than true positives (actual eagles). It was concluded that the significant cost of both the system, and falsely triggered curtailments, needed to be carefully weighed up against potential conservation benefits (avoided collisions with vulnerable species), and that this balance is likely to depend upon both the ecological setting and the objectives of the operator. The other study, conducted in Germany by Mund (2023), was a field-based comparison (though not within a wind energy facility) of detection rates of Red Kites and White-tailed Eagles achieved by human observers, the BirdScan Radar system, and what they call 'mobile IdentiFlight'. It is unclear whether this IdentiFlight is the same system as the proprietary technology discussed above, though detection rates appear to be >90% for both it and the BirdScan Radar (used separately) to a distance of ~1400 m.

Data collected by IdentiFlight® and other ADS can be used for purposes other than just informing curtailment decisions in real-time. For example, Rolek et al. (2022) used IdentiFlight® images to model the probability that Golden and Bald Eagles would enter the RSA, based on the distance to the nearest turbine and flight height. This model and the associated parameters could, in turn, inform future curtailment criteria. McClure et al. (2021) also analysed IdentiFlight® images, but focussed instead on identifying temporal and spatial patterns of eagles entering the RSA, so specific high-risk turbines and seasons could be identified.

While the IdentiFlight® system was the most well-represented ADS in the peer-reviewed literature, other approaches have been proposed. Scholz et al. (2016) presented a new algorithm (the random bounce algorithm) for processing video imagery, and demonstrated its potential with small pilot tests for Sebas Short-tailed Fruit Bats in flight tunnels, and raptors at a wind energy facility in Germany. Alqaysi et al. (2021) tested the ability of a YOLOv4 (artificial intelligence) model to detect birds in greyscale videos collected from wind energy facilities in Denmark, and found it achieved average precision rates of 60–92%. Finally, Sheppard et al. (2015) propose the use of a ‘geofence’ telemetry system, which represents a risk zone around a wind energy facility. In this system target species (Californian Condors, in their case study) are fitted with GPS tags, such that when they cross the geofence and enter the risk zone, an SMS alert with the animal’s identifier and GPS location is triggered, and mitigation actions (such as shutdowns) can be implemented. While this may be effective, it is resource-intensive and relies on the capture and tagging of at-risk individuals, so is unlikely to be suited to many target species, and the study did not actually field test its effectiveness in reducing collisions.

As noted above, the Principato et al. (2023) review highlighted the need for standardised assessment of ADS, to facilitate comparison of performance between different systems. Ballester et al. (2024) addressed this need by developing an assessment protocol, which evaluates the performance of ADS against four criteria: a) the temporal and spatial coverage achieved; b) the detection rates for target species; c) the rate of accurate classification; and d) the reaction time, or how quickly a shutdown is triggered once the target species has been detected. They field-tested the protocol on three different types of ADS (one using 2D cameras, one using 3D cameras, and another using radar), across five different wind energy facilities in France, comparing detection rates with those achieved by human observers with laser range finders. They found that while the protocol was generally practical to implement, there were some limitations. Most notably, it was difficult for humans to estimate distance to a bird when there was a background of vegetation (because it was difficult to focus the range finders on the bird), and this is also a known issue for ADS systems, so the detection probabilities of the humans and ADS may not have been independent in these cases. Comparisons were also difficult to make when groups of birds were present, because it was unclear which system was referring to which bird in a group.

More generally, however, the Ballester et al. (2024) paper is a good example of how challenging it can be to interpret the wind energy and mitigations literature. These studies often involve the use of proprietary software or hardware, so specific details about settings, parameters etc. are purposefully omitted to protect intellectual property rights, and tests are conducted at private facilities that may be concerned about scientific findings potentially jeopardising operations. Ballester et al. (2024) state “*Because these tests were performed under an agreement of confidentiality with private wind power plant operators, we cannot explicitly mention the precise site locations, the ADS brand and model tested, or the exact performance results obtained*.” Without these details, it is very difficult to determine how applicable findings may be to a different geographic context, or where different target species may be involved.

Knowledge gaps for Victoria

38. No published findings are available on the performance of any type of ADS in Victoria. While the IdentiFlight® system appears to be effective at preventing collisions with Golden Eagles, which are broadly comparable morphologically to Wedge-tailed Eagles, it is unclear whether the issue of false detections (and resulting curtailments) triggered by ravens may also be an issue.
39. Although the IdentiFlight® system is currently operating at one wind farm in Tasmania, with anecdotally promising results for Wedge-tailed Eagles, the results of this study are not yet available in the scientific literature to examine the details of these findings.
40. Almost all published studies of the effectiveness of ADS in reducing bird collisions have focussed on large raptor species, so it is difficult to determine how they may perform for smaller and less visually distinguishable bird species.
41. It is possible that ADS systems could be used for flying-foxes in a similar way to birds during dawn or dusk periods, noting that flying-foxes are active throughout the night and these systems are currently

ineffective when it is dark. Further developments to incorporate other technologies that work in night-time conditions (such as thermal or infrared imaging) could help to address this, but these are not currently available.

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6.4 Deterrents and increasing turbine visibility

Moderate number of papers (21 papers)	No taxonomic bias	Low geographic bias (USA 48%)

A range of deterrents have been proposed as a means of discouraging birds and bats from flying in the airspace around turbines, to reduce fatalities. These commonly involve the use of lights, sounds, and radar, either individually or in combination. While we did not identify a strong taxonomic bias in the literature *per se*, there were far more experimental studies that had tested deterrents *in-situ* for bats than there were for birds, so we discuss bat-specific studies first and present a summary of the evidence in Table 15. It is worth noting that the risk of using a wildlife deterrent in any management context, including at wind facilities is that it will cause habituation (Arnett 2016), or worse, attraction (Cryan et al. 2022). Many of the studies discussed below have only been conducted over relatively short time-frames, so do not necessarily provide insights into whether this has occurred.

For bats, the bulk of the experimental evidence pertained to acoustic deterrents, which have been tested as both continuous and pulsing sounds ranging between 20–100 kHz (Table 15). Three BACI experiments (Arnett et al. 2013; Romano et al. 2019; and Weaver et al. 2020), all from the USA, assessed whether acoustic deterrents reduced bat fatalities, and found them to be effective for some of the species and in some years. Estimated fatality reductions ranged from ~25–80%, and were most consistent for the Hoary Bat. Two further studies from the UK (Gilmour et al. 2020, 2021) assessed reductions in bat activity (as opposed to fatalities) associated with the use of acoustic deterrents. These studies also concluded that deterrents were more effective for some species than others, in this case most consistently for the Common Pipistrelle. Despite these studies showing some promise, Arnett et al. (2013) cautioned that the effectiveness of ultrasonic deterrents will be limited by the distance that ultrasonic signals can be broadcast, and that they will attenuate more rapidly in humid conditions. It is also worth noting that Good et al. (2022) found that curtailment alone led to the greatest reduction in fatalities, and the effect of the addition of the acoustic deterrent was marginal.

Four further studies tested the effectiveness of different types of lighting as a potential deterrent for bats. Two of these (Cryan et al. 2022; Gorresen et al. 2015) focussed on ultraviolet light, and two (Jain et al. 2011; Bennett and Hale 2014) on the effect of the red aviation lighting that is often required to be mounted on turbines to enhance visibility to aircraft. The findings of these studies were inconclusive, and they were lacking a before-versus-after study design. Two other studies (Nicholls and Racey 2009; Gilmour et al. 2020) investigated a radar deterrent, with the former finding it to be ineffective compared with the acoustic deterrent tested in the same study, and the latter reporting a 30–39% reduction in activity when medium pulse lengths were used.

Evidence for the effectiveness of various deterrents in mitigating collisions for birds is very patchy. The most well-known example comes from the Smøla wind energy facility in Norway (May et al. 2020), where the authors found that painting a single blade black to increase visibility reduced bird fatalities by over 70%. These findings have garnered considerable media attention globally, which has advocated blade painting as a simple and effective solution for reducing bird collisions (May 2023). This is despite the authors cautioning in the original study that Smøla is a very specific context ecologically, and that more research is required in a variety of settings with different target species before such generalisations should be made. Another study from Smøla has demonstrated that painting not only the turbine blades but also the bottom 10 m of the towers can reduce collisions for Willow Ptarmigan by 48% (Stokke et al. 2020, see Figure 7). However, this is a medium-sized grouse species that spends time on the ground in areas where the white bases of the turbines blend into the surrounding snow cover, quite dissimilar to Victorian bird Species of Concern.

Besides these Smøla examples, which are more about enhancing visibility as opposed to creating a deterrent, only a handful of other approaches have been tested for birds. Boycott et al. (2021) tested the efficacy of an acoustic deterrent (4–8 kHz), deployed in front of communication towers in Virginia, for migrating birds. They recorded a 12–16% reduction in activity levels, and observed birds reducing their flight velocities and deflecting flight trajectories. Kerlinger et al. (2010) synthesised post-construction fatality monitoring data from across the US and Canada and concluded that there was no difference in fatalities between turbines mounted with and without flashing red aviation lights. Dorey et al. (2019) trialled a simulated predator deterrent (an animatronic owl, as well as playback of predator and alarm calls). Findings from this study were inconclusive, though sample sizes were small.

Knowledge gaps for Victoria

42. No deterrents have been experimentally trialled for birds or bats in Victoria. While acoustic deterrents could potentially reduce bat fatalities, international evidence suggests that responses are species-specific and there are limits to the areas over which sounds can be broadcast, so are unlikely to be able to cover the whole RSA. These will need to be tested *in-situ*, within the Victorian environment.
43. While a single, well-publicised study in Norway has indicated that increasing turbine blade visibility was an effective means of reducing bird collisions, it remains uncertain whether such an approach would work in Victoria.
44. The deterrent effect of lighting is unknown, as is the potential for insects to be attracted to artificial lighting on turbines, and therefore increasing the risk to microbats and nocturnal insectivorous birds.
45. There is no information on whether there may be changes in bat and bird behaviour in response to deterrents being used over the long term (e.g. if they become desensitised to the deterrents over time).

Key references

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Figure 7. Smøla vindpark, showing snowy conditions and a turbine (right) painted at the base to reduce collisions for Willow ptarmigan. Source: Emelysjosasen, CC BY-SA 4.0.

Table 15. Bat deterrents: Field experiments assessing the effect of acoustic, light and radar deterrents on bat activity or number of fatalities.

Rows are arranged by deterrent type, then the response variable measured (fatalities, then activity), and then by author. 'Study design' indicates the contrasts used, and corresponds to combinations of: 'B' before, 'A' after, 'C' control, 'I' impact (single or multiple deterrent treatments).

Study	Country	Deterrent type	Settings	Study design	Response	Summary
Arnett et al. (2013b)	USA	Acoustic (ultrasonic)	Continuous broadband ultrasound, 20–100 kHz	BACI	Fatalities	First year: significant difference for 1/7 species, 2.1 times as many Hoary Bats killed at controls compared to treatments. Second year: significant difference for 2/7 species: 1.9 times as many Hoary Bats, and 3.8 times as many Silver-haired Bats killed at controls compared to treatments.
Romano et al. (2019)	USA	Acoustic (ultrasonic)	Broadband 30–100 kHz (effective to ~30 m). Continuous year 1 and 2, pulsed year 3.	BACI	Fatalities	The deterrent resulted in significant overall bat fatality reductions of 29.2% and 32.5% in year 1 and 2, but not in year 3. Reduced Hoary Bat fatalities in year 1 (26%), year 2 (36%), and year 3 (34%). Reduced Silver-haired Bat fatalities in year 2 (57%), and year 3 (73%). Reduced Eastern Red Bat fatalities in year 1 (39%) only.
Weaver et al. (2020)	USA	Acoustic (ultrasonic)	Continuous broadband 20–50 kHz	BACI	Fatalities	Use of deterrents reduced fatalities for all bats combined (by 50%), and Hoary Bats (78%) and Mexican Free-tailed Bats (54%) individually. Mexican Free-tailed Bats made up 78% of fatalities. No significant reduction in fatalities of Northern Yellow Bat.
Gilmour et al. (2021)	UK	Acoustic (ultrasonic)	Continuous broadband 20–100 kHz	CI	Activity	Overall bat activity was reduced by 30%, bat flight speed also increased, and they flew more directly. Significant reduction in the number of passes of the Soprano Pipistrelle (by 27%), Daubenton's Bat (26%), and *Nyctalus* spp. and *Eptesicus* spp. (68%).
Good et al. (2022)	USA	Acoustic (ultrasonic) [combined with curtailment]	Continuous broadband 20–50 kHz. Curtailment to 5.0 m/s.	CI	Fatalities	Overall bat fatality rates were 66.9% lower at curtailed turbines with acoustic deterrents compared to turbines that operated at manufacturer cut-in speed. The addition of the deterrent treatment to curtailed turbines led to a significant decrease in fatalities of Big Brown Bats and Silver-haired Bats, and a marginal decrease for Hoary Bats and Eastern Red Bats. [note: no deterrent-only treatment]
Gilmour et al. (2020)	UK	Acoustic (ultrasonic), radar	Radar: pulse length 0.3 μs, repetition rate 1200 Hz, peak power 6 kW Sound: continuous broadband 20–100 kHz	CI	Activity	Ultrasonic deterrents decreased overall bat activity by approximately 80%, when deployed alone and in combination with radar. Difference driven by the fact that *Pipistrelle* spp. were deterred by the ultrasonic treatment (40–80% reduction), while *Myotis* spp. were not. Radar alone had no significant effect on bat activity.

Study	Country	Deterrent type	Settings	Study design	Response	Summary
Werber et al. (2023)	Israel	Drone mounted with light and acoustic deterrent	Drone: DJI Phantom 4 Pro. Sound: Sweeping chirps 15-80 kHz. Lights: Flashing LEDs (50W each, 400–780 nm, 6200–6800 K).	BA	Activity	There was a significant ~40% decrease in bat activity below the drone, and a ~50% increase in activity above it, while it was in flight, compared to the period before or after. [Note: not tested in a wind energy facility]
Jain et al. (2011)	USA	Lights (aviation)	Blinking red beacons, non-blinking red beacons, and combined non-blinking red beacons and blinking white beacons.	I	Fatalities and activity	No relationship between types of turbine lights and either collision mortality or echolocation activity [Note: no controls]
Bennett and Hale (2014)	USA	Lights (red aviation)	Red strobe, peak intensity of 2000 candela, ~30 pulses/min, mounted at 82.5 m	CI	Fatalities	Aviation lights reduced fatalities of Eastern Red Bats by 46%, no significant effect for five other species
Cryan et al. (2022)	USA	Lights (UV)	UV light, peak wavelength of 365 nm, power density ~1 $\mu W/cm^2$ over a 20 m radius at 30 m from the light source	CI	Activity	No significant effect of lights on total bat, bird or insect activity, but observed an increase in high-risk bat behaviours (viewed through thermal cameras) indicated a possible attractant effect
Gorresen et al. (2015)	USA	Lights (UV)	UV light, power density ~1 $\mu W/cm^2$ over a 20 m radius at 30 m from the light source	CI	Activity	Dim UV light reduces Hawaiian Hoary Bat activity (44% fewer detections) despite an increase in insect numbers, but did not completely inhibit bat activity near trees. [Note: tested in trees, not at turbines]
Nicholls and Racey (2009)	UK	Radar	Pulse length of 0.08 μs, repetition rate 2.1 kHz, and pulse length of 0.3 μs, repetition rate 1.2 kHz. Peak power 6 kW	CI	Activity	Bat counts and bat passes significantly dropped by 38.6% and 30.8%, respectively using a medium pulse length signal (0.3 μs) from a fixed antenna. Bat counts and bat passes were also lower (though not significantly) using the short pulse length signal (0.08 μs) from either a fixed or rotating antenna.

6.5 Other mitigations

Few studies suggested or presented alternative mitigation measures to those discussed above, but that is not to say that other, novel approaches that take into account the unique ecology of Victorian landscapes and species should not be considered. Indeed, these may be necessary for species, like the Grey-headed Flying-fox, that are not readily comparable to species studied elsewhere, and that may require tailor-made mitigations. An international example of such an approach comes from Spain, where fatalities of the Lesser Kestrel were reduced by 75–100% across three wind energy facilities by tilling the area surrounding the towers of the most problematic turbines (Pescador et al. 2019). This tilling disturbed the vegetation and soil in the area in such a way as to reduce availability of the kestrel's prey (predominantly Orthoptera, i.e. crickets and grasshoppers), subsequently making it less attractive to the kestrels for hunting. Hunt and Watson (2016) also suggest making areas close to turbines less attractive to raptors by reducing the availability of prey (in their example, California Ground Squirrels) as well as sites for nesting and perching. What these measures may look like for species like the Grey-headed Flying-fox are unclear, but it does warrant creative thinking or perhaps consideration of combining mitigation measures in novel ways. However, any such destructive changes would need to be balanced against the benefits that these habitats provide to other biodiversity in the area.

6.6 Compensation and adaptive management

Small number of papers (12 papers)	Low bird bias (75%)	Moderate geographic bias (USA 58%)

The mitigation hierarchy comprises three steps that need to be followed when attempting to address adverse impacts to biodiversity that are caused by human activities, and specifically developments (Peste et al. 2015). For wind energy, the first 'avoid' step takes place during regional planning processes (see Section 5.2) and through siting decisions (see Section 5.4.2). The mitigation measures described above (see Section's 6.1– 6.5) represent the second 'minimise' step, and are intended to reduce the number of fatalities caused by collisions with turbines. The final 'compensate and offset' step occurs when residual impacts remain that cannot be avoided or minimised despite all efforts, so measures are applied either on- or off-site to enhance populations of the impacted species. Given the difficulties of predicting fatalities from pre-construction surveys (see Section 5.5.4), compensation measures may also be required when there are unforeseen impacts on species at risk that are detected during the post-construction phase (Arnett et al. 2016).

The literature related to compensation (and offsets) in general is large, complex, and well-established, and applies to a suite of impacts including those that can occur as a result of the construction and operation of a wind energy facility, such as habitat loss and shifts in community composition, and sublethal effects from sensory pollution (noise and vibrations). While compensation was not specifically intended to fall within the scope of our review, and it was not included as a key term in the literature search (Table S1), it was nonetheless mentioned or explicitly addressed in several of the papers yielded by the search. Here we provide a brief overview of the most pertinent studies and perspectives.

All authors emphasised the importance of treating compensation measures as a 'last resort' that are only implemented once the other steps of the mitigation hierarchy have been followed, and only when fatalities cannot be reduced to acceptable levels by avoidance and true mitigation (e.g. Marques et al. 2014; Peste et al. 2015; Arnett and May 2016; Agha et al. 2020). Compensation and offset measures are intended to result in a net-neutral or positive outcome for populations of impacted species. However, this is extremely difficult to demonstrate when the impacts of collision fatalities are immediate and pronounced, while the benefits of compensatory measures are typically challenging to quantify and occur over much longer time frames. It is almost impossible to achieve no-net-loss when the main cause of population declines for a species is collision fatalities, or it is long-lived, rare, or declining (Carrete et al. 2009; Arnett et al. 2016; Voigt et al. 2024). Therefore, some authors stress that the 'no go' option should be applied in cases where irreversible losses will occur (Peste et al. 2015), because evidence that demonstrates the achievement of no-net-loss for wildlife populations is lacking in general (Voigt et al. 2024).

The requirement for offsets or compensatory measures applies in only a few countries globally, namely in the EU (Voigt et al. 2024). Peste et al. (2015) reviewed both the scientific and the grey literature concerning bat fatalities at wind energy facilities in Europe, and developed a list of thirteen potential compensatory measures that fell within five broad categories: a) management of autochthonous (i.e. indigenous) forest; b) diversification of forest and agriculture; c) preservation of existing roosts; d) provision of new roosts; and e) creation of ponds. Peste et al. (2015) also acknowledged that these measures were yet to be adopted in European compensation schemes, and that their effectiveness in offsetting bat fatalities had not been measured. Because of the aforementioned difficulties in predicting the potential impact of a facility (see Section 5.5.4), and in turn quantifying the extent of compensation that may be required, they advocate for an adaptive management approach where the mitigation hierarchy is re-evaluated as new information is acquired during the construction process and through post-construction monitoring.

The USA has adopted one such adaptive management approach (New et al. 2021), specifically in response to the large number of Golden and Bald Eagle fatalities that were being recorded in areas such as the Altamont Pass Wind Resource Area, where an estimated 40–60 Golden Eagles were being killed each year (Hunt and Watson 2017). These species are protected under the *Bald and Golden Eagle Protection Act 1940,* such that any potential fatalities may violate US law. To address this, the US Fish and Wildlife Service (FWS) issues permits to operators to 'take' a limited number of eagles, in the same way they might issue permits to a hunter. The take limits for each facility are informed by the number of fatalities predicted by a collision risk model (New et al. 2015), and the estimated capacity of the species' populations to withstand additional mortality. If the allowed take exceeds this estimated capacity, then compensatory measures must be taken (New et al. 2021). If a facility is found to be 'taking' species at a faster rate than anticipated, then under the adaptive management framework they may be required to apply additional mitigation measures to avoid having excessive impacts on the population (Huso et al. 2016). In the adaptive framework, post-construction fatality data that are collected over time are also intended to allow the FWS to update and improve the collision risk model, so they may more accurately assess the risk that a facility may pose to a species and what offsets or compensatory measures are required (New et al. 2015).

Only one compensatory measure has been approved for use in the US eagle scheme, which is the retrofitting of powerpoles to prevent electrocutions, and we did not identify any US-based studies that had assessed this approach. However, Cole et al. (2013) used a Resource Equivalency Analysis, and data representing the number of White-tailed Eagles killed by both collisions with wind turbines and electrocutions from powerpoles around the island of Smøla in Norway, to estimate compensatory requirements. Their modelling suggested that 348–2,209 pylons would need to be retrofitted, at a cost of $1.2–7.9 million USD, to offset the impact on eagle populations and the social values attached to them.

Lonsdorf et al. (2018, 2023) assessed the efficacy of one other potential mitigation measure that is not currently incorporated into the US scheme, which is the removal of ungulate carcasses from roads to prevent scavenging raptors from colliding with vehicles. Lonsdorf et al. (2018) developed a model of the relationship between eagle fatalities, carcass densities and traffic volume, and predicted that increasing the frequency of removals from zero to five per month resulted in a 30% reduction in eagle mortality. They also suggest how this model could be used to inform mitigation credits. Following on from this, Lonsdorf et al. (2023) updated the model to incorporate new data from both additional monitoring and also camera trap observations, and concluded that up to seven eagles per county could be saved each year through carcass removals.

While these measures are quite specific, and particularly focussed on high-profile raptors, Marques et al. (2014) provide a more general review of the impact of wind energy facilities on birds and some potential compensatory approaches. They emphasise that specific actions should be selected on the basis of the threats limiting target populations in each area. They note that bird populations can be enhanced by: a) creating roosting, foraging and nesting habitats; b) increasing the availability of prey; c) controlling predators; d) removing invasive species; e) reintroducing species; and f) supplementary feeding. They also note that impacts on populations can be minimised by: a) applying minimisation measures to other human infrastructure (e.g. reducing electrocutions, collisions with vehicles); b) minimising human disturbances; and c) conducting awareness campaigns. Measures such as these may be appealing from an implementation perspective, but it is unlikely that the associated benefits could compensate for the immediate and known impacts of collision fatalities.

Knowledge gaps for Victoria

46. The extent to which recovery actions could compensate for the residual impacts of fatalities on Victorian Species of Concern populations is unclear.

Key references

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7 Post-construction monitoring

7.1 Field monitoring

Large number of papers (89 papers)	Low bird bias (71%)	Low geographic bias (USA 41%)

Once a wind energy facility has been constructed, there is typically a monitoring period during which carcass searches take place. It is important to maximise the probability of carcass detection during this time, in order to accurately estimate actual fatalities. Several authors (e.g. Huso 2011; Korner-Nievergelt et al. 2011) have found that low searcher detection probabilities can strongly bias estimates such that it appears that facilities cause fewer fatalities than they actually do, especially when the number of fatalities per year is low. It is also important to achieve an accurate profile of the species being impacted, and especially those that are rare or threatened.

The monitoring period can span one to many years. In the case of the US where incidental take permits for protected species have been issued since 2006 to account for wind energy related fatalities (Huso et al. 2015), monitoring must occur at set intervals (e.g. every 3 months) throughout the permit term, which is up to 30 years (Hallingstad et al. 2023). Both the frequency and duration of carcass searches can affect the number of individual carcasses and species detected, and the resulting estimates of the facility's impact. For example, Smallwood et al. (2017) compared the findings from searches that were conducted every five days to those that were conducted on average every 39 days. They found that the five-day frequency searches resulted in more species, and also a greater number of individual carcasses being detected, which in turn yielded fatality estimates that were 39% higher. Similarly, Beston et al. (2015) compared data from searches conducted across 50 different US wind energy facilities and also found that a greater number of species were detected where searches were conducted more frequently (ideally weekly). This is likely because smaller species (such as microbats and small birds) are scavenged more quickly (see Section 7.1.3 below), and because it is difficult to identify carcasses that are in an advanced state of decomposition, which occurs when there is too long a period between searches.

The other question is how long to survey for post-construction. While regulations and guidelines may stipulate that monitoring is only required in the years immediately following construction, studies have shown that the number of species detected underneath turbines at a facility can continue to grow for many years, as each subsequent survey is carried out. This may especially be the case as species' attraction to, or deterrence from, the turbines changes over time (see Section 5), or when the environmental drivers of flight patterns such as the availability of water, food sources, and nesting/roosting sites vary substantially between years (e.g. Martínez-Abraín et al. 2012). In the same US data synthesis mentioned above, Beston et al. (2015) found that for two of the facilities in their dataset that had continued monitoring up to and beyond a year, neither appeared to have reached a plateau in the number of species being found. Likewise, in Tasmania, Hull et al. (2013) found that new bird species continued to be detected during carcass searches up until around the 7-year mark, when the species accumulation pattern started to taper off.

Hence, one can't assume that the species detected in carcass surveys in the years immediately following construction are the only ones that are being, or will be, impacted by collisions. This is especially important in the Victorian context, where the ranges of threatened species such as Grey-headed Flying-foxes continue to shift. These species may move to new areas and start to collide with turbines even once the required monitoring period is complete.

Once a survey schedule has been set, there are several reasons why a searcher may fail to detect a carcass during a search:

- It has fallen outside of the search area after collision.
- The searcher fails to detect it.
- It has been removed from the search area by a scavenger in the period between the collision occurring and the search taking place.
- The turbine that it has collided with is not searched as part of the monitoring program.

Each of these is a potential source of bias that must be accounted for, and we outline the approaches used to correct for these biases below (see Sections 7.1.1, 1.1.1, and 7.1.3, summarised in Figure 8). A fatality estimator uses data from the carcass search, searcher efficiency and carcass persistence trials to derive estimates of the number of individuals actually being killed in a given period (as opposed to how many are being detected). We discuss how the choice of estimator may also influence findings (see Section 7.1.4).

As noted above, imperfect detection is particularly problematic for rare and threatened species. While these species may only collide with turbines infrequently, they are also more likely to go undetected, and when there are few individuals left of that species there will be a disproportionately large impact on the population. A Victorian example of this might be the Critically Endangered Orange-bellied Parrot. While it has not yet been recorded colliding with turbines, as a small-bodied species it is more likely to be scavenged quickly and subsequently disappear before the next survey is conducted, and there are so few individuals remaining that any losses would be significant. In Section 7.1.4, we also highlight approaches that have been proposed to address this issue of demonstrating 'Evidence of absence'.

Once a carcass has been detected, another source of bias can be identification, which becomes especially difficult once carcasses have started to decay. In their study, Smallwood et al. (2018) found that 44% of bird species (and especially small species) placed as carcasses were subsequently misidentified to species by experienced searchers when found some time later. Sex is also important to record because inherent differences in morphology or behaviours might lead to individuals of one sex being more vulnerable to collisions than the other, which could, in turn, have flow-on effects on overall population structure and effective population size (e.g. Hueck et al. 2020). However, sex can also be difficult to deduce from carcass remains.

For this reason, some authors advocate for the use of DNA barcoding approaches as a means of validating field identification. Chipps et al. (2020) used molecular approaches to assess 439 bat carcasses collected from wind energy facilities across southern Texas, and in doing so improved species identification from 83% to 97%, improved sex assignment from 35% to 94%, and detected two species (39 individuals) outside of their previous known range. These species would not have been considered in pre-construction risk assessments and/or acoustic surveys. Korstian et al. (2016) also used DNA barcoding to analyse 892 bat carcasses salvaged from the US (mostly north Texas), and in doing so improved identification by 3%. They also noted that rates of misidentification occur more frequently when carcasses were more than a day old, and were worse for some species (i.e. the Tricolored Bat) than others. In these studies, surveys were conducted relatively frequently (every 1–6 days), but in Victoria searches typically occur at monthly intervals, so DNA barcoding should improve identification rates even further. For example, in Victoria for data collected up until 2018, 22% of birds and 4% of bats were not identified to the species level (Moloney et al. 2019), and it is unclear what the rates of misidentification were for those carcasses where species identifications were assigned.

Here we focus our discussion predominantly on survey protocols that prescribe separate carcass searches, searcher efficiency, and scavenger/carcass persistence trials, as is standard globally. However, alternatives have been proposed where some of these elements are combined or are not needed. Smallwood et al. (2018) advocate for the use of 'integrated detection trials', where detection errors associated with searcher efficiency and carcass persistence are estimated at the same time, and both placed and found carcasses are left for the entire monitoring period. They suggest that conducting surveys this way is more cost effective, reduces sources of bias more effectively than conducting the trials separately, and generally results in lower estimated fatality rates. Others (Etterson 2013; Péron et al. 2013) have proposed the use of double-sampling approaches to estimate detection, thereby also removing the need for separate searcher efficiency and carcass persistence trials (discussed below in Section 7.1.4). While elements of these approaches may seem appealing, they do not appear to have been widely adopted and tested, making it difficult to ascertain how their use may affect inference about fatalities in different environmental contexts.

Knowledge gaps for Victoria

47. It is unclear what survey schedules (with regards to both frequency and duration) for post-construction mortality monitoring will be optimal in different regions of Victoria, to both maximise detection probabilities of target Species of Concern, and ensure cost efficiency.
48. Little has been published on how decisions about the number or proportion of turbines searched at each facility can introduce biases. This is despite evidence that some individual turbines may present much greater risks than others, and if these are missed as part of a search protocol, fatalities may be greatly underestimated. This warrants further analysis and exploration.
49. Current rates of misidentification of carcasses (both species and sex) in Victoria are unknown. DNA metabarcoding could be used to test the accuracy of fatality monitoring identifications.
50. Integrated detection trials have the potential to reduce post-construction mortality monitoring efforts, but it is not clear whether these would be suitable for the Victorian context and how fatality estimates may be affected.

Key references

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7.1.1 Search area and fall distance distributions

An important decision that must be made in any post-construction monitoring program is the area underneath turbines that will be searched, and in turn, how to statistically correct for the carcasses that fall outside of this area and therefore go undetected. This is also assuming that the animal dies when it is struck by the turbine blade, when some individuals are likely wounded and will survive long enough to move away from the search area (Hull and Muir 2010; Smallwood and Bell 2020) and cannot be detected.

Various plot shapes are used in these searches including squares and circles of various sizes centred on the turbine, or arcs downwind of the turbine (where carcasses tend to fall, e.g. Smallwood and Bell 2010). The density of carcasses around a turbine decreases with distance, because more will fall closer to the turbine, and the area that makes up the outer band of a plot is larger than the band closest to the turbine (Huso and Dalthorp 2014).

The relationship between distance from a turbine and the density of carcasses for a particular species at a particular site is the 'fall distribution', and knowing the shape of this distribution allows one to estimate what proportion of carcasses will have fallen outside of a search plot. There are many factors that can influence how far each carcass travels from the turbine, and in turn, the shape of the fall distribution, including characteristics of the turbines (height, diameter and power), the species' traits (mass, surface area, migration and flight behaviour), and the environment (prevailing wind speed and direction). Some of these patterns are summarised in Table 16, but in general, carcasses will fall farther when they are larger, and when collisions occur at higher wind speeds. While several authors have found that fall distance increases with hub height (e.g. Choi et al. 2020; Anderson et al. 2022), Garvin et al. (2024) contest this and state that effects are species-specific, and that there is stronger relationship with ground clearance.

In cases where carcasses are expected to fall farther, one should increase the size of the search area, as precision will decrease when a smaller proportion of total carcasses are detected. Some rules-of-thumb have been developed to help guide decisions about the search area to be used (see Prakash and Markfort 2022 Table 1, for bat-specific recommendations). These include circular search radii ranging from 55 m for bats to 122 m for birds (Hull and Muir 2010, though see assumptions below), a maximum of 80 m for both birds and bats, and a radius equivalent to the blade tip height of the turbine. It is worth noting that because increased wind speeds are associated with extended fall distances, it is reasonable to expect that when low wind speed curtailment is in place, bats will be struck on average at higher wind speeds, and therefore on average will also be falling farther, and be less likely to be detected. Therefore, some authors (e.g. Rabie et al. 2022) have cautioned against assuming the same fall distribution for control versus treatment (curtailed) turbines – because of it potentially leading to inflated estimates of mitigation effectiveness. Conversely, modelled fall distances that are based on observations made at lower wind speeds could underestimate mortalities at the higher wind speeds.

When several carcasses of a species have been detected at a given facility, one can fit a distribution to the raw data to estimate the proportion of carcasses that are likely to have fallen outside the search area. A data synthesis conducted by Choi et al. (2020) suggested that fall distributions are sometimes bimodal, that is, carcass densities peak at two different distances from the turbines, with one peak representing collision with the tower itself and the second collisions with blades. Huso and Dalthorp (2014) tested several approaches to fitting fall distributions to field data, and suggest that polynomial (quadratic) logistic regression models may be best in providing flexibility while minimising bias.

For species where carcasses are rarely detected, it will likely not be possible to fit site-specific distributions. In these cases, mechanistic models can be used to predict the proportion of carcasses that fall outside the search area, based on factors such as the species mass, the turbine height and rotor diameter, and environmental conditions such as wind speed. The Hull and Muir (2010) ballistics model is frequently used for this purpose, and allows users to predict fall densities based on parameters including species' mass and area, turbine height, rotational frequency and rotor radius, and air density. It is worth noting that this model makes a range of assumptions, most of which are unlikely to be met in most circumstances, including that: (1) there is no pre-collision velocity (i.e. that the animal is stationary when hit); (2) there is equal likelihood of strike anywhere in the RSA; and (3) conditions are calm (i.e. there is no turbulence or wind drift) and therefore the turbines are stationary or rotating only slowly at the time of collision.

Work by Prakash and Markfort (2020, 2021, 2022) has attempted to address some of the unsupported assumptions in the Hull and Muir (2010) model. This has resulted in the development of an alternative three-dimensional ballistics model specifically for microbats that better accounts for post-collision dynamics (Prakash and Markfort 2021), and the subsequent use of Monte-Carlo techniques to propagate uncertainty to predict fall distributions (Prakash and Markfort 2022). This work has demonstrated that predictions are quite sensitive to assumptions about where along the length of the turbine blades bat collisions most frequently occur (the 'radial strike location'), which is poorly understood and assumed to be uniform by Hull and Muir (2010). As a result, model predictions, and in turn survey guidelines, could be improved if more empirical information about radial strike location becomes available. They also note that there is limited information available about the flight speeds and trajectories of a range of bat species as they approach turbine blades. Nonetheless, model predictions of the two-dimensional location of carcasses from turbines (i.e. both the distance and direction) during the three-month migration period in Iowa (USA) for species ranging from 5–25 g closely matched the locations of bats found during searches, with fall distances predicted to range from 0–185 m (Prakash and Markfort 2022). This means that with data representing the distributions of wind speed and direction, turbine operational characteristics (yaw, blade rotation rate), and bat aerodynamic characteristics (mass, drag, flight speed, strike angle and locations), we can now hope to better predict the location of bat carcasses and, in turn, efficiently direct survey efforts.

Knowledge gaps for Victoria

51. Improved information about flight behaviours of Victorian species around wind facilities would greatly benefit the ballistic models used to predict fall locations. Specifically, empirical data about where on a turbine blade collisions are occurring (i.e. closer to, or father from the nacelle), and at what speed and angle individuals approach the turbines, are needed.
52. Data collected from current post-construction mortality monitoring in Victoria, where the fall distance has been recorded, could be used to test the ballistic models that inform search areas and correction factors.

Key references

Garvin, JC, Simonis, JL, Taylor, JL (2024) Does size matter? Investigation of the effect of wind turbine size on bird and bat mortality. *Biological Conservation* **291**, 110474.

Hull, CL, Muir, S (2010) Search areas for monitoring bird and bat carcasses at wind farms using a Monte-Carlo model. *Australasian Journal of Environmental Management* **17**, 77-87.

Huso, MMP, Dalthorp, D (2014) Accounting for unsearched areas in estimating wind turbine-caused fatality. *Journal of Wildlife Management* **78**, 347-358.

Prakash, S, Markfort, CD (2022) A Monte-Carlo based 3-D ballistics model for guiding bat carcass surveys using environmental and turbine operational data. *Ecological Indicators* **470**, 110029.

Table 16. Summary of literature focussed on the issue of fall distances and/or required search areas.

Study	Country	Study type	Taxon/species	Factors modelled	Key finding
Anderson et al. (2022)	Canada	Data synthesis	Birds, bats	Turbines: hub height, rotor diameter	A greater proportion of carcasses are expected to fall outside of the standard 50 m radius search area when turbines are taller
Choi et al. (2020)	USA	Data synthesis	Birds, bats	Turbines: hub height, rotor diameter. Species: mass, migration distance, migration timing, taxon.	As blade length and hub height increased, so did fall distance. Birds fall farther than bats, and larger-bodied species fall farther. For birds, short-distance migrants fell farther than long-distance migrants. Bird and bat fall distance distributions were bimodal (i.e. had two peaks), especially for birds, possibly because they also strike towers. Note: bird carcasses >100 m from turbines were removed from the analysis.
Garvin et al. (2024)	USA and Canada	Data synthesis	Hoary Bat, Horned Lark, Red-tailed Hawk	Turbines: rotor diameter, ground clearance, power	Taller turbines do not necessarily lead to farther fall distances, and modelled distributions need to be species-specific. Horned Larks fell farther on average from turbines with higher ground clearances, but Hoary Bats and Red-tailed Hawks fell closer. All three species had closer fall distances with increasing rotor diameter.
Hull and Muir (2010)	Australia, USA	Data synthesis	Birds, bats	Turbines: hub height, rotor diameter. Species: mass, surface area.	Search area required increases with hub height, rotor diameter, and species size. Recommended search distances from base of turbine range from 55 m for bats where turbines are small and 74 m where turbines are large, to 103 m and 122 m for large birds where turbines are small and large, respectively.
Huso and Dalthorp (2014)	USA	Field surveys	Birds, bats	Turbines: hub height, rotor diameter	Tested five estimators of fall distances against field data. Visibility decreased with distance from turbine. Estimated median distance from turbine to carcass ~30 m. Recommend polynomial logistic regression models of relative carcass density as a function of distance to the turbine.
Prakash and Markfort (2020, 2021, 2022)	USA	Simulation and field experiment	Hoary, Eastern Red, Evening, Silver-haired, and Big Brown Bat	Species: mass, surface area, drag coefficient. Turbines: yaw (compass direction turbine is pointing), rotation rate, height. Environment: wind speed, air density.	Present alternative ballistics models to predict the fall distributions of bat species, focus on quantifying drag coefficients. Validate with field data and fall experiments. Find that wind drift and radial strike location significantly influences carcass fall trajectories (and in turn, fall distributions), especially for smaller carcasses. Predicted fall distances range from 0–185 m, though this upper limit may be smaller if radial strike location is skewed towards the nacelle.

7.1.2 Searcher efficiency

Several factors will influence whether or not a carcass that lies within the survey area at the time of a search is detected, including who (or what) is leading the search, how old and decomposed the carcass is, its size, the nature of the vegetation and substrate surrounding the turbines, and environmental conditions.

Detection dogs have consistently been found to achieve higher detection rates than humans (Table 17), which suggests that employing human-only searches will lead to an underestimation of fatalities caused by both individual turbines, and entire facilities. In studies where human and dog searchers were compared side-by-side in field trials, dogs detected on average 69–96% of carcasses, while humans detected only 9–65% (Table 17). Both humans and dogs tend to achieve higher detection rates for larger-bodied compared to smaller-bodied carcasses in general, but the relative difference between dogs and humans is most pronounced for smaller-bodied carcasses. Therefore, it is particularly important to use detection dogs in searches where it is suspected that microbats or small birds are being impacted; Smallwood et al. (2020) demonstrated that fatality estimates based on detection dog searches were 6.4 times higher for bats and 2.7 times higher for small birds compared to searches conducted by humans. To put this into a local context, forest bats (*Vespadelus* sp.) that collide with turbines in Victoria typically weigh 3–8 g, so are unlikely to be detected by humans alone.

Besides carcass size, detection rates achieved by dogs are in general more robust to other environmental factors. Dogs tended to be less affected by reduced visibility associated with more complex vegetation structure, likely because they are more reliant on smell then sight, though this may depend on whether or not they are leashed (del Valle et al. 2020). In their human-led searches, Barros et al. (2022) also found that higher detection rates were achieved in the dry compared to the rainy season. For these reasons, it is important to conduct searcher efficiency trials in the same locations and conditions as the corresponding carcass searches, rather than borrowing estimates from elsewhere.

Key references

Barrientos, R, Martins, RC, Ascensão, F, D'Amico, M, Moreira, F, Borda-de-Água, L (2018) A review of searcher efficiency and carcass persistence in infrastructure-driven mortality assessment studies. *Biological Conservation* **222**, 146-153.

del Valle, JD, Peralta, FC, Arjona, MIJ (2020) Factors affecting carcass detection at wind farms using dogs and human searchers. *Journal of Applied Ecology* **57**, 1926-1935.

Smallwood, KS, Bell, DA, Standish, S (2020) Dogs detect larger wind energy effects on bats and birds. *Journal of Wildlife Management* **84**, 852-864.

Table 17. Summary of studies assessing searcher efficiency.

'Local' species are those that could reasonably be expected to be impacted by collisions at the facility/facilities, whereas 'domestic' species are surrogate carcasses such as chickens, mice and turkeys brought in from elsewhere.

Study	Country	Study type	Targets	Factors modelled	Placed carcasses	Key findings
Barrientos et al. (2018)	Global	Meta-analysis	Birds, bats	Searcher: dogs, humans, level of experience. Species: size (mass). Environment: vegetation type, season.	Mix of locally-sourced and domestic species	Searcher efficiency higher for larger carcasses: mean detection rate for carcasses <100 g was 67%, for carcasses >1 kg it was 77%. Searcher efficiency higher for dogs (mean detection rate 87%) compared to humans (mean rate 65%), particularly for smaller carcass sizes.
Barros et al. (2022)	Brazil	Field experiment	Bats	Species: size (mass). Environment: vegetation type, season. [searches led by humans]	Local: six bat species	On average 58% of bat carcasses were detected. Detection probability affected by vegetation type, season, and carcass size. The median probability of finding small bats in shrub vegetation during the rainy season was eight times lower than for large bats in no or sparse vegetation during the dry season.
del Valle et al. (2020)	Spain	Field experiment	Birds, bats	Searcher: dogs, humans. Species: size (wingspan) Environment: vegetation complexity, wind speed, temperature.	Local: two bat species, 48 bird species	Dogs detected 77.3% of carcasses, humans only detected 21.5%. Human detection rates were strongly affected by carcass size (higher rates for larger carcasses) and vegetation structure (lower rates in more complex vegetation) while dog detection rates were not.
Mathews et al. (2013)	UK	Field experiment	Bats	Searcher: dogs, humans. Species: bat species. Environment: vegetation height, visibility.	Local: ten bat species	Dogs detected 73% of carcasses, humans only detected 20%. Dogs averaged 40 min to complete a survey, humans took 2 hours. Visibility strongly influenced carcass detection rate, minimum vegetation height an adequate surrogate for this.
Nilsson et al. (2023)	Norway	Field experiment	Birds	Species: size (mass). [searches led by dogs]	Local: twelve bird species	Searcher efficiency higher for larger carcasses: mean detection rate for carcasses <24 g was 17%, for carcasses >60 g it was 74%.
Paula et al. (2011)	Portugal	Field experiment	Birds	Searcher: dogs, humans. Environment: visibility.	Local: one bird species	Dogs detected 96% of carcasses, humans only detected 9%. Dogs unaffected by visibility.
Peters et al. (2014)	USA	Field experiment	Birds, bats	Searcher: humans. Species: size (mass). Environment: substrate, temperature.	Local: two bat species, 32 bird species	Detection rates differed between individual human observers. Searcher efficiency higher for larger carcasses, and higher on bare ground.

Study	Country	Study type	Targets	Factors modelled	Placed carcasses	Key findings
Reyes et al. (2016)	USA	Field experiment and simulation	Birds	Searcher: dogs, humans. Species: size.	External: four bird species, and feather spots	Dogs detected 69–86% of carcasses, and 61% of feather spots. Humans detected 40–54% of carcasses, and 23% of feather spots. Searcher efficiency higher for larger carcasses.
Smallwood and Bell (2020b)	USA	Field experiment	Bats	Searcher: dogs, humans.	Local bat species	Dogs detected 95% of bat carcasses, and dog searches resulted in fatality estimates almost 11 times higher than humans alone. Dogs found only one of the four bats directly observed colliding with turbine blades the night prior using thermal camera. Authors suggest this may be because bats were quickly scavenged, or injured bats crawled into fossorial mammal burrows.
Smallwood et al. (2020)	USA	Field experiment	Birds, bats	Searcher: dogs, humans. Species: size (mass).	Local: five bat species, 39 bird species	Dogs detected 96% of bat carcasses, humans only detected 6%. Dogs detected 90% of small bird carcasses, humans only detected 30%. Dog searches resulted in fatality estimates up to 6.4 and 2.7 times higher for bats and small birds, respectively.

7.1.3 Carcass persistence trials

A carcass may be removed from the area beneath a turbine by a scavenger prior to a carcass search taking place, or degrade in the environment, meaning it will not be detected. Camera traps (i.e. heat and motion activated cameras) have been used to record a great diversity of scavengers removing all or parts of carcasses during trials, including various species of native and feral dogs, foxes, coyotes, cats, rodents, mustelids (ferrets, stoats, weasels etc.), birds including raptors and corvids (ravens and crows), reptiles, and arthropods (such as ants and dung beetles). Scavengers will not only consume carcasses, but also move them within the search area; Farfan et al. (2017) found that many of the carcasses of quails and pigeons that they radio-tagged in their experiment were moved >100 m.

How long a carcass remains in place once it has landed is referred to as the persistence time, and assumed or estimated persistence times influence how much one corrects for imperfect detection. In other words, if a particular type of carcass gets scavenged (or decomposes) quickly, then it is more likely to be missed by searchers, so a greater adjustment needs to be applied to fatality estimates (Hallingstad et al. 2023).

An important factor to consider when conducting carcass persistence trials is how many carcasses to use, and how frequently they are placed (Wilson et al. 2022). If too many carcasses are placed at once then the local scavenger community may be satiated, so carcass persistence times will increase relative to normal conditions – a phenomenon known as ‘scavenger swamping’. This could have a substantial impact on subsequent fatality estimates; for example, Smallwood et al. (2010) found that the placement of only 1–5 carcasses at a time, instead of the standard 10 or more, led to estimations of fatalities that were nearly three times higher for Red-tailed Hawks and Barn Owls. While Huso and Erickson (2013) question the validity of the statistical approach that yielded this finding, they nonetheless acknowledge that scavenger swamping may be a phenomenon worth assessing. In general, if carcasses are placed more frequently, then a higher level of precision can be achieved with fewer carcasses (Wilson et al. 2022), but 15–25 carcasses should be placed in total (over the survey period) for each combination of factors that may influence persistence.

Comparative carcass persistence experiments have consistently found that larger carcasses remain on-site for longer (Table 18). Large raptors can persist for a very long time; for example, in their global meta-analysis Wilson et al. (2022) found a median persistence time of 420 days, with trial-specific median estimates ranging from 14–1586 days. In comparison, trials focussed on microbats, and/or House Mice that are commonly used as proxies, typically find mean persistence times to be around 2 days (e.g. Villegas-Patraca et al. 2012; Paula et al. 2014; Barros et al. 2022). Larger carcasses persist for longer because they are only able to be removed by larger scavengers, while smaller scavengers may only eat part of the carcass so it remains detectable to a searcher (Barrientos et al. 2018). After approximately one to two weeks, carcasses become significantly decayed, so are no longer as attractive to vertebrate scavengers; removals then become driven more by invertebrates and abiotic processes, which are typically slower (Kitano et al. 2020). For birds, sometimes all that remains are feathers, termed ‘feather spots’, which can last a long time in the landscape.

While some researchers have stated that mammals are scavenged faster than birds, Barrientos et al. (2018) concluded from their global meta-analysis that this was an artefact of the comparison between microbats and larger birds. Once differences in body size had been accounted for, they found that mammals actually persisted for longer, which may have implications for large volant mammals such as flying-foxes. They also could not find any evidence that there were differences in persistence times between equally-sized wild versus domestic species, or fresh versus thawed carcasses of the same taxon.

Another consistent finding across studies was that carcasses persist for longer in rainy conditions, so the timing of trials (and the associated weather patterns) needs to match the full suite of conditions across which fatality estimates are being made. There may be certain periods of the year when fewer alternative prey are available for scavengers, and during which removal times will be faster (e.g. Kitano et al. 2020).

While carcass searches can be performed by human observers, many now use camera traps (or trail cameras) for monitoring, which can allow one to pinpoint the exact time a carcass is removed. This improves precision, and also often means that the scavenging species can be identified. However, there are trade-offs associated with costs and other practicalities, as discussed by Paula et al. (2014). Use of cameras, and in turn the reduced presence of humans on a site, may also partly help reduce biases associated with scavengers being attracted or deterred by the human scent left at the moment of carcass placement.

Key references

Barrientos, R, Martins, RC, Ascensão, F, D'Amico, M, Moreira, F, Borda-de-Água, L (2018) A review of searcher efficiency and carcass persistence in infrastructure-driven mortality assessment studies. *Biological Conservation* **222**, 146-153.

Wilson, D, Hulka, S, Bennun, L (2022) A review of raptor carcass persistence trials and the practical implications for fatality estimation at wind farms. *PeerJ* **10**, e14163.

Table 18. Summary of studies assessing carcass persistence.

Rows are arranged according to targets (bats, bats and birds, then birds), then study types, and then author.

Study	Country	Study type	Targets	Factors modelled	Placed carcasses	Key findings
Barros et al. (2022)	Brazil	Field experiment	Bats	Carcass: species. Environment: vegetation type, season.	Domestic chicken chick, Flat-faced Fruit-eating Bat, House Mouse	Longer persistence times in rainy season. Persistence is lower for chicks (1.2 days) compared to bats and mice (2.1 days). Scavengers include canids (especially crab-eating fox), birds of prey, and insects (especially dung beetles). 86% of carcasses removed within the first week.
Grodsky et al. (2012)	USA	Field experiment	Bats	Carcass: species	House Mouse, bats	The persistence (survival) distribution of mice was no different from bats in scavenger removal trials
Paula et al. (2014)	Portugal	Field experiment	Bats, birds	Carcass: species. Scavengers: guild. Environment: season, rain exposure. Survey method: camera trap versus human checks.	Red-legged Partridge, Common Quail, House Mouse	Rodents removed carcasses three times faster than carnivores. Mean persistence time in dry conditions was 2.3 days, in rainy conditions it was 8.4 days.
Peters et al. (2014)	USA	Field experiment	Bats, birds	Carcass: size (mass), taxa. Environment: season, temperature, distance to marshland.	Hoary Bat, Eastern Red Bat, 32 bird species	31.5% of carcasses are predicted to persist after 14 days. The longer a carcass has been placed for, the higher its daily persistence probability (i.e. older carcasses are less likely to be scavenged).
Villegas-Patraca et al. (2012b)	Mexico	Field experiment	Bats, birds	Carcass: type (small bird, large bird, bat). Environment: season, vegetation type.	Six bat species, small and large chickens	Estimated seasonal survival rates, defined as a 20-day period, were 0.0% during the dry season (i.e. none ‘survived’, all were removed) and 21% during the rainy season. Average removal time was 2.0 days for bat carcasses. Average removal time ranged from 2.1–4.4 days for birds.
Barrientos et al. (2018)	Global	Review	Bats, birds	Carcass: size (mass), fresh versus frozen, wild versus domestic, taxon	Both wild and domestic	Larger carcasses persist for longer. Mammals persist longer than birds (once size is controlled for). No difference in fresh versus frozen carcass. No difference in wild versus domestic.

Study	Country	Study type	Targets	Factors modelled	Placed carcasses	Key findings
Bernardino et al. (2011)	Portugal	Field experiment	Birds	Carcass: size (mass). Environment: season.	Parakeets, quails, partridges	On average carcasses were removed faster in spring (3.9 days) versus autumn (4.6 days). No difference in removal rates between the three size classes/species used. Recommended daily checks for carcasses for at least 15 days. >80% of carcasses removed within the first week.
DeVault et al. (2017)	USA	Field experiment	Birds	Carcass: species, wild versus domestic. Environment: vegetation type, temperature.	American Kestrels, Red-tailed Hawk, Northern Bobwhite, domestic chicken, Rock Pigeon	Average persistence time across all carcass types was 8.7 days. Red-tailed Hawk carcasses persisted for significantly longer than chickens (average removal time 10.3 compared to 6.9 days). Northern Bobwhite persisted for significantly shorter periods than chickens (average removal time 10.4 compared to 6.9 days). No significant effect of temperature or habitat type on removal times.
Farfán et al. (2017a)	Spain	Field experiment	Birds	Carcass: species. Environment: vegetation type. Location: turbine versus power line.	Rock Pigeon, Common Quail	Persistence lower for quails (1.5 days) than pigeons (4.6 days). All quails scavenged by the third day, 45% of pigeon carcasses scavenged by the 14th day. 6/10 of the pigeon carcasses that weren't fully removed by scavengers were displaced >100 m from where they were deposited.
Henrich et al. (2017)	Germany	Field experiment	Birds	Environment: vegetation type, season	Domestic chicken chick	Two-thirds of chicks removed within 5 days, 20% were buried by beetles in summer, 40% removed by mammals or birds. Median persistence time was 2.79 days. No significant effect of habitat type, persistence on average two days longer in autumn than in summer.
Kitano et al. (2020)	Japan	Field experiment	Birds	Carcass: size (length). Environment: season.	36 wild bird species	In winter 64% of all carcasses scavenged within one day. In comparison, in autumn and summer no carcasses scavenged within one day. Carcass persistence time much shorter in snow possibly because scavengers are hungrier, and carcasses are more visible.

Study	Country	Study type	Targets	Factors modelled	Placed carcasses	Key findings
Smallwood et al. (2010) [see Huso and Erickson 2014 reply]	USA	Field experiment	Birds	Carcass: size (mass). Environment: season.	Salvaged wild bird species	First scavenging event for all carcasses averaged 4.45 days. Removal rates were slower for large versus small carcasses, and in winter versus summer. When fewer caresses were placed to avoid scavenger swamping, substantially more were removed by the 15-day mark.
Urquhart et al. (2015)	UK	Field experiment	Birds	Carcass: species. Environment: season.	Buzzard, Ring-necked Pheasant	On average, Buzzard carcasses persisted for 63.5 days and pheasant carcasses for 9.2 days. Strongest model included an interaction between season and species.
Hallingstad et al. (2023b)	USA	Field experiment and meta-analysis	Birds (raptors)	Carcass: species (game birds versus wild raptors). Environment: vegetation type.	Ring-necked Pheasant, Mallard, eleven raptor species (hawks, owls, falcons, osprey, vultures)	Raptor persistence significantly higher than game birds for 95% of the sampled strata. 30-day probability of persistence ranged from 0.44–0.99 for raptors and from 0.16–0.79 for game birds. Raptor carcass persistence varied by season, habitat, and region. There was a strong positive relationship between raptor and game bird average probability of persistence.
Wilson et al. (2022)	Global	Meta-analysis	Birds (raptors)	Environment: biome. Study design: number of placed carcasses, trial length.	Multiple	No significant relationship between either the number of carcasses in the trial or trial duration and estimated carcass persistence. Because raptor carcass persistence rates are generally much longer than recommended search intervals, variation in persistence rates between trials doesn’t have a large effect on fatality estimates.

7.1.4 Analysis of field monitoring data to derive fatality estimates

The goal of post-construction mortality monitoring is ultimately to estimate the size of the 'super-population', that is, the number of individuals of a given species that have collided with turbines over a specific period of time. A range of approaches to estimate the super-population have been developed over several decades (Huso et al. 2016). These estimators are typically a modification of a Lincoln-Petersen equation, where estimated total fatalities at a facility are a function of the number of observed carcasses, the number and frequency of searches, the area searched (both around the individual turbines, and the proportion of turbines searched), and the probability of detection (based on the searcher efficiency and carcass persistence trials).

This literature is quite technical, and is not necessarily readily accessible to policy makers and regulators responsible for decisions related to wind energy developments. Nonetheless, the ramifications are important to understand because globally, and through time, different facilities have used different approaches to estimation, which in turn affects perceptions of just how big an impact wind energy developments have had on bird and bat populations.

Bernadino et al. (2013) conducted a comparative analysis of seven of the most commonly-used approaches, including the Erickson et al. (2000), Shoenfeld (2004), Kerns et al. (2005), Jain et al. (2007), Pollock (2007), Huso (2011), and Korner-Nievergelt et al. (2011) estimators, and provided a helpful overview of their associated assumptions and limitations. Since that time, fatality estimators have progressively been adapted to address some of the less realistic assumptions of the earlier models, and in turn, improve the accuracy of fatality estimates. Huso et al. (2016) provide a concise overview of these improvements, which we present below (Table 19). As is often the case in addressing some of the underlying assumptions, model complexity has increased, which in turn can make the models less accessible to everyday practitioners. However, in 2018, the GenEst ('Generalised Estimator') platform was released, which can be implemented through either an R package (also called GenEst) or an online graphic user interface. In our review, this platform was often cited as the means by which authors analysed their survey data (e.g. Weaver et al. 2020; Wilson et al. 2022; Rnjak et al. 2023).

The statistical models underlying the GenEst platform are also extensions of the Huso et al. (2011) estimator. A Canadian study by Thurber et al. (2022) compared the estimates yielded by GenEst with those of Huso et al. (2011), the Shoenfeld-Erickson (2004) estimator, and an approach developed by the Ontario Ministry of Natural Resources and Forestry (OMNRF). They found that while the results of the first three approaches were broadly comparable, the OMNRF estimates tended to be higher, and assumptions related to carcass persistence were regularly violated. Consequently, and because of the flexibility offered by GenEst, they recommended that it be used in future studies to ensure consistency. However, the GenEst platform can only accommodate regular, fixed search intervals (e.g. every 7 days), and not the 'pulse' type searches (where the fixed intervals are punctuated by searches 2–3 days later, typically used for microbats) that are regularly conducted in Victoria.

An alternative set of estimators have also been developed that instead adopt open population capture-recapture approaches (Cormack-Jolly-Seber models, e.g. Etterson et al. 2013; Péron et al. 2013). The advantage of these is that: (1) surveys are conducted using double observer sampling; (2) carcasses can be left on the ground; and (3) the requirement for separate searcher efficiency and carcass persistence trials can be removed (Péron 2018). However, this would mean that field data also have to be collected in a different way that does not necessarily conform to existing survey protocols, and may not be comparable to past studies. There are also concerns that these models may only be suited to situations where large numbers of individuals are killed, and less-so to those where carcass data are sparse (Huso et al. 2016).

Table 19. Comparison of Lincoln-Petersen fatality estimators – adapted from Huso et al. (2016). Note that some of these papers were not included in our review because our results were limited to peer-reviewed papers dating from 2009 onwards, and some of these estimators pre-date this time.

Estimator	Advancements	Assumptions and comments
Erickson et al. (2000)		The carcass population is stable, with 'arrivals' (from collisions) and 'removals' (from scavengers) being in balance. Tends to underestimate total fatalities.
Schoenfeld (2004)	Introduced an adjustment to account for removal of carcasses by teams after each search	Scavenging rate is constant. Searcher efficiency is the same for all searches. Carcasses that have been missed in one or more searches are no more difficult to find than carcasses that are found on the first search after arrival. Tends to underestimate total fatalities.
Huso (2011)	Relaxed the assumption that carcass persistence follows an exponential distribution. Accounts for changes in searcher efficiency.	Carcasses that are missed in one search are not discoverable in later searches
Korner-Nievergelt et al. (2011)	Accounts for changes in searcher efficiency	Carcasses arrive only at the beginning of search intervals. Persistence distribution is exponential (Korner-Nievergelt et al. 2015 allows for other distributions).
Wolpert (2013)	Can accommodate non-constant scavenging rates and searcher efficiency that varies continuously with season and/or carcass age	Incorporates elements of the estimators listed above as special cases
Dalthorp et al. (2014)	Allows for non-constant carcass arrival functions	

In Victoria, two approaches have been used to analyse and synthesise post-construction fatality monitoring data from across facilities. Symbolix Pty Ltd. (2020) applied the Huso (2011) approach to observed survey data to estimate the proportion of carcasses detected. However, they then also use a Monte Carlo approach to simulate survey data and associated searcher efficiency and scavenger rates that could have yielded the observed data, and from this simulated dataset estimate underlying mortality rates of both detected and undetected species. Moloney et al. (2019) also use an adapted version of the Huso (2011) estimator, but implemented within a Bayesian framework, so again could model both detected and undetected species.

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7.1.5 Evidence of absence

One of the issues that arises during post-construction mortality monitoring is how to interpret the finding that a species that could be at risk of collisions hasn't been detected during carcass searches (i.e. the count for that species = 0). This frequently occurs in the case of rare and threatened species and could be the result of no collisions having actually occurred, or searches failing to detect carcasses. Either way, the '0' presents a unique problem for data analyses that cannot be addressed though the estimation approaches detailed above.

This problem is especially important in the context of the USA's incidental take permits, where regulators must have confidence that operators are not exceeding their permitted take, even if the species are not being detected during post-construction monitoring. To address this, Huso et al. (2015) developed a Bayesian analytical approach to assess 'Evidence of Absence' (EoA). That is – how confident can we be that a 0 represents a true 0, given the probability of detection that will have been achieved by the survey effort? They demonstrate that a probability of detection of approximately 0.45 or higher needs to be achieved in order to be confident that species of interest have not been missed. To quote Huso et al. (2015):

"*When probability of detection is high, observing no carcasses can be construed as evidence that no or few animals have been killed, i.e., evidence of absence. When probability of detection is low, however, finding zero carcasses does not credibly rule out the possibility of a large take. We are left with only absence of evidence*."

The EoA approach is now commonly adopted as part of standardised fatality monitoring in the USA, in cases where take permits apply (Hallingstad et al. 2023). McDonald et al. (2021) have subsequently extended these methods with their 'Evidence of Absence Regression' (EoAR) approach, which also allows one to fit covariate relationships (e.g. differences in vegetation between sites) and adds an extra model term to account for facilities of varying sizes.

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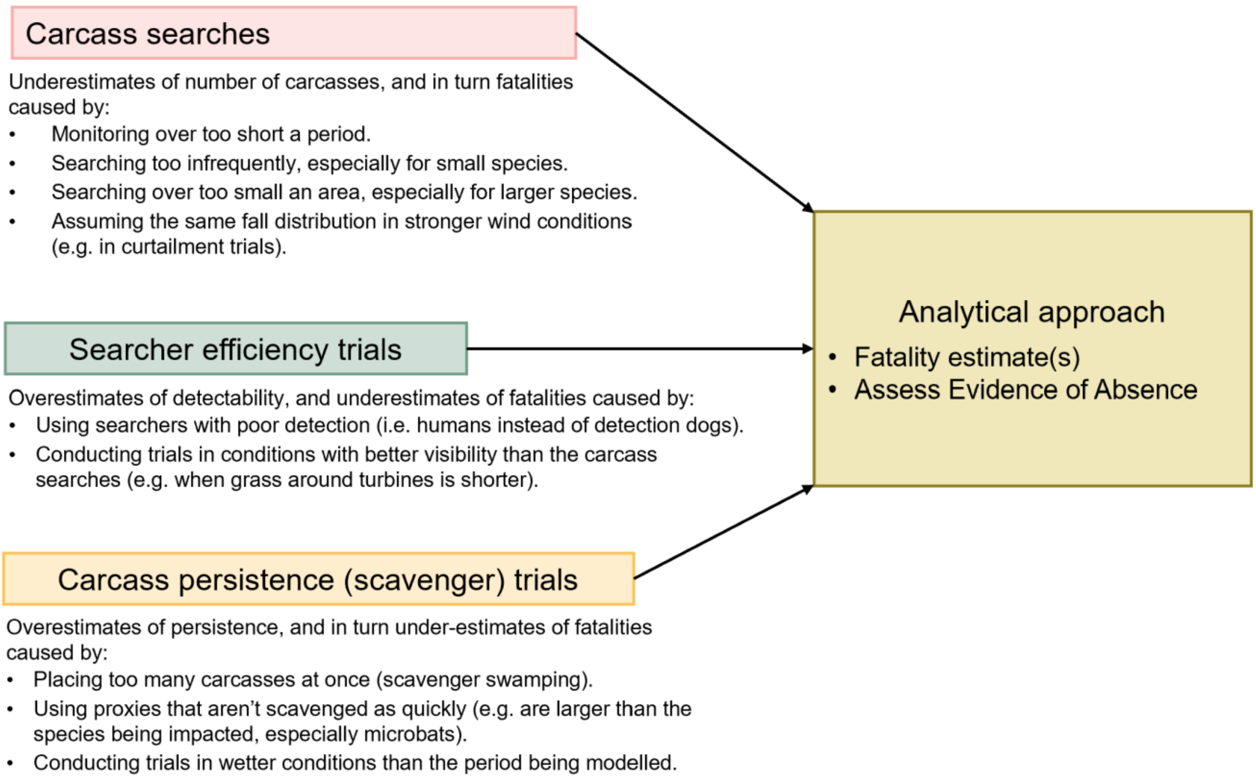

Figure 8. Sources of bias in wind energy fatality estimates that should be addressed in a robust monitoring program.

To have confidence that fatality estimates are accurate, and that threatened or rare species are not being missed in carcass searches, probabilities of detection have to be as high as possible.

7.2 Remote monitoring of mortalities

Very small number of papers (4 papers)	High bird bias (100%)	High geographic bias (Europe 75%)

As we have described above, field surveys consisting of carcass searches, searcher efficiency and carcass persistence trials are the standard approach used for monitoring collisions and associated fatalities at wind energy facilities. There are recognised limitations associated with these approaches, but they are now fairly well-studied and understood.

In contrast, several authors have suggested that remote monitoring methods could be used to automatically detect collisions and fatalities, as a means of either complementing or replacing standard field surveys. These approaches have been tested via simulation, and in lab and field experimental settings using various proxies for carcasses (e.g. balls and bundles of feathers), though none have been trialled *in-situ* with turbines and actual carcasses. Therefore, this field of research is still very much developing.

Mälzer et al. (2020) proposed the use of a radar system, mounted to the turbine tower at various heights such that carcasses were automatically detected as they fell and interrupted the radar barrier. Happ et al. (2021) present an approach that instead involved analysis of imagery of the ground where carcasses would fall, captured via visual, near infra-red and thermal cameras. Their field drop trials (from 75 m) yielded sensitivities (the true positive rate) of 76% for night-time detections and 84% for daytime detections. Two further studies attempted to capture the moment of impact of individuals with turbine blades – Kang et al. (2018) based their system on piezoelectric paint, which detected 100% of collisions in lab tests with small models of turbines, while a system proposed by Hu et al. (2018) used vibration sensors, cameras, and acoustic and contact microphones. This latter study conducted field tests with operational turbines, and simulated collisions with tennis balls, but found that the system's detection rate was only 50% due to signal masking from background noise.

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8 Conclusions

This systematic review of the scientific literature has indicated that turbine collisions have the potential to cause substantial direct (Section 4.1), ongoing, and far-reaching impacts on bird and bat populations (Section 4.2). Much of what is known about these impacts, as well as associated assessments, mitigations and monitoring, is based on a subset of very well-studied species in temperate systems in the northern hemisphere (Section 3.1). There are parallels and similarities between these areas and Victoria, giving some confidence that strong and consistent results found elsewhere will hold here (Section 3.2). Nonetheless, there is still a great need for more species- and context-specific knowledge for Victoria. In particular, some unique Victorian Species of Concern have no comparable well-studied 'surrogates' in the available wind energy literature, such as flying-foxes and small migratory parrots.

It must also be noted that the trends and results identified here from past international research are generally based on shorter and lower-capacity turbines (Section 5.4.1), many of which are significantly smaller than the newer-generation turbines currently in use or under construction in Victoria (typically 6 MW or greater). It is difficult to know whether trends in fatality rates, fall distances, mitigation effectiveness etc. will scale linearly with turbine size, or if these relationships will change. If they do not, this means that relationships based on associated literature cannot be reliably extrapolated to larger turbines established in Victoria.

Bats are greatly impacted by wind energy facilities, often experiencing higher fatality rates than birds (Section 4.1). It is likely that small-bodied microbat mortality rates are underestimated, because they are readily missed in carcass searches and are quickly scavenged (Sections 7.1.2 and 7.1.3). There is also some evidence that some microbat species can be attracted to turbines for reasons that remain unclear (Section 4.3.1). It appears that pre-construction acoustic surveys poorly predict post-construction fatality risks (Section 5.5.4), underlining the importance of rigorous mortality monitoring (Section 7.1) to determine impacts and inform mitigations. Low windspeed night-time curtailment is a highly effective and well-studied mitigation option for microbats that significantly reduces fatality rates (Section 6.1.1); however, little is known about how to assess and mitigate risks to flying-foxes.

Some bird groups of interest in Victoria, such as passerines and parrots, are poorly represented in the wind energy literature (Section 3.2). Even so, it appears that raptors are the bird group most impacted by collisions (Section 4.1.1). Raptors seem to have the potential to learn to avoid turbines (Section 4.3.2), and are less likely to be missed in mortality searches (Sections 7.1.2 and 7.1.3). Some promising mitigations have been trialled elsewhere for this group (e.g. automated detection systems discussed in Section 6.3, and increased blade visibility discussed in Section 6.4), but these mitigations require experimental testing in Victoria. For both birds and bats, an issue for rare and threatened species is that 'absence of evidence' (i.e. failing to detect a species) in mortality searches should not be interpreted as 'evidence of absence' (i.e. assuming no collisions have occurred) unless high detection probabilities have been achieved (Section 7.1.5).

It is unclear how to quantify and account for cumulative impacts on populations (though see Bastos et al. 2016). As with other disturbances such as mining, forestry, and urban development, impacts (in this case, fatalities) will accrue across an expanding spatial footprint as new wind energy facilities are added to a landscape. However, wind turbines also have a long temporal footprint, and can continue to remove individuals from a population as long as they are operational (i.e. potentially several decades). Therefore, the cumulative impacts of wind energy are of particular concern in both space and time (Katzner et al. 2016, Masden et al. 2010) and should be a focus of future research.

Given that we are currently unable to confidently predict the number of fatalities that will be associated with a facility based on typical pre-constructions assessments (Section 5.5.4), there is strong support for the value in adopting an adaptive approach to monitoring and mitigation. This could accommodate and account for improved information as data continue to be collected. Such an approach has been adopted in the USA, specifically for Golden and Bald Eagles, and the underlying models that inform 'take' limits and mitigations continue to be reviewed and updated (New et al. 2015). Finally, as noted by other authors (e.g. Hutchins et al. 2016), the identification of trends in data collected across regions, and indeed across the scientific literature in general, would be greatly helped by increased transparency (e.g. improved information about turbine operation periods during mitigation trials), and use of consistent post-construction survey techniques and reporting. This can be challenging and is dependent on the cooperation of, and collaboration with, private operators who have to balance multiple interests. Nonetheless, this will be critical in balancing the overall environmental benefits of wind energy in reducing greenhouse gas emissions, with the costs to biodiversity and threatened species.

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Supplementary tables and figures

Table S1. Search terms used to identify the potentially relevant pieces of literature.

Theme	Keywords	Boolean
Wind energy facilities	“wind energy” OR “wind farm” OR “windfarm” OR “wind power” OR “turbine”	AND
Volant vertebrates	“avian” OR “bird” OR “ground-nesting” OR “passerine” OR “raptor” OR “seabirds” OR “shorebirds” OR “waterbird” OR “waterfowl” OR “bat” OR “bats” OR “Chiroptera” OR “flying fox” OR “flying-fox” OR “microbat”	AND
Methods and assessments	“CRM” OR “population model” OR “population viability analysis” OR “PVA” OR “cumulative” OR “desktop” OR “mapping” OR “spatial” OR “detect*” OR “dog” OR “evaluation” OR “monitor” OR “search” OR “survey” OR “sweep” OR “pre-construction” OR “post-construction” OR “risk” OR “scavenger”	OR
Determinants of risk	“habitat” OR “migratory” OR “threatened” OR “trait”	OR
Impacts	“fatality” OR “mortality” OR “death” OR “carcass” OR “collision” OR “impact” OR “injury” OR “interaction” OR “attraction” OR “avoidance” OR “displacement”	OR
Management	“curtailment” OR “deterrent” OR “mitigation” OR “monitoring” OR “siting”	

Table S2. Individual species and taxa that formed the focus of studies in the review, and the number of research items that they appeared in.

Table is sorted by taxon (bats, then birds) and then count (number of studies that focussed on that species).

Common name	Taxon	Scientific name	Count
'Multiple species'			251
Hoary Bat	Bat	*Lasiurus cinereus*	40
Silver-Haired Bat	Bat	*Lasionycteris noctivagans*	28
Eastern Red Bat	Bat	*Lasiurus borealis*	26
Common Pipistrelle	Bat	*Pipistrellus pipistrellus*	12
Little Brown Bat	Bat	*Myotis lucifugus*	12
Big Brown Bat	Bat	*Eptesicus fuscus*	11
Common Noctule	Bat	*Nyctalus noctula*	10
Leisler's Bat	Bat	*Nyctalus leisleri*	6
Nathusius' Pipistrelle	Bat	*Pipistrellus nathusii*	5
Indiana Bat	Bat	*Myotis sodalis*	4
Mexican Free-tailed Bat	Bat	*Tadarida brasiliensis*	4
Northern Myotis	Bat	*Myotis septentrionalis*	4
Soprano Pipistrelle	Bat	*Pipistrellus pygmaeus*	4
Tri-colored Bat	Bat	*Perimyotis subflavus*	4
Western Red Bat	Bat	*Lasiurus blossevillii*	3
Cape Serotine	Bat	*Neoromicia capensis*	2
Evening Bat	Bat	*Nycticeius humeralis*	2
Kuhl's Pipistrelle	Bat	*Pipistrellus kuhlii*	2
Savi's Pipistrelle	Bat	*Hypsugo savii*	2
Southern Bent-winged Bat	Bat	*Miniopterus orianae bassanii*	2
Southern Yellow Bat	Bat	*Lasiurus ega*	2
Western Barbastelle	Bat	*Barbastella barbastellus*	2
Argentine Brown Bat	Bat	*Eptesicus furinalis*	1
Brazilian Brown Bat	Bat	*Eptesicus brasiliensis*	1
Daubenton's Bat	Bat	*Myotis daubentonii*	1
Eastern Bent-winged Bat	Bat	*Miniopterus orianae oceanensis*	1
Eastern Small-footed Myotis	Bat	*Myotis leibii*	1
Egyptian Free-tailed Bat	Bat	*Tadarida aegyptiaca*	1
European Free-tailed Bat	Bat	*Tadarida teniotis*	1
Hawaiian Hoary Bat	Bat	*Lasiurus cinereus semotus*	1
Hodgson's Bat	Bat	*Myotis formosus flavus*	1
Japanese House Bat	Bat	*Pipistrellus abramus*	1
Lesser Asiatic Yellow Bat	Bat	*Scotophilus kuhlii*	1
Little Bent-winged Bat	Bat	*Miniopterus australis*	1
New-Caledonia Wattled Bat	Bat	*Chalinolobus neocaledonicus*	1
Northern Bat	Bat	*Eptesicus nilssonii*	1
Northern Yellow Bat	Bat	*Dasypterus intermedius*	1
Parti-coloured Bat	Bat	*Vespertilio murinus*	1
Sebas Short-Tailed Fruit Bat	Bat	*Carollia perspicillata*	1
Serotine Bat	Bat	*Eptesicus serotinus*	1
Small Melanesian Bent-winged Bat	Bat	*Miniopterus macrocneme*	1
Taiwan Serotine Bat	Bat	*Eptesicus serotinus horikawai*	1

Common name	Taxon	Scientific name	Count
Theobald's Tomb Bat	Bat	*Taphozous theobaldi*	1
Western Yellow Bat	Bat	*Dasypterus xanthinus*	1
Golden Eagle	Bird	*Aquila chrysaetos*	40
White-Tailed Eagle	Bird	*Haliaeetus albicilla*	19
Griffon Vulture	Bird	*Gyps fulvus*	16
Bald Eagle	Bird	*Haliaeetus leucocephalus*	14
‘Raptors’	Bird		9
Red Kite	Bird	*Milvus milvus*	9
Red-Tailed Hawk	Bird	*Buteo jamaicensis*	5
Skylark	Bird	*Alauda arvensis*	5
Cape Vulture	Bird	*Gyps coprotheres*	4
Capercaillie	Bird	*Tetrao urogallus*	4
Ferruginous Hawk	Bird	*Buteo regalis*	4
Greater Prairie-chicken	Bird	*Tympanuchus cupido*	4
Mallard	Bird	*Anas platyrhynchos*	4
Whooping Crane	Bird	*Grus americana*	4
Bearded Vulture	Bird	*Gypaetus barbatus*	3
Black Kite	Bird	*Milvus migrans*	3
Cinereous Vulture	Bird	*Aegypius monachus*	3
Common Buzzard	Bird	*Buteo buteo*	3
Egyptian Vulture	Bird	*Neophron percnopterus*	3
Hen Harrier	Bird	*Circus cyaneus*	3
Horned Lark	Bird	*Eremophila alpestris*	3
Lesser Black-backed Gull	Bird	*Larus fuscus*	3
Lesser Kestrel	Bird	*Falco naumanni*	3
Montagu's Harrier	Bird	*Circus pygargus*	3
Northern Pintail	Bird	*Anas acuta*	3
Swainson's Hawk	Bird	*Buteo swainsoni*	3
Turkey Vulture	Bird	*Cathartes aura*	3
White Stork	Bird	*Ciconia ciconia*	3
American Kestrel	Bird	*Falco sparverius*	2
Bewick's Swan	Bird	*Cygnus columbianus bewickii*	2
Black Grouse	Bird	*Lyrurus tetrix*	2
Blue-winged Teal	Bird	*Anas discors*	2
California Condor	Bird	*Gymnogyps californianus*	2
Dickcissel	Bird	*Spiza americana*	2
Eastern Spot-billed Duck	Bird	*Anas zonorhyncha*	2
European Golden Plover	Bird	*Pluvialis apricaria*	2
Greater Sage-grouse	Bird	*Centrocercus urophasianus*	2
Greater White-fronted Goose	Bird	*Anser albifrons*	2
Lesser Prairie-chicken	Bird	*Tympanuchus pallidicinctus*	2
Marbled Murrelet	Bird	*Brachyramphus marmoratus*	2
Northern Shoveler	Bird	*Anas clypeata*	2
Piping Plover	Bird	*Charadrius melodus*	2
Sandhill Crane	Bird	*Grus canadensis*	2
Sharp-shinned Hawk	Bird	*Accipiter striatus*	2

Common name	Taxon	Scientific name	Count
Tasmanian Wedge-tailed Eagle	Bird	*Aquila audax fleayi*	2
Western Marsh Harrier	Bird	*Circus aeruginosus*	2
White-bellied Sea-Eagle	Bird	*Haliaeetus leucogaster*	2
Willow Ptarmigan	Bird	*Lagopus lagopus*	2
Golden Eagle	Bird	*Aquila chrysaetos*	1
African Bearded Vulture	Bird	*Gypaetus barbatus meridionalis*	1
African Collared Dove	Bird	*Streptopelia roseogrisea*	1
African White-backed Vulture	Bird	*Gyps africanus*	1
Andean Condor	Bird	*Vultur gryphus*	1
Bean Goose	Bird	*Anser fabalis*	1
Bicknelle's Thrush	Bird	*Catharus bicknelli*	1
Bird-like Noctule	Bird	*Nyctalus aviator*	1
Black-tailed Godwit	Bird	*Limosa limosa*	1
Black Harrier	Bird	*Circus maurus*	1
Black Stork	Bird	*Ciconia nigra*	1
Black Tern	Bird	*Chlidonias niger*	1
Blue-gray Gnatcatcher	Bird	*Polioptila caerulea*	1
Bonelli's Eagle	Bird	*Aquila fasciata*	1
Brolga	Bird	*Grus rubicunda*	1
Burrowing Owl	Bird	*Athene cunicularia*	1
Cantabrian Capercaillie	Bird	*Tetrao urogallus cantabricus*	1
Columbian Sharp-tailed Grouse	Bird	*Tympanuchus phasianellus columbianus*	1
Common Kestrel	Bird	*Falco tinnunculus*	1
Common Loon	Bird	*Gavia immer*	1
Common Pochard	Bird	*Aythya farina*	1
Common Starling	Bird	*Sturnus vulgaris*	1
Common Tern	Bird	*Sterna hirundo*	1
Cooper's Hawk	Bird	*Accipiter cooperii*	1
Dupont's Lark	Bird	*Chersophilus duponti*	1
Eastern Meadowlark	Bird	*Sturnella magna*	1
Eurasian Eagle Owl	Bird	*Bubo bubo*	1
Eurasian Spoonbill	Bird	*Pilatalea leucorodia*	1
European Robin	Bird	*Erithacus rubecula*	1
Franklin's Gull	Bird	*Leucophaeus pipixcan*	1
Gadwall	Bird	*Anas strepera*	1
Galapagos Petrel	Bird	*Pterodroma phaeopygia*	1
Grasshopper Sparrow	Bird	*Ammodramus savannarum*	1
Great Horned Owl	Bird	*Bubo virginianus*	1
Great White Pelican	Bird	*Pelecanus onocrotalus*	1
Grouse	Bird	Galliformes	1
Harris's Hawk	Bird	*Parabuteo unicinctus*	1
Lark Sparrow	Bird	*Chondestes grammacus*	1
Little Swift	Bird	*Apus affinis*	1
Loggerhead Shrike	Bird	*Lanius ludovicianus*	1
Magpie	Bird	*Pica pica*	1

Common name	Taxon	Scientific name	Count
Marbled Godwit	Bird	*Limosa fedoa*	1
Mccown's Longspur	Bird	*Rhynchophanes mccownii*	1
Mountain Hawk-eagle	Bird	*Nisaetus nipalensis orientalis*	1
New Zealand Falcon	Bird	*Falco novaeseelandiae*	1
Northern Bobwhite	Bird	*Colinus virginianus*	1
Northern Cardinal	Bird	*Cardinalis cardinalis*	1
Northern Harrier	Bird	*Circus hudsonius*	1
Painted Bunting	Bird	*Passerina ciris*	1
Pheasant	Bird	*Phasianus colchicus*	1
Pink-Footed Goose	Bird	*Anser brachyrhynchus*	1
Purple Martin	Bird	*Progne subis*	1
Red-breasted Goose	Bird	*Branta ruficollis*	1
Red-winged Blackbird	Bird	*Agelaius pheoniceus*	1
Ruddy-headed Goose	Bird	*Chloephaga rubidiceps*	1
Saker Falcon	Bird	*Falco cherrug*	1
Sanderling	Bird	*Calidris alba*	1
Tree Swallow	Bird	*Tachycineta bicolor*	1
Tundra Bean Goose	Bird	*Anser serrirostris*	1
Verreaux's Eagle	Bird	*Aquila verreauxii*	1
Wedge-tailed Eagle	Bird	*Aquila audax*	1
White-eyed Vireo	Bird	*Vireo griseus*	1
Whooper Swan	Bird	*Cygnus cygnus*	1
Willet	Bird	*Catoptrophorus semipalmatus*	1
Wilson's Phalarope	Bird	*Phalaropus tricolor*	1
Wood Pigeon	Bird	*Columba palumbus*	1
Woodcock	Bird	*Scolopax rusticola*	1

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