A review of the impacts of onshore wind energy development on biodiversity

Research and Development Series 14/02
A report commissioned by the Northern Ireland Environment Agency

A review of the impacts of onshore wind energy development on biodiversity

Authors: David Tosh, W. Ian Montgomery & Neil Reid

Contractors: *Quercus*, Queen's University Belfast (QUB)

Quercus hosts the Natural Heritage
Research Partnership (NHRP)
between the Northern Ireland
Environment Agency and
Queen's University Belfast.
www.quercus.ac.uk

This report should be cited as:

Tosh, D.G., Montgomery, W.I. & Reid, N. (2014). *A review of the impacts of wind energy developments on biodiversity.* Report prepared by the Natural Heritage Research Partnership (NHRP) between *Quercus*, Queen's University Belfast and the Northern Ireland Environment Agency (NIEA) for the Research and Development Series No. 14/02

For further information on this report please contact:
Claire Hempsey,
Northern Ireland Environment Agency,
Countryside and Coast Unit,
Klondyke Building,
Gasworks Business Park,
Lower Ormeau Rd,
Belfast.
BT7 2JA.

EXECUTIVE SUMMARY

1. The UK Government is committed to the conservation of indigenous biodiversity as well as renewable energy targets. Renewable electricity currently represents nearly 19% of total electricity consumption with a target of 40% by 2020. At present there are 34 operational onshore wind energy developments (> 1 turbine) in Northern Ireland consisting of 313 turbines with a capacity of 533.10MW. In addition there are currently 38 single turbines operating with a capacity of 15.61MW. A further 291 wind energy developments are consented (35 wind developments (> 1 turbine) and 258 single turbines); consisting of 486 turbines with a capacity of 598.12MW (82.77MW from single turbine developments).

2. In Northern Ireland, the Department of Enterprise, Trade and Investment (DETI) is responsible for the promotion of renewable energy whilst the Natural Heritage Directorate (NHD) of the Northern Ireland Environment Agency (NIEA) is responsible for habitat and species conservation targets. Potential conflicts of interest require an assessment of the impact of onshore wind energy developments on biodiversity.

3. We reviewed the evidence for the impact of wind energy developments on biodiversity, examining scientific peer-reviewed publications and relevant grey literature, to evaluate effects on habitats (e.g. peatlands), birds, bats and other organisms.

4. Our results indicate that onshore wind energy construction and operation can have significant negative effects on local and regional biodiversity. However, the occurrence and magnitude of these effects varies between taxa, species, habitats and site. It should be recognised that publication bias is likely to favour the dissemination of negative results.

5. The interaction between onshore wind energy facilities and birds has been the focus of the majority of research to date. However, there has been a recent increase in research on the interaction between bats and wind energy facilities particularly in North America and continental Europe. The recognition of the potential negative impacts on migratory and tree roosting bats has increased in British and Irish research which has focused on their interactions with centralised wind energy development (i.e. wind farms). However, there is limited research on both the immediate and cumulative impacts that single wind turbines may have on biodiversity.

6. In general, the impact of wind energy developments can be summarised in 3 categories:
    i. *Displacement through disturbance,*
    ii. *Direct mortality through collision with operational turbine blades or powerlines,*
    iii. *Direct habitat loss through construction of windfarm infrastructure.*

7. The majority of studies report varying levels of bird mortality or displacement at onshore wind energy facilities. However, impacts varied within- and between-sites and were highly species-specific. Birds of prey (particularly soaring species) were notably vulnerable to collision with rotating blades and direct mortality whilst other aerial species may be vulnerable to barrier effects and/or displacement.

8. An increasing body of evidence also indicates that wind energy developments can have a negative impact on bats, which are more vulnerable than previously thought. In particular, tree roosting species and those that migrate appear to be at greatest risk from wind farm related mortality especially if wind energy developments are close to utilized habitats or migratory routes (e.g. ridges). There may also be impacts in the vicinity of bat swarming sites but this has, hitherto, received little or no research.

9. Although there is little direct research on the effects of siting wind energy development on peat bogs, inferences can be made about the potential impacts from research into the effects of peat cutting and forestry. Studies suggest that site drainage can be affected which affects the ecology of the bog and also has implications for downstream river catchments and consequently water quality.

10. Compared to birds and bats there has been relatively little work conducted on the impacts, if any, on terrestrial mammals, reptiles, amphibians, invertebrates, vascular plants, habitats or ecosystems. Therefore, it is difficult to generalise the wider impacts of wind energy on biodiversity *per se*.

Contents

1.0 Introduction

In 2009, the European Commission passed legislation that required member states to produce 20% of electricity consumed from renewable energy sources (EU Renewable Energy Directive 2009/29/EC). The Directive set a 15% target for the total amount of energy across the UK that should come from renewable sources by 2020 (DETI, 2013).

In 2010, the Northern Ireland Executive agreed renewable energy targets within Department of Enterprise, Trade and Investment (DETI)'s Strategic Energy Framework (SEF) which included a target of 40% electricity consumption from renewable resources by 2020 (DETI, 2010).

To date, onshore wind has been the main contributor to the SEF targets and at February 2014, the renewable electricity level stood at just under 19%. At present, there are 34 operational onshore wind energy facilities (> 1 turbine) in Northern Ireland consisting of 313 turbines (capacity of 533.1MW) and a further 35 wind energy facilities are consented consisting of 228 turbines with a capacity of 576MW (www.renewablesuk.com). In addition there are a further 38 single turbines operational (15.61MW) and a further 258 consented (82.77MW) (www.renewableuk.com accessed on 6th March 2014).

In December 2013 DETI published the Onshore Renewable Electricity Action Plan (OREAP) 2013-2020 which was subject to a Strategic Environmental Assessment (SEA) and a Habitats Regulations Appraisal (HRA) (DETI 2013). While it is expected that the primary contribution from onshore technologies would continue to come from wind, other renewable sources such as biomass/ anaerobic digestion, landfill gas and solar were also considered. In order to assess the potential impacts of increasing renewable electricity deployment on the environment, a number of scenarios were developed for a range of onshore technologies and various combinations which could contribute to the 40% target by 2020.

With regard to possible further onshore wind development, the SEA proposed the consideration of regional capacity studies focussing on receptors where there is potential for significant adverse cumulative effects to occur once development exceeds the low generation scenario for onshore wind (800-1000MW) (DETI, 2013). These related to landscape, ecology and birds.

DETI in conjunction with DOE and NIEA considered these proposals and, with regard to the latter two issues, the following action was included within the OREAP; "*DOE/ NIEA will review existing NI, UK and international data/studies/research into the impacts of wind farms on biodiversity, including birds, bats and habitats. In light of the outcome of this review, to be completed by March 2014, DOE/NIEA will consider the need and scope for any further work at regional level.*"

NIEA is responsible for achieving habitat and species conservation requirements set down in the EU Habitats Directive, EU Birds Directives, EU Water Framework Directive, UK Biodiversity Action Plans (BAP) and regional strategies that include the Natural Heritage Biodiversity Implementation Plan 2008/09 and Wildlife (Northern Ireland) Order (1985). NIEA are consulted for proposed wind turbine installations with respect to planning, mitigation and advise on potential impacts on biodiversity through Environmental Impact Assessment (EIA) reports.

NIEA commissioned the Natural Heritage Research Partnership (NHRP) with Quercus, Queen's University Belfast to undertake this review.

2.0 Methods

2.1 Literature review

An initial search of peer reviewed literature on the impacts of wind energy developments on biodiversity was conducted using articles obtained from the ISI

Web of Knowledge and Google Scholar (white literature). Search terms were tailored for each subject area (Table 1) and results within ISI Web of Knowledge were further restricted to the subject areas of ‘Environmental Sciences & Ecology’, ‘Biodiversity & Conservation’, ‘Marine & Freshwater Biology’ and ‘Zoology’. Unpublished reports (grey literature) were consulted and where appropriate results were included. Not all reviewed literature was included in this document as not all were relevant to biodiversity in Ireland e.g. impacts of wind energy on tortoises (e.g. Lovich *et al.,* 2011). As the majority of wind farm developments occur on peatlands in Northern Ireland, additional searches were made on the impacts they may have in these habitats. Little direct research of the impacts wind energy facilities have on peatlands has been undertaken to date. Therefore, related research into other human related activities informed the review of literature. Search terms encompassing drainage on peatlands arising from the conversion of peatland to forestry, for agriculture and peat harvesting were used. Additional searches were also made for peat restoration practices.

Table 1 Search terms used to collate peer reviewed publications in ISI Web of Knowledge and Google Scholar.

Wind term		***Biodiversity term***
Windfarm	**AND**	Agriculture
Wind energy		Amphibian (s)
Wind farm		Bat (s)
Wind turbine		Bird (s)
		Forestry
		Habitat (s)
		Invertebrate (s)
		Mammal (s)
		microclimate
		Peat bog/land
		Peat cutting/harvesting

3.0 Results

3.1 Birds

Birds are the most well studied taxa with respect to the impact of wind energy development with over 150 peer-reviewed papers published since 1996 (Figure 1). In addition, several unpublished reports, reviews and meta-analyses have been conducted (e.g. Crockford, 1992; Gill *et al.,* 1996; Hötker *et al.,* 2006; Kuvlesky *et al.,* 2007; Stewart *et al.,* 2007). The majority of peer-reviewed studies on wind energy and birds originate from Europe (57%) with British (18.3%) and Spanish (14.7%) research constituting the bulk of this output. A further 48 (33.8%) studies originate from North America and an additional 13 (9.2%) from elsewhere. This trend likely reflects the level of installed capacity throughout Europe and North America which account for 38.6% and 24.0% of the worlds installed wind power capacity respectively (GWEC, 2013). Three wind development areas were notable foci of a large amount of the published research; Altamont Pass Wind Resource Area (APWRA) in California (*n*=10); Buffalo Ridge Wind Resource Area (BRWRA) in Minnesota (*n*=6), and; Tarifa in southern Spain (*n*=11).

Estimating bird mortality rates constitute 31.0% of all published research efforts followed by investigations of impact on behaviour and breeding success (20.4%), planning, modelling and cumulative impacts (17.6%) and other studies including reviews and meta-analyses (29.5%). Studies focusing on single species constitute the bulk of research with raptorial birds or Falconiformes (*n*=34) the most commonly studied order (Figure 2).

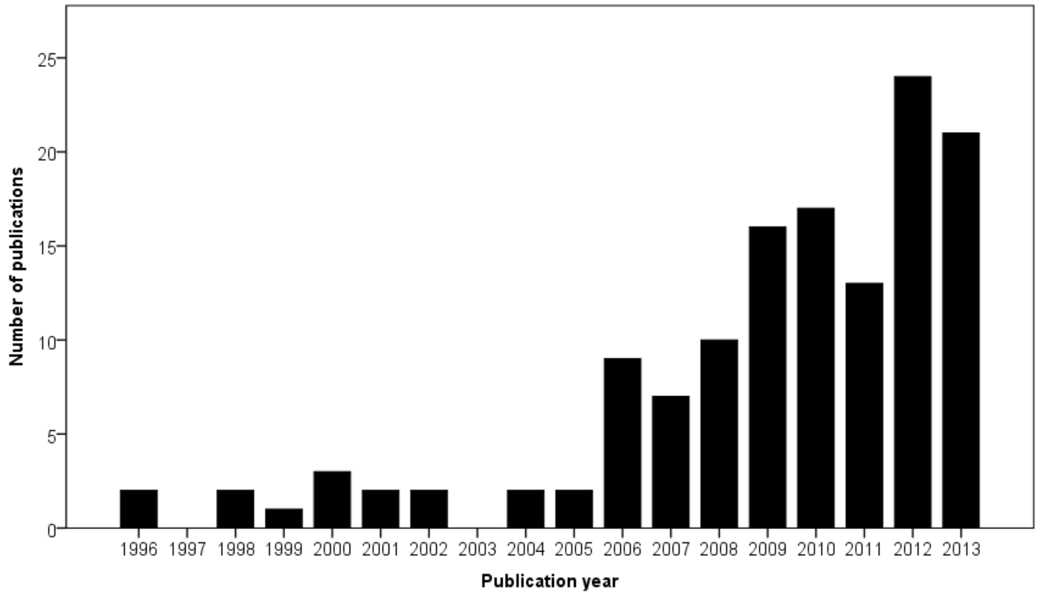

Figure (1). Number of peer reviewed publications since 1996 that involve birds and wind energy

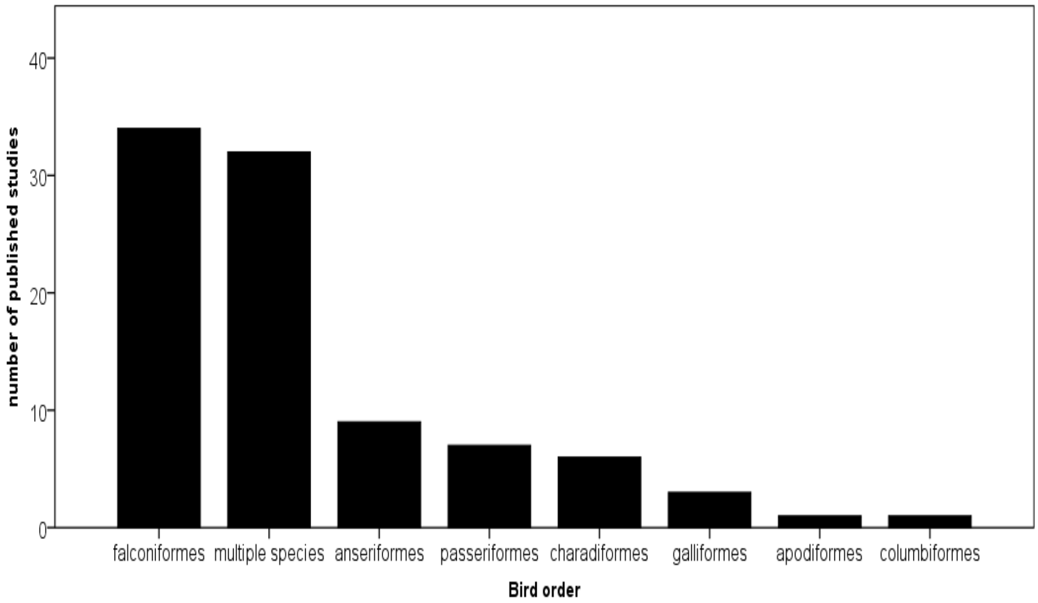

Figure (2). Number of peer reviewed publications per bird order.

Interactions between birds and wind energy facilities

Previous reviews of wind energy and birds typically divide impacts into direct and indirect effects or lethal and non-lethal impacts (Crockford *et al.,* 1992; Gill *et al.,* 1996; Hötker *et al.,* 2006). Direct or lethal impacts are mortality caused by collisions with turbines and their associated infrastructure and indirect impacts, which are not necessarily non-lethal, are habitat loss or displacement. The occurrence, level and causes of mortality are the focus of the vast majority of research while displacement and the cumulative impacts of wind energy facilities are less well studied. The effects of wind energy on birds are time-, species- and site-specific with multiple factors contributing or explaining why mortality or displacement occurs.

Direct or lethal impacts (i.e. mortality)

Mortality of birds at wind energy facilities can arise from direct collisions e.g. with wind turbine blades, towers or nacelles, meteorological masts and power-lines (Drewitt & Langston, 2008; Kerlinger *et al.,* 2012) or from being forcibly thrown to the ground by air turbulence produced in a turbines wake (Winkleman, 1992). Although the majority of studies do not differentiate between causes of death at wind energy facilities it is typically assumed that death arises from collisions with turbine blades, towers or nacelles (e.g. Barrios & Rodriguez 2004). Therefore, the proportion of birds killed by each hazard at a wind energy facility cannot be differentiated with certainty.

The majority of peer reviewed literature, and post-construction mortality monitoring studies, report bird mortality at wind farm facilities. The level of mortality reported varies markedly (Table 2) and few examples exist where no mortality is recorded e.g. Kerlinger (1997). In peer-reviewed literature, researchers either assume that the number of birds found during monitoring reflects true mortality (e.g. Barrios & Rodriguez, 2004; de Lucas *et al.,* 2012) or those found underestimate the total number of birds killed by wind turbines (e.g. Johnson *et al.,* 2002; Smallwood & Karas, 2009). When the latter is the case, estimators (calculations estimating total numbers of birds killed) are typically used to extrapolate annual mortality rates at wind energy facilities based on the number of birds found (see Bernardino *et al.,* 2013 for examples). The results (number of birds killed) produced by estimators can be affected by a number of factors including; habitat, searcher efficiency, the size of the search area, the number of search intervals, timing of searches, the persistence of carcasses and rates of scavenging (Smallwood, 2007; Arnett *et al.,* 2008; Huso *et al.,* 2011; Bernardino *et al.,* 2013). It should be noted that as different estimators weight contributory factors and calculate them differently, comparisons of mortality estimates using different estimators should be made with caution. In contrast to peer reviewed publications, the majority of post-construction monitoring studies assume the number of birds found during carcass searches underestimates the level of mortality at a wind energy facility and estimators are used to calculate total mortality. Regardless of how estimates of bird mortality are reached, the results can be presented in a number of ways. The most common form is to present the total number of birds killed per year and subsequently present these results as either; the number of birds killed per turbine per year (turbine/year^{-1}); or the number of birds killed per Mega Watt (MW) per year (MW/year^{-1}). Reporting of mortality per MW/ year^{-1} is recommended by Natural England as it allows easier comparisons of the effects of wind energy facilities of different size or output (Anon., 2010a).

Table 2. Comparison of estimated bird mortality rates at wind energy facilities in North America.

Location	Year	No. of Turbines	Total Capacity (MW)	Estimated mortality	Deaths /MW/year	Deaths /turbine year	Source
Mount Storm, USA	2010	132	264.0	889	3.4	6.7	Young *et al.,* 2010: 2011
Hatchet Ridge, USA	2011	44	101.2	545	5.4	12.4	Anon, 2013a
Hatchet Ridge, USA	2012	44	101.2	210	2.1	4.7	Anon, 2013a
Montezuma, USA	2012/13	34	78.2	84	1.2	2.5	Anon, 2013b
Wolf Island, Canada	2011	86	197.8	466	2.3	5.4	Anon, 2011a: 2011b
FEC, USA	2009	86	127.5	471	3.7	5.6	Grodsky & Drake 2012
FEC, USA	2010	86	127.5	77	0.6	0.9	Grodsky & Drake 2012
Wild Horse, USA	2007	127	229.0	354	1.5	2.8	Erickson *et al.,* 2008

Peer reviewed studies and post-construction monitoring reports usually state the species found dead and the number (e.g. Hull *et al.,* 2013; Anon., 2013a) but rarely do they provide an estimate for the total number of each species estimated to be killed. Estimates of total annual mortality typically report total number of bird casualties at a site and do not differentiate between species (e.g. Erickson *et al.,* 2001). However, exceptions to this do occur with the number of small birds, medium birds, large birds and number of raptors killed at a facility in a year reported in post construction monitoring reports (e.g. Anon., 2013a). This restriction likely arises from the low numbers of each species found during carcass searches making more accurate estimates difficult if not impossible. Further exceptions to how mortality is reported do occur in the case of raptors. Total numbers killed for the order are produced and where data exists for individual species (Smallwood & Thelander, 2004; Erickson *et al.,* 2005). Zimmerling *et al.* (2013) is a rare example of a study reporting annual mortality estimates for multiple species. This study from Canada details annual mortality estimates for the 20 species most commonly found dead at wind energy facilities (Zimmerling *et al.,* 2013).

At present, national mortality estimates exist only for the USA (Erickson *et al.,* 2001; 2005; Smallwood, 2013) and Canada (Zimmerling *et al.,* 2013). Mortality rates in the USA were estimated at between 20,000 to 37,000 birds during 2004 (Erickson *et al.,* 2005) and in Canada between 20,000 and 28,300 birds were killed in 2012 (Zimmerling *et al.,* 2013). However, more recent estimates put mortality in the USA at up to 573,000 birds a year (Smallwood, 2013). The difference is thought to reflect increased capacity but also the improvement of mortality estimation methods and increased mortality monitoring at wind energy facilities (Smallwood, 2013). However, this disparity may equally be related to varying correction factors rather than higher numbers of carcasses found (Zimmerling *et al.,* 2013). Whatever the reasons for the difference, bird mortality at North American wind energy facilities accounts for < 1% of total avian collision fatalities i.e. including strikes with buildings, vehicles, power lines (Erickson *et al.,* 2005; Zimmerling *et al.,* 2013). At present there are no annual mortality estimates for birds in the UK or Ireland. This is presumably a result of the limited amount of publicly available data on the occurrence of bird kills and the assumed low number and species of birds killed at wind energy facilities in the British Isles.

Reasons for mortality: siting of wind energy development

A variety of factors have been identified as increasing mortality, one of which is the wind energy development's location within a landscape. Those located along migratory routes, flight corridors, on slopes of hills, mountain ridges and within quality foraging or breeding grounds pose the greatest collision risk to birds and often have the highest recorded mortality rates (Osborn *et al.,* 1996; Drewitt &

Langston, 2006; 2008; Everaert & Steinen, 2007; de Lucas *et al.,* 2008; Hötker, 2008; Smallwood *et al.,* 2007; Smallwood *et al.,* 2009; Telleria, 2009a; b). Consequently, poor spatial planning can result in specific wind resource areas accounting for a disproportionately high number of bird mortalities e.g. Tarfia and Navarra in Spain and Buffalo Ridge and APWRA in the USA. It is argued that in the UK and Ireland, the siting of wind energy facilities in habitats with low levels of bird activity e.g. upland bogs may account for the low levels of recorded mortality compared to the USA or Europe (Fielding *et al.,* 2006; Bright *et al.,* 2008).

Mortality rates can also be turbine specific leading to within wind farm differences in mortality e.g. majority of turbines cause no deaths (Martinez-Abrain *et al.,* 2012; de Lucas *et al.,* 2012). Seaward facing turbines at coastal sites (Everaert & Steinen, 2007) or end-of-row turbines can cause high rates of mortality (Orloff & Flannery, 1992; Barrios & Rodriguez, 2004; Smallwood & Thelander, 2004; Drewitt and Langston, 2008; Smallwood & Karas, 2009; Smallwood *et al.,* 2009b). Similarly, turbines located on gentle slopes that have less lift or on high wind facing ridges that deflect updrafts, pose a greater risk to soaring species e.g. raptors (Barrios and Rodriguez, 2004; Hoover & Morrison, 2005; de Lucas *et al.,* 2008;). Active avoidance of wind turbines post-construction would be expected to cause reduced bird mortalities over time (Gavin *et al.,* 2011); however, the majority of studies that consider long-term effects do not support this view, with prolonged high mortality rates being frequently reported over successive years (Smallwood, 2013).

Reasons for mortality: turbine features

Results from a small number of individual wind energy facilities (Tarfia in Spain and AWPRA in the USA) indicate that bird mortality increases with turbine age, turbine size, slow to intermediate wind tip speeds and rotor diameter (e.g. de Lucas *et al.,* 2008; Smallwood & Thelander, 2004). However, the majority of studies (mainly meta-analyses) do not indicate that a clear statistical relationship exists between bird mortality and the features of a turbine (Hötker *et al.,* 2006; Barclay *et al.,* 2007; Stewart *et al.,* 2007; Pearce-Higgins *et al.,* 2012). Currently, there is no compelling evidence that repowering (increasing the capacity) of older turbines will increase the collision risk for birds (Stewart *et al.,* 2007; Drewitt & Langston, 2008) as there is no clear relationship between turbine height and capacity (MW) (Pearce-Higgins *et al.,* 2012). It is more likely that the location of a particular turbine within the landscape has a greater effect on mortality on birds rather than turbine-specific features (Hötker *et al.,* 2006).

The lighting on man-made structures has been shown to attract birds (Erickson *et al.,* 2001), particularly when visibility is poor (Drewitt & Langston, 2008), and it has been suggested that lighting on turbines may increase collision risk although no supporting evidence exists at present (Kuvlesky *et al.,* 2007; Drewitt & Langston, 2008). The effect of tower type e.g. lattice or tubular on mortality of birds is also not clear. Older lattice towers were originally thought to increase the risk of bird mortality due to increased perching opportunities (Orloff & Flannery, 1992). However, higher mortality has been reported at tubular towers (Thelander *et al.,* 2003; Smallwood & Thelander, 2004), or is found to be similar to that of lattice towers (Barrios & Rodriguez, 2004; Smallwood & Neher 2004; Drewitt & Langston, 2008).

Reasons for mortality: time of year

Bird mortality at wind energy facilities can occur throughout the year (Kuvlesky *et al.,* 2007; Drewitt & Langston, 2008; de Lucas *et al.,* 2008; Carette *et al.,* 2012; de Lucas *et al.,* 2012) but peaks in mortality of passerines and raptors have been recorded in the USA (Johnson *et al.,* 2002), Europe (de Lucas *et al.,* 2008; Carette *et al.,* 2012; de Lucas *et al.,* 2012) and Australia (Bull *et al.,* 2013) during the migrating (non-breeding) season. The vulnerability of migrating species may arise from large aggregations of birds in a small area, unfamiliarity with an area (Drewitt & Langston, 2008) and the propensity for some migrating species to fly at lower altitudes (Kemp *et al.,* 2013). In addition, the reduction in the occurrence of thermals in autumn and winter months may also lead to increases in soaring bird mortalities at wind energy facilities e.g. raptors (Barrios & Rodriguez, 2004; de Lucas *et al.,* 2008). This arises from soaring birds switching to updrafts created by hill slopes and ridges for lift instead of thermals.

Reasons for mortality: weather

Weather conditions are also believed to influence the risk of birds colliding with wind turbines. More bird collisions are recorded during poor weather (high winds, rain, fog, low cloud) than good weather (e.g. Winkleman 1992; Drewitt & Langston, 2006). This may be due to behavioural responses of birds to inclement weather conditions e.g. flying lower during certain weather (Drewitt & Langston, 2008). In stronger head winds birds fly at lower heights and soaring raptor species are observed more often in high winds than passerines and, therefore, may be at greater risk of collisions when at lower speed (Drewitt & Langston, 2008; Forfan *et al.,* 2009).

Reasons for mortality: species morphology and behaviour

Although a wide variety of bird species have been recorded dead at wind energy facilities, not all species of birds are at risk of collision. This is aptly demonstrated by Hull *et al.,* (2013) who reported that only 18 -21% of bird species recorded at wind farm sites were found dead during searches around wind turbines. This study, and others before it, have identified that Anseriiformes (swans, geese, ducks), Charadiformes (waders), Falconiformes (raptors), Strigiformes (owls) and Passerines are at greatest risk of collision (Johnson *et al.,* 2002; Stewart *et al.,* 2007; Kuvlesky *et al.,* 2007; Drewitt & Langston, 2008; Ferrer *et al.,* 2012; Bull *et al.,* 2013). Therefore, what puts a species of bird at risk of collision is likely influenced by a combination of factors that include morphology, ecology and foraging strategy (Smallwood *et al.,* 2009; Carette *et al.,* 2012; de Lucas *et al.,* 2012; Herrera-Alsina *et al.,* 2013; Hull *et al.,* 2012).

Although many species of birds e.g. raptors have a high visual acuity (acuteness and sharpness of vision), the lateral placement of eyes in birds means that the direction of best vision in birds is not forwards, as in the binocular vision of humans, but perpendicular (Martin, 2012; Martin *et al.,* 2012). Thus, birds better perceive their environment laterally than directly in front or behind them where they have limited perception. This may be further accentuated during flight, particularly for raptors, which typically orientate their heads downward whilst foraging (Martin & Shaw, 2010; Martin, 2012).

In addition to vision, the flight height and foraging strategy, which are linked to morphology, is associated with the risk of collision in birds. Some species may not avoid flying through or close to turbines and, therefore, remain at high risk of collision (Musters *et al.,* 1996; Ahlen, 2002; Pearce-Higgins *et al.,* 2009). Additionally, those species that fly within the rotor swept area, particularly large birds including swans, geese and birds of prey, may also have a limited ability to alter their speed in response to the sudden appearance of obstacles such as turbine blades (de Lucas *et al.,* 2008; Martin, 2012). The lack of manoeuvrability of some species is linked to longer wings and higher wind loadings and has been linked to collisions with other man-made objects (Bevanger, 1998). These species have been found to be more prone to collision as they are more likely to fly in the "risk zone" (rotor swept area) and are at subsequent risk of collision (Herrera-Alsina *et al.,* 2013). However, smaller birds with smaller wind loadings have been found to more likely to collide with turbines and this has been linked to foraging strategy (Herrera-Alsina *et al.,* 2013). Similar findings from Australia also suggest that those birds that are aerial foragers are at greatest risk of collision with wind turbines (Hull *et al.,* 2013).

As the behaviour of different bird species varies in response to wind turbines, Collision Risk Models (CRM) have been developed to assess the potential impact of wind energy developments on bird populations. Scottish Natural Heritage's CRM or Band Model (Band, 2000; Band *et al.,* 2007) is used widely to assess risk to different species by incorporating the probability of collision (a measure of bird dimensions, flight speed and turbine characteristics) with the number of birds flying through the collision zone (the rotor swept area) at a site and avoidance rates of birds (mortality rate divided by number of birds at risk) (Anon., 2010b). Avoidance rates are considered to be species specific but at present are poorly defined and inferences are made for similar species (Chamberlain *et al.,* 2006; Anon., 2010b). Avoidance rates have been derived for red kites (Whitfield & Madders, 2006), golden eagle (Whitfield, 2009), geese (Anon., 2013c) and others (Anon., 2010b) but it is argued that rates should be derived *in situ* for similar species under similar conditions (time of day, weather, observers) (Chamberlain *et al.,* 2006; Anon., 2010b). A further consideration is survey effort as this has been shown to increase the variability of collision risk models in white tailed sea eagles (*Haliaeetus albicilla*; Douglas *et al.,* 2012).

Reasons for mortality: species abundance

Pre-construction abundance bird surveys have been typically used in Environmental Impact Assessments to assess the risk posed. However, there is conflicting evidence whether the abundance or density of birds (breeding and/or foraging) prior to wind energy development construction is a useful predictor of post-construction mortality rates (Barrios & Rodriguez, 2004; Smallwood *et al.,* 2009b; Whitfield & Madders, 2006; de Lucas *et al.,* 2008; Hull *et al.,* 2012; Carette *et al.,* 2012; Ferrer *et al.,* 2012). Mortality of raptors (e.g. kestrel (*Falco tinnunculus*) and griffon vultures (*Gypus fulvus*)) in Spain (Barrios & Rodriguez, 2004; Carette *et al.,* 2012) has been reported to be correlated to abundance whilst in other studies from the same region no correlation has been found (de Lucas *et al.,* 2008). Additionally, particular topographic and the layout of wind energy facilities may be of greater concern than a particular species that occurs or their abundance (de Lucas *et al.,* 2004; Smallwood *et al.,* 2009). Therefore, any assessment and monitoring of individual developments is of particular importance since published results may not be transferable to other sites (e.g. de Lucas *et al.,* 2004).

Indirect impacts: displacement

In addition to mortality, the construction of wind energy facilities can indirectly impact resident bird populations via disturbance (Langston & Pullan, 2003). Disturbance can lead to the displacement of birds which effectively leads to the loss of habitat available to birds (Langston & Pullan, 2003). Opinions on the occurrence of displacement have varied over time. Early studies suggested it did not occur (Gill *et al.,* 1996) whilst others either agree on its negative impacts in the majority of cases (Kuvlevsky *et al.,* 2007; Stewart *et al.,* 2007; Drewitt & Langston, 2008), or suggest that it is complex and that its occurrence varies between species and site with site specific responses within species (Langston & Pullan, 2003; Hötker *et al.,* 2006). The latter appears to be the general consensus as more studies indicate variation in how different species respond to wind farm construction (e.g. Garvin *et al.,* 2011; Hull *et al.,* 2013; Stevens *et al.,* 2013).

Displacement can typically occur in two ways; from the avoidance of suitable habitats occupied by wind energy facilities (Larsen & Madsen, 2000; Madsen & Boertmann, 2008; Pearce-Higgins *et al.,* 2008) or via extended flights arising from wind energy developments acting as barriers to movement (Winkelman, 1985; Still *et al.,* 1997; de Lucas *et al.,* 2004; Madsen *et al.,* 2009). The “barrier effect” has been recorded for 81 species and includes those that migrate long distances or those making localised movements e.g. from breeding grounds to foraging grounds (Hötker *et al.,* 2006; Masden *et al.,* 2009a; Sugimoto & Matsuda, 2011; Plonczhier & Simms, 2012). Concern was initially raised on the effects that extended flights would have on the energetics of long-distance migrants but the consequent increase in energetic expenditure appears negligible, at least for geese (Masden *et al.,* 2009a). It should be noted, that Masden *et al* (2009a) report that energetic effects arising from the avoidance of a single wind energy facility are negligible but that the cumulative effect may be larger. However at present this is untested.

Avoidance of suitable habitats can occur particularly during construction and subsequent operation of wind energy facilities and is typically measured via assessments of the distribution or abundance of a species within an area before and after construction (e.g. Dahl *et al.,* 2012). As might be anticipated, abundance of birds post-construction has been shown to vary between species and sites. Golden plover (*Pluvialis apricaria*) and red grouse (*Lagopus lagopus scotica*) numbers have been found to recover (to pre-construction levels) but snipe (*Gallinago gallinago*), curlew (*Numenius arquata*) and kestrel did not (Farfan *et al.,* 2009; Gavin *et al.,* 2011; Douglas *et al.,* 2011; Pearce-Higgins *et al.,* 2012). Notably, increases in stonechat (*Saxicola rubicola*) and skylark (*Alauda arvensis*) abundance have been recorded from upland areas of the UK which has been attributed to favourable alteration of habitats following construction (Pearce-Higgins *et al.,* 2012). It should be noted that little research has been conducted on small wind turbines (those below 50kw) but what has been done shows that they have no effect on bird activity (Minderman *et al.,* 2012). No research has been conducted on the effects of the erection of single large wind turbines, whose numbers have increased over the past 6 years due to grant support from the Northern Ireland Rural Development Program which has encouraged diversification of the rural economy (www.dardni.gov.uk/index/rural-development.htm). In addition, higher levels of support have been offered in recent years for small scale wind development under the Northern Ireland Renewables Obligation.

Avoidance of wind farms by breeding birds has been demonstrated in a variety of species but susceptibility to disturbance can show a substantial degree of intra-specific variation (Table 3). However, the negative impacts of wind energy facilities appear to be greatest on species within the orders Anseriiformes (swans, geese, ducks) and Charadriiformes (waders; Langstan & Pullan, 2003; Hötker *et al.,* 2006; Stewart *et al.,* 2007) with lesser effects observed in members of the Falconiiformes (birds of prey) and Passeriformes (perching birds; Madders & Whitfield, 2006; Devereaux *et al.,* 2008; Farfan *et al.,* 2009; Pearce-Higgins *et al.,* 2009; Hull & Muirs, 2013). Avoidance behaviour (habitat abandonment) appears to be particularly pronounced in wading species, although the distance that this occurs within is much debated.

Table 3. Comparison of breeding disturbance distances of Northern Ireland Priority Species for which information has been estimated or derived empirically. SD = Standard deviation.

		Disturbance distances m ± SD (breeding season)		
Common name	**Scientific name**	Pearce-Higgins *et al* (2009)	Hötker *et al* (2006)	Ruddock & Whitfield (2007)
Golden Plover	*Pluvialis apricaria*	<200		
Curlew	*Numenius arquata*	<800		
Buzzard	*Buteo buteo*	<500		
Hen harrier	*Circus cyaneus*	<250		<10 - 750
Skylark	*Alauda arvensis*	<200	100 ± 71	
Redshank	*Tringa tetanus*		188 ± 111	
Black tailed godwit	*Limosa limosa*		300 ± 357	
Lapwing	*Vanellus vanellus*		100 ± 110	
Yellow wagtail	*Motacilla flava*		50 ± 107	
Reed bunting	*Emberiza schoeniclus*		25 ± 70	
Linnet	*Carduelis cannabina*		125 ± 29	
Redwing	*Turdus iliacus*			<10 - 300
Common scoter	*Melanitta nigra*			<10 - 313
Goshawk	*Accipiter gentilis*			<10 - 500
Barn owl	*Tyto alba*			<10 - 100
Short eared owl	*Asio flammeus*			<10 - 500
Nightjar	*Caprimulgus europaeus*			<10 - 150
Fieldfare	*Turdus pilaris*			<10 - 150

Breeding curlew have been observed to avoid areas within wind energy facilities up to 800m (Pearce-Higgins *et al.,* 2009). Similarly, avoidance by lapwing (*Vanellus vanellus*) ranges from 500m to 850m depending on the season and the study (Hötker *et al.,* 2006). Snipe, wheatear (*Oenanthe oenanthe*), hen harriers (*Circus cyaneus*), buzzards (*Buteo buteo*) and golden plover have also been shown to avoid areas next to turbines to varying degrees within the British uplands (Pearce-Higgins *et al.,* 2009). However, the latter have also been found to not be affected at a single wind farm (Douglas *et al.,* 2011). Therefore, although avoidance behaviour does occur it varies between seasons, sites and between and within species. Despite the recognition by Ruddock & Whitfield (2007) that further research is required to evaluate avoidance distances for bird species little progress has been made in determining displacement distances of species at risk from wind energy developments. Restricting construction to outside the bird breeding season has been suggested as mitigation for displacement (Pearce-Higgins *et al.,* 2012) but, as yet, the effectiveness of this measure has not been tested. There is also no information on the response of returning migratory breeders to the presence of turbines constructed at nesting sites between breeding seasons.

Evidence for habituation (when a bird becomes accustomed to a wind turbine or wind farm over a period of time) of birds to wind energy developments over time is not clear. High mortality rates can continue over time but birds remaining within a wind farm area post-construction can avoid turbines (Hötker *et al.,* 2006; de Lucas *et al.,* 2008; Smallwood & Karas, 2009; Garvin *et al.,* 2011). Hötker *et al.,* (2006) indicate in their review that behaviour, indicative of habituation, was reported in wind farm studies without implicitly being stated as such but that if it occurs it is not widespread or a strong phenomenon. In their meta-analysis Stewart *et al.,* (2005) go further and suggest that habituation does not occur as wind farms have a persistent negative impact on bird abundance over time. The pink-footed goose (*Anser brachyrhynchus*) is the only species where apparent habituation has been reported in the peer reviewed literature (Madsen & Boertmann, 2008). Pink footed geese foraged 40 - 50% closer to turbines eight years after their initial installation despite avoidance still being evident within 50 - 100 m from operational turbines (Larsen & Madsen, 2000; Madsen & Boertmann, 2008). Long-term research focused on determining whether habituation occurs in birds has been recommended by previous reviews to determine its existence. However, as of yet, little progress has been made in determining its existence which is likely a consequence of the difficulty in separating the effects of other factors that affect bird distribution and abundance (Hötker *et al.,* 2006; Madders & Whitfield, 2006).

Despite declines in abundance being shown for some bird species breeding within the vicinity of wind developments there has been little research on their impact on breeding success. Although it is widely believed that mortality at wind energy facilities has little effect on most common bird populations, concern has been raised over the implications of wind farm mortality on long lived species that have low productivity and slow maturation rates e.g. raptors (Bright *et al.,* 2008; Carette *et al.,* 2009). Displacement of birds from suitable habitat has been suggested as causing decline in breeding success of white-tailed sea eagles in Norway and Griffon vultures in Spain (Dahl *et al.,* 2012). At present these are the only studies to demonstrate an effect on breeding success despite the suggestion by others that raptors are susceptible to these effects (e.g. Bright *et al.,* 2008; Carette *et al.,* 2009). The only other study examining breeding success showed that there was no difference within wind farm sites and control areas for dicksissel (*Spiza americana)*; Hatchet *et al.,* 2013).

Mitigating impacts

The removal of wind turbines that kill a disproportionately high number of birds (Martinez-Abrain *et al.,* 2013; Smallwood 2013), selectively stopping turbines during periods of high bird activity (de Lucas *et al.,* 2012), temporary curtailment during winter months and repowering of old turbines (Smallwood, 2013), have all been implemented as management actions to reduce bird mortality at wind energy facilities with varying levels of success. A number of studies have investigated increasing the visibility of turbines to reduce bird mortality, by painting blades with UV paint as birds are better able to perceive UV light (McIsaac, 2001; Young *et al.,* 2003). Although McIsaac (2001) demonstrated greater detection of blades painted with UV paint, field studies showed there was no difference in mortality rates with those of control sites (Young *et al.,* 2003).

Current Research

At present, research investigating the impacts of wind energy on birds is being undertaken by a number of independent groups. These include researchers under Maria Zwart at Newcastle University are investigating the effects of wind energy on black grouse and other birds in a project that is due to end in April 2014. At Stirling University, Drs Kirsty Parks and Jeroen Minderman are investigating the impacts of Small Wind Turbines (under 50kW) on bird populations. Natural Research Ltd, an independent consultancy, are undertaking ongoing research (self-funded) investigating the impacts of wind energy on curlew breeding success and the level of avoidance that occurs. In 2013, the Scottish Windfarm Bird Steering Group initiated a 5 year research project that will investigate a variety of questions pertaining in relation to birds and onshore wind energy. Research themes include a review of existing knowledge on the displacement of birds by wind turbines; a comparative study to investigate displacement in different bird species; a study to update and refine avoidance rates of key species; a PhD investigating the factors that affect bird collisions with turbines; a study into the cumulative impacts of turbines and; an evaluation of how habitat management affects bird use of wind energy.

3.2 Bats

The earliest published evidence of wind farm related bat mortality originates from Australia in 1972 when white-striped mastiff bats (*Tardarida australis*) were found dead under a turbine (Hall & Richards, 1972). Further evidence did not appear until the late 1990s when incidental reports of bat mortalities were reported in North America (Osborn *et al.,* 1996; Anderson *et al.,* 1999) and Europe (Dürr & Bach, 1996) during bird focused research. Since 2002, over 70 peer-reviewed publications (Figure 3), and numerous unpublished (grey literature) studies, have been completed. The majority of research originates from the USA and Canada (58%) and Europe e.g. Germany, Italy , Portugal, Slovenia, Spain, Sweden, UK (34%) with little originating from the rest of the world, principally Australia, Mexico, South Africa, India (8%). Research can be divided into four broad categories; i) mortality (46%); ii) behaviour (16%); iii) mitigation studies (21%); and iv) reviews (17%).

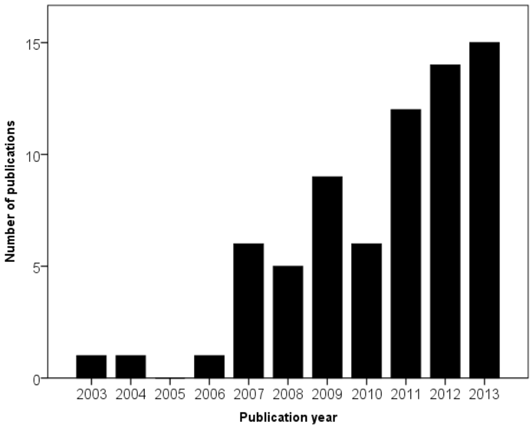

Figure 3. Number of peer reviewed publications published since 2003 on bats and wind turbines. (source: WoS and Google Scholar)

Bat mortality

Two studies, one each from North America (Arnett *et al.,* 2008) and Europe (Rydell *et al.,* 2010a), have reviewed estimates of bat mortality at wind energy. After initial underestimation (Kunz *et al.,* 2007a), bat-turbine collision rates have been shown to be higher than bird-turbine collision rates (Barclay *et al.,* 2007; Arnett *et al.,* 2008; Cohn, 2008; Sovacool, 2009) (Table 4). Bat mortality rates vary markedly between sites and are generally higher in North America than Europe. Arnett *et al.,* (2008) estimated mean numbers of bat deaths between 0.1 - 70 per turbine and 0.2 - 53 per MW per year in North America. In contrast, Rydell *et al.* (2010a) reported death rates between 0.4 - 18 per turbine and 1.1 - 11 per MW per year using data from 40 European (located in Austria, England, France, Germany and Switzerland) wind energy facilities. Techniques to estimate bat fatalities differ (see Bernardino *et al.,* 2013) and a number of factors, for example, the number of searches, the methods employed to search for carcasses, the observer variability in detection rates, the rate of carcass scavenging, and the estimator used, all influence estimates of bat fatalities (Bispo *et al.,* 2010), making direct comparisons difficult. At present, there is limited evidence of bat mortality at wind energy in Great Britain or Ireland although it is suspected to occur (Eurobats, 2013).

Table 4. Comparison of bat mortality estimates at wind farms in North America.

Location	Year	No. of Turbines	Total Capacity (MW)	Estimated mortality	Deaths /MW/year	Deaths /turbine year	Source
Mount Storm, USA	2010	132	264.0	4272	16.2	32.4	Young *et al.,* 2010: 2011
Hatchet Ridge, USA	2011	44	101.2	226	2.2	5.1	Anon, 2013a
Hatchet Ridge, USA	2012	44	101.2	529	5.2	12.0	Anon, 2013a
Montezuma, USA	2012/13	34	78.2	71	0.9	2.1	Anon, 2013b
Wolf Island, Canada	2011	86	197.8	534	2.7	6.2	Anon, 2011a: 2011b
FEC, USA	2009	86	127.5	2218	17.4	26.2	Grodsky & Drake 2012
FEC, USA	2010	86	127.5	1759	13.8	20.6	Grodsky & Drake 2012
Wild Horse, USA	2007	127	229.0	89	0.4	0.7	Erickson *et al.,* 2008

Causes of mortality

Causes of bat mortality at wind turbines can be separated into proximate and ultimate causes (Cryan & Barclay, 2009; Ellison, 2012). Proximate causes are those that are directly responsible for death e.g. collision, whilst ultimate causes explain why bats entered the area. Ultimate causes are further divided into three broad types: i) random collisions; ii) coincidental collisions; and, iii) collisions that arise from bats being attracted to the turbines (Cryan & Barclay, 2009).

Proximate causes

Bat deaths arise from either trauma via direct collision primarily with wind turbine blades (Horn *et al.,* 2008a) or via barotrauma (Baerwald *et al.,* 2008). The latter occurs when a bat enters low pressure vortices created in the wake of a turbine blade resulting in internal haemorrhaging following eruption of the lungs (Baerwald *et al.,* 2008). Bats appear to be able to avoid stationary meteorological masts as collisions are reported infrequently (Barclay *et al.,* 2007; Sovacool, 2009). Barotrauma was listed as the primary cause of death in 90% of bats found dead at wind energy installations (Baerwald *et al.,* 2008) but this has been challenged recently due to the complexity of the injuries often sustained (Grodsky *et al.,* 2011; Rollins *et al.,* 2012). Injuries consistent with collisions were found in 73% and 74% of bats examined by Rollins *et al.* (2012) and Grodsky *et al.* (2011), respectively. In contrast, the occurrence of injuries consistent with barotrauma, were found in only 20% and 52% of bats by Rollins *et al.* (2012) and Grodsky *et al.* (2011).

Ultimate causes

Multiple hypotheses have been proposed to explain why bats are killed by wind turbines (Barclay & Brown, 2009). It is possible that bats are visually, thermally and acoustically attracted to turbines as potential foraging sites, roost locations and/or lekking sites, or, simply, as prominent landmarks in a wider landscape (Ahlen, 2002; Dürr & Bach, 2004; Szewczak & Arnett, 2005; Kunz *et al.,* 2007a; Cohn, 2008; Sterze & Pogacnik, 2008; Cryan, 2008; Cryan, 2009 Cryan & Barclay, 2009), thereby creating population sinks or ecological traps (Cryan, 2009). There is little consensus towards one explanation over another. This is likely due to the small number of studies testing such hypotheses and the often conflicting results reported in the few studies that have been conducted as well as the interaction of multiple factors.

Reasons for mortality: insect activity attracting bats

Flying insects may be attracted to the paint or colour of wind turbine blades (Long *et al.,* 2011a) or the heat generated by nacelles (Ahlen, 2003) whilst insect numbers may accumulate around towers during migration (Rydell *et al.,* 2010b). Studies have shown that white and grey, the two colours turbines are typically painted, are the most attractive colours to insects after yellow (Long *et al.,* 2011a). Consequently, bats may be attracted to wind turbines for feeding opportunities. However, if this was a primary cause of mortality then higher levels of feeding at blade height would be expected but this is not the case (Long *et al.,* 2011a).

Reasons for mortality: habitat creation/modification attracting bats

The modification of the landscape, for example, creation of open spaces, linear features (i.e. access roads) and resulting creation of favourable foraging habitat for insectivorous bats has also been suggested as a reason for wind turbines attracting bats (Kunz *et al.,* 2007a). The installation of lowland and urban turbines and the felling of trees around turbine bases create habitat edges which may increase the numbers of foraging individuals (Limpens & Kapteyn, 1991; Grindal & Brigham, 1998; Erickson & West, 2002; Russ & Montgomery, 2002; Fiedler *et al.,* 2007). Moreover, turbine towers and nacelles may attract bats which investigate these features as potential roost locations (Horn *et al.,* 2008a). However, support for hypotheses on wind turbine attractiveness for insects and landscape attractiveness for bats has been limited. Jain (2005) and Gillespie (2013) found that bat activity did not vary between turbine and non-turbine sites. In addition, a number of studies have found that bat activity falls in the vicinity of active turbines (Minderman *et al.,* 2012).

Reasons for mortality: investigative behaviour

Curiosity and investigative behaviour may play a role in bat-turbine collisions with bats having been observed (via thermal imagery) alighting on turbines (Ahlen, 2002; Barclay *et al.,* 2007; Curry, 2009; Horn *et al.,* 2008a; 2008b). It is also possible that the ultrasonic emissions of turbines may induce exploratory behaviour (Szewczak & Arnett, 2006), but results of studies investigating turbine ultrasound have been inconclusive (Szewczak & Arnett, 2006; Nicholls & Racey, 2007; 2009). However, it is clear that bats are not attracted to turbine lights, as no differences have been found in the numbers of casualties under lit and unlit turbines (Arnett *et al.,* 2008; Baerwald, 2008).

Reasons for mortality: time of year

Although mortality of bats has been recorded throughout the year at wind energy facilities, peaks in recorded mortality at wind energy facilities occur in late summer in both North America (Cryan & Brown, 2007; Kunz *et al.,* 2007a; Baerwald & Barclay, 2007; 2008), Europe (Dürr & Bach, 2004 ; Ahlen 2003; Rydell *et al.,* 2010a) and Australia (Hull & Cawthen, 2012). This, combined with a greater occurrence of migratory species amongst fatalities, has led researchers to conclude that wind turbines are likely to have a greater negative effect on bat species that disperse and migrate during autumn (Johnson *et al.,* 2003; Dürr & Bach, 2004; Cryan & Brown, 2007; Kunz *et al.,* 2007a; Arnett *et al.,* 2008; Jain *et al.,* 2011) compared to locally resident populations (Dürr & Bach, 2004; Johnson, 2004).

Reasons for mortality: weather conditions

Risk of mortality is also likely to vary temporally as a result of weather conditions. Bat activity is highest when wind speeds are low (Rydell *et al.,* 2010a; Baerwald & Barclay, 2011; Gillespie, 2013; Minderman *et al.,* 2012; Korner-Niervergelt *et al.,* 2013; Horn *et al.,* 2008a). Consequently, low wind speeds (< 6-8m/s) are correlated with higher estimated rates of bat mortality in European and North American wind energy (Rydell *et al.,* 2010a; Arnett *et al.,* 2008; Baerwald & Barclay, 2011) with collision rates predicted to be low above 6 m/s (Korner-Kievergelt *et al.,* 2013). Bat activity is also correlated with higher ambient temperatures (Baerwald & Barclay, 2011; Gillespie 2013; Lundy *et al.,* 2013), lower barometric pressures (Horn *et al.,* 2008a; Baerwald & Barclay, 2011) and the passage of storm fronts (Arnett *et al.,* 2008).

Reasons for mortality: siting of wind energy developments

The difference in mortality rates observed between turbine developments (Dürr & Bach, 2004; Arnett *et al.,* 2008; Baerwald & Barclay, 2009; Rydell *et al.,* 2010a) and specific turbines (Piorkowski & O'connell, 2010) suggest that bat kills are non-random events (Dürr & Bach, 2004; Barclay *et al.,* 2007; Baerwald & Barclay, 2009) compounded by poorly sited turbines (Telleria, 2009b; c). Siting of turbines on bat migration routes (Baerwald & Barclay 2009; Voigt *et al.,* 2012) and in habitats with high bat density (Arnett *et al.,* 2007; Horn *et al.,* 2008a; Piorkowski & O'connell, 2010) appears to affect the mortality rates and species affected (Kunz *et al.,* 2007a; b). In north-western Europe, mortality rates in lowland (pastoral and open landscapes) wind farm sites appear to be lower than upland and coastal sites (Rydell *et al.,* 2010a). Similarly, in North America, mortality rates are higher at wind energy sites on forested ridge tops than in open agricultural areas (Kunz *et al.,* 2007a; Arnett *et al.,* 2008). Baerwald & Barclay (2009) also recorded higher bat mortality rates at wind energy sited along migration routes. Since bats often select linear or riparian landscape features for foraging (Limpens & Kapteyn, 1991; Walsh & Harris, 1996; Reynolds, 2006), mortality rates may be higher at wind energy situated near such features. However, Dürr & Bach (2004) examined the proximity of developments to wooded landscape features and found specialist edge-foragers were killed up to 700m from woodland features. However, the majority of kills (77%) were at turbines sited within 50m of trees.

Reasons for mortality: wind turbine and farm size

In contrast to birds, bat-turbine collisions are correlated with turbine size i.e. larger turbines result in higher rates of bat mortality (Dürr & Bach, 2004; Barclay *et al.,* 2007, Kunz *et al.,* 2007a). In their review of north western European studies, Rydell *et al*. (2010a) found that mortality rates increased with turbines greater than 60m in height. Similarly, in North America, mortality rates are superficially different between larger and smaller turbines (Arnett *et al.,* 2008) with mortality rates higher at wind turbines greater then 65m in height (Baerwald & Barclay, 2009). In the same studies, an increase in the area swept by rotor blades is also linked to higher mortality rates (Rydell *et al.,* 2010a; Arnett *et al.,* 2008). The emergent pattern of the installation and/or replacement of decommissioned turbines with larger, higher-rated turbines appear to increase the frequency of bat kills (Barclay *et al.,* 2006; 2007; Smallwood & Karas, 2009). In contrast to the height and rotor area of turbines there is no evidence to suggest that the size of a wind farm facility i.e. number of turbines has an impact on bat fatalities (Kunz *et al.,* 2007a; Arnett *et al.,* 2008; Rydell *et al.,* 2010a).

Species affected

Mortality of 11 (24%) of the 45 bat species found in the USA (Kunz *et al.,* 2007a), 27 (60%) of the 45 that occur in Europe (Eurobats, 2012) and 20% of the 8 species that occur in Tasmania (Hull & Cawthen, 2012), has been recorded at wind energy facilities. Three factors; long distance migration, a propensity to roost in trees and Frequency Modulated (FM) echolocation, have been suggested as factors linking the species effected (Cryan & Barclay, 2009; Horn *et al.,* 2008a). Horn *et al.* (2008a) suggested that bats whose echolocation is Frequency Modulated (FM) are at greatest risk of mortality in the vicinity of wind energy facilities due to reduction in bandwidth of returning echoes limiting the bats ability to detect obstacles in the environment (Simmons *et al.,* 2004). Migratory, tree roosting bats constitute 75% of bat species found dead in North America (Kunz *et al.,* 2007a; Arnett *et al.,* 2008) and

50% of the eight species most commonly found dead in Europe (Rydell *et al.,* 2010a); namely, Leisler's bat (*Nyctalus leislerii*), Nathusius pipistrelle (*Pipistrellus nathusii*), the Noctule bat (*Nyctalus noctula*) and part coloured bat (*Vespertillo murinus*). Similarly, there is a bias in the species recorded dead at wind energy in Australia with 84% of identified bats from one species (Hull & Cawthen, 2012). Although there seems to be a bias in the type of bat recorded dead at wind energy installations, it should be noted that non-migratory, local or short distance migratory bat species are also found dead in the USA (Grodsky *et al.,* 2012) and European (Voigt *et al.,* 2012) wind energy facilities. At present, due to the lack of knowledge regarding migratory habits of Irish bats, we are unable to determine whether Nathusius pipistrelle and Leisler's bat (species known to migrate in Europe) are at a greater risk of mortality due to migration. Table 5 lists the species that occur in Northern Ireland and the current knowledge relating to mortality at wind energy facilities.

The ineffectiveness of echolocation over distances larger than 10 – 25m (see Eurobats, 2012) combined with the reduced bandwidth of calls reflected off turbines (Long *et al.,* 2009; 2010) and the highly directional sonar emitted by bats (Surlykke *et al.,* 2009), could explain why particular bat species collide with, or are struck by wind turbine blades. In addition to the greater risk of mortality for bats that use Frequency Modulated (FM), Long *et al.* (2010) found that echolocation calls of a simulated common pipistrelle were scattered by the lateral edge of a turbine rotor, but not on the horizontal edge, which suggests this may impede the bats ability to detect turbine/blade. Also, probability of pulse reflection is lower at slower rotational velocity of turbines (4-5m/s).

Mitigating wind turbines impacts on bats

The use of ultrasound (Horn *et al.,* 2008b; Arnett *et al.,* 2013a) and electromagnetic (Nicholls & Racey, 2007; 2009) deterrents were first methods tested with variable success. Horn *et al.* (2008a) reported bat activity at one tower fitted with an ultrasonic acoustic deterrent was lower than a control tower but activity at another was not. Similarly, although the use of broadband ultrasonic broadcasts at turbines by Arnett *et al.* (2013b) resulted in a reduction of activity at some turbines, the results were highly variable. Bat activity has been shown to be lower within the vicinity of radar stations (Nicholls & Racey, 2007) and subsequent field trials of a portable radar devices with a fixed antennae resulted in lower levels of bat activity (Nichols & Racey, 2009). However, bats continued to forage within the beam of the portable radar device and insect activity was unaffected (Nicholls & Racey, 2009). Therefore, acoustic deterrents, at present, do not appear to be a reliable, effective means of deterring bats to mitigate the risk of wind turbine collision.

Table 5. Comparison of collision evidence for bat species that occur in Northern Ireland.

Species	**Tree roosting?**[1]	**Migratory?**[1]	**% NI bat population**[2]	**Evidence of mortality in UK or Ireland?**[3]	**Evidence of mortality in Europe?**[3]	**% of recorded mortalities in Europe (%)**[3]
Pipistrellus pipistrellus	YES	NO	50.3	NO[#]	YES	18.40
Pipistrellus pygmaeus	YES	NO	25.4	YES	YES	2.90
Pipistrellus nathusii	YES	YES*	0.5	NO[#]	YES	11.60
Nyctalus leislerii	YES	YES*	0.8	NO	YES	7.40
Plecotus auritus	YES	NO	1.9	NO	YES	0.09
Myotis daubentonii	YES	NO	17.9	NO	YES	0.13
Myotis nattereri	NO	NO	2.1	NO	NO	0.00
Myotis mystacinus	YES	NO	1.0	NO	YES	0.06

Sources: [1]Altrincham, J (2003). Bats, New Naturalist Series [2]Russ, J.M. (1999). The Microchiroptera of Northern Ireland: community composition, habitat associations and ultrasound. Unpublished PhD thesis. Queen's University, Belfast. [3]Anon. 2013. Report of the IWG on Wind Turbines and Bat Populations. 17th meeting of the Eurobats Advisory Committee, Dublin, Ireland. *No evidence of migration in Ireland. [#]Not recorded for this species but bats identified to genus found.

A reduction in the 'cut-in' speeds (the lowest speed that wind turbines generate power) appears to reduce mortality. Arnett *et al.* (2010) reported a reduction in mortality rates of 44 - 93% when cut-in speeds were raised from 3 - 4 m/s (typical cut-in speed) to 5.5 - 6.5m/s. Table 6 details the results of curtailment experiments from North America. Increased cut-in speeds have also been implemented as mitigation in Germany and Portugal but, at present, there is no readily accessible information on the outcome (Eurobats, 2013). In Canada, if mortality is estimated to be above a certain threshold (10 bats per turbine per year) the cut-in speed of the development will be changed to 5.5m/s or blades will be feathered below this speed to reduce the chances of bat mortality (Eurobats, 2013). These conditions will be enforced at all turbines within a site between the months of July and September between the hours of sunset and sunrise.

In France, Lagrange *et al.* (2013) have developed a system called Chirotech that regulates wind turbines according to bat activity with minimum loss of productivity.

This system has been tested at 39 sites in France since 2006 and a further 2 sites in Canada in 2012 (Lagrange *et al.,* 2013). Chirotech monitors bat activity at the hub-height of a turbine and models bat behaviour based on time, wind speed at site, season, temperature at site and height of activity. The trials in Canada have led to a reduction in bat mortality of 78% at turbines with the system installed when compared to control turbines within the site (Lagrange *et al.,* 2013). In Germany, a similar system has been developed that uses an algorithm to stop wind turbines when temperature, wind speed and season are associated with high levels of bat activity (Behr *et al.,* 2011). In Poland, between 2009 and 2010, 42 planned wind turbine sites were surveyed for bats and activity was quantified at each site (Kepel *et al.,* 2011). This work resulted in the production of a scale of bat activity (low, medium, high) based on the number of bat contacts per hour (Table 7). Values differ between individual bat genera and Table 7 lists the results for genera found in Ireland. Turbine feathering has also been reported to reduce bat mortalities (see Arnett *et al.,* 2013b for a synthesis of unpublished work).

Table 6. Comparison of cut-in speed mitigation experiments conducted in North America to reduce bat mortality at wind energy facilities. Data from Arnett *et al.* (2013).

Location	Control		Experimental		Reduction in mortality (%) per turbine
	Cut-in speed (m/s)	Bat deaths per turbine	Cut-in speed (m/s)	Bat deaths per turbine	
Ontario, Canada	4.0	5.2	4.5	2.7	60
Maryland, USA	4.0	28.7	5.0	10.9	62
Alberta, Canada	4.0	19.0	5.5	7.6	60
Vermont, USA	4.0	2.7	6.0	1.0	60
Indiana, USA	3.5	14.0	6.5	3.0	79

Table 7. Results of a Polish study (Kepel *et al.,* 2011) attempting to quantify bat activity at potential wind farm sites. Numbers relate to the number of bat contacts per hour.

Bat activity	*Nyctalus* species	*Pipistrellus* species	All bat species
Low	2.5	2.5	3
Medium	4.3	4.1	6
High	8.6	8.0	12

Current research in the UK and Ireland

At present, there is one study in Ireland that is investigating the effects of wind energy on bats. This is a PhD at University College Dublin that is being conducted by

Una Nealon and is due for completion in 2015. In 2008, DEFRA funded the University of Bristol in conjunction with the Bat Conservation Trust (BCT) to review the known impacts of wind energy on bats and provide recommendations on research protocols (Jones *et al.,* 2009). The second phase of this project is currently being undertaken by Dr Fiona Matthews at the University of Exeter to research the impacts of wind turbines on bat populations in Great Britain. This project is due to finish in 2014. Dr Matthews is also undertaking research, funded by NERC, to determine the impacts that mid-sized wind turbines (50-300kW and 25-50m in height) have on bat populations. This research is due to be completed in 2015. In 2008, Dr Kirsty Park of the University of Stirling was funded by the Leverhulme Trust for 2 years to undertake research on the impacts of microturbines (<50kW) on bats and birds. This project resulted in publications in *PLoS ONE* (Minderman *et al.,* 2012) and the *Journal of Applied Ecology* (Park *et al.,* 2013) and is currently funded (in 2013) by the People's Trust for Endangered Species (PTES) UK mammals grants. A conference on "Wind energy and Wildlife Impacts" has been held twice (2011 and 2013) and both have hosted multiple presentations on impacts of wind energy on bats.

3.3 Cumulative Impacts of wind turbines on birds and bats

Assessing the cumulative impacts of developments as part of an Environmental Impact Statement is a requirement under the Planning (Environmental Impact Assessment) Regulations (NI) 1999 (as amended). However, a review of Environmental Impact Statements in 2002 found that only 48% had mentioned cumulative impacts or effects (Cooper & Sheate, 2002). At the time, this was largely attributed to a lack of definition and guidance regarding the contents and context of such an assessment (Cooper & Sheate, 2002; Masden, 2009). Since then, reports by the Department for Business, Enterprise and Regulatory Reform (Anon., 2008a), Masden *et al.* (2009b) and Scottish Natural Heritage (Anon., 2012) have reviewed the assessment procedure and attempted to clarify the requirements of a cumulative impact assessment. Consequently, SNH have defined cumulative impacts as "additional changes caused by a proposed development in conjunction with other similar developments or as the combined effect of a set of developments, taken together" (Anon., 2012b). Traditionally, the receptor in cumulative impact studies of wind energy has been the landscape (e.g. Anon., 2008b).

EIA have focused on assessing the landscape and visual amenity of wind farm developments and how they affect the character (physical attributes of the land e.g. landform, land cover and settlement pattern) of a landscape (e.g. Anon., 2009). This typically involves taking into consideration the distance between wind energy facilities, the distance they are visible, the overall character of the landscape and its sensitivity to wind energy, the siting and the design of the wind energy and the way in which the landscape is experienced (Anon., 2010). To date, cumulative impact assessments have largely ignored the impacts of wind energy on wildlife receptors (Masden, 2010). However, a small number of studies have begun to address this issue for birds of prey and bats (e.g. Pearce-Higgins *et al.,* 2008; Masden, 2010; Santos *et al.,* 2010; Schaub, 2012; Bellebaum *et al.*, 2013; Roscioni *et al.,* 2013). Although wind farm mortality may be low in comparison to other causes of death (Erickson *et al.,* 2001; Drewitt & Langston, 2008; Willis *et al.,* 2010) these species groups have been the focus of research as they are long lived, have low productivity, slow maturation rates, are often species of conservation concern and additional adult mortality can further inhibit population growth (Whitfield *et al.,* 2004). Consequently, their populations are at risk from the additional mortality that may be caused by wind energy developments (Drewitt & Langston, 2006; Bright *et al.,* 2008; Carette *et al.,* 2009; Martinez *et al.,* 2010).

A number of approaches have been used to investigate cumulative effects on birds and bats. These entail identifying areas and species at risk or determining what the cumulative effects of wind energy developments may be on a population. The former approach simply infers risk from the degree of overlap between the distribution of a species or its migration routes and the distribution of existing or proposed wind farm developments (Pearce-Higgins *et al.,* 2008; Bright *et al.,* 2008; Telleria, 2009a; Martinez *et al.,* 2010; Liechti *et al.,* 2013; Roscioni *et al.,* 2013). The second approach has used more complicated models (e.g. population viability analysis and collision risk models) to predict the impacts of wind energy development associated mortality on population survival (Carette *et al.,* 2009; Masden, 2010; Schaub, 2012).

Due to the high levels of wind farm development in Scotland, Species Distribution Models (SDMs) have been used to produce sensitivity maps for species of conservation concern (Bright *et al.,* 2008; Pearce-Higgins *et al.,* 2008). Comparing the distribution of 16 priority species in Scotland with the location of existing and proposed wind energy developments identified two species (red kites and hen harriers) at high risk due to the level of overlap (Bright *et al.,* 2008). Similarly, Roscioni *et al*. (2013) identified foraging areas of bat communities in Italy using SDMs. They found a high risk to bat communities in Italy as over 50% of current and planned turbines were in areas suitable for foraging Leisler's (*Nyctalus leislerii*) and Common pipistrelles (*Pippistrellus pipistrellus*) e.g. species at risk of collision from wind turbines. In Spain, similar SDMs have shown that although there is low spatial overlap between proposed wind energy sites and the nesting sites of golden (*Aquila chrysaetos*) and Bonelli's (*Aquila fasciata*) eagles, the golden eagle in Almeria, Murcia and Alicante is at greater risk from cumulative impacts due to the degree of overlap with their territories (Martinez *et al.,* 2010).

The use of more complicated models to determine the cumulative effects of wind energy on the populations of species has been restricted to birds of prey and swans. Although using different approaches, all have shown that higher numbers of wind energy facilities or related mortalities have the potential to impact the growth rate of populations (Carette *et al.,* 2009; Masden, 2010; Schaub, 2012). Using a theoretical model, Schaub (2012) showed that increased numbers of single wind turbines, distributed widely over the landscape resulted in a population decline in red kites (*Milvus milvus*) whilst a greater aggregation of turbines (as farms) resulted in fewer casualties. Similar results were demonstrated by Eichhorn *et al.* (2012) and further work on red kites by Bellebaum *et al.* (2013) demonstrated that mortality predicted by cumulative models has the potential to affect long-term survival of red kite populations. Trinder (2012) used a similar approach to Schaub (2012) to assess the impact of mortality, including that caused by wind turbines, on the population growth of the Scottish population of whooper swan (*Cygnus cygnus*). He found that if mortality equated to 4% of the population then the population declined. Similarly, Masden (2010) showed that hen harrier populations on Orkney would decline over the next 50 years based on the location of current and future wind energy. However, unlike Shaub's (2010) model, Masden (2010) looked at factors that might reduce a population decline. She showed that locating turbines away from nest sites caused the magnitude of the effect of wind energy to vary and that considering the ecology of a species may mitigate effects totally (Masden, 2010). Using population viability analysis models, Carette *et al.* (2009) demonstrated that Egyptian vultures (*Neophron percnopterus*) that had territories fewer than 15km from wind energy had an increased probability of mortality. This study also shows that the risk varied within Spain and that mortality associated with turbines had the potential to increase the chances of extinction of some Spanish sub-populations (Carette *et al.,* 2009).

Many bat populations across the world are already declining (Mickleburgh *et al.,* 2009; Ingersoll *et al.,* 2013). Therefore, concern over the frequency and widespread occurrence of bat mortality at wind energy has led to speculation to what the long-term impacts are on species that have a relatively low reproductive rate. Studies from Sweden suggest that bat populations there will decline by as much as 30-50% by 2020 if the rate of wind turbine development continues at its current pace (Hedenstrom & Rydell, 2011; 2013). A regional estimate of annual bat mortality rate in a 204,609km^2 region of the US (the Mid-Atlantic highlands) was estimated to reach 111,000 by 2020 (Kunz *et al.,* 2007a). Similarly, Arnett & Baerwald (2013) (reported in Arnett *et al.,* 2013b) estimate that 650,000 to 1.3 million bats were killed by wind turbines in North America between 2001 and 2011. However, a recent study suggests that mortality in the USA could be higher with over 600,000 bat fatalities in 2012 (Hayes, 2013). The lack of research into the impacts of bat mortality rates at wind energy and their effects on bat populations is likely a result of the difficulty in estimating bat populations and the number of assumptions that have to be made regarding fatality rates and projected wind farm capacity. However, it is highly likely that an increase in wind energy developments will continue to lead to losses of bats which in a species that has a low reproductive rate could have potential consequences for populations of the species at greatest risk.

Therefore, predictive and theoretical tools including spatial and/or constraint mapping (Osborn *et al.,* 1996b; Fielding *et al.,* 2006; Bright *et al.,* 2008; Tapia, 2009; Telleria, 2009a; b), modelling cumulative effects of regional or national developments on populations, particularly of endangered or rare species (Kerlinger, 2003; Masden *et al.,* 2009b; Pearce-Higgins *et al.,* 2009), collision risk modelling (e.g. Chamberlain *et al.,* 2005; 2006; Madders & Whitfield, 2006), and population modelling using theoretical or empirically derived measures of mortality (Dillingham & Fletcher, 2008; Bekessy *et al.,* 2009; Carrete *et al.,* 2009), are important tools in planning the location and effects of developments on biodiversity to avoid undue conflict with avian and bat interests. However, species should be prioritised in accordance with their conservation importance or vulnerability and mortality from other sources (Desholm, 2009), since additive mortality from windfarms can critically alter demographics and drive population declines (Smallwood & Neher, 2004; Carrete *et al.,* 2009).

3.4 Terrestrial mammals

Excluding bats, there were few (*n* = 6) published studies on the effects of wind energy developments on other, terrestrial mammals. Terrestrial mammals obviously are not subject to direct mortality due to turbine blade strikes. Most effects are as a result of associated development causing habitat fragmentation and deterioration which are the principal threats to ground-dwelling, semi-fossorial and fossorial species (Walter *et al.,* 2006; Mouton *et al.,* 2007). However, noise pollution may also affect some species. Three of the six studies demonstrated no effect of wind turbines (and in one case their construction) on ungulate ranging behaviour, diet or vigilance or small mammal abundance. A number of authors suggest that disturbance is unlikely to cause major problems for highly mobile mammals (Linnell *et al.,* 2000; Sauvajot *et al.,* 2004). Only one study suggests that terrestrial mammals were displaced by wind energy developments and moved to alternative habitats (Walter *et al.,* 2006). None of these species occur in Northern Ireland.

Red deer (*Cervus elaphus*) have been shown to be unaffected by wind energy development (pre-construction versus post-construction) by examining home range size and foraging behaviour preferences (Walter *et al.,* 2006). However, home range centres did shift away from turbines (± 700m), possibly due to limited loss of habitat or direct avoidance of turbines. Hablinger (2004) cited in Kusstatscher *et al.* (2005), also suggested that ungulate movement along habitat corridors may be disrupted by avoidance of turbine structures within 150m. However, such studies are confounded by seasonality and extrinsic factors (precipitation, temperature and the selection of agricultural crops) making the quantification of avoidance difficult. Whilst they found no significant effect on large ungulates, Walter *et al.* (2006) suggested that the identification of key resources and important areas for deer, for example foraging or calving sites, is necessary during pre-construction surveys. In an experimental study of semi-domestic reindeer behaviour, foraging was found to be unaffected by the presence of rotating wind turbines in comparison to areas without turbines, but further studies are required (Flydal *et al.,* 2004).

Only one study demonstrated a significant effect of acoustic noise from turbine blades on a species of small mammal; the Californian ground squirrel (*Spermophilus beecheyi*) increasing its vigilance and anti-predator behaviour (Rabin *et al.,* 2006; Kikuchi, 2008). Turbine noise may have masked communication calls and may even lead to auditory impairment (Rabin *et al.,* 2006). Nevertheless, there was no apparent effect on species abundance but the authors suggest that negative effects on anti-predator behaviour may have longer term effects. Moreover, reduction in vigilance may attract predators (for example, golden eagles predate ground squirrels) which may themselves be killed by rotating turbine blades (Hoover & Morrison, 2005; Smallwood *et al.,* 2007). Other species dependent on burrows of ground squirrels may also be impacted (Rabin *et al.,* 2006; Smallwood & Thelander, 2004; Smallwood *et al.,* 2009).

In a study of vertebrate community structure, Santos *et al.* (2010) examined 18 mammal species (although individual species results are not presented) and concluded that overall species richness was impoverished in close proximity to wind energy facilities. De Lucas *et al.* (2005) in an impact gradient (IG) study found no effect of wind energy facilities on the density and abundance of small mammal species, but this study was confounded by small mammal population fluctuations over time and the results may not be transferable to other regions or developments.

Grey literature reports indicate “slight or no significant disturbance” of small mammal species or locally habituated mammal species, for example the red fox (*Vulpes vulpes*), European hare (*Lepus europaeus*) and roe deer (*Capreolus capreolus*) in close proximity to turbines (Kusstatscher *et al.,* 2005). The Irish hare (*Lepus timidius* *hibernicus*) is known to tolerate highly active human environments, such as airports and golf courses, and has been observed to behaviour normally in the middle of wind farm developments in Northern Ireland (Neil Reid, *pers. obs.*). Conversely, other reports indicate that some small mammal populations, particularly fossorial species including prairie dogs, cottontail rabbit and prairie hare may increase due to habitat perturbation during construction activity, whilst others, for example pronghorn and ground squirrel, remain unaffected up to 800m from turbines (Johnson *et al.,* 2000; Hötker *et al.,* 2006).

There were numerous studies on the effects of human sensitivity to windfarm noise (e.g. Elthem *et al.,* 2008; Harding *et al.,* 2008; Pedersen *et al.,* 2009) resulting in national regulation including noise thresholds or minimum setback distances ranging from 350m to 2km to minimise "annoyance". Setback distances have also been applied to wildlife protection and conservation (e.g. Blumstein *et al.,* 2005; Whitfield *et al.,* 2008) and it is conceivable that noise intolerable to humans will be similarly intolerable to wildlife. Consequently, mitigation prescriptions can be used to protect wildlife from anthropogenic disturbance including wind energy development developments (Ruddock & Whitfield, 2007).

3.5 Other vertebrates

Santos *et al.* (2010) conducted an assessment of the effect of wind energy developments on vertebrate biodiversity in general, including, birds, mammals and herpetofauna with measures of species richness. Their conclusion suggests an overall negative impact of wind energy. There are no published studies of the effects of wind energy developments on herpetofauna, but as with other vertebrate species, the direct loss of habitats or, specifically, hibernaculae, may affect species occurrence.

Moreover, there are no published studies on the effect on aquatic ecosystems or species although there is unpublished evidence of fish mortality during wind energy development construction, but these incidences are usually connected with the failure and slippage of construction materials (i.e. over-burden) or peat slippage

(Lindsay & Bragg, 2005) rather than direct effect of turbines or operation *per se*. Sedimentation of rivers or lakes can detrimentally affect adult fish, eggs and larvae by inhibiting growth, development and movement or migration and also alter food resources (Birtwell, 1999). Aquatic invertebrates are also affected by sedimentation, notably filter feeders, and lead to severe population declines or local extinctions (see review in Newcombe & MacDonald, 1991).

3.6 Invertebrates

A minority of studies (*n* = 4) examined the impacts of wind turbines on invertebrates and were usually incidental to bat research (Horn *et al.,* 2008a). A number of these report multi-species mortality through direct collision (Corten & Veldkamp, 2001; Shankar, 2001). However, these studies were conducted by engineers to optimise wind turbine aerodynamic performance and no assessments of species-specific impacts were given. Insect fouling and debris attached to turbine blades may reduce turbine power output (8 – 55%) due to decreased aerodynamic performance (Corten & Veldkamp 2001; Dalili *et al.,* 2009). This may result in increased investment in anti-foulant application and cleaning of blades (Shankar, 2001; Dalili *et al.,* 2009).

Insect congregations are usually ephemeral and weather related. Insects may be attracted to turbines with warning (aviation) lights (Frost, 1958; Horn *et al.,* 2008a). Insect occurrence at turbines can attract insectivorous bat species (Arnett *et al.,* 2005; Horn *et al.,* 2008; Reimer *et al.,* 2008), and presumably birds, which may increase mortality of those groups. This mortality cascade may be amplified by the habitat in which wind energy developments are sited, particularly within forested areas, or clear-felled turbine sites which can increase insects occurrence and thereby increase insectivore occurrence (Horn *et al.,* 2008a). Since the use of thermal imagery at wind energy developments has revealed considerable insect activity around turbines further investigations on temporal and spatial trends of insect occurrence at wind energy developments are important to understand the effects on both invertebrates and their predators.

The number of publications on direct or indirect effects of wind energy developments on invertebrates is small but the effects of any such development can result in the loss and/or fragmentation of important habitats. This may displace species where particular nest sites or foraging habitats or food plants are destroyed (for example, the loss of larval food plants for the marsh fritillary butterfly; Nelson 2001). The grey literature included reference to two unpublished studies which concluded that insect-turbine collisions were “negligible” after assessment of dead insect on wind turbine blades and experimental release of honeybees and blowflies. However, it was unclear exactly how and what was assessed (see Kusstatscher *et al.,* 2005).

3.7 Effects on microclimate

Wind turbines are known to create an area of reduced wind speed and increased turbulence in their shadow, an area that is typically referred to as the ‘wake’ (Rajewski, 2013). The wake can persist up to 15 rotor diameters down-wind of a turbine (Meyers & Meneveau, 2012) but the effects on wind speed and turbulence within the wake typically decrease with increasing distance from the turbine (Magnusson & Smedman, 1999). Factors that affect the characteristics of the wake (e.g. wind speed and turbulence) include; hub height, atmospheric stability, wind speed, turbine design and wind farm size and layout (Rajewski, 2013).

Mathematical modelling has also been used to demonstrate that changes in temperature and humidity (Baida Roy *et al.,* 2004) and heat fluxes (Baida Roy, 2011) occur within the wake of wind turbines. These and other studies have shown that near surface air temperatures are higher during the night and lower during the daytime within the wake compared to areas upwind of wind energy installations (Baida Roy *et al.,* 2004; Baida Roy & Traiteur, 2010). Theoretically, heat fluxes should increase in the wake of wind energy up to 20km down-wind of a wind farm (Baida Roy, 2011). Field observations (Baida Roy & Traiteur, 2010; Rajewski, 2013) and analysis of satellite imagery (Zhou *et al.,* 2012; Walsh-Thomas *et al.,* 2012) have verified some of the temperature, humidity and heat transfer changes predicted by the mathematical modelling. Although studies observing temperature changes concur on the effects observed within the wake of the turbine, the reported temperature differences vary. Baida-Roy & Traiteur (2010) reported differences of less than 1°C between upwind and down-wind wind farm edge air temperatures at a wind farm in California, whilst Rajewski (2013) recorded a difference of less than 0.5°C at a single wind turbine in Iowa. Zhou *et al.* (2012) reported that a wind farm in Texas was 0.5 – 0.72 ° C warmer over a decade than surrounding areas. In contrast, Walsh-Thomas *et al*. (2012) reported temperature differences of 4 to 8 °C in upwind and down-wind air temperatures. Similarly, the distance over which heat transfer occurs is variable; Walsh-Thomas *et al.* (2012) reported differences up to 22km (the extent of their sampling) whilst Rajewski (2013) reported transfer less than 10 rotor blade lengths down-wind (< 800m).

3.8 Effects of siting wind turbines on agricultural land and productivity

The practice of locating wind energy on farmland in the USA, combined with the observed changes in temperature, humidity and heat transfer in the wake of turbines, has prompted a limited number of investigations into the impacts this may have on agriculture. The Crop Wind-energy Experiment (CWEX) is the only study, to date that has reported the effects an altered microclimate within the wake of wind turbines has on processes that affect crop productivity and growth e.g. fluxes of heat, moisture and CO_2 (Rajewski, 2013). In this study, no biophysical measurements of the crops were made and corn yield (the crop within which the study wind farm was located) within the study site was comparable to that beyond the site boundaries (Rajewski, 2013). Therefore, no inferences were drawn on the influence of wind turbines on the productivity of crops. However, the authors did suggest that the increased downward flux of CO_2 caused by turbines during the day may increase CO_2 uptake by plants but that any positives this may bring to crop yield would be counteracted at night by increased respiration rates caused by the upward flux of CO_2 (Rajewski, 2013). Furthermore, the stability of the atmosphere (a factor that changes with time of day and growing season) and turbine arrangement (staggered or aligned) within a wind farm determines whether a turbines wake interacts with the surface and at what distance (Rhodes *et al.,* 2010; Rajewski, 2013; Zhang *et al.,* 2013). Any potential benefit derived from the siting of wind energy on agricultural land is variable in both time and space.

3.9 Impacts of siting wind turbines on peat bogs

Blanket and raised bogs are currently listed in Annex 1 of the EC Habitats Directive and are subject to a Northern Irish Habitat Action Plan (Anon., 2003). Peatland accounts for 14% of the land area of Northern Ireland and as in other parts of the UK Northern Irish peatlands contain many current, and proposed, wind farm developments (Lilly *et al.,* 2009; NIEA, 2010). This is a result of their low agricultural productivity and their occurrence on exposed sites that have large wind resources that will provide higher financial returns than other locations (Lilly *et al.,* 2009). The non-market value of the ecosystem services provided by peatlands (e.g. water storage, filtration and flood mitigation, carbon sequestration and biological process like pollination) is high (Bain *et al.,* 2011). In addition, Northern Irish soils contain 28% and 6% of Ireland's and the UK's total soil carbon stocks respectively (Bradley *et al.,* 2005; Eaton *et al.,* 2008) and, therefore, have an important role in regulating the climate by acting as a carbon sink (Bain et al., 2011). However, when peatlands are disturbed by human activities e.g. peat harvesting, drainage for agriculture and forestry, they can become a carbon source as their physical and hydrological conditions change (Holden *et al.,* 2004; Van Seters & Price, 2001; Lane & Milledge, 2013). Consequently, siting a renewable resource on an important carbon sink may lead to limited green-house gas savings (although this is contested in a forthcoming publication by Smith *et al.,* 2014), poorer water quality and the loss of natural flood defences (Nyak *et al.,* 2010; Bain *et al.,* 2011).

Disturbance during the construction of wind turbines (and associated infrastructure) arises in a number of ways that arise from; forestry operations (to improve airflow), road building, creation of hard-standings and the insertion of drains in the peatland (Murray, 2012). Two types of drainage ditches, temporary and permanent, are used during and post wind farm construction (NE, 2010). Temporary ditches are used during construction to deal with surface runoff arising from earthworks and to prevent uplift of turbine foundations, whilst permanent ditches are installed to deal with runoff from access roads (NE, 2010). There is no specific research on the impacts of drainage at wind energy facilities on peatland. However, inferences can be drawn from research on the effects of drains on peatland converted for other land uses e.g. forestry and agriculture.

Historically, drainage ditches were used to lower the water table in an attempt to make peatland more suitable for farming or forestry (Armstrong *et al.,* 2009). The water table is naturally high on peatland and lowering it, in the first instance, changes the ecology of a peatland by simplifying the micro-topography and species composition of the bog (Lindsay, 2010). A very small change in water table can result in substantial alterations to the bog moss communities (Lindsay, 2010). Reducing the water table also exposes peat to more aerobic conditions (altering the microbial community) which increases decomposition and mineralisation rates (Holden *et al.,* 2007; Anderson *et al.,* 2013). Increased mineralisation from the underlying mineral material (not the peat itself) would result in the leaching of nutrients (calcium, magnesium, manganese, aluminium and potassium) and increases in suspended sediment (from erosion), ammonium and Dissolved Organic Carbon (DOC) in water ways (Leeks & Roberts, 1987; Duxbery & Peverly, 1978; Holden *et al.,* 2007; Worral *et al.,* 2007). The hydrological response of peatland also changes with the lowering of the water table as water pathways change. In undisturbed peatland, most water movement occurs in the upper acrotelm layer (the living plant layer) and is controlled by the amount of water held there (NE, 2010). However, lowering the water table can result in the settlement of peat (drying) and a reduction in its porosity (Holden *et al.,* 2007). This leads to a reduction in the storage capacity of the peat and faster discharge of water as it and the acrotelm layers ability to store water will be reduced. Ditches create more sub-surface storage but they also provide a rapid conduit for run-off (Holden *et al.,* 2006; Ballard *et al.,* 2012) which results in changes in the volume of runoff (Leeks & Roberts, 1987) and the frequency of flooding peaks (Holden *et al.,* 2004; Ballard *et al.,* 2012). However, the magnitude of change is variable and is linked to factors that include; density of ditches, the soil properties of the specific peat and the slope of the site (Holden *et al.,* 2004; Lane & Milledge, 2013). In addition, drainage can increase the occurrence and or efficiency of naturally occurring soil pipes (tubes within the peat that can transport water) which can further increase sedimentation and runoff (Holden *et al.,* 2007).

Another potential consequence of drainage ditches at wind energy facilities on blanket bogs is the risk of peat slides. These have been recorded at a number of wind energy facilities in Ireland and Britain (e.g. Derrybrien, Co. Galway and Garvagh Glebe, Co. Leitrim) and the construction of drainage ditches associated with roads have been linked as a secondary causal factor (Lindsay & Bragg, 2005 ; Long *et al.,* 2011b). In their paper classifying peat movements, Dykes & Warburton (2007) define a peat slide as "failure of blanket bog involving sliding of intact peat on a shearing surface at the interface between the peat and the mineral substrate material or immediately adjacent to the underlying substrate". Although peat slides have been recorded for at least 150 years across Britain and Ireland, the exact mechanisms of what cause a peat slide are poorly understood (Warburton *et al.,* 2004; Dykes & Jennings, 2011; Long *et al.,* 2011b). Peat slides can occur naturally following periods of heavy rainfall but they are also associated with secondary factors such as human disturbance e.g. peat cutting or road building (Dykes & Warburton, 2008; Dykes & Jennings, 2011; Long *et al.,* 2011b). Anthropogenic factors that can trigger peat slides include; alteration of drainage patterns, cutting of peat at the toe of a slope, loading of peat mass with heavy plant machinery, digging or tipping, and changes in vegetation cover.(Anon., 2006). In the first instance, peat failure can affect the ecology of the failure site (Dykes & Warburton, 2007). However, if it reaches water courses, as occurred at Derrybrien, Co. Galway, substantial fish-kills and loss of other aquatic life can occur (Lindsay & Bragg, 2005; Dykes & Jennings, 2011). Peat failures further disrupt bog hydrology, lead to deep erosion of the peat and consequent hydrological disruption and, thus, biodiversity loss and or change (Anon., 2006).

The felling of forest trees within the vicinity of a wind energy development can also be undertaken to improve air flow (Murray, 2012). Although no direct studies have investigated the effects of this during wind farm construction on peatland habitats, inferences can be made from what is known about felling associated with commercial forestry on peatland. Clear felling operations at upland sites have been shown to have the greatest impact on fresh water streams fed by peatland. A range of impacts are recorded that include; increased run off, mobilization of fine sediment and increased nutrient leaching (Leeks & Roberts, 1987; Roberts & Crae, 1997; Muller & Tankere-Muller, 2012). Although nutrient levels in streams fluctuate seasonally, concentrations of nitrates (NO_3), DOC, iron, phosphate, potassium and aluminium, increase in streams following felling as does acidity e.g. lower pH (Leeks & Roberts, 1987; Neal *et al.,* 2004 a,b; Tetzlaff *et al.,* 2007; Muller *et al.,* 2012). However, the occurrence and concentration of nutrients in run-off depends on the composition of the parent materials of the peatland. In commercial forestry, increased levels of nutrients typically reverse after 2-4 years and nutrient levels return to, or near, pre-harvesting levels as a result of vegetation regrowth on felled sites (Neal *et al.,* 1992; Neal *et al.,* 2004a). These impacts have been observed on a local scale (streams) and it is largely unknown what effect, if any, these changes have on wider catchments (Neal *et al.,* 2004a). Increased DOC levels can result in drinking water not meeting European Water Framework Directive (WFD) requirements and increased costs associated with its removal (Bain *et al.,* 2011). Additional infilling of reservoirs by sediment can also add to the cost of water treatment and the covering of gravel bed spawning grounds, particularly of salmonid fish (Holden *et al.,* 2007).

A limited number of studies have monitored stream water chemistry and water quality to see if similar changes occur during wind farm construction. These have shown that the disturbance of shallow soils on peatlands facilitates the export of organic carbon and phosphate (P) into streams fed by peatland and can increase concentration of DOC, particulate organic carbon (POC) and phosphorus (Grieve & Gilvear, 2008; Waldron *et al.,* 2009; Murray, 2012). As a consequence, wind farm construction activities, like other land use conversion of peatland, can have an indirect effect on water quality levels and nutrient levels. However, Murray (2012) suggests that, as the felling of forestry and the practice of brash mulching is associated with increased DOC and phosphorus concentrations in stream water (Neal *et al.,* 2003; Rodgers *et al.,* 2010),this factor alone, rather than the disturbance of soils associated with the construction of the actual turbines, explains changes in stream hydrology.

In addition to impacts on stream water chemistry and water quality, roads on peatland can impact the hydrological function and structural integrity of peat (Lyndsey, 2010; Grace *et al.,* 2013). The type of road constructed at wind energy developments depends on the depth of peat, with deeper peats having floating roads and shallow peats excavated roads (Lyndsey, 2010). Floating roads or tracks can alter the structural integrity of peat by compressing the peat and reducing hydraulic conductivity. This alters drainage by increasing surface runoff and reducing or slowing sub-surface flow (Lindsay, 2010; Grace *et al.,* 2013). The fill material of shallow roads can act as a dam or drain depending on their conductivity compared to the surrounding peat (Lindsay, 2010). In addition, the construction of drainage ditches associated with excavated roads can further lead to the settlement of peat and instability (Grace *et al.,* 2013). Overall, Grace *et al.* (2013) suggest that the damage to peat from roads depends on the type of peat, weight of vehicles using the road, the number of movements on the road and type or track or tread used.

Although the release of carbon via streamwater contributes to the carbon cycle, respiration is the largest carbon flux between peatland and the atmosphere (Limpens *et al.,* 2008). The rate of respiration within peatland is regulated by temperature and water table depth which has prompted a small amount of research into the effects of microclimatic changes, caused by wind energy on carbon cycling within peatland (Richardson *et al.,* 2013; Rea *et al.,* 2013; Armstrong *et al.,* 2012; 2013). These studies have found that fluxes of CO_2 and $CH_{4,}$ the two major greenhouse gases, are affected by a variety of factors within wind energy developments located on peatland. These include; temperature, water table depth, plant functional type (graminids, heathers), time of year (season), and location within wind farm sites (Richardson *et al.,* 2013; Rea *et al.,* 2013; Armstrong *et al.,* 2012; 2013).

In addition, these studies have shown that microbial biomass and soil carbon, two variables that are affected by temperature, increase along a hypothetical, microclimatic gradient within a wind farm located on peatland within Scotland (Richardson *et al.,* 2013; Rea *et al.,* 2013). Furthermore, mesocosm experiments using peat cores taken from within a wind farm located on peatland show that temperature increases of 4°C were found to increase CO_2 fluxes in conjunction with lower water tables (Richardson *et al.,* 2013).

If inferences can be made from existing studies on the impacts of land use change on peatland, the construction of wind energy on peatland could have negative consequences. Peatland plant communities, water pathways and nutrient dynamics can change where trees are felled and drainage ditches inserted. However, the magnitude of change to this environment and the processes it is responsible for will depend on the level of disturbance and the site that the wind farm is constructed.

3.10 Restoration of damaged peatlands

Restoration is often required to return disturbed peatland to its previous ecological and hydrological function. There are many examples of attempts to restore raised bogs in the UK (see UK peatland restoration and management compendium http://www.peatlands.org.uk/) but there are few examples in the published scientific literature (Anon., 2008c). In contrast, there are many examples of restoration programmes on lowland, cutover or former, forested sites in North America and Europe (e.g. Lamers *et al.,* 2002; Anderson *et al.,* 2013). Therefore, examples of blanket bog restoration are few and assumptions will undoubtedly have to be made with little empirical data in approaching restoration. Those guides that exist for bog restoration all state that prior to any restoration work, clear objectives should be established to determine the final goal of such work (Quinty & Rochefort, 2003). Restoration projects rarely take place with full knowledge of previous site-specific ecological function or conservation state. Therefore, approximations of previous conditions have to be made.

Natural colonisation of severely damaged bog sites (particularly cutover and over-grazed sites) by peat forming plant species e.g. *Sphagnum* mosses can take decades or may not happen at all as recolonisation of peat forming species can be poor (Money, 1995; Campeau & Rochefort 1996; Price *et al.,* 2003). Therefore, restoration of upland bogs typically begins with attempts to raise the water table to levels that support peat formation. This is primarily attempted by reducing the amount of rainwater lost from a site by partially blocking drains by creating dams and/or installing bunds (Charman, 2002; Anon., 2008). Damming has been done using a variety of materials that include; peat, heather bales, plastic piling, corrugated perspex, plywood, wooden planks, straw bales, stones or a combination (Armstrong *et al.,* 2009; 2010). Plastic piling is regarded as the most effective and reliable damming material (along with plywood and heather bales) but peat is most commonly used due to on-site availability, cost, aesthetics and land manager preferences (Armstrong *et al.,* 2009; Ballard *et al.,* 2012). Straw bales are generally no longer used as they leach nutrients into streams (Armstrong *et al.,* 2010). Different factors determine what materials should be used to block dams: access, number of people, distance to site, slope, wetness of peat, extent of exposure of mineral substrate and width of ditch (Armstrong *et al.,* 2009). More dams make rewetting more uniform across a site and will increase the area suitable for recolonisation by peat forming species (Van Seters & Price, 2001). However, this can also lead to a loss of habitat heterogeneity across the site, a factor often associated with peatlands and seen as important for maintaining a high level of *β*-biodiversity (Verberk *et al.,* 2010). More dams also reduce the pressure on existing dams and reduce the chances of dam failure (Ketcheson & Price 2011). However, the wetness of the peat, slope of the site, and drain geometry and number, ultimately determine the effectiveness of dams (Armstrong *et al.,* 2009; Ballard *et al.,* 2012). Damming of drains will also lead to the blocking of drains by recolonisation of plants due to the build-up of sediment (Charman 2002).

Once management actions have been implemented in an attempt to restore the natural water table, it may take anywhere from 1 to 20 years (Lunt *et al.,* 2010) for the water table to rise and even longer, 20 to 50 years, for hydrological function to return (Lunt *et al.,* 2010; Wilson *et al.,* 2010; Bain *et al.,* 2011; Ballard *et al.,* 2012). Few long-term monitoring studies exist due to the relatively recent nature of such restoration programmes (Lunt *et al.,* 2010). Topographical variability and location of dams strongly influence the magnitude of change in the water table (Van Seters & Price 2001). Bog vegetation can return quickly (Rochefort *et al.,* 2012) following rewetting but species indicative of pristine bog conditions may fail to re-establish even after a decade post-restoration management (e.g. van Duinen *et al.,* 2003; Verberk *et al.,* 2010; Haapalehto *et al.,* 2011). Drain blocking can also be effective in reducing the further loss of nutrients with decreases in DOC and nutrient export recorded (e.g. Wallage *et al.,* 2006; Wilson *et al.,* 2010; Strack *et al.,* 2011; Anderson *et al.,* 2013; Turner *et al.,* 2013). Similarly, microbial function can return (Anderson *et al.,* 2013). However, Holden *et al.* (2007) warns that blocking all drains is not necessarily effective and the water table may not return to previous levels (Holden *et al.,* 2007). Despite this assertion there are multiple examples of ditch blocking resulting in rising water table levels and the start of processes that may lead to restoration of peatland ecological function (e.g. Ketcheson & Price 2011; Haapalehto *et al.,* 2011; Anderson *et al.,* 2013 and see examples in Cris *et al.,* 2011).

If water table levels are not returned to more favourable conditions for recolonisation, or the site is damaged badly, then reseeding and application of mulch can be utilised (Quinty & Rochefort, 2003; Holden *et al.,* 2007). However, it is recommended that prior to restoration of *Sphagnum* and other peat forming species, that the hydrological function of the site is returned (Quinty & Rochefort, 2003; Lunt *et al*., 2011). Only when this is achieved, is the restoration of peat forming vegetation recommended. Similarly, depending on the extent and type of damage at the site, other actions can also be taken such as stabilisation and peat reprofiling (Anon., 2008). However, what is reported here are the processes likely to be involved in repairing damage associated with wind farm construction and not that required to repair already damaged peatland.

3.11 Northern Ireland Priority Species

Other than birds and bats, there is little information on the effect the construction and operation of wind energy is likely to have on the majority of priority species in Northern Ireland (Table 8). However, it is very likely that habitat loss and disturbance during construction will be the most significant impacts. For example, hydrological changes associated with drainage of blanket bog pre-construction are likely to affect priority species such as the yellow marsh saxifrage (*Saxifraga hirculus*) which requires a high water table (Vittoz *et al.,* 2006) and whose habitat in Ireland has already declined dramatically (Beatty *et al.,* 2013). Similarly, lowering the water table as a result of drainage will also likely affect key invertebrate groups, some of whose larvae are specialist hosts on specific plants, for example, Lepidoptera such as the argent and sable moth (*Rheumaptera hastata*), large heath (*Coenonympha tullia*) and marsh fritillary (*Euphydryas aurinia*) butterflies. However, a paucity of information on these species, in general, and the wider impacts of wind turbines on invertebrates limit confidence in making predictions.

Nutrient leaching as a result of soil disturbance during construction could have an effect on nutrient levels in plants and those organisms feeding on them. For example, mollusc species such as Geyer's whorl snail (*Vertigo geyeri*), require high calcium levels for shell growth but this can be easily leached from drained bogs (Schenkova *et al.,* 2012). Turbine construction could also change adjacent water quality by, for example, increasing sedimentation by disturbing soils. In Northern Ireland, priority species such as the freshwater pearl mussel (*Margatifera margatifera*) and Atlantic salmon (*Salmo salar*) could conceivably be affected and turbine construction should be considered when creating catchment-level management plans (for example, those currently being drawn up by Donegal County Council for cross-border catchments occupied by the freshwater pearl mussel under INTERREG funding). Additionally, increased pH, variation in nutrient levels and DOC could impact wider freshwater invertebrate communities. Overall, more information is required on the distribution and ecology of many of the priority species in Northern Ireland before the impacts of wind farm and turbine construction and operation can be evaluated.

Table 8 List of Northern Ireland Priority Species found on blanket bog vulnerable to development through the construction of wind energy or turbines.

Species group	Species name	Common name	Location
Vascular plants	*Rubus chamaemorus*	Cloudberry	Recorded in upland bogs in Co. Tyrone
	Corex pauciflora	Few flowered sedge	Recorded in two upland sites, 1 in co. Down and 1 in Co. Antrim
	Saxifraga hirculus	Yellow marsh saxifrage	Upland sites in co. Antrim
Mosses	*Cinclidium stygium*	Lurid cupola moss	Recorded on Garron plateau
Fungi	*Armillaria ectypa*	Marsh honey fungus	Recorded on Garron plateau
	Dencoeliopsis johnstonii	None	Found on birch trees at Ballynahone Bog, co. Derry
Lichen	*Cladonia peziziformis*	None	Recorded on peaty soils at Fair Head, Co. Antrim
Dragonflies	*Coenagrion lunulatum*	Irish damselfly	Recorded at small mesotrophic lakes and cutover bogs
Molluscs	*Vertigo angustior*	Narrow mouthed whorl snail	Recorded at 2 sites in Co. Antrim
	Vertigo antivertigo	Marsh whorl snail	Recorded at 2 sites in Co. Fermanagh
	Vertigo geyeri	Geyer's whorl snail	Recorded at five sites in NI on raised bogs
	Vertigo lilljeborgs	Lilljeborg's whorl snail	Recorded in Fermanagh and Armagh
Moths	*Orthonama vittata*	Oblique carpet moth	Occurs on bogs and fens
	Xanthorhoe decoloraria	Red carpet moth	Upland species
	Xanthorhoe ferrugata	Dark barred twin spot carpet	Bog species
	Entephria caesiata	Grey mountain carpet moth	Upland areas in Co.s Fermanagh, Down and Antrim
	Rheumaptera hastate	Argent and sable Moth	Occurs in bogs with stands of bog myrtle = food plant
	Chiasmia clathrata	Latticed heath moth	Common on bogs
	Paraswmia plantaginis	Wood tiger moth	Occurs on acid grassland on bogs
	Xestia castanea	Neglected rustic moth	Occurs in bogs and Heaths of Fermanagh
	Xestia agathius	Heath rustic	Occurs in bogs and heaths
	Blepharita adusta	Dark brocade moth	Associated with bogs and heaths
	Calaeria haworthii	Haworth's minor moth	Common on bogs and heaths
Butterfly	*Eurodryas aurina*	Marsh fritillary	Occurs on mires and heaths
	Coenonymphs tullia	Large heath butterfly	Occurs on bogland in North and west of Ireland
Beetles	*Carabus claratus*	None	Occurs on blanket bog
	Hydrochus brevii	None	Occurs on bog of Fermanagh and Down
	Otiorhynchus auropunctatus	None	Occurs in upland Sphagnum bogs
Mammals	*Lepus timidus hibernicus*	Irish hare	Widespread occurrence

4.0 Discussion

This review follows others in suggesting that wind turbine construction and operation can have significant negative effects on local and regional biodiversity. However, the occurrence and magnitude of negative effects is not consistent and varies between taxa, species, site/locations and habitats and is, therefore, highly site-specific. Whilst wind energy facilities may affect a large range of species, birds and bats are two groups that are notably vulnerable. They are also the most identifiable groups at risk from onshore wind energy development and to date have been the focus of much of the research conducted on wind farms and biodiversity.

Additional mortality poses the greatest risk to species of conservation concern that have low rates of productivity and slow maturation rates e.g. all bat species in NI and birds of prey. Therefore, additional mortality arising from wind farm collisions may put species already at risk at a greater risk. It is imperative that the level of mortality at wind energy facilities for birds and bats be determined as a priority to establish what effects current development has on their populations. This is currently being done at wind energy facilities in North America where post and pre-construction mortality of birds and bats is a requirement. At present, pre-construction monitoring is currently the only requirement under Northern Ireland legislation to inform the siting of wind energy developments. However, until post-construction monitoring is a requirement, it is unlikely the true impact of onshore wind energy on bird and bat populations in Northern Ireland will be known and continued inference based on research conducted elsewhere will be inevitable.

A further impact on birds and bats is the barrier effect created by wind energy developments impeding movements. The negative effects of this are perceived to be minimal at present but if Northern Ireland continues to rely on onshore wind energy to meet renewable energy targets then there may be implications for migrating and foraging species. Therefore, it is imperative that cumulative impacts of wind energy developments include ecological elements and are not limited to visual impacts. Predictive modelling and mapping will be a particularly useful tool to inform the development of wind energy and there remains a need, particularly within a Northern

Ireland context, to make strategic assessments to inform developers of suitable and non-suitable sites for development from a biodiversity perspective.

A largely overlooked and arguably forgotten impact is that of locating wind energy developments on blanket bog. In addition to having negative implications for net carbon emission targets for Northern Ireland and the UK as a whole, damaging peatland by building wind energy developments has wider implications for flood management and water quality. The potential knock-on effects of using the peatland resource for wind turbines are considerable and it is arguable that the impacts on this facet of biodiversity will have the most noticeable and greatest financial implications for Northern Ireland. Further research, via post-construction monitoring, is undoubtedly required to quantify the impacts of wind energy developments on peatland ecology. Until this is done, continued inferences based on the impacts of other anthropogenic activities on peat habitats are the only option and this may be inappropriate.

In contrast to birds and bats, there is a paucity of information on the impacts that energy facilities have on other terrestrial mammals, other vertebrates, invertebrates and plants. It is arguable that these groups are much less likely to be negatively impacted by wind energy development. However, species that require specific habitats and the conditions they provide are at risk particularly if these are associated with blanket bog. Appropriate siting and planning can mitigate the worst of the effects on habitats.

The majority of our knowledge comes from wind energy facilities that contain multiple turbines. Our understanding of the effects of large single wind turbines on biodiversity is limited although it is likely that the same risks exist e.g. collision, displacement but these are smaller, as are the effects. The cumulative aspect of single wind turbines in the wider countryside must be taken into consideration during planning as at present this effect is inadequately assessed in relation to biodiversity.

Further research into the ranging behaviour and habitat preferences of particular species at particular sites is needed. Modelling based on empirical data from the field and/ or using remote sensing data are valuable tools for the assessment of impacts of wind energy and warrant wider applicability. It is also recommended that population-scale research into population dynamics and demography of priority bat and bird species is initiated to establish the effects, if any, of predicted or known collisions at the population level. Such research should also meet the requirement to understand the regional trends in productivity, survival, migration and dispersal movements of bats and birds and to obtain estimates of effective population sizes, population stability and the connectivity of populations with other parts of the UK and Europe.

5.0 Acknowledgments

This project was funded by the Natural Heritage Research Partnership (NHRP) between the Northern Ireland Environment Agency (NIEA) and *Quercus*, Queen's University Belfast (QUB). NIEA Client Officer was Claire Hempsey with input from Neill McCulloch and Ian Enlander. Rob Tomlinson and Chris Hunt reviewed the sections on peatland impacts. Thanks also to Una Nealon, Eimear Rooney and Irena Tomankova for their helpful comments on earlier drafts of the document.

6.0 References

AECOM. (2010). Scoping Report: Strategic Action Plan for Onshore Renewable Electricity Generation: Strategic Environmental Assessment. A Report for the Department of Enterprise, Trade and Industry, Belfast, Northern Ireland.

Ahlen, I. (2002). Bats and birds killed by wind power turbines. Fauna och Flora (Stockholm) 97: 14-21.

Ahlen, I. (2003). Wind turbines and bats - a pilot study. Final Report 11 December 2003 (Dnr 5210P-2002-00473, P-nr P20272-1) for the Swedish National Energy Administration. Sveriges Lantbruks Universitet, Sweden. [Report in Swedish translated into English].

Alberts, D.J. (2007). Stakeholders or subject matter experts, who should be consulted? Energy Policy 35: 2336-2346.

Anderson, R., Morrison, M., Sinclair, K., Strickland, D. (1999). Studying wind energy/bird interactions: a guidance document. Metrics and methods for determining or monitoring potential impacts on birds at existing and proposed wind energy sites. National Wind Coordinating Committee, Washington, DC. 94 p

Anderson, R., Wells, C., Macrae, M., Price, J. (2013). Nutrient mineralisation and microbial functional diversity in a restored bog approach natural conditions 10 years post restoration. Soil biology and Biochemistry, 64: 37-47.

Anonymous. (2003). Northern Ireland Habitat Action Plan – Blanket Bog. Northern Ireland Environment Agency, Belfast, UK.

Anonymous. (2006). Peat landslide hazard and risk assessments. Best Practice guide for proposed electricity generation developments. Produced by the Scottish Executive, Edinburgh, Scotland.

Anonymous. (2008a). Review of Guidance on the Assessment of Cumulative Impacts of Onshore Windfarms. Phase 1 Report. Report by Entec for the Department for business, Enterprise and Regulatory Reform.

Anonymous. (2008b). Angus Windfarms. Landscape Capacity and Cumulative Impacts Study. Report by Ironside Farrar for Angus Council.

Anonymous. (2008c). A compendium of UK peat restoration and management projects. DEFRA Research Project.

Anonymous. (2009). Cumulative Impact Assessment for two additional wind turbines at Myreton, near Keith, Moray. Report prepared by Seren energy on behalf of Myreton Renewable Energy Ltd.

Anonymous. (2010a). Assessing the effects of onshore wind farms on birds. Natural England Technical Information note TIN069.

Anonymous. (2010b). Use of avoidance rates in the SNH Wind Farm Collision Risk Model. SNH Avoidance Rate Information and Guidance Note. Scottish Natural Heritage.

Anonymous. (2011). Wolfe Island Wind Plant Post Construction Follow-up plan Bird and Bat Resources. Monitoring Report No. 5. January – June 2011. Prepared for Canadian Renewable Energy Corporation by Stantec Consulting Ltd.

Anonymous. (2012b). Wolfe Island Wind Plant Post Construction Follow-up plan Bird and Bat Resources. Monitoring Report No. 6. July – December 2011. Prepared for Canadian Renewable Energy Corporation by Stantec Consulting Ltd.

Anonymous. (2012b). Assessing the Cumulative impacts of onshore wind energy developments. Guidance Note. Scottish Natural Heritage.

Anonymous. (2013a). Hatchett Ridge wind Farm Post-Construction Mortality Monitoring Year Two Annual Report. A report by Tetra Tech to Hatchett Ridge Wind, LLC.

Anonymous. (2013b). Montezuma II Wind Energy Center Post-construction Monitoring Report, Year - 1. Prepared for NextEra Energy Montezuma II Wind, LLC by H.T.Harvey & Associates.

Anonymous. (2013c). Avoidance rates for wintering species of geese in Scotland at onshore wind farms. Scottish Natural Heritage Guidance document.

Armstrong, A., Holden, J., Kay, P., Foulger, M., Gledhill, S., McDonald, A.T., Walker, A. (2009). Drain-blocking techniques on blanket peak: a framework for best practice. Journal of Environmental Management, 90: 3512-3519.

Armstrong, A., Holden, J., Kay, P., Francis, B., Foulger, M., Gledhill, S., McDonald, A.T., Walker, A. (2010). The impact of peatland drain-blocking on dissolved organic carbon loss and discolouration of water: results from a national survey. Journal of Hydrology, 381: 112-120.

Armstrong, A., Waldron, S., Ostle, N., Whitaker, J. (2012). The effect of wind turbine-induced microclimates and plant functional types on peatland greenhouse gas emissions and pore water dissolved organic carbon concentrations. European Geophysical Union Research Abstract, 2012.

Armstrong, A., Waldron, S., Ostle, N., Whitaker, J. (2013). Vegetation, soil property and climatic controls over greenhouse gas fluxes in a blanket peatland hosting a wind farm. European Geophysical Union Research Abstract, 2013.

Arnett, E.B., (2005). Relationships between bats and wind turbines in Pennsylvania and West Virginia: an assessment of bat fatality search protocols, patterns of fatality, and behavioural interactions with wind turbines. A final report submitted to the Bats and Wind energy Cooperative. Bat Conservation International, Austin, Texas, USA.

Arnett, E.B., Brown, W.K., Erickson, W.P., Fiedler, J.K., Hamilton, B.L., Henry, T.H., Jain, A., Johnson, G.D., Kerns, J., Koford, R.R., Nicholson, C.P., O'Connell, T.J., Piorkowski, M.D. & Tankersley, R.D. (2008). Patterns of bat fatalities at wind energy facilities in North America. Journal of Wildlife Management 72: 61-78.

Arnett, E.B., Huso, M.M.P., Schirmacher, M.R., Hayes, J.P. (2011). Altering turbine speed reduces bat mortality at wind energy facilities. Frontiers in Ecology and the environment, 9: 209-214.

Arnett, E.B., Hein, C.D., Schirmacher, M.R., Huso, M.M.P., Szewczak, J.M. (2013a). Evaluating the effectiveness of an ultrasonic acoustic deterrent for reducing bat fatalities at wind turbines. PloS One, 8: e65794.

Arnett, E.B., Johnson, G.D., Erickson, W.P., Hein, C.D. (2013b). A synthesis of operational mitigation studies to reduce bat fatalities at wind energy facilities in North America. A report submitted to the National Renewable Energy Laboratory. Bat Conservation International. Austin, Texas, USA.

Baerwald, E.F. (2008). Variation in the activity and fatality of migratory bats at wind energy facilities in southern Alberta: causes and consequences. Thesis, University of Calgary, Calgary, Alberta, Canada.

Baerwald, E.F. & D’Amours, G.H., Klug, B.J., Barclay, R.M.R. (2008). Barotrauma is a significant cause of bat fatalities at wind turbines. Current Biology, 18; R695-R696.

Baerwald, E.F. & Barclay, R.M.R. (2007). Migratory bats and wind turbines in Alberta: Temporal and spatial variation in bat activity and fatality. Bat Research News, 48: 176.

Baerwald, E.F. & Barclay, R.M.R. (2008). Patterns of activity and fatality of migratory bats at a wind energy facility in Southwestern Alberta. Bat Research News, 49: 98-99.

Baerwald, E.F. & Barclay, R.M.R. (2009). Geographic variation in activity and fatality of migratory bats at wind energy facilities. Journal of Mammalogy, 90: 1341-1349.

Baerwald, E.F.& Barclay, R.M.R. (2011). Patterns of activity and fatality of migratory bats at a wind energy facility in Alberta, Canada. Journal of Wildlife Management, 75: 1103-1114.

Baida Roy, S., Pacala, S.W., Walko, R.L. (2004). Can large wind energy affect local meteorology? Journal of Geophysical Research, 109: D19101.

Baidya Roy, S. & Traiteur, J.J. (2010). Impacts of wind energy on surface air temperatures. Proceedings of National Academy of Sciences, 107: 17899-17904.

Baidya Roy, S. (2011). Simulating impacts of wind energy on local hydrometeorology. Journal of wind Engineering and Industrial Aerodynamics, 99: 491-498.

Bain, C.G., Bonn, A., Stoneman, R., Chapman, S., Coupar, A., Evans, M., Geary, B., Howat, M., Joosten, H., Keenleyside, C., Labadz, J., Lindsay, R., Littlewood, N., Lunt, P., Miller, C.J., Moxey, A., Orr, H., Reed, M., Smith, P., Swales, V., Thompson, D.B.A., Thompson, P.S., Van de Noort, R., Wilson, J.D., Worral, R. (2011). IUCN UK Commission of Inquiry on Peatlands. IUCN UK Peatland Programme.

Ballard, C.E., McIntyre, N., Wheater, H.S. (2012). Effects of peatland drainage management on peak flows. Hydrology and Earth Systems Sciences, 16: 2299-2310.

Band, W. (2000). Windfarms and birds: calculating a theoretical collision risk assuming no avoiding action. Scottish Natural Heritage Guidance Note.

Band, W., Madders, M., Whitfield, D.P. (2007). Developing field and analytical methods to assess avian collision risk at wind farms. In De Lucas, M., Janss, G., Ferrer, M. (eds) Birds and Wind Poer. Barcelona, Spain: Lynx Edicions.

Barclay, R.M.R., Baerwald, E.F. & Gruver, J.C. (2007). Variation in bat and bird fatalities at wind energy facilities: assessing the effects of rotor size and tower height. Canadian Journal of Zoology-Revue Canadienne De Zoologie, 85: 382-387.

Barrios, L. & Rodriguez, A. (2004). Behavioural and environmental correlates of soaring-bird mortality at on-shore wind turbines. Journal of Applied Ecology, 41: 72-81.

Beatty, G.E., Reid, N., Provan, J. (2013). Retrospective genetic monitoring of the threatened Yellow marsh saxifrage (*Saxifraga hirculus*) reveals genetic erosion but provides valuable insights for conservation strategies. Diversity and Distributions, Online Early: DOI:10.1111/ddi.12158.

Beddoe, M. & Chamberlin, A. (2003). Avoiding confrontation: securing planning permission for onshore wind energy developments in England: comments from a wind energy developer. Planning Practice & Research, 18: 3-17.

Behr, O., Hochradel, K., Mages, J., Nagy, M., Korner-Nievergelt, F., Niermann, I., Simon, R., Stillerm F., Weber, N., Brinkman, R. (2013). Bat friendly operation algorithms: reducing bat fatalities at wind turbines in central Europe. Presentation to the German-French Bureau for Renewable Energy, 26th September 2013.

Bekessy, S.A., Wintle, B.A., Gordon, A., Fox, J.C., Chisholm, R., Brown, B., Regan, T., Mooney, N., Read, S.M. & Burgman M.A. (2009). Modelling human impacts on the Tasmanian wedge-tailed eagle (*Aquila audax fleayi*). Biological Conservation, 142: 2438-2448.

Bellebaum, J., Korner-Nievergelt, F., Dürr, T., Mammen, U. (2013). Wind turbine fatalities approach a level of concern in a raptor population. Journal for Nature Conservation, 21: 394-400.

Bernardino, J., Bispo, R., Costa, H., Mascarenhas, M. (2013). Estimating bird and bat fatality at wind energy: a practical overview of estimators, their assumptions and limitations. New Zealand Journal of Zoology, 40: 63-74.

Bevanger, K. (1998). Biological and conservation aspects of bird mortality caused by electricity power lines: a review. Biological Conservation, 86: 67-76.

Birtwell I.K. (1999). Effects of sediment on fish and their habitat. Pacific Scientific Advice Review Committee (PSARC) Research Document HAB 99-1. Fisheries and Oceans Canada, Canadian Stock Assessment Secretariat, Ottawa 34 p

Bispo, R., Palminha, G., Bernardino, J., Marques, T., Pestana, D., (2010). A new statistical method and a web based application for the evaluation of the scavenging removal correction factor. In Proceedings of the VII Wind Wildlife Research Meeting, Denver, EUA.

Blumstein, D., Fernandez-Juricic, E., Zollner, P., Garity, S. (2005). Inter-specific variation in wildlife responses to human disturbance: a review and synthesis. Journal of Applied Ecology, 42: 943-953.

Bradley, R.I., Milne, R., Bell, J., Lilly, A., Jordan, C., Higgins, A. (2005). A soil carbon and land use database for the United Kingdom. Soil Use and Management, 21: 363-369.

Bright, J. Langston, R., Bullman, R., Evans, R., Gardner, S., Pearce-Higgins, J. (2008). Map of bird sensitivities to wind energy in Scotland: A tool to aid planning and conservation. Biological Conservation, 141: 2342-2356.

Bull, L. S., Fuller, S., Sim, D., (2013). Post-construction avian mortality monitoring at Project West Wind. New Zealand Journal of Zoology, 40: 28-46.

Campeau, S. & Rochefort, L. (1996). Sphagnum regeneration on bare peat surfaces: field and greenhouse results. Journal of Applied Ecology, 33: 599-608.

Carrete, M., Sanchez-Zapata, J.A., Benitez, J.R., Lobon, M. & Donazar, J.A. (2009). Large scale risk-assessment of wind-farms on population viability of a globally endangered long-lived raptor. Biological Conservation, 142: 2954-2961.

Carette, M., Sanchez-Zapata, J.A., Benitez, J.R., Lobon, M., Montoya, F., Donazar, J.A., (2012). Mortality at wind-farms in positively related to large scale distribution and aggregation in griffon vultures. Biological Conservation, 145: 102-108.

Chamberlain, D., Freeman, S., Rehfisch, M., Fox, T., Desholm, M. (2005). Appraisal of Scottish Natural Heritage's Wind Farm Collision Risk Model and its Application. BTO Research Report 401. British Trust for Ornithology, Thetford

Chamberlain, D.E., Rehfisch, M.R., Fox, A.D., Desholm, M., Anthony, S.J. (2006). The effect of avoidance rates on bird mortality predictions made by wind turbine collision risk models. Ibis, 148: 198-202.

Charman, D. (2002). Peatlands and Environmental Change. John Wiley and Sons Ltd, Chichester, UK. pp 301.

Cohn, J.P. (2008). How eco-friendly are windfarms? Bioscience, 58: 576-578.

Cooper, L.M. & Sheate, W.R., 2002. Cumulative effects assessment: A review of UK environmental impact statements. Environmental Impact Assessment Review, 22: 415-439

Corten, G.P. & Veldkamp, H.F. (2001). Aerodynamics - Insects can halve wind-turbine power. Nature, 412: 41-42.

Cowell, R. (2009). Wind power, landscape and strategic, spatial planning - the construction of 'acceptable' locations in Wales. Land Use Policy doi:10.1016/j.landusepol.2009.01.006

Cris, R., Buckmaster, S., Bain, C., Bonn, A. (Eds). (2011). UK Peatland Restoration – Demonstrating success, IUCN UK National Committee Peatland Programme, Edinburgh.

Crockford, N. (1992). A review of possible impacts of wind energy on birds and other wildlife. JNCC Report 27. Peterborough.

Cryan, P.M. (2008) Mating behaviour as a possible cause of bat fatalities at wind turbines. Journal of Wildlife Management, 72: 845-849.

Cryan, P.M. (2009). Mating behaviour as a possible cause of bat fatalities at wind turbines. Journal of Wildlife Management, 72: 845-849.

Cryan, P.M. & Brown, A.C. (2007). Migration of bats past a remote island offers clues toward the problem of bat fatalities at wind turbines. Biological Conservation, 139: 1-11.

Cryan, P.M. & Barclay, R.M.R. (2009). Causes of bat fatalities at windfarms: hypotheses and predictions. Journal of Mammalogy, 90: 1330-1340.

Curry, A. (2009). Deadly flights. Science, 325: 386-387.

Dahl, E.L., Bevanger, K., Nygard, T., Roskaft, E., Stokkem B.G. (2012). Reduced breeding success in white-tailed sea eagles at Smöla windfarm, western Norway, is caused by mortality and displacement. Biological Conservation, 145: 79-85.

Dalili, N., Edrisy, A. & Carriveau, R. (2009). A review of surface engineering issues critical to wind turbine performance. Renewable Sustainable Energy Reviews, 13: 428-438.

Desholm, M. (2009). Avian sensitivity to mortality: prioritising migratory bird species for assessment at proposed wind energy. Journal of Environmental Management, 90: 2672-2679.

Department of Enterprise, Trade and Industry (2010). Energy: A Strategic Framework for Northern Ireland. A Report by the Department of Enterprise, Trade and Industry, Belfast, Northern Ireland.

Department of Enterprise, Trade and Industry (2013). Onshore Renewable Electricty Action Plan 2013-2020. A Report by the Department of Enterprise, Trade and Industry, Belfast, Northern Ireland.

Devereux, C.L., Denny, M.J.H., Whittingham, M.J. (2008). Minimal effects of wind turbines on the distribution of wintering farmland birds. Journal of Applied Ecology, 45: 1689-1694.

de Lucas, M., Janss, G.F.E., Ferrer, M. (2004). The effects of a wind farm on birds in a migration point: the Strait of Gibraltar. Biodiversity and Conservation, 13: 395-407.

de Lucas, M., Janss, G.F.E., Whitfield, D.P., Ferrer, M. (2008). Collision fatality of raptors in wind energy does not depend on raptor abundance. Journal of Applied Ecology, 45: 1695-1703.

de Lucas, M., Ferrer, M., Bechard, M.J., Munoz, A.R. (2012). Griffon vulture mortality at wind energy in southern Spain: Distribution of fatalities and active mitigation measures. Biological Conservation, 147: 184-189.

Dillingham, P.W. & Fletcher, D. (2008). Estimating the ability of birds to sustain additional human-caused mortalities using a simple decision rule and allometric relationships. Biological Conservation, 141: 1783-1792.

Doty, A.C. & Martin, A.P., (2013). Assessment of bat and avian mortality at a pilot wind turbine at Coega, Port Elizabeth, Eastern Cape, South Africa. New Zealand Journal of Zoology, 40: 75-80.

Douglas, D.J.T., Bellamy, P.E., Pearce-Higgins, J.W., (2011). Changes in the abundance and distribution of upland breeding birds at an operational wind farm. Bird Study, 58: 37-43

Douglas, D.J.T., Follestad, A., Langston, R.H.W., Pearce-Higgins, J.W. (2012). Modelled sensitivity of avian collision rate at wind turbines varies with number of hours of flight activity input data. Ibis, 154: 858-861.

Drewitt, A.L. & Langston, R.H.W. (2006). Assessing the impacts of wind energy on birds. Ibis, 148: 29-42.

Drewitt, A.L. & Langston, R.H.W. (2008). Collision effects of wind-power generators and other obstacles on birds. Year in Ecology and Conservation Biology, 1134: 233-266.

Dürr, T. & Bach, L. (2004). Bat deaths and wind turbines - a review of current knowledge and the information available in the database for Germany. Bremer Beiträge für Naturkunde und Naturschutz, 7: 253-264.

Duxbury, J.M. & Peverly, J.H. (1978). Nitrogen and Phosphorus Losses from Organic soils. Journal of Environmental Quality, 7: 566-570.

Dykes, A.P. & Jennings, P. (2011). Peat slope failures and other mass movements in western Ireland, August 2008. Quarterly Journal of Engineering Geology and Hydrogeology, 44; 5-16.

Dykes, A.P. & Warburton, J. (2007). Mass movements in peat: A formal classification scheme. Geomorphology, 86: 73-93.

Dykes, A.P. & Warburton, J. (2008). Characteristics of the Shetland Islands (UK) peat slides of 19 September 2003. Landslides, 5: 213-226.

Eaton, J.M., McGoff, N.M., Byrne, K.A., Leahy, P., Kiely, G. (2008). Land cover change and soil organic carbon stocks in the Republic of Ireland 1851-2000. Climatic Change, 91: 317-334.

Eichhorn, M., Johst, K., Seppelt, R., Drechsler, M. (2012). Model-based estimation of collision risks of predatory birds with Wind Turbines. Ecology and Society, 17: 1. dx.doi.org/10.5751/ES-04594-170201

Ellison, L.E. (2012). Bats and wind energy – A literature synthesis and annotated bibliography: US Geological Survey Open File Report 2012 -1110, pp57.

Elthem, D.C., Harrison, G.P. & Allen, S.J. (2008). Change in public attitude towards a Cornish wind farm: implications for planning. Energy Policy, 36: 23-33.

Erickson, W.P., Johnson, G.D., Strickland, M.D., Kroner, K. (2000). Avian and bat mortality associated with the Vansycle wind project, Umatilla County, Oregon; 1999 study year. Technical Report prepared by WEST Inc. for Umatilla County Department of Resource Services and Development, Pendleton, OR, pp25.

Erickson, W.P., Johnson, G.D., Strickland, M.D., Young, D.P., Sernka, K.J., Good, R.E. (2001). Avian collisions with wind turbines: a summary of existing studies and comparisons to other sources of avian collision mortality in the United States. National Wind Coordinating committee, c/o RESOLVE, Washington, D.C., USA.

Erickson, J.L., & West, S.D. (2002). The influence of regional climate and nightly weather conditions on activity patterns of insectivorous bats. Acta Chiropterologica, 4: 17-24.

Erickson, W.P., Johnson, G.D., Young, D.P. (2005). A summary and comparison of bird mortality from anthropogenic causes with an emphasis on collisions. US Department of Agriculture Forest Service General Technical Report PSW-GTR-191, Albany, California, USA.

Erickson, W.P., Jeffrey, J.D., Poulton, V.K. (2008). Puget Sound Energy Wild Horse Energy Facility Post-construction Avian and Bat Monitoring First Annual Report January – December 2007. Prepared for Puget Sound Energy, Washington, USA.

Eurobats. (2012). Report of the IWG on wind turbines and bat population. 17th Meeting of the Advisory committee, Dublin, Ireland, 2012.

Eurobats (2013). Report of the IWG on Wind Turbines and Bat Populations. 18th meeting of the Eurobats Advisory Committee, Sofia, Bulgariam 15th-17th April 2013.

Everaert, J. & Stienen, E.W.M. (2007). Impact of wind turbines on birds in Zeebrugge (Belgium). Biodiversity and Conservation, 16: 3345-3359.

Farfan, M.A., Vargas, J.M., Duarte, J., Real, R. (2009). What is the impact of wind energy on birds? A case study in southern Spain. Biodiversity and Conservation, 18: 3743-3758.

Ferrer, M., de Lucas, M., Janss, G.F.E., Casado, E., Munoz, A.R., Bechard, M.J., Calabuig, C.P. (2012). Weak relationship between risk assessment studies and recorded mortality in wind energy. Journal of Applied Ecology, 49: 38-46.

Fiedler, J.K., Henry, T.H., Tankersley, R.D. & Nicholson C.P. (2007). Results of bat and bird mortality monitoring at the expanded Buffalo Mountain Windfarm, 2005.

Fielding, A.H., Whitfield, D.P., McLeod, D.R.A. (2006). Spatial association as an indicator of the potential for future interactions between wind energy developments and golden eagles Aquila chrysaetos in Scotland. Biological Conservation, 131: 359-369.

Flydal, K., Eftestol, S., Reimers, E.,Colman, J.E. (2004). Effects of wind turbines on area use and behaviour of semi-domestic reindeer in enclosures. Rangifer, 24: 55-66.

Frost, S.W. (1958). Insects attracted to light traps placed at different heights. Journal of Ecological Entomology, 51: 550-551.

Garvin, J.C., Jennelle, C.S., Drake, D., Grodsky, S.M. (2011). Response of raptors to a windfarm. Journal of Applied Ecology, 48: 199-209.

Gill, J.P., Townsley, M., Mudge, G.P. (1996). Review of the impacts of windfarms and other aerial structures on birds. Scottish Natural Heritage Review 21, SNH, Edinburgh, UK.

Gillespie, M.K. (2013). Bird and bat responses to wind energy development in Iowa. Graduate Theses and Dissertations, Iowa State University.

Grace, M., Dykes, A., Thorp, S., Crowle, A. (2013). The impacts pf tracks on the integrity and hydrological function of blanket peat. Natural England Evidence review NEER002. Natural England.

Grant, A., Clarke, P., Lynch, S. (2010). Landscape capacity studies in Scotland – a review and guide to good practice. Scottish Natural Heritage Commissioned Report No. 385.

Grieve, I. & Gilvear, D. (2008). Effects of wind farm construction on concentrations of dissolved organic carbon and suspended sediment from peat catchments at Braes of Doune, central Scotland. Mires and Peat, 4: 1-11.

Grodsky, S.M., Behr, M.J., Gendler, A., Drake, D., Dieterle, B.D., Rudd, R.J., Walrath, N.L. (2011). Investigating the causes of death for wind-turbine associated bat fatalities. Journal of Mammalogy, 92: 917-925.

Grodsky, S.M.& Drake, D. (2012). Assessing Bird and Bat mortality at the Forward Energy Center. Prepared for Forward Energy LLC.

Gue, C. T., Walker, J.A., Mehl, K.R., Gleason, J.S., Stephens, S.E., Loesch, C.R., Reynolds, R.E., Goodwin, B.J. (2013). The effects of a large-scale wind farm on breeding season survival of female mallards and blue-winged teal in the Prairie Pothole Region. Journal of Wildlife Management, 77: 1360-1371.

GWEC, 2013. Global Wind Statistics; 2012. Global Wind Energy Council.

Haapaletho, T.O., Vasander, H., Jauhiainen, S., Tahvanainen, T., Kotiaho, J.S. (2011). The effects of peatland restoration in water-table depth, elemental concentrations, and vegetation: 10 years of changes. Restoration Ecology, 19: 587-598.
Hall, L.S. & Richards, G.C. (1972). Notes on Tadarida australis (Chiroptera: Molossidae). - Australian Mammalogy, 1: 46.

Harding, G., Harding, P., Wilkins, A. (2008). Wind turbines, flicker, and photosensitive epilepsy: characterising the flashing that may precipitate seizures and optimising guidelines to prevent them. Epilepsia, 49: 1095-1098.

Hatchet, E.S., Hale, A.M., Bennett, V.J., Karsten, K.B. (2013). Wind turbines do not negatively affect nesting success in the Dickcissel (*Spiza Americana*). Auk, 130: 520-528.

Hayes, M. (2013). Bats killed in large numbers at United States Wind Energy Facilities. Bioscience, 63: 975-979.

Herrera-Alsina. L., Villegas-Patraca, R., Eguiarte, L.E., Arita, H.T. (2013). Bird communities and wind energy: a phylogenetic and morphological approach. Biodiversity and Conservation, 22: 2821 – 2836.

Hedenstrom, A. & Rydell, J. (2011). Effect of wind turbine mortality on Noctule bats in Sweden: predictions from a simple population model. Conference on wind energy and wildlife impacts, 2-5 May 2011, Trondheim, Norway.

Hedenstrom, A. & Rydell, J. (2013). Effect of wind turbine mortality on bat populations in Sweden: predictions from a simple population model. Conference on wind energy and wildlife impacts, 5-7 February 2013, Stockholm, Sweden.

Herrera-Alsina, L., Villegas-Patraca, R., Eguiarte, L.E., Arita, H.T. (2013). Bird communities and wind farms: a phylogenetic and morphological approach. Biodiversity Conservation, 22: 2821-2836.

Holden, J., Chapman, P.J., Labadz, J.C. (2004). Artificial drainage of peatlands: hydrological and hydrochemical process and wetland restoration. Progress in Physical Geography, 28: 95-123.

Holden, J., Evans, M.G., Burt, T.P., Horton, M. (2006). Impact of land drainage on peatland hydrology. Journal of Environmental Quality, 35: 1764-1778.

Holden, J., Shotbolt, L., Bonn, A., Burt, T.P., Chapman, P.J., Dougill, A.J., Fraser, E.D.G., Hubacek, K., Irvine, B., Kirkby, M.J., Reed, M.S., Prell, C., Stagl, S., Stringer, L.C., Turner, A., Worral, F. (2007). Environmental change in moorland landscapes. Earth-Science Reviews, 82: 75-100.

Horn, J.W., Arnett, E.B. & Kunz, T.H. (2008a). Behavioural responses of bats to operating wind turbines. Journal of Wildlife Management, 72: 123-132.

Horn, J.W., Arnett, E.B., Jensen, M., Kunz, T.H. (2008b). Testing the effectiveness of an experimental acoustic bat deterrent at the Maple Ridge Wind Farm. A report for the The Bats and Wind Energy Cooperative and Bat Conservation International, Austin, Texas.

Hötker, H., Thomsen, K.-M. & Jeromin, H. (2006): Impacts on biodiversity of exploitation of renewable energy sources: the example of birds and bats - facts, gaps in knowledge, demands for further research, and ornithological guidelines for the development of renewable energy exploitation. Michael-Otto-Institut im NABU, Bergenhusen

Hoover, S.L. & Morrison, M.L. (2005). Behaviour of red-tailed hawks in a wind turbine development. Journal of Wildlife Management, 69: 150-159.

Hull, A. (1995). New models for implementation theory: striking a consensus on windfarms. Journal of Environmental Planning and Management, 38: 285-305.

Hull, C.L., Stark, E.M., Perruzzi, S., Simms, C.C. (2013). Avian collisions at two wind energy in Tasmania, Australia: taxonomic and ecological characteristics of colliders versus non-colliders. New Zealand Journal of Zoology, 40: 47-62.

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

Hull, C.L. & Cawthen, L. (2013). Bat fatalities at two wind energy in Tasmania, Australia: bat characteristics, and spatial and temporal patterns. New Zealand Journal of Zoology, 40: 5-15.

Huntley, B., Collingham, Y.C., Green, R.E., Hilton, G.M., Rahbeck, C. & Willis, S.G. (2006). Potential impacts of climatic change upon geographical distribution of birds. Ibis, 148: 8-28.

Huso, M. (2010). An estimator of wildlife fatality from observed carcasses. Environmetrics, 22, 318-329.

Ingersoll, T.E., Sewall, B.J., Amelon, S.K. (2013). Improved analysis of long-term monitoring data demonstrates marked regional declines of bat populations in the Eastern United States. PLoS ONE; DOI: 10.1371/journal.pone.0065907

Jain, A.A. (2005). Bird and bat behaviour and mortality at a northern Iowa windfarm. Masters Thesis, Iowa State University.

Jain, A.A., Koford, R.R., Hancock, A.W., Zenner, G.G. (2011). Bat mortality and activity at a Northern Iowa Wind Resource Area. American Midland Naturalist, 165: 185-200.

Johnson, G.D., Erickson, W.P., Strickland, M.D., Sheperd, M.F., Sheperd, D.A., Sarappo, S.A. (2002). Collision mortality of local migrant birds at a large-scale wind-power development on Buffalo Ridge, Minnesota. Wildlife Society Bulletin, 30: 879-887.

Johnson, G.D., Erickson, W.P., Strickland, M.D., Sheperd, M.F., Sheperd, D.A., Sarappo, S.A. (2003). Mortality of bats at a large-scale wind power development at Buffalo Ridge, Minnesota. American Midland Naturalist, 150: 332-342.

Johnson, G.D., Perlik, M.K., Erickson, W.I.P., Strickland, J.D. (2004). Bat activity, composition, and collision mortality at a large wind plant in Minnesota. Wildlife Society Bulletin, 32: 1278-1288.

Jones, G., Cooper-Bohannon, R., Barlow, K., Parsons, K. (2009). Determining the potential ecological impact of wind turbines on bat populations in Britain.

Kemp. M., Shamoun-Baranes, J., Dokter, A.M., van Loon, E., Bouten, W. (2013). The influence of weather on the flight altitude of nocturnal migrants in mid-latitudes. Ibis, 155: 734-749.

Kepel, A., Ciechanowski, M., Jaros, R. (2011). How to assess the potential impact of wind turbines on bats using bat activity surveys? A case study from Poland, XII European Bat Research Symposium, August 22-26, 2011, Vilinusm Lithuania.

Kerlinger, P., Guarnaccia, J., Hasch, A., Culverm R.C.E., Curry, R.C., Tran, L., Stewart, M.J., Riser-Espinoza, D. (2012). Avian collision mortality at 50 and 60m guyed towers in central California. Condor, 114: 462-469.

Ketcheson, S.J. & Price, J.S. (2011). The impact of peatland restoration on the site hydrology of an abandoned block-cut bog. Wetlands, 31: 1263-1274.

Kikuchi, R. (2008). Adverse impacts of wind power generation on collision behaviour of birds and anti-predator behaviour of squirrels. Journal for Nature Conservation, 16: 44-55.

Korner-Niervergelt, F., Brinkmann, R., Niermann, I., Behr, O. (2013). Estimating bat and bird mortality at wind energy turbines from covariates and carcass searches using mixture models. PLoS One, 8: e67997.

Kunz, T.H., Arnett, E.B., Erickson, W.P., Hoar, A.R., Johnson, G.D., Larkin, R.P., Strickland, M.D., Thresher, R.W., Tuttle, M.D. (2007a). Ecological impacts of wind energy development on bats: questions, research needs, and hypotheses. Frontiers in Ecology and the Environment, 5: 315–324.

Kunz, T.H., Arnett, E.B., Cooper, B.A., Erickson, W.I.P., Larkin, R.P., Mabee, T., Morrison, M.L., Strickland, J.D. & Szewczak, J.M. (2007b). Assessing impacts of wind energy development on nocturnally active birds and bats: a guidance document. Journal of Wildlife Management, 71: 2449-2486.

Kusstatscher, K., Kofler, K., Ackerl, M.H. (2005). Alpine Windharvest: an Interreg III B Alpine Space Programme: Impact on wildlife and plant life

Kuvlesky, W.P., Brennan, L.A., Morrison, M.L., Boydston, K.K., Ballard, B.M. & Bryant, F.C. (2007). Wind energy development and wildlife conservation: challenges and opportunities. Journal of Wildlife Management, 71: 2487-2498.

Lagrange, H., Rico, P., Bas, Y., Ughetto, A-L., Melki, F., Kerbirou, C. (2013). Mitigating bat fatalities from wind-power plants through targeted curtailment: results from 4 years of testing of CHIROTECH©. Conference on wind energy and wildlife impacts, 5-7 February 2013, Stockholm, Sweden.

Lamers, L.P.M., Smolders, A. J.P., Roelofs, J.G.M. (2002). The restoration of fens in the Netherlands. Hydrobiologia, 478: 107-130.

Lane, S.N. & Milledge, D.G. (2013). Impact of upland open drains upon runoff generation: a numerical assessment of catchment-scale impacts. Hydrological Processes, 27: 1701-1726.

Langston, R.H.W. & Pullan, J.D. (2003) Windfarms and birds: an analysis of windfarms on birds, and guidance on environmental assessment criteria and site selection issues. Royal Society for Protection of Birds/Report by BirdLife International to the Council of Europe (Bern Convention), Council of Europe Report T-PVS/Inf (2003) 12.

Larsen, J.K. & Madsen. J. (2000). Effects of wind turbines and other physical elements on field utilization by pink-footed geese (*Anser brachyrhynchus*): a landscape perspective. Landscape Ecology, 15: 755-764.

Leeks, G.J.L. & Roberts, G. (1987). The effects of forestry on upland streams – with special reference to water quality and sediment transport. In:Good, J.E,G. (Ed) Environmental Aspects of Plantation Forestry in Wales. Grange-over-Sands, NERC/ITE, 64-69. (ITE Symposium, 22).

Lenzen, M. & Munksgaard, J. (2002). Energy and CO_2 life-cycle analyses of wind turbines - review and applications. Renewable Energy, 26: 339-362.

Liechti, F., Guelat, J., Komenda-Zehnder, S. (2013). Modelling the spatial concentrations of bird migration to assess conflicts with wind turbines. Biological Conservation, 162; 24-32.

Lilly, A., Grieve, I.C., Jordan, C., Baggaley, N.J., Birnie, R.V., Futter, M.N., Higgins, A., Hough, R., Jones, M., Noland, A.J., Stutter, M.I., Towers, W. (2009). Climate change, land management and erosion in the organic and organo-mineral soils in Scotland and Northern Ireland. Scottish Natural Heritage Commissioned Report No. 325 (ROAME No. F06AC104 – SNIFFER UKCC21).

Limpens, H.J.G.A. & Kapteyn, K. (1991). Bats, their behaviour and linear landscape elements. Myotis, 29: 39-48.

Limpens, J., Berrendse, F., Blodau, C., Canadell, J.G., Freeman, C., Holden, J., Roulet, N., Rydin, H., Schaepman-Strub, G. (2008). Peatlands and the carbon cycle: from local processes to global implications – a synthesis. Biogeosciences, 5, 1475-1491.

Lindsay, R. & Bragg, O. (2005). Windfarms and blanket peat: the bog slide of 16th October 2003 at Derrybrien, Co. Galway, Ireland. Unpublished report by the Derrybrien Development Cooperative Ltd. 2nd Edition.

Lindsay, R. (2010). Peatbogs and carbon: a critical synthesis to inform policy development in oceanic peat bog conservation and restoration in the context of climate change. A Report for RSPB Scotland.

Linnell, J.D.C., Swenson, J., Andersen, R., Barnes, B. (2000). How vulnerable are denning bears to disturbance? Wildlife Society Bulletin, 28: 400-413.

Long, C.V., Flint, J.A., Lepper, P.A., Dible, S.A. (2009). Wind turbines and bat mortality: interactions of bat echolocation pulses with moving turbine rotor blades. In: Fifth international conference on Bio-acoustics. Proceedings of the Institute of Acoustics, 31, 183-190.

Long, C.V., Flint, J.A., James, A., Lepper, P.A. (2010). Wind turbines and bat mortality: Doppler shift profiles and ultrasonic bat like pulse reflection from moving turbine blades. Journal of Acoustical society of America, 128; 2238-2245.

Long, C.V., Flint, J.A., Lepper, P.A. (2011a). Insect attraction to wind turbines: does colour play a role? European Journal of Wildlife Research, 57: 323-331.

Long, M., Jennings, P., Roselyn, C. (2011b). Irish peat slides 2006-2010. Landslides, 8; 391-401.

Lovich, J.E., Ennen, J.R., Madrak, S., Meyer, K., Loughran, C., Bujurlin, C., Arundel, T., Turner, W., Jones, C., Groenendaal, G.M. (2011). Effects of wind energy production on growth demography, and survivorship of a desert tortoise (*Gopherus agassizii*) population in southern California with comparisons to natural populations. Herpetological Conservation and Biology, 6: 161-174.

Lundy, M.G., Harrison A., Buckley, D.J., Boston, E.S., Scott, D.D., Teeling, E.C., Montgomery, W.I., Houghton, J.D.R. (2013). Prey field switching based on preferential behaviour can induce Levy flights. Journal of the Royal Society Interface, 10: UNSP 201220489.

Lunt, P., Allot, T., Anderson, P., Buckler, M., Coupar, A., Jones, P., Labadz, J., Worrall, P. (2010). Peatland restoration: A scientific review. This review was commissioned by the IUCN UK Peatland Programmes Commission of Inquiry on Peatlands.

Madders, M. & Whitfield, D.P. (2006). Upland raptors and the assessment of wind farm impacts. Ibis, 148: 43-56.

Madsen, J. & Boertmann, D. (2008). Animal behavioural adaptation to changing landscapes: spring-staging geese habituate to wind energy. Landscape Ecology, 23: 1007-1011.

Magnusson, M. & Smedman, A.S. (1999). Air flow behind wind turbines. Journal of wind Engineering and Industrial Aerodynamics, 80, 169 – 189.

Marris, E. & Fairless, D. (2007). Windfarms deadly reputation hard to shift. Nature, 447: 126.

Martin, G.R. & Shaw, J.M. (2010). Bird collision with power lines: Failing to see the way ahead? Biological Conservation, 143:2695-2702.

Martin, G.R. (2012). Through birds' eyes: insights into avian sensory ecology. Journal of Ornithology, 153, Supplement 1: 23-48.

Martin, G.R., Portugal, S.J., Murn, C.P. (2012). Visual fields, foraging and collision vulnerability in Gypus vultures. Ibis, 154: 626-631.

Martínez, J.E., Calvo, J.F., Martinez, J.A., Zuberogoitia, I., Cerezo, E., Manrique, J., Gomez, G.J., Nevado, J.C., Sanchez, M., Sanchez, R., Bayo, J., Pallarés, A., Gonzalez, C., Gomez, J.M., Perez, P., Motos, J. (2010). Potential impact of wind energy on territories of large eagles in southeastern Spain. Biodiversity and Conservation, 19: 3757-3767.

Martinez-Abrain, A., Tavecchia, G., Regan, H.M., Jimenez, J., Surroca, M., Oro, D. (2012). Effects of wind energy and food scarcity on a large scavenging bird species following an epidemic of bovine spongiform encephalopathy. Journal of Applied Ecology, 49: 109-117.

Masden, E.A., Haydon, D.T., Fox, A.D., Furness, R.W., Bullman, R., Desholm, M. (2009a). Barriers to movement: impacts of wind energy on migrating birds. Ices Journal of Marine Science, 66: 746-753.

Masden, E.A., Fox, A.D., Furness, R.W., Bullman, R., Haydon, D.T. (2009b). Cumulative impact assessments and bird/wind farm interactions: Developing a conceptual framework. Environmental Impact Assessment Review, 30: 1-7.

Masden, E.A. (2010). Assessing the cumulative impacts of wind energy on birds. Unpublished PhD thesis, University of Glasgow.

McIsaac, H.P. (2001). Raptor acuity and wind turbine blade conspicuity. Pp. 59–87 in Schwartz, S.S. (Ed.): Proceedings of the National Avian-Wind Power Planning Meeting IV, Carmel, California, 16–17 May 2000. Unpublished report prepared for the Avian Subcommittee of the National Wind Coordinating Committee by RESOLVE Inc., Washington, DC

Minderman, J., Pendlebury, C.J., Pearce-Higgins, J.W., Park, K.J. (2012). Experimental evidence for the effect of small wind turbine proximity and operation in bird and bat activity. PLoS One, 7: e41177.

Money, R.P. (1995). Restoration of cut-over peatlands: the role of hydrology in determining vegetation quality. In: Hughes, J., Heathwaite, L. Hydrology and Hydrochemistry of British Wetlands. 383-400. Wiley-Blackwell, UK.

Mouton, P.F.N., Meyer, A. & Hayward, J. (2007). Proposed wind energy facility and associated infrastructure: terrestrial fauna scoping report. Report to Savannah Environmental Ltd.

Muller, F.L.L., Tankere-Muller, S.P.C. (2012). Seasonal variations in surface water chemistry at disturbed and pristine peatland sites in the Flow Country of northern Scotland. Science of the Total Environment, 435: 351-362.

Murray, H.S. (2012). Assessing the impact of windfarm-related disturbance on streamwater carbon, phosphorus and nitrogen dynamics: A case study of the Whitelee catchments. Unpublished PhD Thesis, University of Glasgow.

Musters, C.J.M., Noordervliet, M.A.W. & Tekeurs, W.J. (1996). Bird casualties caused by a wind energy project in an estuary. Bird Study, 43: 124-126.

Natural England. (2010). Assessing impacts of wind farm development on blanket peatland in England. Natural England.

Nayak, D.R., Miller, D., Nolan, A., Smith, P., Smith, J.U. (2010). Calculating carbon budgets of wind energy on Scottish peatlands. Mires and Peat, Volume 4 (2008-2010), Article 09; 1-23

Neal, C., Fisher, R., Smith, C.J., Hill, S., Jeffery, H.A. (1992). The effects of tree harvesting on stream water quality at an acidic and acid sensitive spruce forested area: Plynlimon, mid-Wales. Journal of Hydrology, 135: 305-319.

Neal, C., Reynolds, B., Neal, M., Hill, L., Wickham, H., Pugh, B. (2003). Nitrogen in rainfall, cloud water, throughfall, stemflow, stream water and ground water for the Plynlimon catchments of mid-Wales. Science of the Total Environment, 312: 121-51.

Neal, C., Reynolds, B., Neal, M., Wickham, H., Hill, L., Williams, B. (2004). The impact of conifer harvesting on stream water quality: the Afon Hafren, mid-Wales. Hydrology and Earth System Sciences, 8: 503-520.

Nelson, B. (2001). The distribution, status, habitat of the Marsh Fritillary butterfly *Euphydryas aurinia* (Lepidoptera: Nymphalidae) in Northern Ireland Irish Naturalists' Journal, 27: 129-139.

Newcombe, C.P. & MacDonald, D.D. (1991). Effects of suspended sediments on aquatic ecosystems. North American Journal of Fisheries Management, 11: 72-82

Nicholls, B. & Racey, P.A. (2007). Bats avoid radar installations: could electromagnetic fields deter bats from colliding with wind turbines? PLoS One, 2: e297.

Nicholls, B. & Racey, P.A. (2009). The aversive effect of electromagnetic radiation on foraging bats: a possible means of discouraging bats from approaching wind turbines. PLoS One, 4: e6246.

Northern Ireland Environment Agency. (2010). Wind Energy Development in Northern Ireland's Landscapes: Supplementary Planning guidance to accompany Planning Policy Statement 18 'Renewable Energy'. NIEA Research and Development Series No 10/01, Belfast.

Orloff, S. & Flannery, A. (1992). Wind turbine effects on avian activity, habitat use, and mortality in Altamont Pass and Solano County Wind Resource Areas Tiburon, the Planning Departments of Alameda, Contra Costa, and Solano Counties and the California Energy Commission. Report prepared for the California Energy Commission

Osborn, R.G., Higgins, K.F., Dieter, C.D. & Usgaard, R.E. (1996). Bat collisions with wind turbines in southern Minnesota. Bat Research News, 37: 105-108.

Park, K.J., Turner, A., Minderman, J. (2013). Integrating applied ecology and planning policy: the case of micro-turbines and wildlife conservation. Journal of Applied Ecology, 50: 199-204.

Pasqualetti, M. J., Gipe, P & Righter R.W (2004). Wind power in view: energy landscapes in a crowded world, Academic Press, San Diego.

Pearce-Higgins, J.W., Stephen, L., Langston, R.H.W., Bright, J.A. (2008). Assessing the cumulative impacts of wind energy on peatland birds: a case study of golden plover *Pluvialis apric*aria in Scotland. Mires and Peat, Volume 4 (2008-2010), Article 01: 1-13.

Pearce-Higgins, J.W., Stephen, L., Langston, R.H.W., Bainbridge, I.P., Bullman, R. (2009). The distribution of breeding birds around upland wind energy. Journal of Applied Ecology, 46: 1323-1331.

Pearce-Higgins, J.W., Stephen, L., Douse, A., Langston, R.H.W. (2012). Greater impacts of wind energy on bird populations during construction than subsequent operation: results of a multi-site and multi-species analysis. Journal of Applied Ecology, 49: 386-394.

Pedersen, E., van den Berg, F., Bakker, R. (2009). Response to noise from modern wind energy in the Netherlands. Journal of the Acoustical Society of America, 126: 634-643.

Piorkowski, M.D. & O'Connell, T.J. (2010). Spatial pattern in summer mortality from collisions with wind turbines in mixed grass Prairie. American Midland Naturalist, 164: 260-269.

Plonczhier, P. & Simms, I. C. (2012). Radar monitoring of migrating pink-footed geese: behavioural responses to offshore wind farm development. Journal of Applied Ecology, 49: 1187-1194.

Powlesland, R.G. (2009). Impacts of wind energy on birds: a review. Science for Conservation 289. Department of Conservation, Wellington. 51p.

Price, J.S., Heathwaite, A.L., Baird, A.J. (2003). Hydrological processes in abandoned and restored peatlands: an overview of management approaches. Wetland Ecology and Management, 11: 65-83.

Quinty, F. & Rochefort, L. (2003). Peatland Restoration Guide. 2nd Edition. Canadian Sphagnum Peat Moss Association and New Brunswick Department of Natural Resources and Energy, Quebec, Quebec.

Rabin, L.A., Coss, R.G., Owings, D.H. (2006). The effects of wind turbines on anti-predator behaviour in California ground squirrels (*Spermophilus beecheyi*). Biological Conservation, 131:410-420.

Rajewski, D.A. (2013). CWEX (Crop/Wind-energy experiment): Measurements of the interaction between crop agriculture and wind power. Unpublished PhD Thesis, Iowa State University.

Rea, H., Ostle, N., Waldron, S., Whitaker, J. (2012). Investigating the impacts of a wind farm induced microclimate on peatland carbon cycling. European Geophysical Union Research Abstract, EGU General Assembly 2012, 22-27April 2012, Vienna, Austria. p 909.

Reimer, J., Baerwald, E.F., Barclay, R.M.R. (2008).Feeding by migrating Hoary Lasiurus cinereus and Silver-haired Bats *Lasionycteris noctivagans*: are they attracted to insects at wind turbines? Bat Research News, 49: 162-163.

Reynolds, D.S. (2006). Monitoring the potential impact of a wind development site on bats in the northeast. Journal of Wildlife Management, 70: 1219-1227.

Richardson, H., Whitaker, J., Waldron, S., Ostle, N. (2013). Peatland carbon cycling at a Scottish wind farm: the role of plant-soil interactions. European Geophysical Union Research Abstract, EGU General Assembly 2013, 7-12 April, Vienna, Austria. p 636.

Roberts, G. & Crane, S.B. (1997). The effects of clear-felling established forestry on stream-flow losses from the Hore sub-catchment. Hydrology and Earth System Sciences, 1: 477-482.

Rochefort, L., Campeau, S., Bugnon, J.L. (2002). Does prolonged flooding prevent or enhance regeneration and growth of Sphagnum? Aquatic Botany, 74: 327-341.

Rodgers, M., O'Conner, M., Healy, M.G., O'Driscoll, C., Asman, Z., Nieminen, M., Poole, R., Muller, M., Xiao, L. (2010). Phosphorus release from forest harvesting on an upland blanket peat catchment. Forest Ecology and Management, 260: 2241-2248.

Rollins, K.E., Meyerholz, D.K., Johnson, G.D., Capparella, A.P., Loew, S.S. (2012). A forensic investigation into the etiology of bat mortality at a wind farm: barotrauma or traumatic injury? Veterinary Pathology, 49: 362-371.

Roscioni, F., Russo, D., Di Febbraro, M., Frate, L., Carranza, M.L., Loy, A. (2013). Regional-scale modelling of the cumulative impact of wind energy on bats. Biodiversity and Conservation, 22; 1821-1835.

Ruddock, M. & Whitfield, D.P., (2007). A review of disturbance distances in selected bird species. A report from Natural Research (Projects) Ltd to Scottish Natural Heritage.

Russ J.M. & Montgomery, W.I. (2002). Habitat associations of bats in Northern Ireland: implications for conservation. Biological Conservation, 108: 49-58.

Rydell, J., Bach, L., Dubourg-Savage, M.J., Green, M., Rodrigues, L., Hedenstrom, A. (2010a). Bat mortality at wind turbines in northwestern Europe. Acta Chiropterologica, 12: 261-274.

Rydell, J., Bach, L., Dubourg-Savage, M.J., Green, M., Rodrigues, L., Hedenstrom, A. (2010b). Mortality of bats at wind turbines links to nocturnal migration? European Journal of Wildlife Research, 56: 823-827.

Santos, H., Rodriques, L., Jones, G., Rebelo, H. (2013). Using species distribution modelling to predict bat fatality risk at wind energy. Biological Conservation, 157: 178-186.

Sauvajot, R.M., Buechner, M., Kamradt, D., Schonewald, C. (2004). Patterns of human disturbance and response by small mammals and birds in chaparral near urban development. Urban Ecosystems, 2: 279-297.

Simmons, J.A., Neretti, N., Intrator, N., Altes, R.A., Ferragamo, M.J., Sanderson, M.I. (2004). Delay accuracy in bat sonar is related to the reciprocal of normalized echo bandwidth, or Q. Proceedings of the National Academy of Sciences of the United States of America, 101: 3683-3643.

Schaub, M. (2012). Spatial distribution of wind turbines is crucial for the survival of red kite popualtions. Biological Conservation, 155: 111-118.

Schenkova, V., Horsak, M., Pleskova, Z., Pawlikowski, P. (2012). Habitat preferences and conservation of *Vertigo geyeri* (Gastropoda: Pulmonata) in Slovakia and Poland. Journal of Molluscan Studies, 78: 105-111.

Shankar, P.N. (2001). Can insects seriously affect the power output of wind turbines? Current Science, 81: 747-748

Smallwood, K.S. & Neher, L. (2004). Repowering the APWRA: forecasting and minimising avian mortality without significant loss of power generation. Report to California Energy Commission.

Smallwood, K.S. & Karas, B. (2009). Avian and bat fatality rates at old-generation and repowered wind turbines in California. Journal of Wildlife Management, 73: 1062-1071.

Smallwood, K.S. & Thelander, C.G. (2004). Developing methods to reduce bird mortality in the Altamont Pass Wind Resource Area. Prepared for the California Energy Commission: 1-363

Smallwood, K.S., Rugge, L., Morrison, M.L. (2009). Influence of behaviour on bird mortality in wind energy developments. Journal of Wildlife Management, 73: 1082-1098.

Smallwood, K.S. (2007). Estimating wind turbine caused bird mortality. Journal of Wildlife Management, 71: 2781-2791.

Smallwood, K. S. (2013). Comparing bird and bat fatality-rate estimates among North American wind-energy projects. Wildlife Society Bulletin, 37: 19-33.

Smith, J., Nayak, D.R., Miller, D., Nolan, A., Smith, P. (2009). Wind energy on peatland: Effect of management practices on carbon emissions. European Geophysical Union Research Abstract, EGU General Assembly 2009, 19-24 April, Vienna, Austria. p 12484..

Smith, J., Nayak, D.R. Smith, P. (2014). Wind farms on undegraded peatlands are unlikely to reduce future carbon emissions. Energy Policy, 66: 585-591.

Sovacool, B.K. (2009). Contextualizing avian mortality: A preliminary appraisal of bird and bat fatalities from wind, fossil-fuel, and nuclear electricity. Energy Policy, 37: 2241-2248.

Stewart, G.B., Pullin, A.S., Coles, C.F. (2005). Effects of wind turbines on bird abundance. CEE Review 04-002 (SR4). Collaboration for Environmental Evidence: www.environmentalevidence.org/SR4.thml

Stewart, G.B., Pullin, A.S., Coles, C.F. (2007). Poor evidence-base for assessment of windfarm impacts on birds. Environmental Conservation, 34: 1-11.

Still, D., Painter, S., Lawrence, E.S., Little, B., Thomas, M. & Anderson, M. (1997). Birds, wind energy, and Blyth Harbour. Wind Energy Conversion, 1996: 175-183.

Strack, M., Toth, K., Bourbonniere, R., Waddington, J.M. (2011). Dissolved organic carbon production and runoff quality following peatland extraction and restoration. Ecological Engineering, 37: 1998-2008.

Sugimoto, H. & Matsuda, H. (2011). Collision risk of white-fronted Geese with wind turbines. Ornithological Science, 10: 61-71.

Surlykke, A., Pederson, S.B., Jakobsen, L. (2009). Echolocating bats emit a highly directional sonar beam in the field. Proceedings of the Royal Society B-Biological Sciences, 276: 853-860.

Sutherland, W.J., Bailey, M.J., Bainbridge, I.P., Brereton, T.; Dick, J.T.A., Drewitt, J., Dulvy, N.K., Dusic, N.R., Freckleton, R.P., Gaston, K.J., Gilder, P.M., Green, R.E., Heathwaite, A.L., Johnson, S.M., Macdonald, D.W., Mitchell, R., Osborn, D., Owen, R.P., Pretty, J., Prior, S.V., Prosser, H., Pullin, A.S., Rose, P., Stott, A., Tew, T., Thomas, C.D., Thompson, D.B.A., Vickery, J.A., Walker, M., Walmsley, C., Warrington, S., Watkinson, A.R., Williams, R.J., Woodroffe, R. & Woodroof, H.J. (2008). Future novel threats and opportunities facing UK biodiversity identified by horizon scanning. Journal of Applied Ecology, 45: 821-833.

Szewczak, J.M. & Arnett, E.B. (2006). An acoustic deterrent with the potential to reduce bat mortality from wind turbines. Bat Research News, 47: 151-152.

Szewczak, J.M. & Arnett, E.B. (2008). The potential for acoustic broadcasts to deter bats from approaching wind turbines. Bat Research News, 49: 178-179.

Tapia, L., Dominguez, J. & Rodriguez, L. (2009). Using probability of occurrence to assess potential interaction between wind energy and a residual population of golden eagle *Aquila chrysaetos* in NW Spain. Biodiversity and Conservation, 18: 2033-2041.

Telleria, J.L. (2009a). Wind power plants and the conservation of birds and bats in Spain: a geographical assessment. Biodiversity and Conservation, 18: 1781-1791.

Tellería, J.L. (2009b). Overlap between wind power plants and Griffon Vultures *Gyps fulvus* in Spain. Bird Study, 56: 268-271.

Telleria, J.L. (2009c). Potential impacts of wind farms on migratory birds crossing Spain. Bird Conservation International, 19: 131-136.

Tetzlaff, D., Malcolm, I.A., Soulsby, C. (2007). Influence of forestry, environmental change and climatic variability on the hydrology, hydrochemistry and residence times of upland catchments. Journal of Hydrology, 346: 93-111.

Thelander, C.G., Smallwood, K.S., Rugge, L. (2003). Bird risk behaviours and fatalities at the Altamount Pass and Wind Resource Area, Period of Performance: March 1998 – December 2000/ BioResource Consultants, Ojai, California.

Toke, D. (2003). Windpower in the UK: how planning conditions and financial arrangements affect outcomes. International Journal of Sustainable Energy, 23: 207-216.

Trinder, M (2012). The potential consequences of elevated mortality on the population viability of whooper swans in relation to wind farm developments in Northern Scotland. Scottish Natural Heritage Commissioned Report No. 459.

Turner, E.K., Worral, F., Burt, T.P. (2013). The effect of drain blocking on the dissolved organic carbon (DOC) budget of an upland peat catchment in the UK. Journal of Hydrology, 479: 169-179.

Van Duinen, G.A., Brock, A.M. T., Kuper, J.T., Leuven, R.S.E.W., Peeters, T.M.J., Roelofs, J.G.M., van der Velde, G., Verberk, W.C.E.P., Esselink, H. (2003). Do restoration measures rehabilitate fauna diversity in raised bogs? A comparative study on aquatic invertebrates. Wetlands Ecology and Management, 11: 447-459.

Van Seters, T. & Price, J.S. (2001). The impact of peat harvesting and natural regeneration on the water balance of an abandoned cutover bog, Quebec. Hydrological Processes, 15, 233-248.

Verberk, W.C.E.P., Leuven, R.S.E.W., van Duinen, G.A., Esselink, H. (2010). Loss of environmental heterogeneity and aquatic macroinvertebrate diversity following large-scale restoration management. Basic and Applied Ecology, 11: 440-449.

Vittoz, P., Gobat, J-M., Wyss, T. (2006). Ecological conditions for *Saxifraga hirculus* in Central Europe: a better understanding for a good protection. Biological Conservation, 131: 594-608.

Voigt, C.C., Popa-Lisseanu, A.G., Niermann, I., Kramer-Schadt, S. (2012). The catchment area of wind energy for European bats: a plea for international regulations. Biological Conservation, 153: 80-86.

Waldron, S., Flowers, H., Arlaud, C., Bryant, C., McFarlane, S. (2009). The significance of organic carbon and nutrient export from peatland-dominated landscapes subject to disturbance, a stoichiometric perspective. Biogeosciences, 6: 363-374.

Wallage, Z.E., Holden, J., McDonald, A.T. (2006). Drain blocking: An effective treatment for reducing dissolved organic carbon loss and water discolouration in a drained peatland. Science of the Total Environment, 387: 811-821.

Walsh, A.L. & Harris, S. (1996). Factors determining the abundance of vesptertillionid bats in Britain: geographical, land class, and local habitat relationships. Journal of Applied Ecology, 33: 519-529.

Walsh-Thomas, J.M., Cervone, G., Agouris, P., Manca, G. (2012). Further evidence of impacts of large-scale wind energy on land surface temperatures. Renewable and Sustainable Energy Reviews, 16: 6432-6437.

Walter, W.D., Leslie, D.M. & Jenks, J.A. (2006) Response of Rocky Mountain elk (*Cervus elaphus*) to wind-power development. American Midland Naturalist. 156: 363-375.

Warburton, J., Holden, J., Mills, A.J. (2004). Hydrological controls of surficial mass movements in peat. Earth-Science Reviews, 67: 139-156.

Warren, C.R. & McFadyen, M. (2009). Does community ownership affect public attitude to wind energy? A case study from south-west Scotland. Land Use Policy doi:10.1016/j.landusepol.2008.12.010

Whitfield, D.P. (2009). Collision avoidance of Golden Eagles at wind farms under “Band” Collision Risk Model. Report to Scottish Natural Heritage by Natural Research Ltd, Banchory, UK.

Whitfield, D.P. & Madders, M. (2005). Flight height in the hen harrier Circus cyaneus and its incorporation in wind turbine collision risk modelling. Natural Research Information Note 2. Natural Research Ltd, Banchory, UK.

Whitfield, D.P. & Madders, M. (2006). A review of the impacts of wind farms on hen harriers *Circus cyaneus* and an estimation of collision avoidance rates. Natural Research Information Note 1 (Revised). Natural Research Ltd, Banchory, UK.

Whitfield, D.P., Ruddock, M., Bullman, R. (2008). Expert opinion as a tool for quantifying bird tolerance to human disturbance. Biological Conservation, 141: 2708-2717.

Wilson, L., Wilson, J., Holden, J., Johnstone, I., Armstrong, A., Morris, M. (2010). Recovery of water tables in Welsh blanket bog after drain blocking: discharge rates, time scales and the influence of local conditions. Journal of Hydrology, 391: 377-386

Willis, C.K.R., Barclay, R.M.R., Boyles, J.G., Brigham, R.M., Brack, V., Waldien, D.L., Reichard, J. (2010). Bats are not birds and other problems with Sovacool's (2009) analysis of animal fatalities due to electricity generation. Energy Policy, 38: 2067-2069.

Winkelman J.E. (1985). Bird impact by middle size wing turbines on flight behaviour victims and disturbance. Limosa, 58: 117-121.

Winkelman, J.E. (1992). The impact of the Sep wind park near Oosterbierum, The Netherlands, on birds, 2: nocturnal collision risks. Unpublished RIN report 92/3. DLO-Instituut voor Bos-en Natuuronderzoek, Arnhem, The Netherlands.

Worral, F., Armstron, A., Holden, J. (2007). Short-term impact of peat drain-blocking on water colour, dissolved organic carbon concentration, and water table depth. Journal of Hydrology, 337: 315-325.

Young, D.P., Erickson, W.P., Strickland, M.D., Good, R.E. & Sernka, K.J. (2003). Comparison of avian responses to UV-light-reflective paint on wind turbines. Subcontract report July 1999 – December 2000. National Renewable Energy Laboratory, Colorado.

Young, D.P., Nomani, S., Tidhar, W.L., Bay, K. (2010). NedPower Mount Storm Wind Energy Facility Post-Construction Avian and Bat Monitoring. April -July 2010. Prepared for NedPower Mount Storm, LLC.

Young, D.P., Nomani, S., Tidhar, W.L., Bay, K. (2011). NedPower Mount Storm Wind Energy Facility Post-Construction Avian and Bat Monitoring. July – October 2010. Prepared for NedPower Mount Storm, LLC.

Zhang, W., Markfort, C.D., Porte-Agel, F. (2013). Experimental study of the impact of large-scale wind energy on land-atmosphere exchanges. Environmental Research Letters, 8: 015002.

Zhou, L., Tian, Y., Baidya Roy, S., Thorncroft, C., Bosart, L.F., Hu, Y. (2012). Impacts of wind energy on land surface temperature. Nature Climate Change, 2, 529-543.

Zimmerling, J.R., Pomeroy, A.C., d'Entremont, M.V., Francis, C.M. (2013). Canadian estimate of bird mortality due to collisions and direct habitat loss associated with wind turbine developments. Avian Conservation and Ecology, 8: 10-22.

Appendix 1 A summary of studies on the impact of wind energy developments on birds cited in the main text of the review.

Species	Country	No. of study sites	No. of replicates	No. of controls	Factor	Analysis	Effect	Conclusion	Reference
Multiple spp.	Sweden	4	-	0	Mortality	Yes	Yes	Direct mortality of 33 species of birds, most notably insectivores (e.g. swifts & swallows). Thermal imagery revealed birds flying close to turbine blades.	Ahlen, 2002
Multiple spp.	USA	22	-	0	Mortality	Yes	Yes	Direct morality increased with tower height but unaffected by blade size or MWh output.	Barclay *et al*., 2007
Multiple spp.	Spain	2	68	0	Mortality	No	Yes	Direct mortality of storks, raptors & owls with griffon vulture and kestrel most vulnerable. Highest at turbines than powerlines. Mortality varied seasonally and with wind-topography interactions.	Barrios & Rodriguez, 2004
Red kite	Germany	69	7428 searches conducted over 10 year period	n/a	estimated probability of collision and mortality levels	yes	yes	The authors suggest that cumulative wind farm related mortalities of red kites may negatively affect long-term viability of German populations.	Bellebaum *et al.*, 2013
Multiple spp.	Netherlands	n/a	49 studies used - 90 datasets extracted resulting in 2107 data points.	n/a	Mean Species Abundance	yes	yes	Mammal and bird population densities declined with proximity to infrastructure (which includes wind parks). The effect on birds lasted up to 1km and mammals up to 5km.	Benitez-Lopez *et al.*, 2010
Multiple spp.	USA	1	survey point every 200m surveyed 4 times in a 1 year period.	n/a	suitability of a location for a wind turbine	yes	n/a	Modelling was undertaken to predict best location for wind turbines based on wind and bird activity. Placement of exclusion zones around areas of high bird activity had a negative impact on energy generation scenarios.	Bohrer *et al.*, 2013

Species	**Country**	**No. of study sites**	**No. of replicates**	**No. of controls**	**Factor**	**Analysis**	**Effect**	**Conclusion**	**Reference**
Multiple spp.	New Zealand	1	24 turbines	n/a	mortality	n/a	n/a	First study from New Zealand to report bird mortality at an operating wind energy facility.	Bull *et al.*, 2013
Egyptian vultures	Spain	27 wind farms -	Carcass searches once a week at each wind farm. 1279 occupied breeding territories and 433 extinct	n/a	n/a	yes	n/a	Up to 40% of occupied Egyptian vultures considered at risk due to proximity to wind farms. Modelling indicates that the entire meta-population of the Spanish Egyptian vultures is declining due to wind farms.	Carette *et al.*, 2009
Griffon vultures	Spain	34	Mortality data from 799 turbines over 10 years. 122 griffon vulture colonies	n/a	mortality	yes	yes	Turbines located in areas with higher aggregations of griffon vultures killed more birds. This contradicts the findings of de Lucas et al., 2008 but supports those of Barrios and Rodriguez 2004.	Carette *et al.*, 2012
Galapagos petrel	Ecaudor/Spain	1	50 observation points	43 non wind turbine points	movements and behaviour	yes	n/a	Petrels have a clear bimodal pattern of activity associated with dawn and dusk. Petrels flew below rotor swept areas but the species does fly higher during courtship. Suggest that petrels not at great risk at study site.	Cruz-Delgado *et al.*, 2010
White-tailed sea eagle	Norway	1	47 territories were identified over 10 years	n/a	territory size and breeding success	yes	yes	The number of successful breeding attempts in territories close to and within the windfarm were lower than eagles who had territories outside the wind farm.	Dahl *et al.*, 2012

Species	Country	No. of study sites	No. of replicates	No. of controls	Factor	Analysis	Effect	Conclusion	Reference
Raptor & passerine spp.	Spain	1	5740m of transect; 435 hours of focal observation	2	Abundance & behaviour	Yes	Species dependent	Abundance of passerines and kestrels higher in windfarms. No difference in abundance of raptors or storks. Lower number of passerine nests but greater productivity at windfarms. Greatest effect from operational turbines as barriers.	de Lucas *et al*., 2004
Multiple (mainly passerine spp.)	Spain	1	3	1	Abundance & behaviour	Yes	No	No effect on abundance or flying height except during construction.	de Lucas *et al*., 2005
Raptor spp.	Spain	2	400 hours focal observations	0	Mortality	Yes	Yes	Direct mortality greatest in winter and pre-breeding season and not associated with abundance. No evidence of habituation.	de Lucas *et al*., 2008
Griffon vulture	Spain	13	10 wind farms selectively stopped	3 wind farms as normal	mortality	yes	yes	Mean mortality declined at wind farms with selective stopping compared to when they were running normally. Mortality was higher at the 3 wind farms running normally compared to the selective stopping turbines.	de Lucas *et al.,* 2012
Griffon vultures	Spain	n/a	176 hours of observation	n/a	mortality and presence	yes	yes	Vultures had preferred trajectories in an areas which were determined by wind speed which in turn was related to the underlying topography	de Lucas *et al.,* 2012
Farmland spp. (mainly corvid, gamebird & passerine spp.)	UK	1	11	0	Abundance & behaviour	No	No	No effect on the abundance of 33 species, however, common pheasant (*Phasianus colchicus*) avoided turbines. Skylark and corvids found significantly closer to turbines than expected but effect confounded by habitat.	Devereux *et al*., 2008

Species	Country	No. of study sites	No. of replicates	No. of controls	Factor	Analysis	Effect	Conclusion	Reference
Multiple spp.	South Africa	1	154 inspections	n/a	mortality	n/a	n/a	This study reports the findings of the monitoring of a wind farm in South Africa. Only one bird was found dead during searches over a year.	Doty & Martin, 2013
red grouse and european golden plover	UK	1	10 visits made over 2 years	1 (7 visits made)	abundance and distribution	yes	yes	Red grouse occurred closer to wind turbines than predicted by chance. There was no effect of wind turbines on golden plover distribution. Turbines appeared to have no negative effect on abundance	Douglas *et al.*, 2011
White-tailed sea eagles	UK/Norway	1	175 Vantage Point observations over 1 year	n/a	collision rate	yes	yes	A greater number of observation hours reduced the variability of calculated collision rates for white tailed sea eagles at a wind farm in Norway.	Douglas *et al.*, 2012
Red kite	Germany	n/a	n/a	n/a	n/a	n/a	n/a	The authors used bird records and the weather data to determine whether existing wind turbines were located in energy efficient areas. They found that for energy production and bird protection existing wind turbines were poorly sited.	Eichhorn & Drechsler, 2010
Red kite	Canada	n/a	n/a	n/a	n/a	n/a	n/a	The authors developed a tool that models collision risk based on foraging behaviour and landscape structure. The authors suggest it will be useful for planning future wind farms	Eichhorn *et al.*, 2012
Seabird spp.	Belgium	1	-	0	Mortality	No	Yes	Direct mortality of terns and gulls (19.1 birds/turbines/year). Greatest morality at seaward turbines.	Everaert & Stienen, 2006

Species	Country	No. of study sites	No. of replicates	No. of controls	Factor	Analysis	Effect	Conclusion	Reference
Tern & gull spp.	Belgium	1	-	0	Mortality	No	Yes	High direct mortality when situated close to breeding colonies (6.7 - 19.1 birds/turbine/year). High level of avoidance. Greatest morality at seaward turbines (<27.6 birds/turbine/year).	Everaert & Stienen, 2007
Multiple (mainly gull & duck spp.)	Belgium	3	40	0	Behaviour	Yes	Yes	Roosting or foraging waterbirds avoided turbines by 150-300m. Direct mortality greatest during breeding season between 0 - 125 birds/turbine/year with seaward turbines presenting greatest risk	Everaert, 2003
Multiple spp.	Spain	1	209 hours of observation and 186 hours of searches	0	bird density, abundance, behaviour and mortality	yes	yes	Raptor numbers declined post-construction. No effect on densities of the one species studied was found. Disturbance of passerines was negligible.	Farfan et al., 2009
Multiple spp.	USA	n/a	n/a	n/a	n/a	n/a	n/a	The authors used models to identify areas for wind farms that would have minimal additional risk to wildlife receptors.	Fargione *et al.,* 2012
Multiple spp.	Spain/USA	53 proposed sites (20 of these eventually active)	Mean observation effort 107 to 228 hours (total effort: 7267 hours).	n/a	bird abundance and mortality	yes		There was no relationship between bird mortality and bird activity. Bird mortality showed no difference to the level of threat predicted by the wind farm.	Ferrer *et al.,* 2012
Golden eagle (*Aquila chrysaetos*)	UK	-	-	-	Spatial analysis	No	-	No perceived threat as <4% of territories overlapped windfarm sites.	Fielding *et al*., 2006
Eurasian griffon vulture and Egyptian vulture	Spain	n/a	5 sub-populations which included 19 Spanish provinces	n/a	population growth rates	yes	yes	Modelling revealed declines in growth rates of sub-populations when including wind farms in population models for both species but the exact effects could not be distinguished clearly from other factors e.g. pollutants	Garcia-Ripolles & Lopez-lopez, 2011

Species	Country	No. of study sites	No. of replicates	No. of controls	Factor	Analysis	Effect	Conclusion	Reference
Raptors	USA	1	249 surveys over 3 years	n/a	abundance and behaviour	yes	yes	Raptor abundance declined by 47% compared to pre-construction levels. Flight height varied between species but most individuals remained 100m away from turbines.	Garvin *et al.*, 2011
Mallard and blue-winged teal	USA	2	77 mallard and 88 teal at wind farm. 70 mallard and 75 teal at Ref.	1	female survival	yes	mixed	There was a low number of collisions recorded at the wind farm site and thus the authors suggest wind farms will have no direct effect on female survival.	Gue *et al.*, 2013
Dickcissel	USA	6	192 nests	n/a	nest success	yes	no	There was no evidence of nest displacement and no negative effect of wind turbine on nest survival.	Hatchett *et al*, 2013
Multiple spp.	Mexico	1	1 vantage point used number of days surveyed not clear	n/a	multiple	yes	yes	The authors suggest that birds colliding with wind turbines is a result of a two step process. Certain species of bird are more likely to fly close to turbines and a subset of these are prone to collision with turbines. Each step has a different probability of occurring.	Herrera-Alsina *et al.*, 2013
Multiple spp.	Global	Review	127 published/ unpublished studies	Mortality & avoidance	Species dependent			Main hazard identified are: disturbance (displacement/exclusion) and direct mortality. No significant effect found during meta-analysis on breeding species except for waders, but evidence of avoidance by geese, ducks and waders during winter of up to 800m	Hoetker *et al*., 2005
Red-tailed hawk (*Buteo jamaicensis*)	USA	1	15	0	Behaviour	Yes	Yes	Species-specific flight behaviour increased perceived risk of direct mortality.	Hoover & Morrison, 2005

Species	Country	No. of study sites	No. of replicates	No. of controls	Factor	Analysis	Effect	Conclusion	Reference
Birds in general	Australia	n/a	n/a	n/a	modelling	yes	n/a	The authors model the "fall-zone" of birds and bats colliding with different sized turbines. The authors suggest the size of the search area is important to estimate levels of mortality.	Hull &Muir, 2010
Multiple spp.	Australia	2	1228 surveys conducted over 11 years	none	utlilisation and collision risk	yes	yes	Collision counts of species were slightly correlated with rank abundance. Presence/absence on the sites were not a good indicator if collision risk.	Hull *et al,* 2013
Multiple spp.	USA	1	-	0	Mortality	No	-	Direct mortality of migrants and to less extent residents and mostly passerines. Radar indicated 3.5 million birds migrating over the windfarm annually	Johnson *et al*., 2002
Golden eagles	USA/Canada	3	8 (2 nestlings and 6 adults)	n/a	behavioural and flight differences	yes	yes	Golden eagles flew at lower heights over cliffs and steep slopes or summits and ridgetops. This has implications for risk of collision with turbines in such locations.	Katzner *et al.,* 2012
birds	USA	30	30	n/a	mortality	yes	yes	The authors examined collision rates of night migrant birds with turbines and towers and the effects of lighting. Lighting and weather conditions may have been causative factors in four mortalities but flashing red lights (on turbines) were not involved.	Kerlinger *et al.,* 2010
multiple spp.	USA	2	18 met. towers. 1632 searches	n/a	mortality	no	n/a	85 carcasses were found during searches around the meteorological towers over an 18month period.	Kerlinger *et al.,* 2012

Species	Country	No. of study sites	No. of replicates	No. of controls	Factor	Analysis	Effect	Conclusion	Reference
multiple spp.	Netherlands	3	14 turbines searched every 2 to 3 days over 3 months . 3 nights of radar surveillance at each site to determine flight behaviour	n/a	collision rate	yes	n/a	The collision risk for all species observed at the wind farms was 0.14%. Majority (84%) of birds that flew in risk areas were migratory species.	Krijgsveld *et al.,* 2009
White-tailed eagle (*Haliaeetus albicilla*)	Germany	2	-	-	Mortality	No	Yes	Direct mortality of eagles	Krone & Scharnweber, 2003
Whooper swan (*Cygnus cygnus*)	Denmark	1	-	0	Mortality	No	-	Swans prone to collision with small turbines in poor visibility; larger turbines probably avoided	Larsen & Clausen, 2002
Pink-footed goose (*Anser brachyrhynchus*)	Denmark	2	-	0	Behaviour	Yes	Yes	Geese avoided lines of turbines by 100m and clusters by 200m compounded by associated habitat loss.	Larsen & Madsen, 2000
Multiple (mainly passerine spp.)	USA	1	3	1	Abundance	Yes	Yes	Bird density 4 times lower in windfarm grasslands; linear relationship between density and distance from turbines. Trend for higher densities during non-operational phases.	Leddy *et al*., 1999
Red kite	Switzerland	n/a	n/a	n/a	n/a	yes	n/a	The authors produced a model simulating autumn and spring migration of birds across Switzerland. This was done in order to identify areas that would pose a risk to migratory species	Liechti *et al.,* 2013

Species	Country	No. of study sites	No. of replicates	No. of controls	Factor	Analysis	Effect	Conclusion	Reference
5 species of ducks	USA	4	10.338 wetland visits between 2008 and 2010	2	duck densities	yes	yes	Estimated densities were lower on wetland wind farm sites compared to reference sites. Level of reduction varied widely.	Loesch *et al.*, 2013
Raptor spp.	Global	Review	Unknown	Avoidance	No			Upland species are most at risk due to windy locations and reduction in conflict with human habitation. Insufficient long-term studies, but displacement in raptors appears negligible. Important to use modelling studies to reduce conflict of turbine location.	Madders & Whitfield, 2006
Pink-footed goose (*Anser brachyrhynchus*)	Denmark	2	-	2	Behaviour	Yes	Yes	Geese habituate to turbine presence avoiding turbines by 40-100m.	Madsen & Boertmann, 2008
Golden and Bonelli's eagles	Spain	n/a	128 golden eagle and 152 Bonelli eagle territories	n/a	proximity of territories to proposed turbines	n/a	n/a	There is a low spatial overlap between the proposed wind farms in the study area and the nesting sites of both species of Eagle. Golden eagles are at greatest risk of the two species as most proposed wind farms are in the proximity of territories.	Martinez *et al.*, 2010
Vultures	Spain/USA	1	422 birds tagged - 1195 re-sighted	n/a	fecundity and survival	yes	yes	Fecundity and survival of vultures were negatively affected by the presence of wind turbines. However, this changed when problem turbines were removed.	Martinez-Azbrain *et al.*, 2012

Species	Country	No. of study sites	No. of replicates	No. of controls	Factor	Analysis	Effect	Conclusion	Reference
Eider ducks	UK/Denmark	1	13323 flights of eiders were recorded	n/a	curvature of trajectories	yes	yes	The trajectories of Eider ducks post-construction were greater in relation to the wind farm than post construction. Birds flew an additional 174 from wind turbines post-construction. Space used by eiders also reduced post-construction.	Masden *et al.,* 2009
Birds in general	UK/Denmark	n/a	n/a	n/a	n/a	n/a	n/a	Authors describe the issues surrounding cumulative impacts and propose a conceptual framework to promote transparency in assessing cumulative impacts of wind energy developments.	Masden *et al.,* 2010
Moorland spp.	UK	1	6	1	Abundance	Yes	No	No effect on population trends of ducks, waders, skuas, gulls, passerines or red grouse 8 years post-construction. Decline in red-throated diver but likely to be an artefact of disturbance rather than direct mortality	Meek *et al*., 1993
Multiple spp.	UK	20	354hrs from 178 Vantage point surveys	n/a	bird activity	yes	no	There was no evidence that bird activity was affected by turbine proximity or turbine operation over several distances up to 30m.	Minderman *et al.*, 2012
Estuarine spp.	Netherlands	1	5	0	Mortality	Yes	Yes	Direct mortality consistent throughout the year and correlated with number of birds present	Musters *et al*., 1996
Multiple spp.	UK	1	searches made 2 days once a week for 11 years	n/a	mortality	no	n/a	The authors searched a beach adjacent to an onshore wind farm and recorded the species found dead over an 11 year period.	Newton & Little

Species	Country	No. of study sites	No. of replicates	No. of controls	Factor	Analysis	Effect	Conclusion	Reference
7 species	USA	4	10321 visits to wetlands	2	presence and abundance of species	yes	no	Wetland birds were surveyed at wind farm and non-wind farm sites. There was no evidence for avoidance in 4 of the species studied. Authors suggest wetlands with wind turbines retain conservation value.	Niemuth *et al.,* 2013
Raptors	Spain	1	50 linear transects and 20 minute observations in each cell (1km2) repeated every 2 weeks in one year period.	n/a	n/a	n/a	n/a	The authors developed a sensitivity score for raptors in Spain based on flight behaviour and other attributes. They found the sensitivity of raptor species to wind farms varied.	Noguera *et al.,* 2010
White-tailed sea eagle	Norway	1	50 fledglings satellite tagged	n/a	n/a	no	n/a	The authors satellite tagged white tailed eagle fledglings and found that birds used airspace inside and outside the wind farm. Four of the tagged birds were killed by the turbines. The authors suggest displacement was negligible and that avoidance rates of Golden eagles are not suitable for WTSEs.	Nygard *et al.,* 2010
Multiple (mainly raptor, duck, geese & passerine spp.)	USA	1	414 focal observations	2	Abundance & mortality	Yes	Yes	Significant difference in species occurrence and relative abundance between operational windfarms and proposed windfarm sites. 85% avoidance of operational turbines. Raptors (notably kestrels) and waterfowl were at greatest risk of direct morality due to collision.	Osborn *et al.*, 1998

Species	Country	No. of study sites	No. of replicates	No. of controls	Factor	Analysis	Effect	Conclusion	Reference
12 species	UK	12	Six visits at each wind farm, 3 visits to control sites.	12	bird occurrence	yes	yes	7/12 species had lower frequencies of occurrence close to turbines (after accounting for habitat variation). Equivocal evidence of avoidance in another 2 species was also found. Breeding bird densities were reduced within 500m of turbines.	Pearce-Higgins *et al.,* 2009
10 upland spp.	UK	18	two to three bird surveys per site	12	population density	yes	yes	Densities of 5/10 species changed during construction and 4 /10 post-construction. Red grouse and curlew densities declined during construction. Skylark and Stonechat densities were greater during and post-construction.	Pearce-Higgins *et al.,* 2012
Multiple spp.	Global	Review	Unknown	Review	NA			Main effects are collision and avoidance of windfarms and surrounding area. No apparent significant effect in UK and provided each development considers and avoids i) high density raptor populations, ii) high densities of species vulnerable to additive models	Percival, 2005
Lesser and greater prairie chickens	USA	2	463 lesser (over 7 years) and 216 (over 3 years) greater prairie chickens tracked.	n/a	avoidance	yes	n/a	Low levels of mortality of these species recorded at power lines and none recorded for wind farms. Both species avoided areas with powerlines and lesser chickens avoided roads. Suggest these features associated with wind farms may fragment habitat.	Pruett *et al.,* 2009

Species	**Country**	**No. of study sites**	**No. of replicates**	**No. of controls**	**Factor**	**Analysis**	**Effect**	**Conclusion**	**Reference**
Swans and geese	UK	n/a	n/a	n/a	n/a	n/a	n/a	A total of 72 swans and geese have been reported killed by offshore and onshore turbines. Flight avoidance has been recorded for a variety of species and displacement has been recorded but at varying distances.	Rees, 2012
Scissor tailed flycatchers	USA	1	32 nests	n/a	success	n/a	no	Flycatchers nesting closest to turbines with less canopy cover over nests had the highest nesting success. Overall reproductive success of flycatchers at the wind farm was low.	Rubenstahl *et al.*, 2012
Vertebrates including multiple bird spp.	Portugal	4	198	0	Species richness	Yes	Yes	Lower vertebrate species richness, including birds, associated with windfarms probably due to direct disturbance, structural habitat changes and induced behavioural segregation.	Santos *et al*., 2010
Birds in general	Portugal	1	198 quadrats surveyed	0	species richness	yes	yes	The authors created a model to predict trends in vertebrate species richness in response to changes in environmental conditions (habitat structure) by wind farm installation. The authors found that wind farms had a negative effect on species richness of vertebrates.	Santos *et al*., 2010
Red kite	Switzerland	n/a	Simulations were run 1000 times	n/a	population dynamics of red kite population	yes	yes	Simulations revealed that red kite populations decreased over time with increasing number of individual turbines and the greater the aggregation of turbines the lower the impact on red kite population growth rates.	Schaub, 2012

Species	Country	No. of study sites	No. of replicates	No. of controls	Factor	Analysis	Effect	Conclusion	Reference
Multiple spp.	USA	1	4074 turbines	0	Mortality	No	Yes	Direct mortality (67 golden eagles, 349 kestrels, 440 burrowing owls, 1127 raptors and 2710 other birds per annum).	Smallwood & Thelander, 2008
Multiple spp.	USA	1	multiple studies	n/a	mortality	no	yes	Comparison of fatality rates of old-generation turbines (1998-2003) with fatality rates of repowered turbines (2005-2007) at APWRA led to increased bat fatalities but reduced bird fatalities.	Smallwood & Karas, 2009
Burrowing owl (*Athene cunicularia*)	USA	Multiple	28 (4074 turbines)	0	Mortality	Yes	Yes	Direct mortality greatest in winter and associated with cattle grazing and ground squirrel abundance. 29% of turbines caused 71% of mortality.	Smallwood *et al.*, 2007
Multiple spp.	USA	1	65 burrows and 10 mortalities - sampling effort not clearly defined	n/a	n/a	yes	n/a	The model developed corresponded well to actual mortalities and the distribution of burrows of target species.	Smallwood *et al.*, 2009
Multiple spp.	USA	1	28	0	Behaviour	Yes	Yes	Before-after design. Species-specific morality rates. Non-operating turbines used regularly for perching	Smallwood *et al.*, 2009

Species	Country	No. of study sites	No. of replicates	No. of controls	Factor	Analysis	Effect	Conclusion	Reference
Multiple spp.	USA	NA	NA	Review of methods to assess mortality	NA			Review of methods and biases used for calculating and correcting mortality estimates. Searcher detection trials are biased by species and the placement and position of carcasses by trial participants. Scavenger trials can be affected by the number of carcasses, the species used, frozen/fresh carcasses, and intactness of carcass, season, and distance from turbine. Models derived from other studies can increase rigour of future studies	Smallwood, 2007
birds	Singapore	n/a	n/a	n/a	mortality caused by renewable energy	no	n/a	This study suggests that wind power in the US responsible for between 0.3 and 0.4 "avian" fatalities per GWh.	Sovacool, 2009
birds in general	USA	n/a	n/a	n/a	n/a	n/a	n/a	The author updates estimates of bird mortality caused by energy generation.	Sovacool, 2012
Multiple (mainly wader & diving duck spp.)	Netherlands	1	-	0	Behaviour	Yes	Species dependent	Bird detecting radar showed that duck species avoided turbines with mortality linked to poor visibility. Foraging/roosting birds (e.g. curlew) avoided turbine up to 500m. Breeding birds unaffected.	Spaans *et al*., 1998; Dirksen *et al*., 1998
Multiple spp.	USA	1	93 1ha plots in 2009/10 and 117 in 2010/11	random placement of survey plots	habitat use by birds	yes	yes	The authors investigated habitat use and displacement of 3 bird species around wind turbines in a wind farm. They showed that displacement was species specific and the effects depended on the statistical method used.	Stevens *et al.,* 2013

Species	**Country**	**No. of study sites**	**No. of replicates**	**No. of controls**	**Factor**	**Analysis**	**Effect**	**Conclusion**	**Reference**
Multiple spp.	Global	Review	19 studies	Review & meta-analysis	Yes			Meta-analysis of taxon, turbine number, power, location, latitude, habitat, WF area, operational time, species status and study design indicates that bird abundance is significantly affected by the number of turbines, power of turbines, and time since operation.	Stewart *et al*., 2007
Common tern (*Sterna hirundo*)	Belgium	1	-	0	Mortality	No	Yes	Sex biased mortality; greater numbers of male terns killed. Possible risk of population decline.	Stienen *et al*., 2008
Multiple (mainly wetland & raptor spp.)	UK	1	-	0	Behaviour	Yes	No	Birds avoided turbines including in bad visibility with low levels of mortality. Cormorants displaced during construction phase only.	Still *et al*., 1997
Geese	Japan	1	n/a	n/a	mortality and collision risk	yes		The collision risk of geese was assessed at a recently built wind farm. The modelling suggested 0 to 2 geese would be killed a year and permissible amount was 75.	Sugimoto & Matsuda, 2011
Golden eagles	Spain	1	101 10km squares.	n/a	occurrence	yes	no	The authors modelled the risk posed to golden eagles by the presence of wind farms, number of turbines and prospective wind farms. There was no relationship between current developments but there was with future developments and the occurrence of golden eagles.	Tapia *et al.,* 2009
9 bird species	Spain	n/a	269 10 km squares with wind plants and 5174 without	n/a	spatial overlap and coincidence	yes	yes	Spatial analysis revealed that the occurrence of wind power plants in Spain overlapped with the richest areas of birds. There were more wind farms than expected in the range of two species of birds. 44% of bird hotspots were located within 30km of wind farms.	Telleria, 2009

Species	**Country**	**No. of study sites**	**No. of replicates**	**No. of controls**	**Factor**	**Analysis**	**Effect**	**Conclusion**	**Reference**
Wood pigeons (Columba palumbus)	Spain	n/a	165 wood pigeons	n/a	n/a	n/a	n/a	The author used wood pigeons as a model species to examine risk posed to migratory species by wind turbines in Spain. They showed where mortality in migratory species occurred in relation to turbines in Spain.	Telleria, 2009
Griffon vulture (*Gyps fulvus*)	Spain	Multiple	-	-	Spatial analysis	Yes	Yes	Breeding ranges overlapped with windfarms. Perceived as high risk.	Telleria, 2009b
Multiple spp.	Canada	3	radar surveys for 3-6 nights a week in 2006 and 2007. Also 2 VP surveys at each site and dawn and dusk activity surveys.	n/a	migratory passage rates	yes	yes	The authors examined how weather affected passage rates of spring and autumn migrants. Higher wind speeds were associated with higher activity of spring and autumn nocturnal migrants. Diurnal migrants fly lower than nocturnal migrants and flew more on warmer days with low cloud cover with wind from a westerly direction.	Thomas *et al.,* 2011
Raptor spp.	USA	1	43	0	Behaviour	No	Yes	Avoidance suggested as no nests were located within the windfarm site despite availability of suitable habitat.	Usgaard *et al*., 1997
Multiple (tern & gull spp.)	Netherlands	2	6	0	Behaviour	Yes	No	Avoidance during winter but no effect on foraging or commuting in the breeding season	van den Bergh *et al*., 2002
Multiple spp.	Mexico	1	60 carcasses per season (2 seasons)	n/a	carcass persistence rate	yes	yes	The authors looked at the persistence of bird and bat carcasses at a wind farm in the rainy and wet seasons. Carcasses were scavenged by mammals, birds and arthropods. Average removal time of small bird carcasses was 2.7 to 4.4 days and 2.1 to 4.4 days for large birds.	Villegas-Patraca *et al.,* 2012

Species	Country	No. of study sites	No. of replicates	No. of controls	Factor	Analysis	Effect	Conclusion	Reference
Multiple spp.	Mexico	1	24 points were surveyed in each of the following: wind turbines; 200m away from wind turbine; secondary forest and croplands	n/a	Diversity, composition and structure. Of bird communities	yes	yes	Species richness was lowest at the Wind farm points and highest in cropland and secondary forest. Bird abundance was highest in secondary forest and cropland and lowest at wind farm and 200m from wind farm.	Villegas-Patraca *et al.,* 2012
Tern & gull spp.	USA	1	-	0	Behaviour	Yes	Yes	Breeding terns flew within 50m of turbines during chick rearing period but at low wind speeds and only infrequently	Vlietstra, 2007
Golden eagle (*Aquila chrysaetos*)	UK	1	1	0	Mitigation	Yes	Yes	Windfarm avoided in preference for mitigation area provided. No effect on range size.	Walker *et al*., 2005
Multiple spp.	Netherlands	6	341 hours focal observation	0	Behaviour	Yes	Yes	Evasive manoeuvres in 97-100% of species and 7-19% of flocks. Barrier effect evident. No direct mortality.	Winkelman, 1985
Waterbirds	Netherlands	1	-	0	Behaviour	Yes	Species dependent	Decreased occurrence of mallard, tufted duck, pochard and goldeneye up to 300m from turbines. No effect on great-crested grebe, coot and gulls. Increased numbers of black-headed gull and scaup on windfarm. Displacement of whooper swans.	Winkelman, 1989
Waders & waterbirds	Netherlands	1	-	0	Abundance & behaviour	Yes	No	No effect on distribution or breeding numbers of oystercatcher, lapwing, black-tailed godwit or redshank.	Winkelman, 1992a

Species	Country	No. of study sites	No. of replicates	No. of controls	Factor	Analysis	Effect	Conclusion	Reference
Multiple spp.	Netherlands	1	-	0	Behaviour	Yes	Species dependent	Mallard, common gull and oystercatcher avoided construction phase. Curlew avoided operational turbines up to 500m. Lesser effect on gulls, ducks or waders. No effect on starlings, corvids or black-headed gulls.	Winkelman, 1992b
Multiple spp.	Canada	43	n/a	n/a	mortality and loss of habitat	n/a	n/a	The authors assessed mortality on Canadian bird populations by using post-construction carcass search data from across the country. They compare mortality to other causes and doubt wind farms will have population level impacts.	Zimmerling *et al.*, 2013

Appendix 2 A summary of studies on the impact of wind energy developments on bats cited in the main text of the review.

Species	Country	No. of study sites	No. of replicates	No. of controls	Factor	Analysis	Effect	Conclusion	Reference
bats	Portugal	1	20 turbines	n/a	mortality	yes	yes	Authors looked at activity at mortality at a single wind farm in Portugal. They found activity was positively correlated to temperature, wind speed and relative humidity. 94% of mortality occurred between august and October at temps higher than 13C and at wind speeds lower than 5m/s.	Amorim *et al.,* 2012
bats	USA	1	3 treatments repeated 5 times	4 in each replicate	mortality	yes	yes	Mean mortality at control turbines (cut in speed = 3.5m/s) was 5.4 times greater at turbines with greater cut in speeds (5.0m/s and 6.5m/s). There was no difference in mortality between cut in speed treatments.	Arnett *et al.,* 2011
bats	USA	n/a	n/a	n/a	n/a	n/a	n/a	The authors comment on the Canadian province of Ontario's decision to initiate thresholds for curtailment based on the annual mean number of bats killed per turbine (max 10). The authors do not support this move.	Arnett *et al.,* 2013
bats	USA	2	10	15	mortality	yes	yes	The authors installed ultrasonic acoustic deterrent at wind turbines in order to reduce mortality. Mortality at control turbines was higher than the experimental turbines but this varied between year and species. Authors suggest these may work but have limitations.	Arnett e*t al.,* 2013

Species	Country	No. of study sites	No. of replicates	No. of controls	Factor	Analysis	Effect	Conclusion	Reference
Hoary (*Lasiurus cinereus*), Silver-haired bats (*Lasionycteris noctivagans*)	Canada	9 (7)	309 turbines for mortality data and (7 sites for activity)	n/a	mortality and activity	YES	YES	Bats were found to be migrating at sites located near to foothills of mountain ranges. Mortality increased with turbine height and increased with turbine height and migratory activity at 30m. Mortality was positively related with activity at 5 sites. Mortality varied between the sites, partly due to turbine height and differences in activity of migratory bats	Baerwald & Barclay, 2009
Hoary (*Lasiurus cinereus*), Silver-haired bats (*Lasionycteris noctivagans*)	Canada	1	not clear: 8 detectors in pairs used but number of sampling points not stated.	n/a	activity AND fatality	yes	yes (activity and activity)	Bat mortality highest in July and August when species studied migrating. Level of mortality was positively related to migratory bat activity and increased moon illumination. Bat activity was highest at 30m but differed for species. Activity was also highest during migratory period of July to September. Activity of bats was greatest at low wind speeds and lower when wind from north or northeast.	Baerwald & Barclay, 2011
Lasiurus cinereus & Lasionycteris noctivagans	USA	1	75 bats	0	Mortality	Yes	Yes	Mortality due to direct impact of rotor (10%) and barotrauma (90%)	Baerwald *et al.*, 2008
Hoary (*Lasiurus cinereus*), Silver-haired bats (*Lasionycteris noctivagans*)	Canada	1	188 bats collected, 89 post-mortem necropsy	n/a	n/a	no	n/a	Signs symptomatic of barotrauma were found in over 90% of 88 bats necropsied in the field	Baerwald *et al.*, 2009
bats	Canada	1	15 turbines start up speed 5.5 m/s, 6 additional turbines a low speed idle strategy	8 control turbines	mortality	YES	YES	The implementation of measures to reduce the time blades were rotating at low wind speeds resulted in lower levels of bat mortality at experimental turbines compared to the controls where no measures were implemented.	Baerwald *et al.*, 2009

Species	Country	No. of study sites	No. of replicates	No. of controls	Factor	Analysis	Effect	Conclusion	Reference
bats	Portugal	n/a	n/a	n/a	n/a	n/a	n/a	The authors tested the applicability of a Stochastic Dynamic Methodology to estimate bat fatalities. The authors suggest this approach will improve the quality of mortality estimates at onshore wind energy facilities.	Bastos *et al.*, 2013
bats	Portugal	n/a	n/a	n/a	n/a	n/a	n/a	The authors review 7 estimators widely used to estimate the mortality of birds and bats at wind energy facilities. The authors suggest that due to variability of the methods employed to inform the estimators that until a universal estimator is created each case study should use the most appropriate methods and estimator.	Bernardino *et al.*, 2013
bats	USA	1	12 turbines	n/a	mortality	no	n/a	The authors conducted 6 surveys at 12 turbines over the space of 3 months and found 9 bats of 2 species dead.	Bicknell & Gillam, 2013
bats	USA	n/a	n/a	n/a	Review	72 cited references	n/a	Authors review hypotheses that explain bat mortality at wind turbines and suggest further work is required to test the different hypotheses.	Cryan & Barclay, 2009
Lasiurus cinereus	USA	1	295 days observations over 38 years	0	Behaviour	No	Yes	Attraction to tall landscape features including turbines; natural environmental parameters used to predict migration patterns in relation to wind farms	Cryan & Brown, 2007
Hoary (Lasiurus cinereus), Silver-haired bats (Lasionycteris noctivagans) and eastern red bats (L. borealis)	USA/Canada	9	opportunistic sampling between 2007 and 2011	n/a	n/a	no	n/a	The authors examined the breeding condition of bats found dead at turbines. The majority of bats found were male and they were in breeding condition.	Cryan *et al.*, 2012
bats	USA	n/a	n/a	n/a	n/a	n/a	n/a	The author discusses the mortality of migratory bats at wind turbines in a legal context and explains that those species killed are not protected by US law.	Cryan, 2011

Species	**Country**	**No. of study sites**	**No. of replicates**	**No. of controls**	**Factor**	**Analysis**	**Effect**	**Conclusion**	**Reference**
bats	South Africa	1	1 turbine	n/a	mortality	no	n/a	This study monitored mortality over a year at a single wind turbine. They also conducted carcass scavenger experiments. They report the first mortalities (18) of bats at a wind turbine in sub-Saharan Africa.	Doty & Martin, 2013
bats	USA	n/a	n/a	n/a	n/a	yes	n/a	The authors set forth to identify areas suitable for wind energy development that would have minimal impact on wildlife receptors in the Northern Great Plains, USA. This was done using spatial data.	Fargione *et al.*, 2012
bats	Italy	2	46 turbines searched	n/a	n/a	no	n/a	The authors found 7 bat carcasses during searches of two wind farms. These are the first reported instances of bat mortality at a wind farm in Italy.	Ferri *et al.*, 2011
bats	Greece	9	88 turbines searched	n/a	n/a	n/a	n/a	The authors searched 88 turbines at 9 facilities in Greece between 2009 and 2010. They found 181 dead bats over this period. Leisler's were the species found most often dead followed by P.pygmaeus/pipistrellus and P. nathusii.	Georgiakakis *et al.*, 2012
5 species - most hoary and silver haired bat	USA	1	41 bats found (39 radiographed, 33 necropsied, 28 histopathology)	n/a	n/a	no	n/a	Suggest that Baerwald et al 2009 were premature to declare that Barotrauma main cause of death of bats at wind turbines due to indiscernable combination of signs of barotrauma and collision.	Grodsky *et al.*, 2011
bats	USA	1	29 wind turbines (277 days of searches over 2 years)	n/a	mortality	yes	yes	A total of 122 dead bats were found during the study leading to an estimated mortality rate of 4454 over the survey period. Mortality was greatest in August to September and was positively associated with higher temperatures.	Grodsky *et al.*, 2012

Species	Country	No. of study sites	No. of replicates	No. of controls	Factor	Analysis	Effect	Conclusion	Reference
bats	USA	21	n/a	n/a	mortality	yes	n/a	The author estimated 600,000 bats were killed at wind energy facilities in the USA in 2012. This is similar to results produced from other authors using other methods.	Hayes, 2013
Unknown	USA	1	19 (171 hours focal observations)	0	Behaviour	Yes	Yes	Insect activity greater at lighted turbines; bat activity influenced by rotor speed; no effect of turbine lighting	Horn *et al.* 2006
Unknown	USA	1	19 (171 hours focal observations)	0	Mortality	Yes	Yes	Mortality due to direct collisions; 21 bats/wind farm/year correlated with blade vortices during low wind conditions	Horn *et al.*, 2008b
bats	Australia	2	62 turbines in year 1 and 16 in year 2.	n/a	mortality	yes	yes but small due to sample size	Bat mortality at two wind farms in Tasmania was monitored over a two year period. 75 bats were found. They found that a higher number of bats were found dead in March and April.	Hull & Cawthen, 2013
bats	Australia	n/a	n/a	n/a	modelling	yes	n/a	The authors model the "fall-zone" of birds and bats colliding with different sized turbines. The authors suggest the size of the search area is important to estimate levels of mortality.	Hull & Muir, 2010
bats	USA	n/a	n/a	n/a	n/a	n/a	n/a	The author introduces a new estimator that simulations show that it performs better than two older estimators Schoenfeld and Johnson.	Huso, 2011
bats	USA	1	14 (activity), 26 (mortality)	14 (activity)	bat passes	yes	no	There was no different in bat activity at wind turbines and adjacent crop fields (control sites). Dead bats were found in both years of the study (2003, 2004) and 83% and 67% of dead bats found and in good enough condition to examine showed signs of trauma.	Jain *et al.*, 2011
Multiple spp.	USA	2	77	16	Mortality	Yes	Yes	Mortality greatest in migratory than resident species, notably greater in	Johnson *et al.*, 2004

Species	Country	No. of study sites	No. of replicates	No. of controls	Factor	Analysis	Effect	Conclusion	Reference
								autumn.	
Hoary (Lasiurus cinereus), Silver-haired bats (Lasionycteris noctivagans) and eastern red bats (L. borealis)	USA	14	2 detectors (20m and 40m) on met masts at each site. Number of survey days not consistent between site.	n/a	activity	yes	yes	The authors found correlations between bat activity and mortality at a nearby turbine site. They also obtained evidence that suggested acoustic detection rates were good indicators of seasonal movements of the species studied.	Johnson *et al.,* 2011
10 bat species	USA	1	3	n/a	bat migration activity	yes	yes	The authors monitored bat activity on an island to inform the future development of offshore and onshore wind power development. The authors found bat activity higher on nights with higher temperatures and low wind speeds. Additionally bat activity was higher during the migratory period.	Johnson *et al.,* 2011
bats	USA	2	unknown	n/a	Daily Fatality index	yes	yes	The authors studied mortality at 2 wind farms in Texas. Mortality was better explained by wind speed in combination with wind direction. Compared to wind speed on its own	Karsten & Hale, 2012
bats	Switzerland	n/a	n/a	n/a	n/a	n/a	n/a	The author introduces a new mortality estimator that accounts for the bias resulting from carcass removal by predators.	Korner-Nievergelt, 2011
bats	Switzerland	n/a	n/a	n/a	n/a	n/a	n/a	The authors combined carcass search data with animal density indices to investigate collision rates. Their results suggest that acoustic bat activity and wind speed can be used as predictors of collision rates. The method correlated well with traditional estimators.	Korner-Nievergelt., 2013

Species	**Country**	**No. of study sites**	**No. of replicates**	**No. of controls**	**Factor**	**Analysis**	**Effect**	**Conclusion**	**Reference**
bats	India	1	51 turbines searched over a year	n/a	mortality	no	no	Two bats were found dead during searches conducted at a wind farm in India. This is the first records of bat mortality at a wind farm in India.	Kumar *at al.,* 2013
bats	UK	n/a	n/a	n/a	How bats echolocation respond when they hit blades			The authors used a simulated FM bat pulse to determine how bat calls behaved when "fired" at rotating turbine blades both laterally and horizontally at a distance of 0.5m and 1.0m. All horizontal calls were returned to emitter but lateral calls at 1m were not returned to the emitter.	Long *et al., 2010*
bats	UK	1	59 (10 minute sessions)	1 in each session	Insect count	yes	yes	Colour was found to have a significant effect on insect count. White and light grey (common colours for turbines) were significantly more attractive than the other colours (except yellow) and purple was least attractive.	Long *et al.,* 2011
bats	UK	20	2 X 20 (24 hr periods)	40 (24 hr periods)	bat activity	yes	yes	No overall effect of operating Small wind turbines on bat activity. However, activity was lower when turbines were running. Bat activity decreases with increasing wind speed when the turbines are running but not when they are braked. The decrease is substantial at short distances but declines with distance.	Minderman *et al.,* 2012
bats	Germany	10	10 (5 sample heights for 10 nights at each location)	n/a	activty of bat guild	yes	yes	The authors decided to monitor bat activity at montane forests in Germany, areas that are increasingly being considered for wind farm development. They found that surveying under the canopy missed species foraging above the canopy plus those that foraged above the canopy were are greatest risk from any wind farm development e.g. Nyctaloid and Pipistrelle bats	Muller *et al.,* 2013

Species	Country	No. of study sites	No. of replicates	No. of controls	Factor	Analysis	Effect	Conclusion	Reference
bats	UK	20	20h rotating antennae, 20h fixed pulse rate 1, 18h fixed pulse rate 2	58 hours	Index of bat activity (passes, feeding buzzes and counts)	YES	YES but ns	An electromagentic signal from a radar unit with a fixed antennae reduced bat foraging activity within 30m but the result was not significant. Activity was not significantly reduced with the rotating radar.	Nicholls & Racey, 2009
bats	USA	n/a	n/a	n/a	n/a	n/a	n/a	The authors identify areas to avoid developing wind energy facilities in Kansas by using a number of spatial data sets (habitat, land cover etc). They also develop methods to quantify the impacts of wind developments that need to be offfset.	Obermeyer *et al.*, 2011
bats	n/a	n/a	n/a	n/a	n/a	n/a	n/a	The authors talk about the issues surrounding wind turbines and wild life and talk about the situation in New Zealand whilst also talking about the other papers in the special edition of the New Zealand Journal of Zoology.	Parsons & Battley, 2013
bats	USA	n/a	n/a	n/a	n/a	n/a	n/a	The authors describe the use of superpopulation capture-recapture models based on carcass searches to estimate mortality at wind turbines. The method is flexible and can accommodate a range of sampling designs.	Peron *et al.*, 2013
7 bat species	USA	1	68 turbines searched 6 times over 2 years	n/a	mortality	no	n/a	Bat carcasses were found at 68% of turbines and in 2004 mortality of Brazilian free tailed bats were positively associated with ravine topography. Three turbines were hotspots for mortality - located near the head of a forested ravine.	Piorkowski & O'Connell, 2010

Species	Country	No. of study sites	No. of replicates	No. of controls	Factor	Analysis	Effect	Conclusion	Reference
bats	USA	n/a	n/a	n/a	n/a	n/a	n/a	In 2009 30 scientists convened to identify key research priorities concerning the impacts of wind farm development on wildlife. These were: 1. need to standardise work, 2. need for new models to assess risk, 3. increased documentation of mortality at existing wind farm, 4. And data	Piorkowski *et al.,* 2012
bats	USA	1	262 bats from wind farms	52 bats from non-wind farm site	Various	yes	yes	This study found that freezing animals and post-mortem decomposition cause tissue damage that is similar to signs indicative of barotrauma. They therefore suggest that previous claims of barotrauma being the primary cause of death of bats and wind farms untrue due to this and the multiple injuries sustained by bats at wind turbines.	Rollins *et al.,* 2012
bats	Italy	n/a	n/a	n/a	n/a	n/a	n/a	The authors used Species Distribution Models for Leisler's and Common pipistrelles and overlayed these with current and proposed turbine locations in Italy. 41% of the region modelled offers suitable foraging habitat for bats and 50% of these areas host existing or planned wind farms.	Roscioni *et al.,* 2013
bats	Sweden	80 cited references	n/a	n/a	Review	no	n/a	The authors reviewed bat mortality in Europe and compared to results from the US. Levels of bat mortality were lower than US, taller turbines killed more bats as did those on coastal areas and ridge tops. Nyctalus and Pipistrellus species are at greatest risk.	Rydell *et al.,* 2010
bats	Sweden	41 cited references	n/a	n/a	Review	no	n/a	Authors suggest that insect attraction to wind turbines and subsequent attraction of bats may be contribute to observed bat mortality but is unlikely to be sole explanation.	Rydell *et al.,* 2010

Species	Country	No. of study sites	No. of replicates	No. of controls	Factor	Analysis	Effect	Conclusion	Reference
Common pipistrelle, Leisler's, Kuhl's pipistrelle and Hypsugo savii.	Portugal/UK	n/a	21 mortality sites and 75 with no mortality data	n/a	n/a	n/a	n/a	The authors combined Species Distribution Models with mortality data and ecological data from wind farms in Portugal to create mortality risk models. Wind farms in humid areas with mild temperatures, closer than 5km to forested areas and within 600m of steep slopes showed higher probabilities of mortality.	Santos *et al.,* 2013
bats	USA	1	multiple studies	n/a	mortality	no	yes	Comparison of fatality rates of old-generation turbines (1998-2003) with fatality rates of repowered turbines (2005-2007) at APWRA led to increased bat fatalities but reduced bird fatalities.	Smallwood & Karas, 2009
bats	Singapore	n/a	n/a	n/a	mortality caused by renewable energy	no	n/a	This study classes birds an bats as "avian" and thus suggests that wind power in the US responsible for between 0.3 and 0.4 "avian" fatalities per GWh.	Sovacool, 2009
bats	USA	n/a	n/a	n/a	n/a	n/a	n/a	The authors of the work developed a likelihood-based approach for modelling heterogeneity in count data from wildlife collision mortality surveys.	Stevens & Dennis, 2013
9 bat species	Spain	n/a	269 10 km squares with wind plants and 5174 without	n/a	spatial overlap and coincidence	yes	yes	Spatial analysis revealed that the occurrence of wind power plants in Spain did not overlap the richest areas of bats except in the north. Also, 54% of bat hotspots were within 30km of wind power plants.	Telleria, 2009
bats	Mexico	1	60 carcasses per season (2 seasons)	n/a	carcass persistence rate	yes	yes	The authors looked at the persistence of bird and bat carcasses at a wind farm in the rainy and wet seasons. Carcasses were scavenged by mammals, birds and arthropods. Average removal time of bat carcasses was 2 days irrespective of season.	Villegas-Patraca *et al.,* 2012

Species	Country	No. of study sites	No. of replicates	No. of controls	Factor	Analysis	Effect	Conclusion	Reference
Nyctalus noctula, Nyctalus leislerii, P. pipistrellus, P. Nathusii	Germany	5	47 carcasses sampled	Deuterium from 178 bats from across Europe used	Deuterium in hair	yes	yes	The authors collected hair from bat carcasses collected in Germany and used stable isotopes to demonstrate they were not from populations in Germany but had migrated from more northern latitudes thus demonstrating that migrating species in Europe are also at risk from mortality.	Voigt *et al.*, 2012
bats	USA	1	12 towers holding 28 detectors divided between 2 plots	0	bat passes	yes	n/a	Authors built a model that successfully predicted conditions when Low F bats were present on the turbine site. Bat activity was higher at lower wind speeds and higher temperatures but the size of the effect varied between season and on moon phase.	Weller & Baldwin, 2012

ISSN 1751-7796 (Online)