# PhD Thesis

# The effect of wind turbines on bats in Britain

Submitted by Suzanne Mary Richardson, to the University of Exeter as a thesis for the degree of Doctor of Philosophy in Biological Sciences, November 2015.

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## Summary

The increase in wind energy production has been relatively rapid and is expected to continue at a global scale. However, numbers of bat carcasses found at wind turbines in North America in the early 21st century raised concern about the plight of this taxon with the growth in wind-energy generation. This led to carcass searches for bats becoming commonplace at wind farms globally. However, few large scale systematic studies have assessed the effects of wind turbines on bats, especially for species considered potentially at higher risk in Europe. In this thesis the number and species of bats killed from wind farms were estimated across Britain, and the important predictors (i.e. activity, turbine characteristics and habitat) of fatality were determined. Insect abundance, biomass and bat activity was also quantified at turbine and control locations, to assess if insects and hence bats were attracted to turbines. In addition, assessments were made of the effects of increasing temporal and spatial replication of acoustic monitoring on estimates of species composition and bat activity. This was assessed for activity monitored at ground and at the centre of the rotor sweep area (the nacelle).

Carcass searches were conducted using trained search dogs and concurrently bats were surveyed acoustically at three randomly selected turbines at ground and from the nacelle at 48 wind farms throughout Britain. Bats were also monitored acoustically at paired controls (with a randomly selected turbine) at 20 of the wind farms sites. In addition, nocturnal Diptera were sampled at 18 of the sites using a paired turbine and control design.

Across 139 wind turbines, 188,335 bat passes were recorded and 2,973 carcass searches performed. Edge and open aerial foraging species, in particular *Pipistrellus pipistrellus* and *P. pygmaeus* were most at risk of fatality

at wind farms in Britain. The number of *Pipistrellus pipistrellus* killed annually in Britain between mid-July and mid-October was estimated at 2,373 95% CI 513 to 4,233 and the number of *P. pygmaeus* at 3,082 95% CI 1,270 to 4,894*.* When compared to population estimates, the number of *Pipistrellus pygmaeus* killed was 57% higher than the number of *P. pipistrellus* killed (0.19% of the population versus 0.43%, respectively). This may be due to *Pipistrellus pygmaeus* flying more often within the rotor sweep area compared to *P. pipistrellus*.

Activity measured at the nacelle, which is generally assumed to be a better predictor of fatalities, was not a significant predictor of the probability of a fatality for all species combined, *Pipistrellus pipistrellus,* or *P. pygmaeus*. *Pipistrellus pipistrellus* activity and *P. pygmaeus* activity, measured at ground level were not good predictors of their respective fatalities. Whilst there was some evidence that *Pipistrellus pipistrellus* and *P. pygmaeus* activity monitored at ground level, was a significant predictor of the probability of their respective fatalities occurring, across wide ranging turbine types, fatality estimates were large. This is presumably due to the importance of turbine characterises (the wind speed that turbines become operational (cut-in speeds) turbine and the distance between the ground and blade tip at the bottom of the rotor sweep area) both being important negative predictors of fatalities for *Pipistrellus pipistrellus*. Predicting from models, if the cut-in speed is increased from 3.5 to 5 m $s^{-1}$ the number of *Pipistrellus pipistrellus* fatalities would be reduced by 76% (0.23 fatalities per turbine per month to 0.06). These findings have important implications for guidance, since activity is the ubiquitous measure used to assess fatality risk for all species. Since, *Pipistrellus pipistrellus* and *P. pygmaeus* were detected at 98% and 92% of sites respectively; it could be assumed that these species would be detected at the majority of wind farms within their range. Therefore, in a British context, curtailing wind turbines below 5 m $s^{-1}$ could be an effective mitigation strategy without the costly requirement to monitor activity.

*Pipistrellus pipistrellus* and *P. pygmaeus* activity was 46% (6.3 ± 1.3 SE mean passes per night *c.f.* 3.4 ± 1.3 SE) and 34% (4.0 ± 1.4 SE *c.f.* 2.7 ± 1.4 SE) higher at turbines compared to controls, respectively. Given that habitat and elevation were consistent between paired turbines and controls and monitoring was conducted on the same nights, higher activity at turbines compared to controls provides evidence that these two species are attracted to wind turbines. Furthermore, since the biomass of nocturnal Diptera, the main insect prey for *Pipistrellus* spp., was higher at controls compared to turbines, and bat foraging at turbines was not predicted by insect abundance or biomass, attraction is unlikely to be due to insects. Evidence presented here shows that bats are attracted to turbines, and therefore measuring activity at pre-construction sites for environmental impact assessments is unlikely to be effective.

In conclusion, these results provide further evidence that common species are killed but generally in relatively low numbers, they also support the view that monitoring activity for assessing fatality risk at wind farms is ineffective. It is imperative that wind energy is developed using an evidence based approach. However, it also important that wind energy continues to contribute to an increasing renewable energy sector. In conclusion, results presented here, support that wind turbines are likely to be having a small impact on bat populations in Britain.

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## Acknowledgements

Firstly, I would like to thank my supervisor Prof. Fiona Mathews who has provided much support and encouragement during the PhD. I would like to thank my second supervisor, Prof. Dave Hosken, who has given valuable comments on the thesis, as well as important support and advice. I am very grateful to Dr. Paul Lintott who made very useful contributions and comments on the thesis and Dr. Theo Economou who provided statistical advice.

The research was funded by Department for Environment, Food and Rural Affairs, Department of Energy & Climate Change, Natural England, Natural Resources Wales, Scottish Natural Heritage, and RenewableUK. I would like to thank the site owners and operators who allowed access to the wind farm sites and made this research possible. I am indebted to the exceptional hard work from all my field assistants: Adam Rhodes, Amy Campbell, Amy Huff, Cate Jackson, Cecily Goodwin, Charlotte Marshall, Chris Kaighin, Jacelyn Hawkey, Joanna Kohyt, Kath Evans, Luke Gibbons, Marta Vondrova, Tristen Kelsall-Spurr, and Vicky Stent. I am thankful for the help and tutoring given by Adrian Bailey on insect identification. I would also like to thank my research group, Amy, Ali, Betina, Jess, Julie, Philippa, Patrick and Robin, with whom I shared my happy memories and cups of coffee. Finally, I would like to dedicate this thesis to my partner and best friend, James Baker, without whom I would not have finished, and to my family, who have always provided unconditional love and support throughout my life.

# Chapter one:

# 1. Introduction

## 1.1. Growth of the wind energy sector

The growth in renewable energy has been driven by changes in government policies around the world in reaction to climate change. Climate change is predominately the result of burning fossils fuels, particularly since the industrial revolution (Mann, Bradley & Hughes 1998). The ‘Kyoto Protocol’ agreed at the United Nation’s Earth Summit in 1992, was signed by 192 countries and aimed “to stabilize greenhouse gas concentrations in the atmosphere, at a level that would prevent dangerous anthropogenic interference with the climate system” (Koyoto Protocol 1997). To fulfil this agreement there has been a major drive to increase energy production from renewable sources. Specific legislation was also introduced by some countries. In the UK, the increase in the renewable sector was instigated by the introduction of the Renewable Obligation (RO) in 2002 (Renewable Obligation Order 2009). The RO meant that energy suppliers were obliged to increase the proportion of electricity sales attributed to renewable sources from 3% in 2002 to 15% in 2020.

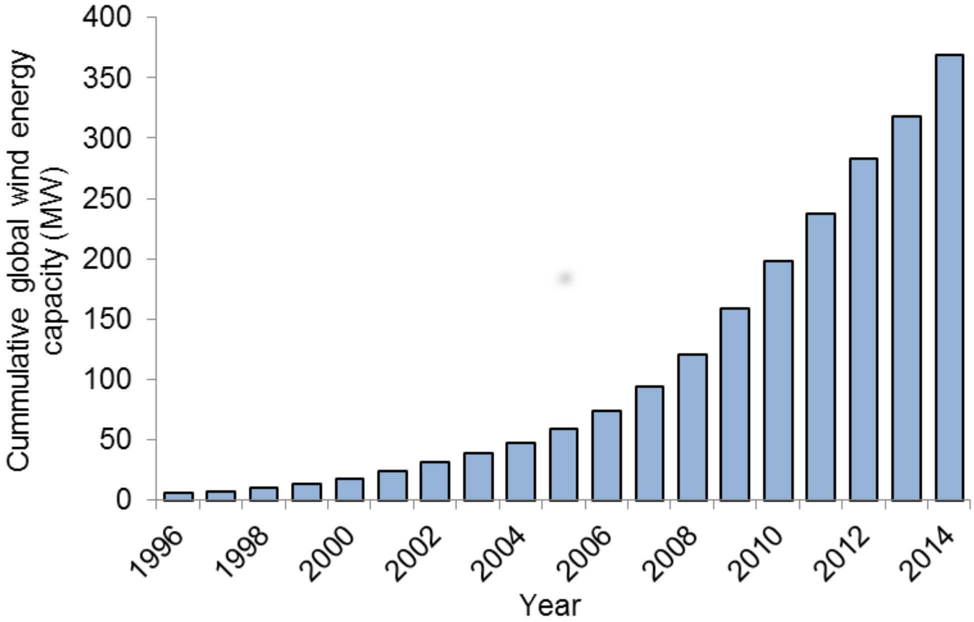

Figure 1.1. Globally installed wind power capacity [adapted from Global Wind Energy Council (2010)].

Within the renewable sector, wind power dominates, and is currently the fastest growing renewable energy technology in most countries (Global Wind Energy Council 2014). In less than two decades global wind power capacity has increased from 6 MW to 369 MW across 80 countries (Global Wind Energy Council 2014) (Figure 1.1). Within Europe, the UK is currently the third highest producer of wind energy (Global Wind Energy Council 2014; Figure 1.2), having some of the best wind resources in Europe (Sustainable Development Commission, 2005), and previously receiving high investment through government incentives. This enabled advancement of turbine technology and more efficient energy production.

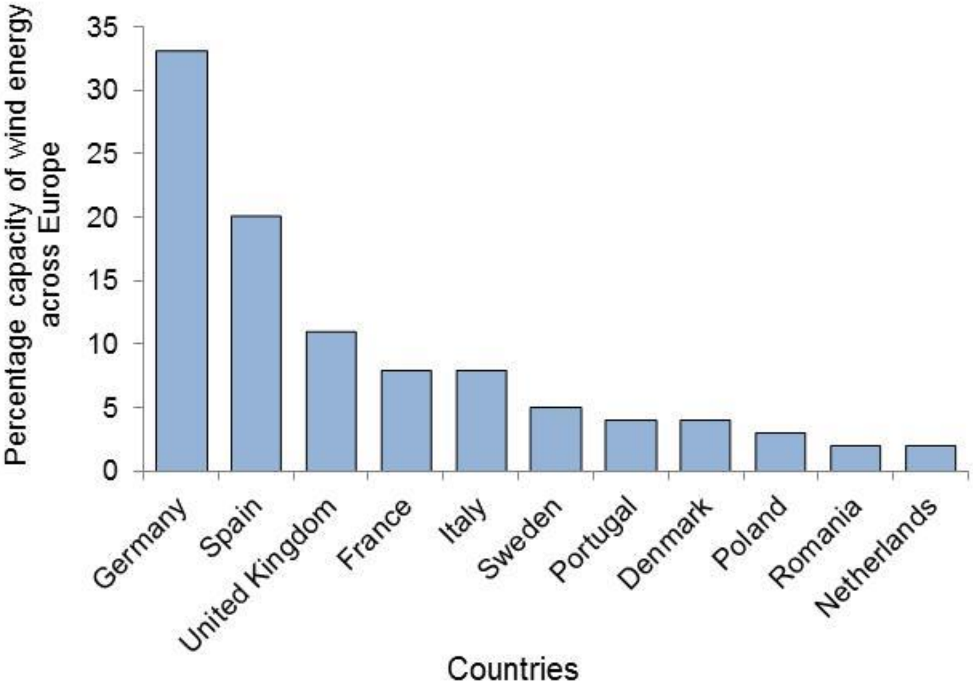

Figure 1.2. Contribution of European Union member states to total installed European wind energy capacity in 2014 (adapted from European Wind Energy Association 2015).

The first onshore wind farm (defined as more than one wind turbine in an area with an individual capacity greater than 50 kW; Department of Energy & Climate Change 2004) in the UK became operational in 1991 (RenewableUK 2010). As of November 2015 there were 912 onshore wind energy facilities in the UK, producing 8.3 GW of energy (RenewableUK 2015). Across the UK, Scotland produces 62% of wind energy capacity, with much of the growth occurring between 2010 and 2015 (2.5 GW to 5.1 GW; Scottish Government 2015). Total UK capacity is projected to rise to 10 GW by 2020, which could generate up to 10% of the UK's energy supply (RenewableUK 2015).

### 1.2. Impacts of wind energy

Energy derived from wind power was initially considered an environmentally friendly option. However, during the expansion of the sector, some negative environmental and anthropogenic impacts of wind-farming emerged as a result of; their aesthetics (Pedersen & Waye 2007), noise pollution (Julian, Jane & Davis 2007), loss of important habitats, such as peatland and forest (e.g. Nayak et al. 2010), a reduction of plant diversity (Fraga et al. 2008) and direct and indirect impacts on wildlife (Drewitt & Langston 2006; Thomsen et al. 2006; Arnett et al. 2008). The most notable cost to wildlife has been to birds and bats, indirectly through loss of habitat (Fox et al. 2006) and disturbance (Leddy, Higgins & Naugle 1999; Masden et al. 2009; Pearce-Higgins et al. 2009), and directly as a result of fatalities (Arnett et al. 2008; Rydell et al. 2010b). The direct impact to birds was first identified in the early 1990s during the initial expansion of the sector (Global Wind Energy Council 2014). High numbers of bird carcasses were found beneath wind turbines at some wind farms in North America, such as at Altamont Pass, California (Smallwood & Thelander 2008). Here it was estimated that 1,127 raptors and 3,837 birds were killed annually, including golden eagles (*Aquila chrysaetos*), which like most raptor species, has a long life span and low reproductive rate (Smallwood & Thelander 2008). In addition to the high fatality numbers, a high range of species were killed (42 different species), many of which are protected under the Migratory Bird Treaty

Act (Smallwood & Thelander 2008). As a result of these high fatalities, carcass searches for birds at new and existing wind farms became commonplace (Erickson et al. 2001). The mean number of bird fatalities across wind farms in North America was estimated at 2.19 fatalities turbine (t)$^{-1}$ year (y)$^{-1}$; but with exceptionally high spatial variation between wind farms (Erickson et al. 2001). Extrapolating to all wind farms in North America, it was estimated that 33,000 birds were killed annually (Erickson et al. 2001). Similarly in Europe, high bird fatalities were found at some wind farms, particularly where extensive wind farms were built in topographical bottlenecks (e.g. mountain passes) where large numbers of birds fly through a restricted area during migration and on local flights (Barrios & Rodriguez 2004). In the Strait of Gibraltar, Southern Spain, De Lucas et al. (2012) recorded 221 dead griffon vultures, *Gyps fulvus*, at 13 wind farms (297 wind turbines) between 2006 and 2009, equating to 0.2 vultures killed t$^{-1}$ y$^{-1}$. This spatial clustering of fatalities for species of high conservation concern raised bird fatalities at wind farms as an important conservation issue.

### 1.3. Bat fatalities at wind farms

In the early 1990s, a small number of bats were found at wind energy facilities during carcass searches for birds. For example, at Buffalo Ridge Wind Farm, Minnesota, USA, five bat carcasses were found between April and December 1994 (Osborn et al. 1996). In 2003, at Mountaineer Wind Farm, West Virginia, USA, the number of bat fatalities were estimated to be between 31.78 and 91.62 bats t$^{-1}$ during the search period (spring to autumn) (Kerns & Kerlinger 2004). Large differences in bat fatality numbers, both spatially and temporally, became apparent. In response to accumulating evidence that bats could be killed in high numbers at wind farms, the number of post-construction carcass searches for bats increased. Over the last decade, bat fatalities at wind turbines have been reported globally (Arnett et al. 2008; Rydell et al. 2010b). In North America, mean bat fatality rates were estimated at an average of 14 ± SE 4.0 $t^{-1}$ $y^{-1}$ (range 0.5 to 70), based on results from 19 wind farms (Arnett et al. 2008). In Europe, mean estimates from 41 sites, are less, at 6 ± SE 1.4 fatalities $t^{-1}$ $y^{-1}$ (range 0 to 41) (Rydell et al. 2010b). In Germany in particular, the number of fatalities is higher, estimated at 10 to 12 fatalities $t^{-1}$ $y^{-1}$ (Brinkmann et al. 2011). It is not known whether this difference is as a result of varying population sizes, increased risk of fatality, or variations in search effort and efficiency.

### 1.4. Conservation status of European bats at higher risk of fatality

European bat populations underwent a drastic decline in the second half of the 20th century due to: habitat loss, agricultural intensification, an increase in pesticide use (Harris et al. 1995), intentional killing, and destruction of roosts (Haysom 2013). As a result, in 1994 the agreement known as EUROBATS came into force under the Convention on the Conservation of Migratory Species of Wild Animals (CMS 2002). EUROBATS aims to conserve all 53 European bat species through legislation, monitoring and education. Among 63 range states[1], 36 have signed up to the agreement (CMS 2002).

In 2008, when the impact of wind farms on bats became apparent, signatories to EUROBATS were encouraged to initiate national wind farm monitoring strategies (EUROBATS 2014). In the European Union, Environmental Impact Assessments (EIA) became a legal requirement at all

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[1] Any state (whether or not it is a Party to the Convention) that exercises jurisdiction over any part of the range of a species covered by the EUROBATS Agreement (CMS 2002).

wind farm installations under the EIA Directive 2014/52/EU (EIA Directive 2005). The purpose of the EIA is to assess the extent of the impact, and where necessary, to specify avoidance, mitigation or compensation protocols that will ensure that there are no overall negative impacts. Under the Council Directive 92/43/EEC, all member states are required to restore or maintain their bat populations in favourable conservation status and monitor any potential threats, such as collisions with wind turbines.

European studies of bat casualties from wind turbines have reported that 11 genera are at increased risk (Table 1.1). However, data were collected on an *ad hoc* basis, potentially biased by sampling effort and therefore reported fatalities may not be a true reflection of relative risk. The most common genera of bats killed by wind turbines are *Pipistrellus* (54%) and, to a lesser extent, *Nyctalus* (22%; EUROBATS 2014). This suggests that open aerial and edge foragers are most at risk of fatality. Bats in the *Pipistrellus* and *Nyctalus* genera that occur in Europe (*P. khuli*, *P. pipistrellus, P. pygmaeus, P. nathusii, N. leisleri,* and *N. noctula*) are classified by the IUCN as least concern; whereas *N. lasiopterus* (0.6 % of reported casualties) is classified as near threatened across its range (IUCN 2014).

Table 1.1. Genera of bat fatalities reported from carcass searches at wind energy facilities across Europe between 2003 and 2011 (adapted from EUROBATS 2014).

| Genus | Number | Percentage |
|---|---:|---:|
| *Barbastella* | 4 | <1 |
| *Eptesicus* | 250 | 4 |
| *Hypsugo* | 180 | 3 |
| *Miniopterus* | 9 | <1 |
| *Myotis* | 32 | 1 |
| *Nyctalus* | 1,257 | 22 |
| *Other* | 753 | 13 |
| *Pipistrellus* | 3,069 | 54 |
| *Plecotus* | 12 | <1 |
| *Rhinolophus* | 2 | <1 |
| *Tadarida* | 46 | 1 |
| *Vespertilio* | 108 | 2 |
| **Total** | **5,722** | |

## 1.5. How are bats killed by wind turbines?

Direct bat collisions with moving blades have been recorded on infra-red imagery (Horn, Arnett & Kinz 2008). In addition, *post-mortem* examinations of bat carcasses found beneath turbines revealed lesions and injuries consistent with collisions, in particular, broken wings (Grodsky et al. 2011; Rollins et al. 2012). Presumably, collisions occur due to insufficient time between a bat detecting a blade, obtaining an accurate perception of rotor movement and reacting to it. This time frame may be between 1 and 30 seconds, dependent on the species of bat; angular velocity of the rotor; blade width; and number of blades (Long et al. 2009; Long et al. 2010a). The difficulty in accurately detecting and avoiding moving blades is potentially a consequence of the Doppler shift effect, whereby a frequency shift of up to ±700-800 Hz under wind speeds <6 m $s^{-1}$ can be created by the blades (Long et al. 2010b).

The second cause of fatalities, barotrauma, is the damage to body tissues caused by a sudden change in pressure (Baerwald et al. 2008).

Barotrauma was first identified from external examination of carcasses found beneath wind turbines. Around the blade-tips of turbines, sudden pressure changes occur due to the downward movement of air creating a vortex around the tips of the blades (Bertin & Smith 1989). Blade tips can reach up to 80 m $s^{-1}$ when turbines are at top speeds, which is equivalent to a pressure change of 10 kPa (Baerwald et al. 2008). Tests on rats revealed that pressure changes of 4.4 kPa were sufficient to cause fatality from barotrauma (Baerwald et al. 2008).

Among bat carcasses examined for barotrauma, 46% (n = 188) had no external injuries, and of the 75 bats examined internally, 92% had internal haemorrhaging (Baerwald et al. 2009). Among a subsample of bats killed the previous night (a requirement for effective histologies to be conducted), 100% (n = 17) had pulmonary lesions and 88% had pulmonary haemorrhaging, all consistent with barotrauma. Since, the difficultly of distinguishing between injuries caused by barotrauma and collision based on *post-mortem* injuries alone has been highlighted (Grodsky et al. 2011). Not only is it impossible to detect barotrauma from all but very fresh carcasses, radiology is required for robust detection of bone fractures (Grodsky et al. 2011).

### 1.6. Why do bats encounter wind turbines?

There are two main contrasting hypotheses to explain large numbers of bats killed at some wind farms; (1) bats are killed during migration, where wind farms coincide with their migratory routes (Baerwald & Barclay 2009; Baerwald & Barclay 2011; Cryan 2011; Lehnert et al. 2014); and (2) bats are attracted to wind turbines either to investigate turbines as potential roost or mating sites (Cryan 2008; Cryan et al. 2014) or to feed on insect accumulations (Corten & Veldkamp 2001; Long, Flint & Lepper 2011; Cryan et al. 2014). Whilst there are two main hypotheses, these are not mutually exclusive and the importance of each may vary between wind farms or geographical areas.

Among the general hypothesis that bats are attracted to wind farms, some specific hypotheses have been refuted, such as, that lights mounted on turbines can attract bats to investigate them (Arnett 2005) or remain inconclusive, namely, that bats are attracted to the ultrasounds produced by turbines (Szewczak & Arnett 2005). Another general attraction hypothesis is that bats are attracted to the wind farm site rather than to wind turbines specifically. Attraction to the site may be due to the construction of new roads creating linear features, such as, woodland edges (Cryan 2009; Kunz 2007b) which are well known to facilitate foraging and commuting (Walsh & Stephen 1996; Lesinski 2008; Cryan & Barclay 2009).

### *1.6.1. Migration*

Bat fatalities at wind farms peak during late summer and early autumn, which coincides with the timing of migration (Arnett et al. 2008, Rydell et al. 2010b). Recent evidence supports the hypothesis that migrant, as well as local bat populations are killed at wind turbines (Voigt et al. 2012; Lehnert et al. 2014). In North America, among the 45 species of bats killed by wind turbines, ~75% of fatalities are made up of three species of migratory bat; eastern red bat (*Lasiurus borealis*)*,* hoary bat (*L. cinereus*)*,* and silver haired bat (*Lasionycteris noctivagans*) (Johnson et al. 2003b; Kunz et al. 2007b; Cryan & Barclay 2009). In Europe, 64% of known fatalities are accounted for by five species; common pipistrelle (*Pipistrellus pipistrellus*)*,* soprano pipistrelle (*P. pygmaeus*)*,* Nathusius' pipistrelle (*P. nathusii*)*,* noctule (*Nyctalus noctula*) and Leisler's (*N. leisleri*) (EUROBATS 2014), among which, all are considered migratory with the exception of *P. pygmaeus* which are considered partially migratory across mainland Europe (Ahlén, Baagøe & Bach 2009; Britzke et al. 2009). In mainland Europe bats typically follow a northeast - southwest direction when migrating between maternity and hibernation roosts (Hutterer 2005). However, little is known about bat migration between the UK and continental Europe. Recently, the first direct evidence was found for a bat (*Pipistrellus nathusii*), migrating 370 miles between the UK and the Netherlands (F. Mathews, *pers. comm.*).

The origins of bats killed at wind turbines have been determined by assessing stable hydrogen isotopes from their fur (Voigt et al. 2012; Lehnert et al. 2014). Among carcasses collected at wind farms in the north and west of Germany, *Pipistrellus nathusii* (n = 10) were from Estonian or Russian origin, and *P. pipistrellus* (n = 16) were from local origins (Voigt et al. 2012). Whereas, *Nyctalus noctula* (n = 14) and *N. leisleri* (n = 7) originated from Scandinavia or other north eastern parts of Europe (Voigt et al. 2012). Among a larger sample of *Nyctalus noctula* carcasses (n = 136) from wind farms in east Germany, found that 72% were of local origin and the remainder (28%) were long distance migrants from Scandinavia, eastern Baltic countries, Belarus and Russia (Lehnert et al. 2014). In the UK, fatalities at turbines could be a mixture of both migratory and local populations similar to in mainland Europe. However, the proportion of migrants is likely to be less due to its island geography, with the English Channel and the North Sea providing a substantial barrier (Moussy et al. 2012) with migration most likely to occur within Britain.

### *1.6.2. Roosting and mating*

Given that in North America most bat fatalities at wind turbines are migratory tree-roosting species (Kunz et al. 2007b) and peak fatalities coincide with the mating period (September to October), it has been hypothesized that bats are attracted to turbines for roosting or mating (Cryan & Barclay 2009). Tree roosting bats investigate tall, mature trees to find appropriate features such as lifted bark and deep crevices for roosting (Sedgeley & O'Donnell 1999; Menzel et al. 2002). Cryan (2008) suggested that turbines may be mistaken as large trees and used as mating sites. Consistent with this, a positive relationship between tower height and the total number of bat fatalities has been found in both North America (n = 21 sites, tower height range 25 to 80 m) and Europe (n = 37 sites, tower height range 24 to 98 m) (Barclay, Baerwald & Gruver 2007; Rydell et al. 2010b). In North America the number of fatalities increased exponentially with tower heights above 65 m (Barclay, Baerwald & Gruver 2007). Adult and juvenile males examined from a sample of carcases found at turbines were reproductively ready to mate, although there was no evidence that they had recently copulated with females (Cryan et al. 2010). If fatalities were resulting from mating behaviour, a male sex-bias would be expected, which has not been found among most studies in Europe (Rydell et al. 2010b); however, a male-sex bias in fatalities was found among nine wind farms monitored in Greece (Georgiakakis et al. 2012).

### *1.6.3. Prey accumulations and bat foraging*

Bat distributions are determined by a multitude of factors including, roost availability, habitat, microclimate, and the abundance of prey (Fukui et al. 2006; Threlfall, Law & Banks 2012). Given the importance of prey abundance and distribution in determining bat distributions, bats foraging on accumulations of insects at turbines, has been postulated as a main hypotheses for the occurrence of bat fatalities at wind farms. Accumulations of insects at turbines are evidenced by efficiency losses due to the build-up of insect remains on blades (Corten & Veldkamp 2001). Reasons for such accumulations include: (1) attraction to turbine colour, (2) creation of microclimates around moving blades, (3) a behaviour known as ‘hill-topping’ and, (4) insect migration.

Insect accumulations as a direct result of attraction to turbines specifically may occur. Long, Flint & Lepper (2011) used paint coatings on card to investigate insect attractiveness to colour, and found that white coatings attracted insects the most and purple the least. From this they implied that turbines, which are commonly painted white, may attract insects. However, the study was conducted on day flying insects and results may not be transferable to nocturnal insects. Insect accumulations, specifically at the height of the nacelle (housing of the gearing and generating components of a wind turbine located at the top of the tower, at the centre of the rotor sweep area) may occur as a result of microclimates created behind moving turbine blades (Cryan et al. 2014). Insects often aggregate on the down-wind side of large structures, acting as wind breaks, especially as wind speed increases (Lewis 1965; Lewis 1969) and turbines may create a similar microclimate for insects (Cryan et al. 2014). Under certain conditions bats orientate downwind of the nacelle, perhaps as a result of insects gathering in the airspace on the leeward side of the turbine (Cryan et al. 2014), although no insect sampling has been conducted at the nacelle to confirm this.

Insect accumulations may also occur on hill-tops, where turbines are often sited, due to a behaviour known as ‘hill-topping’ (Rydell et al. 2010a). This is where male insects follow a hill (or a turbine tower) upwards and congregate (Shields 1967) to attract females for mating (Alcock 1987). Most wind farm sites with high fatalities in North America and Europe are situated on hill tops and ridges (Fiedler 2004; Kerns, Erickson & Arnett 2005; Rydell et al. 2010b).

Rydell et al. (2010a) also suggested that insect migrations may be linked to bat fatalities at wind farms. Insect migrations occur on warm, low wind speed nights which typically follow the passage of cold fronts. Insect migrations generally occur between late summer and early autumn, both of which coincide with the timing of bat fatalities. High altitude feeding in some bat species, such as *Nyctalus noctula* (Kronwitter 1988)*,* occurs at similar elevations to that of the mass movement of migrating insects during late summer (Taylor 1974; Reynolds, Smith & Chapman 2008). However, species which forage at the highest altitudes in Europe (*Tadarida teniotis, Miniopterus schreibersii, and Nyctalus lasiopterus*) appear to be rarely killed by turbines (EUROBATS 2014; Dubourg-Savage et al. 2011). This may be due to two of the species being classified as near threatened (IUCN 2014), and a paucity of surveys at wind farms within their range.

The foraging activity of bats has been found to correlate with insect abundance across a wide array of geographical areas and habitats across the bat season (e.g. Wickramasinghe et al. 2004; Fukui et al. 2006). Specifically at turbines, positive correlations between the number of insect passes and bat passes have been recorded at elevations of up to 70 m (Horn, Arnett & Rodriguez 2004), providing evidence for a relationship between prey and bat foraging at turbines. The general hypothesis that bats forage on insects at turbines is further supported by a study in North America where carcasses of the hoary bat (*Lasiurus cinereus*) were found to have full stomachs (Valdez &

Cryan 2013). However, this evidence is far from conclusive, since they could have foraged elsewhere prior to their death.

## 1.7. Mitigation measures

Over the last decade mitigation strategies have been tested to reduce the number of bat fatalities at wind turbines. These include the use of acoustic deterrents (Szewczak & Arnett 2006; Arnett et al. 2013b), radar (Nicholls & Racey 2007), and increasing the wind speed at which turbines become operational (curtailment) (Arnett et al. 2013a). To date, only curtailment has reduced the number of fatalities at a small number of wind farms in North America. Ultraviolet light has also been recently considered as a method to deter bats, but trials have yet to be conducted at turbines (Gorresen et al. 2015).

Species of bats killed by wind turbines navigate and forage by ultrasonic echolocation. Devices which produce ultrasonic sounds have been tested with the aim of disrupting or jamming the echoes of bats to deter them from the close vicinity of wind turbines (Szewczak & Arnett 2006; Arnett et al. 2013b). Szewczak and Arnett (2006) found that bat activity and foraging was significantly reduced when using acoustic deterrents. However, the study was not conducted at wind turbines and controls and treatments were on different nights; therefore, direct comparisons are difficult to make due to high temporal variability in bat activity (Hayes 1997). Additionally, problems with rapid attenuation of sounds (particularly in humid conditions) were encountered, limiting their effective range (Szewczak & Arnett 2006; Arnett et al. 2013b). It is not known whether the use of deterrents is currently a viable and cost effective method. While development costs of deterrents are high, operational costs over the long term have not yet been established (Arnett et al. 2013b).

The use of radar to repel bats has been assessed as a potential mitigation option (Nicholls & Racey 2009). Radar was used to transmit radio waves at pulse lengths of between 0.08 ms and 0.8 ms, positioned at 2 m above ground level. Bat activity was compared at the radar and control locations across 20 foraging sites (not wind farms). Activity was reduced significantly when the radar was fixed (i.e. not rotating and therefore the strength of the unidirectional field was increased) with a pulse length of 0.3 µs. Even under these conditions, bats still continued to be active within 10-30 m of the radar (Nicholls & Racey 2009), which is well within the ‘at risk area’ of the rotor sweep of large commerical wind turbines.

Most bat fatalities have occurred on nights with low wind speeds (< 6 m $s^{-1}$) (Brinkmann, Schauer-Weisshahn & Bontadina 2006; Arnett et al. 2008; Rydell et al. 2010b). Increasing the cut-in speed that turbines become operational is a mitigation option that has been explored (Arnett et al. 2010; Baerwald et al. 2009). The wind speed that turbines become operational is usually between 3.5 to 4.0 m $s^{-1}$, depending on the manufacturer and individual site specification. In the study by Baerwald et al. (2009) the turbine cut-in speed was increased to 5.5 m $s^{-1}$, which reduced fatalities by 60% compared to control turbines using standard manufacturer specifications. Arnett et al. (2010) tested two curtailment treatments, increasing cut-in speed to 5.0 m $s^{-1}$ and 6.5 m $s^{-1}$ compared to controls. This resulted in a reduction in fatalities ranging from 44% to 93% among curtailed turbines. The total energy loss by implementing these higher cut-in speeds was estimated to be ≤ 1% of the total annual output (Arnett et al. 2013a). However, the percentage loss may be much higher for lower energy production sites.

## 1.8. Objectives of research

Despite growing awareness of the impact of wind farms on bats, few large scale systematic studies have been conducted to ascertain what features of sites pose the most risk to bats. The continual expansion of wind energy, faster than any other renewable energy source (Global Wind Energy Council 2014), provides an added threat to bats, which are already undergoing pressures from habitat loss and intensification of agriculture (Jones, Purvis & Gittleman 2003). Minimising fatalities is paramount to ensure populations are sustainable, particularly those likely to migrate through or forage at high risk wind farm sites.

A national scale study across 48 wind farm sites (Appendix: Table 1.2) was conducted. Bat activity was monitored acoustically and trained search dogs were used to find bat carcasses, which reduce bias in estimates compared to using human searchers as per previous studies. Results will help to improve guidance and develop effective mitigation.

In particular the aims of this research were to:

- assess the spatial and temporal variability of bat activity at wind farms to determine minimum survey effort required to estimate species composition and bat activity robustly. These results will inform guidelines for surveying for bats at wind farms (Chapter two).
- determine whether bat activity is higher at turbines compared to controls, using a case-control design, and to identify and if so, characterise the conditions under which higher activity at turbines occurs (Chapter three).

- determine whether insect abundance and biomass is higher at turbines compared to controls, using a case-control design, and if bat foraging at turbines can be predicted by insect abundance and/or biomass (Chapter four).
- assess whether bat fatalities can be predicted from bat activity and determine the most appropriate height (ground or nacelle) for sampling to inform current guidance and potentially refine curtailment mitigation (Chapter five).
- examine if two cryptic, and largely sympatric species, which are killed at wind farms in Europe, have different predictors and levels of fatality risk. These results will inform guidance and mitigation using a species specific approach (Chapter six).
- estimate the number and species of bats killed by wind turbines across Britain (Chapter six).

## 1.9. Appendix I

Table 1.2. Summary of the location and turbine characteristics of the 48 wind farm sites in the study. The number of turbines, tower heights and blade diameters are rounded to the nearest 5 units to anonymize sites.

| Site ID | Country | Number of turbines | Tower height (m) | Blade diameter (m) | Cut-in speed (m $s^{-1}$) |
|---|---|---|---|---|---|
| 1 | England | 1 | 40 | 55 | 3.5 |
| 2 | England | 5 | 50 | 50 | 5.0 |
| 3 | England | 5 | 60 | 80 | 4.0 |
| 4 | England | 5 | 40 | 50 | 4.0 |
| 5 | England | 5 | 50 | 60 | 4.0 |
| 6 | England | 5 | 70 | 80 | 2.5 |
| 7 | England | 10 | 60 | 80 | 2.5 |
| 8 | England | 10 | 60 | 80 | 2.5 |
| 9 | England | 10 | 80 | 90 | 4.0 |
| 10 | England | 10 | 80 | 90 | 3.5 |
| 11 | England | 10 | 60 | 80 | 6.0 |
| 12 | England | 10 | 80 | 90 | 3.5 |
| 13 | England | 10 | 60 | 80 | 2.5 |
| 14 | England | 10 | 60 | 80 | 2.5 |
| 15 | England | 10 | 65 | 50 | 5.0 |
| 16 | England | 10 | 80 | 90 | 4.0 |
| 17 | England | 15 | 60 | 80 | 2.5 |
| 18 | England | 15 | 80 | 80 | 6.0 |
| 19 | England | 15 | 30 | 35 | 4.0 |
| 20 | England | 15 | 40 | 40 | 4.0 |
| 21 | England | 20 | 65 | 50 | 5.0 |
| 22 | England | 20 | 80 | 90 | 3.5 |
| 23 | England | 20 | 80 | 90 | 3.5 |
| 24 | Scotland | 5 | 45 | 80 | 5.0 |
| 25 | Scotland | 5 | 75 | 100 | 4.0 |
| 26 | Scotland | 5 | 80 | 90 | 3.5 |
| 27 | Scotland | 10 | 80 | 90 | 4.0 |
| 28 | Scotland | 10 | 60 | 80 | 4.0 |
| 29 | Scotland | 10 | 65 | 55 | 4.0 |
| 30 | Scotland | 10 | 65 | 100 | 4.5 |
| 31 | Scotland | 15 | 45 | 60 | 3.0 |
| 32 | Scotland | 15 | 50 | 50 | 2.5 |
| 33 | Scotland | 15 | 60 | 100 | 4.5 |
| 34 | Scotland | 20 | 60 | 80 | 3.5 |
| 35 | Scotland | 20 | 50 | 100 | 4.5 |
| 36 | Scotland | 20 | 80 | 40 | 3.5 |
| 37 | Scotland | 20 | 60 | 60 | 3.0 |
| 38 | Scotland | 25 | 50 | 50 | 5.0 |
| 39 | Scotland | 25 | 50 | 60 | 4.5 |
| 40 | Wales | 10 | 50 | 50 | 5.0 |
| 41 | Wales | 10 | 60 | 100 | 4.5 |
| 42 | Wales | 10 | 50 | 50 | 5.0 |
| 43 | Wales | 10 | 50 | 60 | 4.0 |
| 44 | Wales | 15 | 80 | 80 | 3.0 |
| 45 | Wales | 15 | 60 | 100 | 4.5 |
| 46 | Wales | 29 | 50 | 50 | 2.5 |
| 47 | Wales | 20 | 35 | 40 | 5.0 |
| 48 | Wales | 20 | 70 | 90 | 4.0 |

# Chapter two:

# 2. Evaluating acoustic surveys for bats: assessing species composition and activity at wind farms

### 2.1. Summary

Reliable ecological surveys are fundamental for effective wildlife conservation. There is currently little research evaluating survey design to estimate activity levels for bats. This is of current importance given that high numbers of bat fatalities occur at wind farms globally, and bat activity is predominantly used to assess risk levels at sites. Ecological consultants use desk studies of local roost locations, previous activity surveys, as well as activity levels measured at a potential site to produce an environmental impact assessment. If high risk species are present (e.g. *Nyctalus* and *Pipistrellus* spp.) and their activity levels are high a site may be classified as high risk. It is therefore important to evaluate current survey guidelines (10 survey nights within the peak fatality period) to ensure they are sufficient, temporally and spatially, to robustly detect species and estimate their activity levels.

A nationwide study of bat activity at 139 turbines across 48 wind farm sites was conducted throughout Britain from 2011 to 2013. Surveys were conducted for one month at each site, thereby controlling for variations in seasonality, during the peak period of fatality (late summer to early autumn). Activity at turbines was paired to monitor at ground and from the centre of the rotor sweep area (nacelle). Survey effort was assessed spatially and temporally to determine how increasing effort influenced estimates of species composition and bat activity.

This study provides evidence to support current EUROBATS guidelines for the design of bat surveys at wind farms. The probability of detecting a species at the nacelle and not ground was <0.01 for all species and tower heights, whereas the probability of detecting a species at ground and not at the nacelle ranged from 0.09 to 0.5. When monitoring at ground at three turbines, after ten survey nights (comparable with EUROBATS guidelines) all high risk species (open and edge foragers) were detected for at least 90% of sites (95% CI 91% to 100%) and maximum nightly activity ranged from 16% to 23% (lower 95% CI) below maximum site activity (maximum activity during 14 nights). These results have important implications for the design of bat surveys at wind farms and hence mitigation strategies to reduce fatalities at wind farms. It is recommended that monitoring at ground is prioritized over monitoring at the nacelle. To determine whether monitoring at the nacelle is necessary at all wind farms, further investigation is required to assess which monitoring height is a better predictor of fatalities, for high risk species.

## 2.2. Introduction

Reliable ecological surveys to assess animal abundance and diversity are fundamental in the field of wildlife conservation (Spellerberg 1994). Survey guidelines have been developed in many areas to ensure minimum standards are met and consistency between surveys is maintained (Consortium 1993; Institute of Environmental Assessment 1995; Romano et al. 2006). The need for an evidence based approach when developing survey guidelines has been well acknowledged (e.g. Sutherland et al. 2004). For bats, which are the second largest order of mammals globally and provide important ecosystem services (Shilton et al. 1999; Winter & von Helversen 2001; Boyles et al. 2011), acoustic surveys are increasingly used to survey populations (e.g. Walters et al. 2012). Although species composition has been assessed in relation to survey design (Hayes 1997; Rodhouse, Vierling & Irvine 2011; Skalak, Sherwin & Brigham 2012), there is little research evaluating surveys to estimate activity levels. Whilst species composition is an important measure, bat activity levels are widely used by practitioners to assess the relative importance of sites (e.g. Roche et al. 2011).

Effective surveying to estimate bat activity levels is of particular current importance given the risk wind turbines pose to bats (Johnson et al. 2003b; Niermann, Behr & Brinkmann 2007). Bat fatalities at wind farms have been reported in high numbers globally (Arnett et al. 2008; Rydell et al. 2010b). In response, survey guidelines have been developed to assess fatality risk at proposed and operational wind farm sites (e.g. Rodrigues et al. 2014; Sowler & Stoffberg 2011; Hundt 2012). Site specific risk is assessed by a combination of species composition and activity levels. Species composition is an important measure since fatality risk differs between species, and activity is important since there is some evidence that activity is correlated with fatality (Johnson et al. 2004; Korner-Nievergelt et al. 2013). Together, these results, in addition to other environmental impacts, are used to assess the extent of the impact, and where necessary to determine if any mitigation is required at the site (Rodrigues et al. 2014).

To date, most research evaluating survey design at wind farm sites has been conducted at pre-construction sites (Weller 2007; Collins & Jones 2009). Furthermore, there is accumulating evidence that wind turbines alter the behaviour of bats (Horn, Arnett & Rodriguez 2004; Horn, Arnett & Kinz 2008; Cryan et al. 2014) and therefore a scientific assessment of survey design needs to be undertaken at operational wind farms.

Current European legislation protecting bats (EUROBATS) advises that automatic acoustic surveys should be conducted at all pre-construction wind farm sites. One survey visit should be conducted every second week between 15 May and 31 July, and every 10 days between 1 August and 31 October

(Rodrigues et al. 2014). The guidelines advise that ground level surveys should be conducted at all turbine locations, and surveys should ideally be conducted within the centre of the rotor-sweep area (herein referred to as ‘the nacelle’), for the same duration. Given the expense and logistical difficulties of surveying at the nacelle, it is likely that most survey effort is conducted at ground level.

Accuracy and precision of estimates of species composition and their activity are determined by the amount of survey effort employed (e.g. Gorresen et al. 2008, Weller & Baldwin 2012). To assess estimates of species composition and bat activity with differing survey effort, both spatially and temporally, a large scale study of bat activity was conducted at wind farm sites across Britain. Bat activity was surveyed for three times the duration of EUROBATS guidelines (equivalent to 10 survey nights between the same survey period: mid- July to mid-October) at both ground and the nacelle. Specifically the following questions were addressed:

1. Where is best to detect species and estimate activity; at ground or at the nacelle?
2. When is night time curtailment of turbines most effective?
3. How is species composition and activity estimates affected by survey effort (spatial and temporal)?
4. What are the conservation implications and how do estimates of species composition and bat activity compare to those that would have been obtained using EUROBATS guidelines?

## 2.3. Methods

### *2.3.1. Site selection*

Bats were surveyed acoustically at 48 wind farm sites (herein referred to as a site) across Britain. The distribution of sites covered the full extent of Britain with 23 sites in England, 16 sites in Scotland and 9 sites in Wales. Sites comprised moorland (n = 17), and farmland habitats (n = 31). The mean number of wind turbines at each site was 13 SD 7 and the mean tower height (centre of the rotor sweep area) was 60 SD 14 m (range 30 to 80).

### *2.3.2. Acoustic monitoring*

Static acoustic surveys were conducted in 2011, 2012, and 2013 between mid-July and mid-October. This survey period was chosen to coincide with when most bat fatalities have been recorded at wind farm sites in mainland Europe (Rydell et al. 2010b). Three turbines at each site were selected at random (using a random number generator). Bat activity was monitored acoustically at these turbines, using a paired design, from ground level (~2 m) and from the nacelle (includes monitoring at the centre of the rotor sweep area at all tower heights). Bat passes were recorded using full spectrum real-time acoustic recorders (SM2BAT, Wildlife Acoustics, Massachusetts, USA), in combination with omni-directional SMX-II microphones. The ground microphone was positioned at a 90° angle pointing away from the turbine. The microphone at height was extended approximately 1.5 m from the hub using rigid conduit tubing to hold the microphone in position and to reduce movement. Acoustic recorders were programmed to record at a sampling frequency of 192 kHz, when triggered by a signal to noise threshold above 36 db (year 1 and 2) or 48 db (year 3), adjusted in-line with manufacturer recommendations and above 16 kHz. Recorders were programmed to record from 30 minutes before sunset until 30 minutes after sunrise. Bat activity was monitored for a mean of 29 SD 6 nights per site at ground and 31 SD 8 nights per site at the nacelle.

### *2.3.3. Bat identification*

Bat calls were processed using Kaleidoscope Pro (v.1.1.20, Wildlife Acoustics, Massachusetts, USA) with British bat classifiers (v.1.0.5). This created individual sonograms classified as potentially a bat or noise. All potential bat sonograms were manually verified, classifying them to species level (with the exception of *Myotis* spp. and *Plecotus* spp. which were classified to genus because the similarity of call structure makes them difficult to distinguish reliably to species) using call parameters given in Russ (2012). Uncertain calls were classified to genus, *Nyctalus*/*Eptescisus* (due to the similar call structure for species in these genera) or unidentified.

A bat pass, equivalent to an individual sonogram, was defined as a continuous run of pulses not separated by a time gap of more than one second (Fenton, Jacobson & Stone 1973). To reduce analysis time, more than two bat passes from the same species in each sonogram were not counted. Therefore, when three or more passes from the same species occurred within a sonogram the number of passes was always counted as two. This occurred in 4% of all sonograms. All passes for each species were counted in a sonogram for a subset of the data (42%). The total number of bat passes in this subset was only 2% higher compared to if only 2 passes per sonogram had been counted for the same subset. The number of passes for each species was then calculated per night and per hour at ground and at height, to assess activity within and between nights.

### 2.3.4. *Environmental indicators*

At each site, weather data [rainfall (mm), wind speed (m $s^{-1}$), temperature (°C) and humidity (%)] were sampled using an automated weather monitor (Maplins, UK), located central to the site in an open location at ~2 m high. Recordings were taken every 10 minutes and average, minimum and maximum values were calculated for the same period that acoustic monitoring occurred (30 before sunset until 30 minutes after sunrise).

## 2.4. Statistical analyses

Analyses for each species included only sites considered to be within the geographical range according to IUCN (2014). All statistics were performed with R v.3.1.2 (R Core Team 2014) using the package lme4 (v.1.1.7.; Bates et al. 2014). All models were assessed for model fit, where appropriate, by checking that standardised residuals were normally distributed, predictor variables were not correlated (predictors were not included in the model if Pearson correlation coefficients were > 0.6 and $p < 0.05$) and there was no evidence of over-dispersion or heteroscedasticity. If two variables were correlated, two full models were run, with one variable in each model and the variable with the highest coefficient was selected. In all models, significance was assessed using log-likelihood ratio tests of the full model compared to the alternative model.

### *2.4.1. Where is best to detect species and estimate activity; ground or* nacelle?

The probability of detecting a species at the nacelle but not at ground was modelled using a generalized linear mixed model (GLMM) with a binomial error structure and logit link function. Using paired turbine, ground and nacelle height data (number of sites = 40), the dependent variable fitted was true (1) if per night, detection occurred at the nacelle but not at ground level, and false (0) if detection occurred at either, ground but not the nacelle, or at both ground and the nacelle. Site and turbine nested within site were fitted as random effects to account for the spatial autocorrelation in the data. The height (m) of detectors at the nacelle was fitted as a fixed effect. Only nights which met minimum weather conditions for bat surveys (minimum temperature ≥ 10°C, ground level wind speed ≤ 8m s$^{-1}$ and average rainfall  10°C, gro$^{-1}$) were included. Analyses were conducted separately for: *Pipistrellus pipistrellus, P. pygmaeus, P. nathusii, Nyctalus noctula* and *Nyctalus/Eptesicus,* because these species/species groups vary in their flight height (Russ 2012).

To assess how species detection would be affected if sampling only occurred at the nacelle and not ground the same model as above was fitted but with the dependent variable reversed. Per night, the dependent variable was true if detection occurred at ground and not the nacelle, and false if detection occurred at either, the nacelle but not ground, or at both the nacelle and ground. A third model was run, to assess the nightly probability of detection of each species at either ground or nacelle height detectors (true if detection occurred at either ground or the nacelle). In all models, the mean probability of detecting a species at the nacelle and not ground and at ground but not the nacelle, was predicted for turbine tower heights of 30 m, 60 m and 90 m.

### *2.4.2. How is* species composition and activity affected by survey effort?

Repeated random sampling was used to assess how increasing the number of survey nights affects estimates of species composition and bat activity. Species composition was assessed by estimating the mean probability of detection across sites for each species and height category (ground or nacelle). Sites where the species of interest was not detected after 21 nights at ground or the nacelle were excluded. For each species and height category (ground or nacelle) bat activity was assessed using two measures: the maximum nightly bat activity obtained during 14 survey nights which met minimum weather conditions (see section 2.4.1) and to overcome problems with extreme outliers the 70th percentile of nightly activity was also used (the 70th percentile for some sites was zero, analysis was only possible for *Pipistrellus pipistrellus, P. pygmaeus,* and *Nyctalus noctula* at ground level). The mean difference between each measure of site activity (maximum and 70th percentile) and the maximum nightly activity for increasing number of survey nights was then calculated. Analysis was conducted for *Pipistrellus pipistrellus, P. pygmaeus, P. nathusii* and *Nyctalus noctula,* which are most at risk of fatality at wind farms and with sufficient data*.* Sites where 14 nights of survey effort under minimum weather conditions were not met were excluded from analyses. Survey nights beyond 14 nights were excluded from analyses of survey effort in relation to bat activity.

For the random sampling process, one site, one turbine and one survey night were randomly selected and whether the species was present or absent and the percentage difference between the maximum nightly pass rate of the sample with the maximum and 70th percentile site activity was recorded. The data were then replaced and the sampling process repeated 1000 times. This was repeated for randomly sampling 1 to 21 survey nights (inclusive) for species composition analyses and 1 to 14 (inclusive) for bat activity analyses. When more than one survey night was sampled nights were not replaced within a site, so the same night could not be sampled twice within one of the 1000 reiterations. The 1000 reiterations were then repeated for two randomly selected turbines and all three turbines within a site. For each site the mean probability of detection and the mean percentage difference across the 1000 repeated samples was calculated. The mean probability of detection and the mean percentage difference across sites for each survey night ± SE was then calculated.

## 2.5. Results

Bat activity was recorded at 139 wind turbines across 48 sites on 6,535 detector nights. In total, 188,335 individual bat passes were recorded; 173,515 at ground and 14,820 at the nacelle. Among all passes, 97% were identified to species. The following genera were detected: *Barbastella*, *Eptesicus*, *Myotis*, *Nyctalus*, *Pipistrellus*, and *Rhinolophus. Pipistrellus* spp. were the most common (94% of all bat passes), and *P. pipistrellus* accounted for 72% of *Pipistrellus* spp. activity.

### *2.5.1. Where is best to detect species and estimate activity; ground or nacelle?*

Among low flying species, *Rhinolophus hipposideros, R. ferrumequinum* and *Barbastella barbastellus* were only detected at ground level and *Myotis* spp. and *Plecotus* spp. were rarely detected at the nacelle. Medium and high flying species were also more likely to be detected at ground rather than at the nacelle (Table 2.1). Among all species and tower heights, the probability (per night) of detecting a species at the nacelle and not ground was always <0.01. However, the probability of detecting a species at ground and not the nacelle ranged from 0.04 (*Nyctalus noctula)* to 0.50 (*Pipistrellus pipistrellus*) at all turbine heights. Mean activity was highest at ground compared to the nacelle for all species (Figure 2.1). Among those most at risk of fatality (*Pipistrellus pipistrellus, P. pygmaeus, P. nathusii* and *Nyctalus noctula*), mean activity respectively was 74%, 63%, 67% and 57% higher at ground compared to the nacelle (all tower heights). Activity was most variable at both ground and the nacelle for *Nyctalus noctula* and least variable for *Pipistrellus pipistrellus*.

Table 2.1. The number of sites surveyed within each species' range, their typical flight height ([*]estimates given when known) and the total number of detector nights at ground and at the nacelle are shown. Results from modelling the probability of detection at the nacelle and not ground, ground and not the nacelle, and both ground and the nacelle are also given; [***] $P$ <0.001

| Species | Flying height of species[1] | No. sites in range (% sites spp. detected at ground and nacelle) | Total number of detector nights at ground | Total number of detector nights at nacelle | Probability of detection at ground and not nacelle[3] | Probability of detection at nacelle and not ground[3] | Probability of detection at either ground or nacelle[3] |
|---|---|---|---|---|---|---|---|
| *B. barbastellus* | low[2] | 25 (36, 0) | 2,016 | 1,251 | na | na | na |
| *R. hipposideros*[*] | low[2] (< 5 m) | 13 (8, 0) | 1,140 | 839 | na | na | na |
| *R. ferrumequinum*[*] | low[2] (< 5 m) | 11 (55, 0) | 966 | 762 | na | na | na |
| *Myotis* spp. | low[2] | 48 (88, 14) | 3,658 | 2,809 | 0.25, 0.16, 0.10 | <0.01 | 0.26, 0.16, 0.10 |
| *Plecotus* spp. [*] | low[2] (5-6 m) | 48 (79, 19) | 3,658 | 2,809 | 0.09 | <0.01 | 0.09 |
| *P. pipistrellus*[*] | medium (5-10 m) | 48 (98, 74) | 3,658 | 2,809 | 0.50, 0.47, 0.45 | <0.01 | 0.72, 0.62, 0.49 |
| *P. pygmaeus*[*] | medium (5-10 m) | 46 (96, 64) | 3,532 | 2,705 | 0.41, 0.31, 0.23 | <0.01 | 0.59, 0.39, 0.22 |
| *P. nathusii*[*] | medium (4-15 m) | 42 (88, 46) | 3,238 | 2,397 | 0.07, 0.09, 0.12 | <0.01 | 0.09, 0.10, 0.12 |
| *N. noctula* | high | 37 (89, 66) | 2,858 | 2,080 | 0.04, 0.08, 0.19 | <0.01 | 0.15, 0.17, 0.19 |
| *Nyctalus/Eptesicus* | high | 38 (93, 62) | 3,658 | 2,809 | 0.05, 0.06, 0.08 | <0.01 | 0.15, 0.10, 0.06 |

[1]Russ (2012), [2]Relative to vegetation height, [3]if tower height was significant in models, the probabilities of detection are predicted for turbine tower heights of 30 m, 60 m and 90 m; na = no detections were made at the nacelle.

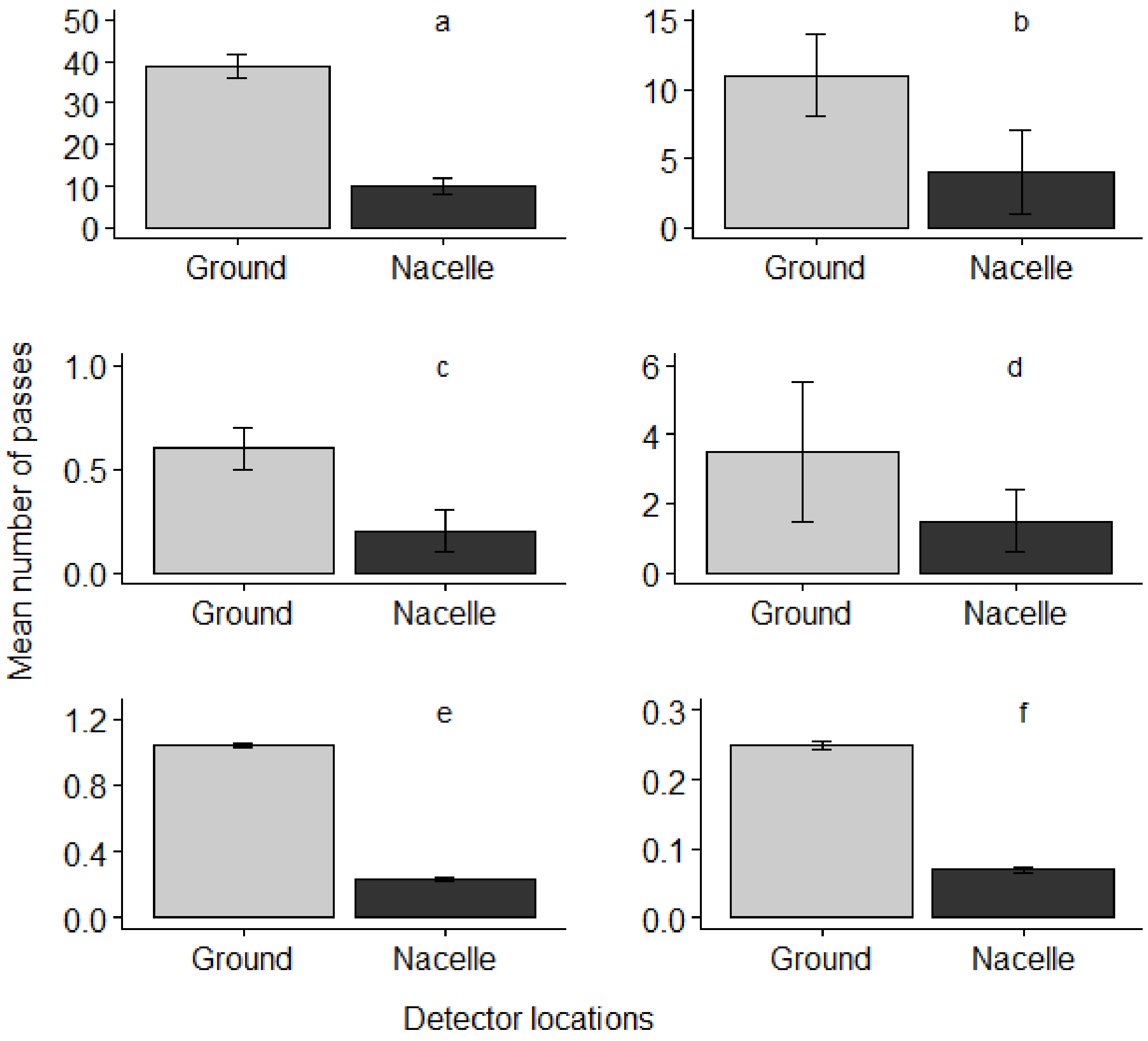

Figure 2.1. Mean nightly activity per turbine ± SE for paired ground and the nacelle detectors at 48 sites; (a) *Pipistrellus pipistrellus,* (b) *P. pygmaeus,* (c) *P. nathusii,* (d) *Nyctalus noctula,* (e) *Myotis* spp., and (f) *Plecotus* spp.

### 2.5.2. When is night time curtailment of turbines most effective?

For *Pipistrellus pipistrellus, P. pygmaeus, P. nathusii* and *Nyctalus noctula,* among hours of the night, the mean number of passes was highest between the first and second hour after sunset (Figure 2.2). For *Pipistrellus pipistrellus,* the peak in activity in this hour compared to the rest of the night was notably higher than the peak activity for the other three species. In addition, for *Pipistrellus pipistrellus,* the decline in activity after the second hour continually diminished, whereas for the other species, between the second and sixth hour after sunset activity remained more constant. Across a night (mean length 10 hrs), depending on the species, between 17% and 27% of bat activity occurred between the first and fifth hour after sunset. Between this time, at least 50% of activity (range 52% to 66% between species) occurred and at least 75% of activity (range 75% to 87%) between the first and sixth hour after sunset.

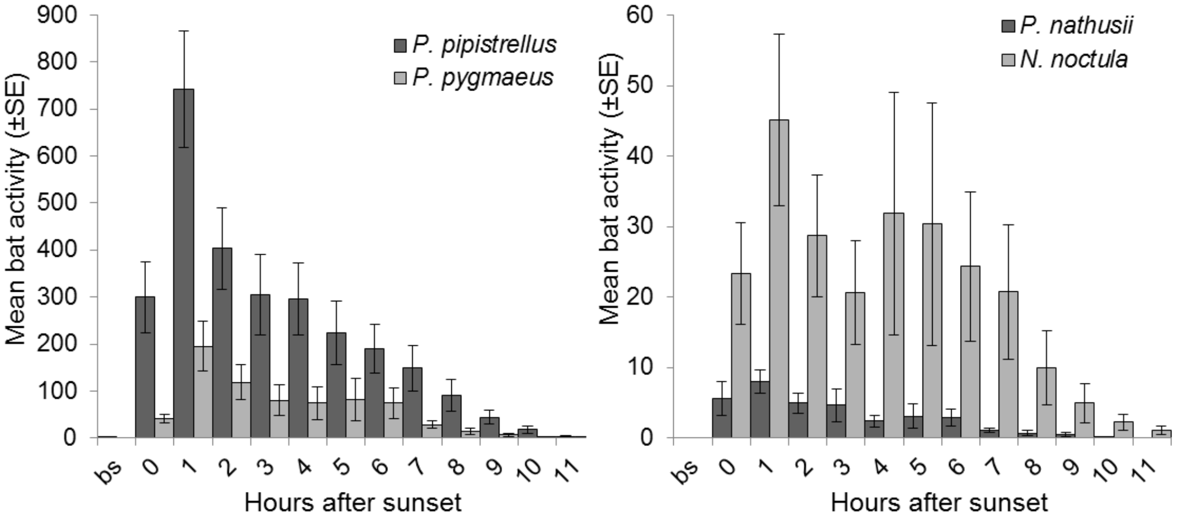

Figure 2.2. The mean hourly variation across sites in bat activity from 30 minutes before sunset until 30 minutes after sunrise for, *Pipistrellus pipistrellus* (n = 46), *P. pygmaeus* (n = 44), *P. nathusii* (n = 38) and *Nyctalus noctula* (n = 36); bs = before sunset.

### 2.5.3. How is species composition and bat activity affected by survey effort?

Estimates of species composition varied considerably depending on the height that monitoring was conducted and the amount of spatial and temporal replication. With the exception of *Nyctalus noctula,* the probability of detection was always higher at ground compared to the nacelle (Figure 2.3). When monitoring three turbines at ground, for three survey nights, the probability of detection for each species was 1.00 (95% CI 1.00 to 1.00) for *Pipistrellus pipistrellus,* 0.97 (95% CI 0.91 to 1.00) for *P. pygmaeus,* 0.90 (95% CI 0.92 to 1.00) for *P. nathusii,* and 0.99 (95% CI 0.97 to 1.00) for *Nyctalus noctula*. In comparison, at height (all tower heights) after ten survey nights, this probability reduced to 0.85 (95% CI 0.73 to 0.97) for *Pipistrellus pipistrellus,* 0.87 (95% CI 0.77 to 0.97) for *P. pygmaeus,* 0.59 (95% CI 0.43 to 0.78) for *P. nathusii* and 0.99 (95% CI 0.96 to 1.00) for *Nyctalus noctula*. Reducing spatial replication had the most impact on the probability of detection for *Pipistrellus nathusii* and least for *P. pipistrellus.* If spatial replication at ground is reduced, from three turbines to two turbines to achieve a 0.80 probability of detection (a common threshold used in power analyses, Cohen 1988), the following increase in temporal replication was required: *Pipistrellus pipistrellus*: no increase; *P. pygmaeus*: an increase from 3 to 4 survey nights; *P. nathusii*: 9 to 13; *Nyctalus noctula*: 5 to 7*.*

The amount of survey effort required to obtain nightly activity within the 70th percentile of the site activity (Table 2.3) was similar to the survey effort of that required to detect a species (Figure 2.4). The relationship between the number of survey nights and the percentage of the maximum site activity (for each height category) recorded varied little between height categories and species (Figure 2.5). Indeed, the relationship was almost linear, for each species and height category, between the number of survey nights and the percentage of the maximum site activity recorded. When monitoring three turbines for ten nights, at ground level, activity was 16%-23% (lower 95% CI among species) below the maximum site activity (obtained during 14 nights) and at the nacelle, activity was 25%-29% below maximum site activity. When survey effort was reduced, from three turbines to two for ten survey nights, the percentage difference from the maximum site activity increased to 32%-42% among species at ground level, and between 31%-39% at the nacelle. When survey effort was further reduced, from three turbines to one for ten survey nights, the percentage difference from the maximum site activity increased to 53%-66% among species at ground level, and 60%-74% at the nacelle.

Table 2.3 The 70th percentile and maximum detector night activity during 14 survey nights which met minimum weather conditions for bat surveys. Activity is shown for *Pipistrellus pipistrellus, P. pygmaeus, P. nathusii* and *Nyctalus noctula* for each height category, across all turbines at all sites used in analysis of the relationship between activity and survey effort.

| | *P. pipistrellus* | | | *P. pygmaeus* | | | *P. nathusii* | | *N. noctula* | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Site ID | 70th percentile | maximum | | 70th percentile | maximum | | maximum | | 70th percentile | maximum | |
| | ground | ground | nacelle | ground | ground | nacelle | ground | nacelle | ground | ground | nacelle |
| 1 | 0 | 5 | 0 | 0 | 7 | 0 | 0 | 0 | 0 | 0 | 0 |
| 2 | 4 | 79 | 0 | 14 | 124 | 2 | 1 | 0 | 0 | 0 | 0 |
| 3 | 6 | 54 | 9 | 0 | 0 | 0 | 1 | 0 | 1 | 3 | 1 |
| 4 | 8 | 520 | 0 | 0 | 2 | 0 | 2 | 0 | 0 | 0 | 0 |
| 5 | 8 | 210 | 10 | 3 | 31 | 0 | 0 | 0 | 0 | 0 | 0 |
| 6 | 13 | 89 | 3 | 2 | 30 | 0 | 0 | 0 | 2 | 11 | 0 |
| 7 | 17 | 53 | 0 | 2 | 11 | 0 | 3 | 0 | 3 | 7 | 0 |
| 8 | 19 | 65 | 18 | 39 | 285 | 12 | 2 | 0 | 6 | 15 | 20 |
| 9 | 22 | 52 | 4 | 0 | 2 | 0 | 1 | 0 | 1 | 23 | 0 |
| 10 | 24 | 388 | 0 | 26 | 464 | 0 | 1 | 0 | 0 | 5 | 0 |
| 11 | 25 | 371 | 1 | 1 | 1 | 0 | 6 | 2 | 3 | 13 | 0 |
| 12 | 50 | 187 | 88 | 0 | 4 | 3 | 2 | 3 | 0 | 0 | 0 |
| 13 | 54 | 621 | 221 | 9 | 47 | 62 | 1 | 1 | 0 | 4 | 0 |
| 14 | 79 | 256 | 3 | 3 | 33 | 7 | 6 | 1 | 1 | 1 | 1 |
| 15 | 146 | 947 | 33 | 13 | 63 | 3 | 7 | 3 | 0 | 1 | 0 |
| 16 | 181 | 470 | 2 | 40 | 70 | 3 | 4 | 0 | 103 | 271 | 396 |
| 17 | 308 | 591 | 2 | 8 | 25 | 0 | 2 | 0 | 3 | 9 | 3 |
| 18 | 403 | 1,950 | 0 | 14 | 190 | 0 | 30 | 0 | 1 | 23 | 0 |
| 19 | 549 | 1,074 | 682 | 176 | 578 | 802 | 52 | 20 | 2 | 14 | 11 |

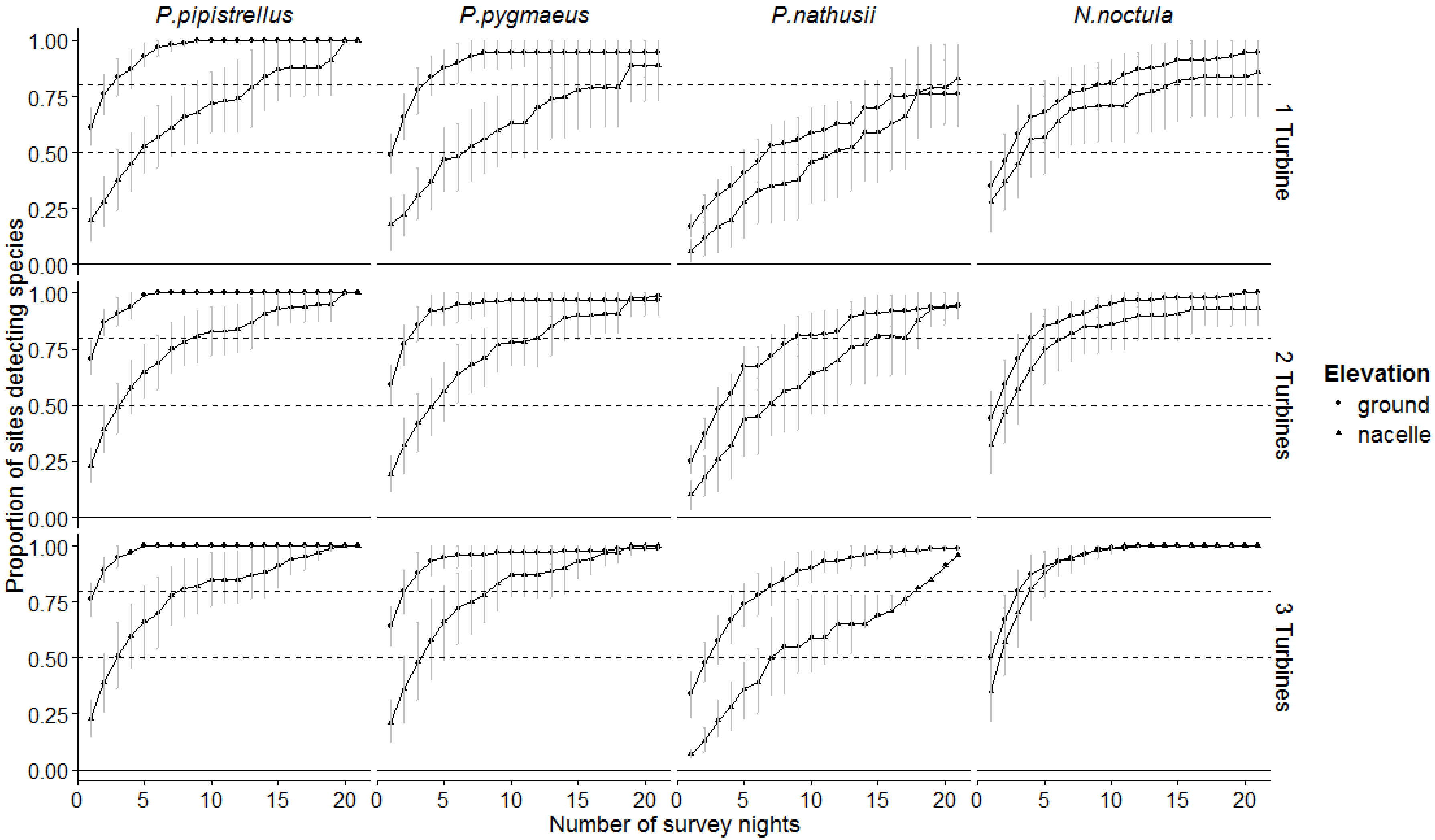

Figure 2.3. The probability of detecting *Pipistrellus pipistrellus* (ground: n = 42, nacelle: m = 21)*, P. pygmaeus* (ground: n = 37, nacelle: n = 17), *P. nathusii* (ground: n = 29, nacelle: n = 9)*,* and *Nyctalus noctula* (ground: n = 23, nacelle: n = 14) (columns) relative to number of survey nights at ground and the nacelle detectors (all tower heights) when sampling at 1, 2, and 3 turbines (rows).

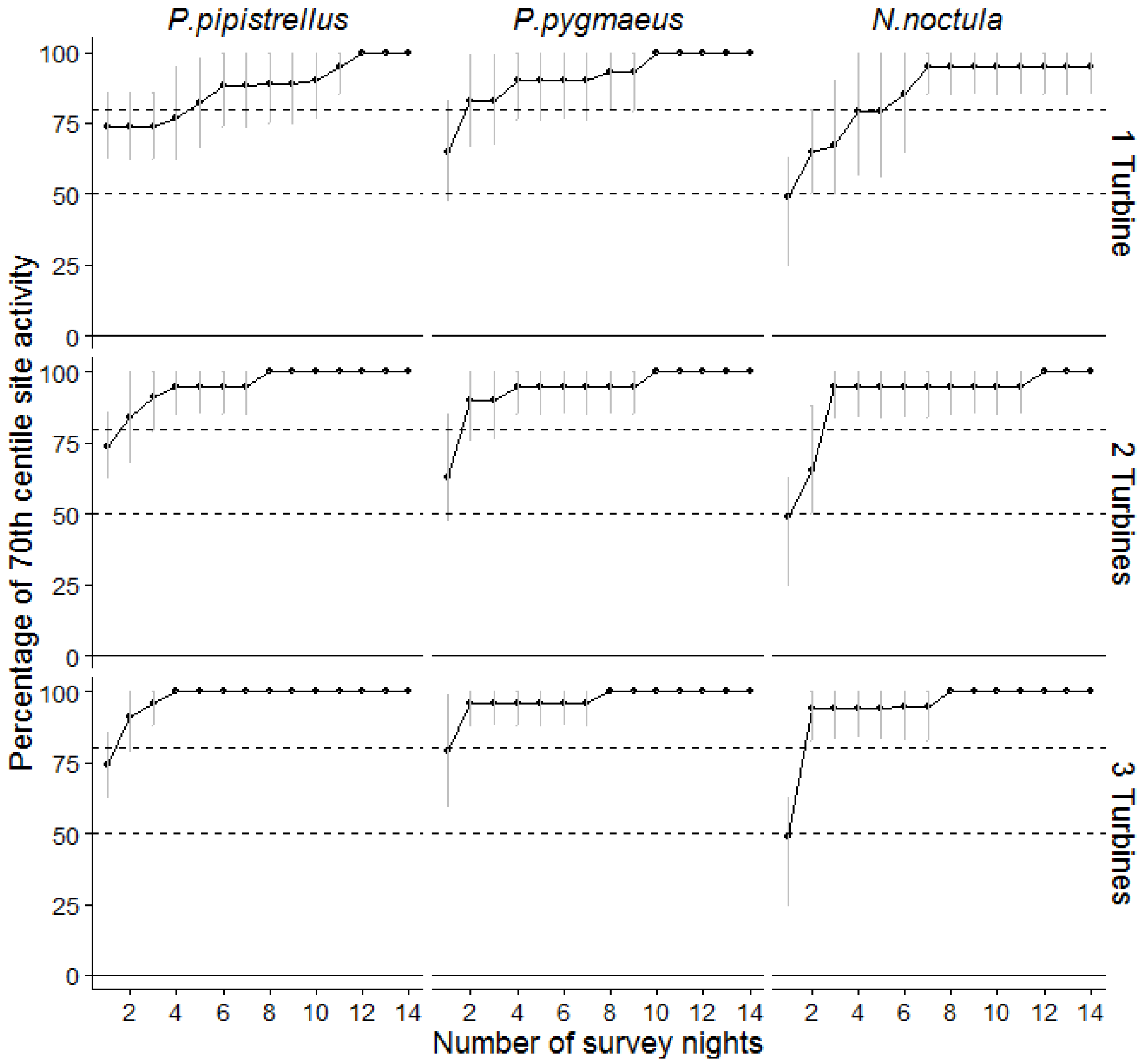

Figure 2.4. The percentage of the 70th site activity percentile (activity for the 70th percentile of nights during 14 survey nights which met minimum weather conditions) obtained relative to the number of survey nights at ground detectors (all tower heights) when sampling at 1, 2, and 3 turbines (rows) for *Pipistrellus pipistrellus* (ground: n = 19), *P. pygmaeus* (ground: n = 14), and *Nyctalus noctula* (ground: n = 11) (columns).

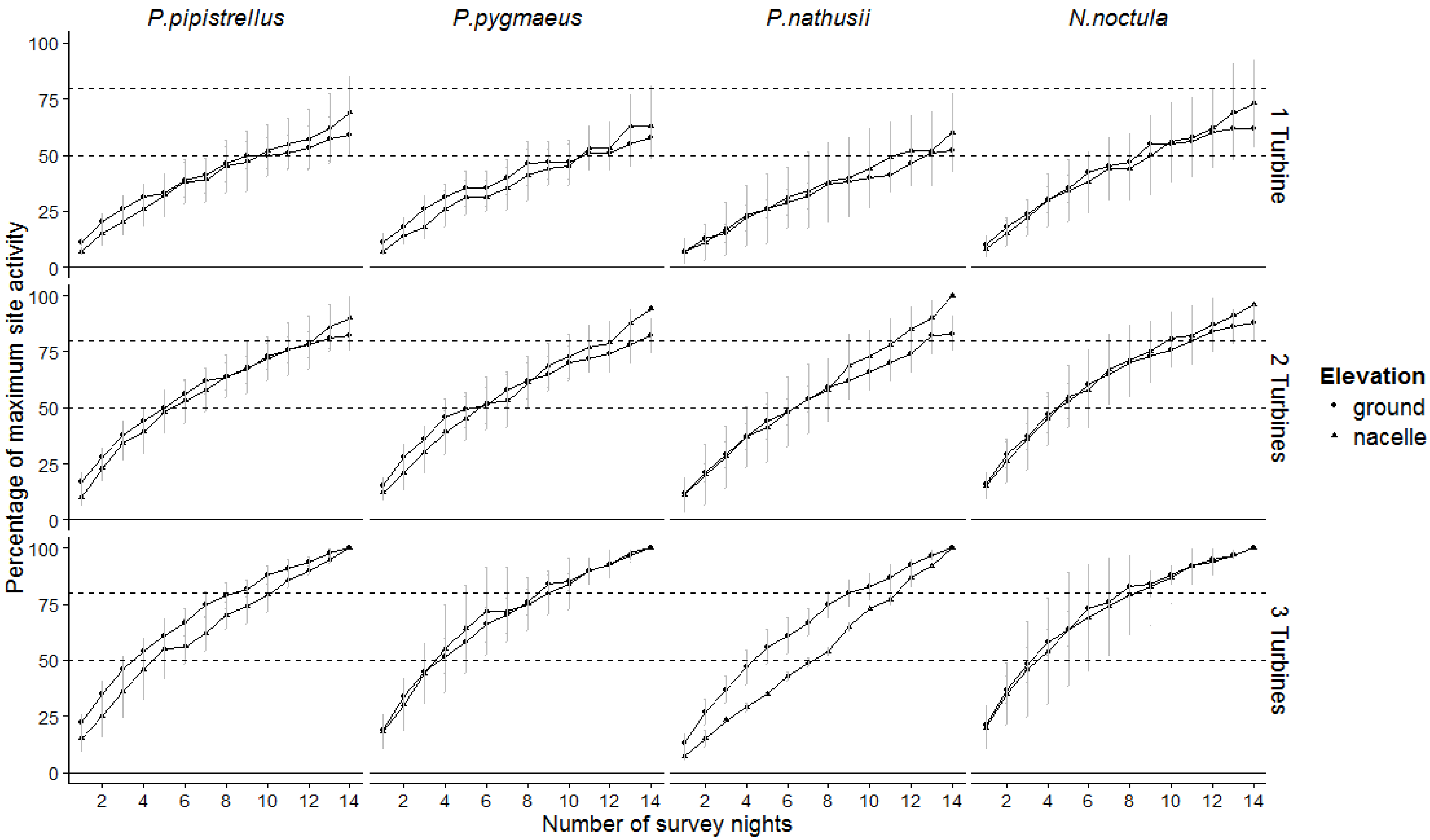

Figure 2.5. The percentage of maximum nightly site activity (maximum activity during 14 survey nights which met minimum weather conditions) obtained relative to the number of survey nights at ground and the nacelle detectors (all tower heights) when sampling at 1, 2, and 3 turbines (rows) for *Pipistrellus pipistrellus* (ground: n = 19, nacelle: n = 14), *P. pygmaeus* (ground: n = 18, nacelle: n = 14), *P. nathusii* (ground: n = 17, nacelle: n = 14), and *Nyctalus noctula* (ground: n = 15, nacelle: n = 12) (columns).

## 2.6. Discussion

Assessing the absolute number of bats at a site is difficult and so practitioners and scientists often use acoustic measures to estimate species composition and measure their activity (e.g. Roche et al. 2011). Whilst studies have assessed survey effort in relation to species composition (e.g. Skalak, Sherwin & Brigham 2012), little research has been conducted to assess how increased survey effort affects estimates of activity. This is of particular current importance given the high number of bat surveys conducted across the globe to determine the risk of fatality of bats at wind farm sites (e.g. Davy et al. 2004, Ahlén et al. 2007). The results in this study show that the survey effort required to detect a species was similar to that required for robust estimates of nightly activity (i.e. obtaining the 70th percentile of nightly activity during 14 survey nights). However, to estimate the maximum nightly activity an increase of between 18% and 80% (across species) in survey duration was required.

### *2.6.1. Where is best to detect species and estimate activity; ground or nacelle?*

Designing robust ecological surveys is difficult for cryptic taxon, such as bats, but especially when the area of interest varies considerably in a vertical plane (e.g. turbine blade tips can range from 10 m to over 160 m above ground level; RenewableUK 2015). Most bat surveys are conducted at ground level, however for wind farms, guidance has recently been updated to stipulate that surveys at the nacelle (centre of the rotor sweep area) are more important than surveys conducted at ground (e.g. Rodrigues et al. 2014). In this study, species detection was always higher at ground compared to at the nacelle and the probability of detection at the nacelle and not ground was <0.01 for all species, even high flying open aerial foragers such as *Nyctalus noctula*. If monitoring was only conducted at the nacelle, detection for *Pipistrellus nathusii* and *P. pygmaeus,* both incurring high numbers of fatalities from wind turbines (EUROBATS 2014), would not have occurred at 42% and 32% of sites respectively. This would be inconsequential if monitoring at the nacelle detected species within the full rotor sweep area. However given the diameters of blades (e.g. up to 100 m, RenewableUK 2015) and their close proximity to the ground (e.g. 10 m, RenewableUK 2015), species detected at ground and not the nacelle may still occur within the rotor sweep area and be at risk of fatality. Due to the cost and practicalities of monitoring at the nacelle and that there is no significant increase in the information gained even for high flying species, it is questionable whether the updated survey requirements for monitoring at the nacelle are effective. In terms of fatality risk, further research is required to determine whether activity monitored at ground or the nacelle is a better predictor for each high risk species. Site specific monitoring at the nacelle may be beneficial at sites where turbines are ‘keyholed’ into woodland habitat. Studies have shown that the flight height of bats increases to above the canopy in woodland environments (Adams, Law & French 2009; Staton & Poulton 2012; Müller et al. 2013), which may result in more activity at the nacelle comapred to ground at wind turbines keyholed in woodland.

### *2.6.2. When is night time curtailment of turbines most effective?*

The patterns of bat activity throughout the night observed in this study were similar to those observed by Swift (1980) at non wind farm sites during summer and early autumn. These activity patterns have important implications for the timing and duration of curtailment (switching-off) of wind turbines (Arnett et al. 2009; Lagrange

2013). For curtailment mitigation, there is a trade-off between reducing the risk of bat fatalities and optimizing power output. Curtailing turbines only during the peak hours of bat activity is of considerable interest to consultants, planning authorities and wind operators. Presuming fatality risk is directly related to activity, based on results from this study, a reduction in fatality risk by 50% would require turbines to be curtailed between the first and fifth hour after sunset. To achieve a further 25% reduction, turbines would need to be curtailed between the first and six hour after sunset.

### *2.6.3. How is species composition and activity affected by survey effort?*

The probability of detection was consistent with studies at non-wind farm sites (e.g. Skalak, Sherwin & Brigham 2012), where common species were detected with high certainty (at least 80% of sites within 95% CI) after 2-5 nights and rare species after 7-14+ nights dependent on the spatial replication. For rarer species, such as *Pipistrellus nathusii,* the probability of species detection was improved substantially by increasing the number of turbines monitored, whereas for other species this had less effect. This suggests that for common species site coverage is greater, due to higher numbers, than the rarer species. Following current EUROBATS guidelines all potential turbine locations should be monitored at a site (Rodrigues et al. 2014). However, the importance of monitoring all locations is likely to be dependent on the variability of habitat within a site and whether rare species are within range. If habitat variability is low and only common species are within range, monitoring all potential turbine locations may not be necessary.

Although acoustic methods are widely used, little attention has been given to the survey effort required for robust estimates of bat activity. This is perhaps due to the high variability of activity, dependent on many variables, such as; seasonality

(Russ et al. 2003), weather (Erickson & West 2002), habitat (Lintott et al. 2015) and the monitoring device used (Adams et al. 2012). Despite this variability, if activity is to be used to assess the importance of a site for bats and determine mitigation in reponse to the development of a site, an evidence based approach for robust survey methods is required. This has become of increasing concern given that high bat fatalities that occur at some wind farm sites and measuring species composition and their activity is the main method by which mitigation is determined (e.g. Rodrigues et al. 2014). After monitoring activity across a wide number of sites within the same season, this study found that the minimum survey effort required to detect a species is sufficient for robust estimates of activity (i.e. obtaining the 70th percentile of nightly activity).

When monitoring to obtain the maximum activity, it was found that the effects of increasing spatial and temporal effort was similar when monitoring at both ground and the nacelle and across four species known to be at high risk of fatality at wind farms (EUROBATS 2014). In terms of temporal replication, surveying beyond ten nights (comparable with current EUROBATS guidelines within the peak fatality period; Arnett et al. 2008, Rydell et al. 2010b) for a further four nights, resulted in a increase of the maximum nightly activity of at most 24% at ground and 29% at the nacelle. This suggests, the amount of temporal replication in EUROBATS guidelines are currently robust for estiamtes of activity. However, it is currently unclear which measure (mean, 70th percentile or maximum activity) is the better predictor of fatality.

Bat surveys are costly therefore there is often a trade-off between monitoring for a sufficient duration of time and monitoring a sufficient number of turbines to achieve the best estimate of species composition and their activity levels. For example, if resources are limited to 12 detector nights (maximum possible example from this study), surveys could be conducted at either: one turbine for 12 nights, two turbines for six nights or three turbines for four nights. For monitoring at ground level, the optimal survey design found in this study to obtain the highest levels of nightly activity was to monitor two turbines for six nights for *Pipistrellus pipistrellus*, *P. nathusii* and *Nyctalus noctula,* and monitoring three turbines for four nights for *P. pygmaeus*. Therefore an intermediate protocol of moderate spatial and temporal monitoring, as opposed to monitoring all potential turbines locations, as per current EUROBATS guidelines (Rodrigues et al. 2014), may result in better estimates of bat activity. For monitoring at the nacelle, the optimal survey design for all four species was to monitor one turbine for 12 nights. This suggests that temporal over spatial replication was more important when monitoring at the nacelle. Survey design may need to be adjusted on a site specific basis to obtain optimal estimates. This may be dependent on the amount of variation in habitat composition and density of linear features which influences bat density and distribution (Swift, Racey & Avery 1985; Limpens & Kapteyn 1991; Verboom & Huitema 1997). In this study, spatial replication was limited and therefore recommendations for survey design based on spatial replication are restricted.

EUROBATS guidelines suggest surveying for ten survey nights within the three month peak fatality period. In this study, the ten survey nights were condensed to within a one month period due to survey limitations. Given seasonal (Russ et al. 2003) and weather (Erickson & West 2002) variation in bat activity, species composition and bat activity are likely to vary more within a three month period than within a one month period used in this study. If this increased variability conspires to be true, then there may be some differences between this study and those obtained following the guidelines. Following EUROBATS guidelines, the greater spread of survey nights across the full survey period, may mean that estimates of species composition and bat activity produced, better represent the three month fatality period than presented here. However, these estimates may be less accurate and precise than results here, due to the potentially higher variability in species presence and activity across a longer time period.

### *2.6.4. Conservation implications*

This study has provided additional evidence to support the development of guidelines for surveying for bats at wind farm sites. Using EUROBATS survey guidelines as a model, this study has shown that current guidelines for survey duration are likely to be sufficient at most sites. Results here show the importance of monitoring at ground level for four of the high risk species and that monitoring at the nacelle provides little additional information from monitoring at ground. Indeed, if the recent trend towards prioritizing monitoring at the nacelle is established as best survey practice, then evidence from this study shows that in the absence of ground monitoring, species composition and activity levels will be underestimated. This may lead to insufficient mitigation and potentially higher fatalities than expected.

The results presented in this study establish the survey effort required to determine robust estimates of species composition and activity at operational wind farms. However, current evidence finds no relationship exists between activity at pre-construction and operational sites (Hein, Gruver & Arnett 2013). If monitoring continues to be required at pre-construction sites to determine post-construction fatality risk and mitigation strategies (e.g. Rodrigues et al. 2014), a relationship needs to be substantiated between pre and post construction activity, or monitoring needs to be conducted at operational wind farms to determine fatality risk and design appropriate mitigation.

# Chapter three:

# 3. Evidence of bat attraction to wind turbines

## 3.1. Summary

Despite over a decade of research on bat fatalities at wind farms, little is known about the proximate cause of fatalities. It is currently unknown whether bats are killed coincidentally or due to their attraction to turbines. This is a critical knowledge gap for conservation efforts and for wind operators and planning authorities attempting to minimize fatalities.

Bats were monitored acoustically for 555 detector nights using a paired design (controls located at a mean distance of 586 SD 275 m from turbines) at 20 wind farms between July and October in 2011-2013 across Britain. The focus of the study is on *Pipistrellus pipistrellus* and *P. pygmaeus* which have incurred the most fatalities at wind farms across Europe. Results here show that *Pipistrellus pipistrellus and P. pygmaeus* activity at turbines compared to controls is 46% (6.3 ± 1.3 SE mean passes per night *c.f.* 3.4 ± 1.3 SE) and 34% (4.0 ± 1.4 SE *c.f.* 2.7 ± 1.4 SE) higher respectively. Given habitat and elevation were consistent between paired turbines and controls and monitoring was conducted on the same nights, higher activity at turbines compared to controls, provides evidence for the first time that these two species are attracted to wind turbines. This attraction may be linked to increased insect abundance at turbines or their exploratory behaviour around large prominent structures in the landscape where there are typically few linear features. These results have important implications for mitigating bat fatalities at wind farms, since most monitoring to inform mitigation is conducted at pre-construction sites, yet these results suggests that this activity will not relate to activity at operational wind turbines. Results here provide evidence that monitoring should be conducted at operational wind farms to inform mitigation to reduce fatalities. These findings also

provide some explanation as to why bats are being killed, at some sites in high numbers.

## 3.2. Introduction

The number of wind turbines across the globe is increasing rapidly as the demand for renewable energy increases (Global Wind Energy Council 2014). Whilst wind power is important to minimize our use of fossil fuels and reduce anthropogenic impact on the climate, other environmental issues with wind power have become apparent, including, noise and visual pollution (Julian, Jane & Davis 2007; Harrison 2011) and negative impacts on wildlife (Drewitt & Langston 2006; Thomsen et al. 2006). Wildlife impacts can be both indirect (e.g. habitat fragmentation and displacement; Minderman et al. 2012) and direct (e.g. collision risk to birds and bats; Arnett et al. 2008; Rydell et al. 2010b) with direct impacts on bats having raised the most concern. Given the projected increase in wind power (Global Wind Energy Council 2014), these impacts are likely to increase.

Despite over a decade of research on bat fatalities at wind farms, little is known about the proximate cause of fatalities. Both local and migrant bat populations are killed at wind farms in Europe (Voigt et al. 2012). It is assumed that migrants are killed incidentally (Baerwald 2006; Cryan & Barclay 2009), and local populations are killed due to their attraction to turbines (Kunz et al. 2007b; Horn, Arnett & Kinz 2008). Possible reasons for attraction include the following: the emission of ultrasound (Schmidt & Joermann 1986; Kunz et al. 2007b), the movement of the blades (Kunz et al. 2007b); increased prey availability (Rydell et al. 2010a; Long, Flint & Lepper 2011; Cryan et al. 2014); and investigation of turbines as potential roosts (Cryan & Barclay 2009).

The evidence for bats being attracted to turbines includes; infra-red imagery of bats interacting with blades (Horn, Arnett & Kinz 2008) and changing their orientation relative to turbines on approach (Cryan et al. 2014). Tree-roosting bats have also been found to be attracted to tall structures (e.g. telecommunication towers) in comparison to other habitats (i.e. open fields and woodlots during migration, Jameson and Willis 2014). Attraction to turbines has only been experimentally assessed at small wind turbines, where bat activity was lower within 5 m of operational turbines compared to non-operational turbines at low wind speeds (e.g. $\leq 5$ m $s^{-1}$; Minderman et al. 2012).

Fatalities among European studies, are positively correlated with elevation (Rydell et al. 2010b), the percentage cover of woodland (Rydell et al. 2010b) and negatively correlated with the distance to forested areas (Santos et al. 2013). Overall, topography, defined by broad habitat classifications (i.e. crop, forest, marsh and pasture) was also significant at predicting the numbers of fatalities (Rydell et al. 2010b), which is perhaps explained by different foraging habitat preferences between species (e.g. Nicholls & Racey 2006a; Walsh & Harris 1996). The importance of elevation for bat activity is however, less well studied. Generally, as elevation increases fewer insects and bats occur (e.g. Grindal & Brigham 1999). The higher numbers of bat fatalities at increased elevations, has been related to high altitude feeding on migratory insects (Rydell et al. 2010a). Elevation generally, also affects the sex-ratio of bats, where a male-bias occurs at higher elevations, postulated to be due to females preferentially selecting lower elevations with optimal foraging, particularly during the reproductive period (Grindal et al. 1999; McGuire & Boyle 2013).

Understanding if bats are attracted to large turbines is fundamental to minimising fatalities and understanding the scale of wind-farm related bat deaths. This evidence will also aid in mitigating of bat fatalities, since current methods to assess risk are predominately conducted pre-construction, which assumes bat activity does not significantly differ at sites with and without operational turbines. Here, a paired turbine and control design was used across a large sample of wind farms in Britain, to assess whether bat activity is higher at turbines compared to controls and whether this is occurs universally across all sites, or is dependent on turbine and site features.

## 3.3. Methods

### *3.3.1. Site selection*

Bat activity was surveyed using static detectors at 20 wind farm sites across Britain in farmland (n = 9) and moorland (n = 11) habitats. The numbers of sites survey in each year were 4 in 2011, 6 in 2012 and 10 in 2013. In each year, respectively, surveys were conducted between 29 July and 29 September, 10 July and 9 October and 11 July and 20 October. Overall, the mean number of wind turbines at each site was 13 SD 5 (range 6 to 22), the mean tower height was 58 SD 16 m (range 35 to 80 m) and the mean blade length was 37 SD 9 (range 20 to 50 m).

### *3.3.2. Acoustic monitoring and bat identification*

Acoustic monitoring of bat activity was conducted at ground level as described in section 2.3.2. Among the three acoustically monitored turbines, a single turbine was randomly selected and paired with a control location. The control location was similar in habitat, elevation, distance the nearest linear feature (e.g. hedgerows or treelines), and under similar management regimes. As turbines were selected randomly, the control was located as far away as possible whilst complying with the above restrictions. The paired turbine was therefore not always the closest turbine to the control; at a mean distance of 586 SD 275 m (range 222 to 1,369) from the nearest turbine and 1,330 SD 993 m (range 222 to 4,150) from their paired turbine. Although, in some cases, the control site was relatively close to the paired turbine, there were no circumstances where the control site was within the rotor-sweep area of the turbine. There was sufficient distance between paired controls and turbines to enable an assessment of any attraction to the rotor-sweep area (where fatalities are possible) rather than to the wind farm site in general. Bat calls were identified as described in section 2.3.3.

### *3.3.3. Habitat analyses*

All habitat analyses were conducted in ArcGIS v.10 (ESRI 2011). Land Cover Map 2007 (LCM07, Morton et al. 2011) and OS MasterMap Topography Layer (EDINA Digimap Ordnance Survey Service 2015) were used to extract the following habitat variables within the surrounding landscape; bog, fen, marsh and swamp, farmland, freshwater, rough grassland, scrub and woodland (total and coniferous). The OS MasterMap Topography Layer was used to derive data on roads, buildings, structures, streams and vegetated channels. Habitat categories important for bat activity and foraging (e.g. Walsh & Harris 1996; Davidson-Watts et al. 2006) were identified and measured to show that landscape metrics were similar between paired turbines and controls (Table 4.1). The distance to linear features, important for bat commuting and foraging routes (e.g. Limpens & Kapteyn 1991), was calculated for all paired turbine and control locations. Linear features were defined as woodland edge, streams, hedgerows, scrub edges and vegetated channels (identified using Google Earth; ©2007 GoogleTM). Due to linear features being slightly closer to controls compared to turbines (Table 4.2), this landscape metric was included in all models. Using LCM07, habitats were classified into broad habitat classes; farmland, lowland moorland and upland moorland, based on the dominant habitat type and elevation (centre of the wind farm, extracted from Google maps; ©2007 GoogleTM). Moorland sites above 250 m were classified as upland and sites below 250 m were classified as lowland (Figure 4.1; Backshall et al. 2001).

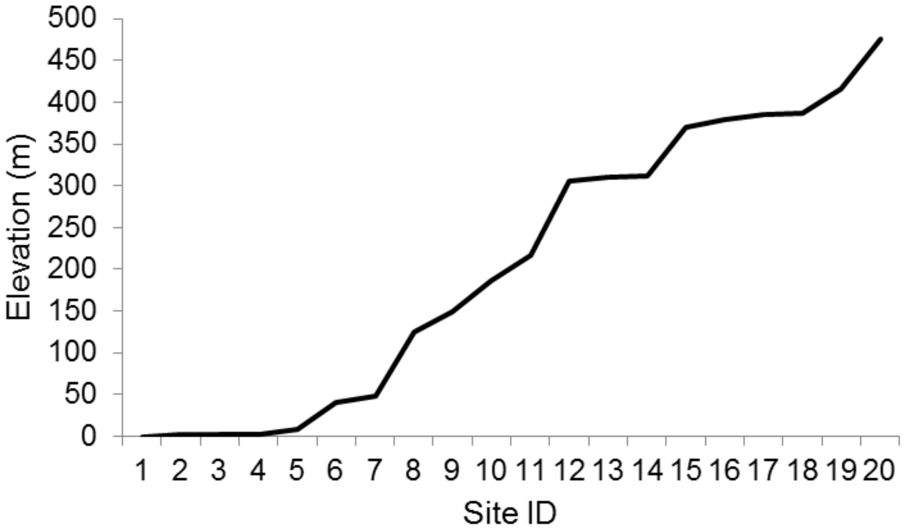

Figure 4.1. Elevation (m) of the 20 sites showing the natural break between sites located at 217 m and 306 m which divides farmland and lowland moorland with upland moorland sites (the exception being one farmland site at 310 m). Site ID is not comparable to Table 1.2 to prevent identification of wind farm sites.

Table 4.1. Definition of habitats used in models, extracted from the [1]Land Cover Map 2007 (Morton et al. 2011) and from [2]OS MasterMap (EDINA Digimap Ordnance Survey Service 2015).

| Habitat | Description |
|---|---|
| Built environment | Buildings, structures and roads[2] |
| Coniferous woodland | Coniferous woodland[1] |
| Farmland | Arable and pastoral farmland[1] |
| Freshwater | Inland water and coastal water[1] (includes waterways and standing water[2]) |
| Riparian | Inland water, coastal water (including waterways and standing water), bog, fen, marsh and swamp[1] |
| Semi-natural habitat | Rough grassland and scrub[1] |
| Total woodland | Coniferous, deciduous or mixed woodland[1] |

Table 4.2. Summary of landscape metrics (minimum distance and percentage cover within 50 m of detector) for paired turbine and control locations across 20 wind farm sites.

| Landscape metric | Turbines mean SD | Control mean SD |
|---|---:|---:|
| %Built environment 50 m | 6 (7) | 3 (4) |
| %Farmland 50 m | 39 (47) | 57 (46) |
| %Riparian 50 m | 3 (11) | 5 (13) |
| %Semi-natural 50 m | 11 (28) | 7 (17) |
| %Woodland 50 m | 2 (9) | 0 (0) |
| Minimum distance built environment (m) | 84 (132) | 109 (102) |
| Minimum distance coniferous woodland (m) | 1,723 (1,846) | 1,511 (1,593) |
| Minimum distance farmland (m) | 325 (434) | 230 (427) |
| Minimum distance freshwater (m) | 3,005 (2,823) | 3,072 (2,823) |
| Minimum distance to LF (m) | 281 (296) | 186 (189) |
| Minimum distance riparian (m) | 164 (122) | 120 (96) |
| Minimum distance semi-natural (m) | 692 (894) | 569 (918) |
| Minimum distance woodland (m) | 704 (489) | 565 (395) |

### 3.4. Statistical analyses

All statistics were undertaken using R (v.3.0.3; R Core Team 2014) with the package lme4 (v.1.1.7; Bates et al. 2014). A GLMM was used to assess whether there was any difference in the nightly activity of *Pipistrellus pipistrellus* at turbines compared to controls. The number of *Pipistrellus pipistrellus* passes was fitted as the dependent variable, with a negative binomial error structure and log-link function. The following variables were identified from the literature (Cryan and Barclay 2009, Jameson & Willis 2014, Minderman et al. 2012, Rydell et al. 2010b) as potentially important predictors influencing bat activity at turbines: distance to linear features (m), mean nightly wind speed (m $s^{-1}$), tower height (m), distance between ground to blade tip (m) and elevation (m).

There was high collinearity between the two variables; tower height, and the distance from the ground to the blade tip. Therefore, the predictor that had the highest $R^2$ value was used when modelled separately. Elevation was also strongly associated with habitat type (above 250 m, 90% of sites were upland moorland and 10% farmland; below 250 m, 67% of sites were farmland and 33% lowland moorland) therefore elevation, which has fewer degrees of freedom was used in models.

Since differences between bat activity at turbines compared to controls may be dependent on wind speed (i.e. prey accumulations which are influenced by wind speed), tower height (i.e. investigation of turbines as potential roosts) and elevation (i.e. prey accumulations which are influenced by elevation), two-way interactions between each variable and location (a factor with two levels: control and turbine) were fitted as fixed effects. Site and night nested within site were fitted as random effects to account for spatial and temporal autocorrelation in addition to an observational level random effect to account for the over-dispersion. Nights within a site with no activity were excluded to improve model fit. The same model was fitted for the nightly activity of *Pipistrellus pygmaeus*.

There was sufficient bat activity recorded to permit analysis of the two most common species in Britain; *Pipistrellus pipistrellus* and *P. pygmaeus,* both of which are at high risk of fatality at wind farms (Rydell et al. 2010b). For *Pipistrellus*

*pipistrellus,* distance between the ground and the blade was used and for *P. pygmaeus* tower height was used as a predictor variable. For ease of model interpretation, any two-way interactions which were not significant were dropped from the full model. All models were assessed for model fit by checking that standardised residuals were normally distributed, predictor variables were not correlated and there was no evidence of over-dispersion or heteroscedasticity. Predictions were made from the reduced model without non-significant two-way interactions and predictors other than those of interest were set at their mean values. Standard errors of predictions were calculated across the mean of the random effects. In all models, significance was assessed using log-likelihood ratio tests of the full model compared to the alternative model.

## 3.5. Results

In total, 28,368 bat passes were recorded at turbines and 10,465 bat passes were recorded at controls, over 766 detector nights and 238 unique nights. Bats from the following genera were detected: *Barbastella, Eptesicus, Myotis, Nyctalus, Pipistrellus, Plecotus* and *Rhinolophus. Pipistrellus* spp. activity accounted for 93% of all activity (*P. pipistrellus* 71%, *P. pygmaeus* 21%, and *P. nathusii* 1%).

For both *Pipistrellus pipistrellus* and *P. pygmaeus* there were no significant interactions between activity at the turbine and control and any predictor variables assessed, hence the effect of predictor variables (i.e. distance to linear features, elevation, tower height and wind speed), on *P. pipistrellus* and *P. pygmaeus* activity is the same at both turbine and control sites. For both species activity was significantly higher at turbines compared to controls (Table 4.3). For *Pipistrellus pipistrellus* mean activity per night was 46% higher (6.3 ± 1.3 SE passes *c.f.* 3.4 ±

1.3 SE) and for *P. pygmaeus* mean activity per night was 34% higher at (4.0 ± 1.4 SE passes *c.f.* 2.7 ± 1.4 SE).

Table 4.3. Summary of GLMM to assess *Pipistrellus pipistrellus* (n = 20 sites) and *P. pygmaeus* (n = 18 sites) activity at turbines compared to controls in relation to predictor variables. Metrics of all predictors are in m with the exception of wind speed which is measured in m $s^{-1}$.

| Species | Predictor variables | Estimate ± SE | Log-likelihood | $\chi^2$ | df | AIC | *P* value |
|---|---|---|---|---|---|---|---|
| *P. pipistrellus* | dist. linear feature | -0.80 ± 0.47 | -2139 | 18.65 | 1 | 4296 | <0.001 |
| | elevation | 0.57 ± 0.26 | -2132 | 4.44 | 1 | 4242 | 0.035 |
| | ground to blade | -0.96 ± 0.47 | -2132 | 3.88 | 1 | 4282 | 0.049 |
| | turbine *c.f.* control | 0.61 ± 0.16 | -2137 | 14.74 | 1 | 4292 | <0.001 |
| | wind speed | -1.19 ± 0.13 | -2169 | 77.73 | 1 | 4355 | <0.001 |
| *P. pygmaeus* | dist. linear feature | -0.91 ± 0.19 | -1275 | 24.88 | 1 | 2567 | <0.001 |
| | elevation | 0.92 ± 0.31 | -1266 | 8.14 | 1 | 2550 | 0.004 |
| | tower height | -1.32 ± 0.38 | -1267 | 1.00 | 1 | 2552 | 0.002 |
| | turbine *c.f.* control | 0.42 ± 0.18 | -1265 | 5.65 | 1 | 2548 | 0.017 |
| | wind speed | -1.09 ± 0.15 | -1287 | 49.26 | 1 | 2591 | <0.001 |

Mean nightly wind speed, the distance to the nearest linear feature, elevation and the distance between the ground and turbine blades (*Pipistrellus pipistrellus*) or turbine tower height (*P. pygmaeus*) were all significant predictors of *P. pipistrellus* and *P. pygmaeus* activity. For both species, activity increased, as the distance to the nearest linear feature decreased (Figure 4.2), as elevation increased (Figure 4.3), as the distance between the ground and blade tip (*Pipistrellus pipistrellus*) or tower height (*P. pygmaeus*) decreased (Figure 4.4) and as wind speed decreased (Figure 4.5). Among these predictor variables for *Pipistrellus pipistrellus* wind speed was most important predictor and elevation the least. *Pipistrellus pipistrellus* activity was 91% higher at turbines when wind speed was 2 m $s^{-1}$ compared to 6 m $s^{-1}$ (8.2 passes per night ± 1.3 SE *c.f.* 0.8 ± 1.4 SE). For *Pipistrellus pygmaeus* turbine tower height was the most important predictor and the distance to the nearest linear feature the least. *Pipistrellus pygmaeus* activity was 94% higher at 40 m turbine towers compared to 80 m turbine towers (14.0 passes per night ± 1.6 SE, *c.f.* 0.8 ± 1.8 SE).

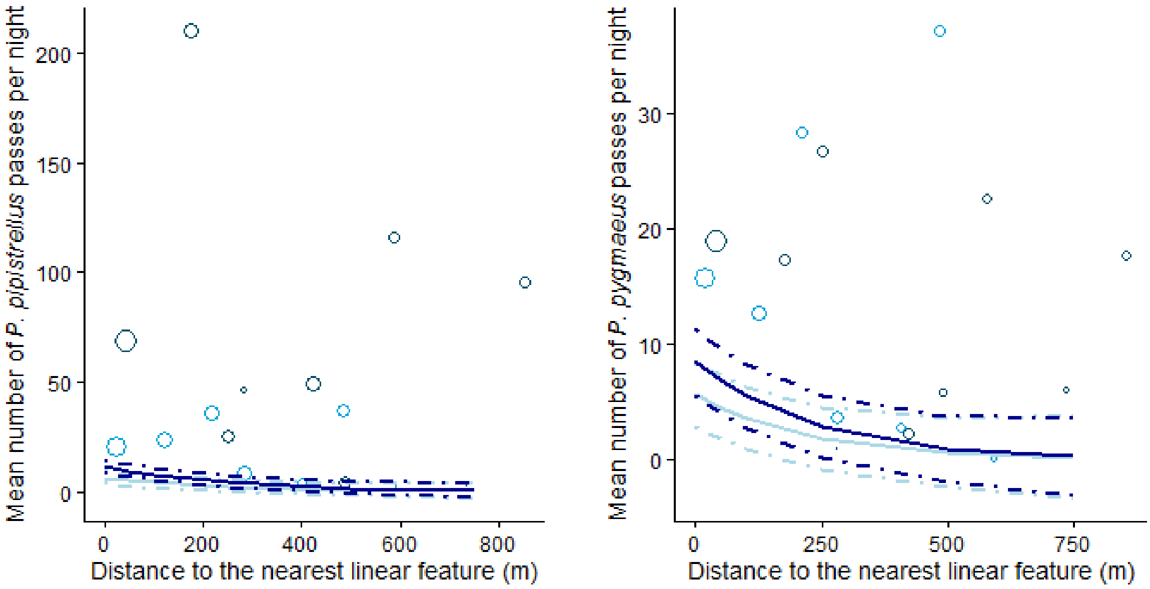

Figure 4.2. Predictions from a GLMM ± 95% CI of the nightly activity of; *Pipistrellus pipistrellus* (n = 20), and *P. pygmaeus* (n = 17), in relation to the distance to the nearest linear feature at turbine (dark blue) and at control (light blue) locations. 95% CI are estimated for the mean across all random effects.

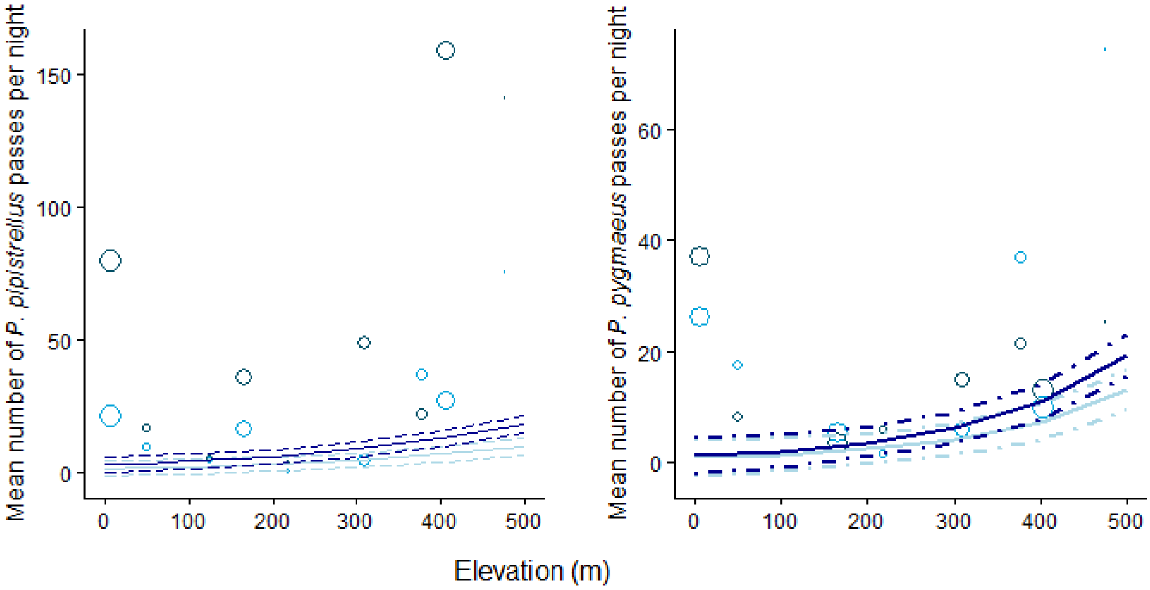

Figure 4.3. Predictions from a GLMM ± 95% CI of the nightly activity of; *Pipistrellus pipistrellus* (n = 20), and *P. pygmaeus* (n = 17), in relation to elevation at turbine (dark blue) and at control (light blue) locations. 95% CI are estimated for the mean across all random effects.

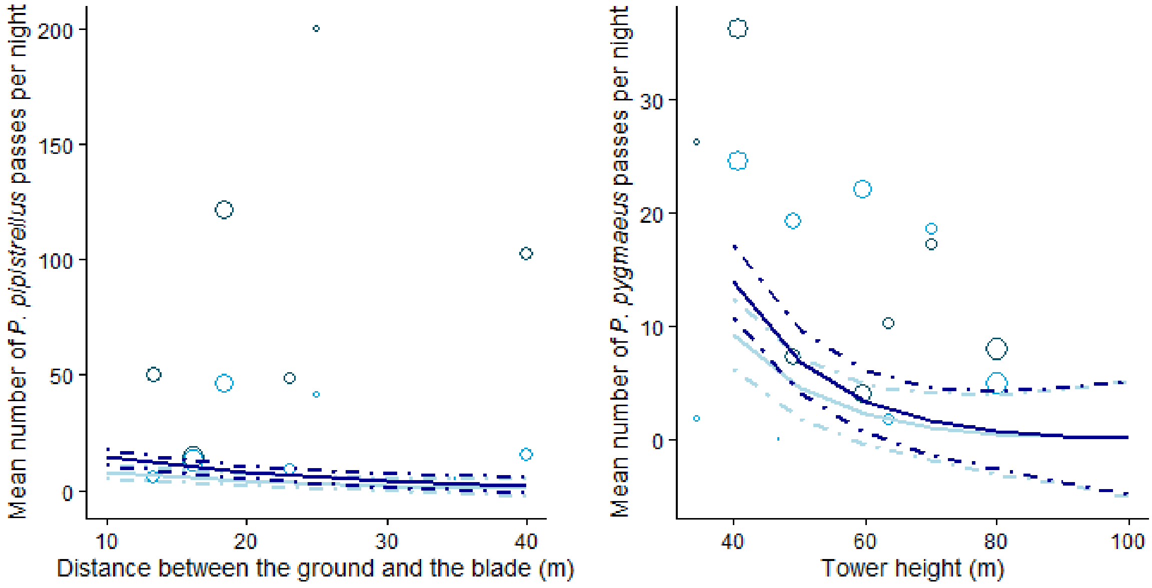

Figure 4.4. Predictions from a GLMM ± 95% CI of the nightly activity of; *Pipistrellus pipistrellus* (n = 20), and *P. pygmaeus* (n = 17), in relation to the distance between the ground to the blade tip (*P. pipistrellus*) and the turbine tower height (*P. pygmaeus*) at turbine (dark blue) and at control (light blue) locations. 95% CI are estimated for the mean across all random effects.

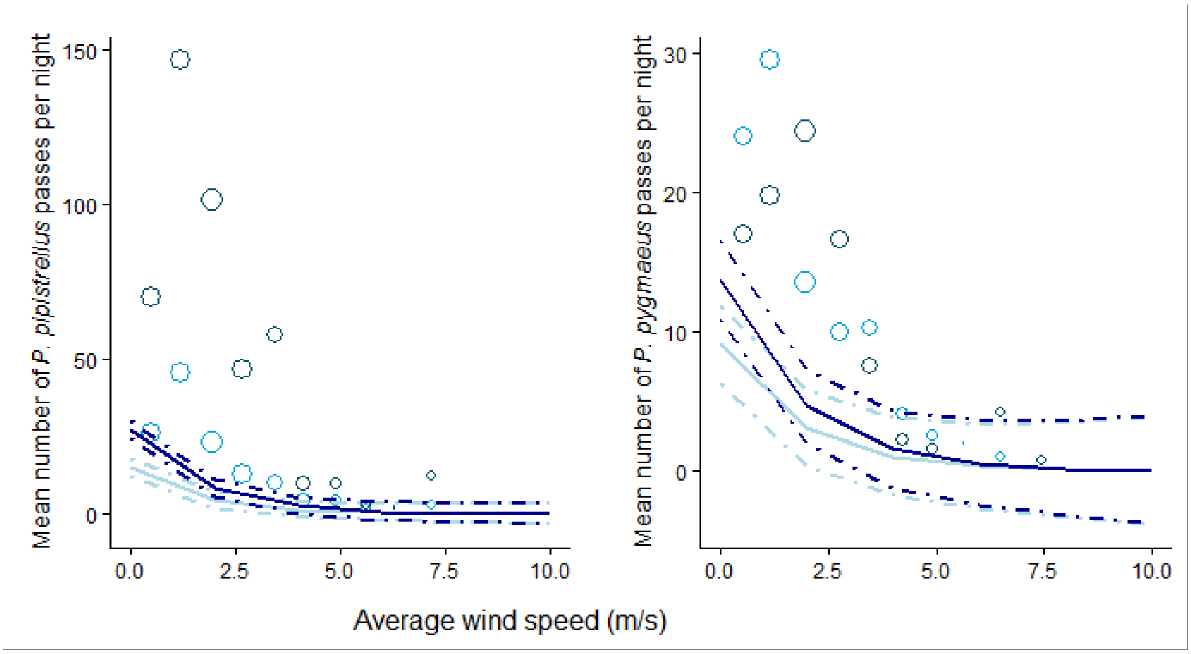

Figure 4.5. Predictions from a GLMM ± 95% CI of the nightly activity of; *Pipistrellus pipistrellus* (n = 20), and *P. pygmaeus* (n = 17), in relation to average nightly wind speed at turbine (dark blue) and at control (light blue) locations. 95% CI are estimated for the mean across all random effects.

## 3.6. Discussion

This study shows for the first time that the activity of two bat species (*Pipistrellus pipistrellus* and *P. pygmaeus*), which incur high numbers of fatality at wind farms*,* (EUROBATS 2014) is higher at turbines compared to controls (located at a mean distance of 586 m from the nearest turbine). For both species, higher activity at turbines was not dependent on elevation, the distance to linear features, wind speed or turbine tower heights. Considering turbine and control locations were paired in terms of habitat and elevation and were also monitored on the same nights, increased activity at turbines is likely to be explained by the attraction of bats to wind turbines. The attraction found in this study could be attributed to either; increased numbers of bats at turbines or an increase in the duration of time spent at turbines by an equal number of bats relative to the control. In either scenario, the risk of collision or barotrauma, as a result of attraction is likely to be increased. The high numbers of bats killed by wind turbines, especially for species killed locally rather than during migration, such as *Pipistrellus pipistrellus* (Voigt et al. 2013), may be attributed to this attraction.

Bat attraction to turbines has been suggested in other studies; Cryan et al. (2014) found that tree-roosting species in North American changed their orientation to approach operational wind turbines from downwind at low wind speeds. This was speculated to be due to streams of air flowing downwind of the nacelle, which facilitates bat foraging consistent with behaviour observed at trees (Cryan et al. 2014). Similarly Horn et al. (2008) observed bats interacting with moving turbine blades and suggested bats were attracted to them. The attraction of bats to other large anthropogenic structures has also been found (Jameson and Willis 2014). In their study, they found that fewer feeding buzzes occurred at telecommunication towers compared to other habitats, suggesting bats were attracted to structures for reasons other than foraging, such as investigation for roosting or mating sites. Whilst in this study, attraction to turbines was not increased at taller towers there was higher activity at turbine and control sites as elevation increased. This is contrary to many other studies at non-wind farm sites, where the reverse is often found (e.g. Grindal et al. 1999, Cryan et al. 2000). In these studies, higher activity was found at lower elevations and speculated to be due to increased foraging opportunities and females were shown to preferentially select low elevation sites particularly during lactation. This difference between wind farm and non-wind farm sites may therefore reflect bats investigating turbines, which are located at higher elevation due to their prominence in the landscape, as potential roost or mating sites. The higher activity at higher elevation sites found in this study, supports previous studies where there was a correlation between fatalities and elevation (e.g. Rydell et al. 2010b) and increased fatalities at sites located on hilltops or ridgelines (Brinkmann, Schauer-Weisshahn & Bontadina 2006; Dulac 2008). The main hypothesis to date explaining this is that high elevation sites coincide with bat migratory routes (Arnett et al. 2008; Cryan & Barclay 2009; Cryan 2011). However in the study by Rydell et al. (2010b) one of the fatality species was *Pipistrellus pipistrellus* which is not considered migratory and carcasses found at turbines are generally from local populations (Voigt et al. 2013), hence, the increased activity in the present study is consistent with Rydell et al. (2010b) and may be attributed to greater activity at higher elevation wind farm sites.

In the last few decades turbines have increased in height (RenewableUK 2015), with older wind farm sites being more likely to have shorter turbine towers. Planning restrictions at wind farms have increased over time and bat surveys have only become a requirement as part of the Environmental Impact Assessment in the last decade (e.g. Rodrigues et al. 2008). Older sites which have shorter turbine towers may therefore be more likely to have been developed in more optimal foraging habitats for bats, potentially supporting higher densities. This may explain why both *Pipistrellus pipistrellus* and *P. pygmaeus* activity recorded in this study, was higher at sites (turbine and control locations) with shorter turbine towers. The relationship between activity and tower height was stronger for *Pipistrellus pygmaeus,* which are considered to be more selective in their habitat preferences compared to *P. pipistrellus* (Vaugh et al. 1997). Whilst higher activity at shorter towers may contradict the hypothesis that bats are attracted to large prominent structures, at a wind farm site, turbines are typically all one height and therefore there isn't a preferential attraction to the tallest turbine, but rather, turbines are the tallest structures in the landscape.

The negative relationship between activity and turbine tower height may also be explained by bats being attracted to moving blades as opposed to the turbine towers (e.g. Cryan et al. 2014). The blades of shorter turbines are within the typical flight height of *Pipistrellus pipistrellus* and *P. pygmaeus* (Russ 2012) (e.g. 40 m towers had blade tips 13 m to 20 m above ground level), whereas at taller turbines blades are above their typical flight height (e.g. 80 m towers had blade tips 34 m to 59 m above ground level) and less interaction is likely to occur. This contradicts previous research investigating predictors of fatalities (e.g. Rydell et al. 2010b) where the total number of fatalities increased at sites with higher turbine towers. This difference may be due to the study comprising fatalities from additional species (i.e. *Nyctalus noctula, Pipistrellus nathusii* and *N. leisleri*) which have higher flight heights than *P. pipistrellus* and *P. pygmaeus* (Russ 2012) and are long distance migrants (Hutterer et al. 2005).

Overall, bat activity was negatively correlated with wind speed, consistent the relationship between wind speed and fatality from many other studies (e.g. Kerns, Erickson & Arnett 2005, Arnett et al. 2008, Rydell et al. 2010b). The importance of weather in predicting bat activity is found in most studies at wind farm and non-wind farm sites (e.g. Erickson & West 2002), perhaps the importance of wind in particular at wind farm sites is due to the typical higher exposure of wind farm sites compared to more typical bat surveys at sheltered locations. At wind farm sites wind speed will have partiulcar importance for *Pipistrellus pipistrellus* and *P. pygmaeus* which forage on small Diptera which are easily displaced with the wind. If foraging is an important reason for their attraction to turbines this explains why low wind speeds predict high activity levels for both species.

In a study by Minderman et al. (2012) bats were shown at close proximity (<5 m) to avoid small wind turbines (typically with toer heights up to 12 m, RenewableUK 2011). Avoidance of small wind turbines may be as a result of an inability of bats' to build an effective acoustic image in close proximity, due to noise created by the relatively fast moving blades and their echolocation calls being erratically reflected (Long, Flint & Lepper 2010a; Minderman et al. 2012). It is not known if obtaining accurate acoustic images of operating turbines differs between small and large wind turbines, but it is likely that larger and slower moving blades are easier to detect, with less echolocation reflection and this may explain the differences found between the two studies.

### *3.6.1. Conservation implications*

This is the first study to show that bats are attracted to turbines. With this fact established the validity of surveying at pre-construction wind farm sites to establish fatality risk is questionable. This is supported by Hein, Gruver & Arnett (2013) who found a lack of relationship between activity at pre-construction and operational sites. Results in the present study suggest that since bats are attracted to turbines, determining fatality risk can only be accurately estimated at operational wind farms sites. In the absence of post-construction monitoring appropriate mitigation may not be implemented. This may lead to fatality numbers exceeding estimates from pre-construction surveys.

This study shows that bat activity and ultimately fatality are due to direct attraction to turbines and not as a result of incidental encounters. This confirms many hypotheses which have speculated this to be the case based on the composition and high numbers of carcasses found beneath turbines over the last two decades (e.g. Cryan & Barclay 2009). This study only considered two of the most common species and further research would be required to establish if attraction occurs in other species incurring high numbers of fatalities at turbines. Furthermore, understanding the distance at which attraction occurs will aid in siting turbines beyond the attractive zone of important roosts and bat habitats.

# Chapter four:

# 4. Assessing insect abundance and bat attraction to turbines

## 4.1. Summary

Bat casualties at wind turbine sites pose conservation concerns. Bats are attracted to turbines and fatality rates vary between sites, therefore, understanding why attraction occurs and the factors predisposing particular sites to increased fatality risk is essential to reduce fatalities and mitigate for them. An increase in insect accumulations around the nacelle (centre of the rotor sweep area) and foraging activity of bats is one of the main hypotheses used to explain why bats are attracted to wind turbines. To test this hypothesis, malaise traps were used at 18 wind farm sites across Britain to sample insect abundance and biomass. Traps were deployed at one randomly selected turbine and a paired control location at each site. Concurrently, bat foraging rate (feeding buzz ratio) and activity were compared with nocturnal Diptera abundance and biomass. The analysis focussed on *Pipistrellus pipistrellus* and their main prey (nocturnal Diptera), since this species has incurred the highest recorded numbers of fatalities across Europe. Nocturnal Diptera abundance declined as wind speed increased, but this rate of decline was significantly greater at turbines compared to controls. At low wind speeds of 4 m $s^{-1}$ (above the cut-in wind speed when most turbines are operational and high numbers of fatalities are found to occur), nocturnal Diptera abundance was 61% lower at turbines compared to controls. Nocturnal Diptera biomass was significantly lower at turbines compared to controls regardless of wind speed. Nocturnal Diptera abundance and biomass did not predict the activity or foraging rate for *Pipistrellus pipistrellus* at turbines when monitored at ground level. Given that *Pipistrellus pipistrellus* fatalities are more likely to occur within the lower rotor sweep area, close to ground level, it is unlikely that increased insect abundance at the nacelle is the reason for attraction to turbines (evidenced in Chapter three). These results

contradict the hypothesis that bats are attracted to turbines to forage on insects, particularly for lower flying specialists such as *Pipistrellus pipistrellus,* and other reasons, such as investigating turbines as potential mating sites, may explain the attraction to turbines.

## 4.2. Introduction

In the last two decades 225,000 wind turbines have become operational in 80 countries across the globe (Global Wind Energy Council 2014). Whilst this is a positive move toward reducing $CO_2$ emissions, fatalities to some bird and bat species have raised conservation concerns (Kerns & Kerlinger 2004; Arnett 2005; Kunz et al. 2007a; Carrete et al. 2009). Understanding the reasons for fatalities is fundamental in addressing this conservation issue. Findings from Chapter three suggest that bats are attracted to turbines. The attraction to turbines has previously been speculated, with a range of reasons for which having already been proposed (Kunz et al. 2007b). Since bat abundance and distribution is typically governed by their prey (e.g. Fukui et al. 2006), one of the main hypotheses for why bats are attracted to turbines, is due to the accumulation of insects there (Corten & Veldkamp 2001; Rydell et al. 2010a; Cryan et al. 2014). The evidence to support this includes, bats observed foraging for insects around wind turbines (Ahlén 2002; Ahlén, Baagøe & Bach 2009; Rydell et al. 2010b) and carcasses found with full stomachs suggesting that they had been foraging relatively recently before being killed (Reimer, Baerwald & Barclay 2008; Valdez & Cryan 2013). Whilst congregations of insects have been observed at wind turbines (Ahlén 2002; Ahlén et al. 2007; Horn, Arnett & Kinz 2008) and insects have been found to accumulate on turbine blades

(Corten & Veldkamp 2001; Dalili, Edrisy & Carriveau 2009; Sagol, Reggio & Ilinca 2013), no known studies have sampled nocturnal insects at wind farms***.***

In Europe and North America, peaks in bat fatalities have coincided with the seasonal migration (August-September) and nightly movements of insects (Arnett et al. 2008; Rydell et al. 2010a). An increase in bat fatalities at taller wind turbines has also been attributed to migratory bats foraging on insects (Barclay, Baerwald & Gruver 2007; Rydell et al. 2010a). High numbers of bat fatalities have also been recorded at wind farms located on hills and along ridges (Fiedler 2004; Kerns, Erickson & Arnett 2005; Rydell et al. 2010b). It has been proposed that this may be due to ‘hill-topping’, a behaviour where male insects, including; butterflies and flies, follow a hill upwards and congregate at the top (Shields 1967) to attract females for mating (Alcock 1987), thereby attracting bats to forage in close proximity to turbines.

Several hypotheses have been proposed to explain insect accumulations around turbines; (1) insects are attracted to the heat generated by turbines (von Hensen 2004; Wright 2004); (2) insects are attracted to the colours commonly used on turbines (Long, Flint & Lepper 2011), which may be partly influence by the ultra-violet reflectiveness of the paints (Young Jr et al. 2003; Long, Flint & Lepper 2011); (3) insects are attracted to the microclimates created around the turbine blades, in particular, the streams of air flowing downwind from turbines, similar to those created around trees at night (Cryan et al. 2014).

Understanding mechanistic drivers for bat distribution at wind turbines is important in developing effective mitigation strategies. This study is novel in sampling for nocturnal insects at wind farms, whilst concurrently measuring bat activity and foraging rates to answer the following questions:

1. Are nocturnal Diptera attracted to turbines?

2. Does habitat and weather predict nocturnal Diptera abundance and biomass at wind farms?
3. Can nocturnal Diptera abundance and biomass at turbines be used to predict the extent of bat foraging activity at turbines?

## 4.3. Methods

### *4.3.1. Site selection*

Insects were sampled and bats surveyed acoustically at 18 wind farm sites across Britain. Sites comprised moorland (n = 7) and farmland (n = 11) and were surveyed between 5 July 2013 and 10 October 2013 (Table 3.1).

Table 3.1 Summary of sites (n = 18) by habitat and country, that were surveyed between 5 July and 10 October 2013 to sample bat activity and insect abundance and biomass.

| Country | Farmland | Lowland moorland | Upland moorland | Totals |
|---|---|---|---|---|
| England | 11 | 0 | 0 | 11 |
| Scotland | 0 | 2 | 1 | 3 |
| Wales | 0 | 0 | 4 | 4 |
| Total | 11 | 2 | 5 | 18 |

### *4.3.2. Insect sampling*

Malaise traps (ez-malaise Traps, BugDorm, Taiwan) were used to catch flying insects, as this method is biased towards sampling Hymenoptera and Diptera, a high component of the diet of many European bat species, especially *Pipistrellus* spp. (Barlow 1997; Vaughan 1997). At each site, a randomly selected turbine and a control location was surveyed. Control locations were selected as far from a turbine as possible within the wind farm, whilst remaining in the same habitat type, land management and elevation as the surveyed turbine (Table 3.2, Table 3.3). Controls were at a mean distance of 597 SD 329 m from the nearest turbine. Two traps were erected perpendicular to each other (north-south direction and east-west direction) to enable sampling from all directions, thereby reducing bias due to differences in wind direction. Traps were opened 30 minutes before sunset and closed 30 minutes after sunrise (in-line with the acoustic surveys). Insects were collected and stored after each night into 70% ethanol, for later identification in the laboratory. Sampling was conducted for a mean of 6 SD 2.2 nights at each site.

All insects were identified to order, with the exception of Diptera, which were identified to the suborder Nemotocera and Brachycera. All Nemotocera were then identified to family and Brachycera were identified to one of the families or categories: Bibionidae, Dolichopodiae, Heleomyzidae, Muscidae, Syrphidae, and ‘Brachycera Other’, following an insect identification guide (Chinery 1993). Diptera were defined as Nemotocera and Brachycera excluding ‘Brachycera Other’. These taxa were chosen for detailed classification due to their importance in the diet of *Pipistrellus pipistrellus* (Swift 1985; Hoare 1991; Vaughan 1997) the most common species in the study and considered vulnerable to fatality at turbines (Rydell et al. 2010b; EUROBATS 2014). Identified Diptera were dried at 60°C for 48 hours to calculate the dry weight biomass. All samples were weighed, and the total abundance and biomass were calculated.

### *4.3.3. Bat acoustic monitoring and identification*

Bat activity was monitored at ground level (~2 m) from the same turbine as the insect sampling was conducted (or nearest turbine if sampling was not possible e.g. due to particular crops grown around turbines). The methods for acoustic sampling were followed as described in section 2.3.2 and bat identification followed the protocols given in section 2.3.3. The presence of a feeding buzz in each sonogram for *Pipistrellus pipistrellus* was also recorded as a proxy for feeding activity. The total number of passes and feeding buzzes per night at the sampled turbine was calculated.

### *4.3.4. Environmental indicators*

Weather data were collected at each site as described in section 2.3.4. Insects were only sampled for nights when; dusk temperatures ≥ 10°C, wind speed ≤ 8 m $s^{-1}$ and rain ≤ 2.5 mm $hr^{-1}$ (Hundt 2012).

### *4.3.5. Habitat analysis*

All habitat analyses were conducted in ArcGIS v.10 (ESRI 2011). Land Cover Map 2007 (LCM07; Morton et al. 2011) and OS MasterMap Topography Layer (EDINA Digimap Ordnance Survey Service 2015) were used to extract the following habitat variables within the surrounding landscape; bog, fen, marsh and swamp, farmland, freshwater, rough grassland, scrub and woodland (total, coniferous and deciduous/mixed). The OS MasterMap Topography Layer was used to derive data on roads, buildings, structures, streams and vegetated channels. Habitat categories were then classified (Table 3.2). Habitats were selected based their importance for foraging (e.g. Walsh & Harris 1996; Davidson-Watts et al. 2006).

The percentage of each habitat category within a radius around each turbine and control site was calculated at a range of spatial scales (50 m, 250 m, 500 m, 1000 m and 2500 m). These spatial scales were selected to assess both, local and landscape level importance. The local spatial scales (50m to 500 m) are important to assess the immediate habitat around traps. The landscape spatial scales (1000 m and 2500 m) were selected as they represent the core sustenance zone of *Pipistrellus pipistrellus* (Davidson-Watts & Jones 2006; Nicholls & Racey 2006b). In addition, the minimum distance of each treatment to each habitat category was calculated (Table 3.3).

Table 3.2. Landscape metrics used in models extracted from the [1]Land Cover Map 2007 (Morton et al. 2011) and [2]OS MasterMap (EDINA Digimap Ordnance Survey Service 2015).

| Habitat | Description |
|---|---|
| Built environment | Buildings, structures and roads[2] |
| Coniferous woodland | Coniferous woodland[1] |
| Deciduous/mixed woodland | Deciduous or mixed woodland[1] |
| Farmland | Arable and pastoral farmland[1] |
| Freshwater | Inland water and coastal water[1] (includes waterways and standing water[2]) |
| Riparian | Inland water, coastal water (including waterways and standing water), bog, fen, marsh and swamp[1] |
| Semi-natural habitat | Rough grassland and scrub[1] |
| Total Woodland | Coniferous, deciduous or mixed woodland[1] |

Table 3.3. Summary of the landscape metrics within 50 m and minimum distances, for paired turbine and control locations, across 18 wind farm sites.

| Landscape metrics | Turbines mean SD | Controls mean SD |
|---|---:|---:|
| %Farmland 50 m | 56 (51) | 66 (48) |
| %Riparian 50 m | 9 (26) | 1 (3) |
| %Freshwater 50 m | 1 (2) | 1 (2) |
| %Built environment 50 m | 4 (6) | 3 (9) |
| %Semi-natural 50 m | 0 (0) | 10 (30) |
| %Woodland 50 m | 0 (0) | 11 (31) |
| Minimum distance to coniferous woodland (m) | 2,401 (2,244) | 2,257 (2,338) |
| Minimum distance to woodland (m) | 802 (594) | 718 (686) |
| Minimum distance to deciduous/mixed woodland (m) | 1,045 (802) | 1,039 (885) |
| Minimum distance to farmland (m) | 465 (937) | 380 (694) |
| Minimum distance to riparian (m) | 123 (124) | 191 (189) |
| Minimum distance to built environment (m) | 126 (144) | 115 (128) |
| Minimum distance to semi-natural (m) | 988 (1,326) | 940 (1,101) |
| Minimum distance to freshwater (m) | 3,879 (5,837) | 2,675 (1,346) |

## 4.4. Statistical analysis

All analyses were conducted using R (v.3.1.2; R Core Team 2014) with the lme4 (v.1.1.7; Bates 2014) package. All models were assessed for model fit by checking that standardised residuals were normally distributed, predictor variables were not correlated and there was no evidence of over-dispersion or heteroscedasticity. Significance was assessed using log-likelihood ratio tests of the full model compared to the alternative model.

### *4.4.1. Selection of landscape metrics*

Due to high co-linearity between some of the landscape predictor variables, initial screening was conducted to identify the landscape metric (habitat and spatial scale combinations) that provided the best model fit following methodology outlined in Zuur et al. (2009). Separate models were tested, one for each landscape metric for both dependent variables (Diptera abundance and biomass). For Diptera abundance, generalized linear mixed models (GLMMs) with negative binomial error structure and log-link function were fitted, and for Diptera biomass, general linear mixed models (LMMs) were fitted with biomass log-transformed and a normal error distribution and logit link function. All models included site and night as random effects, to account for the spatio-temporal autocorrelation in the data. For nocturnal Diptera abundance models, an observational level random effect was also fitted to account for the over-dispersion in the data. The landscape metric with the highest $R^2$ value for each dependent variable was selected (Figure 3.1). If outliers caused a sudden increase in the $R^2$ value (>5%) from one spatial scale to the next, models were run with and without outliers to assess their effect. Pseudo and marginal $R^2$ were calculated following Nakagawa et al. (2013).

### *4.4.2. Insect attraction to turbines*

GLMMs with a negative binomial error structure and log-link function were used to assess whether nocturnal Diptera abundance was higher at the turbine compared to the control. Diptera abundance per night was specified as the dependent variable. Site, night and an observational level factor were fitted as random effects. The interaction between location as a fixed factor (two levels: control or turbine) and wind speed (m $s^{-1}$), and the habitat variable selected in the process above, were included as continuous predictor variables. In another model, assessing $\log_e$ (biomass), the same model structure was used, but with a normal error distribution and logit-link function. Only site and night were fitted as random effects.

### *4.4.3. Insects as predictors of foraging at turbines*

Feeding activity was assessed from the ratio of feeding buzzes to passes (buzz ratio; Vaughan, Jones & Harris 1996; Wickramasinghe et al. 2003). Four general linear models (GLM) were used to assess whether the abundance and biomass of nocturnal Diptera could predict *Pipistrellus pipistrellus* (1) bat activity, and (2) buzz ratio. Total activity and the $\log_e$ transformed ratio of total feeding buzzes to activity was calculated for each site and fitted as the dependent variable with the $\log_e$ of the number of survey nights fitted as an offset. In separate models the abundance and biomass of Diptera was fitted as a predictor variable. Total activity and buzz ratio was used across the site, since insect and bat surveys were conducted on the same night and model fit was improved, compared to when modelling activity and buzz ratio per night. Only those nights with data for both insect abundance and bat activity were included in the analyses.

## 4.5. Results

### *4.5.1. Insect abundance and diversity at wind farm sites*

In total, 7,666 insects were sampled across 21 orders. Most samples were Diptera (60%), and within Diptera; Cecidomyiidae (21%), Psychodidae (19%) and Chironomids (17%) were the most abundant families and Tipulidae had the highest biomass (86%) (Appendix: Table 3.4).

### *4.5.2. Landscape metrics and predicting insect abundance and biomass*

The percentage of built environment within 500 m of insect traps was the most important (highest $R^2$ value) landscape metric for nocturnal Diptera abundance; as built environment cover increased nocturnal Diptera abundance also increased. However, at other spatial scales this habitat was less important. Percentage woodland cover (total and mixed/deciduous) at the landscape scale (1000 m) and the minimum distance to riparian habitat were also important for nocturnal Diptera abundance (Figure 3.1a). For Diptera biomass, the percentage cover of farmland at the local scale (250 m) had the highest $R^2$ value. The percentage cover of freshwater became increasingly important at increasing scales from the local to the wider landscape (Figure 3.1b).

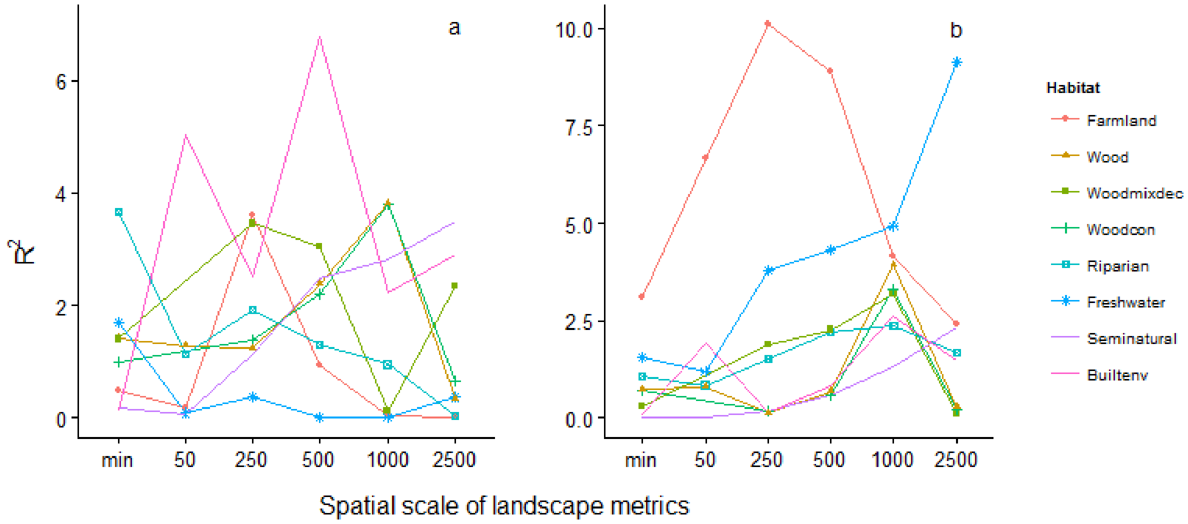

Figure 3.1. Marginal $R^2$ values from separate GLMMs for Diptera (a) abundance and (b) biomass. A range of landscape metrics were examined, min = the minimum distance to the landscape metric; 50-2500 refers to the radius around insect trap, for the percentage cover estimates. For landscape definitions see table 6.1; Builtenv = build environment, Wood = total woodland, Woodmixdec = mixed and deciduous woodland, Woodcon = coniferous woodland.

### *4.5.3. Insect abundance and biomass at turbines compared to controls*

Wind speed was a significant predictor of nocturnal Diptera abundance. Predicting from the model, as wind speed increased from 0 to 6 m $s^{-1}$, abundance declined by 21% at the controls (11 to 9 individuals) and 83% at the turbines (12 to 2 individuals; Figure 3.2). Nocturnal Diptera biomass was significantly higher at controls compared to turbines, although biologically the difference was small (Table 3.5, Figure 3.3). Nocturnal Diptera biomass was not predicted by wind speed or landscape metrics.

Table 3.5. Summary table showing the influence of landscape metrics, wind speed and sampling location on nocturnal Diptera abundance and biomass from GLMMs. Significant predictors are highlighted in bold. T = turbine, C = control, built = built environment, w.s. = wind speed, *c.f.* = compared to.

| Dependent variables | Predictor variables | Estimate ± SE | Log-likelihood | $\chi^2$ | *df* | AIC | *P-value* |
|---|---|---|---|---|---|---|---|
| Abundance | **%built 500 m** | **0.23 ± 0.11** | **-658** | **4.55** | **1** | **1333** | **0.033** |
| | **T *c.f.* C** | **-0.50 ± 0.16** | **-663** | **13.37** | **2** | **1340** | **0.001** |
| | **w. s.** | **-0.08 ± 0.22** | **-660** | **8.24** | **2** | **1335** | **0.016** |
| | **T *c.f.* C x w.s.** | **-0.50 ± 0.21** | **-659** | **5.64** | **1** | **1334** | **0.018** |
| Biomass (g) | %farm 250m | 0.43 ± 0.21 | -332 | 4.10 | 1 | 676 | 0.129 |
| | **T *c.f.* C** | **-0.63 ± 0.21** | **-344** | **9.06** | **2** | **681** | **0.011** |
| | w. s. | 0.10 ± 0.27 | -330 | 0.35 | 2 | 672 | 0.838 |
| | T *c.f.* C x w.s. | 0.06 ± 0.31 | -330 | 0.04 | 1 | 674 | 0.845 |

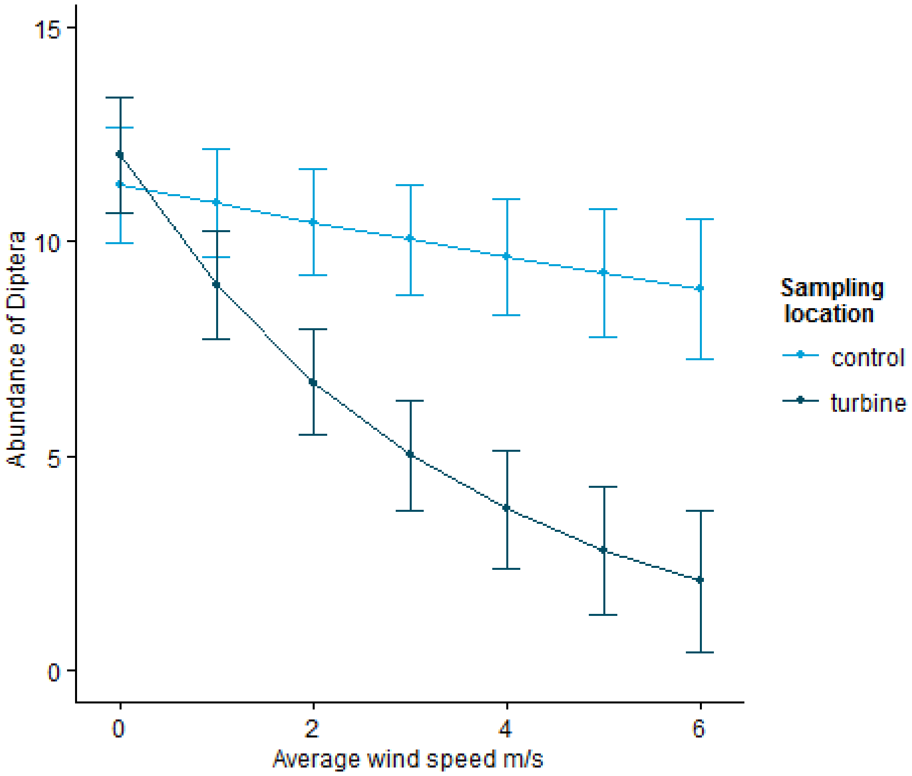

Figure 3.2. Predictions from a GLMM of nocturnal Diptera abundance ±SE at turbines and controls in relation to average wind speed (m $s^{-1}$). Models are based on samples from 18 sites surveyed in 2013.

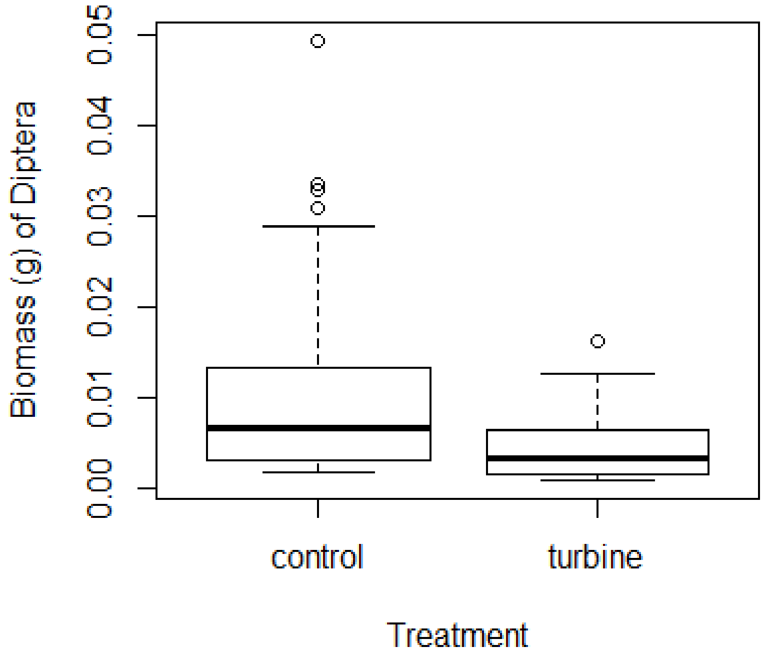

Figure 3.3. Box plot of the predictions from a LMM of the biomass of nocturnal Diptera at turbines compared to controls. Models are based on samples from 18 sites surveyed in 2013.

### *4.5.4. Insects as predictors of foraging at turbines*

In total, 2,657 *Pipistrellus pipistrellus* passes and 123 feeding buzzes were recorded over 97 nights. The mean number of passes per night was 30 SD 74, the mean number of feeding buzzes per night was 7 SD 14 and the mean feeding buzz ratio was 0.03 SD 0.03 across sites. Neither Diptera abundance nor biomass, were significant predictors of *Pipistrellus pipistrellus* feeding buzz ratio or activity (Table 3.6).

Table 3.6. Summary table of (G)LM outputs predicting the *Pipistrellus pipistrellus* feeding buzz ratio from nocturnal Diptera abundance and biomass in two separate models (site included as a fixed factor). LR = likelihood ratio statistics, F = F-statistic.

| Dependent variables | Predictor variable | Estimate ± SE | LR/F | df | *P* value |
|---|---|---|---|---|---|
| Bat activity | Insect abundance | 0.003 ± 0.22 | 0.0005 | 1 | 0.989 |
| Bat activity | Insect biomass | 0.33 ± 0.33 | 0.979 | 1 | 0.339 |
| Feeding buzz | Insect abundance | 0.003 ± 0.22 | 0.0002 | 1 | 0.990 |
| Feeding buzz | Insect biomass | 0.27 ± 0.66 | -0.63 | 1 | 0.690 |

## 4.6. Discussion

One of the main hypothesis proposed for why bats are attracted to turbines is that insects may accumulate at turbines due to their colour (Long, Flint & Lepper 2011) or the unique microclimates created around the nacelle (Cryan et al. 2014), which provide increased foraging opportunities for bats. In this study the abundance and biomass of nocturnal Diptera, the main prey taxa for *Pipistrellus pipistrellus,* was quantified for the first time at turbines and control locations. Biomass was higher at controls compared to at turbines. Furthermore, at wind speeds when turbines are operational (>2 m $s^{-1}$), nocturnal Diptera abundance was also lower at turbines compared to controls, and this difference increased as wind speed increased. This indicates that nocturnal Diptera are displaced away from turbines at and above operational wind speeds. These results contradict suggestions that bats are attracted to turbines to forage on insects that aggregate there (e.g. Horn, Arnett & Kinz 2008; Long, Flint & Lepper 2011), and supports other hypotheses for their attraction, such as investigating turbines as prominent structures in the landscape for potential roosting or mating locations. In the case of *Pipistrellus pipistrellus*, the latter is more likely, since they typically roost in buildings in the UK (Russ 2012).

The attraction to turbines for social reasons was similarly suggested by Jameson & Willis (2014). They monitored bat activity (North American species) at tall anthropogenic structures (telecommunication towers) and compared this to woodlots and open fields (other habitats), and found increased activity at towers compared to other habitats during summer and early autumn. They also monitored feeding buzzes and found a lower proportion at towers compared to other habitats. Peaks in the number of bat fatalities at wind farms (Arnett et al. 2008, Rydell et al. 2010b) support the theories of attraction linked to social rather than foraging behaviour, since their timings coincide with mating periods rather than the full extent of the bat active season.

Results from this study cannot reject the foraging behaviour hypothesis completely, since insects were not sampled at the nacelle where they are speculated to accumulate (Cryan et al. 2014). However, *Pipistrellus pipistrellus* are rarely detected at the nacelle (Chapter two) and hence fatalities are more likely within the lower rotor sweep area, close to ground level and their typical flight height (Russ 2012). Insect aggregations at the nacelle are therefore unlikely to be the reason for attraction for *Pipistrellus pipistrellus* and similar lower flying aerial hawkers, but it may be a more significant factor for other higher flying species.

Despite insect abundance and biomass being lower at turbines compared to controls, bats may obtain an increased foraging benefit around such large structures in an open landscape. Due to the high frequency echolocation calls used by bats and the inherent attenuation of their calls (Verboom & Huitema 1997), foraging around structures may increase the success rate of prey capture. *Pipistrellus pipistrellus* which are aerial hawkers normally avoid very open habitats, with their activity being proportional to the density of linear features (Swift, Racey & Avery 1985; Limpens &

Kapteyn 1991; Verboom & Huitema 1997). Bat attraction to turbines occurred regardless of the distance to linear features and site elevation (Chapter three). In an open landscape, typical of many wind farms, wind turbines may therefore provide a substitue linear feature and improved foraging abilities around large structures, particularly on evenings with low wind speeds, when insect dipserion is reduced.

In many studies, foraging activity of bats is correlated with insect abundance (e.g. Wickramasinghe et al. 2004; Fukui et al. 2006); whilst in other studies no correlations are found (e.g. Grindal & Brigham 1999). If bat attraction to turbines is due to a combination of factors (i.e. social and foraging), then the relationships between insect abundance and biomass with activity and foraging will be weakened. In the present study, neither abundance nor biomass significantly predicted bat foraging rate or activity at turbines, which may reflect lack of power to detect weak relationships.

Landscape metrics important for predicting Diptera abundance differed from those important for predicting Diptera biomass. The order Diptera constitutes thousands of species (Chinery 1993). Some Diptera families will have more or less influence on abundance and biomass (due to size and mass differences) which may explain why different landscape metrics are shown to be important for abundance compared to biomass. Differences in the importance of landscape metrics for Diptera abundance and biomass have also been found in other studies (e.g. Fuentes-Montemayor, Goulson & Park 2011).

The positive relationship between the percentage of built environment within 500 m of turbines and Diptera abundance was unexpected. However, there were only small percentage differences in the amount of built environment between sites at the local scale (range 0% to 5%). Given the high percentage of non-built environment cover, this difference is unlikely to be biologically important, compared to more extreme urbanized landscapes which generally have low insect generation (McKinney 2008). For other invertebrate taxa, increases in abundance and decreases in diversity have been found as the percentage of built environment increases at relatively low levels of urbanization (Shochat et al. 2004; Lintott et al. 2014). The positive relationship between Diptera abundance and the percentage of built environment may also relate to favourable microclimates created for Diptera as road density increases. Within 500 m of turbines the built environment is likely to represent roads, which slowly release stored heat generated in the day during the evening, possibly causing aggregation of insects (Hunter & Webster 1973; Kantzioura, Kosmopoulos & Zoras 2012). In addition, roads are often associated with hedgerows (not examined, due to a paucity of data), which provide a natural wind break for Diptera (Lewis 1965; Lewis 1969).

Among the landscape metrics considered, the percentage cover of farmland at the local scale (250 m) explained most variability in biomass. Farmland habitat is known to be important for *Pipistrellus pipistrellus,* possibly due to the high connectivity from hedgerows (Frey-Ehrenbold et al. 2013) typically found. Hedgerows provide insect generating habitat as well as shelter from wind (Lewis 1965; Lewis 1969). In addition to farmland, in this study, woodland habitats were also important and positively associated with Diptera biomass and abundance. Woodland provides similar wind breaks to hedgerows (Verboom & Huitema 1997), providing important foraging sites for *Pipistrellus* spp. (Russ & Montgomery 2002; Downs & Racey 2006).

## 4.7. Conclusions

Bats are most likely to concentrate and forage where insects are at high density (Griffin, Webster & Michael 1960). Despite this fewer insects were found at turbines compared to controls and insect abundance and biomass did not predict foraging or activity in *Pipistrellus pipistrellus.* This contrasts to the positive relationships between insects and activity found at other non-wind farm sites (Grindal & Brigham 1999; Fukui et al. 2006). This suggests that bats are not solely visiting wind turbines to forage on insects, and are attracted to turbines (Chapter three) due to social reasons or a combination of factors. Although a higher abundance or biomass of nocturnal Diptera at turbines compared to controls was not found at ground level, sampling insects at the nacelle would enable an assessment of whether insects are accumulating behind the nacelle in microclimates generated by moving blades (Cryan et al. 2014). For practical reasons this was not conducted in this study and is unlikely to be feasible.

## 4.8. Appendix I

Table 3.4. Summary of nocturnal insect abundance collected at turbines and control sites at 18 wind farm sites across Britain.

| Order | Family | Control | Turbine | Total |
|---|---|---|---|---|
| | | Abundance (% total catch) | Abundance (% total catch) | Abundance (% total catch) |
| Acarina | | 25 (0.5) | 43 (1.4) | 68 (0.9) |
| Coleoptera | | 125 (2.7) | 103 (3.4) | 228 (3.0) |
| Collemboa | | 346 (7.5) | 431 (14.3) | 777 (10.1) |
| Dermaptera | | 12 (0.3) | 50 (1.7) | 62 (0.8) |
| Diptera | Bibionidae (B) | 13 (0.3) | 1 (0.0) | 14 (0.2) |
| Diptera | Dolichopodiae (B) | 22 (0.5) | 8 (0.3) | 30 (0.4) |
| Diptera | Heleomyzidae (B) | 1 (0.0) | 2 (0.1) | 3 (0.0) |
| Diptera | Muscidae (B) | 9 (0.2) | 9 (0.3) | 18 (0.2) |
| Diptera | Others (B) | 487 (10.5) | 324 (10.7) | 811 (10.6) |
| Diptera | Syrphidae (B) | 17 (0.4) | 17 (0.6) | 34 (0.4) |
| Diptera | Unidentified (B) | 2 (0.0) | 1 (0.0) | 3 (0.0) |
| Diptera | Anisopodidae (N) | 60 (1.3) | 52 (1.7) | 112 (1.5) |
| Diptera | Cecidomyiidae (N) | 504 (10.9) | 283 (9.4) | 787 (10.3) |
| Diptera | Ceratopoginidae (N) | 320 (6.9) | 109 (3.6) | 429 (5.6) |
| Diptera | Chironomids (N) | 418 (9.0) | 220 (7.3) | 638 (8.3) |
| Diptera | Culicidae (N) | 18 (0.4) | 20 (0.7) | 38 (0.5) |
| Diptera | Dolichopodiae (N) | 1 (0.0) | 0 (0.0) | 1 (0.0) |
| Diptera | Mycetophilidae (N) | 284 (6.1) | 209 (6.9) | 493 (6.4) |
| Diptera | Psychodidae (N) | 551 (11.9) | 156 (5.2) | 707 (9.2) |
| Diptera | Sciaridae (N) | 4 (0.1) | 2 (0.1) | 6 (0.1) |
| Diptera | Tipulidae (N) | 316 (6.8) | 167 (5.5) | 483 (6.3) |
| Diptera | Trichoceridae (N) | 0 (0) | 1 (0.0) | 1 (0.0) |
| Diptera | Unidentified (N) | 0 (0) | 11 (0.4) | 11 (0.1) |
| Ephemeroptera | | 1 (0) | 0 (0.0) | 1 (0.0) |
| Hemiptera | | 64 (1.4) | 59 (2.0) | 123 (1.6) |
| Homoptera | | 243 (5.2) | 85 (2.8) | 328 (4.3) |
| Hymenoptera | | 319 (6.9) | 241 (8.0) | 560 (7.3) |
| Lepidoptera | | 331 (7.1) | 238 (7.9) | 569 (7.4) |
| Mecoptera | | 0 (0.) | 1 (0.0) | 1 (0.0) |
| Neuroptera | | 12 (0.3) | 9 (0.3) | 21 (0.3) |
| Odonata | | 3 (0.1) | 1 (0.0) | 4 (0.1) |
| Opiliones | | 73 (1.6) | 90 (3.0) | 163 (2.1) |
| Orthoptera | | 1 (0.0) | 2 (0.1) | 3 (0.0) |
| Plecoptera | | 12 (0.3) | 8 (0.3) | 20 (0.3) |
| Psocoptera | | 0 (0.0) | 1 (0.0) | 1 (0.0) |
| Ptchyopteridae | | 0 (0.0) | 1 (0.0) | 1 (0.0) |
| Thysanoptera | | 5 (0.1) | 6 (0.2) | 11 (0.1) |
| Trichoceridae | | 2 (0.0) | 0 (0.0) | 2 (0.0) |
| Trichoptera | | 41 (0.9) | 60 (2.0) | 101 (1.3) |
| Unidentified | | 0 (0.0) | 3 (0.1) | 3 (0.0) |
| Total | | 4,642 | 3,024 | 7,666 |

# Chapter five:

# 5. Can bat activity be used to refine curtailment at wind farms?

## 5.1. Summary

The expansion of global wind power is a relatively recent anthropogenic threat to wildlife. Conservation concern has been raised in light of the numbers of bat carcasses found at wind farms across the world. In Europe, the bats which appear most at risk are within the genera *Pipistrellus*, *Eptesicus* and *Nyctalus*. Currently, the most effective mitigation strategy is to curtail wind turbines at low wind speeds. However, this has yet to be widely adopted by industry. Using bat activity to curtail turbines may reduce the time turbines are not operational and therefore increase the uptake of this mitigation strategy by wind operators.

A national study of bat activity and fatalities at wind farms was conducted at 48 sites across Britain during the peak fatality period (July-October inclusive) from 2011 to 2013. Bat activity was monitored acoustically from three turbines at each site from paired ground and nacelle (centre of the rotor sweep area) height detectors. Concurrently, trained searched dogs were used to detect carcasses at the same turbines plus three additional turbines. The aim of the study was to assess whether acoustic monitoring of bat activity could be used to predict fatalities, thus refining curtailment of turbines for effective mitigation.

Bat activity, whether monitored at ground or at the nacelle, was not a significant predictor of fatality for all species combined and for *Pipistrellus pipistrellus. P. pygmaeus* activity monitored at ground level was a significant predictor of their fatality; however, estimates varied considerably across sites with similar activity levels. The probability of a *Pipistrellus pipistrellus* fatality was negatively related to the distance between the ground and the blade tip at the bottom of the rotor sweep area. These results suggest that general algorithms for curtailing

turbines based on activity in addition to wind speed are unlikely to be effective across all turbine designs.

## 5.2. Introduction

Wind energy is an important renewable energy source and is increasingly used currently in more than 80 countries across the world (Global Wind Energy Council 2014). In the last two decades 225,000 wind farms have become operational globally (Global Wind Energy Council 2014). Whilst wind power is important in reducing $CO_2$ emissions and limiting further anthropogenic changes in climate, it comes at a cost. The negative environmental impacts associated with wind energy include; noise and visual pollution (Julian, Jane & Davis 2007; Harrison 2011) and negative impacts on wildlife. Bats are one group to suffer impacts of wind farms, with fatalities resulting from collisions with turbines (Grodsky et al. 2011) and barotrauma (Baerwald et al. 2008) all raising conservation concerns (Johnson et al. 2003a; Kerns & Kerlinger 2004; Fiedler et al. 2007; Rydell et al. 2010b). In Europe, fatalities have been recorded for 27 bat species; however, the majority of fatalities occur in just five species; *Pipistrellus pipistrellus* (18%), *Nyctalus noctula* (14%), *P. nathusii* (13%), *P. pipistrellus/ P. pygmaeus* (not separated to species level) (11%) and *N. leisleri* (8%) (EUROBATS 2014). All bat species are legally protected (e.g. Annex IV of the EU Habitats and Species Directive; IUCN 2014), and therefore, understanding how to minimise casualty rates is key to complying with legislation that is aimed at conserving a taxon which has undergone severe declines, (Haysom et al. 2013) and yet provides important ecosystem services (Kunz et al. 2011).

High bat fatalities at some wind farms has led to extensive research on mitigation measures, including, the use of radar (Nicholls & Racey 2007), and devices which produce ultrasonic sounds to deter bats (Arnett et al. 2013b). However, these have yet to prove effective. Increasing the cut-in wind speed at which turbines become operational is the only mitigation measure that has resulted in significant reductions in bat fatalities at wind farms (Baerwald et al. 2009; Arnett et al. 2010; Brinkmann et al. 2011). The estimated cost of increasing cut-in speeds to wind operators is minimal at sites with high and consistent winds (<1% reduction of total annual power output) (Arnett et al. 2009). However, at more marginal wind farms, typical of those found in the UK and across Europe, the percentage loss is higher and not financially viable (Simon Pickering, Ecotricity, *pers*. *comm*. March 2016; RenewableUK 2010; Ottinger 2013). Therefore, more focused curtailment strategies have been investigated using pre-set bat activity thresholds to control turbine switch-on (Behr, Niermann & Korner-Nievergelt 2011; Lagrange 2013). These have reduced fatalities for *Nyctalus noctula* (Brinkmann et al. 2011) but their effectiveness for species in the *Pipistrellus* genus, and across wide ranging turbine types, is unknown, and has therefore been questioned (Voigt et al. 2015).

To assess whether bat fatalities can be predicted by activity and wind speed, a large systematic study was conducted at 48 wind farms across Britain. All bat species most at risk of fatality across Europe occur within Britain (Harris & Yalden 2008; Rydell et al. 2010b; EUROBATS 2014; IUCN 2014). In this study, specifically, the following questions were addressed:

1. Are individual species models better than ‘any species’ models for predicting fatalities?
2. Is measuring bat activity at ground level a better predictor of fatality than measuring bat activity at nacelle level?

3. Can activity accurately estimate fatalities and does this vary across turbine designs?

## 5.3. Methods

### *5.3.1. Acoustic monitoring and bat identification*

Bats were surveyed acoustically at 48 sites across Britain. The distribution of sites covered the full extent of Britain (Table 5.1). The mean number of wind turbines at each site was 13 SD 7, the mean tower height (centre of the rotor sweep area) was 60 SD 14 m (range 30 to 80) and the mean cut in wind speed that turbines became operational was 3.9 SD 0.9 m $s^{-1}$ (range 2.5 to 6). Bats were surveyed acoustically as described in sections 2.3.2. and 2.3.3.

Table 5.1 Summary of sites (n = 48) by habitat and country, that were surveyed between 5 July and 10 October 2013 to sample bat activity and insect abundance and biomass.

| Country | Farmland | Lowland moorland | Upland moorland | Totals |
|---|---|---|---|---|
| England | 23 | 0 | 0 | 23 |
| Scotland | 2 | 6 | 8 | 16 |
| Wales | 4 | 0 | 5 | 9 |
| Total | 28 | 6 | 13 | 48 |

### *5.3.2. Carcass searches*

Carcass searches were conducted at the same 48 sites monitored acoustically by four dog-handler teams in each year. Across the study a total of 12 dog-handler teams conducted searches, with the same team used for the full survey period at each site (mean duration 27 SD 6 days). Trained search dogs were used, which have higher detection rates (73%) than human searchers (20%) for carcass detection (Mathews et al. 2013). High detection rates reduce false negatives and bias, which is particularly important when the rate of fatality is low (Bernardino et al. 2014). On average six turbines (SD 1) were randomly selected at each site (including the three monitored acoustically) to be searched by the trained dog-handler team. A 50 m radius was searched every three days (SD 0.03) until eleven searches were conducted at each turbine. During a search, on detection of a carcass, the location was recorded and the condition of the bat (state of decay: fresh, early decay, late decay or desiccated) were used to estimate night of death. Wing samples were stored in 90% ethanol for subsequent genetic analysis. The remainder of the carcass was stored in neutral buffered formalin (in 2011 and 2012) or in 90% ethanol (2013) to permit subsequent *post-mortem* examination. Species identifications were made using a combination of morphometric characteristics, subject to carcass condition, and molecular methods (Hamilton et al. 2015).

### *5.3.3. Meteorological data*

Meteorological data was gathered as described in section 2.3.4. In addition, wind speeds measured from meteorological masts and from the nacelle of each turbine monitored acoustically were obtained for 28 of the sites. The proportion of time the turbine was operational was calculated from the proportion of hours in the night that the average wind speed at height was above the cut-in speed for the specific turbine model.

## 5.4. Statistical analyses

All statistics were performed using R (v.3.1.2; R Core Team 2014), with the package lme4 (v.1.1.7; Bates 2014) to fit GLMMs. In all models, collinearity between predictor variables was assessed and significance was determined using log-likelihood ratio tests of the full model compared to the alternative model.

### *5.4.1. Calculating wind speed at height*

To predict wind speed at height for sites where only ground wind speed was available, a GLMM was used with normal errors to model the relationship between wind speed at height and ground. The elevation of the detector at the nacelle was fitted as a fixed effect, and site as a random effect.

### *5.4.2. Predicting bat fatalities*

The exact night that a fatality occurred was uncertain due to mean 3 SD 0.04 day search intervals and <100% probability of carcass detection during a search. In addition, acoustic data were only recorded at half of the turbines searched. Three binomial mixed models was used to assess the probability of an 'any' species, a *Pipistrellus pipistrellus,* and a *P. pygmaeus* fatality occurring at any of the turbines within a site during each search interval (period of time between carcass searches, mean 3 days between searches and 3 potential nights where a fatality could have occurred). Binomial models were chosen in preference to Poisson models, to avoid problems with over-dispersion (a common problem when modelling rare events; Zuur et al. 2009), and due to the data having few intervals with multiple causalities (e.g. 'any species': 12/456 and *Pipistrellus pipistrellus:* 4/456 search intervals).

Based upon the scientific literature on the influences of bat fatalities (Arnett et al. 2010; Korner-Nievergelt et al. 2013; Cryan et al. 2014); the following potential predictor variables were used; turbine cut-in speeds, turbine tower height, bat activity and wind speed. Turbines varying in the cut-in wind speed that they become operational, which is intrinsic to the manufacture and model type (although they can be modified by wind operators). When turbines are not operational they are either ‘braked’ or ‘feathering’ (i.e. not operational but pitched into the wind and free to rotate). Turbine tower height was correlated with the distance between the ground and the blade tip (bottom of the rotor sweep area) and the rotor sweep area. Therefore, these three variables were assessed in separate models (including all other predictors) and the one with the highest coefficient was selected for the final model. An index of bat activity was calculated by summing the total number of passes across all turbines during the search interval and dividing by the total number of operational turbine hours. Only passes recorded when turbines were operational were included (assuming fatalities only occur when turbines are operational), which was determined when the average hourly wind speed at the nacelle was above the turbine cut-in speed. This gave an hourly pass rate and was calculated for ‘any’ species, *Pipistrellus pipistrellus* and *P. pygmaeus*. The average nightly wind speed during the search interval was calculated at ground level [since *Pipistrellus* spp. occurred more often at ground than the nacelle (Chapter two)]. Given that turbine cut-in speeds vary between turbine models and hence sites, an interaction between the turbine cut-in speed and the average nightly wind speed was also fitted as a predictor variable. In addition, site was fitted as a random effect to account for the spatial autocorrelation in the data. Any differences in the efficiency of the dog-handler team and the level of carcass predation at a site were assumed to be accounted for by the random effects. This model was run to assess the probability of: 'any species', *Pipistrellus pipistrellus,* and *P. pygmaeus* fatalities occurring. For each species the model was repeated for data using activity measured at ground and the nacelle.

## 5.5. Results

In total, 188,335 individual bat passes over 6,535 detector nights at 139 different turbines were recorded. Activity was recorded on 52% of nights, with *Barbastella, Eptesicus, Myotis, Nyctalus, Pipistrellus,* and *Rhinolophus* species detected. Bat activity was highest for *Pipistrellus pipistrellus* (74% of all activity), followed by *P. pygmaeus* (17%) and *Nyctalus noctula* (5%). In total, 2,973 carcass searches were conducted. Bat fatalities were found at 60% of sites and *Pipistrellus* spp. fatalities were found at 54% of sites (Table 5.2).

Table 5.2. Summary of the number of bat fatalities across 48 sites surveyed between mid-July and mid-October in 2011 to 2013.

| Species | Number of fatalities | Sites within species range | Sites with fatalities (%) |
|---|---:|---:|---:|
| All | 88[1] | 48 | 29 (60) |
| *N. noctula* | 9 | 37 | 5 (14) |
| *Pipistrellus* spp. | 78 | 48 | 26 (54) |
| *P. pipistrellus* | 38 | 48 | 17 (35) |
| *P. pygmaeus* | 30 | 46 | 16 (35) |
| *P. nathusii* | 1 | 41 | 1 (2) |
| *Plecotus auritus* | 1 | 37 | 1 (3) |

[1] one unknown spp.; seven additional bats were found incidentally, either outside the full search period and/or at other turbines.

### 5.5.1. The relationship between ground and height wind speed

The following equation predicts the average nightly wind speed at nacelle level (*WSn*): $WSn = 3.192 + 0.56WSg + 0.05h$

Average wind speed at ground (*WSg*) was a significant predictor ($\chi^2_1$ = 133.18, $P < 0.001$) of the wind speed at height (*h*), but the nacelle height was not a significant predictor ($\chi^2_1$ = 1.54, $P = 0.215$), although this was retained in the model.

### 5.5.2. Predicting bat fatalities

Individual species models had higher marginal and conditional $R^2$ values compared to 'any' species models. Therefore, more variability in the probability of a fatality was explained when species were modelled individually rather than all species together. The significance, sign and strengths of predictors of the probability of a fatality differed between 'any', *Pipistrellus pipistrellus* and *P. pygmaeus* models. For 'any' species and *Pipistrellus pipistrellus* models the number of turbines nights between searches was a significant positive predictor of fatality. In addition, for *Pipistrellus pipistrellus* the distance between the ground and blade tip was a significant negative predictor of fatality. As the distance between the ground blade tip increased from 20 m to 30 m the probability of fatality per turbine per month reduced by 47% (0.18 ± SE 0.63 to 0.10 ± SE 0.64; Figure 5.1). For *Pipistrellus pygmaeus* only the pass rate monitored at ground level was a positive significant predictor of fatality (Table 5.3). As the pass rate increased from 0 to 10 passes per hour the probability of fatality per turbine per month increased by 83% (0.24 ± SE 0.0.74 to 0.04 0.66). At zero passes per hour, the probability of *Pipistrellus pygmaeus* fatality per turbine per month was 0.04 ± SE 0.66; for *P. pipistrellus* the probability was 0.11± SE 0.63.

Table 5.3. Summary output of models predicting the probability of an ‘any’, *Pipistrellus pipistrellus,* and *P. pygmaeus* fatalities when monitoring activity at the ground (n = 48) and at the nacelle (n = 40). Site is included as a random effect. Significant predictors are highlighted in bold. If interaction terms were not significant there were dropped from the model and not displayed for ease of interpretation of main predictors.

| Species | Variables | Estimate | Log-likelihood | AIC | $\chi^2$ | df | P-value | Marginal $R^2$ (%) | Conditional $R^2$ (%) |
|---|---|---|---|---|---|---|---|---|---|
| Any | Avg. wind speed (m $s^{-1)}$ | -0.05 ± 0.14 | -128 | 268 | 0.05 | 1 | 0.817 | | |
| | Cut-in speed (m $s^{-1}$) | -0.02 ± 0.20 | -128 | 268 | 0.01 | 1 | 0.903 | | |
| | Distance ground to blade (m) | -0.39 ± 0.25 | -129 | 271 | 2.82 | 1 | 0.093 | 12 | 16 |
| | **No. turbine nights** | **0.51 ± 0.14** | **-134** | **281** | **12.88** | **1** | **<0.001** | | |
| | Pass rate at ground | 0.19 ± 0.14 | -129 | 269 | 1.65 | 1 | 0.199 | | |
| Any | Avg. wind speed (m $s^{-1}$) | -0.05 ± 0.15 | -112 | 236 | 0.05 | 1 | 0.815 | | |
| | Cut-in speed (m $s^{-1}$) | 0.03 ± 0.21 | -112 | 236 | 0.02 | 1 | 0.885 | | |
| | Distance ground to blade (m) | -0.39 ± 0.26 | -113 | 239 | 2.73 | 1 | 0.099 | 13 | 18 |
| | **No. turbine nights** | **0.51 ± 0.15** | **-118** | **247** | **11.55** | **1** | **<0.001** | | |
| | Pass rate at nacelle | 0.22 ± 0.16 | -113 | 238 | 2.24 | 1 | 0.135 | | |
| *P. pipistrellus* | Avg. wind speed (m $s^{-1}$) | -0.10 ± 0.31 | 140 | -64 | 0.10 | 1 | 0.755 | | |
| | Cut-in speed (m $s^{-1}$) | -0.25 ± 0.35 | 140 | -64 | 0.52 | 1 | 0.471 | | |
| | **Distance ground to blade (m)** | **-0.93 ± 0.52** | **-66** | **144** | **3.97** | **1** | **0.046** | 30 | 41 |
| | **No. turbine nights** | **0.81 ± 0.20** | **-73** | **158** | **18.87** | **1** | **<0.001** | | |
| | Pass rate at ground | 0.11 ± 0.18 | -64 | 140 | 0.35 | 1 | 0.553 | | |
| *P. pipistrellus* | Avg. wind speed (m $s^{-1}$) | -0.17 ± 0.37 | -57 | 127 | 0.21 | 1 | 0.646 | | |
| | Cut-in speed (m $s^{-1}$) | -0.14 ± 0.37 | -57 | 127 | 0.15 | 1 | 0.697 | | |
| | Distance ground to blade (m) | -0.92 ± 0.53 | -59 | 130 | 3.60 | 1 | 0.058 | 29 | 46 |
| | **No. turbine nights** | **0.87 ± 0.22** | **-67** | **145** | **18.46** | **1** | **<0.001** | | |
| | Pass rate at nacelle | 0.02 ± 0.17 | -57 | 127 | 0.01 | 1 | 0.905 | | |
| *P. pygmaeus* | Avg. wind speed (m $s^{-1}$) | -0.37 ± 0.34 | -65 | 142 | 1.40 | 1 | 0.236 | | |
| | Cut-in speed (m $s^{-1}$) | 0.31 ± 0.34 | -65 | 142 | 0.89 | 1 | 0.344 | | |
| | Distance ground to blade (m) | -0.19 ± 0.36 | -65 | 141 | 0.32 | 1 | 0.572 | 12 | 19 |
| | No. turbine nights | 0.20 ± 0.24 | -65 | 141 | 0.62 | 1 | 0.430 | | |
| | **Pass rate at ground** | **0.43 ± 0.17** | **-67** | **147** | **6.01** | **1** | **0.014** | | |
| *P. pygmaeus* | Avg. wind speed (m $s^{-1}$) | -0.36 ± 0.38 | -57 | 126 | 2.94 | 1 | 0.086 | | |
| | Cut-in speed (m $s^{-1}$) | 0.32 ± 0.22 | -56 | 124 | 0.63 | 1 | 0.426 | | |
| | Distance ground to blade (m) | -0.34 ± 0.43 | -56 | 124 | 0.72 | 1 | 0.395 | 10 | 26 |
| | No. turbine nights | 0.19 ± 0.26 | -56 | 124 | 0.50 | 1 | 0.482 | | |
| | Pass rate at nacelle | 0.32 ± 0.22 | -57 | 126 | 2.94 | 1 | 0.086 | | |

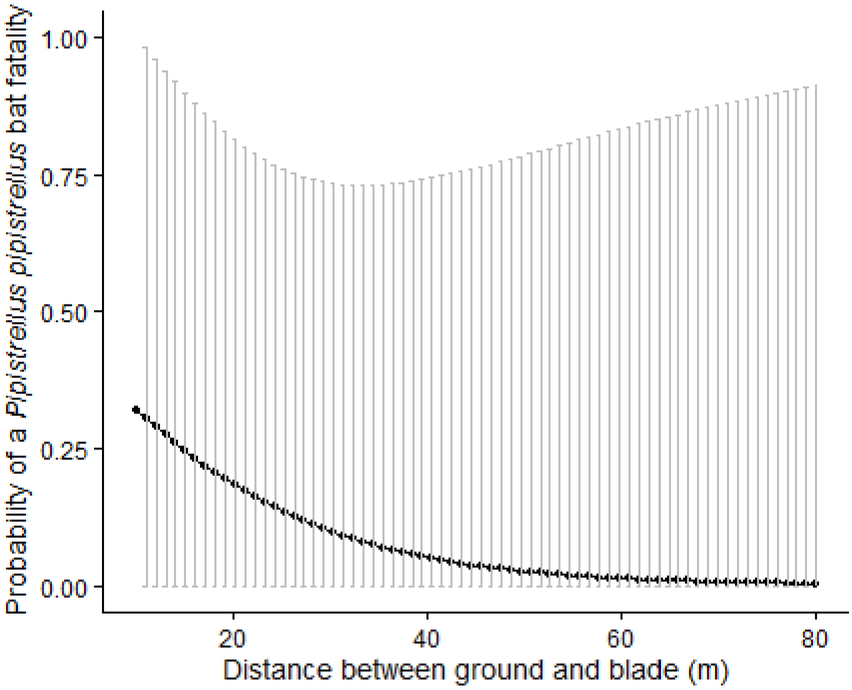

Figure 5.1. The relationship predicted from a GLMM model of the distance between the ground and blade tip (m) and the probability of a *Pipistrellus pipistrellus* fatality per turbine per month ± SE.

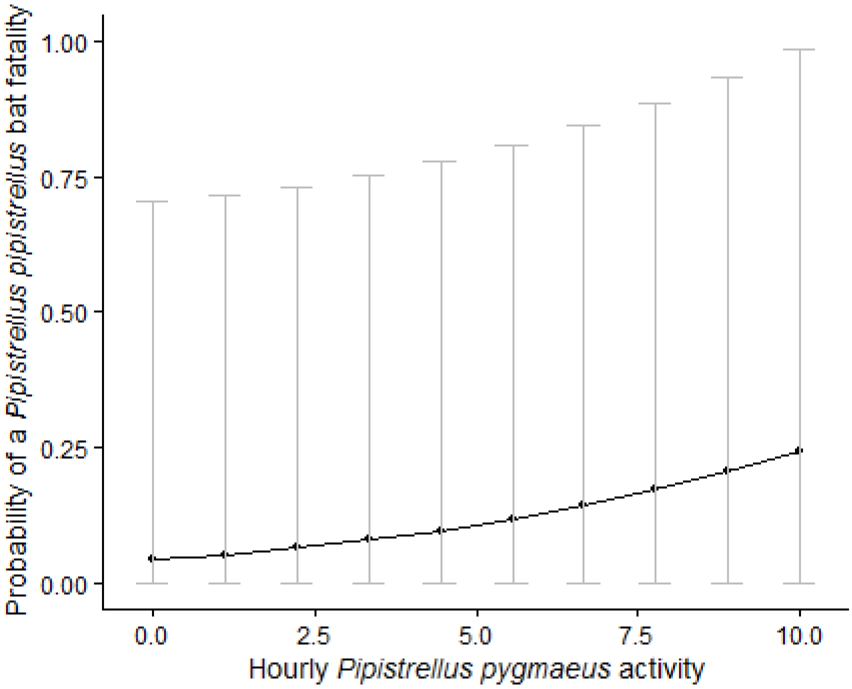

Figure 5.2. The relationship predicted from a GLMM model of the hourly *Pipistrellus pygmaeus* activity and the probability of a *P. pygmaeus* fatality per turbine per month ± SE.

In all models ('any', *Pipistrellus pipistrellus* or *P. pygmaeus*), the coefficient for bat activity was always positive, however, activity was not a significant predictor of the probability of a fatality for 'any' species or *Pipistrellus pipistrellus,* whether measured at ground or at the nacelle. The coefficient was highest for *Pipistrellus pygmaeus* monitored at ground level and lowest for *P. pipistrellus* monitored at the nacelle. When comparing ground and nacelle height models, for 'any' species models the coefficient was higher when monitored at the nacelle, whereas for individual species models (*Pipistrellus pipistrellus* and *P. pygmaeus*) the coefficient was higher when monitored at the ground.

## 5.6. Discussion

In this study, most activity and fatalities were from two species, *Pipistrellus pipistrellus* and *P. pygmaeus*. In a European context, this supports previous findings that these species are especially at risk of fatality (Rydell et al. 2010b; EUROBATS 2014). This is a novel study, using algorithms to predict fatalities for individual high risk species at wind farms, which has shown differences in predictors of fatality between two sympatric species (*Pipistrellus pipistrellus* and *P. pygmaeus*). Results suggest that predicting fatalities for all species combined may confound relationships.

### *5.6.1. Predicting fatalities for any species or individual species*

The relationship between predictors (activity, wind speed, distance between ground and blade tip and turbine cut-in speeds) and the probability of a fatality for 'any species', *Pipistrellus pipistrellus* and *P. pygmaeus,* were not consistent. The 'any' species models included nine *Nyctalus noctula* fatalities and one *Plecotus auritus* in addition to all *Pipistrellus* spp. fatalities. Bats from these genera differ in their size, foraging behaviour and flight height (Davidson-Watts & Jones 2006, Jones 1995, Nicholls & Racey 2006b, Russ 2012). This may have resulted in the lower variability explained in the 'any' species models compared to the individual species models.

The differences found between *Pipistrellus pipistrellus* and *P. pygmaeus* models perhaps reflect the behavioural differences found between the species (Davidson-Watts & Jones 2006; Nicholls & Racey 2006a, Nicholls & Racey 2006b) but also that *P. pygmaeus* occur more frequently at the nacelle compared to *P. pipistrellus* (Chapter two). This may explain the importance of the distance between the ground and the blade tip for *Pipistrellus pipistrellus* but not for *P. pygmaeus*. In addition, if *Pipistrellus pipistrellus* fly more often below the rotor sweep area (Chapter two), this would explain why activity was not a good predictor of fatality for *Pipistrellus pipistrellus* but was for *P. pygmaeus* models.

### *5.6.2. Fatalities and the height of acoustic monitoring*

Bat activity measured at either ground level or from within the centre of the rotor sweep area ('at height') was not a significant predictor of the probability of an 'any' species or *Pipistrellus pipistrellus* fatalities. This contrasts with previous findings of positive relationships between total activity (all species) at the nacelle and the total number of fatalities (Korner-Nievergelt et al. 2013). The difference between the present study and Korner-Nievergelt et al. (2013) may be due to variation in species composition. *Pipistrellus pipistrellus* constitute most activity and fatalities here, whereas *Nyctalus* and *Eptesicus* may have shared a higher representation of activity (species composition not documented) in the study by Korner-Nievergelt et al.

(2013). The wider ranging turbine types in this study (e.g. turbine diameter range 33 to 101 m) compared to the study by Korner-Nievergelt et al. (2013; e.g. Enercon turbines, diameter range 66 to 72 m) could also explain differences in relationships between activity and fatality between the two studies, especially since turbine characteristics (i.e. distance between ground and blade tip which correlates with tower height) are significant predictors of fatality (Barclay, Baerwald & Gruver 2007; Rydell et al. 2010b).

Korner-Nievergelt et al. (2013) considered activity measured at the nacelle to predict fatalities and a significant relationship was found. In the present study no significance was found between activity monitored at the nacelle for all species combined or *Pipistrellus pipistrellus* and *P. pygmaeus* considered individually. However, for all species combined, the coefficient for activity was higher when measured at the nacelle compared to ground, which was not the case for individual species models. This difference was presumably due to the inclusion of *Nyctalus noctula* fatalities, which had a higher rate of detection at the nacelle compared to other species (Chapter two). In this study there were insufficient *N. noctula* fatalities to consider them separately. Typical *Nyctalus noctula* flight height is considerably higher than *Pipistrellus* spp. (~30 m; Russ 2012) thus, activity measured at the nacelle is more likely a better predictor of fatality for this species, supported by findings elsewhere (e.g. Brinkmann et al. 2011).

### *5.6.3. Predicting fatalities across a range of turbine designs*

For all species combined and *Pipistrellus pipistrellus* considered separately, activity was not a significant predictor of fatality across a wide range of turbine designs. Therefore, using activity in addition to wind speed to curtail turbines and reduce fatalities is unlikely to be any more effective than using wind speed alone. Bat activity related curtailment may be an effective strategy to employ at a few high fatality sites (e.g. Lagrange 2013); however, it is unlikely to be effective across all sites where turbine characteristics vary considerably and influence fatality risk, as shown here and in other studies (Barclay, Baerwald & Gruver 2007; Rydell et al. 2010b). At tall turbine towers with blade tips far from the ground, *Pipistrellus pipistrellus* are more likely to fly outside the area of risk; as such, the relationship between activity and fatality will be weakened. Generic algorithms for all turbines, not incorporating these additional attributes will be ineffective. The importance of turbine characteristics in predicting fatalities has also been found for bats killed at wind turbines across several North American wind farms, where all species were combined and turbine characteristics were considered separately in simple algorithms (Barclay et al. 2007).

The interaction of wind speed with the wind speed turbines were operational was not a significant predictor of fatality. In other studies, the wind speed turbines were operational (e.g. Arnett et al. 2009, Arnett et al. 2010) and wind speed (e.g. Brinkmann, Schauer-Weisshahn & Bontadina 2006; Arnett et al. 2008; Rydell et al. 2010b) were significant predictors of fatalities. The lack of significance found in this study may be due to the temporal resolution of the search interval (mean 3 day period). Nightly wind speed was therefore also averaged across the same time period, during which, wind speed may have varied substantially.

Experimental tests of altering cut-in wind speed in a European context, will aid in determining whether curtailing turbines is an effective method for reducing fatalities for high risk species, such as *Pipistrellus pipistrellus* and *P. pygmaeus.* In this study, since site was included as a random effect, any variability between sites would have been accounted for, which may have reduced the strength of the fixed effects, which only varied by site (i.e. cut-in speed). Therefore, additional analyses to determine the predictors of the numbers of fatalities at a site level resolution were also conducted (Chapter six). Data was unavailable for whether a site used a breaking system or a feathering system at wind speeds below the cut-in speed. Whilst there is a difference between the two systems, when turbines are feathering the low rotation speed (*pers. observ.*) is unlikely to cause a significantly increased fatality risk compared to a braked system.

In current environmental impact assessments, fatality risk is determined by pre-construction species composition and activity levels at a site. Results from this study bring into question whether this is an effective method for all species and highlights the importance of incorporating turbine characteristics such as the distance between the ground and blade tip (correlated with tower height) into assessments of risk at wind farms. Results also bring into question whether activity should be used to determine fatality risk, since no relationships between activity at operational turbines and fatality for all species combined and *Pipistrellus pipistrellus* were found. Indeed, even for *Pipistrellus pygmaeus,* where activity was significant predictor of the probability of their fatality, estimates of the probability of fatality varied considerably across sites with similar *P. pygmaeus* activity levels. This may in part be due to false zeros, where carcasses were either not detected or removed prior to detection (Mathews et al. 2013) and the relationship between activity and fatality is weakened. Since trained search dogs were used to find carcasses, the probability of a non-detection was relatively low (Mathews et al. 2013). In addition, stronger relationships may be evident where activity is measured at the same turbine that fatalities occurr, rather than activity estimated across half of the turbines monitored for carcasses. Despite this, the habitats within wind farms used in this study were relatively uniform and activity for *Pipistrellus pipistrellus* and *P. pygmaeus* did not vary substantially between turbines within a site (Chapter two).

# Chapter six:

# 6. Estimating fatality numbers at wind farms in Britain for two high risk bat species

### 6.1. Summary

High bat fatalities at wind farm sites are of global conservation concern. Despite uncertainty in how pre-construction bat activity relates to fatality risk post-construction, extensive surveys are conducted at all proposed wind farms, at high cost. Few systematic studies have been conducted at a large number of wind farms to predict the number of fatalities at a site from turbine characteristics, habitat and bat activity levels. Understanding how to predict the number of fatalities at a site will make ecological assessments more robust. Two sympatric species, *Pipistrellus pipistrellus* and *P. pygmaeus,* were the focus of the study, since both have incurred high numbers of fatalities across Europe.

Acoustic monitoring was conducted, whilst concurrently performing carcass searches using trained search dogs to assess whether fatalities could be predicted at wind farms by activity levels. From 2011 to 2013, 48 wind farm sites were monitored across Britain during the peak bat fatality period. Estimates of the rate of carcass removal and the effectiveness of the search-dog handler team were made at each site to ensure accurate estimates of casualty rates were obtained.

Predicting from models in this study, between July and September, 5,085 *Pipistrellus pipistrellus* (95% CI 2,367 to 7,803) and 3,324 *P. pygmaeus* (95% CI 1,276 to 5,504) fatalities are estimated to occur annually across Britain. Importantly, mean nightly ground activity estimated across the month survey period was not a good predictor of the number of fatalities for *Pipistrellus pipistrellus* or *P. pygmaeus.* The manufacturer specified cut-in wind speed for turbines in this study was a significant predictor of the number of *Pipistrellus pipistrellus* fatalities. If manufacturer cut-in wind speeds was increased from 3.5 to 5 m $s^{-1}$ for all wind farms, the number of *P. pipistrellus* fatalities could be reduced by 76%. This study provides further

evidence for not measuring activity at high cost to predict the numbers of fatalities, but instead curtailing turbines at higher wind speeds, for an effective mitigation strategy. It is recommended that in Britain wind farms are only developed in locations where they are economically viable when using a high cut-in wind speed (e.g. 5 m $s^{-1}$), during the peak fatality period identified for bats.

## 6.2. Introduction

Wind energy is the fastest growing renewable energy source in over 80 countries worldwide (Global Wind Energy Council 2014). It is currently one of the most cost effective forms of renewable energy (RenewableUK 2010), and an important energy source which can help to reduce $CO_2$ emissions and limit further anthropogenic changes to the climate. However, wind energy has some environmental drawbacks, such as noise and visual pollution (Julian, Jane & Davis 2007; Harrison 2011) as well as direct and indirect effects on wildlife (Drewitt & Langston 2006; Thomsen et al. 2006). The negative impact on bats as a result of collisions with turbines (Grodsky et al. 2011) and barotrauma (Baerwald et al. 2008) is of high conservation concern (Johnson et al. 2003a; Kerns & Kerlinger 2004; Fiedler et al. 2007; Rydell et al. 2010b).

To effectively minimize bat fatalities at wind farms, it is important to assess and adapt the methods currently implemented to infer risk. Using European guidelines as an example; assessments at present involve; (1) pre-construction surveys based on desk studies of known roosts and surrounding site characteristics, (2) pre-construction surveys using acoustic monitoring, (3) post-construction surveys using acoustic monitoring, and (4) post-construction carcass surveys (Hundt 2012; Rodrigues et al. 2014). For two of these strategies (2 & 3) to be effective, an assumption is made that there is a strong link between activity and fatality risk. Whilst some studies have found a significant relationship between activity and fatality, these were considered for all species combined, and therefore relationships may not be consistent if species composition differs (Johnson et al. 2004; Korner-Nievergelt et al. 2013). Furthermore, building on previous research (Johnson et al. 2004; Korner-Nievergelt et al. 2013) by considering multiple turbine manufacturers and models, will enable results to be extended across a wide range of wind farm sites. This is particularly important given that tower height (Barclay, Baerwald & Gruver 2007; Rydell et al. 2010b) and blade length (Rydell et al. 2010b) may influence bat fatality risk. In addition, total fatalities, may mask species-specific differences due to inherent differences in behaviour (Jones & Rydell 1994; Davidson-Watts & Jones 2006) and unequal levels of risk between species (Rydell et al. 2010b; EUROBATS 2014).

In North America, ~75% of fatalities recorded have been tree-roosting migratory species, whereas in Europe, ~64% of fatalities have been edge and open aerial foraging species. Within mainland Europe, increased fatalities occur at taller turbines and within larger rotor sweep areas (Rydell et al. 2010b). High fatalities have also been linked to sites located on ridges and hilltops (Behr & Helversen 2005; Brinkmann, Schauer-Weisshahn & Bontadina 2006). Again, in these studies, total numbers of fatalities were assessed and species specific relationships were not investigated. Given that species behaviour (Davidson-Watts & Jones 2006) and preferences in foraging habitat differ (Walsh & Harris 1996; Lintott et al. 2015), relationships between species may not be consistent. Santos et al. (2013) considered species individually (*Hypsugo savii, Nyctalus leisleri, Pipistrellus kuhlii* and *P. pipistrellus*) using species distribution models, and found increased fatality risk was associated with wind farms located within 5 km of forested areas and within 600 m of steep slopes.

Understanding fatality risk associated with turbine characteristics and landscape metrics would aid wind operators and planning authorities to better site wind farms and assist ecological consultants in their assessment of sites for their potential risk to bats. A species driven approach for assessing risk may aid in more effective mitigation, especially where important populations are in the vicinity (resident or migratory) of a proposed wind farm. This study assessed whether bat fatalities on a species level, can be predicted by their activity, turbine characteristics and landscape metrics, by surveying 48 wind farm sites across Britain and addressing a genus of bats incurring the highest numbers of fatalities across Europe (EUROBATS 2014; IUCN 2014). In this study, the following questions were addressed:

1. How many *Pipistrellus pipistrellus* and *P. pygmaeus* fatalities from wind turbines are occurring in Britain?
2. Can *Pipistrellus pipistrellus* and *P. pygmaeus* fatalities be predicted by activity, habitat and turbine characteristics at the site level?
3. Based on these findings, what mitigation options can be implemented to minimize *Pipistrellus pipistrellus* and *P. pygmaeus* fatalities at wind farms?

## 6.3. Methods

### *6.3.1. Site selection*

Bats were surveyed acoustically at 48 sites across Britain (Figure 6.1). The distribution of sites covered the full extent of Britain (England, n = 23; Scotland, n = 16, Wales, n = 9). The mean number of wind turbines at each site was 13 SD 7, the mean tower height (centre of the rotor sweep area) was 60 SD 14 m (range 30 to 80) and the mean cut-in wind speed (m $s^{-1}$) that turbines became operational was 3.9 SD 0.9 m $s^{-1}$ (range 2.5 to 6).

Figure 6.1. Kernel distribution of (a) wind farms sites sampled across the study (n = 48), and (b) operational wind farms in Britain (n = 606), light areas show higher densities. For confidentiality, the participating wind farm locations are not revealed.

### *6.3.2. Acoustic monitoring and bat identification*

Bats were surveyed acoustically as described in section 2.3.2. and 2.3.3.

### *6.3.3. Carcass searches*

Carcass searches were conducted as described in section 5.3.2. In addition, at each site, efficiency trials (ET) were conducted to estimate the efficiency of the dog-handler team at finding bat carcasses (see Mathews et al. 2013). The trials enabled an estimate to be made of the probability that a carcass, if present, would be detected. Carcass removal trials were also conducted at each site to estimate the rate that carcasses decayed, or were removed by predators. For ET, a mean of 6 bats SD 0.3 (predominately *Pipistrellus* spp.) were randomly placed by an independent observer in a 100 m x 100 m area (not part of the study area but of similar habitat). The dog-handler team attempted to locate the carcasses, without knowledge of the number or their location. Carcasses were then left in position for the carcass removal trails: on each site visit, the bat carcasses were checked for their presence. The proportion of bats removed per 24 hours was calculated by dividing the number removed after the first visit by the number of intervening days.

Site specific, carcass removal and efficiency trial results were used to adjust detected carcass estimates per site using the following equation:

$$Estimate = N \,/\, (ET \times PR)$$

*N = number of bats found*

*ET = proportion of bats found in efficiency trials*

*PR = proportion of bats remaining in predator removal trials per day*

Adjusted carcass estimates were compared with unadjusted counts for each site (Figure 6.2).

### *6.3.4.* Habitat analysis

Habitat analyses were conducted in ArcGIS v.10 (ESRI 2011). The Land Cover Map 2007 (LCM07, Morton et al. 2011) and OS MasterMap Topography Layer were used to calculate habitat variables within the surrounding landscape. The minimum distance between surveyed turbines and each habitat category (Table 6.1) was extracted and mean distances were calculated for each habitat category and site.

The percentage cover of the habitat within two spatial scales (1000 m and 2500 m radius around each surveyed turbine) was extracted and the mean across surveyed turbines within a site was calculated (Table 6.1). These spatial scales were selected as they represent nightly foraging distances of the focal species in this study (Davidson-Watts & Jones 2006).

Table 6.1. Landscape metrics used in models extracted from the $^1$Land Cover Map 2007 (Morton et al. 2011) and from $^2$OS MasterMap (EDINA Digimap Ordnance Survey Service 2015).

| Habitat | Description |
|---|---|
| Built environment | Buildings, structures and roads$^2$ |
| Coniferous woodland | Coniferous woodland$^1$ |
| Deciduous/mixed woodland | Deciduous or mixed woodland$^1$ |
| Farmland | Arable and pastoral farmland$^1$ |
| Freshwater | Inland and coastal water$^1$ (includes waterways and standing water$^2$) |
| Riparian | Inland water, coastal water (including waterways and standing water), bog, fen, marsh and swamp$^1$ |
| Semi-natural habitat | Rough grassland and scrub$^1$ |
| Total woodland | Coniferous, deciduous or mixed woodland$^1$ |

## 6.4. Statistical analyses

Statistics were performed using R (v.3.1.2; R Core Team 2014) with the gamlss package (v.4.3.6.; Rigby 2005). All models were assessed for model fit by checking that standardised residuals were normally distributed, predictor variables were not correlated and there was no evidence of over-dispersion or heteroscedasticity. Significance was assessed using log-likelihood ratio tests of the full model compared to the alternative model.

### 6.4.1. Selection of landscape metrics

There was high collinearity between landscape metrics (i.e. between the proportions of different landscape metrics or the same landscape metric at different spatial scales); preliminary analyses to select the landscape metric to be included in the final model was conducted following the methodology outlined by Zuur et al. (2009). One model was fitted for each landscape metric (habitat and spatial scale, excluding other predictor variables) and dependent variable (number of *Pipistrellus pipistrellus* and *P. pygmaeus* fatalities at each site). The model used is described below (section 6.4.2). For the final model, the landscape metric with the lowest AIC value for each dependent variable was selected.

### 6.4.2. Predicting bat fatalities

Two ZIP linear mixed models (with log-link function) were used to predict the number of *Pipistrellus pipistrellus* and *P. pygmaeus* fatalities. The optimal model structure [zero inflated Poisson (ZIP) linear model, Poisson or negative binomial linear model (GLM)] was chosen using log-likelihood ratio tests and assessment of residuals. Normal qq-plots from GLMs exhibited zero inflation; hence, ZIP models were chosen as defined as:

$$\Pr(y) = p + (1-p)\exp(-mu) \quad if\ y = 0$$

$$\Pr(y) = (1-p)Pois(mu) \qquad if\ y = 1,2,\ldots n$$

Analyses for *Pipistrellus pipistrellus* and *P. pygmaeus* included all sites, since both species, although different in population density, are widespread across Britain (detected at 98% and 92% of sites, respectively). Ground level activity (averaged across all turbines and the month survey period) was used in both models, since coefficients for activity measured at ground level were higher than coefficients for activity measured at the nacelle (centre of the rotor sweep area) for predicting the probability of a *Pipistrellus pipistrellus* and *P. pygmaeus* fatality (Chapter five). Fatality estimates were adjusted as described above (see section 6.3.3.) for searcher efficiency and predator removal. The natural log of the potential number of turbine nights between the first and last search was fitted as an offset (total number of nights between first and last search x number of turbines searched). Fatality estimates were rounded to the nearest integer to enable a Poisson error structure to be fitted.

The following predictor variables were assessed as fixed effects for the Poisson error structure (mu): tower height of the turbine (m), average nightly ground activity (averaged across turbines monitored acoustically) and turbine cut-in speed (wind speed at wind turbines become operational based on the turbine model). There was high collinearity between tower height and rotor-sweep area and between tower height and the distance between the ground and the blade tips; thus, to minimize the number of predictor variables, tower height was fitted in models. Since turbines in the study were operational at wind speeds when bats were active (Chapter three) and potentially fatalities could occur at turbines of any height, only ground activity and the product of ET (proportion of bats found in efficiency trails) x PR (proportion of bats remaining in predator removal trials per day) were assessed inclusion in predicting the zero-inflation probability (p).

If convergence was not achieved the predictor variables for *p* were first fitted and the predictor with the lowest scaled coefficient was removed. The predictor variables for the Poisson error structure were then added, and the model was re-run. This was repeated for predictors for the Poisson error structure until model convergence was achieved. With ZIP models, interpretation of coefficients when included in both predicting *mu* and *p* is not possible. Model interpretation for ground activity was therefore made via simulations to generate E(y), the mean number of fatalities, across the mean of all other variables, with the exception of the ET x PR predictor which was predicted at 1 and the log of the offset (number of turbine nights) which was predicted at 30 (to predict across a month period at one turbine). Standard errors of all predictions were generated by simulations, similar to bootstrapping (Efron and Tibshirani 1986).

## 6.5. Results

### *6.5.1. Fatality summary*

Over 244 days, 2,973 carcass searches were conducted at 294 turbines at 48 sites. The mean proportion of bats detected in the efficiency trials was 0.83 SD 0.15 and the mean proportion of bats remaining per 24 hours was 0.89 SD 0.13. There was a strong correlation between the number of carcasses found and the number of fatalities estimated after adjusting for searcher efficiency and predator removal rates, for both *Pipistrellus pipistrellus* and *P. pygmaeus* (Figure 6.2). A total of 88 bat carcasses were found during carcass searches; 1 *Plecotus auritus*, 78 from the *Pipistrellus* genus (38 *P. pipistrellus,* 30 *P. pygmaeus,* 1 *P. nathusii,* and 10 *Pipistrellus* spp. not known), 9 *Nyctalus noctula*, and 1 unknown spp. (7 additional carcasses were found incidentally outside the full search period and/or at other turbines). Bat fatalities were found at 60% of sites, *Pipistrellus pipistrellus, P. pygmaeus* and *Nyctalus noctula,* occurred at 35%, 35%, and 14% of sites, within their distribution range (IUCN 2014), respectively. The 48 sites can be divided into three categories of fatality risk, namely, low, medium and high, using tertiles of the dataset. After adjusting for searcher efficiency and carcass removal at each site and standardising fatalities per turbine per month, low fatality sites would equate to zero fatalities $t^{-1}$ $month^{-1}$ and high fatality sites would equate to >0.5 fatalities $t^{-1}$ $month^{-1}$ (Table 6.2).

Table 6.2. Wind farm sites (n = 48) divided into tertiles of fatality per turbine per month, after adjusting for search efficiency and carcass removal at each site for all species combined, *Pipistrellus pipistrellus* and *P. pygmaeus*.

| Species | Low | Medium | High |
|---|---|---|---|
| All | 0 | >0 to ≤ 0.5 | >0.5 |
| *P. pipistrellus* | 0 | >0 to ≤ 0.18 | >0.18 |
| *P. pygmaeus* | 0 | >0 to ≤ 0.19 | >0.19 |

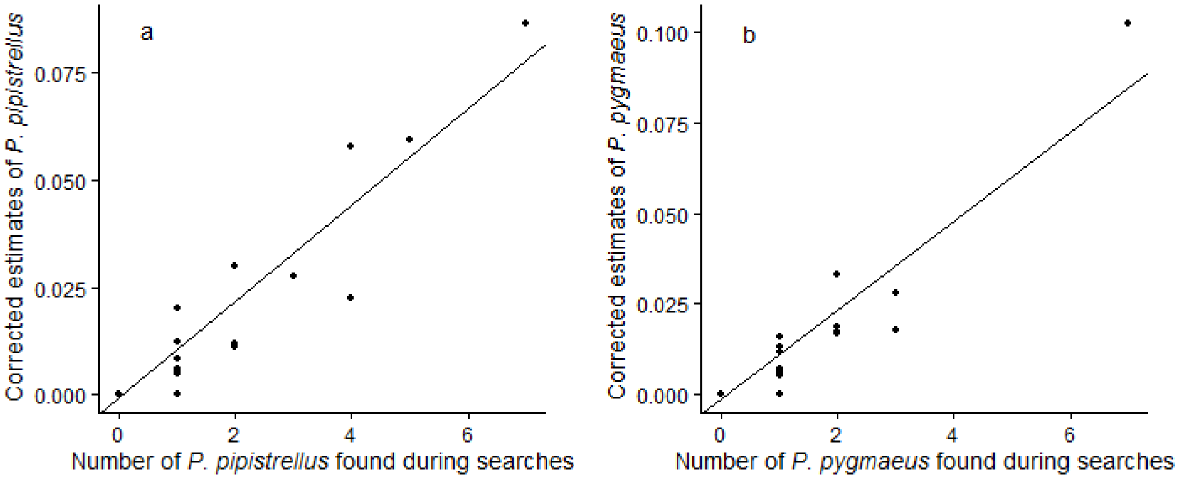

Figure 6.2**.** Correlation between the number of (a) *Pipistrellus pipistrellus* and (b) *P. pygmaeus* carcasses found and the estimated number of corresponding fatalities after adjusting for searcher efficiency, predator removal and search effort. Corrected estimates are given for number of fatalities per turbine per night.

### *6.5.2. Landscape metrics predicting fatalities*

The most important landscape metric in predicting the number of fatalities for *Pipistrellus pipistrellus,* was the percentage cover of total woodland within 1000 m of surveyed turbines (increased fatalities as the percentage of woodland decreased), and for *P. pygmaeus,* it was the minimum distance to riparian habitat (increased fatalities the further turbines were from riparian habitat). For both species, among landscape metrics, the percentage cover of total woodland in the immediate surroundings of the wind farm explained the most variability in predicting the number of fatalities (Figure 6.3). However, in the final models, habitat was not a significant predictor of the number of fatalities for *Pipistrellus pipistrellus* or *P. pygmaeus* (Table 6.3).

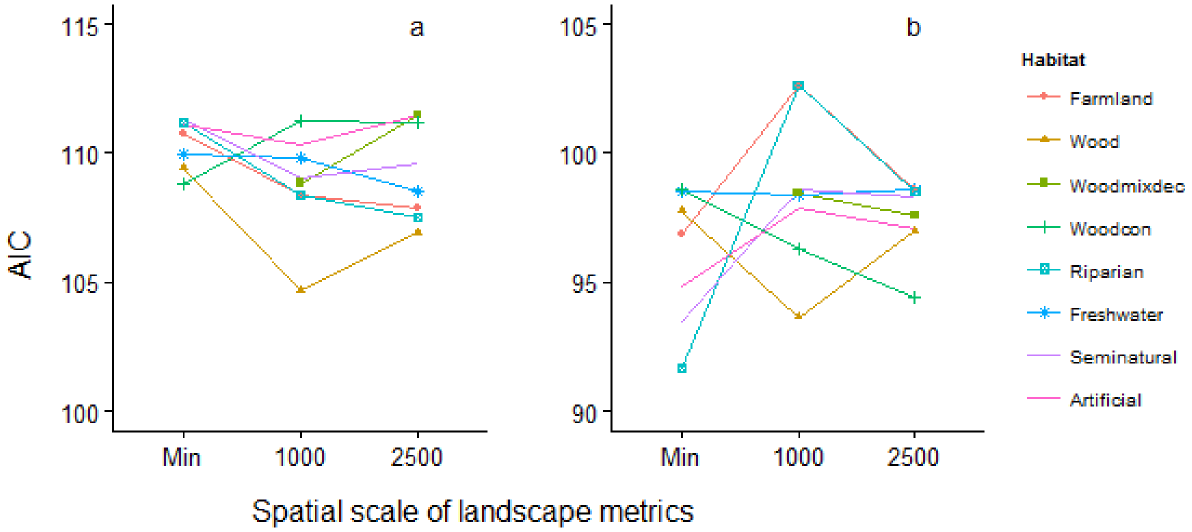

Figure 6.3. AIC values of landscape metrics in zero inflated GLMs predicting fatalities at each site for (a) *Pipistrellus pipistrellus* and (b) *P. pygmaeus*. Min = minimum distance to landscape metric, 1000 and 2500 = mean percentage cover of each habitat within the given radius of the surveyed turbines. The variable selected for each species had the lowest AIC, which explained the most variability in the model. For landscape definitions see table 6.1; Builtenv = build environment, Wood = total woodland, Woodmixdec = mixed and deciduous woodland, Woodcon = coniferous woodland.

### *6.5.3. Predicting bat fatalities*

For *Pipistrellus pipistrellus,* turbine cut-in speed was a significant predictor of the number of *P. pipistrellus* fatalities (Table 6.3). As the cut-in wind speed increased from 3.5 to 5 m s$^{-1}$, the mean number of *Pipistrellus pipistrellus* fatalities t$^{-1}$ month$^{-1}$ declined by 76% (0.23 fatalities per turbine per month to 0.06). The number of fatalities reduced most notably when turbine cut in speeds increased from 2 to 4 m s$^{-1}$ (1.05 fatalities per turbine per month to 0.15), beyond which fatality numbers tend towards zero (Figure 6.4). Mean *Pipistrellus pipistrellus* nightly ground activity across the full search period was not significant at predicting the number of *P. pipistrellus* fatalities (Figure 6.5); however, it was significant at predicting whether fatalities occurred at a site. At sites with fatalities, the mean number of passes per turbine and night (across all turbines and the full month search period) was 59% higher compared to sites with no fatalities (50 ± SE 14 passes per night, *c.f.* 21 ± SE 6). Predicting from the model, when zero activity is found at ground, the probability of a *Pipistrellus pipistrellus* fatality per turbine per night was 0.034 (95% CI 0 to 0.09).

Table 6.3. Summary of model outputs from a zero inflated Poisson linear model assessing predictors of fatalities for *Pipistrellus pipistrellus* (n = 46 sites, excluding two outliers) and *P. pygmaeus* (n = 46, excluding two outliers). Model assessment was made by dropping each predictor in turn from the full model. Mu = Poisson error structure, p = zero inflation probability. Convergence was not achieved with ET (proportion of bats found in efficiency trails) x PR (proportion of bats remaining in predator removal trials per day) included as a predictor for p in the *Pipistrellus pipistrellus* model. If deviance of the reduced model was smaller $\chi^2$ or p-values could not be calculated.

| Dependent variables | Predictor variables | Error | Coefficient | AIC | $\chi^2$ | df | *P* value | $R^2$ (%) |
|---|---|---|---|---|---|---|---|---|
| *P. pipistrellus* | Full model | | | 109 | | | | 46 |
| | **Cut-in speed (m s$^{-1}$)** | ***mu*** | **-0.91** | **104** | **114** | **1** | **<0.001** | **6** |
| | Ground activity | *mu* | - | 108 | 96 | 1 | 0.254 | 2 |
| | %woodland 1000m | *mu* | -0.30 | 110 | 98 | 1 | 0.085 | 4 |
| | Tower height (m) | *mu* | 0.27 | 107 | 86 | 1 | 0.357 | 1 |
| | **Ground activity** | ***p*** | **-** | **116** | **104** | **1** | **0.002** | **12** |
| *P. pygmaeus* | Full model | | | 104 | | | | 9 |
| | Cut-in speed (m s$^{-1}$) | *mu* | -0.50 | 102 | 88 | 1 | 0.564 | <1 |
| | Ground activity | *mu* | - | 101 | - | 1 | - | <1 |
| | Min. dist. riparian | *mu* | 0.0004 | 102 | 88 | | 0.958 | <1 |
| | Tower height (m) | *mu* | -0.05 | 102 | 88 | 1 | 0.802 | <1 |
| | Ground activity | *p* | - | 100 | 88 | 1 | 0.518 | <1 |
| | Observer efficiency | *p* | -0.33 | 104 | 89 | 1 | 0.388 | 1 |

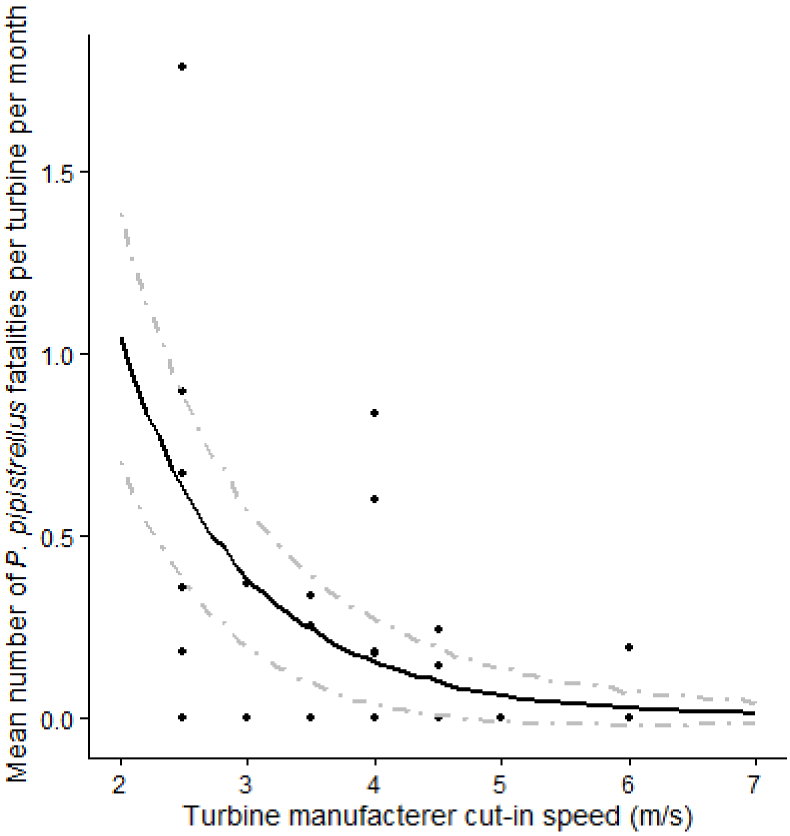

Figure 6.4. Predictions from a zero-inflated Poisson linear model (n = 46, excluding two outliers) of the mean number of *Pipistrellus pipistrellus* $t^{-1}$ month $^{-1}$ ± 95% CI in relation to the turbine manufacturer cut-in wind speed (m $s^{-1}$).

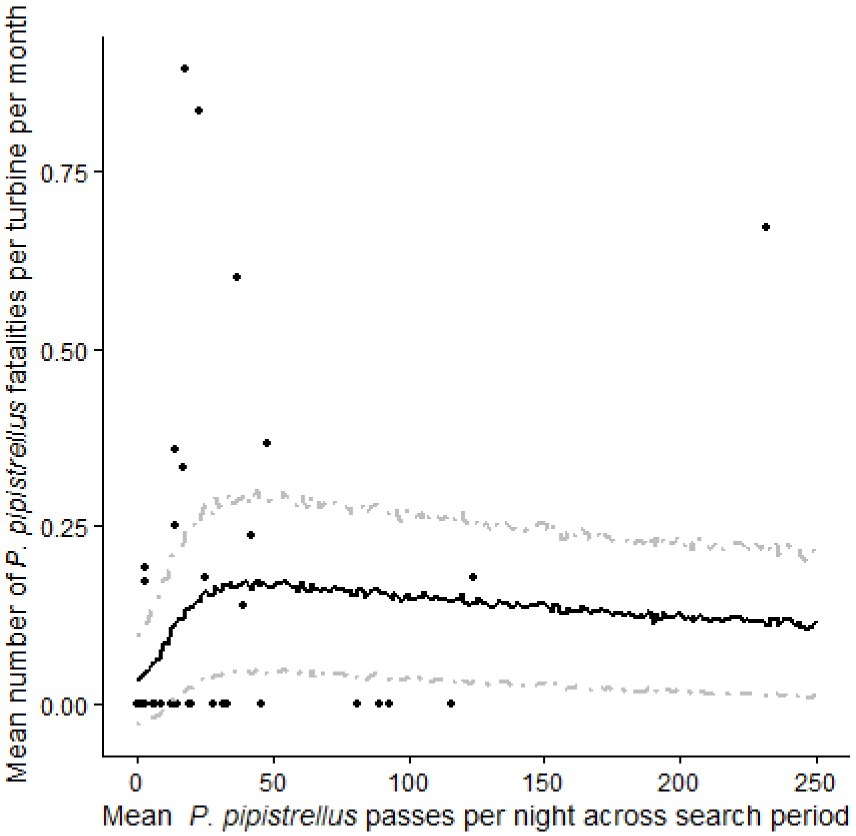

Figure 6.5. Predictions from a zero-inflated Poisson linear model (n = 46, excluding two outliers) of the mean number of *Pipistrellus pipistrellus* fatalities $t^{-1}$ month $^{-1}$ ± 95% CI in relation to the mean nightly activity across the full search period at ground level.

There were no significant predictors for the number of *Pipistrellus pygmaeus* fatalities when one outlier site and one influential site were excluded from the model. With the inclusion of the influential site, mean ground activity (estimated across turbines and the full month search period) was a significant predictor of the number of fatalities (Figure 6.6). At zero activity the probability of a fatality was 0.046 (95% CI 0 to 0.12).

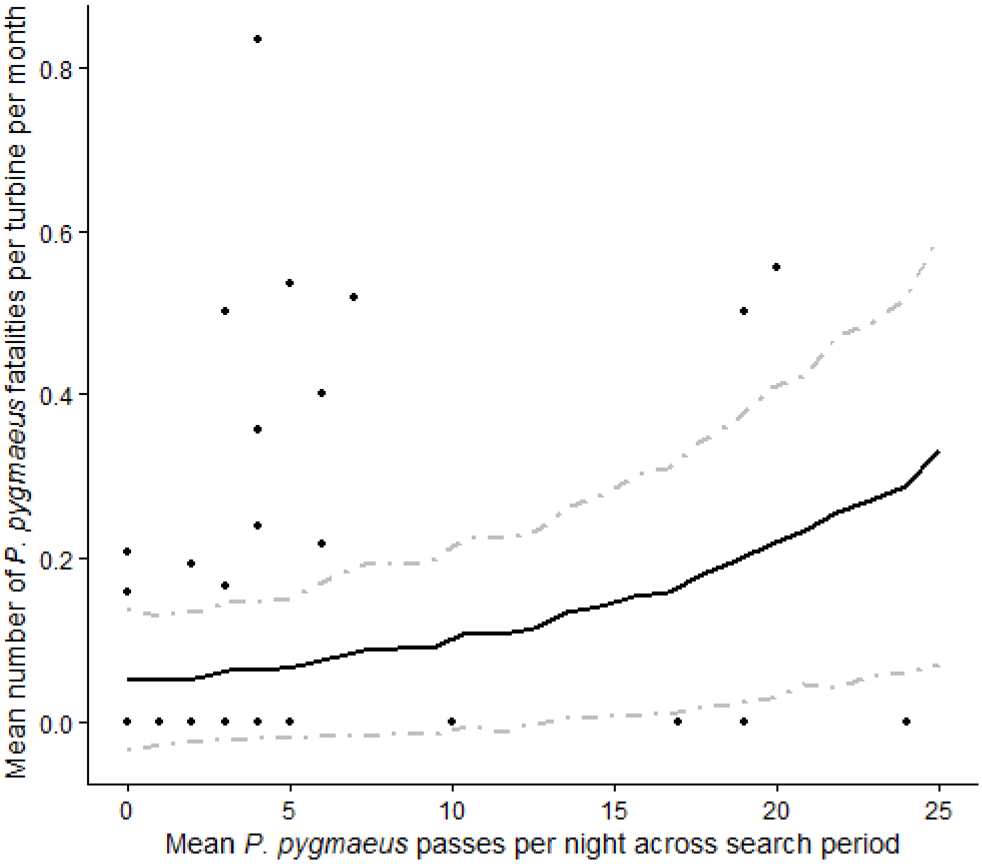

Figure 6.6. Predictions from a zero-inflated Poisson linear model (n = 46, excluding one outlier and one influential site) of the mean number of *Pipistrellus pygmaeus* fatalities t$^{-1}$ month$^{-1}$ ± 95% CI in relation to the mean nightly activity across the full search period at ground level.

### *6.5.4. Estimating bat fatalities from large wind turbines in Britain*

Predicting from the model, the mean number of *Pipistrellus pipistrellus* and *P. pygmaeus* bats killed at turbines was 0.15 95% CI 0.04 to 0.27 t$^{-1}$ month$^{-1}$ and 0.20 95% CI 0.06 to 0.34 t$^{-1}$ month$^{-1}$ respectively. In November 2015, 5,136 large wind turbines were operational across 909 wind farms in the UK (RenewableUK 2015). Extrapolating from the model, the number of *Pipistrellus pipistrellus* and *P. pygmaeus* fatalities month$^{-1}$ was estimated at 791 (95% CI 695 to 886) and 1,027 (95% CI 906 to 1,148), respectively. Across the three month survey period (when most bat fatalities occur), this equates to 2,373 *Pipistrellus pipistrellus* fatalities (95% CI 513 to 4,233) and 3,082 *P. pygmaeus* fatalities (95% CI 1,270 to 4,894). In

Britain, the population of *Pipistrellus pipistrellus* is estimated at 1,280,000 and the population of *P. pygmaeus* at 720,000 (Battersby et al. 2005; Russ 1999). Using these population figures, the proportion of the population killed annually, i.e. during the main fatality period is estimated to be 0.19% for *Pipistrellus pipistrellus* and 0.43% for *P. pygmaeus*.

## 6.6. Discussion

This study is novel in modelling the numbers of bats killed by wind farms for individual high risk species based on a nationwide scale. By modelling species individually, difference between species fatality risk were found. In particular, the manufacturer turbine cut-in wind speed was the most important predictor of the number of *Pipistrellus pipistrellus* fatalities, with more fatalities occurring at turbines with lower cut-in wind speeds. However, for a very similar species, *P. pygmaeus,* the turbine cut-in speed was not a significant predictor of the number of fatalities. For both species mean activity measured at ground level was not a significant predictor of the number of fatalities. These results have important implications for current environmental impact assessments for bats, where currently, turbine characteristics are not incorporated into assessments, and activity is the main measure used to assess risk (Rodrigues et al. 2014). These results provide evidence to support incorporating manufacturer turbine cut-in speeds into wind farm site planning and bring into question whether activity should be used to assess risk for *Pipistrellus pipistrellus* and *P. pygmaeus,* two species incurring the highest numbers of fatalities across Europe (EUROBATS 2014).

### *6.6.1. Predicting bat fatalities*

In this study, turbine manufacturer cut-in wind speeds ranged from 2.5 to 6.0 m $s^{-1}$ and was an important determinant of the number of *Pipistrellus pipistrellus* fatalities*.* Results here support the findings of Arnett et al. (2010), who found that increasing the turbine manufacturer cut-in speeds from 3.5 to 5 m $s^{-1}$ and 6.5 m $s^{-1}$ reduced *Lasiurus* spp. and *Lasionycteris* spp. fatalities by between 44% and 93% at North American wind farms. In the present study, when modelling the increase in turbine cut-in speed from 3.5 to 5 m $s^{-1}$ (equivalent to the lower threshold in Arnett et al. 2010), significant reductions were found in the numbers of *Pipistrellus pipistrellus* fatalities (76%). The significant reductions in the number of fatalities at sites with turbines with high cut-in wind speeds provides additional evidence of the effectiveness of curtailing turbines as a mitigation strategy for a wider range of species. *Pipistrellus* spp., are comparatively low flying generalists whereas in North America, fatality reductions were observed for high flying tree-roosting specialists.

Turbine characteristics, in addition to the cut-in speed, were also assessed as predictors of fatality. Tower height was not a significant predictor of fatality for *Pipistrellus pipistrellus* and *P. pygmaeus.* The lack of significance in tower height for predicting fatality is inconsistent with Barclay, Baerwald and Gruver (2007) and Rydell et al. (2010b). This perhaps highlights the importance of both assessing species specifically (other studies combined all species together) and also modelling fatality using all potential predictors (i.e. habitat, activity and turbine characteristics) which may influence fatality in a single model, rather than simple correlations between single predictors and the numbers of fatalities. In the present study, a lack of significance may however reflect insufficient power due to small sample and effect sizes relative to the number of potential predictors and fatalities. However, given that ground level activity is a better predictor of fatality than activity measured from the nacelle, and the distance between the ground and blade tip negatively predicted the probability of a *Pipistrellus pipistrellus* fatality (Chapter five), it is unlikely that more fatalities occur at taller turbines for *P. pipistrellus* and *P. pygmaeus,* at least for those killed during non-migratory flights (Voigt et al. 2012).

### *6.6.2. Important landscape metrics*

Habitat type and scale (landscape metric) are often assessed for their relative importance for bat activity and density (Fuentes-Montemayor *et al.* 2013; Lintott *et al.* 2014b). Despite habitat being important in determining bat distribution and density (Verboom & Huitema 1997, Walsh & Harris 1996), in the present study, habitat was not a significant predictor of the number of fatalities, and other variables (turbines cut-in speed and activity) were better predictors. This suggests it is not possible to site wind farms to reduce *Pipistrellus pipistrellus* and *P. pygmaeus* fatalities at turbines.

### *6.6.3. Estimating bat fatalities in Britain*

In Britain, estimated numbers of fatalities are similar to previous estimates across Europe (Rydell et al. 2010b; Voigt et al. 2012). However, recently an additional study of fatalities in Germany, estimates higher fatality rates, at 10-12 bats killed $t^{-1}$ $y^{-1}$, extrapolated to ~250,000 bats killed annually (Voigt et al. 2015). This is considerably higher than the fatality numbers estimated in this study. In North America, estimates of several hundred thousand (600,000) have been made and have included species of high conservation concern (Johnson et al. 2004). The higher fatality rates may be due to more bats migrating within North America and mainland Europe (~70% of bats killed at wind turbines in Germany, are killed during migratory flights; Voigt et al. 2015). Little is known about migration in mainland Britain, however due to its island geography providing a substantial barrier the scale of migration between Britain and continental Europe is likely to be relatively low (Moussy et al. 2012) and hence migratory related fatalities are also anticipated to be lower. Species which are common but are killed in low numbers in Britain are *Plecotus auritus* and *Myotis* spp. (IUCN 2014). Results from this study confirm that these species are not at high risk of fatality, which is likely to reflect their differing foraging strategies; close to vegetation compared to species in the *Pipistrellus* genus which forage predominately at edge habitats and in the open (Russ 2012).

Whilst edge and open aerial foragers are most at risk of fatality, even within a genus it was found that species are at differing risk of fatality. Based on the estimates presented here of the number of fatalities across Britain, relative to their population sizes, *Pipistrellus pygmaeus* are at 57% higher risk of fatality compared to *P. pipistrellus.* The differences in fatality risk between *Pipistrellus pipistrellus* and *P. pygmaeus* may be due to *P. pygmaeus* flying higher, more often within the rotor sweep area (Chapter two). Conversely the two species may be at similar risk of fatality, but given the lack of significance of predictors for *Pipistrellus pygmaeus* and the low pseudo $R^2$ value of the model, fatality estimates may be less reliable than those for *P. pipistrellus.* For both species confidence intervals are large (*Pipistrellus pipistrellus* fatalities: 95% CI 513 to 4,233, *P. pygmaeus* fatalities: 95% CI 1,270 to 4,894) and hence, mean risk levels may in fact be more similar than predicted by the model.

From this study, the estimated number of bats (*Pipistrellus pipistrellus* and *P. pygmaeus* combined) killed in the UK at large wind turbines (95% CI 1,782 to 9,127 bats killed $y^{-1}$) was higher than the numbers of bats estimated to be killed by small wind turbines (95% CI 161 to 3,363 bats killed $y^{-1}$; Minderman et al. 2015). This may be due to the attraction of bats to large turbines (Chapter three), in contrast to displacement at small turbines (Minderman et al. 2012). In addition, studies have positively related fatalities to the height of the turbine towers (Barclay, Baerwald & Gruver 2007; Rydell et al. 2010b), indicating that larger rotor sweep areas which are correlated with taller turbines, increase the probability of a bat fatality. This positive correlation with taller towers is also evidenced by the numbers of bats killed which are associated with high altitude feeding and long-distance migration behaviours (Lehnert et al. 2014, Rydell et al. 2010a, Voigt et al. 2012).

### *6.6.4. Conservation implications*

Evidence from this study shows that activity measured at ground level at operational wind farms is not a reliable method for predicting the number of fatalities of *Pipistrellus pipistrellus* and *P. pygmaeus*. Currently, bat activity is monitored at pre-construction wind farm sites for environmental impact assessments of wind turbines. Since, bats were shown to be attracted to turbines (Chapter three); using pre-construction activity for environmental impact assessments is likely to provide an inaccurate prediction of post-construction wind farm fatality risk. Curtailment of all wind turbines at low wind speeds could be an effective mitigation strategy. This study found that the turbine cut-in wind speed was a significant negative predictor of fatality for *Pipistrellus pipistrellus,* but not for *P. pygmaeus*. However, given that most of the activity found in this study occurred at low wind speeds (< 6 m $s^{-1}$, Chapter three); increasing turbine cut-in speeds is likely to be the most effective mitigation strategy for reducing fatalities for *Pipistrellus pygmaeus* and *P. pipistrellus.* Indeed, curtailment of wind turbines at low wind speeds has been the only successfully tested mitigation strategy to date to reduce bat fatalities (Arnett et al. 2010, Arnett et al. 2013a). The results presented here are inconclusive across similar species and therefore an experiment test of the effectiveness of curtailment in a European context is required.

Although mitigation measures are an important option, consideration also needs to be given to the scale of the conservation issue. The actual numbers of bat fatalities caused by wind turbines, for species with the highest fatalities in Europe, are small relative to population sizes. Other conservation issues for bats maybe of a higher concern, such as; loss of habitat through agricultural intensification and change of land use (Harris et al. 1995), and loss of roosts due to development (Walsh & Harris 1996). In terms of fatality numbers at wind farms, estimates presented here are in the order of 30-40 times lower than the number killed by domestic cats in Britain (Woods, McDonald & Harris 2003). Recent evidence suggests that many bats are also being killed by roof membranes used in modern houses where they roost (Waring, Essah & Gunnell 2014). Although little research has been conducted in this area, this threat is likely to be accelerating given the high rate of urbanisation (Seto et al. 2012) and re-development of older buildings more suitable for bats (Smit et al. 2014). Whilst it is important to develop wind power using an evidence based approach, it is also important that fatalities from wind farms are not overstated.

# Chapter seven:

# 7. General Discussion

Wind energy production has rapidly expanded across the globe over recent decades. Current evidence suggests that wind farms may cause declines in bat populations, due to the high numbers of fatalities found, raising bat fatalities at wind farms as a serious conservation issue. The impacts of large commercial wind turbines have been assessed by examining bat fatalities across wind farm sites, but often using different methodologies. The study reported here is one of the few large-scale systematic studies of bat activity and fatalities at wind farms, and is the first major study across Britain.

The major findings in this thesis were; that across Britain, each year during the peak fatality period (July to September), an estimated 2,373 *Pipistrellus pipistrellus* and 3,082 *P. pygmaeus* fatalities from wind turbines occur. Total all species activity, *Pipistrellus pipistrellus* activity and *P. pygmaeus* activity, whether monitored at ground or from the nacelle, were not good predictors of their respective fatalities. Whilst there was some evidence that *Pipistrellus pipistrellus* and *P. pygmaeus* activity monitored at ground level, was a significant predictor of the probability of their respective fatalities occurring, across wide ranging turbine types, fatality estimates were large. This is presumably due to the importance of turbine characteristics (i.e. distance between ground and blade tip and turbine cut-in speed) in predicting fatalities. The probability of a *Pipistrellus pipistrellus* fatality was negatively related to the distance between the ground and blade tip (lower rotor sweep area) and the numbers of *P. pipistrellus* fatalities was negatively related to the turbine cut-in wind speed. *Pipistrellus pipistrellus* and *P. pygmaeus* activity at turbines compared to paired controls was 46% and 34% higher, respectively, providing evidence that these species are attracted to turbines. These findings raise important issues for current environmental impact assessments, where at present, ecological consultants do not consider turbine characteristics in site assessments and monitor bat activity at pre-construction sites in an attempt to estimate fatality risk at operational turbines.

## 7.1. Evaluating current bat survey guidelines and implementing mitigation

At present, UK bat surveys for wind farms require that a desk survey be conducted to identify records of roosts and species composition within the area (e.g. Hundt 2012, Rodrigues et al. 2014). This is used to determine if medium and high-risk species are present (i.e. *Eptesicus* spp., *Nyctalus* spp., and *Pipistrellus* spp.). Bat acoustic surveys, both static and active (i.e. walked transects), are also conducted to assess species composition, activity levels, and locations of foraging and commuting routes. This information is then used to determine whether development of the wind farm can proceed, and if so, if any mitigation is required, such as, further post construction monitoring. Although wildlife surveys are now mandatory and much effort is expended to assess potential risk, it is uncommon that wind farms are refused planning consent based on wildlife reasons (e.g. 8% for small wind turbines; Park, Turner & Minderman 2013 ).

Survey method guidelines have minimum standards, but beyond this, ecological consultants make decisions on survey effort and design. This leads to variability in the quality and effectiveness of the surveys implemented (Hill & Arnold 2012). The present study compared EUROBATS guidelines (10 survey nights during the peak fatality period) with a more extensive survey period. After ten survey nights, when monitoring at ground level, four of the most high risk species in Britain *(Pipistrellus pipistrellus, P. pygmaeus, P. nathusii* and *Nyctalus noctula)* would have been detected at 97% of sites (with 95% confidence) and activity estimates were above the 70th percentile of site activity (determined after 14 nights). If survey restrictions were such that the number of detector nights was limited to 12 (maximum comparable in this study), for ground level monitoring, the optimal survey design for *Pipistrellus pipistrellus, P. nathusii* and *Nyctalus noctula* was to monitor two turbines for six nights, and for *P. pygmaeus* it was to monitor three turbines for four nights (Chapter two).

Assessing turbine characteristics (i.e. distance between ground and blade tip and turbine cut-in wind speed) resulted in better predictions of fatalities compared to activity. This is evidenced by the fact that fatalities occurred on nights when no acoustic activity was recorded. Thus, no recorded activity does not necessarily equate to no fatality risk. It is therefore important that future risk assessments take into account turbine characteristics to aid assessment of risk at a site. These improvements to risk assessment do not require additional survey effort and cost. Given that *Pipistrellus pipistrellus* and *P. pygmaeus* were detected at 98% and 96% of sites respectively; it could be assumed that these species would be detected at all wind farms within their range. Therefore, in a British context, mitigation (e.g. curtail turbines at wind speeds below 5 m $s^{-1}$) could be imposed across all wind farms. This would result in lower pre-construction survey costs for developers, whilst automatically defaulting to a more effective mitigation strategy for bats. If curtailment at low wind speed is not economically viable, then perhaps more restrictions on wind farm planning are required so that wind farms are only developed in locations where higher winds are more consistently above these low cut-in speeds.

Post-construction carcass surveys are becoming increasingly common; however, because fatalities are rare events, it is debatable how informative such surveys will be. Interpretation of the number of fatalities found in relation to populations is critical but these numbers (population) are typically unknown, even to a first approximation. Defining thresholds for the number of fatalities permitted is arbitrary without this information. Currently, reference ranges (bat activity levels within defined percentiles) are being developed where environmental (e.g. habitat and weather) and temporal (e.g. season) variables are used to classify activity as low, medium, and high. One could similarly rank fatalities at wind farms using a categorical approach to identify high fatality sites and target mitigation accordingly. Using this study and dividing sites into tertiles, fatality numbers at sites equated to; low: 0 fatalities $t^{-1}$ $month^{-1}$, medium: >0 to ≤ 0.5 fatalities $t^{-1}$ $month^{-1}$ and high: > 0.5 fatalities $t^{-1}$ $month^{-1}$. Each category could have a devised mitigation strategy (i.e. no mitigation, curtail >5 m $s^{-1}$, curtail > 6.5 m $s^{-1}$, respectively for each category). This method could be experimentally tested to assess whether the number of fatalities was significantly reduced and would be justifiable, given that fatalities found in the study were geographically widespread and occurred across wide ranging habitats. Currently, there is little evidence that curtailment mitigation is being applied in Britain; e.g. in this study, 87% (n = 15) of wind farms operational since 2011, had cut-in wind speeds of ≤ 5 m $s^{-1}$, with one wind farm using a cut-in speed of 2.5 m $s^{-1}$.

The lack of implementation of effective mitigation is perhaps due to the lack of communication between researchers, practitioners and policy makers (Milner-Gulland et al. 2012; Park, Turner & Minderman 2013). The increase in ecological surveys required for environmental impact assessments, has resulted in an increase in the number of ecological consultancies, particularly in the last few decades (Hill & Arnold 2012). The rapid improvements in technology in a relatively short time frame, has enabled large datasets to be gathered more easily. However, this ‘data gathering’ approach in the absence of interpretation and implementation of mitigation has been previously highlighted as a concern (Kaisler et al. 2013). This may have led to a focus on surveying, which is more profitable than implementing effective mitigation, for an industry worth between £1 bn and £3 (Hill & Arnold 2012).

## 7.2. Predicting bat fatalities

In this thesis, it has been shown that bat fatalities are difficult to predict due to one, or possibly a combination of the following reasons: the rarity of fatalities, bats being recorded acoustically outside the rotor sweep area, bats not echolocating when near turbines and a lack of relationship, either between activity and abundance or between abundance and the number of fatalities. On nights when no bat activity was recorded, a mean of 0.03 95% CI 0 to 0.09 *Pipistrellus pipistrellus* fatalities $t^{-1}$ $month^{-1}$ and a mean of 0.05 95% CI 0 to 0.12 *P. pygmaeus* fatalities $t^{-1}$ $month^{-1}$ occurred. These acoustically unrecorded fatalities may have occurred due to bats using vision rather than echolocation for navigation. This has been found when moon illumination or light levels are sufficiently high (Eklöf & Jones 2003; Holland 2009). In forests, bats have been found to be more active in the canopy compared to lower storey vegetation (i.e. shrub or sub-canopy) when moon illumination is higher (Hecker & Brigham 1999), indicating that vision rather than echolocation is used for navigation and foraging. At turbines, activity has also been shown to increase on moon-lit nights (Cryan et al. 2014). These behaviours may explain why some bat fatalities in this study occurred on nights where no activity was recorded. However, it is possible that bats were echolocating, regardless of moon illumination, within the risk zone, but were not recorded acoustically due to limitations of the recording equipment used. Moon illumination was not investigated within this thesis due to insufficient numbers of fatalities relative to predictor variables, and other predictors being more important.

These detection issues are not encountered when predicting fatalities for bird species at wind turbine sites, where absolute numbers are known and behavioural studies are more feasible (e.g. Eichhorn et al. 2012).

## 7.3. Species risk level

The species at risk of fatality at wind-farms in Britain are those that have also been identified elsewhere in Europe (Camina 2012; Rydell et al. 2010b). There was species-specific variation in the relationship between activity and fatality. There were also species differences in the predictors of the probability of a fatality (Chapter five) and the numbers of fatalities (Chapter six). For the same level of nightly activity, the mean risk was higher for *Pipistrellus pygmaeus* than *P. pipistrellus.* Although *Pipistrellus pipistrellus* and *P. pygmaeus* can be recorded from ground detectors, it is speculated here, that *P. pipistrellus* are flying more often below the rotor sweep area where they are at lower risk of fatality, resulting in a reduced activity and fatality association. This is substantiated by activity measured at the nacelle being a more important predictor for the probability of a *Pipistrellus pygmaeus* fatality compared to *P. pipistrellus* (i.e. scaled coefficient was 93% higher). The relationships between activity and fatalities have not been considered at the species level in previous studies (Johnson et al. 2004; Korner-Nievergelt et al. 2013).

## 7.4. Siting wind turbines

Habitat at wind farms may alter the risk of fatality for different species. Although habitat was not a significant predictor of the numbers of fatalities, for *Pipistrellus pipistrellus* and *P. pygmaeus,* the habitat that explained the most variability in the numbers of fatalities in models differed between the species. For *Pipistrellus pipistrellus*, the percentage cover of woodland within 1000 m of turbines explained most variability, with more fatalities at sites with less woodland, whereas for *P. pygmaeus* the minimum distance to riparian habitat explained the most variability, with more fatalities at sites with riparian habitat further away. These differences may be explained by the different habitat preferences between the species (e.g. Nicholls & Racey 2006a; Walsh & Harris 1996; Lintott et al. 2015). The lack of significance of landscape metrics in predicting the number of fatalities for either *Pipistrellus pipistrellus* or *P. pygmaeus* may be due to a large number of variables (e.g. bat activity, habitat, and turbine characteristics) relative to the number of sites and low fatality rates.

On a broader habitat scale, there was no pattern in the type of habitat (i.e. farmland, lowland moorland and upland moorland) that high fatalities and no fatalities occurred. Relatively high bat fatalities ($\geq$ 1 fatality $t^{-1}$ month$^{-1}$) were recorded at 13% of sites and no fatalities were recorded at 40% of sites. There were insufficient sites with predominately woodland habitat, permitting specific investigation of this habitat type. However, *Pipistrellus* spp. risk level is likely to increase at turbines ‘keyholed’ into woodland habitat, where the flight height of bats is typically above the canopy (Staton & Poulton 2012; Müller et al. 2013). It is therefore impossible to draw safe conclusions about where to site wind farms to minimize fatalities. An alternative assumption, based on evidence presented here, is that a wind farm sited in any habitat type may incur relatively high bat fatalities, given that the species suffering most fatalities are widespread generalists (Walsh & Harris 1996).

### 7.5. Are bat fatalities at wind farms in Europe a conservation issue?

Interpreting the numbers of fatalities in relation to population stability, and assessing the extent of any conservation issue is problematic, due to poor population size estimates for bats. Key parameters for individual species are required to enable bat population viability modelling. These include; longevity, fecundity, and survival rate (EUROBATS 2014; Huso & Dalthorp 2014). Estimating bat population sizes is difficult due to; their nocturnal habits, small size, utilization of several roosts, and their ability to cover large areas (including migration) which are difficult to survey (Sabol & Hudson 1995). In the future, the use of genetics may provide an alternative approach to population size estimates through the use of non-invasive capture-mark-recapture studies at a local scale (Puechmaille and Petit 2007), and/or effective population size estimates at a national scale (Razgour et al.2014; Wang 2005). Whilst current estimates are limited, since they are based on few data, they provide some basis for contextualising bat fatality numbers from wind turbines at a population level. Trends in bat populations have been assessed from annual roost surveys and acoustic surveys, where statistically robust temporal trends can be detected (Barlow et al. 2015).

In the UK, the number of wind farms has been growing exponentially over the last 20 years (Global Wind Energy Council 2014). Over this time frame, *Pipistrellus pipistrellus, P. pygmaeus* and *Nyctalus noctula* populations have all been increasing (Barlow et al. 2015). Other species at risk at wind farms in Britain (although not found to be at risk in this study), include, *Eptesicus serotinus, Nyctalus leisleri*, and *Pipistrellus nathusii* (Rydell et al. 2010b), which are also either stable or increasing during this 20 year time frame (Haysom 2013). However, in the study by Barlow et al. (2015), roost surveys, which are generally considered less reliable than field surveys, did not complement results from activity surveys, with declines for *Pipistrellus pipistrellus, P. pygmaeus* and *Eptesicus serotinus.*

Whilst wind turbines alone may not cause significant population declines, cumulative effects from increasing pressures of habitat loss (Harris et al. 1995), increasing light pollution (Stone, Jones & Harris 2009; Hölker et al. 2010), and the potential for more extreme weather under climate change (Meehl et al. 2000) may cause population declines. For a taxon that is relatively long lived, has low reproduction rates, and is slow to recover, small effects on survival can have large impacts on populations (Jones, Purvis & Gittleman 2003). Whilst common generalist species are less likely to become extinct (Jones, Purvis & Gittleman 2003; Rebelo, Tarroso & Jones 2010), there is some concern, that declines in these species can be overlooked, despite their ecological importance (Inger et al. 2015). The importance of undertaking conservation efforts for common and widespread species was highlighted by Gaston and Fuller (2007). Three factors that were identified were; a number of species previously described as common are now highly threatened or extinct, current declines are ongoing for large numbers of common and widespread species, and the processes causing these declines are expected to intensify. In this thesis, it is shown that although perceived to be relatively common, *Pipistrellus pipistrellus* and *P. pygmaeus* incur the highest fatalities from wind turbines. *Pipistrellus* spp. are also negatively impacted by urbanisation (Lintott et al. 2015), agricultural expansion and intensification (Harris et al. 1995), and loss of roosting and foraging habitat (Walsh & Harris 1996). The addition of wind turbine fatalities to these increasing pressures may destabilize populations and cause declines.

## 7.6. Climate change and bat conservation

In this thesis, it has been shown that wind energy can cause fatalities for a small proportion of common species. It is important that wind energy is developed using an evidence-based approach to minimize bat fatalities. Turbine and landscape characteristics that increase the risk level for bats have been outlined in this thesis. However, there is also a need to ensure that the larger picture of the potential impact of climate change is not forgotten. If $CO_2$ levels continue to rise and further destabilize the climate, impacts on a wide range of bats, and indeed all species and their habitats (Barnosky et al. 2011; Bellard et al. 2012), may have far graver consequences than the impacts shown in this thesis. Whilst direct comparisons between negative impacts from renewable compared to non-renewable energy are difficult, comparisons have been attempted for birds in the U.S., where fossil-fuel and nuclear electricity were estimated to kill 98% more birds (e.g. through collision and electrocution, poisoning and death caused by acid rain, mercury pollution and climate change) compared to wind energy (327,000 *c.f.* 7,000; Sovacool 2009). Although the Sovacool (2009) study has received some criticism in its methods and interpretations (Willis et al. 2010), a similar attempt to contextualise bat fatalities from wind energy compared to non-renewables would be valuable.

Due to the extensive investment made in the wind energy sector, it has become one of the most important renewable forms of energy, playing a vital role in producing clean energy. Halting the growth of renewable energy in favour of fossil fuels will undoubtedly result in a more extensive negative impact on the planet. Currently, under mid-level climate warming scenarios (i.e. temperature increases of 1.8–2.0 °C and $CO_2$ increases of 500–550 p.p.m.v. by 2050; Houghton et al. 2001), commitment to extinction is expected in 15-37% of all taxa by 2050 (Thomas et al.

2004). It is anticipated that species most at risk are endemics and those at the top of the food chain (Isaac & Williams 2007). Generalists are expected to be able to adapt most rapidly to the changing climate, and therefore less likely to go extinct (Isaac & Williams 2007). Among Chiropterans, species occurring within colder northern latitudes are predicted to be most at risk of extinction compared to bats in Mediterranean and temperate areas, which are more tolerant to changes in temperature (Rebelo, Tarroso & Jones 2010). We should therefore be cautious in our attempts to micro manage bat fatalities when a macro management approach could be more beneficial. In the words of Sir David King, “Climate change is not the biggest challenge of our time; it’s the biggest challenge of all time”.

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