An investigation of mining impacts on bats in South-West England

Submitted by Emma Theobald to the University of Exeter
as a thesis for the degree of
Masters by Research in Biological Sciences
in March 2018

I certify that all material in this thesis which is not my own work has been identified and that no material has previously been submitted and approved for the award of a degree by this or any other University.

Signature: ……………………………

Acknowledgements:

First of all a massive thank you to my research tutors Prof. David Hosken and Dr. Kelly Moyes for introducing me 'bat world' and for their continued guidance and support, also to Prof. Patrick Foster for his insight into the mining industry. I am very grateful to Wolf Minerals for providing the funding for this research and would like to thank Jess Easterbrook and Hannah Clarke of the Environmental Team and Michel Hughes for introducing me to the Drakelands site and answering my questions. I really appreciate the cooperation of the landowners who let me climb fences and wander through their fields and to set up detectors (David Cobbold in particular) - without your permission this research would not have been possible. Last but not least, I would like to thank Reece and my family and friends for their support and encouragement and for listening to my bat chat over the past year and a half!

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Abstract:

The extraction of minerals through open-pit mining can result in sudden and extensive land use change, often posing threats to local biodiversity. Bats are particularly vulnerable to the impacts of mining, but their metapopulation structure and wide-ranging roosting habits can make it challenging to monitor local populations. Here, we investigated the impacts of habitat loss and disturbance at Drakelands open-pit mine, the first new metal mine to be established within Britain in the past 45 years. This was addressed in two parts, firstly by analysing data collected by contracted ecologists at the site, in order to identify potential short-term shifts in bat activity and to evaluate the efficacy of mitigation measures. Secondly, by monitoring bat activity in the wider landscape to identify potential further-ranging impacts of the mine on local bat populations. In conjunction with this work we incorporated a field trial of a novel bat detector designed for long-term monitoring of bat activity. The results highlighted the multitude of factors which influence bat activity at a local level, and may provide a platform for continued research into the impacts of habitat fragmentation and anthropogenic noise at a species/ genus level. The information presented here will help to inform management decision making in regards to bat conservation, both at the Drakelands site and potentially at mining operations elsewhere.

Table of contents:

List of tables:

List of figures:

List of appendices:

Chapter 1: Introduction

1.1 Order Chiroptera

With more than 1300 species, bats (order Chiroptera) represent the second largest mammalian order next to Rodentia (Voight and Kingston, 2016). Bats occupy a wide range of habitats from deserts to tropical forests and exist on every continent other than Antarctica, including many oceanic islands (Kunz and Parsons, 2009). It is therefore not surprising that bats display huge trophic diversity, with seven feeding associations recognised (piscivory, sanguivory, nectarivory, frugivory, omnivory, carnivory, and insectivory) (Segura-Trujillo *et al.* 2016). Traditionally bats were divided into two suborders, Microchiroptera (microbats) and Megachiroptera (megabats). Microchiroptera use their larynx to emit echolocation pulses for navigation and prey capture, while Megachiroptera have well developed vision (although several Megabats also echolocate using tongue clicks) (Springer, 2013). However, more recent examination of morphological, molecular, fossil and behavioural evidence has led the scientific community to adopt a new chiropteran classification system, of suborders Yinpterochiroptera and Yangochiroptera (Lei and Dong, 2016). Yinpterochiroptera comprises the Old World fruit bats (previously Megabats) plus four microbat families in Rhinolophoidea, while Yangochiroptera includes 12 microbat families. Crucially, this arrangement implies that laryngeal echolocation evolved in the common ancestor of Chiroptera (and was subsequently lost in Old World fruit bats) or evolved independently within two different groups of microbats (Springer, 2013).

Unfortunately, many bat populations are under severe threat worldwide as a result of human activities and some face a growing risk of extinction (Racey, 2009). Economic evaluations of the benefits provided to humans by nature, termed ‘ecosystem services’ are frequently being used to support arguments for species conservation. Recent evaluations of the ecosystem services that bats provide have validated the valuable and unique roles they play in arthropod suppression, pollination, seed dispersal and forest regeneration (Kunz *et al*. 2011).

Over two thirds of known bat species are either obligate or facultative insectivores; however the extent of their control of agricultural pests is not well documented (Kunz *et al*. 2011). Despite there being limited research in quantifying the economic value of these ecosystem services, several studies have focussed on approximating the worth of pest control services provided by *Tadarida brasiliensis* in North America. During the summer *T. brasiliensis* forms enormous breeding colonies in Texas and Northern Mexico, acting as pest control for the cotton industry by feeding on the larvae of destructive agricultural pests such as the cotton bollworm *Helicoverpa zea*. Research by Cleveland *et al*. (2006) estimated that the value of avoided damage to cotton per year is $638,000 with an additional value of $100,000 saved on the avoided use of pesticides within an eight county region of southwestern Texas. However, updated estimates of the forging bat population in this region using thermal imaging suggest that the value of the avoided costs of pesticides in this region may be as high as $500,000 (Betke *et al*. 2008).

In addition to the ecosystem services provided through insect suppression, some bat species play crucial roles in pollination and seed dispersal in tropical and subtropical ecosystems. Approximately 528 species of flowering plant are known to be pollinated by bats, predominantly by species within the families *Pteropodidae* (Old World fruit bats) and *Phyllostomidae* (New World leaf-nosed bats) (Kunz *et al*. 2011). Bats disperse over long-distances and carry large pollen loads relative to many other pollinator groups, therefore depositing a wider variety of pollen genotypes onto plant stigmas. This promotes outcrossing, and maintains the genetic continuity of plant populations of considerable ecological and economic importance (Fleming *et al.* 2009). The fruit eating bats of *Phyllostomidae* and *Pteropodidae* further assist with plant propagation through seed dispersal within tropical forests. Research suggests that fruit eating bats are more likely to disperse seeds within clearings than birds, and therefore play a crucial role in forest regeneration following timber operations and fire (Altringham, 1996; Gorchov *et al*. 1993).

Further, bat guano collected from roost sites can provide a valuable organic fertiliser, particularly in regions such as Cambodia where arable land is frequently nutrient deficient and access to fertiliser is critical for national food security (Sothearen *et al.* 2014). Containing high concentrations of nitrates bat guano acts as an effective soil cleanser, fungicide, nematocide and compost activator and is only required in small quantities to increase the efficiency of plant growth (Shetty *et al.* 2013). The properties of other bat products are being explored in medicine, with ongoing research into the fibrinolytic agent Desmoteplase isolated from the saliva of sanguivorous bats (Jauch *et al*. 2013). Desmoteplase acts as an anticoagulant, and therefore has potential applications in the treatment of ischemic strokes and the prevention of heart attacks (Fernandez *et al*. 1999).

While evaluating the human benefits provided by bats may be beneficial for the conservation of some species (such as the pollinators and pest regulators) others which do not contribute to human wellbeing but have important ecological functions will not be prioritised by the ecosystem services approach (Ingram *et al.* 2012). For example, bats are vital in sustaining cave ecosystems where primary productivity is minimal. In these habitats, the nutrients derived from bat guano is critical for the continuation of fish and salamander populations and invertebrate communities (Kunz *et al*. 2011). Further, the high trophic level occupied by bats means that they are valuable bio indicators as they are particularly responsive to bioaccumulation of toxins, changes in arthropod prey populations and habitat loss. Changes in the abundance and activity of bats may be related to deterioration of water quality, habitat fragmentation, climate change, disease, agricultural intensification and pesticide use, therefore reflecting a wide range of pressures that affect other taxa (Jones *et al*. 2009).

1.2 Bat conservation

The assessment of mammalian data by the IUCN Red list in 2008 classed five bat species as extinct, 15.4% of species as threatened (categories critically endangered, endangered or vulnerable), 6.7% of species as near threatened and 17.7% as data deficient (IUCN Red List, 2008). Arguably one of the most serious threats to bat species worldwide is habitat destruction, which results in direct loss of foraging resources and roost sites. Remaining habitat patches must be large enough to sustain viable populations and habitat corridors are essential in enabling exchange of individuals between populations (Altringham, 1996). Evidence suggests that bat communities are more sensitive to the impacts of habitat degradation than other animal groups. A study by Muylaert *et al*. (2016) estimated that when forest fragmentation reduces forest cover beyond a threshold value of 47% bat species richness will experience steep declines (relative to a fragmentation threshold of 30% for birds and non-flying mammals). The global rate of land-use change may be exacerbated by the agricultural intensification that is predicted over the next 50 years, which in addition to removing roosting and foraging habitat may impact bat populations through the bioaccumulation of pesticides and by changing the availability of arthropod prey (Tilman *et al*. 2002; Voigt and Kingston, 2016; Swanepoel *et al*. 1999; Neuweiler, 2000; Kannan *et al*. 2010; Secord *et al*. 2015).

While habitat destruction has resulted in large-scale loss of natural roosts, urbanisation has led to the adoption of man-made structures as alternative roosting sites for some species. Structures such as barns, attics and churches are favoured by maternity colonies due to the fact they are heated quickly by the sun, while mine shafts can provide important hibernation, maternity and migration rest stops due to their cave-like microclimates (Neuweiler, 2000; Ducummon, 2000). Species which are able to thrive among the challenges and opportunities created by urban areas have been termed ‘synurbic’ and often show refined behavioural or ecological traits allowing them to exploit these niches (Russo and Ancillotto, 2015). The success of bats in responding to urbanisation is highly species-specific; those that possess high wing loadings and aspect ratios (indicators that they forage in open areas) and roost largely in man-made structures appear to adjust relatively well, provided there is sufficient tree cover (Voight and Kingston, 2016). However, even for synurbic species there is evidence that urban sites may act as ecological traps for bats; with factors such as human conflict, high levels of opportunistic predators and higher competition for food leading to negative responses to the urban environment and increased mortality rates (Russo and Ancillotto, 2015; Lintott *et al.* 2016).

Increasing urbanisation has contributed considerably towards dramatic increases in artificial lighting at night, with global levels rising by 6% per year (Hölker *et al*. 2010). Responses to artificial lighting by bats are species-specific, reflecting differences in flight morphology and performance. For light-tolerant species artificial lighting may act as a feeding resource by attracting high densities of insects. However, for light-shy species artificial lighting can result in spatial avoidance, habitat fragmentation and disruptions in the abundance and composition of insect prey (Stone *et al*. 2015). The illumination of buildings has been shown to delay roost emergence times and may reduce reproductive success by slowing juvenile growth rates (Mathews, 2015). Along roads the impacts of artificial lighting, noise and chemical pollution degrade habitat and contribute towards habitat fragmentation, barrier effects and edge effects. This is reflected in decreased levels of bat activity and diversity in proximity to roads, and lower reproductive output in areas where critical commuting routes have been severed (Berthinussen and Altringham, 2012b).

In response to the wide range of threats that bat populations face some countries have granted legal protection to bats, however the level of protection varies widely among nations. There are currently only two international agreements which have been specifically created to protect bats: The Agreement on the Conservation of Bats in Europe and the Program for the Conservation of Migratory Bats of Mexico and the United States (Mickleburgh *et al.* 2002; Hutson *et al*. 2001). In order to stimulate the creation of further action to protect bats at a regional or local level, two IUCN Action Plans (Megachiroptera and Microchiroptera) have been created, which identify conservation priorities and recommend conservation actions (Hutson *et al.* 2001; Mickleburgh *et al.* 1992). Non-governmental organisations (such as the Bat Conservation Trust, Bat Conservation International and The Lubee Bat Conservancy) now play a significant role in bat conservation globally, by drawing attention to the threats faced by bat populations, supporting research and conservation and funding local wildlife officers and public outreach services (Racey, 2009).

Within Europe, all bat species and their roost sites are protected by law under the Convention on the Conservation of European Wildlife and Natural Habitats (making it illegal to deliberately capture, keep, disturb, sell or kill a bat) and the Agreement on the Conservation of Populations of European Bats (Bern Convention, 1979; Bonn Convention, 1983; EUROBATS, 1994). Further, when deciding habitat conservation priorities, governments must aim to protect foraging areas and habitats that may be important to local bat populations. All

European bat species are also protected under the Habitats Directive, and additional protection is afforded to 4 species which are listed under Annex II, making it a requirement to designate roosts and foraging sites as Special Areas for Conservation (SACs) (Council Directive, 1992). Other international treaties such as the Ramsar Convention and the Rio Convention indirectly protect bats by protecting their foraging and roosting habitats (Hutson *et al.* 2001). Within the European Union environmental impact assessments are mandatory for many large developments, to determine the potential impacts that proposed development projects may have on local biodiversity, including habitat loss and pollution (The Environmental Impact Assessment Directive, 2014; Racey, 2009). Any development which may have a detrimental impact on bats or their roost sites must apply for a European Protected Species Licence, which if granted will permit these activities to proceed under certain conditions.

When a European Protected Species Licence is granted the applicant must demonstrate that the work conducted will not have a negative impact on the conservation status of any bat population. Frequently, the licence will specify that mitigation measures proportionate to the scale of works are implemented in order to reduce or compensate for any impacts that the development may have on local bat populations (Mitchell-Jones, 2004). Long term management and maintenance of mitigation measures is usually requested, in addition to post-development population monitoring to assess the success of the scheme. Despite the importance of post-development monitoring in informing future impact assessments and maximising the success of mitigation, research within England found that post-develop monitoring was only carried out at 19% of sites, and only 33% of licensees submitted post-development reports (Stone *et al.* 2013). Depending on the type and scale of impact and the significance of the population affected, mitigation may require sensitive timing of operations, or adhering to a specified lighting regime and particular methods of work. Within the wind energy sector mitigation may include adhering to specified operational parameters, such as stopping turbine blades in periods of low wind to reduce bat fatalities, as trialled in Canada (Baerwald *et al*. 2009).

Where natural roosts have been lost, mitigation may include the creation of artificial roosts such as bat boxes and bat houses. The adoption rates of bat boxes have been shown to range from 7% to 100% across studies, however it is important that the type of box selected matches the requirements of the target species and is designed to exclude non-target species including other mammalian groups, birds, and insects (Mering and Chambers, 2014). Although bat boxes are considered suitable mitigation for the loss of tree roosts, they are not generally considered sufficient compensation for loss of maternity roosts located in buildings (McAney and Hanniffy, 2015). Other compensation measures include the creation and restoration of habitats and foraging areas, and the strengthening of linear features such as hedgerows for commuting purposes. Features such as bat gantries, underpasses, and habitat bridges may be constructed to promote connectivity where development and transport links result in the fragmentation of existing habitats. Evidence suggests that bat gantries are ineffective and bats continue to use select severed but unmitigated commuting routes, however underpasses which are built on pre-construction commuting routes have demonstrated greater success (Berthinussen and Altringham, 2012a). Habitat bridges designated across Europe to connect woodland for medium to large mammals such as deer have been found to act as guiding features for bats, allowing flight paths to cross highways without risk of collision with vehicles (O’Connor and Green, 2011).

The United Kingdom has 18 resident species of bat, 17 of which sustain known breeding populations (Russ, 2012). Of these, the IUCN Red List classifies *Myotis bechsteinii* and *Barbastella barbastellus* as ‘near threatened’, and the remaining species are listed as ‘least concern’ (Bat Conservation Trust, 2016a). All UK bat species and roosts are protected by law, although differences in legislation exist between the constituent countries. Both *Rhinolophus ferrumequinum* and *Myotis myotis* were legally protected in 1975, but protection was later afforded to all UK species in 1981 with the implementation of the Wildlife and Countryside Act, which was transposed from the Bern Convention as a result of UK obligations (Wildlife and Countryside Act, 1981; Bern Convention, 1979; Stebbings, 1988). This was later updated by the UK government to include additional clauses and levels of protection for bats, including the Countryside and Rights of Way Act (2000) and the Natural Environment and Rural Communities Act (2006). All UK bat species are also protected under the Conservation (Natural Habitats, &c.) Regulations (1994), which was later updated to The Conservation of Habitats and Species Regulations (2012). This legislation was transposed into UK law as a result of the Habitats Directive (1992), and requires additional protection under the Natura 2000 Network for the roosts and foraging sites of *Rhinolophus hipposideros, Rhinolophus ferrumequinum, Barbastella barbastellus* and *Myotis bechsteinii* which are listed as Annex II species (Council Directive, 1992).

Within the UK, the Natura 2000 Network is comprised of Special Areas of Conservation (SACs) and Special Protection Areas (SPAs) (European Commission, 2016; JNCC, 2014). It is the responsibility of the UK government to designate SACs to protect the range of species and habitats listed as Annex I and II under the Habitats Directive. The following sites have been designated to protect the maternity and hibernacula roosts of Annex II listed species: *R. hipposideros* (13 sites), *R. ferrumequinum* (11 sites), *B. barbastellus* (9 sites), and *M. bechsteinii* (9 sites) (JNCC, 2016). Other forms of protected areas within the UK declared under the Wildlife and Countryside Act (1981) include Ramsar sites, Sites of Special Scientific Interest (SSSIs) and National Nature Reserves (NNRs), which may be designated to safeguard significant bat roost sites or feeding grounds but may also indirectly conserve general bat habitat (JNCC, 1989).

1.3 Bats in relation to mining activities

The long-life span of bats makes them particularly susceptible to the effects of bioaccumulation of toxins such as heavy metals, pyrites and clays (Wilkinson and South, 2002). These contaminants are frequently produced as mining residue, and upon contact with water can form acid mine drainage which may be ingested by animals through drinking water or contaminated food chains. Assessments of the heavy metal contents in the liver of insectivorous bats in Southern Brazil identified higher levels of chromium, nickel, copper and lead in *Molossus molossus* and of copper and iron in *Tadarida brasiliensis* in individuals living in coal mining areas relative to individuals living in control areas (Zocche *et al*. 2010). A recent assessment of heavy metal concentrations in *Pipistrellus* species sampled from across Britain suggests that intake of these contaminants is due to both recent and long-term exposure. This study found that 21% of the individuals sampled contained high enough concentrations of at least one metal to evoke toxic effects associated with kidney damage (Hernout *et al*. 2016). Metal contamination may act as an additional stressor involved in the continuing bat population declines observed worldwide. Further investigation into water contamination as a result of mining is needed, particularly in Africa where mining activity is abundant but very little research has been carried out regarding water quality (Voigt and Kingston, 2016).

Bodies of open water in desert landscapes form important resources for bats by providing drinking opportunities and foraging sites of emerging aquatic insects. Unfortunately, toxins associated with mining are a major anthropogenic source of water contamination (Voigt and Kingston, 2016). In gold extraction both the processes of vat and heap leaching require the use of cyanide which is commonly stored in open mill-tailings ponds which can stretch over 200 acres in size. Investigation into mammalian deaths at cyanide ponds based on data from 1984-1989 across 75 mines in Arizona, California and Nevada revealed that 34% of fatalities were bats, based on a sample size of 519 mammals. Although not feasible for the larger ponds, netting of tailings ponds to prevent wildlife from coming into contact with contaminated water appears to be the most effective method of mitigation (Clark, 1991).

Despite the negative impacts that mining can have on the environment, abandoned mines serve as important year round sanctuaries for bats. Urban and agricultural developments, deforestation, human disturbance of caves and cave commercialisation have led to the displacement of bats from their traditional roosts. Many displaced bats have gradually moved into abandoned mines, which offer stable microclimates similar to cave environments (Tuttle and Taylor, 1998). The colonial nature of many bat species and their instinctive commitment to certain sites makes populations particularly vulnerable to the closure of mines, and hundreds of thousands of individuals may be lost with a single closure. Until relatively recently, few mines were evaluated for their importance to bats before being permanently closed by backfilling, capping, or blasting (Ducummon, 2000). Presently, abandoned mines are only permanently closed as a last-resort such as when the mine is unstable and in danger of collapse or there is a risk of contaminant release (Brown, Altenbach and Sherwin, 2000). The installation of bat gates can allow bats to continue accessing abandoned mine sites while preventing human entry, however it is crucial that the gate design is tailored to suit the resident species (dependent on the colony size and flight path gates may still be rejected by bats) (Tuttle, 1977; Voigt and Kingston, 2016; Pugh and Altringham, 2005). Further research is needed into the responses of species to gating worldwide, as well as the potential for gates to alter cave and mine microclimates.

At an active mine site the principal physical disturbance is the open pits and associated mine waste facilities (MWF). Underground mining generally results in a relatively small MWF ranging in size from a few acres to tens of acres. However in open pit mining the waste rock to ore ratio is commonly 1:3, therefore very large volumes of waste rock are removed from the pits and deposited in a nearby MWF (Hudson *et al*. 1999). Following the closure of the mine, the disturbed areas are stabilised and habitat is commonly restored either through vegetation activities or conservation offsets (Sonter *et al.* 2014). Comparisons of restored open-pit mine areas with nearby undisturbed areas have found that successful restoration of vegetative communities can occur within 30-50 years (Kuter, 2013; Nielsen and Kelly, 2016). However, for bat species which rely on microhabitat features such as deadwood and tree hollows which require at least 100 years to develop, restored habitat in mining areas may initially have limited value (Vesk, 2008). Several studies have used radio telemetry to track bats to their diurnal roosts and established that many species select tall, mature trees of 150-200 years of age which are in the mid-late stages of decay with exfoliating bark. One such study tracking *Nyctophilus gouldi* and *Vespadelus regulus* in a post-mining landscape in Southwestern Australia found that despite bats travelling through restored forest, all of the tagged individuals roosted in un-mined remnant forest (Burgar *et al.* 2015; Watrous *et al*. 2006).

In the long-term, there is hope that the majority of restored forest will be preserved and allowed to develop into quality roost habitat. Surveys of reclaimed mine sites in the Midwestern United States found that following bond release only 2% of private landowners engaged in the clearing of replanted woodland, which in all cases was to convert the land for either residential or commercial use (Briggeman *et al*. 2007). Additionally, the lack of tree-roost opportunities in young forests can be supplemented with bat boxes to enhance the area for bats (Bat Conservation Trust, 2016b). The landscapes resulting from open-pit mining often contain a highly fragmented mix of forest, agricultural land, water features and wildlife habitat which can offer valuable foraging and roosting opportunities for a wide range of bat species (Briggeman *et al*. 2007).

Despite the potential for site restoration local bat populations in proximity to an active mine site may suffer from habitat loss, shifts in prey availability and fragmentation of the surrounding landscape by link-roads providing access to the mine. Further, there may disturbance impacts associated with mining including noise pollution and vibrations from vehicles, processing machinery and surface mine blasting (ELAW, 2010). Relatively little is known about the effects of anthropogenic noise on bats, and research into the thresholds and characteristics of sound and vibration which influence the daily and seasonal movement of bats is lacking (Player and Keim, 2015). Therefore, this project will investigate the impacts of disturbance, land-use change and efficacy of habitat enhancements on the spatial distribution of local bat populations at the active Drakelands open-pit mine in Southwest England.

1.4 Introduction to the study site: Drakelands Mine

Drakelands Mine is located near the village of Hemerdon in Southwest Britain, in close proximity to China clay mines operated by Imerys Minerals and Sibelco UK Ltd. The tungsten deposit at Drakelands Mine (previously known as Hemerdon) was first discovered in 1867, and mineral working was subsequently carried out during the First and Second World Wars between 1919-1920 and 1934-1944. Following an extended period without mining activity, an application to was submitted to Devon County Council to reopen and extend the mine by Amax Exploration and Hemerdon Mining and Smelting Ltd. However, this application was eventually refused following a public inquiry in 1984. The following year, Amax Exploration submitted a revised application comprising of an opencast pit, a mineral processing plant and associated tip and the construction of a ‘Link Road’ (Michel Hughes Associates, 2009). Although Devon County Council approved the application, due to low tungsten prices the project did not proceed. The planning permission at Drakelands remained valid and the project was later acquired by Australian company Wolf Minerals Ltd in 2007.

Wolf Minerals was established as a metals exploration and development company in 2006, and Drakelands Mine remains their core asset with over 200 permanent on-site employees. Site construction began in 2014 with full production commencing in 2015. The processing plant was first fed in July 2015 with August 2015 being the first full month of production, and currently operates 24 hours per day (McGlinchey, pers.comm, 2017). The final pit will measure 850 metres long, 450 meters wide and ultimately reach a depth of 260 metres (Figure 1.1) (Wolf Minerals Ltd, 2016). A three million tonne per year operation is scheduled for the ten year duration of the mine-life, although this period is likely to be extended further through the conversion of resources to reserves. This translates to the processing of 26.7 million tonnes of ore for a wolframite recovery of approximately 66%, in addition to the annual production of 1,000 tonnes of tin concentrate (Micon International, 2016). This will render the United Kingdom a globally significant tungsten exporter, and as the first metal mine to be established within Britain in the last 45 years Drakelands will provide valuable income to local economies.

Discovered in 1758, tungsten occurs naturally only in the form of chemical compounds and was not isolated as a metal until 1783. Although over 20 tungsten bearing minerals are known to occur, only wolframite and scheelite are important for industrial use. At 3410°C, tungsten has the highest melting point of any metal as well as the highest tensile strength. Tungsten is also a good thermal and electrical conductor, has good resistance to corrosion and possesses the smallest compressibility of any metal (Wolf Minerals Ltd, 2016). Worldwide demand for tungsten continues to grow, with usage in a wide range of industries including mining and construction, chemistry, aviation, and electronics. Cemented tungsten carbides have a diamond-like hardness and are commonly used in cutting and grinding tools for the shaping of metals, alloys, ceramics and plastics. Tungsten metal products include lighting filaments, electrodes, wires, rods and electrical contacts (International Tungsten Industry Association, 2011). Tungsten was recently named as one of 14 critical raw minerals essential for the economy of the European Union; the significant deposit at Drakelands has the potential to meet this demand for the United Kingdom for many years to come (Wolf Minerals Ltd, 2016).

Figure 1.1
Aerial view of the study site, Drakelands Mine. The location of: **a)** the mine waste facility **b)** offices **c)** processing plant **d)** open pit (Photograph credit to Wolf Minerals Ltd, 2016).

Compensating for the impacts that a development may have on local bat populations often poses management challenges for mining companies and the relevant regulatory agencies (Sonter *et al*. 2014). It is understood that the Drakelands mine development has and will result in significant landscape modification and have an impact on local biodiversity in both the short and medium term. However, Wolf Minerals Ltd are working to minimise adverse environmental impacts to ensure that there is no net loss of biodiversity (Michel Hughes Associates, 2009). Baseline habitat surveys of the site by contracted ecologists in 2009 revealed two ecologically distinct but contiguous zones (Figure 1.2). The first, identified as the ‘Hemerdon Mine’ area was predominantly scrub and improved grassland, and included the disused Hemerdon Mine and associated buildings. The second zone identified as the ‘Crownhill Down’ area predominantly consisted of acid grassland, with patches of broadleaf woodland, heath and mire (Michel Hughes Associates, 2009). Planning permission for Drakelands mine spans both the Hemerdon Mine and Crownhill Down area.

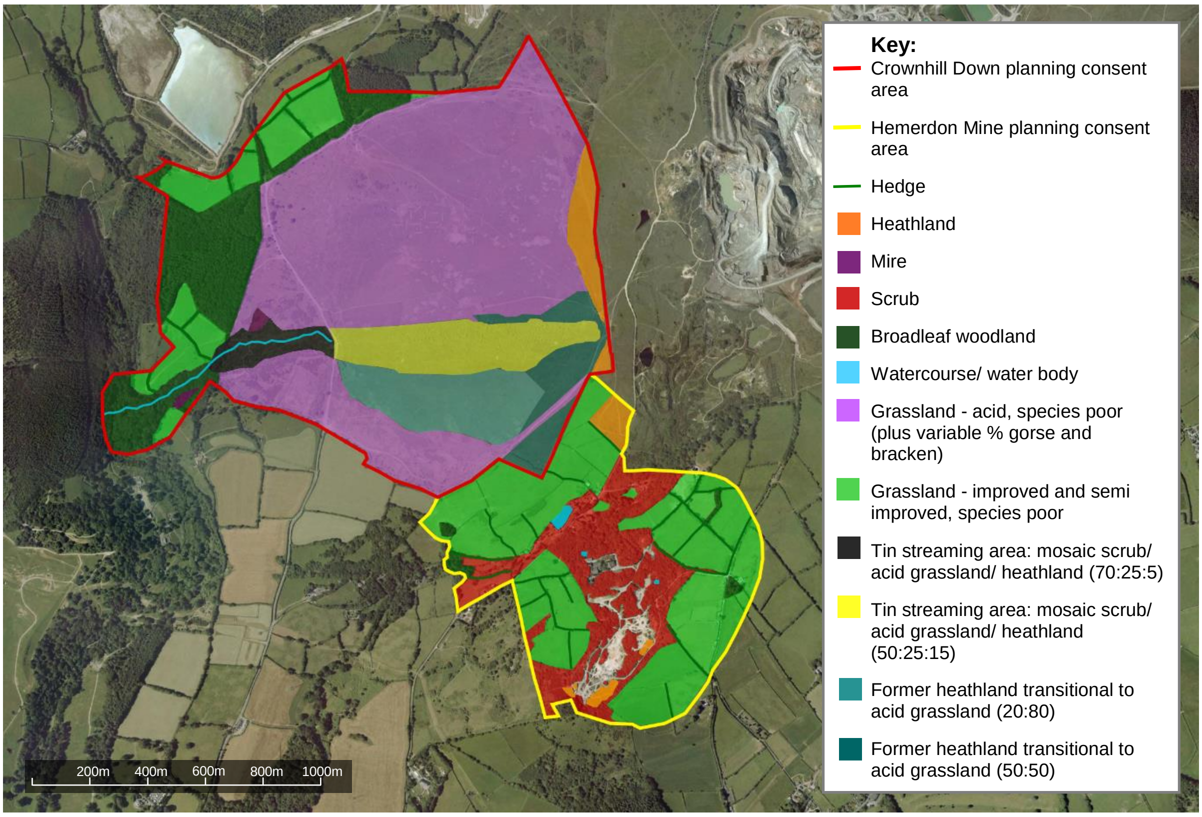


Figure 1.2

Results of the phase 1 habitat survey conducted by Michel Hughes Associates in 2009. This survey revealed two ecologically distinct but contiguous zones, referred to as Crownhill Down and Hemerdon Mine. Planning permission for Drakelands mine spans both regions (Aerial image © Google Earth, 2010)

Baseline bat surveys conducted from 2008-2013 identified at least 11 species utilizing the proposed site of development with low to moderate levels of activity, which was largely dominated by *Pipistrellus pipistrellus* and to a lesser extent *Pipistrellus pygmaeus* and *Myotis* species (Appendix A: table 1) (SLR Consulting Ltd, 2013). Further surveys identified two tree roosts for five *Pipistrellus pygmaeus*, a building roost site for three *Plecotus auritus*, and within a disused mine building a non-breeding summer roost for five *P.pipistrellus* and a hibernation roost for seven *R.ferrumequinum* (Michel Hughes Associates, 2014). Three EPSLs were granted by Natural England to permit the destruction of these roost sites between 2013 and 2015 (Appendix A: figure 1). In order to compensate for the loss of roost sites at Drakelands mine a total of 81 bat boxes were installed prior to June 2014 within retained areas of woodland. A variety of box types were selected to supply potential roost sites for a wide range of bat species, including larger boxes selected to provide breeding and hibernation roost opportunities. In addition, 12 remaining buildings identified as being used by roosting bats were retained from demolishment, one of which was modified to provide a new roost provision (Gillingham, 2014).

In order to maintain and strengthen foraging and commuting habitat for local bat populations, significant habitat creation is being undertaken at Drakelands mine (Gillingham, 2014). This includes the planting of approximately 9 ha of native broadleaf woodland across the Drakelands site, which commenced in 2014 (Michel Hughes Associates, 2013) (approximately 87 ha of pre-existing habitat was lost in the construction of the mine, the majority of which was grassland, heathland and scrub). Standing deadwood obtained from woodland clearing has been transplanted to these nurseries to provide roosting opportunities during the juvenile stages of tree growth (Gillingham, 2014). The areas of woodland planting have been incorporated into a framework of 56 Biodiversity Enhancement Zones, to create a mosaic of appropriately managed habitats which will enhance the site for bats. These zones are made up of retained habitat patches as well as newly created habitats, consisting of woodland, grassland, heath, mire, freshwater habitats and boundary strengthening (Michel Hughes Associates, 2013).

In addition to the impacts of habitat loss and fragmentation, the installation of artificial lighting at the site has the potential to affect the abundance and distribution of local bat populations (Stone *et al*. 2015). In order to minimise impacts a lighting strategy will be implemented throughout the lifetime of the mine, to ensure that habitats potentially used by light-sensitive species remain unlit. Where possible lighting regimes will be triggered by motion sensors and where permanent lighting is required the lamps will be cowled to limit light spill. Lux levels will be monitored at five locations across the site, while levels along boundary vegetation will remain below 4 lux (Gillingham, 2014).

Wolf Minerals believe that long-term success hinges on sustainable development and proactive environmental management. They therefore provided funding for this degree of Masters by Research to explore the potential impacts of Drakelands Mine on local bat populations, as part of the offsite opportunities as required by the Environment Agency (SLR Consulting Ltd, 2013). This project seeks to evaluate the impacts of disturbance and land-use change and efficacy of habitat enhancements on local populations by assessing trends in species abundance and richness in the surrounding area. In addition, analysis will be carried out on the annual monitoring data collected by contracted ecologists following commencement of operations. The deepened understanding into the impacts of open pit mining on local bat populations achieved through this study will allow recommendations to be made for future management and monitoring of this site, while the insight gained into disturbance effects and habitat fragmentation may be relevant across a wide range of developments.

Chapter 2: Investigating short-term trends in bat activity at a new open-pit mining development

2.1 Abstract

Historically, the extraction of minerals through open-pit mining has usually resulted in long-term environmental impacts. However, conserving biodiversity through all stages of a mine’s lifecycle is now an industry priority, with environmental mitigation, monitoring and appropriate site restoration forming an integral part of operations. Bats are particularly vulnerable to the impacts of mining, but their metapopulation structure and wide-ranging roosting habits make it challenging to monitor local populations. Here, we outline the monitoring and mitigation work conducted from 2014 to 2016 for bats at the first new metal mine to be established within Britain in the past 45 years. No significant change was identified in the number of bats emerging from retained building roosts. Data collected by contracted ecologists using remote detector monitoring was used to assess bat activity at the site and to identify potential short-term shifts in the activity levels of local bats. Although species richness was not found to vary over the three year monitoring period, total bat activity was lower in 2016 relative to 2015 due to decreased activity of *P. pipistrellus* and *P. pygmaeus,* thought to be the result of shifts in prey availability. Further, *Plecotus* species experienced sharp declines in activity which may be linked to increased susceptibility to the impacts of disturbance at the mine site, as a result of their foraging ecology. This assessment can be used to inform management decisions on future developments both at this site and elsewhere.

2.2 Introduction

The dramatic alteration of ecosystems by human populations, particularly since the industrial revolution has led to significant impacts on biodiversity worldwide (Díaz *et al.* 2006; Steffen *et al.* 2011). The mining industry has been instrumental in the development of civilisation; playing a crucial role in the iron and bronze ages, the industrial revolution and the infrastructure of today’s digital age (MBendi, 2016). However the discovery, extraction and processing of minerals is widely regarded as an environmentally disruptive activity, with impacts potentially persisting long after the closure of a mine (Bebbington *et al*. 2008). Mining currently occurs on every continent other than Antarctica, and exploration emphasis is now shifting toward areas that have been little explored or previously had restricted accessibility due to politics, legislation or infrastructure (Jébrak, 2012). As a result of this expansion, mining is currently a major anthropogenic source of environmental contamination and habitat loss worldwide (Holden and Jacobson, 2012).

At an active mine site the principal physical disturbance is a result of the open pits and associated mine waste facilities where waste rock is disposed. Open pit mining, in which material is extracted from an open pit to access strategic minerals, commonly results in the production of very large quantities of waste rock (Hudson, Fox and Plumlee, 1999). The resulting mine waste facilities and areas of excavation almost always experience biodiversity loss and habitat fragmentation through the destruction and displacement of resident species (Environmental Law Alliance Worldwide, 2010). Further, the surrounding landscape may be impacted by soil degradation, the release of pollutants, the use of scare water and disturbance impacts such as noise and vibration (Campbell, 2009).

Globally, it is estimated that mining threatens nearly 40 percent of undeveloped tracts of forest (Whitmore, 2006). Even for highly mobile animals such as bats, declines in the size of forest patches can lead to decreased roost availability and foraging opportunities (Campbell *et al.* 1996). The scarcity of population estimates and lack of life history knowledge for many bat species presents a significant challenge when considering the impacts of mining on local populations (Barclay, 2014; Morrison and Fox, 2009). Due to their longevity, low reproductive output and high metabolic rates bats are particularly vulnerable to the impacts of bioaccumulation of toxins produced by mining (Voigt and Kingston, 2016; Wilkinson and South, 2002). Despite the negative impacts that mining may have on bat populations, the landscapes resulting from open-pit mining often contain a highly fragmented mix of habitats and water features which can offer valuable foraging and roosting opportunities for a range of bat species (Briggeman *et al*. 2007). Further, abandoned underground mines provide stable microclimates which act as valuable roost sites (Tuttle and Taylor, 1998).

Although bats make up one fifth of extant mammal species, many populations are now in decline with over 20 percent of species classified as threatened or near-threatened (Gunnel and Simmons, 2005, IUCN Red List, 2008). The Bat Specialist Group of the IUCN Species Survival Commission has compiled global Action Plans for Microchiropteran *and* Megachiropteran bats, which provide a global framework for bat conservation (Mickleburgh *et al*. 2002). National legislation for bat protection varies widely, from no protection to full protection of both roosts and individuals. Bat conservation can benefit greatly from international collaboration, proper enforcement of regulations and the protection of foraging grounds and roost sites (Hutson *et al.* 2001; Mickleburgh *et al.* 1992).

Within the United Kingdom, all 18 resident bat species and their roost sites are protected by both domestic and international legislation (Russ, 2012; Zeale *et al.* 2016). In addition many large developments, including mining operations, must also undertake and Ecological Impact Assessment, with the level of survey required dependent on the size of the development footprint and prior information regarding bat activity at the site (Mclean, 2010). Proposed developments with a large footprint such as mineral extraction operations are required to undertake thorough baseline surveys for bat activity and a desk study of the surrounding area, to identify roost sites, important foraging areas and strategic flyways (Hundt, 2012). If potential detrimental impacts to local bat populations or their roosts sites are identified, then a European Protected Species Licence may be granted to allow the proposed activities to proceed. Frequently, the licence will specify that mitigation measures proportionate to the scale of works are implemented in order to reduce or compensate for any impacts (Natural England, 2010; Mitchell-Jones, 2004).

The rising standards required by both regulators and the general public means that environmental protection is now a priority within the mining industry. As a result, ‘best practice methodologies’ now make up approximately five percent of the capital and operating costs of new mining developments (Environment Australia, 2002). In addition to carrying out habitat restoration following the closure of the mine, environmental enhancement often occurs while mining is still underway (Braker *et al.*

2013). Ecological compensation for local bat populations includes the erection of bat boxes to provide roosting opportunities, the creation of habitat corridors and measures to enhance foraging resources by creating invertebrate habitat (Nielsen and Kelly, 2016; Bat Conservation Trust, 2016).

It is important that the effectiveness of mitigation measures implemented for bats are evaluated throughout the lifetime of large scale mining developments (Berthinussen and Altringham, 2012b). Long-term monitoring using appropriate techniques may minimise the ineffective use of financial resources, by identifying species-specific responses by bats to the impacts of habitat loss and disturbance on a local scale (Grift *et al.* 2013). Here, we analyse data collected by contracted ecologists over a three year period at Drakelands mine, the first new metal mine to be established within Britain in the last 45 years. Ideally, we would determine the effectiveness of mitigation measures in maintaining local bat populations by comparing activity levels post-development to those collected at the site pre-development. However, variation in the data collection methods pre and post-development means that this comparison is not possible. Instead, we identify potential short term trends in local bat activity on a species/ genus level following development of the site, by analysing remote detector and roost survey monitoring data, with the aim to inform management decision making at this site and other future mining developments.

2.21 Mitigation for bats at the study site

Compensating for the impacts that a development may have on local bat populations often poses management challenges for mining companies and the relevant regulatory agencies (Sonter *et al*. 2014). It is understood that the Drakelands mine development has and will continue to result in significant landscape modification, impacting on local biodiversity in both the short and medium term (Michel Hughes Associates, 2009). However, Wolf Minerals Ltd are working to minimise adverse environmental impacts to ensure that there is no net loss of biodiversity. Construction of the Drakelands site was completed in 2013, although the transfer of waste material from the pit to the mine waste facility is ongoing. The habitats that were cleared in the construction of the site were predominantly grassland (acid, improved and semi improved) with smaller patches of scrub and broadleaf woodland.

Three EPSLs were granted by Natural England, to permit the destruction of roost sites within woodland and existing buildings. Collectively, these sites provided roosts for up to three *Plecotus auritus*, seven *Rhinolophus ferrumequinum* and ten *Pipistrellus pipistrellus* (for further details of EPSLs and initial habitat composition see Chapter 1 section 1.5) (Michel Hughes Associates, 2014).

In order to compensate for the loss of these roost sites a total of 81 bat boxes were installed at the site prior to June 2014 (Gillingham, 2014). Further, 12 buildings identified as being used by roosting bats were retained and one new building roost provision was created (Gillingham, 2014). Significant habitat creation has also been undertaken at Drakelands mine, including the planting of approximately 9ha of native broadleaf woodland which commenced in 2014 (Gillingham, 2014; Michel Hughes Associates, 2013). These areas of planting have been incorporated into a framework of 56 Biodiversity Enhancement Zones, to create a mosaic of appropriately managed habitats including woodland, grassland, heath and freshwater habitats which will enhance the site for bats. In order to minimise the effects of artificial lighting on bat activity a lighting strategy has been implemented across the site, to ensure that habitats potentially used by light-sensitive species remain unlit (Gillingham, 2014).

2.3 Methods

2.31 Data collection

Evening emergence surveys were conducted at 12 retained roosts and one new roost provision within the Drakelands site by contracted ecologists, from 2014-2016. Surveys were conducted biannually between May and August, in order to identify potential breeding colonies (Gillingham, 2014). Surveying began 15 minutes before sunset and terminated 90 minutes after sunset. Emerging bats were identified to a species level, based on visual observations and the subsequent analysis of echolocation calls recorded using a range of ultrasonic detectors. Of the two sampling sessions, the greater number of individuals observed of each species was recorded as the annual emergence figure for each roost (Gillingham, 2013).

From 2014-2016 remote monitoring was conducted simultaneously at 14 locations around the Mine Waste Facility by contracted ecologists. However, one location was not monitored during 2014 due to construction activity. In total, 122 five-night sampling sessions were conducted, totalling 610 nights of monitoring. For the purpose of analysis, monitoring locations were classified as ‘grassland’ or ‘woodland’ depending on the habitat in which the detector was situated (note: all woodland monitoring locations were well-established, none were recently planted) (Figure 2.1). Three monitoring sessions were scheduled annually; once during spring, summer and autumn. A Song Meter SM2 detector (Wildlife Acoustics, Massachusetts, USA) was positioned at each monitoring location and left to record from 30 minutes before sunset until sunrise, for five continuous nights. In the case heavy rainfall, high wind speeds or low temperature, detectors were left to record for additional nights and the period of less extreme weather was selected for analysis, as these environmental factors are known to affect levels of bat activity (Voigt *et al.* 2011; Cryan *et al.* 2014; Barros *et al.* 2014).

Data were visualised in Song Scope Bio-acoustics software (Wildlife Acoustics, Massachusetts, USA). Echolocation calls were visually inspected and assigned to a species or genus based on comparison to a reference library of known species. Echolocation calls assigned to *Myotis* and *Plecotus* were only identified to a genus level, due to similarities in call parameters (Kunz and Parsons, 2009; Russ, 2012; Jennings *et al.* 2008). Echolocation calls lasting up to 20 seconds were defined as a single ‘pass’. Species richness was calculated for each five night monitoring session, consisting of the sum of the number of species detected (or genus for *Myotis* and *Plecotus*). Total bat activity was calculated for each five night monitoring session as the sum of the number of passes of all species, and was used as an index of bat activity at each sampling location rather than quantifying actual abundance (Ober and Hayes, 2008). Nightly totals of bat activity were not available, therefore environmental data such as temperature, wind speed and rainfall could not be included in any of the statistical models (unless calculated as a five night average, which was decided to have limited value).

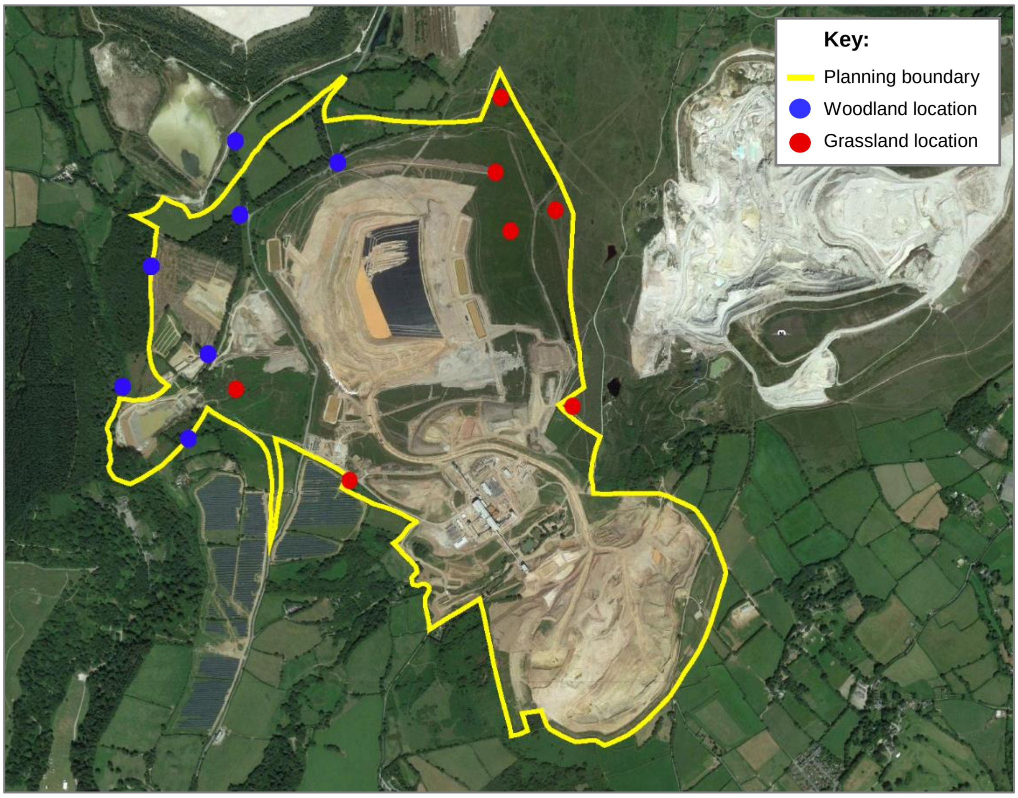


Figure 2.1
Positioning and habitat classification of remote detector monitoring locations at the Drakelands site (Aerial image © Google Earth, 2015).

2.32 Analysis

Statistical analyses were conducted using R version 3.3.1 (R Core Team, 2016). All response variables were assessed for normality, homogeneity of variance and over dispersion using standard diagnostic procedures. Assumptions were checked and appropriate transformations were made to reduce residual variance where necessary (Grueber *et al*. 2011).

Emergence survey data:

Emergence survey data were analysed on a species level, using a Kruskal-Wallis test to examine whether the number of individuals identified leaving the retained roosts varied by monitoring year at the $p < 0.05$ significance level. Monitoring year was entered as a categorical variable, in order to identify non-linear trends in species richness and activity over the three year monitoring period.

Remote monitoring data:

General Linear Mixed Models (GLMs) were constructed using the package “lme4” to investigate whether bat species richness and total activity levels (totals per five night sampling session) varied by monitoring year at Drakelands mine (Bates *et al*. 2015). The response variable of total activity was transformed to ln (passes + 1) to reduce heterogeneity resulting from large variation in activity levels between nights while also accounting for nights with zero passes (Berthinussen and Altringham, 2012a). All global models contained the fixed effects of monitoring year and habitat type. Monitoring year was entered as a categorical variable, in order to identify non-linear trends in species richness and activity over the three year monitoring period. The global models also included an interaction between monitoring year and habitat type, as activity and richness trends may differ within habitat types over the monitoring period. Models were fit using maximum likelihood and a normal error distribution. All models contained the random effect of location (to account for pseudo-replication as multiple monitoring sessions were conducted across 14 locations), and month (to account for temporal autocorrelation as the month of monitoring sessions differed among years). Assessing the potential impacts of spatial autocorrelation at the study site is difficult, because the results are likely to be confounded by the close grouping of woodland monitoring locations in the western area of the site, and the grouping of grassland monitoring locations in the eastern area of the site (Figure 2.1).

Bat activity was also analysed on a species/ genus level using GLMs for *P. pipistrellus, P. pygmaeus* and *Myotis* species which were identified most frequently. The response variable of number of passes was transformed to ln (passes + 1) to reduce heterogeneity resulting from large variation in activity levels between nights while also accounting for nights with zero passes. Again, the global models contained the fixed effects of monitoring year and habitat type with an interaction between the two variables, plus the random effects of location and month. Models were fit using maximum likelihood and a normal error distribution.

GLMs were built using all possible combinations of predictors using the package “MuMIn” and then ranked using Akaike’s Information Criterion corrected for small sample size (AIC_c), which penalizes models with many explanatory variables

(Burnham and Anderson, 2010). Each model was then given an Akaike weight (ΔAIC_c), based on the difference in AIC_c value for that model compared with the AIC_c value of the best fitting model. Models for which $\Delta AIC_c \leqslant 2$ were considered to have equivalent support and were validated by visual examination of residuals and q-q plots (Burnham *et al.* 2011; Bolker *et al.* 2009). According to the rules of parsimony, where multiple models received equivalent support the simplest model containing the fewest factors was selected as the minimum adequate model (Busemeyer *et al.* 2015). The minimum adequate model was then fit using restricted maximum likelihood estimation to obtain unbiased parameter estimates. Tukey's honest significant difference test (HSD) was computed using the "multcomp" package to determine which treatment combinations differed significantly, with $p < 0.05$ considered statistically significant.

On a species/genus level, activity by *N. noctula*, *R. ferrumequinum* and *Plecotus* species was detected less frequently. Therefore, the response variable of activity was converted to binary data of presence/absence, per five night sampling session. We investigated whether monitoring year had an effect on the likelihood of detecting each species/genus by building Generalised Linear Models (GLzMs), with a binomial error distribution. Models were constructed using the package "lme4" and fit using maximum-likelihood with a logit-link function. The global models contained the fixed effects of monitoring year and habitat type. Again, monitoring year was entered as a categorical variable, in order to identify non-linear trends in species richness and activity over the three year monitoring period. All models contained the random effects of location and month.

Using a Chi-Squared difference test the global model was compared with a null model containing only the fixed effect of habitat type and the random effects of location and month. A value of $p < 0.05$ indicted that the global model had a significantly better fit than the null model, and therefore monitoring year was an important predictor of the response variable of presence/ absence. A post-hoc Tukey's honest significant difference test (HSD) was computed using the "multcomp" package with the best-fitting model, to determine whether treatment combinations differed significantly within the variables of year or habitat type.

The following species were detected infrequently: *E. serotinus, R. hipposideros*, *N. leisleri*, *B. barbastellus*, and *P. nathusii*. Due to insufficient data, performing mixed modelling to identify changes in activity levels by monitoring year was not possible. Therefore percentage change was calculated between monitoring years, using the proportion of five-night sessions in which each species was detected.

2.4 Results

Over the three year monitoring period a total of 118,268 bat passes and at least 11 species were identified. Bat species were *Pipistrellus pipistrellus, Pipistrellus pygmaeus, Pipistrellus nathusii, Rhinolophus ferrumequinum, Rhinolophus hipposideros , Barbastellus barbastellus, Nyctalus noctula, Nyctalus leisleri, Eptesicus serotinus, Myotis* species and *Plecotus* species (Table 2.1).

Table 2.1
Summary table for number of bat passes per species, and detection frequency over the sampling period. Sampling was conducted from 2014-2016, over 122 five-night sampling sessions, totalling 610 nights of monitoring.

Bat species	Number of passes	% of total passes	% of sampling sessions detected
P. pipistrellus	94,364	79.79	94.26
P. pygmaeus	17,700	14.97	76.23
P. nathusii	31	0.03	2.46
R. ferrumequinum	190	0.16	46.72
R. hipposideros	92	0.08	17.21
B. barbastellus	18	0.02	8.20
N. noctula	737	0.62	54.10
N. leisleri	66	0.06	7.38
E. serotinus	124	0.10	25.41
Myotis species	4,637	3.92	74.59
Plecotus species	309	0.26	29.51

2.41 Effect of monitoring year on emergence counts

Three species were identified emerging from the retained roost buildings; *P. pipistrellus, P. auritus,* and *R. hipposideros* (Figure 2.2). The mean number of emerging individuals was not found to differ significantly between monitoring years for *P. pipistrellus* ($H_{(2)} = 1.663$, $p = 0.4354$), *P. auritus* ($H_{(2)} = 1.487$, $p = 0.476$) or *R. hipposideros* ($H_{(2)} = 1.028$, $p = 0.598$).

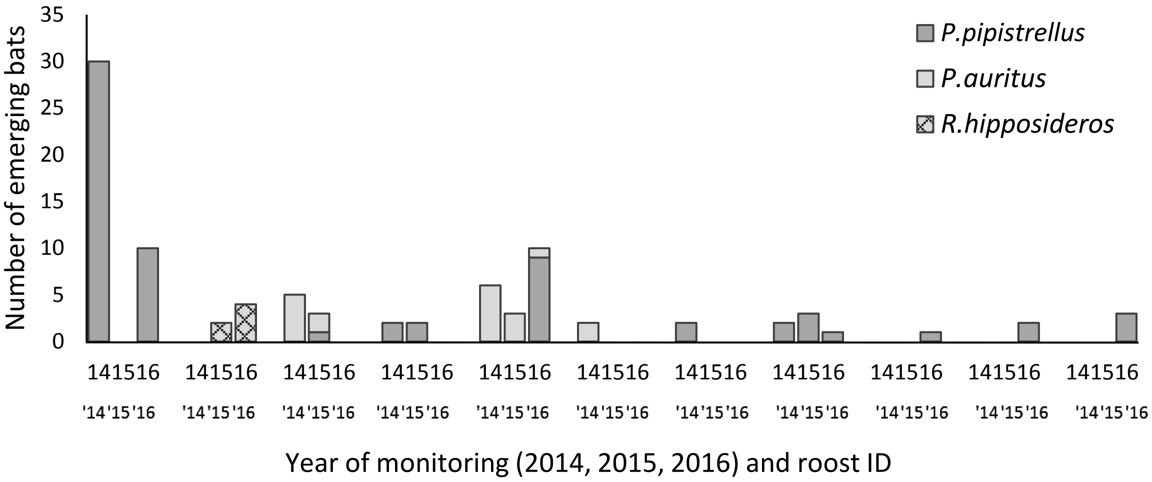


Figure 2.2

The number of bats observed emerging from the retained roost buildings during summer monitoring sessions from 2014-2016. Two sampling sessions were conducted annually, and the results of the sampling session with the greater number of emerging bats of each species is displayed here. Note: no bats were observed emerging from two of the retained roosts during any of the monitoring years so these are not displayed.

2.42 Effect of monitoring year on bat species richness

Monitoring year was not retained in the minimum adequate model investigating the factors which influence species richness, and therefore may not be considered a significant predictor of species richness (Model 1b, Table 2.2; Table 2.3). Post-hoc analysis revealed that species richness was lower in grassland than woodland monitoring locations (Tukey's test: $p < 0.001$). However, given that year was retained within the model with the lowest AICc (Model 1a, Table 2.2) the predicted values of species richness using this model have been presented in Figure 2.3. Species richness declines within both woodland and grassland locations over the three year monitoring period. Using alternative statistical methods such as model averaging this trend may have been found to be significant (Burnham and Anderson, 2010).

Table 2.2
The full set of models for species richness. Shown is the deviance (Dev), the number of model parameters (*K*), the Akaike's Information Criterion (AICc) and the difference in Akaike's Information Criterion between each model and the top-ranked model (ΔAICc) and the Akaike weight (w_i) of each model. Of the top model set ($\Delta AICc \leqslant 2$), the most parsimonious model (with the fewest parameters) was selected as the minimum adequate model and is shown in bold.

Model rank	**Model name**	**Response = species richness; random effects = date and location**	**Dev**	***K***	**AICc**	**ΔAICc**	w_i
1/3	1a	Year + Habitat	462.9	4	476.9	0.0	0.63
2/3	**1b**	**Habitat**	**468.7**	**3**	**478.7**	**1.8**	**0.25**
3/3	1c	Year * Habitat	462.2	5	480.2	3.3	0.12

Table 2.3
Summary of parameter estimates for the minimum adequate model for species richness. Shown is the estimates, standard error (SE), and the lower and upper confidence intervals (5%, 95%) for parameters and the variance of random effects.

Response: Species richness, random effects = location and month			
Parameter	**Estimate**	**SE**	**Confidence interval (5,95)**
Intercept†	3.553	0.459	2.805, 4.308
Habitat: woodland	1.775	0.379	1.151, 2.401
			Random Effects (variance)
Location			0.231
Month			0.806

†Habitat: grassland was the reference category.

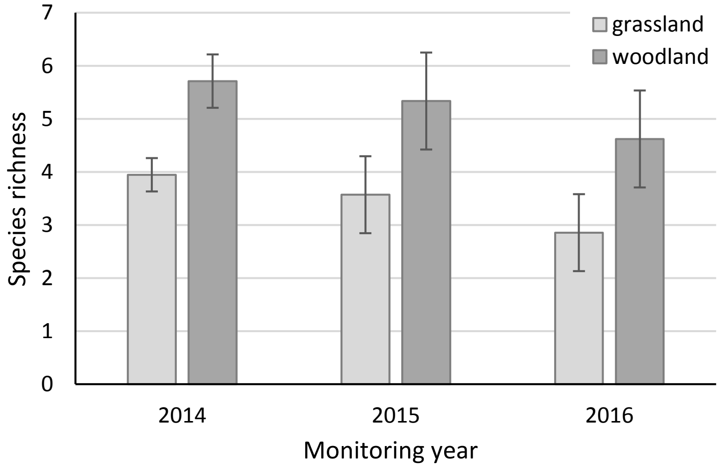


Figure 2.3
Species richness over the three year monitoring period within habitat type, predicted by model 1a (Table 2.2) with standard error.

2.43 Effect of monitoring year on total bat activity

Monitoring year and habitat type were retained in the minimum adequate model for the factors which influence total bat activity and therefore may be considered significant predictors (Model 2a, Table 2.4; Table 2.5). The model predicts that total bat activity was significantly lower in 2016 relative to 2015 (Tukey's test: $p < 0.001$) (Figure 2.4a), but that total activity in 2014 was not significantly different to activity in 2015 or 2016. Total activity was found to be lower in grassland than woodland monitoring locations (Tukey's test: $p < 0.002$). Model 2b contained an interaction between year and habitat and was supported by AICc (Table 2.4), but was not selected as the top model using the minimum adequate model method. However, the raw data have been displayed in Figure 2.4b to allow visualisation of the interaction. The trends appear similar to the model 2a predicted values, with the reduction in activity from 2015 to 2016 within both habitat types remaining apparent.

Table 2.4

The full set of models for total bat activity. Shown is the deviance (Dev), the number of model parameters (K), the Akaike's Information Criterion (AICc) and the difference in Akaike's Information Criterion between each model and the top-ranked model (ΔAICc) and the Akaike weight (w_i) of each model. Of the top model set (ΔAICc $\leqslant 2$), the most parsimonious model (with the fewest parameters) was selected as the minimum adequate model and is shown in bold.

Model rank	Model name	Response = ln(total activity + 1); random effects = date and location	Dev	K	AICc	ΔAICc	w_i
1/3	**2a**	**Year + Habitat**	**462.3**	**4**	**476.3**	**0.0**	**0.61**
2/3	2b	Year * Habitat	459.2	5	477.2	0.9	0.39
3/3	2c	Habitat	477.3	3	487.3	10.0	0.00

Table 2.5

Summary of parameter estimates for minimum adequate model for total bat activity. Shown is the estimates, standard error (SE), and the lower and upper confidence intervals (5%, 95%) for parameters and the variance of random effects.

Response: ln(total bat activity + 1), random effects = location and month			
Parameter	**Estimate**	**SE**	**Confidence interval (5, 95)**
Intercept [†]	3.825	0.701	2.740, 4.910
Year: 2015	0.416	0.629	-0.586, 1.420
Year: 2016	-0.798	0.628	-1.799, 0.204
Habitat: woodland	2.457	0.781	1.195, 3.719
			Random Effects (variance)
Location			1.910
Month			0.445

[†] Year: 2014 and Habitat: grassland was the reference category

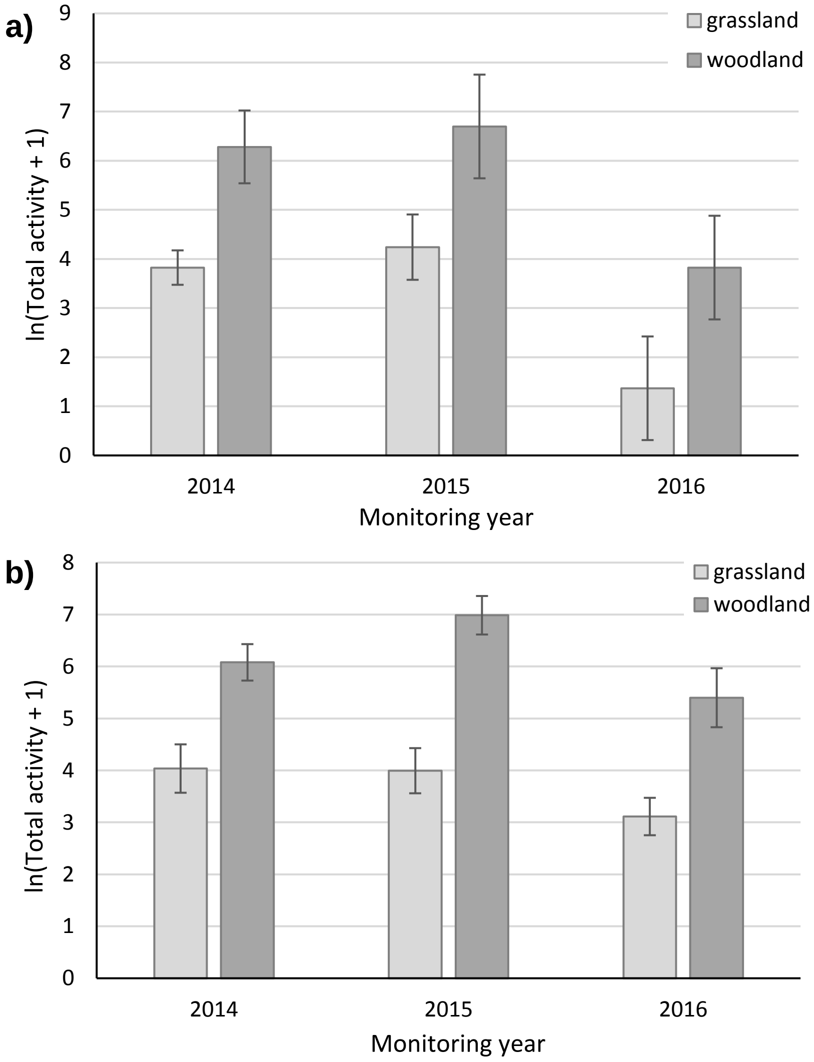


Figure 2.4

a) Total activity levels over the three year monitoring period, predicted by the minimum adequate model within habitat type (Table 2.5) with standard error. Total activity was transformed to ln(passes + 1) to reduce heterogeneity.

b) Total activity levels over the three year monitoring period (raw data) with standard error. Total activity was transformed to ln(passes + 1) to reduce heterogeneity.

2.44 Effect of monitoring year on bat activity on a species/ genus level

Monitoring year and habitat type were retained in the minimum adequate models for *P. pipistrellus* and *P. pygmaeus* activity and therefore may be considered significant predictors of activity for these species (Table 2.6a, model 3b; Table 2.6b, model 4a; Table 2.7a, b). For *Myotis* species the minimum adequate model included habitat type but did not include monitoring year, indicating that year was not a significant predictor of *Myotis* activity (Table 2.6c, model 5b; Table 2.7c).

Post-hoc analysis revealed that *P. pipistrellus* activity was significantly lower in 2016, relative to 2015 (Tukey's test: $p < 0.001$) (Figure 2.5a). *P. pygmaeus* activity was found to be significantly lower in 2016 relative to 2014 and 2015 (Tukey's test: $p<0.02$, $p < 0.002$ respectively) (Figure 2.5b). Across all species, activity was found to be significantly lower at grassland monitoring locations relative to woodland monitoring locations (Tukey's test: *P. pipistrellus* $p<0.004$, *P. pygmaeus* $p<0.001$, *Myotis* $p<0.002$).

Table 2.6

The full set of models for bat activity on a species/ genus level: **a)** *P. pipistrellus* **b)** *P. pygmaeus* **c)** *Myotis* species. Shown is the deviance (Dev), the number of model parameters (*K*), the Akaike's Information Criterion (AICc), the difference in Akaike's Information Criterion between each model and the top-ranked model (ΔAICc) and the Akaike weight (w_i) of each model. Of the top model set (ΔAICc ⩽ 2) the most parsimonious model (with the fewest parameters) was selected as the minimum adequate model and is shown in bold.

a)

Model rank	Model name	Response = ln (*P. pipistrellus* passes+1); random effects = date and location	Dev	*K*	AICc	ΔAICc	w_i
1/3	3a	Year * Habitat	460.4	5	478.4	0.0	0.63
2/3	**3b**	**Year + Habitat**	**465.5**	**4**	**479.5**	**1.1**	**0.37**
3/3	3c	Habitat	484.1	3	494.1	15.7	0.00

b)

Model rank	Model name	Response = ln (*P. pygmaeus* passes+1); random effects = date and location	Dev	*K*	AICc	ΔAICc	w_i
1/3	**4a**	**Year + Habitat**	**446.3**	**4**	**460.3**	**0.0**	**0.70**
2/3	4b	Year * Habitat	444.0	5	262.0	1.7	0.30
3/3	4c	Habitat	460.7	3	470.7	10.4	0.00

c)

Model rank	Model name	Response = ln (*Myotis* passes+1); random effects = date and location	Dev	*K*	AICc	ΔAICc	w_i
1/3	**5b**	**Habitat**	**395.3**	**3**	**405.3**	**0.0**	**0.51**
2/3	5a	Year + Habitat	391.7	4	405.7	0.4	0.42
3/3	5c	Year * Habitat	391.2	5	409.2	3.9	0.07

Table 2.7

Summary of parameter estimates for the minimum adequate models (ΔAICc $\leqslant 2$) for bat activity on a species/ genus level: **a)** *P. pipistrellus* **b)** *P. pygmaeus* **c)** *Myotis* species. Shown is the estimates, standard error (SE), and the lower and upper confidence intervals (5%, 95%) for parameters and the variance of random effects.

a)

Response: ln(*P. pipistrellus* passes+1), random effects = location and month			
Parameter	**Estimate**	**SE**	**Confidence interval (5,95)**
(Intercept) [†]	3.463	0.782	2.254, 4.671
Year: 2015	0.512	0.727	-0.646, 1.671
Year: 2016	-0.853	0.726	-2.010, 0.305
Habitat: woodland	2.439	0.846	1.069, 3.809
			Random Effects (variance)
Location			2.283
Month			0.646

[†] Year: 2014 and Habitat: grassland was the reference category.

b)

Response: ln(*P. pygmaeus* passes+1), random effects = location and month			
Parameter	**Estimate**	**SE**	**Confidence interval (5,95)**
(Intercept) [†]	1.951	0.568	1.063, 2.839
Year: 2015	-0.332	0.475	-1.098, 0.433
Year: 2016	-1.329	0.474	-2.092, -0.565
Habitat: woodland	2.507	0.663	1.439, 3.575
			Random Effects (variance)
Location			1.332
Month			0.203

[†] Year: 2014 and Habitat: grassland was the reference category.

c)

Response: ln(*Myotis* passes+1), random effects = location and month			
Parameter	**Estimate**	**SE**	**Confidence interval (5,95)**
(Intercept) [†]	1.095	0.399	0.453, 1.739
Habitat: woodland	1.528	0.467	0.768, 2.289
			Random Effects (variance)
Location			0.629
Month			0.286

[†] Habitat: grassland was the reference category.

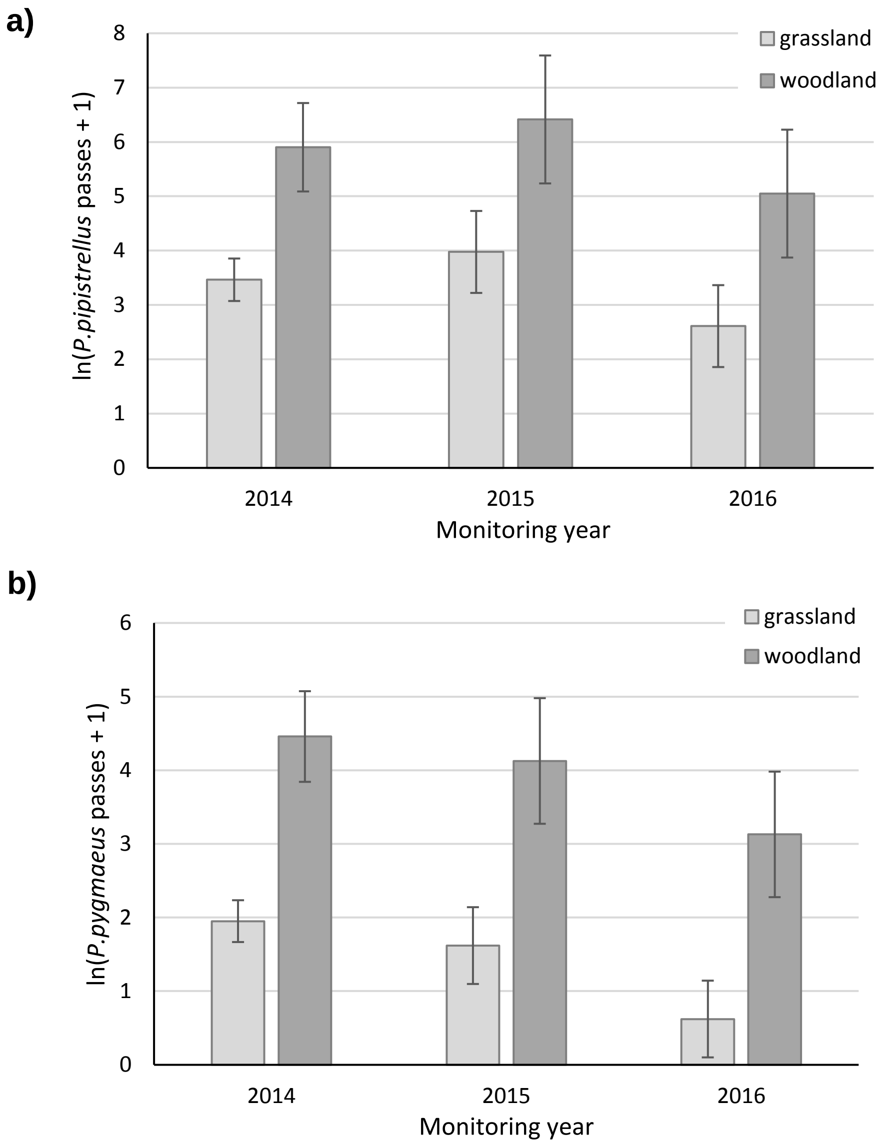


Figure 2.5
The activity levels of **a)** *P. pipistrellus* **b)** *P. pygmaeus* over the three year monitoring period, predicted by the minimum adequate models within habitat type, with standard error (Table 2.7a, b). Activity levels were transformed to ln(passes + 1) to reduce heterogeneity.

Monitoring year was found to be a significant predictor of *N. noctula* and *Plecotus* presence ($\chi^2_{(2)}$ = 6.769, p = 0.033 and $\chi^2_{(2)}$ = 8.228, p = 0.016 respectively, Table 2.8a, c). Post-hoc analysis revealed that *N. noctula* was more likely to be detected in 2016 relative to 2015 (Tukey's test: p=0.039) and that *Plecotus* species were more likely to be detected in 2014, relative to 2015 (Tukey's test: p=0.027) (Figure 2.6a, b). As zero *Plecotus* passes were identified in 2016 it was not possible to calculate a pairwise comparison for this year, however the lack of passes indicates a substantial decrease in activity. Monitoring year was not found to be a significant predictor of *R. ferrumequinum* presence ($\chi^2_{(2)}$ = 0.892, p = 0.640, Table 2.8b).

Post-hoc analysis revealed that *R. ferrumequinum* was more likely to be detected in woodland than grassland (Tukey's test: p<0.001). Habitat type was not a significant predictor of presence of *N. noctula* or *Plecotus* species. The proportion of monitoring sessions in which *R. hipposideros, B. barabastellus*, *E. serotinus* and *P. nathusii* were detected increased in 2015 relative to 2014 (Table 2.9). Only *N. leisleri* was detected less frequently in 2015 relative to 2014. *P. nathusii* and *N. leisleri* were detected more frequently in 2016 relative to 2015, while *R. hipposideros, B. barabastellus* and *E. serotinus* were detected less frequently.

Table 2.8
Summary of parameter estimates for the best fitting model for **a)** *N. noctula* **b)** *R. ferrumequinum* **c)** *Plecotus* species presence**.** Shown is the estimates, standard error (SE), and the lower and upper confidence intervals (5%, 95%) for parameters and the variance of random effects.

a)

Response: Probability of detecting *N. noctula*, random effects = location and month			
Parameter	**Estimate**	**SE**	**Confidence interval (10,90)**
(Intercept) †	-0.415	0.645	-1.631, 0.760
Year: 2015	0.018	0.813	-1.534, 1.569
Year: 2016	1.216	0.829	-0.324, 2.818
Habitat: woodland	0.431	0.462	-0.363, 1.269
			Random Effects (variance)
Location			0.172
Month			0.631

†Year: 2014 and Habitat: grassland were the reference category.

b)

Response: Probability of detecting *R. ferrumequinum*, random effects = location and month			
Parameter	**Estimate**	**SE**	**Confidence interval (10,90)**
(Intercept)[†]	-0.943	0.397	-1.693, -0.271
Habitat: woodland	1.512	0.468	0.762, 2.400
			Random Effects (variance)
Location			0.146
Month			0.242

[†] Habitat: grassland was the reference category.

c)

Response: Probability of detecting *Plecotus* species, random effects = location and month			
Parameter	**Estimate**	**SE**	**Confidence interval (10,90)**
(Intercept)[†]	1.115	1.562	*
Year: 2015	-5.181	2.099	*
Year: 2016	-48.503	0.000	*
Habitat: woodland	1.889	1.023	*
			Random Effects (variance)
Location			1.856
Month			5.193

[†] Year: 2014 and Habitat: grassland were the reference category.

* Unable to compute confidence interval due to singularity, because no *Plecotus* passes were identified in 2016.

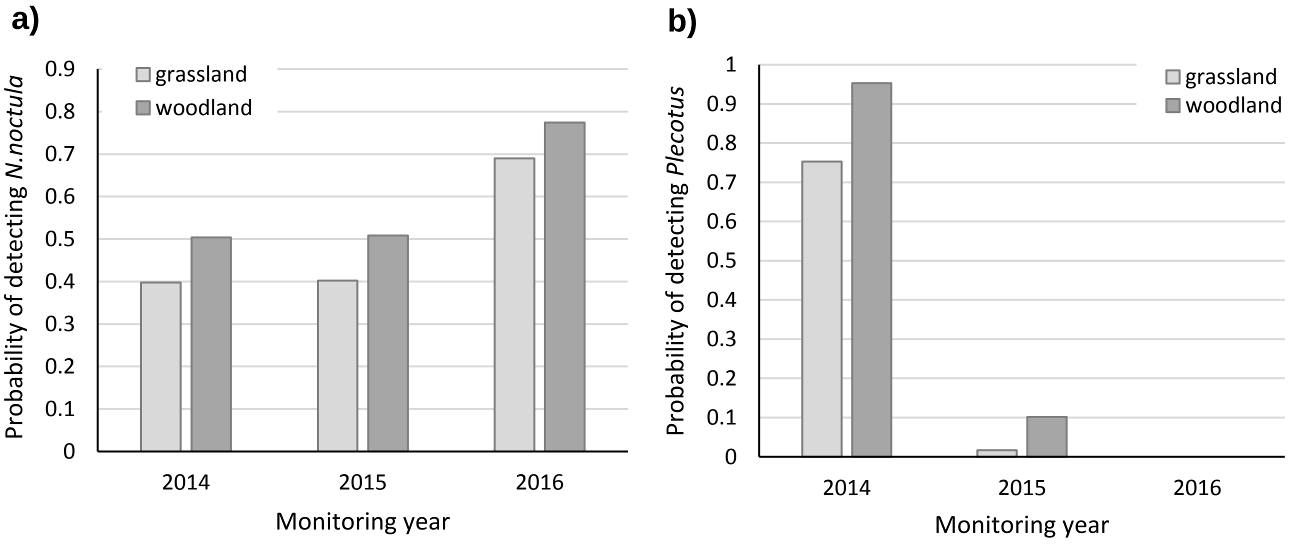


Figure 2.6
The probability of detection of **a)** *N. noctula* and **b)** *Plecotus* species over the three year monitoring period (within a five day monitoring session) within habitat type, predicted by the best fitting model (Table 2.8a, c). For the standard error of model parameters refer to Table 2.8a, c.

Table 2.9
The proportion of sampling sessions in which species were detected per year and the associated percentage change between monitoring years. Where a species was not detected during an entire monitoring year percentage change could not be calculated, and is represented by N/A.

Proportion of sessions detected				**Percentage change**	
Species	**2014**	**2015**	**2016**	**2014-2015**	**2015-2016**
P.nathusii	0.00	0.02	0.05	N/A	95.2
R.hipposideros	0.15	0.22	0.14	42.7	-34.9
B.barbastellus	0.03	0.17	0.05	565.9	-72.1
N.leisleri	0.15	0.00	0.07	-100	N/A
E.serotinus	0.18	0.34	0.24	90.2	-30.3

2.45 Environmental conditions

Although the incorporation of weather parameters into statistical models was not possible due to the lack of nightly activity data, summary statistics are presented in Figure 2.7 to enable a visual examination of annual trends which may have influenced bat activity at the site. The monitoring season of 2015 experienced the lowest average temperature (2014=15.8°C, 2015=14.6°C, 2016=15.6°C), the greatest total rainfall (2014=284mm, 2015=295mm, 2016=242mm) and the greatest average wind speeds (2014=2.9mph, 2015=4.4mph, 2016=4.1mph) (Plymouth Live Weather Station, 2018).

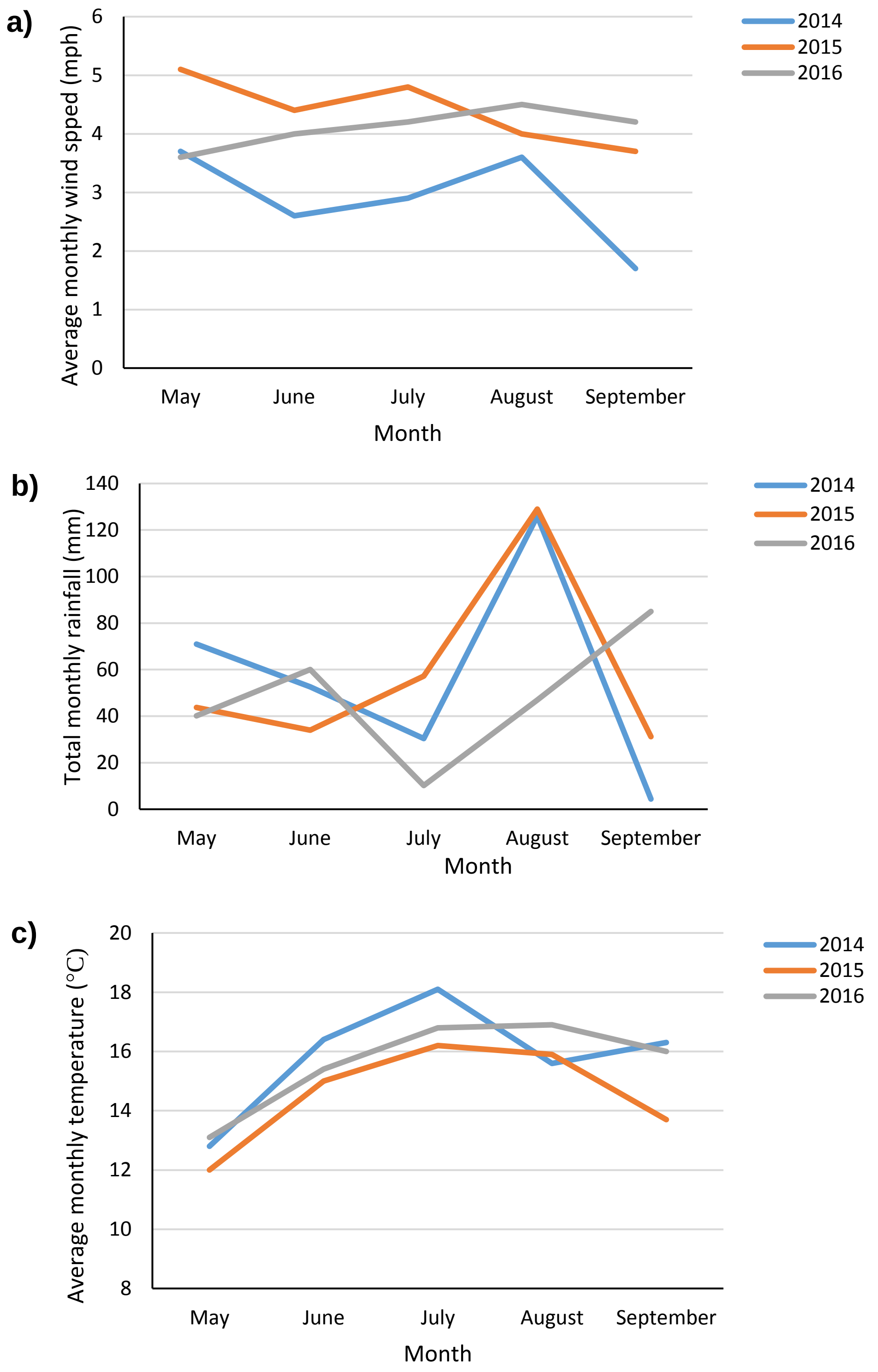


Figure 2.7
Summary statistics for the monitoring season from 2014-2016, displaying **a)** average wind speed, **b)** total rainfall, **c)** average temperature by month (Plymouth Live Weather Station, 2018).

2.5 Discussion

At Drakelands open-pit mine in southwest England the impacts of habitat loss, disturbance and the destruction of roosts within trees and retained buildings may have resulted in recent shifts in the activity of local bat populations. In order to minimise these impacts mitigation has been implemented on-site in the form of habitat creation, bat boxes, bat houses, sensitive lighting strategies and retained building roosts (Gillingham, 2014). Here, we assess the short-term effectiveness of these mitigation measures by analysing data collected by contracted ecologists over a three year period. Analysis of the remote detector monitoring results revealed potential short-term changes in the local abundance of some species; the possible causes of these shifts will be discussed in the following section.

Species richness at the site was found to be higher at woodland monitoring locations than grassland monitoring locations. During the three year monitoring period, woodland areas continued to be cleared to allow for expansion of the mine waste facility and diverted link road. Therefore, it is surprising that species richness did not significantly decline over the three year period and that monitoring year was not a significant predictor of species richness. The tree planting which commenced in 2013 may have provided some additional foraging resources, helping to maintain species richness; to date approximately 38,000 trees have been planted across the site (SLR Consulting Ltd, 2013). As these juvenile woodland areas mature, the foraging and commuting opportunities they provide to local bat populations should be enhanced (Garland and Markham, 2007). Although roosting opportunities within these young woodlands will be limited until microhabitat features such as deadwood and tree hollows develop, roost opportunities are supplemented by the 81 bat boxes which were installed prior to 2014 within retained woodlands, supporting species richness at the site (Vesk, 2008; Burgar *et al.* 2015; Watrous *et al*. 2006; Gillingham, 2014). The bat boxes have been adopted by at least six species, with the rate of adoption rising steadily over the three year monitoring period (See Appendix B: figure 1 for maps displaying the position of adoption of bat boxes by species) (Gillingham, 2016).

In contrast to species richness, monitoring year was found to be a significant predictor of overall bat activity at the site. Total activity remained similar from 2014 to

2015, but then decreased significantly in 2016. Total activity at the site was heavily influenced by *P. pipistrellus* and *P. pygmaeus* passes, which respectively accounted for 79.8 and 15.0 percent of the total number of passes recorded. The number of *P. pipistrellus* individuals observed emerging from the retained roost buildings was not found to differ significantly between monitoring years, therefore the shifts in activity may have been due to prey availability. A substantial component of the diet of both *P. pipistrellus* and *P. pygmaeus* consists of Chironomidae and other families within the order Diptera, many of which have an aquatic larval stage (Vaughan, 1997; Bartonička *et al.* 2008). Therefore, the distribution of *P. pipistrellus* and *P. pygmaeus* and the duration of foraging is likely to be influenced by the availability of water bodies at the site, which are required by Dipteran prey for the early life stages (Kusch *et al*. 2004; Jong and Ahlén, 1991; Altringham, 1996).

The aquatic larval stages of Diptera are able to colonise new water bodies quickly, occurring in both pristine and extremely degraded ecosystems and can progress from egg to flying adult in as little as four days (Sundermann *et al*. 2007; AMCA, 2014). During the summer of 2015 significant new waterbodies were created at the Drakelands site in the form of stream and pool systems, storage ponds and diversion channels (SLR Consulting Ltd, 2013). The rapid colonisation of these habitats by Diptera species would have provided additional foraging resources at the site, which might explain the observed increase in total bat activity in 2015 relative to 2014. However, the subsequent colonisation of the new waterbodies by predators of aquatic Diptera larvae such as Corixidae and *Utricularia* species, Anura tadpoles, Odonata larvae, newts and fish may have resulted in fewer emergent adults the following year (Reilly and McCarthy, 1990; Martens and Grabow, 2011; Klecka and Boukal, 2012; Fasola and Canova, 1992). This reduction in foraging resources may partially explain the observed reduction in *P. pipistrellus* activity and therefore total bat activity in 2016 relative to 2015.

Despite the potential increase in foraging resources linked to the creation of new water bodies, *P. pygmaeus* activity at the site was found to be significantly lower in 2016 than in the previous two years of monitoring. Both *P. pygmaeus* and *P. pipistrellus* activity was found to be higher at woodland than grassland monitoring locations, therefore the reduction in optimum foraging and commuting habitat through the continued clearance of woodland areas could have had led to reductions in the activity levels of both species over the monitoring period. The impacts of woodland clearance may be more significant for *P. pygmaeus*, as it is less mobile and occupies a smaller home range than *P. pipistrellus* (Fuentes-Montemayor *et al.* 2013). In subsequent years *P. pygmaeus* and *P. pipistrellus* activity may begin to increase as mitigation measures such as the woodland planting, bat boxes and habitat corridors become better established (Jenkins *et al*. 1998).

In contrast, only habitat type was found to be a significant predictor of *Myotis* activity, which was higher at woodland than grassland monitoring locations. However, as echolocation calls were only identified to the genus level it is difficult to assess how *Myotis* species are reacting to the site-wide changes, as species-level trends may be present but impossible to detect (Voigt and Kingston, 2016). For example, research suggests that the 'passive listening' *M. bechsteinii* may be particularly vulnerable to the impacts of broadband noise such as that produced by the mine's processing plant, but without being able to identify echolocation calls to a species level it is impossible to monitor potential shifts in activity (Schaub *et al.* 2008*;* Clarke, pers.comm, 2017). Although *Myotis* forage using a range of feeding strategies across varying habitat types, all species tend to favour deciduous woodland habitats close to water (Russ, 2012). While the woodland clearance may have had negative impacts on *Myotis* activity, this could have been offset by the creation of new waterbodies resulting in increased foraging resources, in particular for *M. daubentonii, M. nattereri* and *M*. *brandti* which frequently consume insect taxa which are associated with water (Vaughan, 1997; Shiel *et al.* 1991; Flavin *et al*. 2001). Following the creation of the Tory pond (the most substantial new waterbody) in 2015, the colonisation of two nearby bat boxes in 2016 by a breeding colony of 17-25 *M. nattereri* and a single hibernating *M. daubentonii* suggests that *Myotis* species are indeed utilizing this enhanced foraging habitat (Gillingham, 2016)*.* Further, the adoption of these bat boxes is a positive sign that the Drakeland's lighting strategy is not deterring these light-sensitive species from foraging and roosting within the mine site (Gillingham, 2014; Stone *et al.* 2012).

Unlike *Myotis* species, *N. noctula* and *N.leisleri* are not considered light-shy and may utilize artificial lighting as a feeding resource by preying on insect congregations (Mathews *et al.* 2015). *N. noctula* and *N.leisleri* have also been identified foraging along major roads and using road brides as roost sites, suggesting they may be relatively more tolerant of anthropogenic impacts such as noise (Waters *et al.* 1999; Ceľuch *et al.* 2008). The probability of detecting *N. noctula* during a sampling session was found to be significantly greater in 2016 than 2015, while *N. leisleri* was detected on 15%, 0% and 7% of sampling sessions from 2014-2016 respectively. Monitoring years in which detection rates were low may reflect reduced on-site prey availability or habitat loss, potentially due to woodland clearance and earthworks around the Mine Waste Facility resulting in loss of grassland habitat and woodland disturbance and therefore fewer Lepidoptera and Coleoptera prey items (Vaughan, 1997; Dodd *et al.* 2012). In response, *N. noctula* and *N. leisleri* may have shifted their foraging patterns to more productive areas in the surrounding landscape, as both are capable of commuting long distances; frequently foraging at distances of more than 10km and 13km from the roost respectively (JNCC, 2007; Waters *et al.* 1999). The subsequent increase in detection rates in 2016 relative to 2015 may reflect an increase in local prey availability, which may be due to the establishment of plants in the previously moved earth, leading to a rise in Lepidoptera and Coleoptera prey items in 2016.

The probability of detecting *Plecotus* species decreased over the three year monitoring period, with significantly lower detection rates in 2015 than 2014 and no passes identified in 2016. This drop in activity is reflected in the retained roost monitoring, in which the maximum number of emerging *P. auritus* fell from 13 to five to one individual over the three year monitoring period. Due to the overall low numbers of emerging individuals this trend was found to be non-significant, however it will be interesting to see whether the number of emergent *P. auritus* remains low in subsequent years. The identified reduction in *Plecotus* activity may be in response to the continued loss of woodland and hedgerows across the site resulting in reduced foraging and commuting opportunities (JNCC, 2007). Further, the ongoing loss and disturbance of woodland and grassland habitats in the development of the link-road and Mine Waste Facility may have impacted the local abundance of Lepidoptera species, which forms the majority of diet for both *P. auritus* and *P. austriacus* (Dodd *et al.* 2012; Vaughan, 1997). In addition to the impacts of habitat loss, anthropogenic noise from the processing plant may be degrading the foraging efficiency of the passive listening *P. auritus,* by masking insect generated sounds in the surrounding area (Razgour *et al*. 2013; Anderson and Racey, 1993; Schaub *et al.* 2008; Siemers and Schaub, 2010; Eklof and Jones, 2003). The processing plant at Drakelands operates for 24 hours per day and commenced production in July 2015, coinciding with the fall in *Plecotus* detection rates (McGlinchey, pers.comm, 2017). As only three years of monitoring data are available this downward trend may be short-lived; *Plectous* activity may begin to rise as the juvenile woodland areas and hedgerow planting matures, providing additional foraging and commuting opportunities on-site (Garland and Markham, 2007).

In contrast, year of monitoring was not a significant predictor of the likelihood of detecting *R. ferrumequinum*. It would be interesting to assess whether the detection rate of *R. ferrumequinum* at the site has changed relative to levels recorded in 2013 - prior to the removal of a hibernation and potential transitional building roost for seven *R. ferrumequinum* under a EPSL and the development of the mine waste facility (Gillingham, pers.comm, 2017). Unfortunately, differences in the sampling methodology mean that this is not possible. In order to mitigate for the loss of this roost a replacement roost provision was created, which has been adopted by both *R. ferrumequinum* and *R. hipposideros* since 2015 (Gillingham, 2016).

Unlike *R. ferrumequinum*, *R. hipposideros* has been identified using one of the retained roost buildings, with the number of emerging individuals increasing from zero to two to four over the three year monitoring period. The proportion of sessions on which *R. hipposideros* was detected peaked in 2015, with levels in 2014 and 2016 remaining similar. This peak coincides with the creation of new waterbodies at the site, which could be particularly beneficial for this species given that they may spend half of their active flight time foraging within 600 metres of the roost site and that aquatic-linked Diptera form a substantial part of their diet (JNCC, 2007; Vaughan, 1997; Mitschunas and Wagner, 2015). Although *R. ferrumequinum* may take some Diptera prey, the majority of their diet is composed of Lepidoptera and

Coleoptera (Flanders and Jones, 2009; Vaughan, 1997). The local abundance of Lepidoptera and Coleoptera may have been impacted by the ongoing loss and disturbance of grassland habitats due to the earthworks around the Mine Waste Facility. While the sensitive lighting strategy, new roost sites and maturing woodland planting may help to maintain local *R. ferrumequinum* and *R. hipposideros* populations, their effective conservation of *R. ferrumequinum* in particular will depend on the sensitive management of farmland in the 4km surrounding the roost sites (Froidevaux *et al*. 2017; Flanders and Jones, 2009; JNCC, 2007).

A nationally rare species, *B. barbastellus* was only detected once during the 2014 remote detector monitoring, 11 times during 2015 and six times during 2016 (Harris *et al*. 1995). The continued clearance of vegetation in areas surrounding the Mine Waste Facility is likely to have negative impacts on the local abundance of Lepidoptera which accounts for a substantial proportion of *B. barbastellus* diet (Andreas *et al.* 2012). However, the new waterbodies will have resulted in a local increase in aquatic larval-stage Diptera*,* of which Tipulidae and Nematocera are commonly taken as prey (Rydell *et al*. 1996). As *B. barbastellus* is a crevice dwelling species, the preservation of the remaining broadleaf woodland at the site is particular import for retaining roosting opportunities (Greenaway and Hill, 2005). However, a breeding colony of *B. barbastellus* has been identified in the nearby Dendle's Wood nature reserve in Dartmoor. Given that the reserve is less than 5km from the Drakelands site and the average foraging range of *B. barbastellus* is 6.8km, it is possible that the individuals detected on site were commuting from this location and only use the Drakelands area for foraging or commuting purposes (Zeale *et al.* 2012; Dartmoor National Park Authority, 2009). Finally, local activity levels may be higher than they appear; *B. barbastellus* emits echolocation calls 10-100 times lower in amplitude relative to other aerial hawking bats in order to maximise the success rate of catching tympanate moths, which also reduces their detectability on acoustic equipment (Goerlitz *et al.* 2010).

Similarly to *B. barbastellus, E. serotinus* will very rarely adopt bat boxes. However, as this species favours building roosts located in areas with a high proportion of improved grassland and arable land, *E. serotinus* may adopt the retained roost buildings in subsequent years (Tink *et al*. 2014; Bat Conservation Trust, 2010). The proportion of sessions on which *E. serotinus* was detected peaked in 2015, with levels in 2016 remaining slightly higher than those in 2014. This peak coincides with the creation of new waterbodies at the site and may be a result of increased availability of aquatic-linked *Diptera*, which are commonly taken as prey by *E. serotinus* during the spring months (Zukal and Gajdosik, 2012). The remainder of the diet consists of Coleoptera (particularly chafers and *Aphodius* beetles) and Lepidoptera (Kervyn and Libois, 2008). While the availability of these prey items may have been affected by the loss of grassland habitats around the mine waste facility, their abundance will also be affected by the local agricultural land management practices and livestock in the surrounding area (Catto *et al*. 1996).

P. nathusii has only been detected on three sampling sessions; once in May 2015, May 2016 and September 2016. These timings coincide with peak recordings elsewhere in England, in keeping with the probable migration period of a journey from Britain to continental Europe (Hargreaves *et al*. 2015). All of the monitoring points at which *P. nathusii* was recorded lie on the western boundary of the site; a trend which may be confirmed with data from subsequent monitoring years.

Although the incorporation of weather parameters into statistical models was not possible due to the lack of nightly activity data, summary statistics indicate that the activity trends identified may not be a result of variation in environmental conditions among monitoring seasons. The monitoring season of 2015 received the lowest annual temperatures, greatest total rainfall and greatest average wind speeds – factors which generally correlate with lower insect densities and therefore reduced bat activity (Cryan *et al.* 2014; Horn *et al.* 2008; Barros *et al.* 2014; Taylor, 1963). Despite this, total bat activity and *P. pipistrellus* activity was significantly greater in 2015 relative to 2014, and *P. pygmaeus* activity was significantly greater in 2014 and 2015 than 2016. Subsequent monitoring of the site should collect nightly activity data, in order to enable a more reliable understanding into the potential influence of environmental conditions on bat activity.

While daylight blasting may have some impact on bats roosting in proximity to the mine site, the rate of blasting has remained consistent throughout the three year monitoring period, so is not likely to account for any variation in bat activity. Laboratory studies suggest that bats in torpor are particularly sensitive to anthropogenic noise in the period before dusk, and responded by elevating their skin temperature (in cases where torpid temperature was 10°C lower than their active skin temperature) (Luo et al. 2014). Blasting at the Drakelands site occurs at 15:00, therefore the potential impacts of blasting may be elevated during the winter months when the air temperature is lower and dusk occurs earlier. Despite this, case studies have noted annual increases in cave-roosting bat populations in proximity to blast locations, suggesting that that torpid bats rapidly habituate to regular and prolonged noise exposure and seismic vibration (West Virginia Department of Environmental Protection, 2006; Player and Keim, 2015; Luo *et al.* 2014).

2.6 Conclusion

In order to conserve local bat populations throughout large-scale development, it is crucial that the impacts of development and effectiveness of mitigation measures are evaluated using appropriate monitoring techniques. By identifying short-term trends in bat activity at a new open-pit mine, we discuss potential species-specific responses to mitigation, habitat loss and disturbance on a local scale. While some species appear to be befitting from the implemented mitigation measures, *Plecotus* species in particular have experienced sharp declines in activity which may be linked to their foraging ecology and the impacts of anthropogenic noise. Although the potential impacts of anthropogenic noise on some passive listening bat species has been demonstrated, we could not identify research relating specifically to *P. auritus* which may represent an area for future research (Radford *et al.* 2012; Razgour *et al*. 2013; Anderson and Racey, 1993; Schaub *et al*. 2008; Siemers and Schaub, 2010). As well as informing management decision making at this site, this study may provide insight to other projects involving large scale habitat loss. However, as the data were collected over a short time frame and don’t allow a comparison with bat activity pre-mining they should be treated with caution. Where possible, future mining developments should consider standardising data collection methods for monitoring conducted pre and post development, in order to allow comparison of subsequent bat activity to baseline levels. Finally, this study was limited by the inability to include weather data into statistical models due to the lack of nightly activity totals, as these factors may have had a significant influence on bat activity.

Chapter 3: Investigating the impacts of open pit mining on bats in the wider landscape

3.1 Abstract:

Areas used in open-pit mining often undergo sudden and extensive landscape changes, including habitat fragmentation and disturbance. With mining activity predicted to expand, understanding the impacts of habitat loss on wildlife is fundamental to developing effective mitigation at new developments. Despite this, research into the impacts of open-pit mines on bat populations, a significant component of global biota, is lacking. In order to promote informed management strategies, we investigated potential impacts of a newly established open-pit mine in southwest Britain on bat activity in the surrounding landscape, using remote monitoring techniques. Distance from the site boundary was found to be an important predictor of total bat activity and species richness, with effects varying with habitat type. Within woodland both total bat activity and species richness increased with distance from the site boundary with the effect potentially extending to 400-500 metres, whereas within grassland habitats there was no clear trend. The loss of foraging habitat within the site boundary may reduce the movement of bats in the direction of the mine, leading to habitat fragmentation and barrier effects. In line with other studies, our results suggest that bat species that are adapted to forage within clutter may be more susceptible to barrier effects than species which are adapted to forage within open environments. Further, this study highlights a range of factors which might influence bat activity. This highlights the need for further research into the species-specific responses to habitat fragmentation in the wider landscape.

3.2 Introduction

During the past century increasing anthropogenic pressures have greatly altered the habitat composition of the majority of landscapes, with land use change currently posing the greatest threat to global biodiversity (Fischer and Lindenmayer, 2007; Pekin and Pijanowski, 2012). Mining often causes sudden and extensive land use change, particularly through the process of open-pit mining, which requires the removal of vegetation and overburden (the natural soil and rock that sits above the ore body) prior to accessing the ore body (Sonter *et al*. 2014; ELAW, 2010). Open-pit mines generally operate over a large spatial-scale, with the area of landscape degradation frequently 2-11 times greater than that of underground mining projects (Miao and Marrs, 2000; Kuter, 2013).

At an active mine site local habitats will be directly impacted by the removal of native vegetation and top-soil to form the pit and associated mine waste facility, with the area of degradation potentially extending over several dozen km^2 depending on the mineral being excavated (Edwards *et al.* 2013). Establishing a mine also promotes indirect effects at the landscape level, such as urbanisation and the building of access roads and transport links which may lead to habitat fragmentation and facilitate the spread of invasive species (Castro Pena *et al.* 2017; Calinger *et al.* 2015). Mining can have particularly devastating effects on aquatic ecosystems as a result of leaked mine contaminants, sedimentation and alteration of watercourses (Durand, 2012; Holden and Jacobson, 2012). Mining may also impact the surrounding landscape through acoustic disturbance effects caused by the blasting, transportation and processing of material (Donoghue, 2004; Manwar *et al.* 2016). For acoustic predators such as bats these impacts may result in compromised foraging efficiency, leading to reduced activity levels in the areas most heavily affected by noise (Senzaki *et al*. 2016).

Relatively little is known about the effects of anthropogenic noise on bats, and research into the thresholds and characteristics of sound and vibration which influence the daily and seasonal movement of bats is lacking (Player and Keim, 2015). Studies suggest that species which echolocate at low frequencies (<35kHz) or use passive listening to locate prey may be at greater risk from the impacts of anthropogenic noise (Schaub *et al.* 2008; Bunkley *et al*. 2015; Berthinussen and Altringham, 2012a). There is very little published information on the effects of mine blasting on bats, and the research to date has primarily focused on the effects of vibrations generated by mine blasting on bat hibernacula (Pritchard *et al*. 2012; Player and Keim, 2015). Habitat loss and degradation may also have detrimental impacts on local bat populations if roost sites, important foraging areas and strategic flyways are lost (Hundt, 2012). Further, bats are particularly susceptible to the effects of bioaccumulation of toxins such as heavy metals, pyrites and clays associated with mining, and the impacts of these on bat health and mortality are well documented (Wilkinson and South, 2002; Voight and Kingston, 2016). Due to their long life span and low fecundity, any impacts of mining which increase bat mortality or reduce reproductive success may result in severe population declines and slow rates of recovery (Voight and Kingston, 2016).

As a result of bat population declines over the past century, in many countries it is now mandatory to consider the potential impacts of new mining developments on local bat populations throughout the planning and Ecological Impact Assessment process (Berthinussen and Altringham, 2012a; CIEEM, 2016). In addition to identifying the impacts in the immediate vicinity of developments, it is crucial to evaluate the potential impacts of new open-pit mines on bat populations in the wider landscape, in order to enable informed management recommendations (Brown and Berry, 1997; Buehler and Percy, 2012; Gorresen and Willig, 2004). Despite this, research investigating the impacts of land use change and disturbance on local bat populations in the wider landscape as a result of open-pit mines is lacking (Voight and Kingston, 2016; Ducummon, 2000).

Studying the spatial distribution of bat populations in relation to landscape level disturbance from noise, vibration and habitat loss is therefore an important step in developing effective programs for management and conservation in areas of open pit mining (Russel *et al*. 2014; Senzaki *et al*. 2016). Using remote monitoring techniques we assess the potential impacts of a new open pit mine on local bat populations in southwest Britain, by measuring activity and species richness in relation to distance from the site boundary. With areas of the mining sector predicted to expand in the coming years, this information will be valuable in developing informed management programs, which include mitigating the impacts of open pit mining on local bat populations in the wider landscape (UK Minerals Forum, 2014).

3.3 Methods

3.31 Sampling location:

The study site Drakelands Mine is located near the village of Hemerdon in southwest Britain, in close proximity to China clay mines operated by Imerys Minerals and

Sibelco UK Ltd and borders Dartmoor National Park. Established in 2014, the open-pit tungsten and tin mine has a three million tonne per year operation scheduled for the duration of the mine-life (Wolf Minerals Ltd, 2016). Baseline surveys conducted in 2009 revealed that the planning permission area predominantly consisted of grassland, with patches of broadleaf woodland, mire and heathland (Michel Hughes Associates, 2009). Bat surveys of the planning permission area conducted in 2009 regarded species composition at the site as typical for rural Devon, with a total of 11 species detected (Michel Hughes Associates, 2009a). Development of the site required the destruction of three roosts used by low numbers of *Pipistrellus pipistrellus, Pipistrellus pygmaeus, Rhinolophus ferrumequinum* and *Plecotus auritus*, carried out under European Protected Species Licences (Gillingham 2013a; Gillingham 2013b; Gillingham 2013c). In order to compensate for the roost loss, extensive bat mitigation measures have been implemented across the site including habitat creation, the strengthening of commuting routes by limiting hedge trimming and installation of bat boxes and buildings (Gillingham, 2014; Michel Hughes Associates, 2013). The majority of these measures are contained within the area of the planning boundary, and it is hoped that they will partially offset the habitat loss within the mine site and lessen the impacts of the mining development on bat activity and species richness in the surrounding landscape.

The majority of land within the planning permission boundary has been converted to form the pit and the MWF with the remainder designated to the processing plant, offices, link-roads, storage ponds and wildlife compensation (Michel Hughes Associates, 2013a). Light pollution across the site is strictly controlled, with levels along boundary vegetation consistently below 4 lux (Gillingham, 2014). The processing plant operates for 24 hours per day and is therefore the primary source of noise pollution during the bat active period. Noise emitted from the processing plant largely falls within the low frequency range of 16-140 Hz and the high frequency range of 12-40 kHz (Clarke, pers.comm, 2017). The latter falls within the auditory range of European bat species, which are thought to be most sensitive to frequencies above 10 kHz (Luo *et al.* 2014). Blasting is another significant source of noise and vibration, and occurs several times per week. Although it is conducted during daylight hours, this disturbance could potentially have a subsequent effect on the spatial distribution and activity levels of local bats. Therefore, this study will also investigate whether blasting has an effect on species richness and bat activity the following night.

3.32 Bat activity and species richness:

A series of 17 line transects was designed in order to evenly sample the landscape surrounding the Drakelands site (Figure 3.1). Each transect consisted of 5 sampling locations, positioned at approximately 20, 270, 520, 770 and 1020 metres from the planning permission boundary of the mine. A maximum distance of 1020 metres was selected in order to minimise the potential confounding impacts of the nearby town of Plympton on bat activity. Also, the landscape to the north of the Drakelands site was not sampled, in order to avoid confounding impacts of the adjacent Lee Moor China clay pits. Bat activity at each sampling location was monitored using stationary acoustic detectors, which recorded from 30 minutes before sunset until sunrise for a duration of five consecutive nights. Within each transect all five points were sampled simultaneously in order to avoid bias due to differing weather conditions between locations. Transects were completed in a random order, with four transects sampled from 31st September - 31st October 2016 and the remaining 13 transects sampled from 9th March - 19th May 2017. This sampling period was selected to identify year-round resident species as well as potential seasonal migrants (Hargreaves *et al.* 2015).

The exact location of each sampling point was determined upon deployment with effort made to position detectors close to tree lines, linear features, water sources and habitat boundaries where the activity of foraging and commuting bats may be more concentrated (Russ, 2012; Altringham, 1996). The final placement and elevation (m) of each acoustic detector was recorded with a Garmin eTrex 10 GPS (Garmin Ltd, Southampton, United Kingdom) and the actual distance to the planning permission boundary and was calculated for each location using Google Earth 2016. Due to difficulty obtaining permission from a small number of landowners, the placement of some detectors had to be adjusted to accommodate this. Therefore the maximum distance sampled was 1434 meters from the site boundary. The total precipitation (mm), average temperature (°C) and average wind speed (mph) were calculated for each night of sampling using the weather forecasting programme Wind Guru (which uses the Global Forecast System), as these factors are known to influence levels of bat activity (Voigt *et al.* 2011; Cryan *et al.* 2014; Barros *et al.* 2014). Average nightly humidity levels (%) were also calculated, as greater atmospheric absorption of echolocation calls at high humidity levels is thought to result in lower detection rates (Broken-Brow and Corben, 2015).

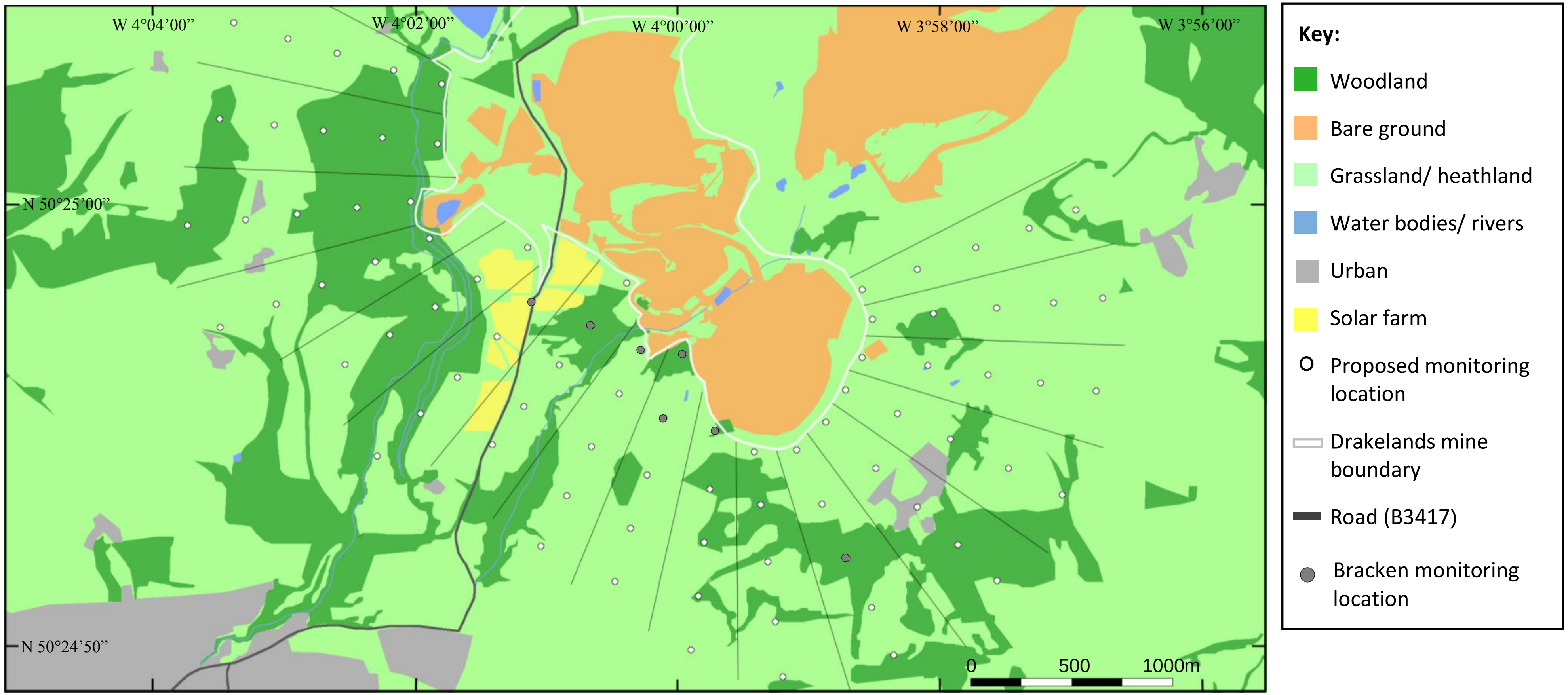


Figure 3.1

Aerial map detailing the 17 proposed monitoring transects, with recording locations positioned at 20, 270, 520, 770, and 1020 metres from the Drakelands site boundary.

The habitat type of each sampling location was recorded and classified into three categories; woodland, grassland and bracken. No other habitat types were identified, however monitoring points located within the solar farm were classed as grassland habitats (Figure 3.2). Although bat activity and species richness has been shown to vary among coniferous, riparian and broadleaf habitats the woodland areas sampled largely consisted of broadleaf, occasionally interspersed with coniferous or riparian habitat (Kirkpatrick *et al.* 2017). Therefore, for the purpose of analysis all woodland habitat types were grouped into one category. Effort was made to place detectors within one of these three habitat types, however occasionally sampling locations were bordered by multiple habitat types, and in these cases the classification of the predominant habitat type was selected (i.e. a small group of trees (15%) surrounded by grassland (85%) would be classified as grassland). Of the locations sampled 9.9% were classed as habitat type bracken, 33.8% as woodland and the remaining 56.3% sites as grassland.

Figure 3.2
Habitat classifications for sampling points: **a)** woodland **b)** grassland **c)** bracken. Where a sampling location was bordered by multiple habitat types the classification of the predominant habitat type was selected. Some grassland sites contained livestock, but none were used for arable purposes. Areas of juvenile tree planting were classed as woodland when the height of the trees exceeded 2m.

Acoustic monitoring was conducted using Anabat ultrasonic detectors (models SD1 and SD2, Titley Scientific, Lancashire, United Kingdom) with directional microphones and internal ZCAIM storage units. Detectors were placed in weatherproof boxes and elevated to 1m above ground level on a tripod, tilted upwards at an angle of approximately 45°. In order to account for potential differences in sensitivity between the detectors a randomised schedule was determined, so that each of the transect distance classes were measured equally by each of the five detectors. Due to environmental factors and equipment failure no data was obtained for 14.1% of the sampling locations. Livestock interfering with the equipment appeared to be the leading cause of data loss, followed by excessive wind speeds; both of which commonly resulted in the tripod falling over. Data obtained on nights preceding this event was included in the analysis, but where data loss occurred part way through a recording session data from the night itself was not included in the analysis.

Data were visualised in AnalookW (Titley Scientific, Lancashire, United Kingdom). Echolocation calls were visually inspected and assigned to a species or genus based on comparison to a reference library of known species and the echolocation parameters provided in the literature by Russ (2012). Echolocation calls for which confident identification was not possible were labelled ‘unknown’ (due to echolocation calls being too faint or in the case that only a short segment of a call was detected)blas and occasionally because the parameters of a call fell between those of two species. Echolocation calls assigned to *Myotis* and *Plecotus* were only identified to a genus level, due to similarities in call parameters (Kunz and Parsons, 2009; Russ, 2012). All echolocation calls of the same species identified within a one minute period were classed as one ‘pass’. Species richness was calculated for each night of monitoring, consisting of the sum of the number of species detected (or genus for *Myotis* and *Plecotus*) and excluding those labelled ‘unknown’*.* Total activity was calculated for each night of monitoring as the sum of the number of passes of all species, including those labelled ‘unknown’ and was used as an index of bat activity at each sampling location (Ober and Hayes, 2008). While bat activity cannot confer abundance, the number of bat passes detected within a unit of time can be a useful measure for comparing the relative functional importance of areas within a study site.

3.33 Analysis:

Statistical analyses were conducted using R version 3.3.1 (R Core Team, 2016). All response variables were assessed for normality, homogeneity of variance and over dispersion using standard diagnostic procedures. Assumptions were checked and appropriate transformations were made to reduce residual variance where necessary (Grueber *et al.* 2011). Further, prior to all analyses the potential for spatial autocorrelation of species richness and bat activity among sampling locations was assessed using Moran's I (Paradis, Claude and Strimmer, 2004). Spatial autocorrelation was found to be non-significant in all analyses ($p<0.05$), with the exception of species richness (Moran's I: 0.050; $p<0.001$). Therefore, the variable of 'Transect ID' was included as a random effect in all species richness models, to account for spatial autocorrelation.

Generalised Linear Mixed Models (GLzM) were constructed using the package "lme4" to investigate whether distance from the site boundary had an effect on bat species richness and total activity (total number of passes) (Bates *et al*. 2015). Species richness models were fit using GLzMs by Laplace Approximation with a Poisson error distribution, BOBYQA optimization and a "log" link function. Total activity models were fit using GLMs by restricted maximum likelihood (REML), with a Normal error distribution. The response variable of total activity was transformed to ln (passes + 1) to reduce heterogeneity resulting from large variation in activity levels between nights while also accounting for nights with zero passes (Berthinussen and Altringham, 2012b).

Activity was also analysed on a species/ genus level for *P. pipistrellus, P. pygmaeus, Myotis* and *R. ferrumequinum* which were identified most frequently. *P. pipistrellus, P. pygmaeus,* and *Myotis* activity was transformed to ln (passes + 1) and modelled using GLMs by restricted maximum likelihood (REML) with a Normal error distribution. *R. ferrumequinum* was detected less frequently, therefore the response variable of total number of passes was converted to binary data of presence/ absence per night and modelled using GLzMs with a Binomial error distribution, fit by maximum-likelihood with a logit-link function.

All global models contained the fixed effects of distance from the site boundary (km), habitat type, average wind speed (mph), total precipitation (mm), average temperature (°C), average humidity (%) and whether blasting was conducted during the preceding day. The global models also included an interaction between distance and habitat type. All models contained the random effects of date (to account for seasonal variation) and location (to account for pseudo replication as multiple nights of monitoring were conducted at each location within a successive five day period). The random effects of date and location also help to mitigate temporal autocorrelation, by accounting for similarities in observations that are due to proximity in the time of sampling. Detector I.D. was not included in models, as preliminary analysis revealed that there was no significant differences in performance between detectors in the range of species detected or the total number of calls detected (Appendix C: Table 1a, b). Where the global model was unable to fully converge, parameters were removed to reduce its complexity using backwards step-wise selection (Grueber *et al.* 2011). Models were validated by visual examination of residuals and q-q plots (Bolker *et al.* 2009).

Models were built using all possible combinations of predictors using the package "MuMIn" and then ranked using Akaike's Information Criterion corrected for small sample size (AIC_c), which penalizes models with many explanatory variables (Burnham and Anderson, 2010*)*. Each model was then given an Akaike weight (ΔAIC_c), based on the difference in AIC_c value for that model compared with the AIC_c value of the best fitting model. Models for which $\Delta AIC_c \leqslant 2$ were considered to have equivalent support and were validated by visual examination of residuals and q-q plots (Burnham *et al.* 2011; Bolker *et al.* 2009). According to some model selection methods (including the principles of parsimony), the model containing the fewest parameters of those with equivalent support would have been selected as the optimal model (Busemeyer *et al.* 2015; Kadane and Lazar, 2004; Forster, 2000). However, given that this is an ecological study with substantial noise, where more than one model received equivalent support the top-ranked models ($\Delta AIC_c \leqslant 2$) were averaged to create the final model (Whittingham *et al.* 2006).

Model averaging was performed by combining parameter estimates from different models in a weighted average to generate a predictive model. This was carried out using the natural average method; the parameter estimate for each fixed effect was only averaged over models which also contained that predictor, and then weighted by the total weight of these models (Grueber *et al.* 2011). Confidence intervals were calculated for all parameters included in the final averaged model. Parameters for which the confidence intervals did not include zero were considered useful predictors of bat species richness and activity levels.

Simple generalised additive models were built in order to demonstrate the potential for a non-linear relationship between distance to the site boundary and species richness/ activity within a particular habitat type. Models used a regression spline with a Gaussian family and identity link function, and were represented visually.

3.4 Results

A total of 13,726 bat passes and at least 11 species were identified. Bat species were *Pipistrellus pipistrellus, Pipistrellus pygmaeus, Pipistrellus nathusii, Rhinolophus ferrumequinum, Rhinolophus hipposideros , Barbastellus barbastellus, Nyctalus noctula, Nyctalus leisleri, Eptesicus serotinus, Myotis* species and *Plecotus* species (Table 3.1).

Table 3.1
a) Summary table for the number of bat passes per species and detection frequency
b) Summary table for the number of monitoring locations per habitat type and the number of monitoring nights conducted in each habitat.

a)

Bat species	Number of passes	% of total passes	% of nights detected
P. pipistrellus	10802	78.70	71.94
P. pygmaeus	1776	12.94	48.36
P. nathusii	16	0.12	2.39
R. ferrumequinum	102	0.74	17.61
R. hipposideros	18	0.13	4.78
B. barbastellus	42	0.31	7.16
N. noctula	109	0.79	10.15
N. leisleri	12	0.09	2.69
E. serotinus	2	0.01	0.30
Myotis species	827	6.03	45.07
Plecotus species	20	0.15	5.07
Unidentified	132	0.96	19.70

b)

Habitat type	Number of monitoring locations	Number of monitoring nights
Woodland	24	111
Bracken	7	34
Grassland	41	189
Total:	72	334

3.41 Effect of distance from the mine on bat species diversity

Species richness ranged from 0-8 species/genus identified per night, with an average value of 2.16 across all locations. Three of the models generated were considered to have equivalent support ($\Delta AIC_c \leqslant 2$) (Table 3.2a). Following model averaging, the final model contained an interaction between distance and habitat type, plus the parameters precipitation, temperature and wind (Table 3.2b).

Increasing levels of precipitation and wind correlated with lower species richness, while increasing temperature correlated with higher species richness. Species richness also appears to decrease with proximity to the site boundary within woodland and bracken habitats, and remain fairly constant regardless of distance within grassland habitats (Figure 3.3a). However, as only 9.9% of the monitoring locations were classed as bracken, this trend is easily influenced by outliers. One bracken monitoring location surrounded by woodland positioned at 668 metres from the site boundary appears to be driving this trend, with consistently higher levels of species richness detected over four nights of monitoring than in the other bracken locations. To visualise the influence of these outliers, the interaction plot was reproduced without the data from this monitoring location (Figure 3.3b). With these four influential points removed from the dataset, only two models received equivalent support ($\Delta AIC_c \leqslant 2$). Of these, one model contained distance as a fixed effect but neither contained an interaction between distance and habitat type (Appendix C: Table 2). Therefore the interaction should be interpreted with caution.

The confidence intervals for the parameters precipitation, temperature, wind and habitat did not include zero. These parameters may therefore be considered the most important predictors of bat species richness at the study site (Table 3.2b). Further, the confidence intervals for the distance: habitat interaction revealed that the trends for both woodland and grassland were significantly different to the trend of the reference category bracken.

Contrary to the interaction plots (Figure 3.3a, b), graphical visualisation of the raw data suggested that within woodland habitats the relationship between species richness and distance from the site boundary may follow an asymptotic rather than linear trend (Figure 3.4). To investigate this further, a simple generalised additive model was used to visually represent species richness within woodland habitats, based on distance from the site boundary. The relationship was found to be significant ($F_{2.11,\ 107.89} = 2.74$, $p < 0.001$) and showed that species richness appears to plateau at approximately 400m from the site boundary (Figure 3.5).

Table 3.2

a) The full set of models for species richness for which ΔAICc ⩽ 2, which were included in model averaging. Shown is the deviance (Dev), the number of model parameters (*K*), the Akaike's Information Criterion (AICc) and the difference in Akaike's Information Criterion between each model and the top-ranked model (ΔAICc) and the Akaike weight (w_i) of each model.

Model rank	Model name	Response = species richness; random effects = date, location and transect	Dev	*K*	AICc	ΔAICc	w_i
2/3	Model 1a	Distance + habitat + wind + precipitation + temperature	1068.3	8	1088.3	0.5	0.313
3/3	Model 1b	Distance * habitat + wind + precipitation + temperature	1064.4	8	1088.4	0.6	0.254
1/3	Null model	Habitat + wind + precipitation + temperature	1069.7	7	1087.8	0.0	0.433

b) Model-averaged parameter estimates: effects of each parameter on species richness in the area surrounding Drakelands mine. Shown is the model-averaged means (Estimate), associated standard error (SE) and confidence intervals (10%, 90%).

Parameter	Estimate	SE	Confidence interval (10,90)
(Intercept) †	0.05892	0.35782	-0.400, 0.518
Distance (km)	0.71359	0.80836	-0.323, 1.750
Habitat (grassland)	0.46545	0.35588	0.009, 0.922
Habitat (woodland)	0.78916	0.37125	0.313, 1.265
Distance: habitat (grassland)	-1.40338	0.71447	-2.321, -0.486
Distance: habitat (woodland)	-1.24249	0.71950	-2.166, -0.319
Precipitation	-0.56496	0.15578	-0.765, -0.365
Temperature	0.46341	0.11151	0.320, 0.607
Wind	-0.32582	0.11576	-0.474, -0.177

†Habitat (bracken) was the reference category.

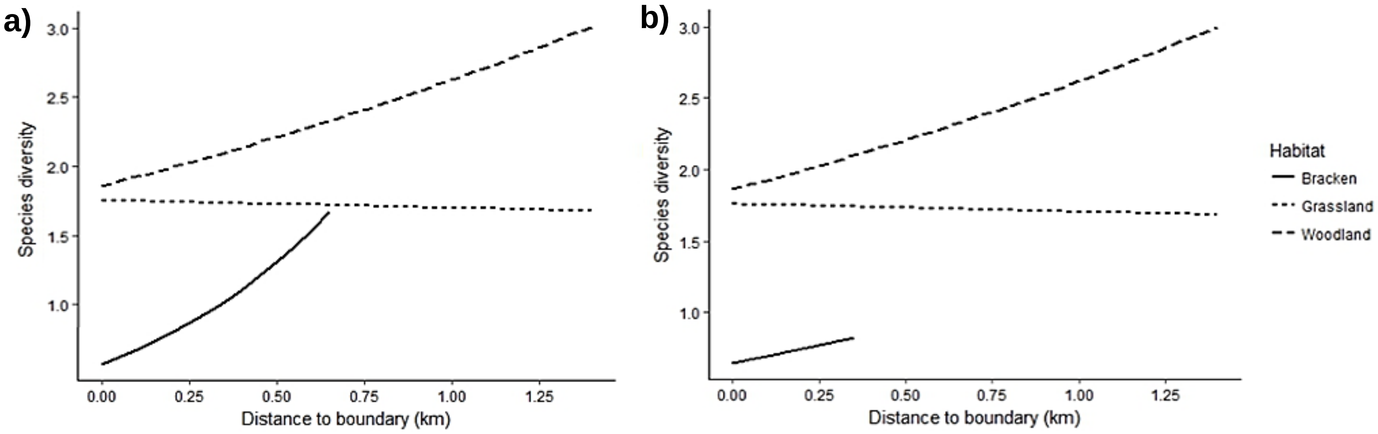


Figure 3.3
The relationships between species richness and distance from the mine in the various habitat types.
a) The predicted relationship based on model 1b (Table 3.2a).
b) The same predicted relationship, but with the influential bracken monitoring location removed from the data set.

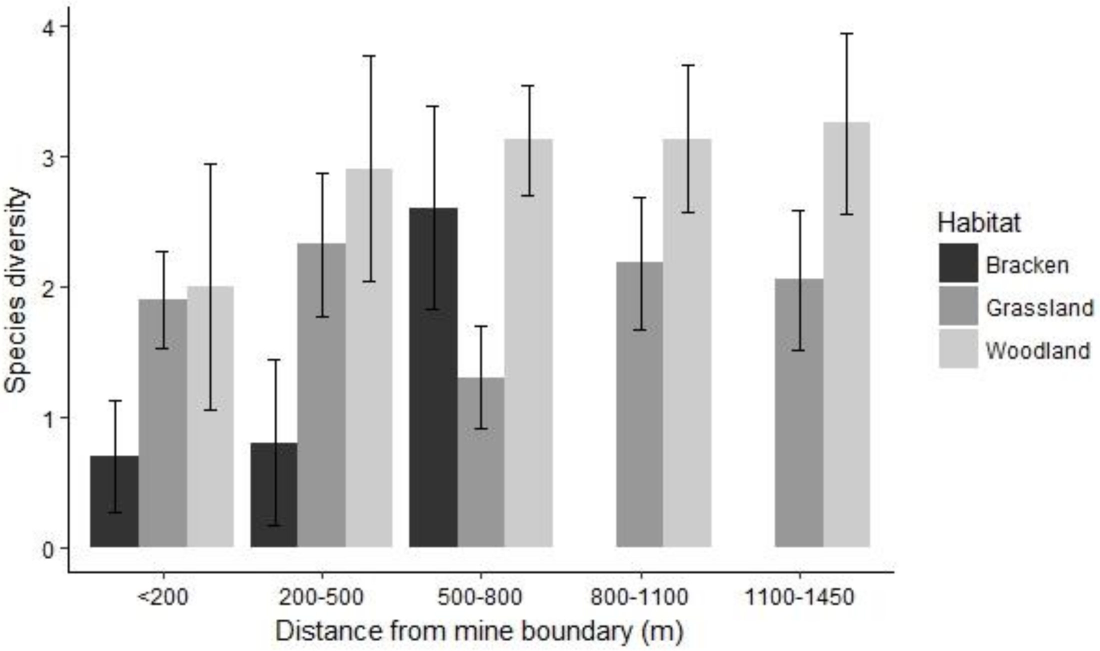


Figure 3.4
The relationship between mean species richness and distance from the site boundary within habitat type, using raw data and displaying standard error bars. While distance was used as a continuous variable in analyses, here it is displayed as a categorical variable.

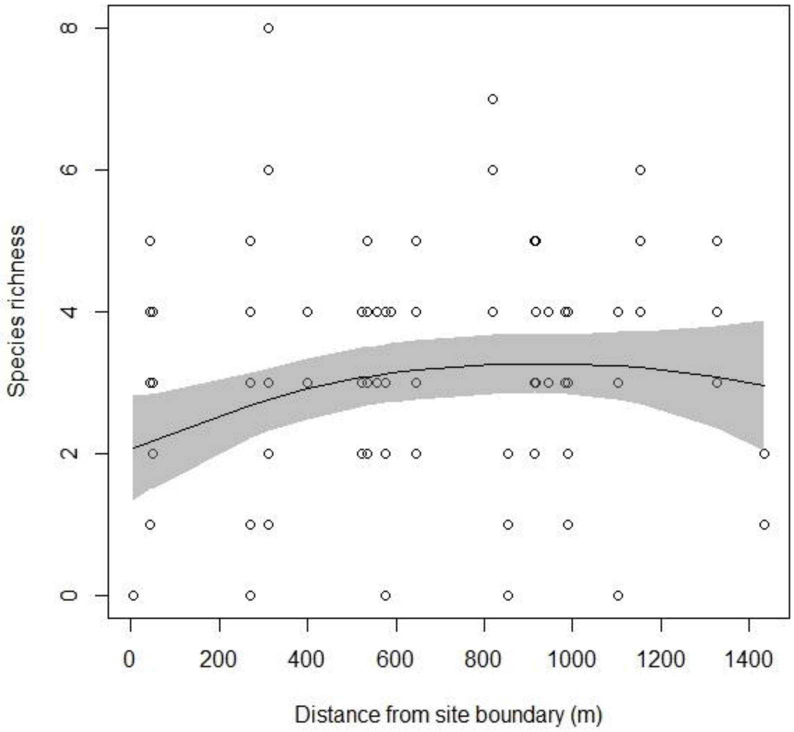


Figure 3.5
A simple generalised additive model was used to visually represent species richness within woodland habitats, based on distance from the site boundary, with 95% confidence interval shown in grey. In demonstrating the potential for a non-linear relationship between species richness and distance, the model suggested that richness may plateau at approximately 400m from the site boundary.

3.42 Effect of distance from mine on total bat activity

Total bat activity ranged from 0-163 passes identified per night, with an average of 41.3 passes identified across all locations. Five of the models generated were considered to have equivalent support ($\Delta AIC_c \leqslant 2$) (Table 3.3a). Following model averaging, the final model contained an interaction between distance and habitat type, plus the parameters precipitation, temperature, wind, humidity and blasting.

The confidence intervals indicate that the distance: habitat interaction (bracken and grassland) and the parameters precipitation, temperature and wind are the most important predictors of total bat activity at the study site (Table 3.3b). Increasing levels of precipitation, wind and humidity correlated with lower total activity, while increasing temperature correlated with higher total activity.

Although total bat activity on nights following blast activity was lower than nights on which blasting did not occur, this effect was at best very small and had little statistical support as confidence intervals overlapped zero. (Appendix C: Figure 1a).

Based on the model predicted trends, total bat activity appears to decrease with proximity to the site boundary within woodland and bracken habitats, and remain fairly constant regardless of distance within grassland habitats (Figure 3.6a). As with the species richness data, the low proportion of bracken monitoring locations mean that this trend was susceptible to outlier effects. Consistent with this, one bracken monitoring location surrounded by woodland positioned at 668 metres from the site boundary appears to be driving this trend, with consistently higher levels of total bat activity detected over four nights of monitoring than in the other bracken locations. To visualise the influence of these outliers, the interaction plot was reproduced without the data from this monitoring location (Figure 3.6b). With these four influential points removed from the dataset, seven models received equivalent support ($\Delta AIC_c \leqslant 2$) and only one of these contained an interaction between distance and habitat type (Appendix C: Table 3). Therefore the interaction effect displayed in Figure 3.6a should be interpreted with caution.

Graphical visualisation of the raw data suggested that within woodland habitats the relationship between total activity and distance from the site boundary may follow an asymptotic rather than linear trend (Figure 3.7). To investigate this further, a simple generalised additive model was used to visually represent log (total activity+1) within woodland habitats, based on distance from the site boundary. The relationship was found to be significant ($F_{2.12,\ 107.88} = 4.61$, $p < 0.01$) and showed that total activity appears to plateau at approximately 500m from the site boundary (Figure 3.8).

Table 3.3

a) The full set of models for total bat activity for which ΔAICc ⩽ 2, which were included in model averaging. Shown is the deviance (Dev), the number of model parameters (*K*), the Akaike's Information Criterion (AICc) and the difference in Akaike's Information Criterion between each model and the top-ranked model (ΔAICc) and the Akaike weight (w_i) of each model.

Model rank	**Model name**	**Response = total activity; random effects = date and location**	**Dev**	***K***	**AICc**	**ΔAICc**	**w_i**
1/5	Model 2a	Distance * habitat + wind + precipitation + temperature	1086.5	8	1111.5	0	0.359
2/5	Model 2b	Distance * habitat + wind + precipitation + temperature + humidity	1085.9	9	1112.9	1.4	0.178
3/5	Model 2c	Distance * habitat + wind + precipitation + temperature + blasting	1086.0	9	1113.0	1.5	0.170
4/5	Model 2d	Distance + habitat + wind + precipitation + temperature	1092.2	8	1113.2	1.7	0.154
5/5	Null model	Habitat + wind + precipitation + temperature	1094.9	7	1113.4	1.9	0.139

b) Model-averaged parameter estimates: effects of each parameter on total bat activity in the area surrounding Drakelands mine. Shown is the model-averaged means (Estimate), associated standard error (SE) and confidence intervals (10%, 90%).

Parameter	**Estimate**	**SE**	**Confidence interval (10,90)**
(Intercept) †	2.209	0.819	1.158, 3.260
Distance (km)	2.213	1.524	0.258, 4.168
Blasting	-0.151	0.211	-0.422, 0.121
Habitat (grassland)	0.029	0.829	-1.035, 1.093
Habitat (woodland)	0.671	0.871	-0.447, 1.789
Distance: habitat (grassland)	-2.661	1.397	-4.454, -0.867
Distance: habitat (woodland)	-1.726	1.428	-3.560, 0.108
Precipitation	-0.715	0.250	-1.036, -0.393
Temperature	0.869	0.241	0.560, 1.178
Wind	-0.735	0.231	-1.031, -0.438
Humidity	-0.202	0.242	-0.512, 0.108

†Habitat (bracken) was the reference category.

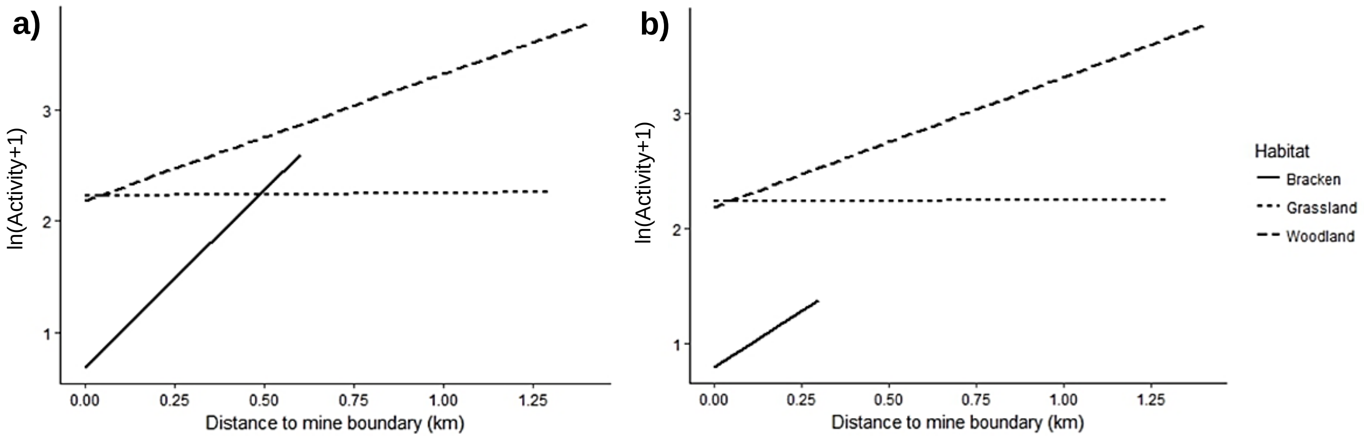
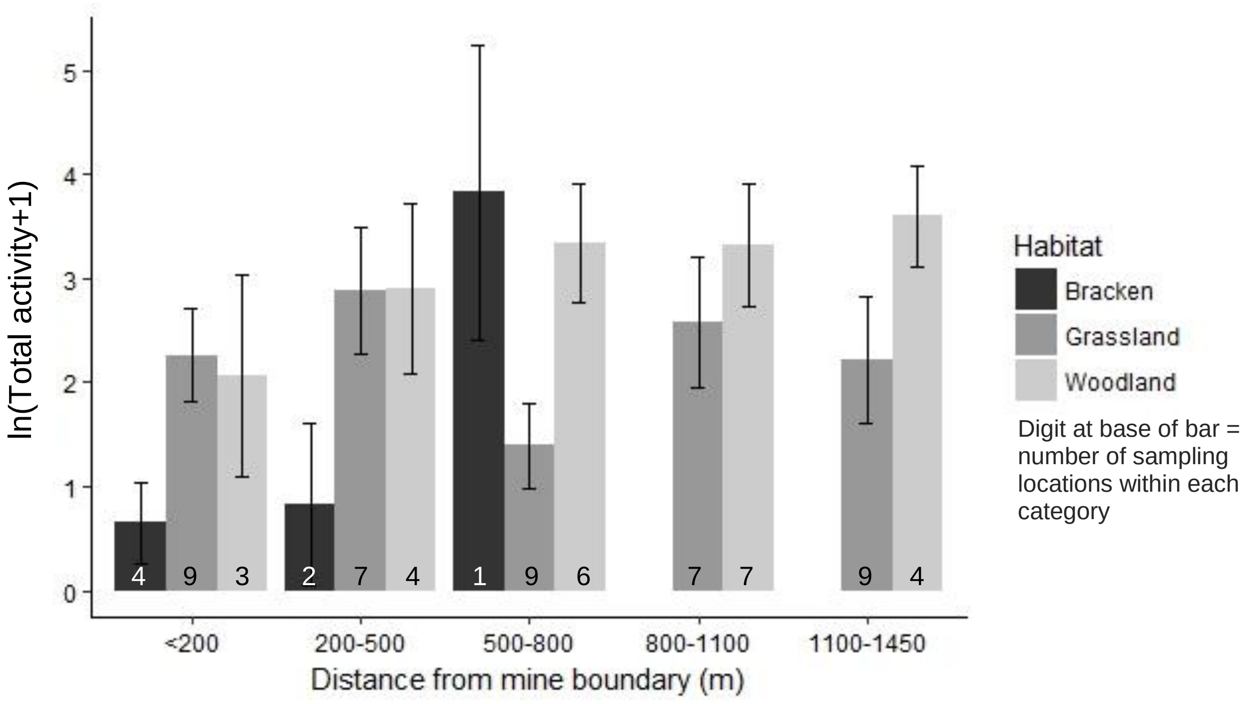


Figure 3.6
The relationship between total bat activity and distance from the mine in the various habitat types.
a) The predicted relationship based on model 2a (Table 3.3a).
b) The same predicted relationship, but with the influential bracken monitoring location removed from the data set.



Digit at base of bar = number of sampling locations within each category

Figure 3.7
The relationship between mean bat activity and distance from the site boundary within habitat type, using raw data and displaying standard error bars. While distance was used as a continuous variable in analyses, here it is displayed as a categorical variable.

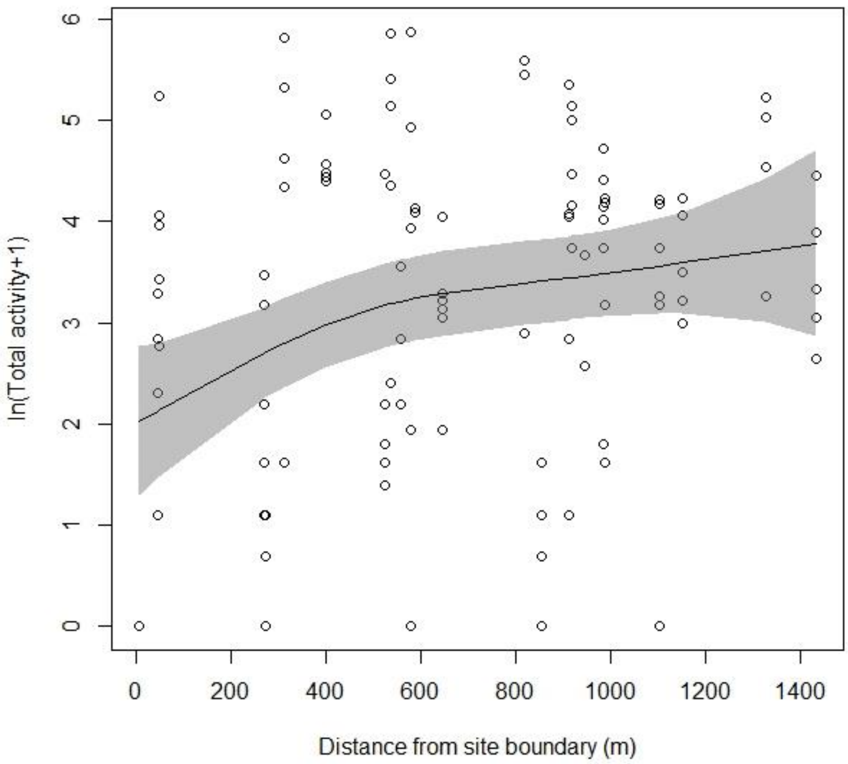


Figure 3.8
A simple generalised additive model was used to visually represent total bat activity within woodland habitats, based on distance from the site boundary with 95% confidence interval shown in grey. In demonstrating the potential for a non-linear relationship between activity and distance, the model suggested that activity may plateau at approximately 400m from the site boundary.

3.43 Effect of distance from mine on bat activity at a species level

The three most commonly identified species/ genus were *P. pipistrellus, P. pygmaeus* and *Myotis* (identified on 71.9%, 48.4% and 45.1% of sampling sessions respectively). *R. ferrumequinum* was only identified on 17.6% of sampling sessions.

The models which were considered to have equivalent support ($\Delta AIC_c \leqslant 2$) within each analysis are displayed in Table 3.4. All eight of the top models for *P. pipistrellus* contained distance to the site boundary as a fixed effect, and three models also contained an interaction between distance and habitat type (Figure 3.9a). Given that *P. pipistrellus* passes accounted for 81.7% of total bat activity, it is unsurprising that the model predicted trends for the distance: habitat interaction are largely similar in both analyses (Figure 3.6a, 3.9a). *P. pipistrellus* activity appears to decrease with proximity to the site boundary within all habitat types, but with a steeper trend predicted within woodland and bracken habitats. Again, the low proportion of bracken monitoring locations mean that this trend is easily influenced by outliers. One bracken monitoring location surrounded by woodland positioned at 668 metres from the site boundary appears to be driving this trend, with consistently higher levels of *P. pipistrellus* activity detected over three nights of monitoring than in the other bracken locations. To visualise the influence of these outliers, the interaction plot was reproduced without the data from this monitoring location (Figure 3.9b). With these four influential points removed from the dataset, four models received equivalent support ($\Delta AIC_c \leqslant 2$) and only one of these contained an interaction between distance and habitat type (Appendix C: Table 4). Therefore the interaction effect displayed in Figure 3.9a should be interpreted with caution.

Contrary to the interaction plots (Figure 3.9a, b), graphical visualisation of the raw data suggested that within woodland habitats the relationship between *P. pipistrellus* activity and distance from the site boundary may follow an asymptotic rather than linear trend (Figure 3.10). To investigate this further, a simple generalised additive model was used to visually represent log(*P. pipistrellus* activity+1) within woodland habitats, based on distance from the site boundary. The relationship was found to be significant ($F_{2.79,\ 107.21} = 6.83$, $p < 0.001$) and showed that *P. pipistrellus* activity appears to plateau at approximately 400m from the site boundary (Figure 3.11).

Within the *P. pygmaeus, Myotis* and *R. ferrumequinum* analyses, none of the models with equivalent support ($\Delta AIC_c \leqslant 2$) contained an interaction between distance and habitat type, however a proportion of the top models contained distance to the site boundary as a fixed effect (Table 3.4b, c, d).

Following model averaging the parameter estimates for the final models predicting bat activity on a species/ genus level are displayed in Table 3.5. Parameters for which the confidence intervals did not include zero were considered the most important predictors of species activity at the study site.

Distance from the site boundary was found to be a significant predictor of *P. pipistrellus* activity, with differing effects depending on the habitat type of the monitoring location. The confidence intervals indicated that the distance: habitat interaction trend within grassland habitats differed significantly from the trend within bracken habitats.

Distance from the site boundary was found to be a significant predictor of *R. ferrumequinum* activity, with increasing distance from the site boundary correlating with a higher probability of detecting *R. ferrumequinum*. Graphic visualisation indicates that relative to the site boundary, *R. ferrumequinum* is approximately twice as likely to be detected in areas 0.8 km from the boundary and three times as likely to be detected in areas 1.4 km from the boundary (Figure 3.12). Distance from the site boundary was found not to be a significant predictor of *P. pygmaeus* and *Myotis* activity.

Table 3.4

The full set of models for bat activity on a species level **a)** *P. pipistrellus* **b)** *P. pygmaeus* **c)** *Myotis* species **d)** *R. ferrumequinum* for which ΔAICc ⩽ 2, which were included in model averaging. Shown is the deviance (Dev), the number of model parameters (*K*), the Akaike's Information Criterion (AICc) and the difference in Akaike's Information Criterion between each model and the top-ranked model (ΔAICc) and the Akaike weight (w_i) of each model.

a)

Model rank	**Model name**	**Response = ln (*P. pipistrellus* passes+1); random effects = date and location**	**Dev**	***K***	**AICc**	**ΔAICc**	**w_i**
1/8	Model 3a	Habitat * distance + precipitation + temperature + wind	1103.7	8	1128.7	0.0	0.200
2/8	Model 3b	Habitat + distance + precipitation + temperature + wind	1108.3	8	1129.0	0.3	0.173
3/8	Model 3c	Distance + precipitation + temperature + wind	1112.8	7	1129.2	0.5	0.156
4/8	Model 3d	Habitat * distance + humidity + precipitation + temperature + wind	1102.9	9	1130.0	1.3	0.105
5.5/8	Model 3e	Habitat + distance + precipitation + temperature + wind + blasting	1107.3	9	1130.1	1.4	0.100
5.5/8	Model 3f	Habitat * distance + precipitation + temperature + wind	1103.0	8	1130.1	1.4	0.100
7/8	Model 3g	Habitat + distance + humidity + precipitation + temperature + wind	1107.6	9	1130.4	1.7	0.086
8/8	Model 3h	Distance + precipitation + temperature + wind + blasting	1111.9	8	1130.5	1.8	0.081

b)

Model rank	**Model name**	**Response = ln (*P. pygmaeus* passes+1); random effects = date and location**	**Dev**	***K***	**AICc**	**ΔAICc**	w_i
1/7	Model 4a	Habitat + precipitation + temperature	855.1	6	871.5	0.0	0.245
2/7	Model 4b	Habitat + humidity + precipitation + temperature	853.5	7	872.1	0.6	0.181
3/7	Model 4c	Habitat + blasting + precipitation + temperature	854.2	7	872.7	1.2	0.134
4.5/7	Model 4d	Habitat + humidity + precipitation + temperature + wind	852.3	8	873.0	1.5	0.116
4.5/7	Model 4e	Habitat + blasting + humidity + precipitation + temperature	852.3	8	873.0	1.5	0.116
6.5/7	Model 4f	Habitat + precipitation + temperature + wind	854.6	7	873.2	1.7	0.105
6.5/7	Model 4g	Habitat + distance + precipitation + temperature	854.7	7	873.2	1.7	0.105

c)

Model rank	Model name	Response = ln (*Myotis* passes+1); random effects = date and location	Dev	*K*	AICc	ΔAICc	w_i
1/8	Model 5a	Habitat + precipitation + temperature	738.0	6	754.5	0.0	0.207
2.5/8	Model 5b	Precipitation + temperature	742.8	7	755.1	0.6	0.153
2.5/8	Model 5c	Habitat + precipitation + temperature + wind	736.6	7	755.1	0.6	0.153
4/8	Model 5d	Precipitation + temperature + wind	741.1	6	755.5	1.0	0.125
5/8	Model 5e	Habitat + humidity + precipitation + temperature	737.1	7	755.7	1.2	0.113
6/8	Model 5f	Humidity + precipitation + temperature	741.8	6	756.2	1.7	0.088
7.5/8	Model 5g	Blasting + habitat + precipitation + temperature	737.8	7	756.4	1.9	0.080
7.5/8	Model 5h	Distance + precipitation + temperature	742.1	6	756.4	1.9	0.080

d)

Model rank	Model name	Response = Presence/absence of *R. ferrumequinum*; random effects = date and location	Dev	*K*	AICc	ΔAICc	w_i
1/10	Model 6a	Distance + temperature + wind	274.7	6	286.9	0.0	0.160
2/10	Model 6b	Temperature + wind	276.8	5	287.0	0.1	0.152
3/10	Model 6c	Habitat + temperature + wind	272.8	6	287.2	0.3	0.138
4/10	Model 6d	Habitat + temperature	275.8	5	288.0	1.1	0.092
5/10	Model 6e	Distance + temperature	277.9	5	288.1	1.2	0.088
6/10	Model 6f	Temperature	280.1	4	288.2	1.3	0.084
7/10	Model 6g	Habitat + distance + temperature + wind	271.8	7	288.3	1.4	0.080
8/10	Model 6h	Habitat	278.2	4	288.4	1.5	0.076
9/10	Model 6i	Habitat + wind	276.3	5	288.5	1.6	0.072
10/10	Model 6j	Distance + wind	278.7	5	288.9	2.0	0.059

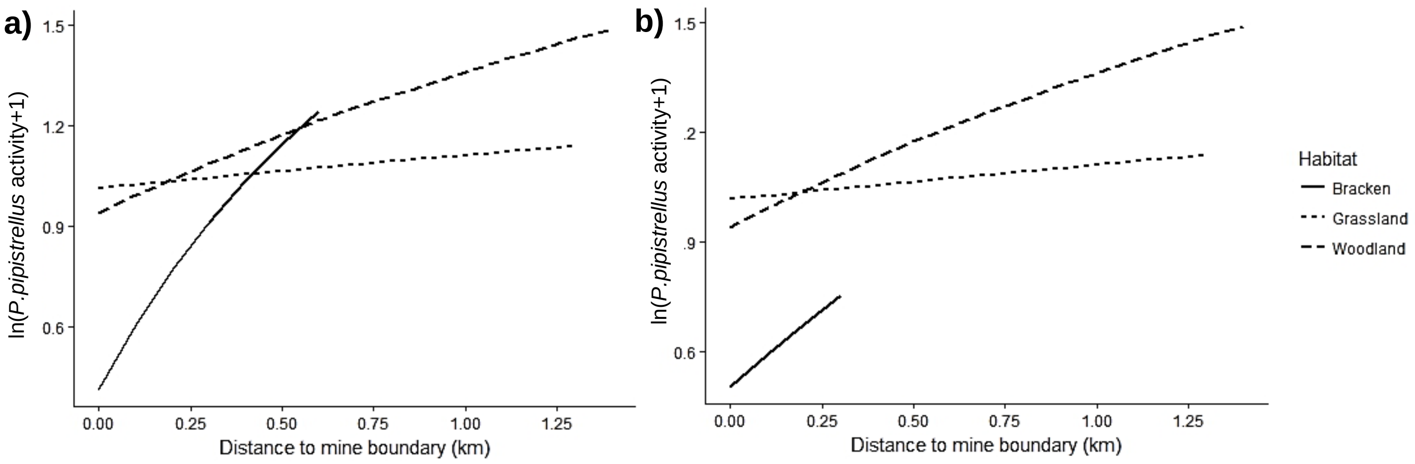


Figure 3.9
The relationship between *P. pipistrellus* activity and distance from the mine in the various habitat types.
a) The predicted relationship based on model 3a (Table 3.4a).
b) The same predicted relationship, but with the influential bracken monitoring location removed from the data set.

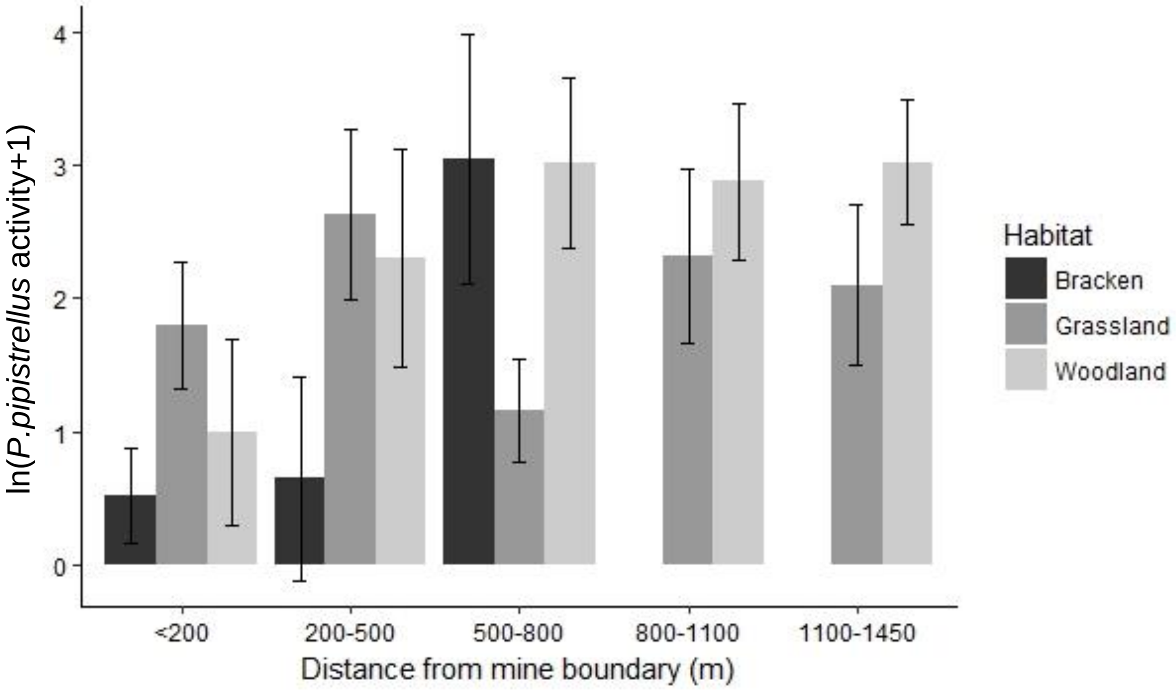


Figure 3.10
The relationship between mean *P. pipistrellus* activity and distance from the site boundary within habitat type, using raw data and displaying standard error bars. While distance was used as a continuous variable in analyses, here it is displayed as a categorical variable.

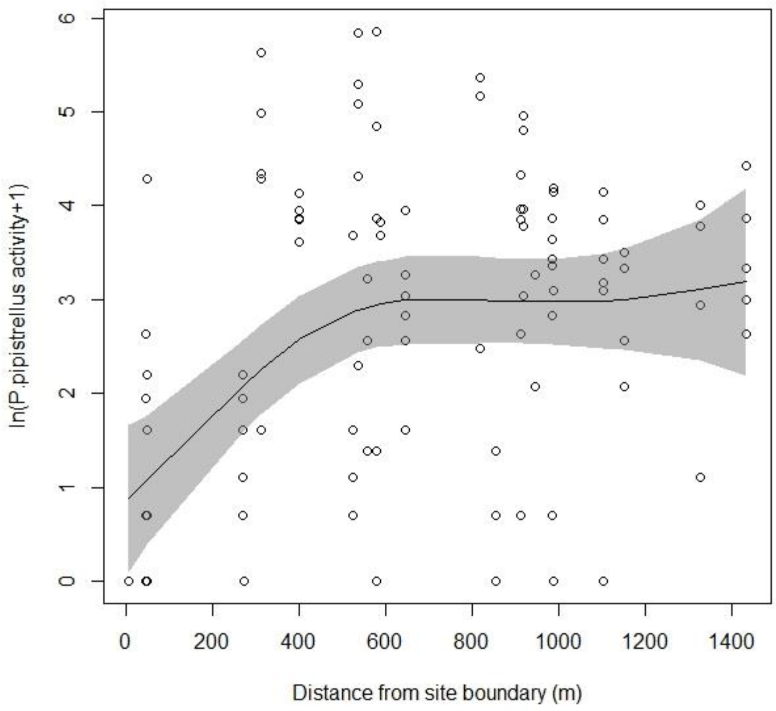


Figure 3.11
A simple generalised additive model was used to visually represent *P. pipistrellus* activity within woodland habitats, based on distance from the site boundary with 95% confidence interval shown in grey. In demonstrating the potential for a non-linear relationship between *P. pipistrellus* activity and distance, the model suggested that activity may plateau at approximately 500m from the site boundary.

Table 3.5

Model-averaged parameter estimates for bat activity on a species level **a)** *P. pipistrellus* **b)** *P. pygmaeus* **c)** *Myotis* species **d)** *R. ferrumequinum* in the area surrounding Drakelands mine. Shown is the model-averaged means (Estimate), associated standard error (SE) and confidence intervals (10%, 90%).

a)

Response: ln(*P. pipistrellus* passes+1), random effects = date and location			
Parameter	**Estimate**	**SE**	**Confidence interval (10,90)**
(Intercept) †	2.006	0.694	1.116, 2.896
Distance (km)	1.531	1.395	-0.259, 3.321
Blasting	-0.206	0.219	-0.487, 0.075
Habitat (grassland)	-0.086	0.798	-1.109, 0.938
Habitat (woodland)	0.393	0.839	-0.685, 1.470
Distance: habitat (grassland)	-2.529	1.466	-4.412, -0.647
Distance: habitat (woodland)	-1.652	1.499	-3.577, 0.273
Precipitation	-0.628	0.260	-0.961, -0.294
Temperature	0.682	0.256	0.354, 1.011
Wind	-0.771	0.239	-1.078, -0.465
Humidity	-0.221	0.250	-0.542, 0.100

†Habitat (bracken) was the reference category.

b)

Response: ln(*P. pygmaeus* passes+1), random effects = date and location			
Parameter	**Estimate**	**SE**	**Confidence interval (10,90)**
(Intercept) †	0.220	0.275	-0.133, 0.572
Distance (km)	0.115	0.179	-0.115, 0.345
Blasting	0.113	0.111	-0.029, 0.255
Habitat (grassland)	0.365	0.296	-0.015, 0.744
Habitat (woodland)	1.333	0.315	0.929, 1.737
Precipitation	-0.336	0.118	-0.487, -0.184
Temperature	0.366	0.139	0.188, 0.544
Wind	-0.114	0.129	-0.279, 0.051
Humidity	-0.173	0.128	-0.337, -0.010

†Habitat (bracken) was the reference category.

c)

Response: ln(*Myotis* passes+1), random effects = date and location			
Parameter	**Estimate**	**SE**	**Confidence interval (10,90)**
(Intercept) †	0.432	0.275	0.080, 0.784
Distance (km)	0.010	0.055	-0.065, 0.322
Blasting	-0.003	0.029	-0.163, 0.081
Habitat (grassland)	0.210	0.267	0.055, 0.707
Habitat (woodland)	0.321	0.353	0.236, 0.930
Precipitation	-0.267	0.104	-0.401, -0.133
Temperature	0.351	0.113	0.206, 0.497
Wind	-0.036	0.081	-0.268, 0.004
Humidity	0.0216	0.065	-0.033, 0.246

†Habitat (bracken) was the reference category.

d)

Response = Presence/absence of *R. ferrumequinum*, random effects = date and location			
Parameter	**Estimate**	**SE**	**Confidence interval (10,90)**
(Intercept)[†]	-3.689	1.543	-5.668, -1.709
Distance (km)	0.853	0.625	0.051, 1.655
Habitat (grassland)	2.289	1.443	0.436, 4.141
Habitat (woodland)	2.671	1.501	0.744, 4.598
Wind	-0.830	0.520	-1.498, -0.161
Temperature	0.968	0.533	0.284, 1.652

[†] Habitat (bracken) was the reference category.

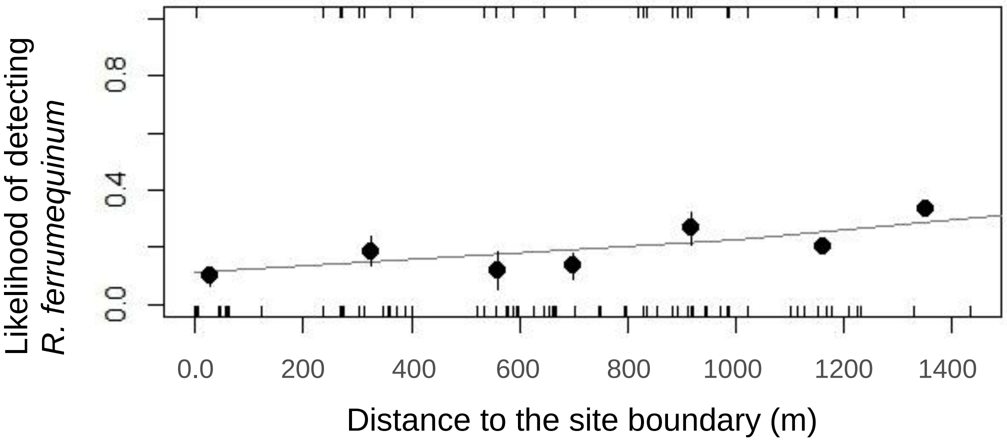


Figure 3.12
The likelihood of detecting *R. ferrumequinum* in relation to distance from the site boundary. The location of data points for presence/ absence are displayed as marks along the top and bottom of the plot. The continuous variable of “distance” was split into seven categories, with the markers showing the likelihood of detection for each, with associated standard error bars. The logistic regression is shown in grey.

3.44 Effect of blasting on bat activity

The categorical variable ‘blasting’ was included as a fixed effect in the global models for all analyses, to determine whether blasting has an effect on bat species richness and activity levels the subsequent night. Within the analysis of total activity, *P. pipistrellus, P. pygmaeus,* and *Myotis* activity, blasting was retained as a fixed effect within a small proportion of the top models ($\Delta AIC_c \leqslant 2$) (1/5, 2/8, 2/7, and 1/8 respectively: Table 3.3a, 3.4a, b, c). Sampling sessions on which blasting was conducted during the previous day had marginally lower bat activity than sessions on which blasting did not occur (Appendix C: Figure 1). However, as indicated by the estimate, standard error and confidence intervals following model averaging, there is little support for this effect and blasting is not considered an important predictor of bat activity at the study site (Table 3.3b, 3.5a, b, c).

3.5 Discussion:

Using simple generalised additive models we found that bat activity and species richness decreased with proximity to the site boundary. However, using model averaging techniques we identified varying trends depending on the habitat type of monitoring location (Grueber *et al.* 2011). Within grassland habitats, both species richness and total activity remained fairly consistent regardless of distance to the site boundary. However within woodland habitats both species richness and total activity were found to increase with distance from the site boundary, with the effect potentially extending to 400 and 500 metres respectively. Based on the model predictions, species diversity within woodland habitats increased by 20% between 0 and 400 meters from the mine boundary (Figure 3.3a). Further, total bat activity within woodland habitats increased by 73% between 0 and 400 meters from the mine boundary based on the model predictions (Figure 3.6a). By comparison, a study by Berthinussen and Altringham (2012) found that total bat activity increased by 100% between 0 and 500 meters from a major road (within the highest grade of habitats sampled, at 60 minutes after sunset). Within bracken habitats, an increase in species richness and total activity was also observed, but due to the small sample size of bracken monitoring locations this trend may be unreliable. Within habitat type, there was a strong correlation between the model predicted trends for species richness and total activity, indicating that areas of high species richness also had high levels of bat activity. This association between richness and activity has been observed in other studies of disturbance and fragmentation (Medellín *et al*. 2000; Estrada *et al.* 1993).

Potential reasons for lower species richness and total bat activity in proximity to the site boundary include habitat degradation due to light, noise and chemical pollution and habitat loss within the site boundary resulting in reduced foraging and commuting opportunities. The impacts of chemical contamination on bat populations as a result of mining are well documented, including the bioaccumulation of metals in prey items such as Chironomidae which may pose threats to Chiroptera over vast areas (Voight and Kingston, 2016; Cain *et al.* 1992). However, chemical contaminants at Drakelands mine are strictly controlled with water sources regularly checked for leached contaminants; therefore pollutants do not explain the decrease in bat activity and richness in proximity to the site boundary. Light pollution is also regulated across the site, with many areas remaining unlit and light levels along boundary vegetation kept below 4 lux (Gillingham, 2014). Although lighting levels as low as 3.6 lux have been shown to significantly reduce activity levels of both *R. hipposideros* and *Myotis* species, any potential impacts of on-site lighting would be limited to areas adjacent to and within the site boundary (Stone *et al.* 2012). However, the range of artificial lighting and noise may have a wider reaching impact within more open habitats such as grassland, relative to more cluttered habitat such as woodland where absorption of light and sound would be more rapid.

The primary source of noise pollution at Drakelands mine is the processing plant which currently operates for 24 hours per day. High frequency noise emitted from the processing plant largely falls within the range of 12-40 kHz; well within the auditory sensitivity range of bat species present at the study site (Luo *et al.* 2014; Clarke, pers.comm, 2017). Studies suggest that anthropogenic noise has the potential to degrade habitats by reducing the foraging efficiency of species which use passive listening or low frequency echolocation (<35 kHz) to locate prey (Schaub *et al*. 2008; Siemers and Schaub, 2010; Bunkley *et al*.

2015). Although anthropogenic noise from the processing plant may be degrading the suitability of habitats for some bat species at the study site, the potential impacts are likely to be short-ranging and largely affect habitats within the site boundary rather those in than the wider landscape. While the impacts of anthropogenic noise may contribute to the observed reduction in species richness and total bat activity in areas close to the processing plant, this does not explain the observed trends which may potentially extend to 400 metres.

However, the loss of foraging areas and commuting routes within the site boundary may explain the reduced species richness and total bat activity observed over greater distances. By reducing the movement of bats in the direction of the mine, this may lead to habitat fragmentation and barrier effects (Rico *et al.* 2009). Evidence suggests that by restricting access to foraging habitats and potentially reducing the size and quality of the home range, barrier effects may lead to lower reproductive output in bats (Voight and Kingston, 2016). On a much larger scale than that of Drakelands, barrier effects could potentially reduce gene flow between populations of rarer bat species with lower mobility, resulting in inbreeding (Meyer *et al.* 2009; Kerth and Petit, 2005). Although there may be barrier effects present at the Drakelands site which restrict the home-range size of individuals roosting near the site boundary, the surrounding landscape remains reasonably well connected therefore metapopulations should be maintained. Furthermore, mitigation measures implemented at Drakelands mine would hope to soften the impacts of the barrier effect, by providing additional foraging and commuting opportunities within the site boundary. These measures include the preservation of habitat fragments, the creation of new habitat corridors via woodland planting, the creation of new water bodies and the installation of 81 bat boxes (Michel Hughes Associates, 2013a). While bat boxes may be occupied quickly, it often take several years for regular occupation to be established and for breeding groups to form (McAney and Hanniffy, 2015). Installation of bat boxes at Drakelands began during the winter of 2013 therefore these boxes should currently be supplementing roost opportunities for established groups, provided the potential impacts of anthropogenic noise form the mine site do not act as a deterrent (Michel Hughes Associates, 2014).

The trends observed at Drakelands mine are unlikely to be due to differences in the quality of woodland habitat, as all sampling locations were within mature, predominantly broadleaf woodlands, with detectors positioned along edges or rides. The woodland sampling locations closer to the boundary generally had more numerous water bodies nearby, which may enhance these areas for bats which prey on insects that are associated with water (Figure 3.1) (Vaughan, 1997). The largest woodland patch was also located close to the site boundary, although size of woodland patch is not necessarily correlated with the level of bat species richness it supports (Estrada and Coates-Estrada, 2002; Law *et al.* 1999; Fuentes-Montemayor *et al.* 2013). The woodland edges nearest the mine are likely to experience higher levels of disturbance, including the felling of nearby trees (for example to create the link roads). Woodland disturbance has been associated with lower abundance of Lepidoptera*,* an important prey item for several British bat species (Dodd *et al.* 2012; Vaughan, 1997).

Interestingly, distance from the site boundary did not appear to influence species richness within grassland habitats. This may be because the wing morphology and foraging strategies of species adapted for open habitats makes them less susceptible to the impacts of habitat fragmentation, relative to species which forage close to surfaces and within cluttered habitats and are therefore reluctant to fly in the open (Jones *et al.* 2003). This is supported by a study by Kerth and Melber (2009) which identified species-specific barrier effects of a motorway on *B. barbastellus* which forages in open spaces and *M. bechsteinii* which gleans insect prey from vegetation. They found that five out of six tagged *B. barbastellus* crossed the motorway both by flying over and using underpasses for foraging and roost switching purposes. In contrast, only three of 34 tagged *M. bechsteinii* crossed the motorway using an underpass, solely for foraging purposes. These results suggest that manoeuvrable species with broad, short wings which are adapted to forage within cluttered environments may be more at risk from barrier effects than long winged, less manoeuvrable species which forage within open environments (Kerth and Melber, 2009; Russ, 2012). This theory is supported by other studies which have observed *Nyctalus* species which forage in open spaces commuting over habitat barriers such as roads more frequently than bats of other genus (Berthinussen and Altringham, 2012a; Berthinussen and Altringham, 2012b).

Further, the foraging flight range for species within the genus *Nyctalus* is typically larger than that of species within the genus *Plecotus, Myotis* and *Rhinolophus* which frequently forage within woodland habitats (Smith and Racey, 2014; Entwhistle *et al*. 2011; Razgour *et al*. 2013; Fisher *et al.* 2005; Shiel *et al.* 2006; Robinson and Stebbings, 1997). A recent study suggests that variation in the size of home range among bat species may explain the identified physiological differences in response to habitat disturbance, depending on roost preference (Seltmann *et al.* 2017). Therefore, open-space specialists such as *Nyctalus* may be less at risk from the impacts of habitat fragmentation and barrier effects than woodland specialists due to their greater mobility; this may contribute to the differing trends observed within woodland and grassland habitat types (Meyer *et al.* 2009). The presence of artificial lighting within the mine site may further aggravate barrier effects for woodland specialists, as species that are reluctant to cross open spaces are commonly averse to lighting (Voight and Kingston, 2016). Species within the genus *Rhinolophus*, *Myotis* and *Plecotus* are thought to be less tolerant of artificial lighting than open-air foragers such as *Pipistrellus*, *Nyctalus* and *B. barbastellus*. Therefore, while artificial lighting within the site boundary may increase the foraging efficiency of open-space specialists, it may result in spatial avoidance and competitive exclusion of some woodland species (Stone, 2013).

Bat activity in relation to distance from the site boundary was examined on a species/ genus level for *P. pipistrellus, P. pygmaeus, R. ferrumequinum* and *Myotis* species. Using model averaging techniques, distance from the site boundary was found to be a predictor of *P. pipistrellus* activity, with differing effects depending on the habitat type of the monitoring location. The model predicted trends revealed that although *P. pipistrellus* activity decreased with proximity to the site boundary within all habitat types, the strength of this trend was greater within woodland habitats than grassland habitats (while a lack of data meant that the bracken predicted trends may be unreliable). The observed decrease in *P. pipistrellus* activity in proximity to the mine within all habitat types may be in response to the lack of foraging and commuting opportunities within the site boundary, leading to barrier effects (Berthinussen and Altringham, 2012b). Further, artificial lighting at the nearby town of Plympton (2km southwest of the site boundary) may act as a feeding resource by attracting high densities of insects, therefore leading to elevated *P. pipistrellus* activity levels in the nearby habitats (Stone *et al.* 2012). *P. pipistrellus* passes accounted for 81.7% of total bat activity, and the model predicted trends for the distance: habitat interaction were largely similar in both analyses. Therefore, these factors may also explain the increase in total bat activity observed in relation to the distance from the site boundary.

However, the differing strength of this trend for *P. pipistrellus* activity dependent on habitat type was unexpected. The elevated levels of *P. pipistrellus* activity within woodland areas further from the site boundary may be a result of inter-specific habitat partitioning with *P. pygmaeus,* leading to spatial segregation of their foraging ranges (Nicholls and Racey, 2006). Contrary to *P. pipistrellus,* distance from the site boundary was not found to be a significant predictor of *P. pygmaeus* activity, which was predominantly detected within woodland areas. Due to differences in dietary composition, *P. pygmaeus* is more strongly associated with water than *P. pipistrellus;* therefore woodland patches located in proximity to the water bodies within the site boundary may provide more optimal foraging habitat for this species (Russ and Montgomery, 2002; Boughey *et al*. 2011). Further, relative to *P. pipistrellus,* studies have found that *P. pygmaeus* is more likely to be detected in landscapes containing a high proportion of woodland and a low proportion of urban grey space (Lintott *et al.* 2015). Despite both species being well adapted to forage within woodland edge and interior habitats, *P. pygmaeus* may preferentially select woodland habitats closer to the site boundary due to the greater proportion of woodland and water bodies. In turn, this may result in spatial avoidance of these foraging areas by *P. pipistrellus;* shifting activity further from the site boundary and towards the areas of urban grey space which are less appealing to *P. pygmaeus.* Several studies have suggested that differential habitat use *P. pipistrellus* and *P. pygmaeus* may be a result of partitioning to avoid competition for foraging resources (Nicholls and Racey, 2006, Lintott *et al.* 2015). However, further research into the habitat preferences of *P. pipistrellus* living in allopatry is needed to assess the extent of this phenomenon (Rachwald *et al.* 2016).

Similarly to *P. pygmaeus,* distance to the site boundary was found not to be a significant predictor of *Myotis* species activity, which was predominantly detected within woodland areas. Some *Myotis* species are thought to favour the interior of dense deciduous woodland in proximity to water, reflecting their gleaning foraging strategies and the increased availability of Diptera prey (Fuentes-Montemayor *et al.* 2013; Russ and Montgomery, 2002). Further, research suggests that woodland patches smaller than 1km^2 do not provide suitable habitat for woodland-adapted bat species (Lesiński *et al.* 2007). Therefore, while *Myotis* species may be experiencing boundary effects in proximity to Drakelands mine, the woodland habitat sampled closer to the mine may represent more optimal habitat (larger woodland patch with greater interior area, located close to water bodies), resulting in elevated *Myotis* species activity in these areas. This is supported by the uptake of roost boxes by *M. daubentonii* and a significant breeding colony of *M. nattereri*, located within the largest woodland patch close to the site boundary and the water bodies (Gillingham, 2016a).

Due to low detection rates, *R. ferrumequinum* activity was examined in a binomial analysis of presence/ absence. Increased distance from the mine was found to correlate with an increase in the likelihood of detection, with *R. ferrumequinum* approximately twice as likely to be detected in areas 0.8 km from the boundary and three times as likely to be detected in areas 1.4 km from the boundary. Studies suggest that *R. ferrumequinum* tends to forage within 4km of the roost site, flies close to commuting features such as hedges, walls and tree lines and is averse to artificial lighting (Flanders and Jones, 2009; Jones and Rayner, 1989; Stone *et al.* 2012). These characteristics may make *R. ferrumequinum* particularly sensitive to habitat loss, fragmentation and disturbance, potentially resulting in a restricted home-range and barrier effects in areas close to the site boundary. Further, insect prey with an aquatic larval stage only makes up a small component of the diet of *R. ferrumequinum*, so the new water bodies created will not provide a significant feeding resource (Jones, 1990). Despite this, *R. ferrumequinum* has adopted a new roost provision for hibernation and continues to utilize the retained roost buildings on the periphery of the site (Gillingham, 2016a). The maturation of woodland planting and hedgerows will enhance the site interior for this species in the coming years, particularly as moth abundance (which forms a substantial part of the diet of *R. ferrumequinum*) is thought to be positively associated with woodland patch size (Fuentes-Montemayor *et al.* 2012; Vaughan, 1997).

Blasting was included as fixed effect in a proportion of the top-ranked models for the analysis of total activity, *P. pipistrellus* activity, *P. pygmaeus* activity and *Myotis* species activity. However, following model averaging there was little support for this effect in any analysis, therefore blasting may not be considered an important predictor of bat activity at the study site. If an effect was present, this may have been masked by weather bias; blasting is only carried out at Drakelands mine on days with low wind speeds and good visibility, conditions which also correlate with higher levels of bat activity (Clarke, pers.comm, 2017). While studies have examined the effects of noise and vibration from mine blasting on daily roosting and hibernation, to our knowledge no other study has investigated the potential for delayed effects of blasting on subsequent nightly foraging activity (Player and Keim, 2015; Luo *et al.* 2014; West Virginia Department of Environmental Protection, 2006). Although blasting was not found to be a significant predictor of bat activity, our data was collected over a relatively wide spatial scale. Therefore, in the analysis of subsequent bat activity data that is collected in proximity to blast locations, it may be worthwhile including blasting as a factor in the analysis to investigate potential short-range effects.

Environmental conditions were included in all global models to account for variation in weather among sampling nights. Increasing wind speeds correlated with reduced species richness, total activity, *P. pipistrellus* activity and *R. ferrumequinum* presence. Other studies have observed that bats are detected more frequently at lower wind speeds and suggest that this may be linked to high insect activity during periods of low wind (Cryan *et al.* 2014; Horn *et al.* 2008). Insect activity is also dependent on temperature, and may explain why higher temperatures correlated with increased levels of species richness, total activity and activity on a species level (or increased presence in the case of *R. ferrumequinum*) (Barros *et al.* 2014; Taylor, 1963). This may also be a result of low ambient temperatures leading to more rapid heat loss in active bats, therefore making foraging during lower temperatures less energetically profitable (Reichard *et al.* 2010). Rainfall may also impair bat flight by increasing energetic costs, in addition to imposing sensory constraints on echolocation (Voigt *et al.* 2011). In line with this, we identified lower levels of species richness, total activity and activity on a species level on sampling nights with higher total rainfall. Finally, lower *P. pygmaeus* activity was detected on nights with higher levels of humidity; this may be a result of high humidity reducing the maximum distance of bat echolocation, therefore resulting in decreased microphone sensitivity (Schnitzler *et al.* 2001).

3.6 Conclusion:

Evaluating the potential impacts of habitat loss and fragmentation on bat populations is crucial in order to enable informed management recommendations for effective conservation. By examining the effect of distance from an open pit mine on activity and species richness in the surrounding landscape, this study suggests that the impacts of habitat loss on bat species may vary depending on their foraging ecology. Our results suggest that species with greater manoeuvrability which are adapted to forage close to vegetation clutter are more susceptible to barrier effects than species which are adapted to forage within open environments. The reduced foraging flight range of woodland adapted bats and aversion to artificial lighting may further contribute towards this effect. Of the four Annex II listed bat species occurring within the UK, *R. hipposideros*, *R. ferrumequinum* and *M. bechsteinii* all forage close to vegetation clutter (JNCC, 2016; Russ, 2012). With areas of the mining sector predicted to expand, this research highlights that habitat fragmentation may pose an elevated risk in the wider landscape to already-threatened species (UK Minerals Forum, 2014). At new mining developments bat mitigation should seek to develop substantial commuting routes to connect isolated woodland habitat patches with the wider landscape and consider enhancing existing woodland habitats that are in proximity to the site boundary. Further, during link-road construction underpasses created on existing commuting routes may reduce barrier effects for clutter-specialists that fly close to ground-level. Clearly, further research into the species-specific responses of bats to barrier effects is needed, in order to better predict the responses of bat populations in the wider landscape to new developments.

Chapter 4: Field trial of a novel bat detector

4.1 Abstract:

Increasing concern about recent declines in bat populations has highlighted the need for population monitoring over extended time periods. This may be achieved using ultrasonic detectors, but in order to sample over large temporal and spatial scales equipment must be energy efficient and able to store large amounts of data. Using a unique recording technology, the BatBug is a novel bat detector with a huge range of potential applications in the field of long-term monitoring. Therefore, it is important to quantify performance in line with other commercially available bat detectors, to identify potential variation in results and equipment limitations. Here, we assessed the relative sensitivity of the BatBug Ranger against the Anabat SD1 and SD2 models from industry leading brand Titley Scientific, by sampling bat activity in the field. While some differences were observed between the results of paired detectors, detector model was not found to have a significant effect on the levels of species richness and total bat activity detected per night. Due to their low power consumption, compact data storage, high quality waterproofing and competitive pricing, the BatBug models from Chelonia Limited represent a valuable new tool in long-term microchiropteran monitoring.

4.2 Introduction

With over 1,300 species, bats make up more than twenty percent of extant mammal species, and most of these are Microchiroptera (Teeling *et al.* 2005; Voight and Kingston, 2016). Despite being widespread, the cryptic nature and metapopulation structure of microbats makes it difficult to monitor changes in abundance over large spatial and temporal scales using traditional techniques such as mark-recapture (Ingersoll *et al.* 2013). Ultrasonic detectors which identify high frequency echolocation calls provide researchers with an invaluable non-invasive tool to investigate bat ecology and these have been employed in a range of studies, from the identification of cryptic species to investigating mating systems and ecological impacts (Parsons, 1996; Jahelkova and Horacek, 2011; Lintott *et al.* 2016). Increasing concern about recent declines in bat populations has highlighted the need for population monitoring over extended time periods (Meyer *et al.* 2010). This may be achieved using passive acoustic monitoring, but in order to sample over large temporal and spatial scales ultrasonic detectors must be energy efficient and able to store large amounts of data, in order to minimise maintenance.

A wide selection of bat detectors are now commercially available. Although prices and features vary widely, the ability to detect bats will primarily be determined by key features such as microphone quality, recording technology and sampling rate (Adams *et al.* 2012). Here, we present the results of a field trial of a novel bat detector - the BatBug$^{©}$, created by acoustic monitoring specialists Chelonia Limited. Chelonia Ltd initially developed acoustic data loggers (C-POD$^{©}$) designed to be left at sea unattended for the purpose of monitoring toothed cetaceans based on their echolocation click trains (Pierpoint, 1999). Other developments include the Banana Pinger$^{©}$, a cost effective fishery tool which emits sound pulses for the reduction of ceteacean by-catch. The BatBug represents Chelonia's first development of an ultrasonic bat detector, and is the initial step in a project seeking to develop robust instruments for accurate long term monitoring of microbat activity (Tregenza, pers.comm, 2017).

Detectors that are currently widely used in static monitoring include the various Anabat models (Titley Scientific), the SM2BAT and successive SM4BAT (Wildlife Acoustics), various Batlogger models (Elekon) and the D500X (Petterson). Previous research has evaluated the relative performance of these commonly used bat detectors, therefore this study will focus on comparing the sensitivity of the prototype BatBug Ranger to the Anabat SD1 and successive SD2 models from industry leading brand Titley Scientific, which are frequently used by researchers and ecological consultants (Adams *et al.* 2012; Parsons, 1996; Fenton *et al.* 2001; Stahlschmidt and Brühl, 2012). The SD1 and SD2 Anabat models were selected for use in this comparative study because their compact data format and low power use makes them suitable for long-term static monitoring (Britzke *et al.* 2013). Therefore, these shared attributes mean that the BatBug Ranger and the Anabat SD2 may occupy a similar niche in the market. The features and parameters of each detector model are listed in Table 4.1.

Today’s commercially available bat detectors generally operate using one of three recording technologies; heterodyne, frequency division and time expansion (Russ, 2016). However, the BatBug operates using a different approach, currently termed ‘high resolution cycle logging’. This process involves high speed acoustic sampling within the range of 17-220 kHz at the effective rate of 4 MHz per second, termed a cycle. With real-time call detection and storage of the amplitude and wavelength of every cycle in a call, this allows the parameters of bat echolocation calls to be visualised in high resolution. Additionally, continuous sound analysis known as ‘micro-triggering’ means that sound between bat calls is not recorded, therefore greatly reducing the file size and hence storage needs. The recorded bat calls are then compressed to wavelength, amplitude and bandwidth data; this compact format means that a single SD card can potentially hold up to one year’s worth of monitoring data. In addition to compact data storage, the low power consumption of the BatBug enables long-term monitoring, therefore minimising maintenance and facilitating deployment in remote monitoring locations.

Similarly, the AnaBat SD1/SD2 has highly efficient data storage capabilities and uses little power (using an external power supply the Anabat will operate for 1 day per Amp hour of battery) (Hourigan and Corben, 2012). The Anabat technology focuses on capturing and analysing information on call structure, using frequency division to reduce the true frequencies of a sound and make bat echolocation calls audible, and zero-crossings analysis to make bat calls visible, with no lag between call detection and audio output or visual display (if the Anabat is connected to a laptop). While this process retains the original structure of the call, some detail such as call amplitude is not preserved. Further, harmonics are not always visible in Analook files, as the zero crossings technology that is used only displays the loudest part of a signal at a given time (Russ, 2012).

Two BatBug models are currently in development. The Ranger, which is intended for mobile or static surveys and the Endurance which is suitable for very long static deployments. Powered by internal lithium primary batteries and operating on a dark-only cycle, the Ranger and Endurance have a running time of approximately nine months and 2.8 years respectively. Both models also record temperature and light levels every minute, and these data are visualised alongside an overview of bat activity in the dedicated BatBug.exe software (Figure 4.1). Given that the BatBug has a huge range of potential applications in the long-term monitoring of bat populations, it is important to quantify performance in line with other commercially available bat detectors.

Although some studies suggest that the Anabat SD2 may detect fewer echolocation calls than detectors which operate using time expansion technology, overall the Anabat is thought to provide a sufficient picture of bat activity and remains a highly effective tool for long-term static monitoring (Fenton, 2000; Adams *et al.* 2012). Identifying variation in recording technology is an important step in assessing the suitability of different ultrasonic detectors for the research questions being addressed. Here, we assess the relative performance of the prototype BatBug Ranger against the Anabat SD1/SD2, by comparing the levels of species richness and total bat activity recorded by each detector model.

Calculating the species richness and total activity detected using each detector model will provide an overview of relative sensitivity. As well as varying in sensitivity, the relative performance of different bat detector models will depend on the frequency of the echolocation call, which may result in detection bias (Adams *et al.* 2012). This detection bias is also influenced by interspecific variation in the structure, frequency and intensity of echolocation calls, leading to under-representation of some species when using acoustic monitoring techniques (Neuweiler, 1989). Therefore, in this study we will compare the total number of passes detected using each detector model on a species level, to investigate potential sensitivity bias among detector types. We will also calculate a Shannon-Weiner diversity index score for each detector type, as this measure accounts for both the number of species detected and the evenness of species present (Spellerberg, 2008).

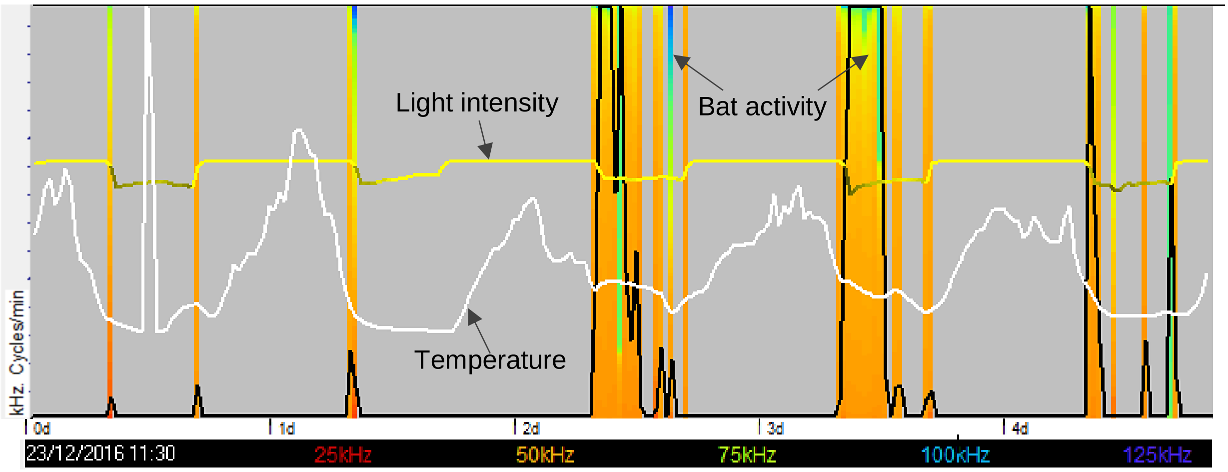


Figure 4.1
Screen shot of the BatBug.exe software displaying five nights of monitoring data. In this low-resolution view, temperature and light intensity data are displayed alongside an overview of bat activity data. In this view echolocation calls are colour-coded according to frequency, while the black line provides a measure of activity levels by displaying the number of octets (a measure of data) of bat data logged. Within BatBug.exe, data from up to three BatBugs may be viewed within one window.

Table 4.1

Summary of the features and parameters of the Anabat SD2 (Titley Electronics) and BatBug Ranger (Chelonia Ltd). Multiple battery options are denoted with a superscript number, which matches that of the corresponding running time. Other models of bat detectors are available from both manufacturers.

Feature	Anabat SD1/SD2	BatBug Ranger
Battery type	^{1} 4 AA NiMH/ 4 AA Alkaline ^{2} 12V lead acid external battery	^{1} 18 AA NiMH or ^{2} 12 AA Alkaline ^{3} 12 AA Lithium primary
Running time (night only)	^{1} Up to 2 days ^{2} 1 day per Amp hour	^{1} 72 days ^{2} 90 days ^{3} 9 months
Recording technology	Zero-crossing	High resolution cycle logging
Sampling rate	N/A	4 MHz
Sound file	Anabat	BB1
Storage system	Compact Flash Card (CF)	Secure Digital Card (SD)
Microphone type	Condenser	Condenser (Electret)
Frequency range	5-200 kHz	17-220 kHz
Microphone directionality	45° from the axis	45° from the axis
Post-process tools	AnaLookW	BatBug.exe
Weatherproof casing	No	Yes
Weatherproof microphone	No	Yes
Light sensor	No	Yes: internal
Temperature sensor	No	Yes: internal
Price from manufacturer	£1,794	To be announced, approximately £750

4.3 Methods

4.31 Field set-up

We simultaneously deployed the prototype BatBug Ranger alongside AnaBat models SD1 and SD2. Two BatBug Ranger detectors were tested over four sampling sessions, with each BatBug paired with one of four Anabat detectors for each session. Each sampling session lasted for five consecutive nights, with detectors set to record from 30 minutes before sunset until sunrise. Testing was conducted in a rural area close to the town of Plympton in southwest Britain, during spring 2017. The Anabat detectors were placed in weatherproof boxes and elevated to 1m above ground level on a tripod, tilted upwards at an angle of approximately 45°. The Anabat stainless steel microphone was protected from water damage with a single sheet of plastic food wrap, which is thought to have no significant effect on microphone sensitivity (Moyes, pers.comm, 2016). The BatBug Ranger was fastened onto the paired Anabat detector, with the corresponding microphones positioned less than ten centimetres apart. The detector settings used in all trials are listed in Table 4.2; for a description of these settings please refer to the relevant detector manuals. Sensitivity on the Anabat is adjusted using the 'gain' control, which may be set to a maximum value of ten. However, insect noise and electromagnetic interference may require the sensitivity to be reduced to minimise undesired signals. Following advice within the Anabat SD2 user manual, a value of 6.5 was selected for these trials (Hourigan and Corben, 2012).

Table 4.2
The detector settings used in this trial relating to the collection and storage of data. Settings were selected according to recommendations within the respective detector manuals.

Anabat SD1 and SD2	**BatBug Ranger**
Gain: 6.5	Max data chunk size: 16,384
Audio Div: 16	High pass filter: 20 kHz
Data Div: 8	N of data chunks per min: 4

4.32 Analysis

Recordings from the Anabat detectors were visualised in AnalookW (Titley Scientific, Lancashire, United Kingdom), while recordings from the BatBug Ranger were visualised in BatBug.exe (Chelonia Ltd, Cornwall, United Kindgom). Echolocation calls were visually inspected and assigned to a species or genus based on comparison to a reference library of known species and the echolocation parameters provided in the literature by Russ (2012). Echolocation calls for which confident identification was not possible were labelled ‘unknown’ (due to echolocation calls being too faint or in the case that only a short segment of a call was detected) and occasionally because the parameters of a call fell between those of two species. Echolocation calls assigned to *Myotis* and *Plecotus* were only identified to a genus level, due to similarities in call parameters (Kunz and Parsons, 2009; Russ, 2012). All echolocation calls of the same species detected within a one minute period were classed as one ‘pass’. Species richness was calculated for each night of monitoring, consisting of the sum of the number of species detected (or genus for *Myotis* and *Plecotus*) and excluding those labelled ‘unknown’. Total activity was calculated for each night of monitoring as the sum of the number of passes of all species, including those labelled ‘unknown’. Using the total number of passes detected for each species, we calculated the overall Shannon-Weiner diversity for each detector model, and compared these two values using the Hutcheson t-test (Krebs, 1985; Hutcheson, 1970).

All other statistical analyses were performed using R statistical software version 3.3.1 (R Core Team, 2016). General Linear Models (GLMs) were constructed using the package “lme4” to investigate whether species richness and total bat activity varied by detector type (Bates *et al*. 2015). Species richness and total activity formed the response variable in each respective analysis, and detector type (Anabat/BatBug) was listed as a fixed effect. The response variable of total bat activity was square-root transformed to reduce heterogeneity resulting from large variation in activity levels between nights. As four Anabat detectors and two BatBug detectors were used in this trial, ‘detector ID’ was included as a random effect. Each BatBug was paired individually with each Anabat for a duration of five nights. Therefore ‘pair ID’ and ‘location’ were also included as random effects, to identify each individual pairing and account for the non-independence of monitoring in the same location for multiple nights. The model was fitted using restricted maximum likelihood (REML) and normal error distribution, and was validated by visual examination of residuals and q-q plots (Bolker *et al.* 2009). To determine if detector type had a significant effect on the level of species richness and total activity detected, a likelihood ratio test (ANOVA) was used to compare the global models with intercept-only models (Crawley, 2014). We present the predicted species richness and predicted transformed total activity calculated for each detector type ± standard error, using the global models.

Linear regressions were calculated to see whether the species richness and total activity detected by the Anabat detectors could be predicted by the species richness and total activity detected by the Batbug detectors.

4.4 Results

A total of 4,892 bat passes and at least ten species were identified. Bat species were *Pipistrellus pipistrellus, Pipistrellus pygmaeus, Pipistrellus nathusii, Rhinolophus ferrumequinum, Rhinolophus hipposideros, Nyctalus noctula, Nyctalus leisleri, Barbastellus barbastellus, Myotis* species and *Plecotus* species.
The total number of calls detected using each detector model over the 40 night sampling period are displayed by species/genus in Figure 4.2.

The Shannon-Weiner diversity index values calculated for each detector model were not found to differ significantly (Anabat = 0.659 ± 0.031, BatBug = 0.670 ± 0.034, $t_{(79)} = 0.377$, $p = 0.706$).

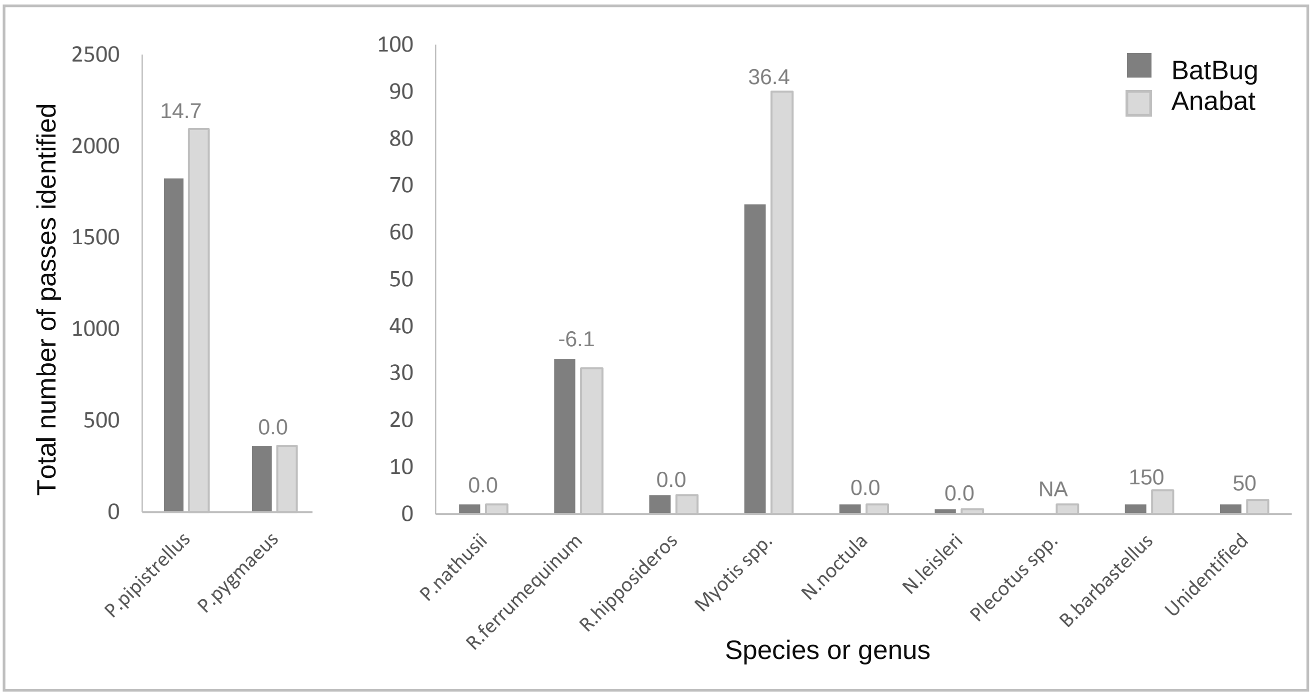


Figure 4.2
The total number of bat passes detected by the BatBug Ranger and the Anabat SD1/SD2 ultrasonic detectors over the 40 night sampling period, displayed by species/genus. The percentage difference in the number of calls detected by Anabat relative to BatBug for each species/ genus is displayed above the bars.

Detector model did not significantly predict the species richness detected, therefore the null model was accepted ($\chi^2_{(1)} = 2.010$, $p = 0.156$). The predicted species richness detected using the Anabat detector was 2.900 ± 0.357, compared to a value of 2.725 ± 0.124 detected using the BatBug (Table 4.3a). Species richness detected by the Anabat detectors could be predicted by species richness detected by the BatBug detectors ($F_{(1,38)} = 93.17$, $p < 0.001$). Variation in the BatBug detector explained 71% of variation in the Anabat detector, with Anabat species richness predicted by the following equation: Anabat richness = 0.451*(BatBug richness) + 0.899, $R^2 = 0.710$ (Figure 4.3a).

Detector model also did not significantly predict total bat activity ($\chi^2_{(1)} = 1.52$, $p = 0.218$). The total activity (back-transformed) detected using the Anabat detector was 45.738 ± 1.016, compared to 38.812 ± 0.210 with the BatBug (Table 4.3b). Total activity detected by the Anabat detectors could be predicted by total activity detected by the BatBug detectors ($F_{(1,38)} = 1170$, $p < 0.001$). Variation in the BatBug detector explained 96% of variation in the Anabat detector, with Anabat total activity predicted by the following equation: Anabat activity = 6.757*(BatBug activity) + 1.011, $R^2 = 0.969$ (Figure 4.3b).

Table 4.3
Output of GLMMs investigating whether detector model (Anabat/BatBug) had an effect on the level of **a)** species richness **b)** total bat activity detected. As four Anabat detectors and two BatBug detectors were used in this trial, 'Detector ID' was included as a random effect. Each BatBug was paired individually with each Anabat for a duration of five nights. Therefore 'Pair ID' and 'Location' were also included as random effects, to identify each individual pairing and account for the non-independence of monitoring in the same location for multiple nights.

a)

Parameters	**Estimate**	**SE**
Fixed effects:		
(Intercept)[†]	2.900	0.357
BatBug Ranger	-0.175	0.124
Random effects		**SD**
Pair ID		0.959
Location		0.879
Detector ID		0.000

[†] Anabat SD1/SD2 was the reference category.

b)

Parameters	**Estimate**	**SE**
Fixed effects:		
(Intercept)[†]	6.763	1.008
BatBug Ranger	-0.533	0.458
Random effects		**SD**
Pair ID		3.806
Location		2.156
Detector ID		0.507

[†] Anabat SD1/SD2 was the reference category.

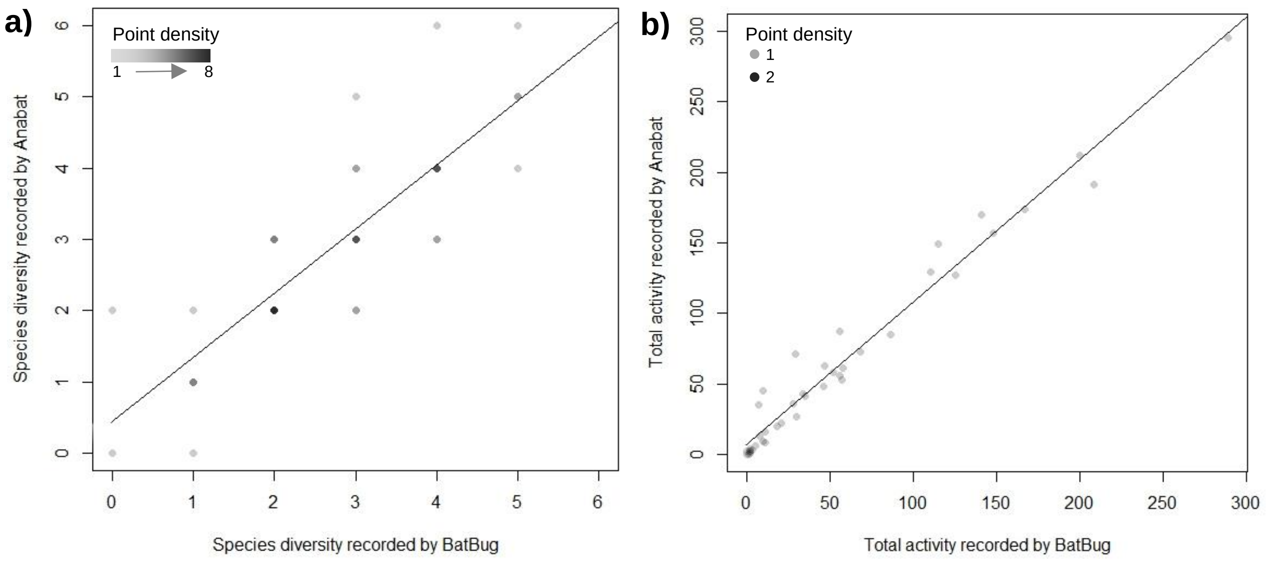


Figure 4.3
Correlation plot showing **a)** species richness **b)** total number of bat passes, detected per night by paired BatBug and Anabat detectors. The density of points is indicated by shade of grey, and the regression line is displayed in black.

4.5 Discussion

Regardless of the recording technology, the microphones of ultrasonic bat detectors will only detect a subset of the echolocation calls present in the environment (Adams *et al.* 2012*)*. However, greater microphone sensitivity allows a larger volume of space to be sampled with a single detector, therefore enabling more echolocation calls to be detected (Corben and Fellers, 2001). In comparing the relative sensitivity of the prototype BatBug Ranger against the Anabat SD1/SD2, we demonstrate that there were no significant differences in the levels of species richness and total bat activity detected using the two detector models. A significant regression equation was found for both species richness and total bat activity detected between the paired detectors, indicating that individual detectors within each model consistently performed to a similar standard. Further, the Shannon-Weiner diversity index scores which account for both the number of species detected and the evenness of species present, were not found to differ significantly by detector model (Spellerberg, 2008). While the BatBug failed to detect *Plecotus* passes, the lower number of

Pipistrellus pipistrellus passes detected resulted in a greater equitability among the species detected than within the Anabat sample and therefore led to a marginally higher diversity index score.

While differences in the amplitude and frequency of bat echolocation calls leads to variation in their detectability, this detection bias may be further compounded by variation in the performance of an ultrasonic detector model over a range of frequencies (Murray *et al.* 2009). Research comparing the detection efficacy of ultrasonic detectors using synthetic call playback found that the Anabat SD2 was less effective at detecting frequencies of 85 kHz, relative to the various full-spectrum detectors tested (Adams *et al.* 2012). Within the United Kingdom, the echolocation calls of *Rhinolophus ferrumequinum* most closely match this frequency, with a maximum energy of 81.3 kHz (Russ, 2012). The Anabat detectors recorded 6.1% fewer *R. ferrumequinum* passes than the BatBug detectors, indicating that a similar detection bias may also be present in this comparison. However, the BatBug detectors recorded fewer *P. pipistrellus* and *Myotis* passes than the Anabat (12.8% and 26.7% respectively), which may indicate that the BatBug samples a marginally smaller area than the Anabat. Although rarely detected, the Anabat also recorded a greater number of *Barbastellus barbastellus* and *Plecotus* passes. Relative to other bat species of Britain, the echolocation calls of *B. barbastellus*, *Plecotus auritus, Plecotus austriacus* and *Myotis* species are lower in amplitude, and in the case of *M.daubentonii* highly directional (Goerlitz *et al.* 2010; Waters and Jones, 1995; Surlykke *et al.* 2009; Russ, 2012; Schaub *et al.* 2008). Based on the total number of passes detected for each species, the BatBug may be less sensitive in detecting quieter calls and those with strong directionality than the Anabat; this could be investigated further in a study of longer duration with more statistical power.

Alternatively, the variation in the number of passes detected may be a result of differences in the detection algorithms between the two models, rather than microphone sensitivity. While both the Anabat and BatBug use sound analysis to select and store ambient sounds that resemble bat echolocation, the BatBug further filters this raw data into files which have been automatically marked as belonging to bat calls. Although this process may result in a slightly more conservative file set, it saves the researcher from having to sift through potential recordings from insects, birds and traffic. With developments in the automated analysis of sound files, automated bat identification software may allow the unbiased processing of large volumes of data (Russo and Voigt, 2016). The BatBug.exe software currently allows the user to filter bat passes based on a large selection of call parameters, but an automated classifier for British bat species is also in development. Although automatic identification programs have been shown to successfully classify some bat species with characteristic echolocation calls, other species are commonly misidentified resulting in high rates of false positives (Russo and Voigt, 2016; Rydell *et al.* 2017; Stathopoulos *et al.* 2018). One way for researchers to minimise this risk is to use classifiers to identify easily recognisable species such as *P. pipistrellus* and *Pipistrellus pygmaeus*, and then manually identify the remaining files (Rydell *et al.* 2017). Alternatively, species can be classified into broader groups such as genus, but social calls and files containing multiple species may still present problems for automatic identification software. The performance of any automatic classifier will depend on the trade-off between sensitivity and specificity, which may be visualised using receiver operating curves (Andreassen *et al.* 2014).

Despite widespread deforestation and habitat fragmentation leading to declines in bat populations in tropical ecosystems, long term monitoring in these geographical areas is lacking (Meyer *et al.* 2010). Due to their low power consumption, compact data storage, high quality waterproofing and competitive pricing, the BatBug models may help to facilitate microchiropteran research in remote monitoring locations such as the rainforest canopy, or at inaccessible locations such as oil rigs and wind turbines. Further, the ability to view data from up to three BatBugs in sync within the BatBug.exe software, alongside temperature and light intensity data could be a valuable tool in identifying seasonal or longer-term trends. On a finer scale, the ability of the BatBug to preserve harmonics and retain the amplitude information of recorded echolocation calls may be valuable in species identification and for studies examining the fine-scale characteristics of acoustic signals (Britzke *et al.* 2013).

4.6 Conclusion

Ultrasonic detectors provide researchers with an invaluable non-invasive tool to investigate bat ecology. However, differences in performance among detector models will provide researchers with varying pictures of bat activity. Identifying variation in recording technology is therefore an important step in assessing the suitability of different ultrasonic detectors for the research questions being addressed. Here, we assessed the relative sensitivity of the novel BatBug Ranger from Chelonia Ltd against the Anabat SD1 and SD2 models from Titley Scientific. While some differences were observed between the results of paired detectors, detector model was not found to have a significant effect on the levels of species richness and total bat activity detected per night. Due to their low power consumption, compact data storage, high quality waterproofing and competitive pricing, the BatBug models represent a valuable new tool in long-term Microchiropteran monitoring. Areas for further investigation could include assessing the durability and performance of the BatBug under varying conditions such as high humidity and rainfall, and testing the battery life in areas of high bat activity.

Chapter 5: Summation

The extraction of minerals through open-pit mining can result in sudden and extensive land use change, often posing threats to local biodiversity (Sonter *et al*. 2014). Bats are particularly vulnerable to the impacts of mining, but their metapopulation structure and wide-ranging roosting habits make it challenging to monitor local populations (Voigt and Kingston, 2016; Wilkinson and South, 2002). The primary goal of this thesis was to investigate the impacts of habitat loss and disturbance on local bat populations at Drakelands open-pit mine, the first new metal mine to be established within Britain in the past 45 years. This goal was addressed in two parts, firstly by analysing data collected by contracted ecologists at the site, in order to identify potential short-term shifts in the activity levels of local bats and evaluate the efficacy of mitigation measures. Secondly, by monitoring bat activity in the wider-landscape to identify potential further-ranging impacts of the mine on local bat populations. In conjunction with this work I also incorporated a field trial of a novel bat detector designed for long-term monitoring of bat activity. Detailed discussions are provided at the end of each chapter. Here, I summarise and aggregate my findings from these sections, discuss potential implications and suggest directions for future research.

In analysing the data collected by contracted ecologists at the Drakelands site over a three year period, potential short term trends in local bat activity were identified. While some species appear to be benefitting from the implemented mitigation measures, the on-site detection rate of *Plecotus* species declined sharply over the monitoring period. Coinciding with the commencement of ore-processing (24 hours per day), this reduction may be a result of acoustic masking of prey resulting in reduced foraging efficiency and therefore spatial avoidance of the local area by the passive listening *Plecotus auritus* (Schaub *et al*. 2008; Siemers and Schaub 2010; Moretto and Francis, 2017; Clarke, pers.comm, 2017). Despite research supporting the use of passive listening by *P. auritus*, to our knowledge no study has investigated the impacts of anthropogenic noise on the foraging efficiency of this species (Eklof and Jones, 2003; Radford *et al.* 2012; Razgour *et al*. 2013; Anderson and Racey, 1993; Schaub *et al*. 2008; Siemers and Schaub, 2010). Although *P. auritus* remains common and widespread throughout Europe, the creation of busy roads and other anthropogenic sources of noise in proximity to broadleaf woodland could have more significant consequences for local *P. auritus* populations relative to species which forage using alternative strategies (Hutson *et al.* 2008). Investigation into the impacts of anthropogenic noise on *P. auritus* may therefore represent an important area for future research.

The development of an open pit mine and associated infrastructure will inevitably lead to habitat loss in the immediate vicinity, often resulting in reductions in local bat activity and diversity (ELAW, 2010; Brown and Berry, 1997). Interestingly, at the Drakelands site species richness did not significantly decrease over the three year monitoring period. We conclude that this may be due to the creation of significant new waterbodies within the site, providing enhanced Dipteran foraging resources and softening the impacts of lost woodland and grassland foraging habitats. The potential short term trends at a species/ genus level identified in this assessment will be useful in guiding management decision making at the Drakelands site, particularly if the trends observed are confirmed with data from subsequent monitoring years. However, due to the limited time frame of data collection (three years), these results should be treated with some caution until further data has been included in the analysis. This study was also limited by the inability to compare post-development bat activity with data collected before the mine was established, due to variation in sampling methodology. Where possible, future mining developments should consider standardising data collection methods for monitoring conducted pre and post development, in order to allow comparison of subsequent bat activity to baseline levels. Ideally, this would also extend to the data collection methodology used in monitoring the site following its restoration. A recent review of derogation licenses granted within England found that post-developmental monitoring was conducted at less than one fifth of sites, and of these only one third of licensees submitted post-development reports (Stone *et al.* 2013). Despite the rapid and widespread impacts that open pit mining can have on local biodiversity, published or openly available work documenting the response of local bat populations to mitigation measures and subsequent restoration is lacking. I feel that disseminating bat mitigation research is a crucial step in maximising the success of future mitigation work at developments involving large scale habitat loss, in order to achieve effective conservation outcomes (Fazey *et al.* 2004).

In addition to assessing the impacts of Drakelands mine on bat activity within the site, I monitored bat activity in the surrounding landscape in order to identify potential further-ranging impacts on local populations. Distance from the site boundary was found to be an important predictor of species richness and total bat activity, with effects varying with habitat type. Interestingly, within grassland habitats species richness and total bat activity remained fairly constant regardless of distance to the mine, whereas within woodland habitats increasing species richness and total activity correlated with increasing distance from the site boundary. In line with other studies, these findings suggest that bat species which are adapted to forage within clutter may be more susceptible to barrier effects (caused by the loss of foraging and commuting habitat within the site boundary) than species which are adapted to forage within open environments (Kerth and Melber, 2009; Berthinussen and Altringham, 2012a; Berthinussen and Altringham, 2012b).

Within the UK, three of the four Annex II listed bat species forage close to vegetation clutter: *R. hipposideros*, *R. ferrumequinum* and *M. bechsteinii*, in addition to *Myotis* and *Plecotus* species (JNCC, 2016; Russ, 2012). With areas of the mining sector predicted to expand, this research highlights that habitat fragmentation may pose an elevated risk in the wider landscape to already-threatened species depending on their foraging ecology (UK Minerals Forum, 2014). At new mining developments bat mitigation should seek to develop substantial commuting routes to connect isolated woodland habitat patches with the wider landscape and consider enhancing existing woodland habitats that are in proximity to the site boundary. Further, during link-road construction underpasses created on existing commuting routes may reduce barrier effects for clutter-specialists that fly close to ground-level. As highlighted in Chapter 2, both *P. auritus* and *M. bechsteinii* may be particularly vulnerable to the impacts of anthropogenic noise due to their passive listening foraging strategy. Both these species also forage close to vegetation clutter, and as identified in Chapter 3 may be more susceptible to experiencing barrier effects as a result of habitat loss (Schaub *et al.* 2008). Therefore, at new developments which will result in both habitat loss and anthropogenic noise it could be beneficial for mitigation for these species to be placed at a greater distance from the development, in areas that are less severely impacted.

Bat activity was also analysed in relation to distance from the mine for the more commonly detected species/ genus. Among the trends identified, *P. pipstrellus* activity was found to increase with distance from the site boundary, with the gradient differing according to habitat type. In contrast, *P. pygmaeus* activity did not vary with distance from the mine. We suggest that this may be a result of inter-specific habitat partitioning between the two cryptic species, leading to spatial segregation of their foraging ranges. As proposed by Nicholls and Racey (2006), *P. pipistrellus* may actively avoid the foraging sites of *P. pygmaeus* in favour of similar sites that are more distantly located, in order to avoid excessive competition. This theory is supported by studies from Lintott *et al.* (2015; 2016) which suggested that habitat partitioning between *P. pipistrellus* and *P. pygmaeus* occurs within habitat types, at a fine spatial scale. Our results further support this hypothesis, and highlight the complexity of assessing species specific responses to habitat loss and fragmentation in Microchiroptera. Further investigation of this potential phenomena at the Drakelands site would require designing transects that cover a larger spatial scale, in order to include the surrounding urban areas. To my knowledge, this study was the first to investigate the potential for delayed effects of blasting on levels of Microchiropteran foraging activity the following evening. Although blasting was not a significant predictor of bat activity in any analysis, the data was collected over a relatively wide spatial scale. Therefore, in subsequent studies that are conducted in proximity to blast locations, it may be worthwhile including blasting as a factor in the analysis to investigate potential short-range effects on bat activity.

In conjunction with the monitoring work carried out in Chapter 3, a field trial of a novel bat detector (the “BatBug”) designed for long-term monitoring of bat activity was incorporated, by comparing sensitivity relative to the Anabat SD1 and SD2 models. The levels of species richness and total bat activity identified in the trial did not differ between the two detector types. However, the Batbug may be less sensitive than the Anabat in detecting quieter calls and those with strong directionality. Due to funding constraints this study was conducted in early spring, and therefore the levels of bat activity were relatively low. Repeating this study later in the year or for a longer duration would result in greater statistical power and may provide greater insight into potential variation between the BatBug and commercially available detectors. The ability of the BatBug to preserve detailed information about the structure of echolocation calls would be valuable in the development of dedicated automated bat identification software. Automated identification software has the potential to allow unbiased processing of large volumes of data, and paired with the BatBug's low power consumption, compact data storage and competitive pricing this would make it a valuable tool in the cost effective long-term monitoring of Microchiroptera (including at active and restored mine sites) (Russo and Voigt, 2016).

To conclude, mitigating the impacts of mining on biodiversity is now an industry priority. Increasing awareness of the vulnerability of Microchiroptera to anthropogenic threats has led to increasing levels of legal protection in many countries. However, implementing effective mitigation for bats at mine sites requires an understanding into how impacts such as habitat loss and disturbance may affect the distribution and activity of local populations. The results of my thesis have highlighted the multitude of factors which influence bat activity at a local level, and may provide a platform for continued research into the impacts of habitat fragmentation and anthropogenic noise at a species/ genus level. I hope that the information presented here will help to inform management decision making in regards to bat conservation, both at the Drakelands site and potentially at mining operations elsewhere.

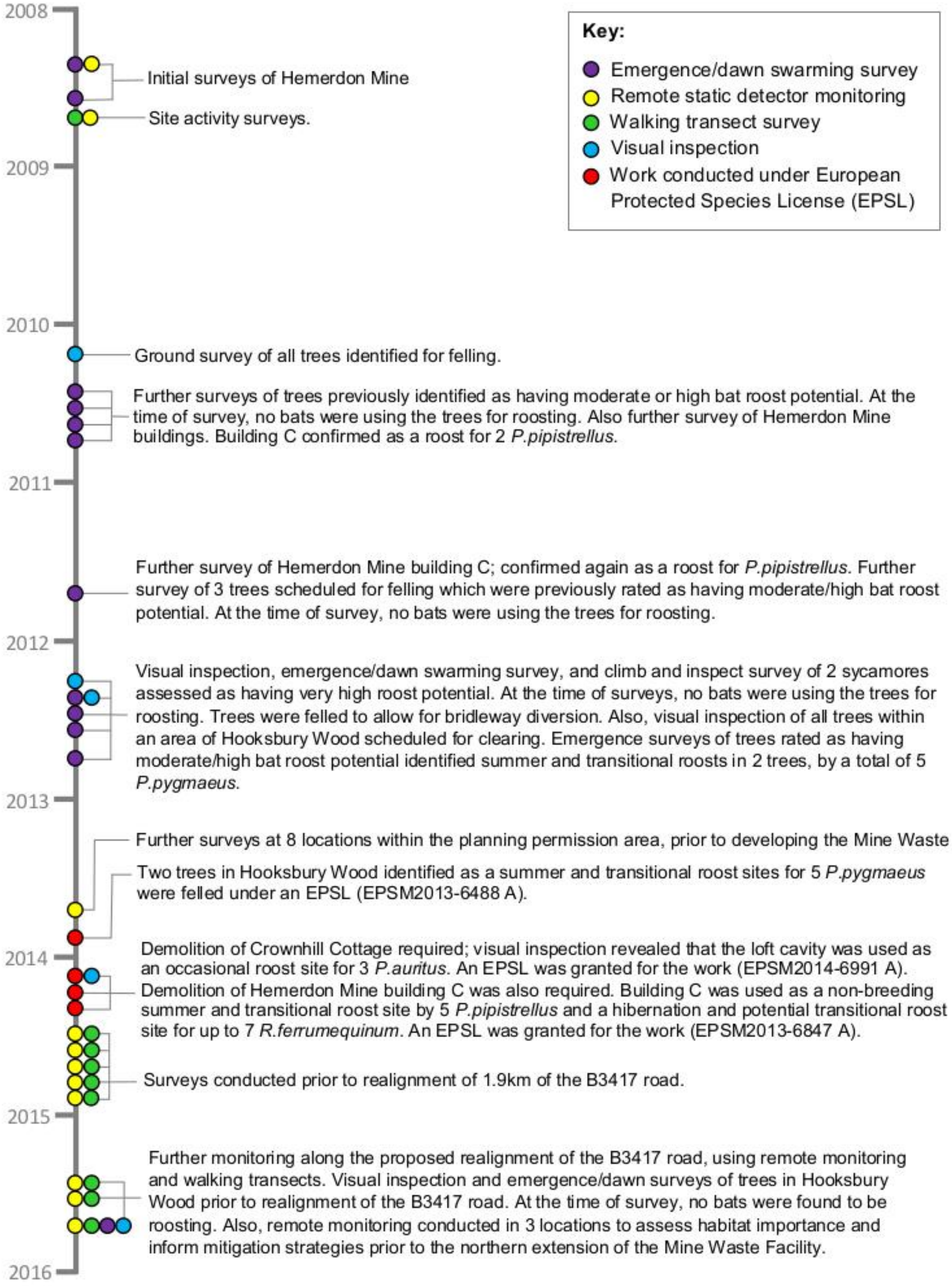
Appendix A: Chapter 1 supplementary material

Table 1
At least 11 bat species have been recorded within the Drakelands site. Details of their foraging and commuting habitat and foraging flight range are provided in the table below. References are provided in the footnote.

Species	Ecology
Pipistrellus pipistrellus	**Foraging and commuting habitat:** Wide variety of habitats including gardens, farmland, parkland, and deciduous woodland rides. Frequently forages along edge habitat and avoids open areas.[1] **Flight range:** Usually within 2 km of roost.[2]
Pipistrellus pygmaeus	**Foraging and commuting habitat:** Common near bodies of water and associated riparian habitat, in addition to broadleaf and mixed woodland edges. Generally less common in farmland.[1] **Flight range:** Usually within 1.5 km of roost.[3]
Pipistrellus nathusii	**Foraging and commuting habitat:** Lowland woodland rides (deciduous, occasionally coniferous) and meadows. Almost always recorded within a few km of large bodies of water.[1] **Flight range:** Unknown.
Myotis species	**Foraging and commuting habitat:** *Myotis* may be found within a range of habitats including woodland, riparian vegetation, parkland, meadows, gardens, and coniferous forests. All *Myotis* species tend to favour deciduous woodland and habitats close to water bodies.[1] **Flight range:** Up to 12 km.[4]
Plecotus species	**Foraging and commuting habitat:** *Plecotus* forages in open deciduous woodland, occasionally coniferous. Other habitats include parkland, gardens, hedges, treelines, and overgrown banks and streams. *Plecotus austriacus* will also hunt in open habitat such as meadows, and occasionally around streetlights and human settlements.[1] **Flight range:** Up to 8.7 km, but commonly 1-5 km.[5,6]
Rhinolophus ferrumequinum	**Foraging and commuting habitat:** Parkland, meadows and woodland, particularly near water. Commutes along rides and footpaths in woodland, and within 10m of hedges and treelines over pasture.[1] **Flight range:** Within 4 km of roost.[7]
Rhinolophus hipposideros	**Foraging and commuting habitat:** Largely deciduous woodland and wetlands, but also pasture, woodland edge, and hedgerows. Shows preference for broadleaf woodland but also forages along riparian vegetation bordering riverbanks.[1] **Flight range:** Usually within 2.5 km but up to 6.4 km.[7]
Barbastella barbastellus	**Foraging and commuting habitat:** Prefers wooded countryside, generally near bodies of water. Also forage around gorse and mercury vapour streetlights.[1]

	Flight range: Average 6.8 km.[8]
Nyctalus leisleri	**Foraging and commuting habitat:** Common above parkland, cattle pasture, meadows, and habitat borders. May be observed foraging over water bodies and around white streetlights.[1] **Flight range:** Up to 13.4 km.[9]
Eptesicus serotinus	**Foraging and commuting habitat:** Woodland edge, parkland, pasture and tall hedgerows. Also forages along rivers, lake shores, and above streetlights.[1] **Flight range:** Average 8 km, but up to 41 km.[10]
Nyctalus noctula	**Foraging and commuting habitat:** Found in a wide range of open habitats including deciduous woodland, pasture, parkland, marshland and rivers.[1] **Flight range:** Up to 26 km.[5]

[1] Russ, 2012; [2] JNCC, 2007; [3] SLR Consulting Ltd, 2013; [4] Smith and Racey, 2014; [5] Entwhistle *et al*., 2000; [6] Razgour *et al*. 2013; [7] Fisher *et al.* 2005; [8] Zeale *et al.* 2012; [9] Shiel *et al*. 2006; [10] Robinson and Stebbings, 1997.



Initial surveys of Hemerdon Mine

Site activity surveys.

Ground survey of all trees identified for felling.

Further surveys of trees previously identified as having moderate or high bat roost potential. At the time of survey, no bats were using the trees for roosting. Also further survey of Hemerdon Mine buildings. Building C confirmed as a roost for 2 *P.pipistrellus*.

Further survey of Hemerdon Mine building C; confirmed again as a roost for *P.pipistrellus*. Further survey of 3 trees scheduled for felling which were previously rated as having moderate/high bat roost potential. At the time of survey, no bats were using the trees for roosting.

Visual inspection, emergence/dawn swarming survey, and climb and inspect survey of 2 sycamores assessed as having very high roost potential. At the time of surveys, no bats were using the trees for roosting. Trees were felled to allow for bridleway diversion. Also, visual inspection of all trees within an area of Hooksbury Wood scheduled for clearing. Emergence surveys of trees rated as having moderate/high bat roost potential identified summer and transitional roosts in 2 trees, by a total of 5 *P.pygmaeus*.

Further surveys at 8 locations within the planning permission area, prior to developing the Mine Waste

Two trees in Hooksbury Wood identified as a summer and transitional roost sites for 5 *P.pygmaeus* were felled under an EPSL (EPSM2013-6488 A).

Demolition of Crownhill Cottage required; visual inspection revealed that the loft cavity was used as an occasional roost site for 3 *P.auritus*. An EPSL was granted for the work (EPSM2014-6991 A). Demolition of Hemerdon Mine building C was also required. Building C was used as a non-breeding summer and transitional roost site by 5 *P.pipistrellus* and a hibernation and potential transitional roost site for up to 7 *R.ferrumequinum*. An EPSL was granted for the work (EPSM2013-6847 A).

Surveys conducted prior to realignment of 1.9km of the B3417 road.

Further monitoring along the proposed realignment of the B3417 road, using remote monitoring and walking transects. Visual inspection and emergence/dawn surveys of trees in Hooksbury Wood prior to realignment of the B3417 road. At the time of survey, no bats were found to be roosting. Also, remote monitoring conducted in 3 locations to assess habitat importance and inform mitigation strategies prior to the northern extension of the Mine Waste Facility.

Figure 1. Timeline displaying the bat monitoring which has been undertaken at the Drakelands site since 2008. In addition to the investigations shown, annual monitoring of the site has been carried out following commencement of operations in 2014.

Appendix B: Chapter 2 supplementary material

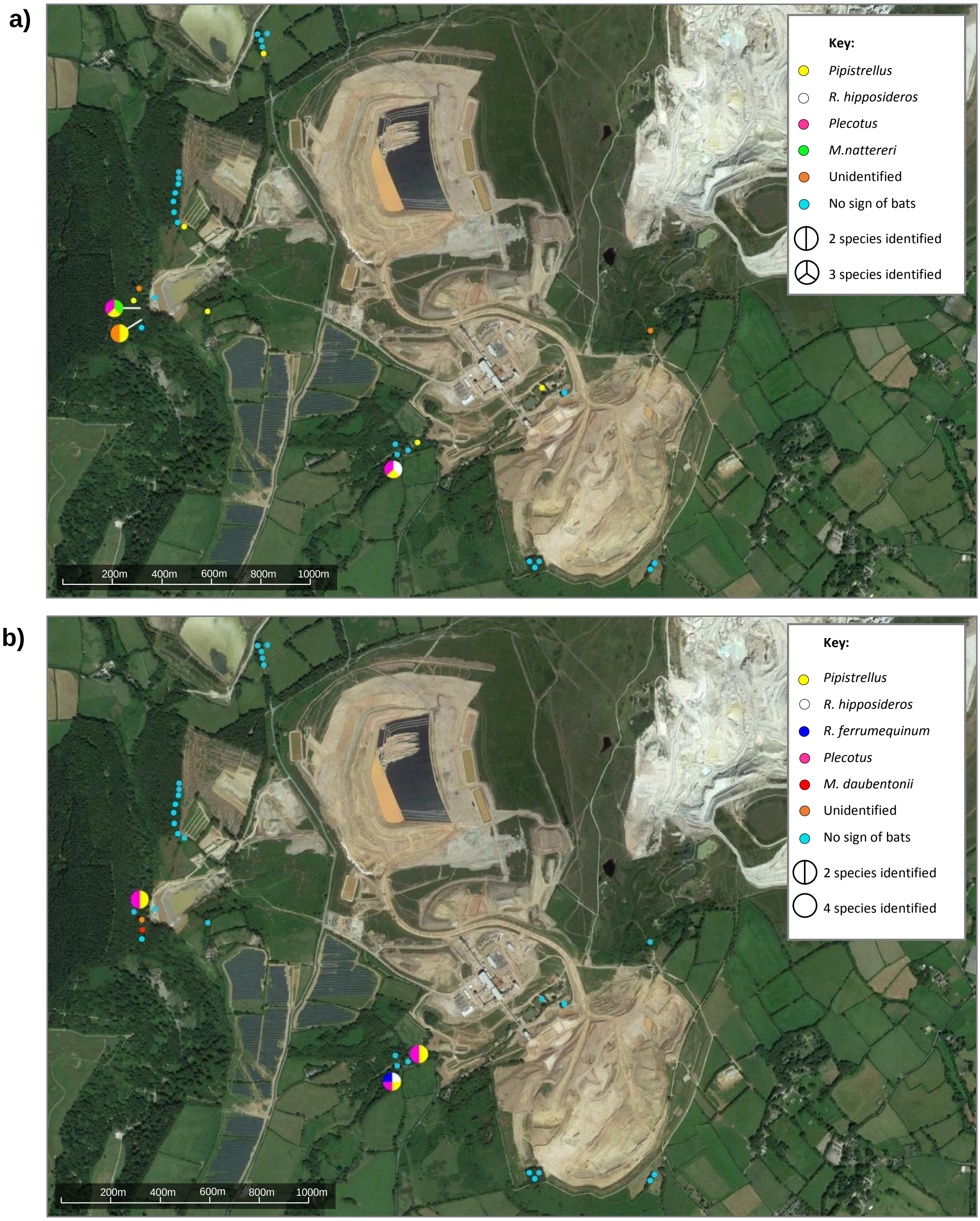


Figure 1. Cumulative results of the roost box monitoring at the Drakelands site from 2014-2016 during **a)** summer **b)** winter. Where bats were absent but droppings were present, identification was carried out to a genus level. (Aerial image © Google Earth, 2015)

Appendix C: Chapter 3 supplementary material

Table 1a

Output of GLzM investigating the performance of each Anabat detector in detecting species richness. The model was fit using Laplace Approximation, with poisson error distribution, BOBYQA optimization and a “log” link function. Random effects included in the model were date and location. Post hoc Tukey HSD tests showed that there was not significant variation in the levels of species richness detected between Anabat detectors.

Parameters	**Estimate**	**SE**
Fixed effects		
(Intercept)†	0.712	0.166
Anabat (B)	-0.273	0.229
Anabat (C)	0.020	0.234
Anabat (D)	-0.079	0.250
Anabat (E)	-0.071	0.225
Random effects		**SD**
Date		0.247
Location		0.526

†Anabat (A) was the reference category.

Table 1b

Output of GLM investigating the performance of each Anabat detector in detecting total bat activity. The model was fit using restricted maximum likelihood (REML) and normal error distribution. The response variable total number of passes was transformed to ln (passes+1) to reduce heterogeneity resulting from large variation in activity levels between nights while also accounting for nights with zero passes. Random effects included in the model were date and location. Post hoc Tukey HSD tests showed that there was not significant variation in the levels of total activity detected between Anabat detectors.

Parameters	**Estimate**	**SE**
Fixed effects		
(Intercept)†	2.490	0.335
Anabat (B)	-0.361	0.441
Anabat (C)	0.290	0.460
Anabat (D)	-0.127	0.488
Anabat (E)	-0.008	0.440
Random effects		**SD**
Date		0.890
Location		1.125

†Anabat (A) was the reference category.

Table 2
The full set of models for species richness (with the dataset modified to remove outlying points) for which ΔAICc ⩽ 2, which were included in model averaging. Shown is the deviance (Dev), the number of model parameters (*K*), the Akaike's Information Criterion (AICc) and the difference in Akaike's Information Criterion between each model and the top-ranked model (ΔAICc).

Model rank	**Model name**	**Response = species richness; random effects = date and location**	**Dev**	***K***	**AICc**	**ΔAICc**
2/2	Model 1a	Distance + habitat + wind + precipitation + temperature	1059.9	8	1077.9	1.3
1/2	Null model	Habitat + wind + precipitation + temperature	1060.6	7	1076.6	0.0

Table 3
The full set of models for total bat activity (with the dataset modified to remove outlying points) for which ΔAICc ⩽ 2, which were included in model averaging. Shown is the deviance (Dev), the number of model parameters (*K*), the Akaike's Information Criterion (AICc) and the difference in Akaike's Information Criterion between each model and the top-ranked model (ΔAICc).

Model rank	**Model name**	**Response = total activity; random effects = date and location**	**Dev**	***K***	**AICc**	**ΔAICc**
3/7	Model 2a	Distance * habitat + wind + precipitation + temperature	1074.7	8	1098.7	0.9
7/7	Model 2b	Distance + habitat + wind + precipitation + temperature + humidity	1077.5	9	1099.5	1.7
6/7	Model 2c	Distance + habitat + wind + precipitation + temperature + blasting	1077.4	9	1099.4	1.6
2/7	Model 2d	Distance + habitat + wind + precipitation + temperature	1078.0	8	1098.0	0.2
5/7	Model 2e	Habitat + wind + precipitation + temperature + humidity	1079.3	8	1099.3	1.5
4/7	Model 2f	Habitat + wind + precipitation + temperature + blasting	1079.1	8	1099.1	1.3
1/7	Null model	Habitat + wind + precipitation + temperature	1079.8	7	1097.8	0

Table 4

The full set of models for *P.pipistrellus* activity (with the dataset modified to remove outlying points) for which ΔAICc ⩽ 2, which were included in model averaging. Shown is the deviance (Dev), the number of model parameters (*K*), the Akaike's Information Criterion (AICc) and the difference in Akaike's Information Criterion between each model and the top-ranked model (ΔAICc).

Model rank	Model name	Response = ln (passes+1); random effects = date and location	Dev	*K*	AICc	ΔAICc
1/4	Model 3b	Habitat + distance + precipitation + temperature + wind	1093.8	8	1113.8	0.0
2/4	Model 3c	Habitat + distance + precipitation + temperature + wind + blasting	1093.0	9	1115.0	1.2
3/4	Model 3d	Habitat + distance + humidity + precipitation + temperature + wind	1093.2	9	1115.2	1.4
4/4	Model 3a	Habitat * distance + precipitation + temperature + wind	1091.2	8	1115.3	1.5

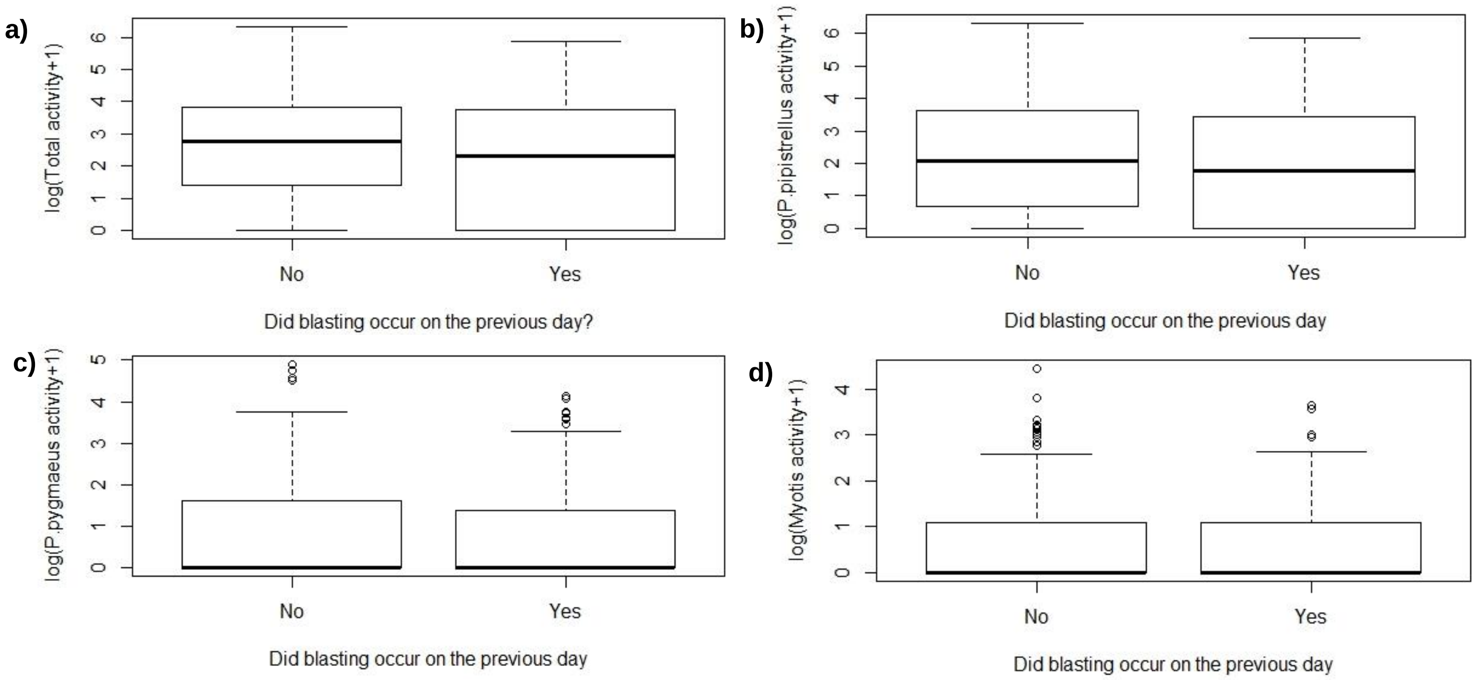


Figure 1

Boxplot displaying the range, upper and lower quartiles, mean and outliers for **a)** total bat activity **b)** *P.pipistrellus* activity **c)** *P.pygmaeus* activity and **d)** *Myotis* species activity, in relation to whether blasting occurred on the previous day.

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