

Article

Mines and bats: the impact of open-pit mining on bat activity.

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Abstract Areas used in open-pit mining often undergo sudden and extensive landscape changes, including habitat fragmentation. With mining activity predicted to expand, understanding the impacts of this habitat loss on wildlife is key to developing effective mitigation at new mine developments. Despite this, research into the impacts of open-pit mines in general, and on bat populations in particular, is largely lacking. Here, we investigated potential impacts of a recently re-established open-pit mine in southwest Britain on bat activity in the surrounding landscape, using remote monitoring techniques. Distance from the mine site boundary was found to be an important predictor of bat species richness and activity levels, with the effect on richness extending to approximately 900m from the site and the effect on activity potentially extending beyond the sampling range (1,200m). The effect of distance from the site boundary on species richness also varied with the proportion of woodland cover. It appears that habitat loss within the mine site boundary may have reduced bat movement because of barrier effects. The study highlights the need for further research into the impacts of mines on bats.

Keywords activity, bat, Chiroptera, disturbance, habitat, industry, mining, richness.

Short running title The impact of open-pit mining on bat activity.

Introduction

The dramatic alteration of ecosystems by human populations, particularly since the industrial revolution, has significantly impacted global biodiversity (Steffen et al. 2011). The mining industry was crucial in industrialisation and remains so in today's digital age, but the discovery, extraction and processing of minerals is environmentally disruptive, with impacts potentially persisting long after the closure of a mine (Bebbington et al. 2008; Spitz and Trudinger, 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 (Protocol on Environmental Protection to the Antarctic Treaty, 1991; Filho, 2003).

As a result of this past and recent expansion, mining can be a major anthropogenic source of environmental degradation and habitat loss worldwide (Kutz, 2007). Surface mining (comprised of strip mining, open-pit mining and mountaintop-removal mining) accounts for over 80% of ore mined each year, and can lead to sudden and extensive land use change as it requires the removal of vegetation and overburden in order to access an ore deposit (Sonter et al. 2014; Ramani, 2012). Given that the extraction stage of mining typically lasts from 10 – 30 years, biodiversity within the development footprint of open pit mines often experiences ongoing habitat fragmentation and loss impacts (Newmont Mining Corporation, 2018). Even for highly mobile animals like bats, habitat loss can decrease roost availability, reduce foraging opportunities and destroy strategic flyways (Campbell et al. 1996; Bambini et al. 2011).

Although bats make up one fifth of extant mammal species, many populations are now in decline with over 21 percent of species classified as Threatened or Near Threatened (IUCN, 2019). Due to their long lifespans and low reproductive output, any mining activity which reduces habitat availability to bats could result in population declines and populations will be slow to recover (Voigt and Kingston, 2016). However, there has been little investigation of potential impacts of mining on bat populations (Barclay, 2014). In addition to habitat loss, the blasting, transportation and processing of materials during open-pit mining could result in acoustic disturbance effects (Manwar et al. 2016). For acoustic predators like bats, noise pollution could compromise foraging efficiency, and therefore reduce activity in the noisy areas (Senzaki et al. 2016). However, relatively little is known about the effects of anthropogenic noise on bats, although studies suggest that species using passive listening to locate prey or those echolocating at low frequencies

(<35kHz) are likely to experience greater disruption from anthropogenic noise (Schaub et al. 2008; Bunkley et al. 2015).

To ameliorate impacts of disturbance and habitat loss within mining landscapes, habitat restoration is increasingly being used as a tool to slow or prevent biodiversity loss (Burgar et al. 2015). In addition to carrying out habitat restoration following mine closure, environmental enhancement is now frequently implemented while mining is still underway (Tischew and Kirmer, 2007). For local bat populations this ecological compensation includes habitat creation or enhancement to provide strategic flyways and additional foraging resources and the provision of bat boxes to offer roosting opportunities (although this only applies to species which roost within tree cavities or buildings or exhibit roost plasticity) (Nielsen and Kelly, 2016; Mering and Chambers, 2014). However, despite the emphasis on promoting best practice methodologies through Environmental Impact Assessments (but see Richardson et al. 2019), research investigating the impacts of open pit mines on bat populations is lacking (Voigt and Kingston, 2016; Carvalho, 2017). Understanding the extent of potential landscape level effects on species richness and bat activity will enable better management recommendations and ultimately more effective conservation of bat populations (Gorresen and Willig, 2004).

Here, we exploited the opportunity to investigate the impacts of a recently re-established open pit mine on bat populations in the U.K. Although tungsten was discovered at the Drakelands site in Southwest England in 1867, historic mineral working only occurred from 1919-1920 and 1934-1944. Mineral working was later re-established in 2014, and Drakelands is now one of only two mines with an annual tungsten concentrate production capacity of more than 3,000 tonnes outside of China (Wolf Minerals Ltd, 2018). Chemical pollution at the site is strictly controlled, with local water sources regularly checked for leached contaminants. Further, light pollution is regulated across the site, with many areas remaining unlit, and light levels along boundary vegetation kept below 4 lux (Gillingham, 2014). The mine processing plant emits noise 24 hours per day, the high frequency portion of this ranging from 12-40 kHz (measured using an Anabat ultrasonic detector (SD2 model, Titley Scientific, Lancashire, United Kingdom)) which falls within the auditory range of European bat species (Luo et al. 2014). High frequency noise attenuates relatively rapidly in an outdoor environment, therefore the range of ultrasonic noise emitted from the processing plant is likely to be fairly limited (Makarewicz, 1998). Other than blasting, which occurs during the daylight hours but may still have some impact on bats roosting in proximity to the extraction pit, the mine processing plant represents that most significant source of noise locally, with background noise levels in the surrounding area remaining low (West Virginia Department of Environmental

Protection, 2006). Therefore, Drakelands represents an ideal study site to investigate the extent of potential impacts of habitat loss and noise on local bat populations at an open pit mine. In order to assess possible impacts, we used remote acoustic monitoring techniques to measure bat activity and species richness in relation to distance from the planning boundary of the mine site.

Study area

The study sampled the landscape surrounding Drakelands mine, in the county of Devon in Southwest Britain (8km²; 50°24'47"N, 4°00'58"W) (Fig. 1). The Drakelands area is largely composed of agricultural land, interspersed with patches of broadleaf woodland (20%) and a small proportion of conifer plantation (4%). The region has a maritime climate, with mean annual temperatures of 10.5°C and an annual rainfall of approximately 1000mm (ClimaTemps, 2017). Bordering Dartmoor National Park, Drakelands mine is situated adjacent to operating China clay mines and is located 2km from the nearest urban area. The extraction pit measures 850m x 450m, with the site footprint covering an area of approximately 4km² (Wolf Minerals Ltd, 2018). Development of the site required the removal of vegetation and destruction of three roosts (one tree roost, two building roosts) used by low numbers of common pipistelle *Pipistrellus pipistrellus* (n=5), soprano pipistelle *Pipistrellus pygmaeus* (n=5), greater horseshoe *Rhinolophus ferrumequinum* (n=7) and brown long-eared *Plecotus auritus* (n=3), carried out under European Protected Species Licences. In order to compensate for the loss of bat habitats, extensive mitigation measures have been implemented across the site including habitat creation, the strengthening of commuting routes by thickening hedges, installation of bat boxes and buildings, and a sensitive lighting regime (SLR consulting, 2013). The majority of these measures are contained within the area of the planning boundary.

Materials and methods

Field procedure

A series of 17 line transects was designed in order to evenly sample the landscape surrounding the Drakelands site (Fig. 2). Each transect consisted of 5 sampling locations, positioned at approximately 20, 300, 600, 900 and 1200 metres from the planning permission boundary of the mine. A maximum distance of 1200 metres was selected in order to minimise the potential confounding impacts of the nearby town on bat activity. The landscape to the north of the Drakelands site was not sampled, in order to avoid confounding impacts of near-by China clay pits. Acoustic monitoring was conducted at each transect point for a duration of five consecutive nights,

using one of five static Anabat ultrasonic detectors (SD1 and SD2 model, Titley Scientific, Lancashire, United Kingdom). Detectors were placed in weatherproof boxes and elevated to 1m above ground level on a tripod, tilted upwards at an angle of approximately 45°, and set to record from 30 minutes before sunset until sunrise. In order to account for potential differences in sensitivity between the detectors a randomised schedule was determined, so that each of the transect sampling points were measured equally by each of the five detectors. All five sampling points within each transect were monitored simultaneously, to avoid bias due to differing environmental conditions between locations. Transects were completed in a random order, with four transects sampled from September - October 2016 and the remaining transects sampled from March - May 2017; the selection of the sampling period was limited by funding constraints. The placement of each acoustic detector was recorded with a Garmin eTrex 10 GPS (Garmin Ltd, Southampton, United Kingdom), to determine the actual distance to the mine site. In order to investigate potential species differences in responses to the impacts of the mine resulting from different foraging ecologies, the proportion of woodland cover in the 100m buffer surrounding each monitoring point was calculated using the Centre for Ecology and Hydrology Land Cover Map 2015 (Rowland et al. 2017) and ArcGIS 10.2.2 (ESRI, 2011). Environmental conditions were recorded for each night of sampling using the Global Forecast System (total precipitation, average temperature and average wind speed), as these factors influence levels of bat activity (Fischer et al. 2009).

Data were visualised in AnalookW (Titley Scientific, Lancashire, United Kingdom), and echolocation calls were visually inspected and assigned to 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 assigned to *Myotis* and *Plecotus* were only identified to a genus level, as many of these calls are too similar for accurate species-level assignment (Kunz and Parsons, 2009). Echolocation calls for which confident identification was not possible were labelled 'unknown', this was largely due to calls being too faint, and occasionally because the parameters of a call fell between those of two species. 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 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 locations within a study site (Ober and Hayes, 2008).

Data analysis

Statistical analyses were conducted in R Studio using R version 3.3.1 (R Core Team, 2018) and the *ggplot2* package (Wickham, 2016) for graphics. 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). Generalised Linear Mixed Models (GLMM) were constructed using the package “lme4” (Bates et al. 2015) to investigate whether distance from the mine site had an effect on species richness and total activity. Species richness models were fit using GLMMs by Laplace Approximation with a Poisson error distribution, BOBYQA optimization and a “log” link function using the package “lme4” (Bates et al. 2015). Activity models were fit using GLMMs by maximum likelihood, with a zero-inflated negative binomial distribution and a “log” link function using the package “glmmTMB” (Brooks et al. 2017).

The global models contained the fixed effects of distance from the mine site (km), proportion of woodland cover, average wind speed (mph), total precipitation (mm), average temperature (°C) and an interaction between distance and woodland cover. All models contained the random effects of date (to account for temporal autocorrelation) and location (to account for pseudo-replication as multiple nights of monitoring were conducted at each location). Potential spatial autocorrelation was assessed for the variables of species richness and total activity using Moran’s I (Paradis et al. 2004). Spatial autocorrelation was only found to be significant in species richness (Moran’s I: 0.064; $p < 0.001$), therefore the variable of ‘Transect ID’ was included as a random effect in all species richness models to account for this. 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. Lunar phobia (changing foraging habitats or activity schedules as a result of moonlight) does not appear to affect behaviour in the six UK-resident bat species in which this phenomenon has been studied (Karlsson *et al.* 2002; Roeleke *et al.* 2018). As a result, we did not anticipate that moonlight intensity would significantly effect on bat activity in our study and this parameter was not included in our analyses. Models were built using all possible combinations of predictors using the package “MuMIn” (Barton, 2017) and then ranked using Akaike’s Information Criterion corrected for small sample size (AICc), which penalizes models with many explanatory variables (Burnham and Anderson, 2010). Each model was then given an Akaike weight ($\Delta AICc$), and based on the difference in these values, the best fitting models were arrived at. Models for which $\Delta AICc \leq 2$ were considered to

have equivalent support and were validated by visual examination of residuals and q-q plots (Bolker et al. 2009). According to some model selection methods, the model containing the fewest parameters of those with equivalent support, would be the optimal model(s) (Busemeyer et al. 2015). However, given that this is an ecological study with substantial noise, where more than one model received equivalent support the top-ranked models ($\Delta AICc \leq 2$) were averaged to create a 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, using the natural average method (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 species richness and total activity. In order to investigate potential non-linear trends between the species richness and distance from the mine site, we ran simple generalised additive models within each tercile of woodland cover separately, using the package “mgcv” with a gaussian error distribution and identity link function (Wood, 2011).

Results

A total of 13,610 bat passes were recorded during the study, and this represented at least 11 species (see Supplementary Table 1 for the detection frequency of each species/ genus). Data were not obtained for 14% of sampling locations, due to equipment failure and livestock interference. In investigating the factors that influence species richness, two of the models generated were considered to have equivalent support ($\Delta AICc \leq 2$) (Table 1). Following model averaging, confidence intervals indicated that distance to the mine site was an important predictor of species richness, with differing trends depending on the proportion of woodland cover (Supplementary Table 2). Within the lower and middle terciles of woodland cover species richness rose gradually with increasing distance from the mine site. However, within the upper tercile of woodland cover species richness increased more steeply with increasing distance (Fig. 3). Visual inspection of the generalised additive models reveals a non-linear trend within the upper tercile of woodland cover, with species richness rising sharply with increasing distance from the mine site, and plateauing at approximately 900m (Fig. 4). Other important predictors of species richness were precipitation, temperature and wind, with high levels of precipitation and wind correlating with lower richness, and high temperatures correlating with greater richness.

In investigating the factors that influence total bat activity, four of the models generated were considered to have equivalent support ($\Delta AICc \leq 2$) (Table 2). Following model averaging, confidence intervals indicated that distance to the mine site was an important predictor of total activity, with the level of activity increasing with distance (Supplementary Table 3; Fig. 5). Other important predictors of activity were precipitation, temperature and wind, with high levels of precipitation and wind correlating with lower activity, and high temperatures correlating with greater activity.

Discussion

An improved understanding of the impacts of open pit mines on bats is crucial for informed management strategies and to help maintain the social contract that exists between mining companies and society at large. There is currently very little information available on these impacts, and we partly address that paucity here. Using model averaging techniques we found that bat species richness and activity levels increased with distance from the Drakeland mine. The effect of distance from the mine site boundary on species richness depended on the woodland cover around the mine. Within areas of low and medium woodland cover species richness rose gradually with

increasing distance from the mine site. However, within areas of relatively high woodland cover (mean value of 62%) species richness rose more steeply, from 1.9 at the mine site boundary to 3.4 at a distance of 900m. Given that the majority of the mitigation measures that have been implemented are contained within the site boundary, the impacts of the mining development on bat activity and species richness in the surrounding landscape may be somewhat softened.

It is important to note that the interaction between species richness and woodland cover was not likely to be driven by differences in habitat quality among monitoring locations. The vast majority of monitoring points that were situated in low cover habitats were within grasslands that were used for the grazing of livestock, and not arable farming which is often associated with reduced bat activity and species richness (Vaughan et al. 1997a). All wooded areas were mature and predominantly broadleaf and furthermore, the wooded areas close to the mine site generally contained more water bodies, which should increase their quality for bat foraging (Vaughan, 1997b). The largest woodland patch was also located close to the mine site boundary, although size of woodland patch is not necessarily correlated with the level of bat richness it supports (Law et al. 1999; Estrada and Coates-Estrada, 2002). Thus, it seems distance from the mine and woodland cover are both important in determining species richness.

The noise emitted from the mine processing plant may contribute towards the relatively low species richness observed in woodland habitats, close to the site boundary. Studies suggest that by masking insect generated sounds, noise pollution may reduce the foraging efficiency of both *Plecotus auritus* and Bechstein's bat *Myotis bechsteinii*, which are thought to use passive listening to locate prey and are specialised to forage within woodland habitat (Anderson and Racey, 1993; Siemers and Swift, 2006; Schaub et al. 2008). Avoiding areas that are degraded by noise and shifting foraging activity further from the site boundary would contribute to the increased species richness further from the site boundary within areas with high woodland cover. A field trial of the impacts of broadband noise at gas compressor stations found some bat activity was reduced to a radius of at least 80m from the plant suggesting noise is important for some bats (Bunkley et al. 2015). However, sound attenuates with distance and we still saw effects up to 900m from the mine.

A more likely explanation for the patterns detected is that the decreased species richness close to the mine could be linked to the loss of foraging areas, roosting habitat and commuting routes within the site boundary, which would reduce movement of bats in the direction of the mine, generating a barrier effect (Fensome and Mathews, 2016). The susceptibility of bat species to barrier effects

varies in accordance with foraging ecology and wing morphology. Long-thin-winged species adapted to fly within low clutter habitats are thought to be less impacted by habitat loss barriers, compared with manoeuvrable species with broad, short wings which forage close to environmental clutter (Norberg and Rayner, 1987; Meyer et al. 2008). Furthermore, studies investigating the barrier effects of roads indicate species that forage in open spaces are willing to cross busy highways, whereas clutter specialists rarely do (Kerth and Melber, 2009; Berthinussen and Altringham, 2012a; Berthinussen and Altringham, 2012b). While we cannot conclusively state that activity decreases were driven by elevated impacts of the mine on clutter specialists, our findings are consistent with the loss of foraging habitats creating barrier effects, especially given the patterns identified (i.e. the interaction between woodland cover and distance). If our interpretation of the patterns detected is correct, effects could be further compounded by impacts on home range size. The short, broad wings of high-clutter specialists impose increased costs on commuting, and therefore the home range of these species tend to be smaller (Norberg and Rayner, 1987; Safi and Kerth, 2004). The relatively small area of habitat loss within the study site is therefore likely to have a greater impact at the local level on high clutter specialists, and a lesser impact on low clutter specialists with greater mobility which may be more affected by landscape level factors at large spatial scales (Fuentes-Montemayor et al. 2013).

Distance from the mine site boundary was also found to positively correlate with total nightly bat activity. As noted above, variation in habitat quality and noise pollution are unlikely to explain this pattern over the distances covered by our surveys. We suggest instead that habitat loss and commuting route disruption because of the mine site probably drives this effect. Consistent with this interpretation, large roads can act as barriers to bats with activity increasing by 23% 1,200 meters from the road (Berthinussen and Altringham, 2012a). Here we focussed on total bat activity rather than species-level activity, but total activity was dominated by *P. pipistrellus* (78.1% of calls). Given that *P. pipistrellus* is a generalist and may less disrupted by human activity and habitat loss, the activity decrease toward the mine site boundary may well be greater for some individual species, but testing this requires additional data (Davidson-Watts et al. 2006; Richardson et al. 2019).

Finally, and consistent with many other studies, we found that that high wind speeds and rainfall resulted in reduced activity generally, while high temperatures correlated with higher species richness and activity (Taylor, 1963; Reichard et al. 2010; Voigt et al. 2011; Barros et al. 2014). These patterns are thought to be due to temperature related shifts in insect activity, variation in the

energetic costs of thermoregulation, and sensory constraints on echolocation as a result of rainfall. Our study did not take place during the peak months of bat activity (June-August) (Russ *et al.* 2006), but despite this, shifts in activity and species richness were still evident. The maternity season occurs during seasonal peak-activity and this may have generated other patterns too. For example, during lactation, female bats often adopt a bimodal (or in some cases trimodal) activity pattern, returning to the maternity roost to suckle offspring during the night (Maier, 1992; Swift, 1980). This could result in elevated activity in the vicinity of maternity roosts, and, depending on the locations of the maternity roosts, potentially alter measured impacts of the mine on bat activity in the surrounding landscape.

To conclude, our findings suggest that open pit mining reduces the activity of some bat species nearby, with negative impacts on species richness potentially extending up to 900m from the mining activity. We also identified a threefold reduction in total bat activity close to the mine. These trends were observed despite the extensive mitigation measures that have been implemented across the site. To our knowledge, this is the first attempt to quantify impacts of open pit mines on bat populations in the wider landscape. At new open-pit 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 located within 1km of the mine site boundary. Additionally, 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 to better predict impacts in the wider landscape of new developments.

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Conflicts of interest None.

Ethical standards Permission for field work was granted by respective landowners. Bat activity was surveyed using passive acoustic monitoring and therefore bats were not affected in any way.

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Tables

TABLE 1 The two top-ranking species richness models for which $\Delta\text{AICc} \leq 2$, which were included in model averaging. Also shown is the null model, which does not contain distance or woodland cover. 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's weight (w_i).

Model name	Model rank	Structure of fixed effects (Response = species richness; random effects = date and location)	Dev	K	AICc	ΔAICc	w_i
Model 1	1	Distance + woodland cover + wind + precipitation + temperature	1052.9	9	1071.4	0.0	0.51
Model 2	2	Distance* woodland cover + wind + precipitation + temperature	1052.6	1 0	1073.2	1.8	0.21
Null	6	Temperature + precipitation + wind	1064.6	7	1078.9	7.5	0.01

TABLE 2 The four top-ranking total activity models for which $\Delta AICc \leq 2$, which were included in model averaging. Also shown is the null model, which does not contain distance or woodland cover. 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 ($\Delta AICc$) and the Akaike's weight (w_i).

Model name	Model rank	Response = Total activity; random effects = date and location	Dev	K	AICc	$\Delta AICc$	w_i
Model 3	1	Distance + precipitation + temperature + wind	2589.4	9	2608.0	0.0	0.25
Model 4	2	Distance + woodland cover + precipitation + temperature + wind	2587.5	10	2608.2	0.2	0.22
Model 5	3	Distance * woodland cover + precipitation + temperature + wind	2586.8	11	2609.7	1.7	0.10
Model 6	4	Distance + precipitation + temperature	2593.3	8	2609.7	1.7	0.10
Null	6	Temperature + precipitation + wind	2594.1	8	2610.6	3.2	0.07

Figure captions

FIG. 1 Map detailing the location of Drakelands mine in Southwest England, UK.

FIG. 2 Map detailing the 17 line transects, positioned at 20, 270, 520, 770, and 1020 metres from the Drakelands mine site boundary.

FIG. 3 The relationship between species richness and distance from the mine site, according to the proportion of woodland cover (predicted by Model 2, using averaged values of other fixed effects). Medians of each tercile of woodland cover are 0, 0.133, 0.576. As can be seen, more species were detected as sampling moved away from the mine site boundary.

FIG. 4 Bat species richness as a function of distance from the mine site boundary. Simple generalised additive models were used to investigate potential non-linear trends between species richness and distance from the mine site, run separately within each tercile of woodland cover. Here, predictions for the three separate models are displayed on one graph to allow easy comparison. Medians of each tercile of woodland cover are 0, 0.133, 0.576. In all models there was an increase in species richness to at least 1km from the mine site boundary.

FIG. 5 The relationship between total activity and distance from the mine site (predicted by Model 3, using averaged values of other fixed effects). As the distance from the mine site boundary increased, there was an increase in total bat activity (the total number of passes recorded).

Appendices

TABLE 1 Summary table for number of bat passes per species, and detection frequency.

Species	Number of passes	% of total passes	% of nights detected
<i>P. pipistrellus</i>	10,629	78.09	71.56
<i>P. pygmaeus</i>	1,753	12.88	47.40
<i>P. nathusii</i>	16	0.12	2.45
<i>R. ferrumequinum</i>	101	0.74	18.04
<i>R. hipposideros</i>	19	0.14	5.20
<i>B. barbastellus</i>	42	0.31	7.34
<i>N. noctula</i>	108	0.79	10.09
<i>N. leisleri</i>	12	0.09	2.75
<i>E. serotinus</i>	2	0.01	0.31
<i>Myotis</i> species	776	5.70	44.34
<i>Plecotus</i> species	20	0.15	5.20
Unidentified	132	0.97	20.18

TABLE 2 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 (5%, 95%).

Parameter	Estimate	SE	Confidence interval (5, 95)
(Intercept) †	0.571	0.092	0.419, 0.722
Distance (km)	0.220	0.106	0.046, 0.394
Proportion woodland	0.328	0.116	0.137, 0.519
Distance: proportion woodland	0.130	0.226	-0.243, 0.503
Precipitation	-0.580	0.162	-0.846, -0.313
Temperature	0.465	0.114	0.276, 0.653
Wind	-0.331	0.116	-0.523, -0.139

TABLE 3 Model-averaged parameter estimates: effects of each parameter on total activity in the area surrounding Drakelands mine. Shown is the model-averaged means (Estimate), associated standard error (SE) and confidence intervals (5%, 95%).

Parameter	Estimate	SE	Confidence interval (5, 95)
(Intercept) †	-1.487	1.018	-3.165, 0.192
Distance (km)	0.868	0.439	0.144, 1.593
Woodland cover	0.306	0.645	-0.702, 1.964
Distance: woodland cover	0.178	0.712	-1.2678, 3.578
Precipitation	-0.506	0.131	-0.722, -0.290
Temperature	0.444	0.099	0.281, 0.608
Wind	-0.049	0.034	-0.107, -0.009
ZI Intercept	-19.396	3051.995	-5054.175, 5015.383