

1 Quantifying the differences in avian attack rates on reptiles between  
2 an infrastructure and a control site

3 Cameron Law<sup>1</sup>, Lesley T Lancaster<sup>1</sup>, Jeanette Hall<sup>2</sup>, Sam Handy<sup>3</sup>, Mike Hinchliffe<sup>1</sup>, Clare O'Brien<sup>4</sup>,  
4 Katie O'Brien<sup>5</sup>, Sam Watts<sup>3</sup>, and David O'Brien<sup>2</sup>.

5 1. University of Aberdeen, School of Biological Sciences, AB24 2TZ, UK

6 2. Scottish Natural Heritage, IV3 5LQ, UK

7 3. Bangor University, LL57 2UW, UK

8 4. Woodlands, Brae of Kinkell, IV7 8HZ, UK

9 5. University of St Andrews School of Biology, KY16 9ST, UK

10 [Email address of corresponding author](#)

11 [David.obrien@nature.scot](mailto:David.obrien@nature.scot)

12 [ORCID numbers:](#)

13 [David O'Brien 0000-0001-7901-295X](#)

14 [Lesley Lancaster 0000-0002-3135-4835](#)

15 [Jeanette Hall 0000-0002-2694-8209](#)

16 [Abstract](#)

17 Increased demand for renewable energy has led to growth in the use of land for electricity  
18 generation and associated infrastructure. Land-based wind farms are amongst the commonest  
19 generators of renewable energy. To date most research on the effects of wind farms on wildlife have  
20 focussed on birds and bats, with very little work on terrestrial taxa. We hypothesised that widely  
21 reported negative effects of wind farms on predatory birds might benefit potential prey species. We  
22 focussed on reptiles due to concerns over worldwide declines in this group. We compared avian  
23 attack rates on clay model snakes at a site in Caithness UK within a wind farm relative to a control

24 site of the same topography and habitat class, 1 km away, using life-sized clay models of adder  
25 *Vipera berus*, a widespread but declining Palearctic species. Attack rates at the control site were  
26 comparable with similar studies elsewhere in Europe. However, we found that attack rates were  
27 lower within wind farm arrays, although several species of bird known to prey on reptiles were  
28 observed both within the wind farm and the control site. Therefore, given the high rate of loss to  
29 avian predators experienced in reptile reintroduction and reinforcement projects, wind farm sites  
30 may offer safe-havens, representing a neglected opportunity in reptile conservation. Grazing by  
31 sheep severely offset this benefit, presumably through removal of plant cover which was apparent in  
32 those areas of the wind farm where sheep were allowed access; grazing must thus be managed  
33 carefully for these benefits to be realised.

## 34 Introduction

35 Infrastructure is an important part of economic development and is sometimes seen as a key to  
36 addressing societal inequalities (UN Sustainable Development Goal 9 2019). Nevertheless, it can  
37 have unintended adverse impacts on wildlife, including through land take, habitat fragmentation,  
38 disturbance, pollution and direct mortality (Laurance 2018; IPBES 2019). In Europe, 50% of land is  
39 within 1.5km of transport infrastructure, leading to predictions of declines in populations of birds  
40 and mammals (Torres et al. 2016). Other studies have suggested that transport infrastructure can  
41 have a disproportionately large effect on reptiles and amphibians (reviewed in Andrews et al. 2015).  
42 There is also growing interest in the potential interactions between other forms of infrastructure  
43 and wildlife, such as that associated with renewable energy (e.g. Sánchez-Zapata et al. 2016).  
44 Along with reduction in energy consumption, renewable energy generation has been supported by  
45 national governments and international bodies as a strategy to combat climate change. For example,  
46 the European Union's Renewable Energy Directive sets a binding target of 20% final energy  
47 consumption from renewable sources by 2020 (EU 2009). Compared to hydrocarbon- or nuclear-  
48 based electricity generation, renewable energy technologies tend to require large areas of land

49 (Perrow 2017; Allison et al. 2019) and may include features potentially hazardous to wildlife (e.g.  
50 wind, wave and tidal turbines, and solar farms). All of these aspects of their infrastructure and  
51 operation can put them at odds with biodiversity and landscape conservation goals. Thus the  
52 benefits of renewable energy, in terms of climate change mitigation, need to be weighed carefully  
53 against infrastructure costs to habitat extent and quality (Sánchez-Zapata et al. 2016; IPBES 2019).

54 Land-based wind farms are amongst the least expensive and most technologically mature deliverers  
55 of renewable energy, resulting in a significant recent expansion of proposed and established wind  
56 farms (Pearce-Higgins et al. 2009). Wind farms can impact birds in two main ways: through direct  
57 mortality from collision with rotor blades and through displacement (Barrios and Rodriguez 2004;  
58 Drewitt and Langston 2006; Everaert and Stienen 2007; de Lucas et al. 2008). These impacts have  
59 been shown for breeding, migrating and over-wintering birds (reviewed in Hötcker 2017; de Lucas  
60 and Perrow 2017). Local population declines in both birds and bats have been partially attributed to  
61 collision mortality (Hunt and Hunt 2006; Perrow 2017). The development of wind farms may also  
62 have a cumulative effect on birds when superimposed onto already existing anthropogenic sources  
63 of disturbance. For long-lived species, with low reproductive rates, more than 0.5% additive  
64 mortality could lead to a considerable population-level impact (Whitfield et al. 2004; Everaert and  
65 Stienen 2007).

66 Displacement of wildlife by wind farms effectively amounts to habitat loss, and this can occur during  
67 both the construction and operational phases of wind farms (Hötcker 2017). Disturbance may be  
68 caused by visual intrusion, noise and vibration impacts, as well as personnel and vehicular  
69 movement for maintenance purposes (Drewitt and Langston 2006). Pearce-Higgins et al. (2009)  
70 found that there was a negative correlation between raptor abundance and proximity to wind  
71 turbine, with significant avoidance observed up to 500m and 200m from turbines observed for  
72 buzzards *Buteo buteo* and hen harriers *Circus cyaneus* respectively. Garvin et al. (2011) observed  
73 similar results, with a 47% reduction in raptor abundance after construction of a wind farm in North

74 America. Furthermore, the time since operation commencement can also have a significant effect on  
75 bird abundance; Stewart et al. (2005) found that the deleterious effect on bird abundance can  
76 persist or worsen over time. Despite the rapid expansion of onshore wind farms, their effects on  
77 wildlife other than birds and bats still remains largely unquantified (reviewed in Perrow 2017, and  
78 see Santos et al. 2010). Lovich and Ennen (2017) found only two published studies of the effects of  
79 wind farms on reptiles, although further papers have been published subsequently (Agha et al. 2017;  
80 Keehn and Feldman 2018; Thaker et al. 2018; Keehn et al. 2019).

81 Avian predation is typically one of the major sources of mortality within reptile populations (e.g.  
82 Andren 1982; Boarman 2003; Thaker et al. 2018) and, since the birds most affected by wind farms  
83 include predators, it seems likely that changes in bird density will have indirect effects on prey  
84 species. In analyses of raptor diets, reptiles occur frequently and are often an important alternative  
85 prey source to small mammals, especially in warmer climates (Martin and Lopez 1996; Reidar et al.  
86 2007; Tapia et al. 2007; Steen et al. 2011), but also in the UK (Cramp and Simmons 1979; Graham et  
87 al. 1995). Predation has an important regulatory role on the abundance of the reptile species; for  
88 example, Anderson and Burgin (2008) reported a strong negative correlation between skink  
89 numbers and predatory bird presence. The converse is also true in population dynamics of reptiles; a  
90 recent meta-analysis has shown that predator removal can induce a 1.6x fold increase of prey  
91 populations (Salo et al. 2010). Furthermore, Buckley and Jetz (2007) found that abundance of island  
92 reptile populations was correlated with lower rates of predation from raptors, and of competition  
93 from insectivorous birds. It might therefore be that phenomena which reduce predatory bird  
94 numbers may lead to an increase in reptile numbers (Agha et al. 2015). Thaker et al. (2018) found  
95 that raptor abundance and predation attempts were lower at sites with wind turbines, and the  
96 population density of the superb fan-throated lizard *Sarada superba* was significantly higher, than at  
97 control sites. Levels of stress-induced corticosterone was lower in lizards on wind farm sites, but so  
98 was body condition (linked to increased density-dependent competition) suggesting that the effects  
99 of predator release are not necessarily wholly positive (Thaker et al. 2018).

100 Reviews suggest that reptiles are subject to global decline, although there is still much to be learned,  
101 with the majority of species considered data deficient by IUCN (Gibbons et al. 2000; Hilton-Taylor et  
102 al. 2004; Meiri et al. 2018). In Europe, the decline in reptile populations resembles the global  
103 position; Reading et al. (2010) found that European snake populations had declined sharply since the  
104 early 1990s, with surveyed populations recording a mean decline of 86.7%. The UK national survey  
105 found adder *Vipera berus* (the only snake present across the whole of the British mainland) in only  
106 7% of squares surveyed and, while there was no baseline against which to make a comparison, this  
107 low occupancy rate may have been a result of the theorised species' decline (Wilkinson and Arnell  
108 2013). There is no clear evidence of decline of the UK's other widespread reptiles (Wilkinson and  
109 Arnell 2013), and the adder is now a conservation priority in the UK (JNCC 2010). Raptors and corvids  
110 are known to be important predators of adders (e.g. Prestt 1971; Beebee and Griffiths 2000) and,  
111 although unlikely to be a cause of their decline, avian predation has been shown to limit  
112 recruitment, and hence to potentially limit recovery following perturbations, in another *Vipera*  
113 species (Halpern et al. 2017).

114 In Scotland, wind farms are increasingly prevalent in habitats used by all native reptiles (the common  
115 lizard, *Zootoca vivipara*; slow-worm, *Anguis fragilis*, and the common European adder, *Vipera berus*)  
116 (NBN 2018), and these wide-spread species provide models for research into the impacts of wind  
117 farms. We hypothesised that avian predation pressure would be lower in wind farms than  
118 neighbouring land, due to established patterns of wind farm avoidance by these predators. Although  
119 not part of the original study design, unauthorised sheep grazing within the turbine array during the  
120 project gave us the opportunity to study combined impacts of anthropogenic land use changes on  
121 attack rates in the context of wind farms. Adders' low occupancy rates (Beebee and Griffiths 2000;  
122 Wilkinson and Arnell 2013), coupled with putative low population densities and high detection  
123 difficulties, make quantifying any effects of wind farms on adders challenging. To quantify the  
124 relative frequency of avian attacks, we used clay models, a well-established means of estimating  
125 avian predation pressure (Wüster et al. 2004; Webb and Whiting 2005; Bateman et al. 2017).

126

## 127 Materials and methods

### 128 Study area and experimental design

129 Fieldwork was carried out at Camster wind farm (constructed in 2011, 25 turbines (Vestas V80), max  
130 blade tip height 120m) and the adjacent Flows of Leanas, Caithness UK (58° 24' N; 3° 16' W, altitude  
131 80-100m asl). This site was selected for its low topographic variation and uniform blanket bog  
132 habitat (dominated by bog moss *Sphagnum* spp; common heather *Calluna vulgaris*; cross-leaved  
133 heath *Erica tetralix* and sedges *Carex* spp, EUNIS (European nature information system) habitat type  
134 D1.2), thus reducing the effect of variation in habitat type and topography. Ecologists employed at  
135 the site were interviewed to confirm that reptiles and potential avian predators were present.  
136 Common buzzard *Buteo buteo*, common kestrel *Falco tinnunculus*, hen harrier *Circus cyaneus*,  
137 carrion/hooded crow *Corvus corone* and northern raven *Corvus corax* were present and all were  
138 common (seen by the authors on every visit), except hen harrier, which was seen often but not  
139 every time.

140 Models were constructed from terracotta non-toxic polymer clay (Sculpey® Original) (Fig. 1).  
141 Polymer clay remains soft enough that it retains beak and talon marks when out in the field, allowing  
142 surveyors to easily distinguish between avian and mammalian attacks (Bittner 2003). Terracotta clay  
143 was chosen because the colour resembles that of female adders to the human eye; whilst it may  
144 appear different to birds, this does not affect our results, as we are not considering overall predation  
145 pressure, but relative predation rates between treatments. The models used approximately 80g of  
146 clay, and measured approximately 20cm in length and 1cm in diameter, which is within the size  
147 range of a yearling adder, the life-stage most vulnerable to predation (Prestt 1971). The models  
148 were constructed in a 'S' shape and had a tapering tail end with a slightly enlarged head distinct  
149 from the thinner neck to make them as realistic as practical (Fig. 1, following Wüster et al. 2004).

150 A total of 100 clay models were used, with 10 models per transect divided evenly between five  
151 transects within the wind farm array and five within the control site. There were four site visits per  
152 transect to examine the models in 2015 (between 14 May and 26 May), and four in 2016 (between 6  
153 August and 2 October), giving a maximum 400 recording events per year. In practice, whereas the  
154 total in 2015 was 400, it was 337 in 2016; due to the dense vegetation, several models could not be  
155 found during one or more surveys. Due to an extreme weather event in 2016, the controls were  
156 only surveyed three times rather than four. Transects were 90m in length, with models placed within  
157 5m either side of the transect line at 10m intervals (Fig. 2). All transects within the treatment area  
158 were within 200m of at least one turbine. To ensure that nearby wind turbines did not affect the  
159 chances of models being attacked within the control site, control transects were placed at least  
160 1000m from the closest turbine, i.e. at a greater distance than that over which avoidance behaviour  
161 had been observed for locally-occurring raptor species (Pearce-Higgins et al. 2009). Site selection for  
162 transects was determined by the surveyor who looked for patches of moorland habitat of a  
163 consistent type to the other transects, i.e. flat areas with short vegetation (<30cm), 100% cover, and  
164 similar structural heterogeneity, particularly the proportions of graminoids and ericaceous plants  
165 (Fig. 3). Models were placed in areas of partial cover similar to typical adder basking sites (Prestt  
166 1971). In 2015, after transect selection and model placement, but before survey took place, sheep  
167 were unexpectedly introduced to the areas occupied by transects 4 and 5 within the wind farm  
168 arrays (TT4 and TT5), while the other wind farm and control sites were fenced, thus excluding sheep  
169 (See lower boundary feature on Fig. 2). No models were trampled, but the presence of sheep  
170 markedly changed vegetation structure, reducing vegetation height and density, and leaving patches  
171 of bare ground. Rather than discard these transects, we considered how sheep grazing may  
172 influence the effects of wind farms on reptile attack rates. TT4 was so badly damaged by trampling  
173 that it was replaced by a new, grazing-free transect (TT6) the following year.

174 The models were left and re-examined after a period of between three and 34 days. In each case,  
175 wind farm and control visits were made on the same day, so that exposure periods, although

176 somewhat variable, were matched across plots. Beak and talon marks were recorded, and were  
177 unambiguous (Fig. 1) and easily distinguished from mammal damage, which was also recorded. If  
178 multiple marks were observed on a model, it was only counted as one attack, since it would be  
179 impossible to know whether more than one attack had taken place. After examination, models were  
180 repaired and returned to their original position.

## 181 Statistical analyses

182 To test the significance of our observations, two different statistical approaches were used. Firstly,  
183 General Linear Mixed Effects Models (GLMMs) using lme4 (Bates et al. 2014) and lmerTest  
184 (Kuznetsova et al. 2016) packages for R version 3.4.2 (R core team 2017) were used to test if there  
185 was a significant difference between attack probability on the wind farm vs. the control site  
186 transects. For this, individual attacks for each model were coded as a binomial response variable,  
187 where for each observation interval, an attack event was scored as (0,1) where 0 = non-attacked and  
188 1= attacked. Fixed explanatory factors were included for wind farm vs. control, sheep presence,  
189 observation interval, and year. Random effects were included to control for individual model snake  
190 (where there were 3-4 observation events per model), transect, and observation date. AICc was used  
191 to choose the best combination of fixed effects (Mazerolle 2015). Model effects were visualised  
192 using the Effects package for R. Second, a partial Mantel test was used to examine the correlation  
193 between the model attack rate matrix and geographic location of transects: each location's attack  
194 rate was calculated as the number of attacks observed in each transect divided by the number of  
195 observations; differences in attack rates among transects were correlated with geographical  
196 distance using the Ecodist package for R (Goslee and Urban 2007).

197

## 198 Results

199 Avian attacks were best explained by a model which included a significant negative effect of wind  
200 farms on attack probability (effect =  $-1.08 \pm 0.49$  s.e.,  $z = -5.53$ ,  $P = 0.04$ ); a significant positive effect of



201 sheep grazing (effect =  $1.64 \pm 0.56$ ,  $z = 2.93$ ,  $P = 0.003$ ). A slightly worse fitting model ( $\Delta AIC = 1$ ) also  
202 included non-significant effects of sampling year (effect =  $-0.50 \pm 0.74$ ,  $z = -0.678$ ,  $P = 0.50$ ) and  
203 observation interval (effect =  $-0.02 \pm 0.03$ ,  $z = -0.77$ ,  $P = 0.44$ ) (Fig. 4). In the absence of correcting for  
204 the effect of sheep grazing in some parts of the wind farm (there was no grazing on any of the  
205 control sites), we were unable to detect the protective effect of wind farms on attack probabilities,  
206 and model fit was significantly worsened ( $\Delta AIC = 22$ ). However, the effect of turbines was also  
207 extremely important in the model, and dropping this term resulted in a  $\Delta AIC$  of 17. This suggests  
208 that grazing on wind farms can severely offset the ecological benefits to reptiles, in terms of  
209 protection from avian attacks. Overall, the average per-observation attack rate over all models was 5  
210 out of 270 (0.018 probability) under the wind farm array in the absence of grazing, and was 12 out of  
211 120 (0.10 probability) under the wind farm where grazing was present. In control sites, none of  
212 which were grazed, attack rates were 20 out of 347 observation intervals (0.058). There was no  
213 spatial signal in attack rates among transects, with a Mantel's  $r$  of  $-0.31$ ,  $P = 0.98$ .

214 Although adders were not found during the study, reliable local observers reported finding them  
215 nearby, and weather conditions during our field visits were seldom conducive to finding this  
216 secretive species. Common lizards *Zootoca vivipara* were seen on the site, with the discovery of a  
217 neonate in the midst of turbine array near transect TT6 confirming breeding. At least one model was  
218 attacked by a small mustelid (least weasel, *Mustela nivalis*; stoat, *M. erminea*; ferret, *M. furo* and  
219 European pine marten, *Martes martes* all occur locally, Scott, 2011), and red fox *Vulpes vulpes* was  
220 observed on one occasion at the wind farm.

221

## 222 Discussion

223 Our hypothesis that avian attack rates would be lower within wind turbine arrays was supported,  
224 but only once the confounding factor of herbivore grazing was taken into account. Attack rate at the  
225 control site (6.4%) was similar to Wüster et al. (2004), who found a rate of 6.5% in a large-scale study

226 looking at 12 sites across Britain, and not dissimilar to studies elsewhere in Europe (9% in Spain,  
227 Valkonen et al. 2011) and in North America (7% to 13% Brodie 1993; Bittner 2003; Farallo and  
228 Forstner 2012). This suggests that our control site had a predation pressure similar to other areas.  
229 Although potential avian predators were observed amongst the turbine array during this study,  
230 previous studies have suggested that birds are less likely to actively hunt in the wind farm and are  
231 more likely to simply pass through (Perrow 2017) and that densities are likely to be lower (Pearce-  
232 Higgins et al. 2009) than outside the array. Any reduction in predation rates, particularly of  
233 immature *V. berus*, might reasonably be assumed to lead to higher numbers of breeding individuals.  
234 Survival rates of immatures to breeding age (three or four years old) is typically under 10%, whereas  
235 adult annual survival for this species has been estimated at between 77% and 80% (Prestt 1971;  
236 Neumeyer 1987). Reduced avian predation may also lead to increased opportunities to bask, and  
237 hence improved body condition in this ectothermic species; Thaker et al. (2018), working on the  
238 superb fan-throated lizard, found lower stress-induced levels of circulating corticosterone, and lower  
239 flight initiation distances within the array. They also found that the benefits of predator release  
240 were partly offset by density dependent competition, and there may be increased incidence of  
241 parasitism or disease at higher population densities. However, windfarms in Scotland do seem to  
242 offer an opportunity to maintain higher population levels of adders, particularly give the abundance  
243 of its main prey species short-tailed vole *Microtus agrestis* in the region; indeed, raptors are also  
244 major predators of voles, so competition for food may be somewhat reduced within the array (Harris  
245 and Yalden 2008).

246 We also found that areas heavily grazed by sheep showed higher attack rates. Sheep both eat cover  
247 plants and churn up the peaty soil leaving bare patches of earth. This leads to lower amounts of  
248 cover which makes the models, and presumably reptiles themselves, more obvious to visual  
249 predators. The impact of herbivores on heathland is complex, with some authors advocating low  
250 density sheep stocking to enhance biodiversity (e.g. Evans et al. 2006) but, in our study site, stocking  
251 levels on the two transects where sheep were present seemed to be high enough to damage the

252 habitat structure necessary to provide reptile cover habitat. Studies elsewhere on the impacts of  
253 sheep grazing on reptiles have also found complex interactions which can vary with habitat type (e.g.  
254 Rotem et al. 2016), intensity and timing (Val et al. 2019). Lindenmayer et al. (2018) found grazing  
255 was linked to reductions in reptile species richness, leading them to recommend that sheep grazing  
256 should be limited or prevented in sites where vegetation restoration for biodiversity conservation  
257 was a goal. The documented negative relationship (Wheeler 2008) between sheep numbers and  
258 abundance of short-tailed vole is a further reason to exclude them from such sites.

259 Whilst our study focussed on predation impacts, there are likely other features of wind farms that  
260 will affect reptile populations. In areas of low solar incidence like northern Scotland, good basking  
261 sites are of great importance to reptiles (McInerney and Minting 2016) and power company workers  
262 and hikers reported that site roads and concrete turbine bases are used by adders and common  
263 lizards for this purpose. Other studies have highlighted the risk of mortality from traffic near  
264 windfarm infrastructure (Lovich et al. 2011). However, at this site, as with others in Scotland,  
265 vehicular access is limited to power company staff and biological surveyors, and site speed limits are  
266 rigorously enforced, although cyclists and walkers have full access (as they do across rural Scotland).  
267 Traffic was very infrequent during our study and, whilst reptiles tend to bask near cover at the road  
268 edge, we found no evidence of roadkill during our visits to this or other sites.

269 While we are unaware of any work specifically on wind farms, there is an extensive literature on how  
270 both naturally and anthropogenically derived acoustic disturbances effect reptile behavioural  
271 ecology (Avery 1993; Randall and Matocq 1997; Burger 1998; Young and Aguiar 2002; Young 2003).  
272 Indeed, Rabin et al. (2006) found that the acoustics of wind farms had a deleterious impact on  
273 Californian ground squirrel *Spermophilus beecheyi* living beneath turbines by altering antipredator  
274 behavioural responses, though this was at least in part due to animals being unable to hear calls  
275 from conspecifics (Rabin et al. 2006). Such acoustic stresses, and particularly low-frequency sound,  
276 are likely to be greatest during the construction phase (Lovich and Ennen 2017) which may lead to

277 temporary displacement. Further work is required to understand if such acoustic (including ground  
278 vibration) disruption by wind farms is relevant in reptiles which may rely on vibrations to detect  
279 terrestrial predators.

280 It has been hypothesised that carcasses of birds and bats killed by turbines might attract  
281 mesocarnivores and that these may prey-switch to reptiles (Lovich et al. 2014). However, Agha et al.  
282 (2017) found fewer mesocarnivore visits to tortoise burrows closer to wind turbines, but more visits  
283 closer to roads within the wind farms. The small number of models attacked by mammalian  
284 carnivores, presumably mustelids, in our study does not illuminate this issue; mammals are more  
285 reliant on olfactory cues and as such may be less likely to be fooled by unscented clay models  
286 (Hansen et al. 2019). Analysis of mammal scats within and outside wind farms may provide a useful  
287 insight.

288 We must be cautious about extrapolating from a study limited to a single site and habitat type over  
289 two years. Studies at other sites and habitat types would be revealing, particularly if carried out in  
290 regions and habitats known to support species of conservation importance, such as steppe grassland  
291 (e.g. meadow viper *Vipera ursinii*) or desert, although work on the desert tortoise *Gopherus agassizii*  
292 by Lovich and Daniels (2000) has already shown that wind-farm development can be compatible  
293 with conservation of this sensitive species. The type of avian predator may also be important. The  
294 main avian potential predators at our study site find prey by soaring (common buzzard *Buteo buteo*),  
295 hovering (common kestrel *Falco tinnunculus*) or slow flying (hen harrier *Circus cyaneus*), making  
296 them susceptible to negative effects from wind farms (Perrow 2017). The locally-present corvid  
297 species rarely prey on live reptiles, though they have been recorded taking carrion such as road-kill  
298 (Cramp and Perrins 1994). Some other reptile specialist predators such as short-toed eagles  
299 *Circaetus gallicus* have been found to be disproportionately vulnerable to collision with turbine  
300 blades (Ferrer et al. 2012). Other important reptile predators not found in northern Scotland, such as  
301 shrikes (Laniidae) which typically hunt from perches near the ground, or roadrunners *Geococcyx* spp,

302 secretary birds *Sagittarius serpentarius* or storks (Ciconiidae) which hunt on the ground itself, would  
303 be less likely to be affected by wind farms (Hötker and Dürr 2017). A similar study by Keehn and  
304 Feldman (2018) on side-blotched lizards *Uta stansburiana* did not find a significant difference  
305 between clay model attack rates at wind farms and control sites. The target area of their study did,  
306 however, contain far more ground-based visual predators that may be affected by wind turbines  
307 compared to soaring and hovering birds which are the main predators of reptiles in the biome we  
308 studied.

309 The use of clay models does have some limitations, which are usefully reviewed by Bateman et al.  
310 (2017). However, we do not think that these will have had an important impact on the results of this  
311 study. A major issue can be confidently attributing attacks to predators, but we found that the signs  
312 of avian attacks were unambiguous, having the appearance of attacks on genuine reptiles (Fig. 2).  
313 Bill and talon marks on the head and neck of the models could be clearly seen and, in some cases,  
314 the head was completely removed. It is also possible that animals might interact with clay in itself,  
315 and this could be relevant for rodents, which might perceive the substance as a food source; eight  
316 incidences of nibbling by rodents were found in 2016. However, we did not analyse data on rodents  
317 as they do not prey on adders, so this is not relevant to our study. Other damage comprised sharp  
318 cuts in the clay, apparently caused by wind-blown grasses or sedges; these were totally unlike marks  
319 of predation. Other concerns relate to the accuracy of the models' appearance, smell and  
320 behaviour. Bateman et al. (2017) note that the visual acuity of birds is different from that of  
321 humans, which is likely to affect predatory behaviour. However, since birds in our study attacked  
322 the models in the same way that they would attack real snakes, we can have some confidence that  
323 they are sufficiently realistic. As stated in the methods, we were looking at comparative attack rates  
324 across the two habitats, rather than overall attack rate, reducing the risk of any bias. Some studies  
325 have used cameras to observe predator/ model interaction; this would have needed one camera per  
326 model, making it prohibitively expensive. Despite the potential limitations, our method has the  
327 advantage of simplicity; surveyors need not be extensively trained, and materials are low cost. The

328 widespread use of clay models to assess predation risk also means that a researcher has a range of  
329 benchmark studies available for comparison.

### 330 Conclusion / Applications

331 As infrastructure projects are using increasing areas of land (IPBES 2019), there is growing interest in  
332 the potential value of novel habitats for conservation (e.g. Maclagan et al. 2018). Our work suggests  
333 that, where the habitat is suitable, wind farms might provide refuges for reptiles from predation.  
334 This clearly does not apply to all reptile species, e.g. forest species will lose habitat where trees are  
335 removed to improve wind-flow, and disturbance-intolerant species will presumably take time to  
336 recolonise after the construction phase. In some cases, road infrastructure may increase access by  
337 recreational users, potentially heightening the risk of persecution or illegal collection. Whilst wind  
338 farms may be seen as problematic for some taxa, there may be opportunities to use them to support  
339 conservation interventions in other cases. For example, since reptile reintroduction programmes  
340 typically face high loss to avian attack (e.g. *Vipera ursinii*, Halpern et al. 2017 and see Halpern 2014),  
341 wind farms may have potential as nuclei for such reintroduction and reinforcement projects. We  
342 propose that wind farms may, indeed, have some advantages over nature reserves, more  
343 traditionally considered for reintroduction projects, as many avian predators are also of  
344 conservation concern and populations are often high in protected areas (McClure et al. 2018).  
345 Creation of artificial hibernacula and basking sites by developers, for example using stones and  
346 cleared vegetation from the construction process, may augment the suitability of wind-farm  
347 infrastructure for reptiles at negligible cost. To maximise the benefits, it will be essential to manage  
348 grazing carefully in all cases. Appropriate levels of grazing will stop succession to woodland, which is  
349 important for the operation of the wind farm and for keeping open habitat suitable for many reptile  
350 species (McInerny and Minting 2016), but excessive grazing prevents establishment of cover plants  
351 essential for reptiles and other small and medium-sized animals (Evans et al. 2006). Control of  
352 invasive species, both non-native (e.g. *Rhododendron ponticum*) and native (e.g. gorse *Ulex*

353 *europaea*) may be necessary; care should be taken to avoid inadvertently bringing non-native  
354 species into the site on construction equipment.

### 355 Acknowledgements

356 We would like to thank Professor Bob Furness for his comments on an earlier draft of this  
357 manuscript, and the staff of Camster Wind Farm for their cooperation. Two anonymous reviewers  
358 provided comments that greatly improved the text.

359

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550

551 **Figures**

552

553 **Fig. 1** Model adder (*Vipera berus*) *in situ*, showing bill and talon marks on head and neck where it has  
554 been attacked by a bird of prey. Constructed from terracotta non-toxic polymer clay (Sculpey®  
555 Original)

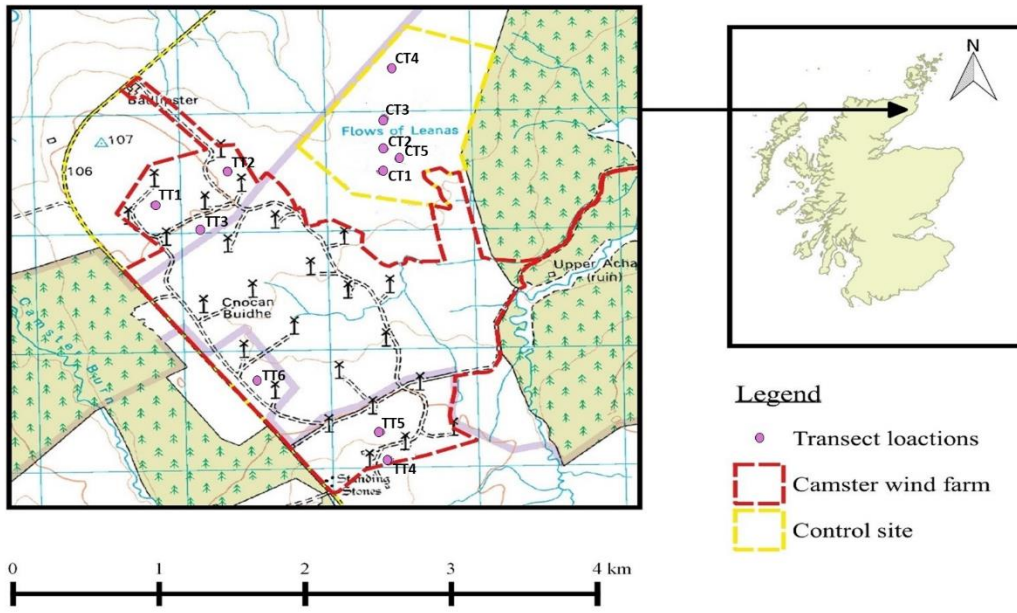


556

557 **Fig. 2** Site map showing locations of transects on wind farm (TT1-TT6) at Camster, Caithness, UK and  
558 control (CT1-CT5) sites at the neighbouring Flow of Leanas (58° 24' N; 3° 16' W). Smaller map shows  
559 location of study area within Scotland. TT1 to TT5 were surveyed between 14 May and 26 May 2015,  
560 and TT1-3, TT5 and TT6 between 6 August and 2 October 2016. CT1-5 were all surveyed in both  
561 years. TT4 was very heavily damaged by sheep, which had been introduced in 2015, and was thus  
562 replaced with TT6 for the 2016 survey.

563 *Contains OS data © Crown copyright and database right (2019).*





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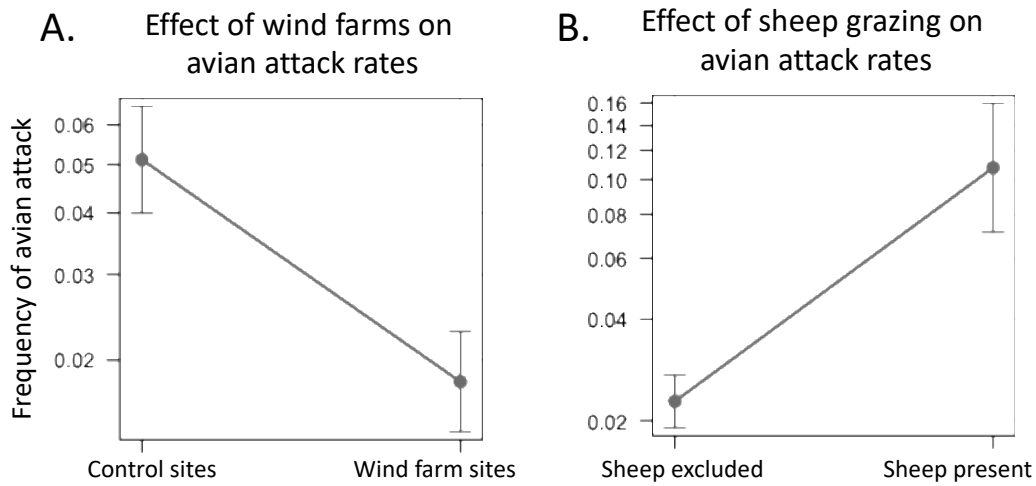
565

566 **Fig. 3** View of typical moorland habitat (blanket bog dominated by *Sphagnum* spp, *Calluna vulgaris*,  
 567 *Erica tetralix* and *Carex* spp, EUNIS code D1.2), with windfarm in background. Taken from control  
 568 site CT1 (See Fig 2), looking south-west.



569

570 **Fig. 4** Per model avian attack rates showing effect of (a) windfarm and (b) sheep-grazing. The  
571 average per-observation attack rate over all models was 5 out of 270 (0.019 probability) under the  
572 wind farm array in the absence of grazing, and 12 out of 120 (0.100 probability) under the wind farm  
573 where grazing was present. In control sites, none of which were grazed, average attack rates were  
574 20 out of 347 observation intervals (0.058).



575

576