

1 Accepted refereed manuscript of:

2

3 Kirkpatrick L, Oldfield IF & Park K (2017) Responses of bats to clear fell
4 harvesting in Sitka Spruce plantations, and implications for wind turbine
5 installation, *Forest Ecology and Management*, 395, pp. 1-8.

6

7 DOI: [10.1016/j.foreco.2017.03.033](https://doi.org/10.1016/j.foreco.2017.03.033)

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15 Responses of bats to clear fell harvesting in Sitka Spruce plantations, and implications for wind
16 turbine installation

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26 **Keywords: Coniferous plantation, management, biodiversity, Chiroptera, bats, wind turbines, clear**
27 **fell harvesting**

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30 Highlights:

- 31 • Keyhole felling for turbine installation is common in upland plantations
- 32 • Despite evidence that bats make use of plantations, the impacts of felling are unknown
- 33 • We found that bat activity either increased or stayed the same post harvesting
- 34 • Species-specific increases in activity were greatest in small/recently felled stands
- 35 • Siting wind turbines in conifer plantations may result in increased bat mortality

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46 **Abstract**

47 Commercial coniferous plantations are often assumed to be poor habitats for bats. As a result, the
48 impact of forest management practices on bats, such as clear felling, has received little attention,
49 particularly in Europe. However, there is growing evidence from multiple regions that bats do make
50 use of plantation landscapes, and as interest in siting onshore wind turbines in upland conifer
51 plantations grows, there is an urgent need to examine whether felling prior to turbine installation is
52 likely to put foraging bats at risk of collision. In the first study of its kind, we use a “before – after –
53 control - impact” study to explore the short-term impacts of clear fell harvest on bat activity in
54 commercial plantations. Thirty-one mature stands of Sitka Spruce were surveyed using acoustic
55 detectors in three large, upland Sitka Spruce plantations in Britain. Eleven stands were felled
56 between 2013 and 2015, and 26 of the original 31 stands were resurveyed in 2015. The change in
57 total bat activity and species- or genus-specific bat activity was modelled before and after felling
58 occurred at both felled and control stands using generalised linear models. There was no change in
59 overall bat activity at felled sites compared to control sites, but activity of *Nyctalus* species was 23
60 times higher following felling. Total *Pipistrellus spp.* activity doubled at felled sites post-harvesting,
61 although this was mainly driven by increased activity at a few felled sites. When *P. pygmaeus* and *P.*
62 *pipistrellus* were considered separately, activity increased slightly but non-significantly. The size of
63 the felled area influenced activity (for bats overall and *Pipistrellus spp.*), with 90% higher activity in
64 smaller felled stands (less than 5ha⁻¹) compared to larger felled stands (greater than 30ha⁻¹). For *P.*
65 *pipistrellus*, activity in felled areas decreased with the duration since harvesting; the greatest activity
66 occurred in stands felled within two months compared to those harvested more than 16 months
67 previously. Higher activity for some groups following felling may occur due to the creation of more
68 edge habitat, which is preferred by both *Pipistrellus* species we recorded. An increase in activity
69 following the small-scale felling (‘key-holing’) required for the installation of turbines could put
70 foraging bats at risk from collisions with turbines. Further investigation of the influence of both size
71 of clear fell patch, timing of felling and changes in invertebrate abundance due to felling are
72 required to establish the potential risk of key-holing and turbine installation to foraging bats.

73 **1 Introduction:**

74 Large scale clear felling is a widely used form of timber extraction in commercial forests which has
75 been heavily criticised for its perceived impacts, particularly on forest dependent flora and fauna
76 (Borkin and Parsons, 2014; Lindenmayer et al., 2006). However, there is little consistency in the
77 literature about the impacts of clear felling on biodiversity, with responses being highly taxa specific.
78 Felling may negatively affect organisms by isolating populations, decreasing resources, increasing
79 predation or changing climatic conditions (Grindal and Brigham, 1998). For example, forest

80 specialists may be negatively affected due to increased predation risk from the lack of cover (e.g.
81 arboreal sciurids; Fisher 2005). However, felling may be beneficial for open and edge adapted
82 species, particularly successional species, which respond positively to the changes in vegetation
83 structure and composition caused by harvesting and preferentially use clear-cuts (e.g. some early
84 successional birds; Loeb and O’Keefe, 2011; Oxbrough et al., 2010; Paquet et al., 2006). Eycott et al.
85 (2006) found that plant species richness is 60% lower in stands with full canopy closure and
86 increased in the first few years post-harvest as regeneration occurs. Regeneration may be positive
87 for some invertebrate taxa (Lin et al., 2006), but less so for canopy specialists (Humphrey et al.,
88 2003). Typically, generalist and open specialist species appear to benefit from clear felling, while
89 forest specialist abundance and diversity decreases (Humphrey et al., 2003; Ohsawa and
90 Shimokawa, 2011; Oxbrough et al., 2010). Therefore, while species richness may not necessarily
91 change in response to felling pressure, community composition can be altered.

92 Many habitat selection studies have found that bats avoid commercial coniferous plantations
93 (Boughey et al., 2011; Jones et al., 2003; Smith and Racey, 2008; Walsh et al., 1996), which is often
94 attributed to low invertebrate density and increased structural complexity, amongst other factors
95 (Haupt et al., 2006; Russo and Jones, 2003; Smith and Racey, 2008). However, often these studies
96 are carried out in areas of extremely low conifer cover (e.g. conifer cover of less than 3%, Davidson-
97 Watts and Jones, 2005; Davidson-Watts et al., 2006) and there is growing evidence that certain bat
98 species are able to make use of intensively managed non-native plantations in landscapes
99 dominated by plantations (Charbonnier et al., 2016; Cistrone et al., 2015; Kirkpatrick et al., 2017;
100 Mortimer, 2006; Russo et al., 2010). While the impacts of logging forests have been investigated for
101 a number of different bat species worldwide (e.g New Zealand: Borkin and Parsons, 2010a, 2010b;
102 USA: Grindal and Brigham, 1998; Australia: Law and Law, 2011), much of the previous work has
103 concentrated primarily on old growth or native forests (Dodd et al., 2012; Grindal and Brigham,
104 1998; Loeb et al., 2006; Loeb and O’Keefe, 2011; Menzel et al., 2002; Patriquin and Barclay, 2003).
105 The impact of forest management practices in non-native commercial plantations has received far
106 less attention. Research that does exist has focused on the impacts management may have on forest
107 specialist bats which rely on tree roosts for much of their life cycle (Borkin et al., 2011; Borkin and
108 Parsons, 2014). For bat species which are adept at using anthropogenically-disturbed habitats and
109 rely on building roosts rather than tree roosts, commercial coniferous plantations may be a
110 landscape which they can exploit (Kirkpatrick et al., 2017).

111 Bats with home ranges dominated by plantation forests, are likely to come into contact with felling
112 operations (Borkin and Parsons, 2011). Features such as standing dead wood, snags, tree damage
113 such as double leaders, and peeling bark all form key roosting habitats for bats and other taxa

114 (Altringham et al., 1996; Arnett, 2007; Russo et al., 2010). However, in some plantation systems,
115 trees are removed before these features develop due to reaching economic maturity, safety
116 concerns, damage, fire risk or to limit the spread of parasites (Russo et al., 2010). Depending on the
117 plantation system, felling may therefore directly cause direct mortality by removal of a roost that is
118 currently occupied by a bat colony or indirect mortality through impacting reproductive fitness and
119 success as the number of roost trees within a colony home range is reduced (Borkin and Parsons,
120 2014). Therefore remnant patches of either native or old growth trees may constitute the only
121 available appropriate natural structures for roosts (e.g. Burgar et al., 2015; Lindenmayer and Hobbs,
122 2004).

123 Clear felling causes an immediate and substantial change to stand structural complexity, which may
124 benefit foraging success in edge and open adapted bats (Adams, 2012; Elmore et al., 2005;
125 Kirkpatrick et al., 2017). In stands with substantial vegetative clutter, bat activity will be reduced due
126 to constraints on both echolocation and manoeuvrability (Dodd et al., 2012; Morris et al., 2010;
127 Patriquin and Barclay, 2003), and bat activity is likely to increase once clear felling has occurred.
128 Felled stands may support a similar invertebrate abundance compared to mature forest (Dodd et al.,
129 2012; Lacki et al., 2007; Ohsawa, 2005; Oxbrough et al., 2010), particularly in non-native plantations,
130 where felled plantations can be bordered with mature stands, resulting in edge habitat which
131 provides protection from wind and predators (Nicholls and Racey, 2006). Furthermore, invertebrates
132 may accumulate passively along edge habitats due to wind (Law and Law, 2011; Verboom and
133 Spoelstra, 1999). Even when invertebrate availability is lower in felled stands compared to mature
134 stands, bat activity may be higher, suggesting that the structure of the habitat may be more
135 important than prey abundance in determining the spatiotemporal foraging patterns of bats (Adams
136 et al., 2009; Dodd et al., 2012).

137 Previously, we found evidence of *P. pipistrellus* and *P. pygmaeus* making widespread use of three
138 large, predominantly Sitka Spruce (*Picea sitchensis*) plantations in Scotland and Northern England,
139 with all other species in this geographic range also detected, albeit in low numbers (Kirkpatrick et al.,
140 2017). There was also evidence that *Pipistrellus spp.* preferentially associated with felled areas
141 (Kirkpatrick et al., 2017), suggesting that some species may increase their foraging activity as a result
142 of harvesting operations. Therefore, providing roost structures are not removed or damaged in the
143 process, felling may result in increased bat activity in commercial plantations.

144 Knowledge of how bats respond to felling practices is important in understand the potential
145 implications of siting wind turbines in plantations, a practice which has greatly increased in recent
146 years. There is overwhelming evidence to suggest that wind turbines cause both direct and indirect

147 mortality through barotrauma, collision, and avoidance resulting in changes to habitat use (Voigt and
148 Kingston, 2015), although the extent to which such effects can exert population level impacts is likely
149 to vary greatly between regions. Therefore, a further consideration of this work was to investigate
150 how bat activity changed in response to the size of the clear felled area and the time since felling,
151 and relate this to forest management practices carried out to install wind turbines in commercial
152 plantations.

153 To our knowledge, the impact of felling on foraging activity of bats in commercial plantations has not
154 been experimentally tested (but see Grindal and Brigham (1998) for a similar study in native forest).
155 In this study we used a before – after – control – impact (BACI) design to quantify the effect of felling
156 on bat activity in the short term (between 1 and 16 months post-felling).

157 Specifically, we aimed to answer the following questions:

- 158 1. What is the short term influence of felling on bat activity and behaviour?
- 159 2. What influence does the size of the felled area have on bat activity?
- 160 3. How does the age of the clearfell (i.e. time since felling) influence bat activity?

161 We predict that in the short-term activity is likely to increase post felling with the creation of new
162 edge habitats. Furthermore, we predict that it is likely that the greater increases in activity will occur
163 in the smaller stands compared to the larger stands. Finally, bat activity could be expected to
164 increase as time since felling increases due to the short term change in vegetative structure which
165 may support more invertebrate prey. As we are looking at changes within two years of harvesting,
166 substantial regeneration is unlikely to have occurred which would be likely to reduce bat activity
167 (Law and Law, 2011).

168 **2 Methods:**

169 The study was conducted in three large, intensively managed plantation forests in Central and
170 Southern Scotland, and Northern England (Cowal and Trossachs: 56.188, -4.509; Galloway Forest:
171 55.117, -4.4728; Kielder Forest: 55.158, -2.442). All three forests were chosen because of their large
172 size (between 30,000 – 114,000 ha), high productivity and predominance of *Picea sitchensis*, which is
173 the most commonly planted and intensively managed coniferous tree species in Europe (Boye and
174 Dietz, 2005). Within each plantation, multiple sites (total n=31) were selected, each with a range of
175 different stand ages including mature stands of harvestable age. Sites were at least 4 km from each
176 other to reduce spatial autocorrelation (Bellamy et al., 2013). Bat activity was surveyed pre- and
177 post-harvesting at a total of 26 mature stands (11 harvested between Autumn 2013 and Spring

178 2015; 15 control stands which were not felled). Access to the remaining five sites was not possible
179 due to changes in ownership or deterioration of access routes into the plantation area.

180 2.1 Bat surveying:

181 Data on pre-harvesting activity was collected in the summer of 2013, and post-harvesting activity in
182 the summer of 2015. Stands were surveyed for a single night, starting 30 minutes after sunset
183 ensuring that recorded individuals would be actively foraging rather than commuting, and
184 continuing for four hours (the length of the shortest night in our study area); surveys were only
185 carried out on dry nights above 8°C and with wind speeds below Beaufort 4. While we recognise that
186 surveying for a single night provides only a snapshot of bat activity, we believe that BACI nature of
187 this experiment provides a robust design with which to address our primary research question. Bat
188 activity was quantified using a SongMeter SM2 Bat+ (Wildlife Acoustics, Inc., Concord, MA). For the
189 initial, pre harvesting surveys, microphones were placed at 1m height, pointed at a 45 degree angle
190 with one at the stand edge (normally adjacent to a track) and another 20 – 40m into the stand
191 interior (see Kirkpatrick et al., 2017 for further details on experimental set up). Data for edge and
192 interior at each site were pooled. Surveying was repeated in the same way following felling, with
193 microphones placed at the exact same locations as previously used. We tried to keep sampling dates
194 as similar as possible between years but this was not always possible due to logistical constraints
195 (date difference between pre and post felling sampling = ± 15 (4 – 33) days).

196 2.2 Bat call analysis:

197 We identified all calls manually to species or genus, counting the number of bat passes per night
198 (four hour period; the duration representing the shortest night during the summer at these sites),
199 which was used as a measure of activity. Analyses were conducted for total bat activity and also
200 separately for bats in the genera *Myotis* and *Nyctalus*, and for *Pipistrellus pygmaeus* and *P.*
201 *pipistrellus*. Bats in the genus *Myotis* have a similar call structure and as such were also identified
202 only to genus. It can be difficult to distinguish between *Nyctalus* calls in cluttered environments
203 (Schnitzler et al., 2003), so again these were only identified to genus. *Pipistrellus* species can be
204 differentiated between due to differences in characteristic call frequency (F_c = frequency of the
205 right-hand end of the flattest part of the call; Russ, 2012) and the call shape, so where possible
206 passes were identified to species level. For passes where it was not possible to assign to either
207 *Pipistrellus* species, we assigned them to genus. We recorded a very small number of *Plecotus*
208 *auritus* calls but did not carry out further analysis; this species has very quiet calls, so their
209 occurrence is greatly underestimated by using acoustic recordings alone.

210 2.3 Statistical analysis:

211 All analyses were carried out in R studio using R version 3.3.1 (R core development team) using the
212 following packages: MASS, lme4, ggplot2. In all models, activity was recorded as passes per four
213 hour period (defined as at least two calls within one second). To assess the effect of felling on
214 foraging activity (question 1), total or species / genus specific activity was modelled with sampling
215 time ("period"; pre / post), treatment ("treatment"; felled / control), and plantation (Cowl and
216 Trossachs, Galloway or Kielder) as fixed factors. To determine if activity changed at harvested sites
217 compared with control sites post-felling, an interaction between pre / post periods and treatment
218 was also included. Models used either a negative binomial (total bat activity, *P. pygmaeus*, *P.*
219 *pipistrellus*, all *Pipistrellus* and *Myotis*) or Poisson (*Nyctalus*) error distribution. Deviance residuals
220 were checked to ensure normality (Crawley, 2007). Two sites with much higher activity for
221 *Pipistrellus spp.* than all other sites were found to be strongly influencing the results, so analyses for
222 total bat and *Pipistrellus* activity were carried out both with and without these sites. Both forest and
223 temperature were included in BACI models to account for differences in bat activity due to
224 geographic variation or climatic conditions.

225 To determine whether the size of the felling area (question 2) or the time elapsed since felling
226 (question 3) influenced bat activity (passes per 4 hour sampling period), we used a generalised linear
227 regression model with a negative binomial error distribution for the 11 harvested sites only. Total
228 bat and *Pipistrellus spp.* activity (including *Pipistrellus spp.* calls we were unable to identify to
229 species) and the two *Pipistrellus* species separately post-felling were response variables, with the
230 size of the felled area (ha) and months since felling included as covariates and plantation ID as a
231 fixed factor. We were unable to model the influence of the size of the felling area or time elapsed
232 since felling for *Myotis* and *Nyctalus* due to the low activity we recorded for these genera. One site
233 was found to be heavily influencing the results, so the analysis was carried out both with and
234 without this site.

235 3 Results:

236 3.1 Influence of felling on bat activity:

237 All activity is expressed as bat passes per night. Total bat activity was more than four times higher at
238 treatment compared to control stands after felling had taken place, after controlling for forest and
239 temperature ($F= 3.10$, $p<0.005$; Control: Pre 17.4 (95% confidence intervals 12.1 – 25.2), Post 22.4
240 (15.6 – 32.3); Felled: Pre 22.0 (12.7 – 29.5), Post 96.4 (63.4 – 146.9). However, after removing the
241 two sites with the highest *Pipistrellus spp.* activity, this difference was no longer significant ($F = 0.59$,

242 p = 0.56; Control: Pre 17.7 (12.5 – 25.0), Post 21.2 (15.0 – 29.9); Felled: Pre 19.3 (12.7 – 29.5), Post
243 29.1 (19.10 – 44.2); Table 1 A, B, Figure 1A, 2A).

244 The impact of felling on bat activity differed between species. *Nyctalus* activity was 23 times higher
245 at the treatment sites post felling (Table 1A, Fig 1E). Overall, *Pipistrellus* activity more than doubled
246 at treatment sites post felling, but only if two sites with high activity levels were included (Table 1A,
247 1B, Figure 1F, 2D). When both *Pipistrellus spp.* were considered separately, *P. pipistrellus* activity
248 increased slightly but non-significantly post felling, and only if the two outliers were included (Table
249 1A, Figure 1C, 2C). Although from Figure 1B it appears that *P. pygmaeus* activity increases post
250 harvesting at treatment rather than control sites, this is dependent upon high activity at one site,
251 and once removed there was no significance difference in *P. pygmaeus* activity (Table 1A, B, Figure
252 2A, 2B). Finally, *Myotis spp.* activity was similar before and after felling at control and treated sites
253 (Table 1A, Fig 1D).

254 3.2 Influence of size of the felled area and time since felling:

255 Total bat activity, consisting largely of *Pipistrellus* species, declined significantly as the size of the
256 clearfell increased (Table 2A, Figure 3E), and this effect persisted after outliers were removed (Table
257 2B, Figure 3A). Considering only felled sites, total bat activity was 97% lower in the largest stands (40
258 ha) compared to smaller stands (3 ha), and similar reductions in activity were seen when *P.*
259 *pygmaeus* and *P. pipistrellus* were modelled separately (Figures 3B, C, F, G). There was no effect of
260 the time elapsed since felling (which ranged from 1 month – 18 months) on bat activity, with the
261 exception of *P. pipistrellus*, where activity was 90% higher in newly felled stands compared to those
262 felled more than 16 months previously (Table 2A, 2B).

263 4 Discussion

264 Currently, there is little information available for managers on how management and harvesting
265 operations affect biodiversity in non-native conifer plantations. To our knowledge, this is one of only
266 a few studies to explicitly test the immediate impact of felling on bat activity using a BACI study (e.g.
267 Grindal and Brigham, 1998), and is the only one conducted in commercially managed plantations
268 rather than native forest. While we recognise that sampling for a single night will only provide a
269 snapshot of bat activity and is likely to underestimate the presence or activity of rare species, our
270 primary interest was in assessing differences in activity levels between treatments rather than
271 quantifying species richness per se (Skalak et al., 2012). Furthermore, the potential implications of
272 installing wind turbines in commercial plantations are likely to be most problematic for common
273 species which would be detected by our sampling regime (Skalak et al., 2012).

274 Across much of Europe, plantation forest systems involve non-native coniferous stands felled before
275 old growth conditions are achieved, rather than the removal of old growth or native forest (FSC,
276 2012), limiting the applicability of studies conducted in old-growth forests. Here, we found that for
277 some bat species, there was an increase in activity following clear-fell harvesting. However, this was
278 dependent on the size of the clear cut, with higher bat, and particularly *Pipistrellus spp.*, activity at
279 smaller clear cuts compared to larger clear cuts. There is little evidence that felling negatively affects
280 bats through the loss of potential roost trees in this system; both *Pipistrellus* species preferentially
281 roost in buildings (Altringham et al., 1996), and Sitka Spruce reach economic maturity (<60 years)
282 and are harvested before suitable roost features form, reducing the likelihood of *Myotis* or *Nyctalus*
283 breeding colonies being present. This is in contrast to other common plantation tree species such as
284 *Pinus nigra* or *Pinus sylvestris*, both of which may form features suitable for bat roosts before
285 reaching economic maturity (Mortimer, 2006). It is likely that Sitka Spruce plantations may be
286 important primarily as foraging habitat, with felling causing little in the way of mortality due to roost
287 loss or reduction in reproductive potential (e.g. Borkin et al., 2011; Borkin and Parsons, 2014).
288 Radiotracking carried out during 2014 and 2015 found no evidence of lactating female *P. pygmaeus*
289 roosting in Sitka Spruce, although individuals did use deciduous trees in remnant patches of
290 broadleaf cover as night roosts (Kirkpatrick, 2017).

291 4.1 Impacts of felling on bat activity

292 Bat responses to felling were species specific and consistent with predictions from ecomorphology
293 (Aldridge and Rautenbach, 1987; Schnitzler and Kalko, 2001). Open adapted bats, such as *Nyctalus*
294 *spp.* which have long thin wings, a high aspect ratio and low-frequency calls, are less manoeuvrable
295 in cluttered conditions, and as expected, increased in activity at felled stands. Similarly, both
296 *Pipistrellus* species, which are typical edge foragers, had a non-significant trend towards higher
297 activity post-harvest at felled stands compared to control stands, although the extent of the
298 response varied with stand size.

299 These results contrast with those of Law and Law (2011), who found that bat activity was reduced in
300 native *Eucalyptus* forests in Tasmania following harvesting, particularly at the stand centre.
301 However, their study was conducted more than five years after felling, during which time substantial
302 changes in stand structure and plant species composition are likely to have occurred. Our study was
303 conducted within two years of felling in a commercially managed system, and as such reflects felled
304 stands rather than stands in which substantial vegetative regrowth has occurred (Law and Law,
305 2011).

306 *4.3 The influence of the size of the felled area on bat activity*

307 The size of the felled area had a significant, negative impact on bat activity for total bat activity,
308 driven by *P. pygmaeus* and *P. pipistrellus* activity, similar to other studies (Law and Law, 2011;
309 Patriquin and Barclay, 2003). It is possible that the large range in cut block size in this study (2.7 –
310 39ha) may be responsible for the somewhat equivocal species-specific responses to felling as this is
311 considerably greater than in previous studies (e.g. 0.5 - 1.5ha; Grindal and Brigham, 1998), who
312 found no effect of the size of felled area on bat activity. However, as this study and others have
313 shown, activity is lower in larger felled or open stands, possibly in response to increased perceived
314 predation risk and exposure to adverse microclimatic conditions which impact invertebrate
315 populations (Baker et al., 2013; Grindal and Brigham, 1998). Therefore, for large cut block sizes, the
316 increase in bat activity due to more edge habitat availability and easier access to invertebrate prey
317 (Law and Law, 2011) may not compensate for the increased perceived predation risk (Baker et al.,
318 2013). Clear fell sizes in this study were more representative of current clear fell forestry practices
319 (in countries where clearfelling is practiced as the method of timber extraction) compared to those
320 surveyed by Grindal and Brigham (1998). Again, further comparisons of both felling techniques and
321 extent would be necessary to fully understand the impacts of felling in a commercial forestry context
322 on bat populations.

323 *4.2 Changes in bat activity in relation to time since felling*

324 The change in structural complexity is likely to be driving the change in bat activity we see here, and
325 may be more important in driving bat activity than invertebrate availability (Dodd et al., 2012;
326 Morris et al., 2010). Physical clutter may impede flight efficiency and be harder to negotiate as dense
327 vegetation will result in increased acoustic clutter (Jung et al., 2012; Morris et al., 2010; Patriquin
328 and Barclay, 2003). We did not measure invertebrate abundance before and after felling in this
329 study, and it is also possible that increased ground disturbance, prevalence of deadwood and
330 stagnant groundwater that remains immediately after clear felling results in a short-lived but
331 substantial increase in Nematoceran Diptera abundance (Blackwell et al., 1994), providing an
332 ephemeral food source which bats are able to exploit (Fukui et al., 2006). Therefore freshly felled
333 areas in Sitka Spruce plantations may represent a patchy and ephemeral food resource in the
334 landscape, with minimal physical and acoustic clutter, which is thereby easier to negotiate and hunt
335 in (Patriquin and Barclay, 2003; Pauli et al., 2015). Similarly, Borkin and Parsons, (2014) hypothesised
336 that the reduced home range of *Chalinobus tuberculatus* in exotic pine plantations in Australia
337 partially reflected an increase in potential foraging areas near roost sites post harvesting.

338 *Pipistrellus pipistrellus* activity decreased as time since felling increased, but *P. pygmaeus* and total
339 bat activity was unaffected. Previous studies have found that the two closely related and

340 morphologically similar *Pipistrellus* species (*P. pipistrellus* and *P. pygmaeus*), whilst similar in foraging
341 ecology, have habitat and dietary differences (Barlow, 1997; Davidson-Watts et al., 2006; Nicholls
342 and Racey, 2006). Both species were commonly recorded in the three Sitka Spruce plantations
343 surveyed as part of this study, and have previously been found to preferentially forage at felled
344 stands, with activity declining in response to stand age and density (Kirkpatrick et al., 2017).
345 Therefore, it is likely that the increase in bat activity post felling will not continue as stand replanting
346 or regeneration occurs.

347 *4.4 Implications for wind turbine installation:*

348 This work has implications for our understanding of how habitat management prior to installing
349 turbines may influence bat activity in plantations. Typically, small areas are felled and kept clear of
350 regenerating vegetation around the turbine site (50m beyond turbine blade tip, Anon., 2015) and
351 access roads are constructed into newly felled areas. Roads are important flyways for some bat
352 species, allowing access into different plantation areas (Grindal and Brigham, 1998; Hein et al., 2009;
353 Law and Chidel, 2002) and may potentially guide bats towards newly installed turbines, particularly
354 *Nyctalus* and *P. pipistrellus*. This may create a patchwork of attractive foraging patches within which
355 turbines have been installed, acting as an ecological trap (Tscharntke et al., 2012), particularly as low
356 structural clutter is maintained in these areas. Since it is assumed that bat activity in upland
357 plantations is low, there are currently no requirements for monitoring post felling and post
358 construction and no guidance on the potential impacts on bats (Mathews et al., 2016). Results from
359 this thesis suggest that some bat species could be at far greater risk from the installation wind
360 turbines in commercial plantations than previously thought. Further investigation of both short and
361 long term responses of bats to keyholing is an urgent priority, particularly for *Nyctalus* species
362 (Mathews et al., 2016).

363 *4.5 Conclusions*

364 This study is the first of its kind to investigate the impact of felling on bats in Sitka Spruce plantations
365 and demonstrates not only that upland commercial coniferous plantations are not devoid of bats,
366 but that there is an urgent need for further studies, especially in light of changing land use in upland
367 plantations. We found some evidence that activity of particular species increased, especially in small
368 stands that have been recently felled. Installing wind turbines in upland plantations after keyhole
369 harvesting could therefore have implications for bat activity and mortality.

370 Acknowledgements:

371 Funding was provided as part of an IMPACT studentship, funded by Forestry Commission and
372 University of Stirling. We thank Dr. Sallie Bailey for assistance in project design and implementation.

373 We thank Bill Fisher, Andrew Jarrott, John Hair, Rajmund Zygadlewicz, Tom Dearnley and Max
374 McLaughlan from the Forestry Commission for assistance with site selection and access.

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Tables

Model A (with outliers)	(Intercept)	Treatment (Felled)	Activity: (Post felling)	Treatment * activity	Temperature	Galloway	Kielder	R ²
All bat activity	0.66 ± 0.87	0.23 ± 0.28	0.25 ± 0.27	1.23 ± 0.40	0.24 ± 0.06	-2.14 ± 0.31	-0.42 ± 0.29	0.22
<i>P. pygmaeus</i>	2.09 ± 1.53	0.21 ± 0.50	0.37 ± 0.47	0.94 ± 0.70	0.17 ± 0.11	-2.07 ± 0.55	-0.64 ± 0.52	0.27
<i>P. pipistrellus</i>	-1.70 ± 1.60	0.44 ± 0.50	0.61 ± 0.48	1.03 ± 0.70	0.32 ± 0.11	-2.51 ± 0.56	0.69 ± 0.50	0.59
<i>Myotis spp.</i>	-2.69 ± 1.69	-0.01 ± 0.53	-0.15 ± 0.51	0.19 ± 0.74	0.22 ± 0.12	0.72 ± 0.65	1.85 ± 0.61	0.20
<i>Nyctalus spp.</i>	-2.94 ± 0.99	-1.14 ± 0.39	-0.24 ± 0.29	1.67 ± 0.48	0.13 ± 0.05	2.66 ± 0.72	0.28 ± 0.79	0.54
All <i>pipistrellus</i> activity	1.39 ± 1.35	0.32 ± 0.44	0.34 ± 0.41	1.16 ± 0.61	0.26 ± 0.10	-2.36 ± 0.49	-0.49 ± 0.45	0.44
Model B (without outliers)								
All bat activity	0.74 ± 0.84	0.09 ± 0.27	0.18 ± 0.25	0.23 ± 0.38	0.23 ± 0.06	-1.81 ± 0.29	-0.63 ± 0.27	0.10
<i>P. pygmaeus</i>	2.18 ± 1.51	-0.25 ± 0.50	0.32 ± 0.44	0.24 ± 0.70	0.17 ± 0.11	-1.62 ± 0.52	-0.93 ± 0.49	0.26
<i>P. pipistrellus</i>	-1.55 ± 1.46	0.58 ± 0.46	0.47 ± 0.42	-0.59 ± 0.65	0.31 ± 0.10	-2.17 ± 0.50	0.51 ± 0.44	0.49
All <i>pipistrellus</i> activity	1.54 ± 1.29	0.18 ± 0.43	0.25 ± 0.38	0.22 ± 0.60	0.25 ± 0.09	-2.04 ± 0.45	-0.72 ± 0.42	0.40

Table 1: Change in bat activity at control and felled sites after harvesting occurred. Presented are model estimates plus standard error for activity per night, including outliers (A) and excluding outliers (B). Parameters with a significant effect are outlined in bold. Models are presented both with (A, n = 26) and without (B, n = 24) two outlying sites.

Model	Term	A			B				
		Estimate ± std error (with outliers)	F- statistic	p value	R2	Estimate ± std error (without outliers)	F- statistic	p value	R2
Total bat activity	(Intercept)	2.55 ± 1.02	2.50	0.01		2.08 ± 1.04	2.00	0.05	
	Size of felled area	-0.10 ± 0.02	-4.57	<0.001		-0.09 ± 0.03	-3.46	<0.001	
	Months since felling	-0.04 ± 0.04	-0.99	0.32	0.83	-0.01 ± 0.04	-0.31	0.76	0.72
	Galloway	3.34 ± 0.97	3.46	<0.001		3.30 ± 0.96	3.42	<0.001	
	Kielder	5.38 ± 1.05	5.13	<0.001		5.08 ± 1.09	4.65	<0.001	
<i>Pipistrellus pygmaeus</i>	(Intercept)	2.16 ± 1.34	1.62	0.11		1.82 ± 1.43	1.28	0.20	
	Size of felled area	-0.11 ± 0.03	-3.53	<0.001		-0.10 ± 0.04	-2.71	0.01	
	Months since felling	-0.03 ± 0.06	-0.55	0.58	0.68	-0.01 ± 0.06	-0.18	0.86	0.54
	Galloway	2.96 ± 1.26	2.34	0.02		2.94 ± 1.31	2.25	0.02	
	Kielder	4.77 ± 1.38	3.45	<0.001		4.57 ± 1.50	3.05	0.002	
<i>Pipistrellus pipistrellus</i>	(Intercept)	2.40 ± 1.45	1.65	0.09		2.10 ± 1.52	1.38	0.17	
	Size of felled area	-0.09 ± 0.03	-3.20	0.001		-0.08 ± 0.03	-2.41	0.02	
	Months since felling	-0.14 ± 0.05	-2.64	0.01	0.85	-0.12 ± 0.06	-2.11	0.03	0.75
	Galloway	1.59 ± 1.42	1.12	0.26		1.58 ± 1.45	1.09	0.28	
	Kielder	4.99 ± 1.48	3.38	<0.001		4.82 ± 1.56	3.08	0.002	
<i>Pipistrellus spp.</i>	(Intercept)	2.62 ± 1.12	2.34	0.02		2.20 ± 1.17	1.88	0.06	
	Size of felled area	-0.11 ± 0.03	-4.23	<0.001		-0.09 ± 0.03	-3.20	0.001	
	Months since felling	-0.04 ± 0.05	-0.94	0.34	0.80	-0.02 ± 0.05	-0.39	0.69	0.67
	Galloway	3.15 ± 1.06	2.97	0.003		3.11 ± 1.08	2.90	0.004	
	Kielder	5.38 ± 1.15	4.67	<0.001		5.12 ± 1.23	4.17	<0.001	

Table 2: Change in total bat activity and *P. pygmaeus* / *P. pipistrellus* / all *Pipistrellus* activity felled sites after felling occurred including the size of felled area and time elapsed since felling in the models. Model estimates plus standard error are presented for activity per four hour sampling period, and are presented both with (A) and without (B) two sites with high activity levels.

Figures

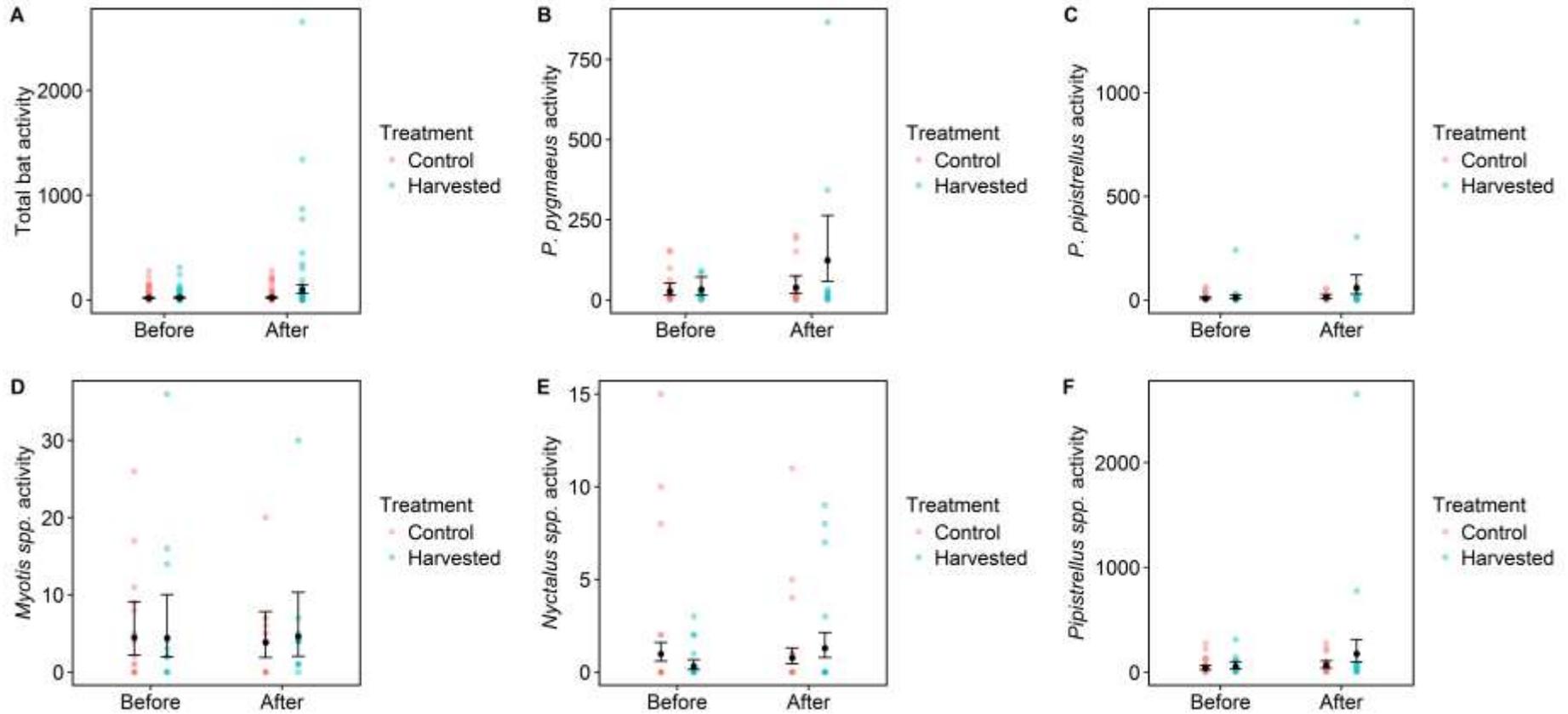


Figure 1: Change in bat activity at control and felled sites before and after harvesting occurs for total and species / genus specific activity. Outlying sites are included ($n = 26$). Coloured dots depict raw data on bat activity, black dot shows model predicted activity and error bars show 95% confidence intervals.

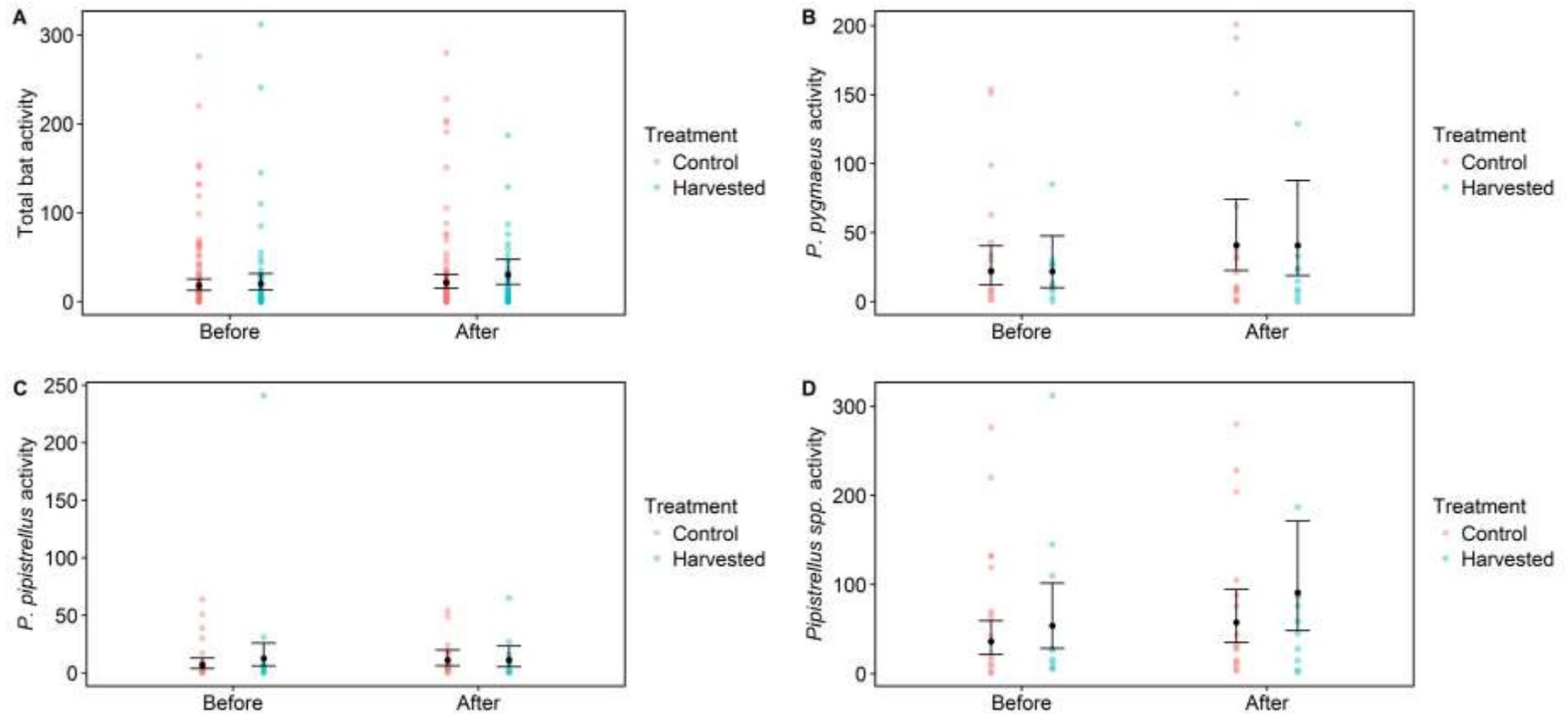


Figure 2 Change in bat activity at control and felled sites before and after harvesting occurs for total and species / genus specific activity. Outlying sites are excluded ($n = 24$). Coloured dots depict raw data on bat activity, black dot shows model predicted activity and error bars show 95% confidence intervals.

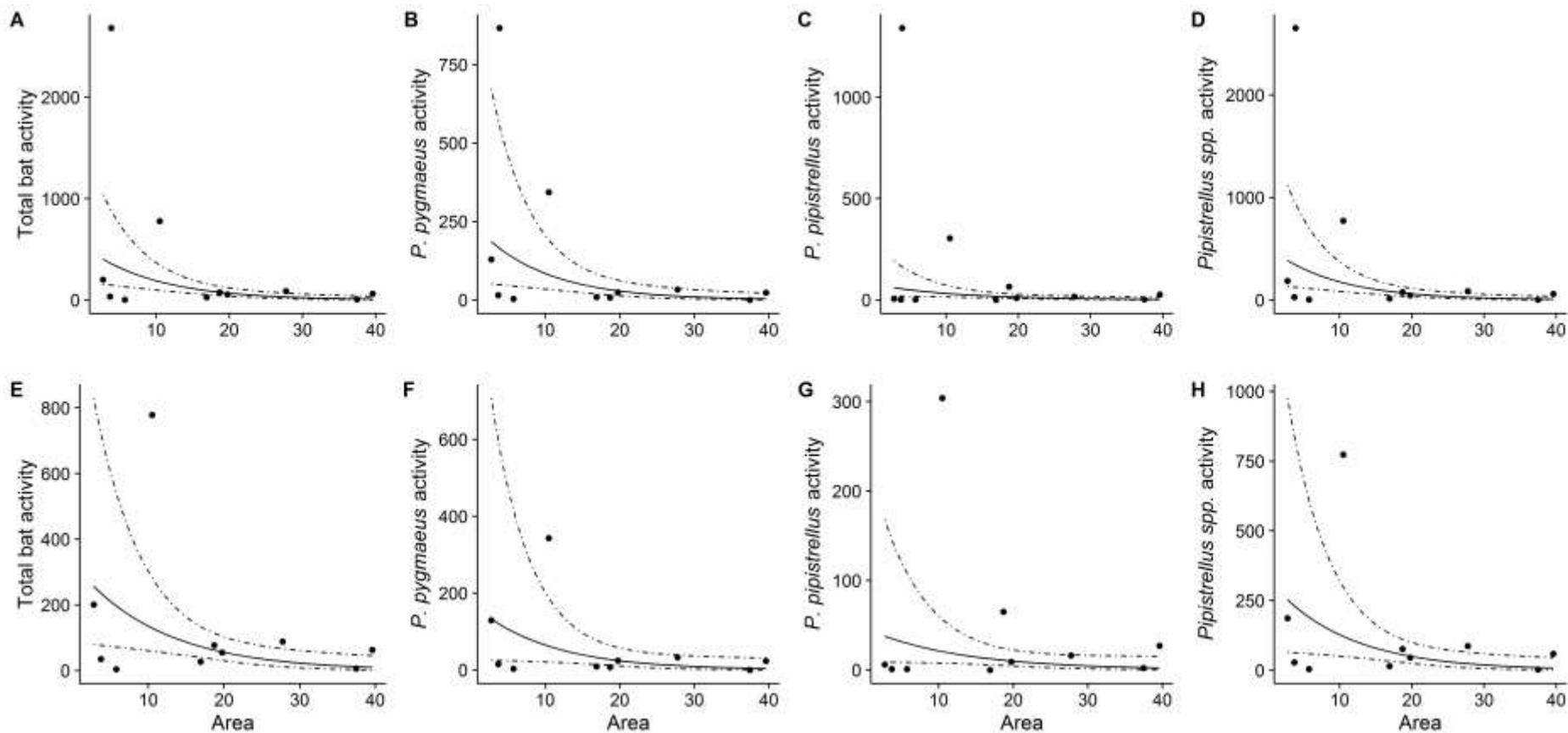


Figure 3: Change in total, *P. pygmaeus*, *P. pipistrellus* and all *Pipistrellus* activity in response to felled stand area ($n = 11$). Black dots are raw data, the solid black line is the model prediction for change in activity, the dashed lines are the 95% confidence intervals. Graphs A – D include one site with very high activity ($n = 11$), graphs E – H exclude outlier ($n = 10$).