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Greg Wood, Duncan Whyatt, David Hackett, Carly Stevens

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- Temporal behavioural fluctuations are important for disturbance susceptibility
- Temporal GIS models can be used to synchronise development with conservation
- Disturbance tolerances can be accounted for at the scale of the individual animal

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Spatio-temporal challenges in representing wildlife disturbance within a GIS

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4   **Greg Wood**[a] (g.wood1@lancaster.ac.uk)
5   **Duncan Whyatt**[a] (d.whyatt@lancaster.ac.uk)
6   **David Hackett**[b] (d.hackett@solumenvironmental.com)
7   **Carly Stevens**[a] (c.stevens@lancaster.ac.uk)

8

9   [a]Lancaster Environment Centre, Lancaster University, Lancaster, UK, LA1 4YQ
10 [b]Solum Environmental, Suite 6, 9 – 11 Princess Street, Knutsford, Cheshire, UK, WA16 6BY

11

12 Corresponding Author: Carly Stevens +44 01524 510211

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14 **ABSTRACT**

15 Assessing the potential environmental impacts of disturbance on protected species during and after

16 the development process is a legislative requirement in most nations. However, the restrictions that

17 this legislation places on developers are often based on limited ecological understanding, over-

18 simplified methodologies, less-than-robust data and the subjective interpretations of field

19 ecologists. Consequently, constraints may be imposed with no transparent methodology behind

20 them to the frustration of, and occasionally large expense to, developers. Additionally, protected

21 species numbers continue to decline and biodiversity continues to be threatened. This paper

22 describes a GIS conceptual model for assessing ecological disturbance vulnerability, based upon a

23 case study development in Scotland. First, uncertainties in traditional methods of recording and

24 representing ecological features with GIS are reviewed such that they may be better accounted for

25 in the disturbance model. Second, by incorporating temporal fluctuations in ecological behaviour

26 into the disturbance susceptibility concept, it is argued that it is possible to synchronise

27 development with conservation requirements. Finally, a method is presented to account for

28 disturbance tolerances at the scale of the individual animal. It is anticipated that this model will

29 enable environmental impact assessors to produce more robust analyses of wildlife disturbance risk

30 and facilitate synchronisation between development and wildlife vulnerability to minimise

31 disturbance and better avoid delays to the works programme.

32 **KEYWORDS**

33 Ecological Disturbance; Development sites; Protected species; GIS representation; Ecological

34 networks; Temporal ecology

35 **1   INTRODUCTION**

36 Increasing demand for housing, commerce and industry, driven by an expanding human population,

37 is perpetuating global urban development (Millennium Ecosystem Assessment, 2005). However, as

38 global landscapes become increasingly urbanised, space available for new development becomes

39 ever more constrained. This constraint is compounded by the need to maintain multifunctional

40 landscapes that promote prosperity for both humans and wildlife (Angold et al., 2006, Rudd et al.,

41 2002) for intrinsic purposes, and to continue the delivery of ecosystem services (Millennium

42 Ecosystem Assessment, 2005). For these reasons, a series of legislative measures have been

43 introduced affording legal protection to selected species and habitats deemed to be under threat or

44 of particular cultural significance. Central to this concept of protection is the notion of 'wildlife

45 disturbance', such as that caused by excessive noise, vibration, loss of food sources or the

46 introduction of new predators. Disturbance is prohibited for certain species under the Habitats

47 Directive (European Commission, 1992) and Birds Directive (European Commission, 2009), and

48 various national laws devolved from them. There is, however, no universally accepted definition of

49 the term 'wildlife disturbance' leaving it open to interpretation in best practice guidance issued by

50 statutory regulators. Consequently, environmental impact assessments with a wildlife disturbance

51 component are open to subjectivity and lack a standardised approach.

52 A more in-depth model of wildlife susceptibility to disturbance is clearly needed to reduce levels of

53 subjectivity and improve the means by which development constraints are integrated into

54 development programmes. Given that the risk of disturbing an animal is largely subject to spatial

55 criteria, Geographical Information Systems (GIS), offer a solid foundation upon which to achieve this

56 task. GIS data are also easily displayed via a website or server system, facilitating communication,

57 and may be viewed at multiple scales to better understand a sites context and broader landscape

58 connections. GIS has already established its credentials as a planning tool in ecology, for example, in

59 the design of wildlife corridors (Jenness et al., 2010), nature reserves (Ball et al., 2009) and habitat

60 restoration schemes (Rempel, 2008).

61 Goodchild (2007) defines three levels of abstraction between real world phenomena and GIS

62 representation - conceptualisation of the processes and interactions inherit to the studied

63 phenomenon, recording of the variables of interest and representation of the variables in

64 appropriate digital form.   In the context of wildlife disturbance on a development site there are

65 challenges at each level of abstraction.

66 Difficulties at the conceptual level are illustrated through a lack of legislative clarity in the definition

67 of wildlife disturbance. Such ambiguity has led to differing requirements for the treatment of badger

68 (*Meles meles*) setts for example, protected from disturbance in the UK by the Protection of Badgers

69 Act 1992. In Scotland, the current requirements for compliance with this legislation involves creating

70 a protection zone of 30 m around the sett, within which potentially disturbing activities are

71 prohibited (Scottish Natural Heritage, 2013a). However, the English approach (English Nature, 2002)

72 leaves the interpretation of disturbance to the field ecologist. Whilst the English approach can

73 facilitate a more complex conceptualisation of wildlife disturbance, its application could be biased by

74 social or cultural values and research specialisations of the individual ecologist. Conversely, the

75 problem with the Scottish methodology is twofold: first, the discrete representation undermines the

76 obvious distance decay in disturbance probability with respect to proximity to the sett entrance;

77 second, the protection zone radius of 30 m, based on tunnel lengths of excavated setts (Raynor,

78 2012) is designed to protect the sett structure, giving little consideration to adverse effects of noise

79 or vibration upon the badger inside. In more extreme cases of physical disturbance such as pile

80 driving or blasting, the protection zone is increased to 100 m radius (Scottish Natural Heritage,

81 2013a), although the justification for this distance is not given in any of the reviewed literature.

82 At the recording level, challenges arise from not being able to monitor wildlife completely and

83 directly in the field. For example, whilst animals are frequently GPS or radio tagged, giving insight

84 into their spatio-temporal positions, which in turn allows the derivation of home range (Powell,

85 2000), interaction patterns (Handcock et al., 2009) and travelling routes (Nams, 2005), the capture

86 and tagging of every animal on a large development site is impractical. Thus, ecological knowledge is

87 often derived from field signs including faeces, hair, prints and scratches, which are easy to miss in

88 the field (Parry et al., 2013). Although such uncertainties are reducing with the introduction of video

89 technology (Moll et al., 2007), surveillance generally covers only a small area of a given site and a

90 sub-selection of individual animals.

91 At the representational level, challenges chiefly arise from the temporally-dynamic nature of

92 ecosystem functioning, thus affecting the severity and likelihood of wildlife disturbance. Bats for

93 example, are extremely reliant upon undisturbed hibernation in order to maintain sufficient fat

94 supplies to last the winter (Thomas, 1995). Similarly, the Forestry Commission (1995) advocates the

95 cessation of works in close proximity to badger setts at dawn and dusk to allow its occupants to

96 move in and out, illustrating the species' dependence upon daylight cycles. The lack of an innate

97 temporal query language within most GIS applications means that answering questions regarding

98 when a particular operation (e.g. pile driving) should be conducted to coincide with periods of low

99 disturbance likelihood is difficult.

100 This paper offers a detailed assessment of the conceptual, recording and representational challenges

101 faced in communicating wildlife disturbance constraints within a GIS for a case study site in the

102 Central Lowlands of Scotland. To begin with, potential receptors to disturbance are discussed, along

103 with uncertainties in their traditional GIS representations. Temporal fluctuations in disturbance

104 susceptibility are then considered, and insights offered into how anthropogenic disturbance may be

105 synchronised with cyclic variations in wildlife activity to minimise disturbance. Finally, in the light of

106 these discussions, a conceptual model for wildlife disturbance vulnerability is proposed. The model

107 accounts for a more detailed understanding of wildlife ecology and encompasses spatio-temporal

108 uncertainties in ecological knowledge.

109 **2   METHODOLOGY**

110 Challenges are illustrated using ecological examples from a large (10 km^2) brownfield site, located 20

111 km west of Glasgow, Scotland. The site has proposals for 2500 units of housing, a 150,000m^2

112 business park, related infrastructure and a community woodland park (Renfrewshire Council, 2014).

113 During the sites' extended period of phased decommissioning, public access has been

114 heterogeneously restricted, meaning that some areas of the site have remained undisturbed for

115 over 75 years, whilst others were in use until 2002. Such conditions have yielded an ecological

116 mosaic exhibiting a wide variety of successional traits, and have encouraged a number of protected

117 species to occupy the site including 12 badger social groups (average 3-4 adults per group plus cubs),

118   3 otters (plus cubs), 2 breeding pairs of barn owls, small populations of common pipistrelle, brown

119   long-eared and Daubenton's bats and 2 breeding pairs of little ringed plover (results of monitoring

120   by the author since 2012).

121   Surveys for a number of species were undertaken using methods described in the Chartered

122   Institute of Ecology and Environmental Management's (CIEEM) Technical Guidance Series (Chartered

123   Institute of Ecology and Environmental Management, 2014). These surveys are designed to be

124   conducted at fine spatial scales, to determine specific landscape features, such as a particular

125   hedgerow or building, considered important for conservation of an individual animal or social group.

126   This scale is equivalent to Johnson's (1980) fourth order of hierarchical habitat selection; the first

127   being the geographical distribution of the species, the second being the selection of a home range

128   within that distribution, and the third being the usage of generalised habitat patches within the

129   home range. By adopting a fine spatial scale we constrain our study to disturbance concepts

130   affecting individual animals and social groups, rather than wider populations or meta-populations.

131   Conceptual, recording and representation challenges were then identified by comparing the

132   practical requirements of field survey and GIS design with ecological disturbance concepts

133   highlighted throughout the literature. These concepts were augmented with our own critical

134   thinking and then consolidated into a conceptual model that encapsulates a more complete idea of

135   the potential for wildlife disturbance due to different activities.

136   **3 RESULTS AND DISCUSSION**

137   **3.1 INFERRING THE UNKNOWN FROM THE KNOWN**
138   Initially, an understanding of how protected species use their available space is necessary in order to

139   determine both the projected impact of a disturbance event across the ecological network, and the

140   likely position of an individual animal within it at any given time. However, each of the features that

141   comprise the ecological network, such as foraging resources, pathways and shelters (collectively

142   referred to hereafter as ecological network components (ENCs)) are often difficult to observe, record

143   and map comprehensively (Parry et al., 2013). In seeking to improve communication of wildlife

144   disturbance constraints, it is important to understand the limitations of the underlying ecological

145   data upon which higher level knowledge can be derived. By integrating associated uncertainties that

146   will be inherent within the disturbance model, decision makers can be better informed as to the

147   reliability of disturbance projections, and field ecologists can learn where best to target their surveys

148   as part of any monitoring efforts. The following section highlights some of these uncertainties and

149   generalisations for different ENCs observed at the case study site. It is not intended to be an

150   exhaustive list but rather to draw attention to nature of uncertainties in ecological data and to

151   stimulate discussion and critical thinking.

152   **3.1.1 Foraging resources**
153   Food and water act as vital nodes within the ecological network and their quantities play a

154   significant role in the estimation of carrying capacities (for a review see McLeod, 1997). However,

155   given the spatial scale of the case study site, it was not possible to record the availability of food

156   directly or completely. For example it was not feasible to count the number of berries available

157   across all woodland and scrub patches, nor was it possible to account for movements of each and

158   every live prey individual between territories of carnivores. Habitat types are therefore often used as

159   a surrogate for food availability (e.g. Anderson et al., 2005, Scottish Natural Heritage, n.d.). Whilst

160   this generalisation forms a convenient measure by which to estimate resource availability, it

161   assumes a homogeneous distribution of resource biomass throughout each mapped habitat parcel.

162   Biomass can however vary significantly between microhabitats (Shakir and Dindal, 1997, Shevtsova

163   et al., 1995).

164   Moreover, the habitat categorisation processes typically used by commercial ecologists (e.g. Phase 1

165   Habitat Classification (Nature Conservancy Council, 1990) and the National Vegetation Classification

166   (NVC) (Rodwell, 2006)) required for a vector-based GIS representation, often renders habitat

167   boundaries that are not always agreed upon by ecologists (Stevens et al., 2004). Consequently, a

168   discrete approach to mapping habitats may yield highly uncertain estimates of foraging resource

169 availability. Whilst there are alternative habitat representations available that use a raster model,

170 such as the Land Cover Map (Morton et al., 2011), these do not reflect the continuous nature of

171 habitat and food availability transitions through space. Rather, they depict the same discrete land

172 cover classes as the vector models, but are spatially generalised over grid cells of 25 m or 1 km

173 (Morton et al., 2011).

174 **3.1.2 Home range and territory**
175 The amount of foraging resources available to an individual animal is constrained by its home range

176 or territory. Individuals with a home range or territory categorised by poor foraging opportunities

177 are more vulnerable to loss of resources (Kitaysky et al., 1999). By using such data to influence land

178 use change, under some scenarios it may be possible to dilute the effects of disturbance across

179 multiple territories or home ranges to the point where it is no longer significant to any one

180 individual. Multiple methodologies have been developed to derive home ranges from field data with

181 varying degrees of complexity and realism. The simplest approach, and the one usually applied in

182 commercial UK ecology, is to utilise a uniform radius buffer around shelters with a distance

183 extrapolated from literature or 'expert' opinion (English Nature, 2002, Scottish Natural Heritage,

184 2013a, b). However, this approach is also the most generalised as it does not account for factors

185 such as habitat composition, which can cause variations in home range size from individual to

186 individual (Börger et al., 2006, Anderson et al., 2005), Consequently, the home range in reality may

187 be markedly different to that formulated using this methodology.

188 Slightly more representative of the real world, Minimum Convex Polygon analysis (MCP) (Nilsen et

189 al., 2008, Börger et al., 2006) can be utilised to derive home ranges and territories from a variety of

190 spatial data (depending on the species) including field signs, direct observations, location tracking,

191 camera trap data and echo-sounding. This has the advantage over the buffering approach in that the

192 analysis is based on data taken directly from the study area, but is nevertheless subject to a

193 significant number of limitations highlighted by Worton (1987) including bias caused by sample size

194 and a tendency to overestimate range sizes. The MCP methodology has, however, been

195 demonstrated to be effective for delineating badger territories from bait marking data (Delahay et

196 al., 2000) as latrines are highly characteristic of badger territory edges (Roper et al., 2001).

197 Conversely, territories of avian species derived from MCP analysis of field signs such as sightings,

198 droppings, down, and in particular, subjective interpretation of song (Bibby et al., 2000) are likely to

199 yield home range boundaries of much lower certainty.

200 A further limitation to MCP analysis is that the derived territories and home ranges are discrete, thus

201 giving no representation of spatial usage within them. Greater insight into the internal structure of

202 home ranges and territories can be gained through the application of utilisation distribution

203 methodologies including kernel density (Worton, 1989, Seaman and Powell, 1996, Powell, 2000,

204 Börger et al., 2006), time-geographic density estimation (Downs et al., 2011) and random walk

205 models (Horne et al., 2007) to generate a representation of probability of usage of space within each

206 home range or territory. GIS representation of this internal 'probability-of-use' gradient then

207 becomes more suited to a raster approach (continuous variation), rather than a vector approach

208 (discrete objects). Whilst these types of analyses are also subject to uncertainties, such as that

209 caused by spatial-autocorrelation (Blundell et al., 2001), the chief concern here is that they require

210 considerable geo-statistical skill to compute and interpret, and also tend to rely upon telemetry

211 data. In the UK protected species require a licence to trap and fit with a GPS device. Combined with

212 the complexity of utilisation distribution methodologies this may lead practicing ecologists to

213 implement the less robust MCP methodologies as at the case study site.

214 **3.1.3 Pathways**
215 Pathways link areas used for shelter and foraging together, thus are an essential component of the

216 ecological network. As with home ranges and territories, pathways are often inferred from field

217 signs yielding different levels of certainty depending upon the species studied. Some species on the

218 case study site, such as badgers, follow well-defined paths and these can often be recognised in the

219 field by the trained ecologist (Kyne et al., 1990, Neal and Cheeseman, 1996). Badger paths also

220 tended to follow prominent linear features, such as hedgerows, woodland edges and walls, which is

221 consistent with other observations made by Feore and Montgomery (1999), and Hutchings and

222 Harris (Hutchings and Harris, 2001). Badgers are also known to show little variation in traveling

223 routes, which in turn facilitates the mapping of these features as discrete lines, but also signifies

224 greater sensitivity of the animals to path disturbance. In contrast, it is more difficult to define an

225 exact flight path for birds and bats since they show more spatio-temporal variability in their

226 travelling routes, their movements leave no field signs such as footprints or flattened grass to

227 observe and their movements must be considered in three dimensions.

228 The inference of such 'fuzzy' pathway locations may be improved by additional observation, location

229 tracking or camera traps to help improve data reliability. Additionally, a least cost path can be

230 inferred, based on known shelters, foraging locations and movement impedances or preferences.

231 Davies et al. (Davies et al., 2012) for example inferred travelling routes for pipistrelle bats

232 *(Pipistrellus pipistrellus)* constrained by lighting sources (known features of avoidance) and

233 hedgerows, which are known to be preferred travelling routes for the species (Mitchell-Jones, 2004).

234 Nonetheless, given the 'fuzzy' nature of these features, vector representation as discrete lines can

235 imply greater accuracy in the data than is appropriate. Murdock and Potts (Murdock and Potts,

236 2009) interviewed ornithologists, and found the estimated positional accuracy of such surveys to be

237 ±50 m. Hence, it may be more appropriate to represent these paths as rasters with a cells size of 50

238 m or generate 25 m buffer zones around the vector lines to reflect this spatial uncertainty.

239 **3.1.4 Shelters**
240 As with other ENCs, acquiring a comprehensive dataset of all protected shelters in a large study area

241 can be difficult. Otter *(Lutra lutra)* holts in particular are notoriously difficult to locate due to their

242 often secluded position, camouflage and underwater entrances (Parry et al., 2013). Even when

243 shelters are discovered, the attribution of a species to that shelter can be problematic. Since many

244 shelters are subterranean, both occupation and species are typically inferred from field signs, such

245 as the shape of the tunnel, foot prints, faeces, hair and scratches found in close proximity to the

246 shelter. Although ecological surveyors are trained in using these field signs to find shelters and infer

247   species occupation, the process is not infallible, meaning that some shelters may be missed or that

248   the wrong species may be assigned to it. The latter is particularly problematic when considering

249   legislative-driven constraints on wildlife disturbance. Consider, for example, that female foxes are

250   known to use unoccupied badger setts to rear their young (Trewby et al., 2008), but that they are

251   not protected from disturbance by wildlife legislation. Moreover, it is difficult for the ecologists to

252   know if the sett is likely to be used by a badger in the future, thus the decision as to whether to

253   continue to control nearby anthropogenic disturbance, is equally problematic from a legislative

254   perspective.

255   Similar legal uncertainties also surround the protection status of otter resting places and bat feeding

256   perches. UK legislation and best practice guidance (e.g. Mitchell-Jones, 2004, Scottish Natural

257   Heritage, 2013b) offers no insight into the frequency of use necessitated for these features to

258   receive protection, thus their explicit inclusion within a disturbance susceptibility model may place

259   unnecessary restrictions on development works if alternative areas are available. Perhaps a more

260   meaningful disturbance analysis would consider areas with the potential to be used by wildlife as

261   resting places or perches, rather than simply attempting to establish those in current use. A similar

262   methodology to that used in habitat suitability modelling could be used (for reviews see Fielding and

263   Bell, 1997, Hirzel et al., 2002, Hirzel and Le Lay, 2008); however, further research would be required

264   to produce such a model for each relevant feature.

265   Uncertainty is not only introduced at the recording level but at the digital representation level too.

266   Shelters are usually represented within a GIS as vector points or generalised into raster cells for

267   larger scale studies (e.g. National Biodiversity Network, 2012). In the latter case the raster can either

268   indicate a count of shelters present, or simply indicate presence in a Boolean format. Whilst the

269   Boolean representation is subject to the most spatial and thematic generalisation, it is less prone to

270   error (see Figure 1). However, in the case of disturbance modelling at the site scale, the locational

271   precision of shelter data offered by the vector representation is imperative to formulating the most

272   robust strategies of avoidance and mitigation. The issue with this, as with pathway representation as

273   vectors lines, is that a degree of precision and accuracy is instilled within the data that may lead to

274   false conclusions regarding its reliability.

275   **Figure 1.**

276   **3.2 THE ISSUE OF TIME**
277   Development at the case study site is being conducted over a multi-phased programme, lasting in

278   excess of a decade. Consequently, temporal ecological factors, which may be overlooked in

279   development projects conducted over shorter timescales, warrant significant attention here. As

280   illustrated in section 3.1.4, GIS models of representation can be used to generalise the spatial

281   dimension; in much the same way, a single GIS layer, in the absence of other thematically related

282   layers, can be considered a generalisation of the temporal dimension. Many species exhibit spatial

283   variation in their home ranges through time including otters (Erlinge, 1967), deer (Börger et al.,

284   2006) and badgers (Cresswell and Harris, 1988). For hibernating species such as bats, the spatial

285   extent of their home ranges will decrease to nothing during winter. Thus, any GIS layer depicting a

286   home range for a single instance in time must, in the absence of supplementary data, be taken by

287   the researcher to be representative of all instances of time. Further, seasonal generalisations occur

288   in modelling food resource availability, since the abundance of nuts and berries for instance are also

289   dependent upon seasonality. This in turn may affect the temporal frequency of pathway use since a

290   community may use different feeding areas in different seasons. Finally, shelter usage may also be

291   dependent on seasonality. A main badger sett, for example, is likely to be active all year round; an

292   outlier however will be more frequently used in the breeding season when territorial behaviour

293   peaks (Cresswell et al., 1992). These concepts are illustrated for different species in Figure 2.

294   Seasonality is not the only driver for temporal generalisation, daily cycles also play a significant role

295   in governing ecological behaviour, and subsequently in determining the reliability of the GIS data

296   representing it. Diurnal shelter usage is chiefly governed by the sleeping cycle of the animal, so for a

297   nocturnal animal, their shelter is likely to be in use during daytime. Thus, in a temporally-enabled

298   GIS, the frequency at which data is recorded, and the granularity at which it is modelled, will govern

299   the generalisation of the time dimension.

300   Temporal generalisation has particular significance when using ecological GIS data to influence

301   landscape management decisions. If the protection of the ENCs described above are considered to

302   be of primary importance in species conservation, then a failure to account for their temporal

303   fluctuations, both spatially and thematically, may result in under or over protection at a given time.

304   Under protection may result in development which will impact negatively upon local wildlife,

305   whereas over protection will place unnecessary strain on developers' time and resources. This has

306   particular relevance for constraining site operations such as cable laying, safety lighting, blasting and

307   pile-driving where wildlife disturbance may only be temporary. Undertaking these works near a

308   winter food source, travelling route or shelter may be considered acceptable in the summer for

309   example, provided alternative resources are available.

310   Unfortunately, the representation of the temporal dimension within GIS can be problematic and is a

311   primary focus of past and current GIS research (Armstrong, 1988, Erwig et al., 1999, Huang and

312   Claramunt, 2005, Pelekis et al., 2004, Raper, 2012). Although a full discussion on approaches and

313   methods of time representation in a GIS is beyond the scope of this paper, it is important to note

314   here that the vast majority of solutions require customised software, bespoke query languages and

315   considerable GIS skill to implement and maintain. In the case of commercial ecology this becomes

316   unfeasible since data sharing between stakeholders, whose specialisms are likely to lie beyond the

317   domain of GIScience, is of paramount importance. We therefore argue that the extension of GIS into

318   the temporal dimension needs to be conducted in conjunction with existing propriety or open

319   source software, making solutions accessible, intuitive and interoperable with other data sources.

320   **Figure 2**.

321   **4   A CONCEPTUAL MODEL FOR REPRESENTING DISTURBANCE VULNERABILITY**

322   The previous sections have illustrated that ecological networks are spatially and temporally dynamic,

323   and that failure to represent these complexities within ecological disturbance models is an over

324   simplification of reality. By recognising these complexities and integrating them into ecological

325   network models, a more solid foundation for disturbance modelling can be formulated. The final

326   conceptual model for wildlife disturbance vulnerability is constructed from variables that fall into

327   one of eight categories discussed below.

328   *Spatial accuracy* – Given that disturbance potential for an ENC warrants analysis at a fine spatial

329   scale, the accurate representation of disturbance source and receptor positions is of paramount

330   importance. In creating a 30 m protection zone for an otter holt, recorded using GPS for example, a

331   positional accuracy of 10 m has the potential for significant misalignment of the mapped and real

332   world protection zones. Whilst differential GPS can act to reduce inaccuracy problems (Rempel and

333   Rodgers, 1997), such resources are not always available in low to medium budget projects.

334   Therefore, positional accuracy, which is often reported by the GPS device itself, can be used to

335   buffer the source or receptor to yield a polygon containing possible positions. In cases where

336   positions have been inferred, such as MCP analysis to yield a home range, accuracy may not be so

337   easily ascertained, and must therefore be interpreted by the ecologist who made the field

338   observations, or calculated statistically.

339   *Two dimensional proximity* – This is the main focus of the current disturbance protection method

340   advocated by Scottish Natural Heritage (2013a), which involves buffering disturbance protection

341   zones around shelters. It should, however, be noted that classification of the study area into binary

342   categories of 'susceptible to disturbance' and 'not susceptible to disturbance' is not reflective of a

343   disturbance magnitude that dissipates continuously with respect to distance from the source (see

344   Reed et al., 2012). It should also be recognised that the distances currently used to draw these

345   protection zones around ENCs, although based on limited ecological knowledge, have already been

346   accepted into best practice guidance and policy. Thus, the conceptual model proposed here seeks to

347   represent this uncertainty rather than encourage a potentially contrary methodology. A disturbance

348   susceptibility surface could be employed here to represent the graduation in disturbance

349   susceptibility with respect to distance from the ENC. This could be interpolated by assigning a 100%

350   susceptibility to the ENC and a 5% susceptibility at the protection zone radius proposed under best

351   practice guidance.

352   *Proximity in the third dimension* – Extending the two dimensional proximity concept to incorporate

353   relative differences in height between disturbance source and receptor can reflect multiple

354   processes and forces. This promotes greater consideration for elevated ENCs such as bat roosts atop

355   tall buildings, which should be considered less susceptible to ground based disturbances such as

356   digging. This can be represented as a modification to the two dimensional susceptibility raster by

357   reducing the susceptibility value proportional to the vertical separation. The rate at which this

358   susceptibility is reduced in this way may also depend on whether the source is above or below the

359   receptor. Where a badger sett is dug into a slope for example, disturbance susceptibility is less

360   significant below the sett tunnels than above, since the compaction force only operates above and

361   not below the structure.

362   *Medium composition* – This factor can be used to represent how resilient an ENC is to disturbance. A

363   shelter dug into sand is more likely to collapse when compared to a shelter dug under concrete slabs

364   when exposed to vibration for example. Equally, pippistrelle bat roosting activity has been shown to

365   vary with light intensity (Downs et al., 2003); thus, introduction of new light sources may need to be

366   more stringently controlled near roosts in buildings with windows, or in trees, than in other types of

367   roost. This would be represented in the same way as foraging resources, i.e. each parcel in a GIS

368   layer depicting the modelled medium can be assigned a score of resistivity, in much the same way as

369   is done in groundwater flow modelling. Separate resistivity values can be given for different types of

370   disturbance. A building for example may have a high resistivity to light pollution but a low resistivity

371 to noise. It is however, unlikely that validated models will exist for all required resistances and as

372 such may need to be given subjective weightings by appropriate experts.

373 *Tolerance* – Some species are more tolerant of disturbance than others. This has been particularly

374 noted through the study of bird nest proximities to main roads (Hockin et al., 1992). It may therefore

375 be beneficial to not only apply resistivity scores to the medium within which the ENC is situated, but

376 also to the ENC itself, based on the species that utilises it. Again, this may need a subjective

377 weighting devised by experienced ecologists. In addition, this parameter can be modified to

378 demonstrate community resilience. An urban otter family, established near a railway embankment

379 for example, may be less susceptible to vibration and noise disturbance than an isolated rural

380 community. This can be found by proximity analysis of existing ENCs to existing disturbance sources,

381 and by considering the magnitude of the disturbance.

382 *Position in breeding cycle* – Breeding periods signify particularly vulnerable times for wildlife,

383 however only partial consideration for this aspect is undertaken in the current disturbance

384 protection guidance. Although statutory regulators may authorise unavoidable disturbance outside

385 of the breeding season, this concept of temporally variable disturbance risk is generally not reflected

386 in the criteria for determining whether disturbance will occur. One exception to this rule is the

387 protection zone of an otter holt which increases from 30 m to 100 m if the holt is found to be natal

388 (Scottish Natural Heritage, 2013b). As previously discussed in section 3.2, the representation of

389 temporal data within a GIS can be problematic. One solution to this may be to utilise custom GIS

390 coding to compare the system clock to known breeding times. The disturbance risk can then be

391 intensified or relaxed around the modelled ENC accordingly.

392 *Position in sleep cycle* – This is also used to represent temporal fluctuations in disturbance

393 susceptibility, but unlike breeding cycles, the period of the fluctuation is variable. Badgers for

394 example are nocturnal creatures and Forestry Commission guidelines (The Forestry Commission,

395 1995) suggest that works in close proximity to setts should not be undertaken two hours either side

396   of dawn and dusk. Thus, two four hour windows of increased disturbance risk are proposed in a 24

397   hour period. However, this is complicated by the fact that dawn and dusk times vary according to

398   the time of year. Additionally, this concept can also be used to highlight changes to disturbance

399   susceptibility during hibernation. These can be modelled and represented in much the same way as

400   is suggested for the position in the breeding cycle parameter.

401   *Disturbance index* – Although best practice guidance tends to account for the intensity of a

402   disturbance activity (Scottish Natural Heritage, 2013a and English Nature 2002), it does not consider

403   the more complex relationship between activity intensity and duration. It is reasonable to assume

404   that a moderate intensity, long duration disturbance event is as likely to disturb wildlife as a high

405   intensity, short duration event. Equally, the frequency of disturbance events may also play a

406   significant role in determining the overall disturbance magnitude. Thus, a disturbance index can be

407   conceived to be a function of disturbance intensity, duration and frequency. Additionally, there may

408   be a rate of disturbance increase where the animal becomes accustomed to the activity rather than

409   experiencing disturbance, meaning that higher levels of noise or light, for instance, could be

410   permitted. Although units for duration are standard for each different disturbance type, units for the

411   magnitude for each will differ. Light for example is measured in candela, noise is measured in

412   decibels, and seismic vibration is measured as a displacement acceleration in metres per second

413   squared. This means that separate disturbance indices must be modelled for each type of

414   disturbance.

415   The concept of the ecological disturbance model builds upon the utilisation distribution

416   methodologies discussed in section 3.1.2 by representing a probability of use gradient within the

417   home range (at a scale that is fine enough to show preferred traveling routes and shelters) for

418   discrete time periods. The granularity of these periods will depend upon the variability of habitat

419   usage for the modelled species at the studied site, and would need to be agreed by ecological

420   experts on a case by case basis. Additional modelling can be conducted on these (raster) surfaces to

421   infer the likelihood of disturbance at a point in time, based on knowledge of the species ecology

422   such as breeding seasons and periods of scarce resource availability. Once the vulnerability gradient

423   map has been created, a disturbance index for a disturbance event can be calculated and compared

424   to the map to finally assess the likelihood of disturbance for animals occupying the site. These steps

425   are illustrated in Figure 3 for a single badger territory at the case study site, during the breeding

426   season.

427   **Figure 3**.

428   **5 CONCLUSION**

429   This paper has outlined a new approach to wildlife disturbance susceptibility conceptualisation,

430   recording and representation within a GIS which accounts for ecological complexities and

431   uncertainties in both space and time. By building up a digital representation of the spatio-temporal

432   relationships between species and their foraging resources, home ranges, territories, pathways and

433   shelters, the effects of disruption to any particular instance of these, at a given time, can be

434   modelled. In addition, by considering spatio-temporal uncertainties in both data recording and GIS

435   representation, insight can be given into the reliability of such generated data. Ultimately, an

436   improved representation of development constraints due to wildlife disturbance considerations will

437   facilitate better synchronisation between development activity and periods of heightened

438   disturbance susceptibility. This may well pave the way for developments that would otherwise have

439   been rejected under static modelling of the worst case scenario, whilst simultaneously acting to

440   protect wildlife when they require it the most.

441   Although adoption of the proposed methodology will necessitate an increase in time, computing

442   resources and data to undertake, these disadvantages are likely to be outweighed by a number of

443   additional advantages. First, the proposed methodology can transparently demonstrate the

444   ecological importance of ENCs, and could alleviate current developer frustrations emanating from a

445 perception that the ecological industry is overly bureaucratic (see Coleridge, 2013), confusing

446 (Department for Environment Food and Rural Affairs, 2013) and inflexible. Second, a GIS approach

447 facilitates not only a more robust approach to disturbance modelling, but also provides a platform

448 upon which modelling results can be shared. This may stimulate a greater adoption of ecological

449 concerns into development planning and ease data integration between other development

450 stakeholders. Third, representing spatio-temporal dynamics of wildlife vulnerability (as opposed to

451 the static, discrete parcels proposed under some current approaches) would mark a move toward

452 data that more accurately reflect the real world processes it represents. Calls for such data are not

453 novel to the ecology industry but have been a criticism of GIS-based approaches to spatial modelling

454 in a wide variety of applications for some time (Goodchild et al., 2007).

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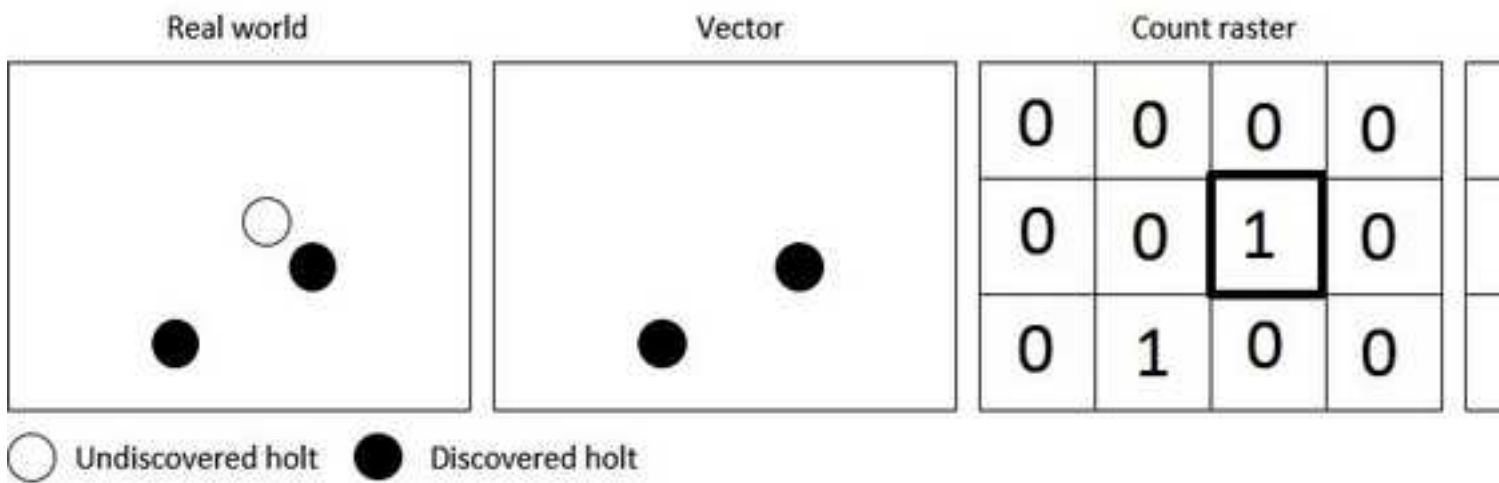
614

LIST OF FIGURE CAPTIONS

Figure 1 – Illustration to demonstrate how an otter holt that is missed during an ecological survey can affect the truthfulness of different representations. The real world shows three holts, one of which has been missed during the survey. The vector representation can be considered to be untruthful since one holt is missing. It does, however, show the greatest degree of positional precision. The count raster is also incorrect since the cell highlighted in bold should contain two holts. The binary raster can be considered truthful since it only shows presence and absence, however this generalisation renders it unsuitable for some types of analyses.

Figure 2 - Temporal differences in disturbance vulnerability illustrated for different areas of a hypothetical site. Due to the temporal variations in disturbance vulnerability, overall disturbance could be minimised by conducting site works in the winter, in the northwest of the site, and during summer in the southeast.

Figure 3 - Increasing levels of complexity in the representation of ecological disturbance vulnerability, illustrated for a badger network. (a) Discrete vectors representing sett entrances, paths and feeding areas. (b) Raster interpolated from discrete protection zones, adjusted for spatial uncertainties and incorporating vertical constraints (c) Raster then adjusted for seasonality giving greater protection to the breeding setts. (d) Vulnerability map created by adjusting for railway. (e) Probability of disturbance for 3 potential disturbance events.

Figure 1

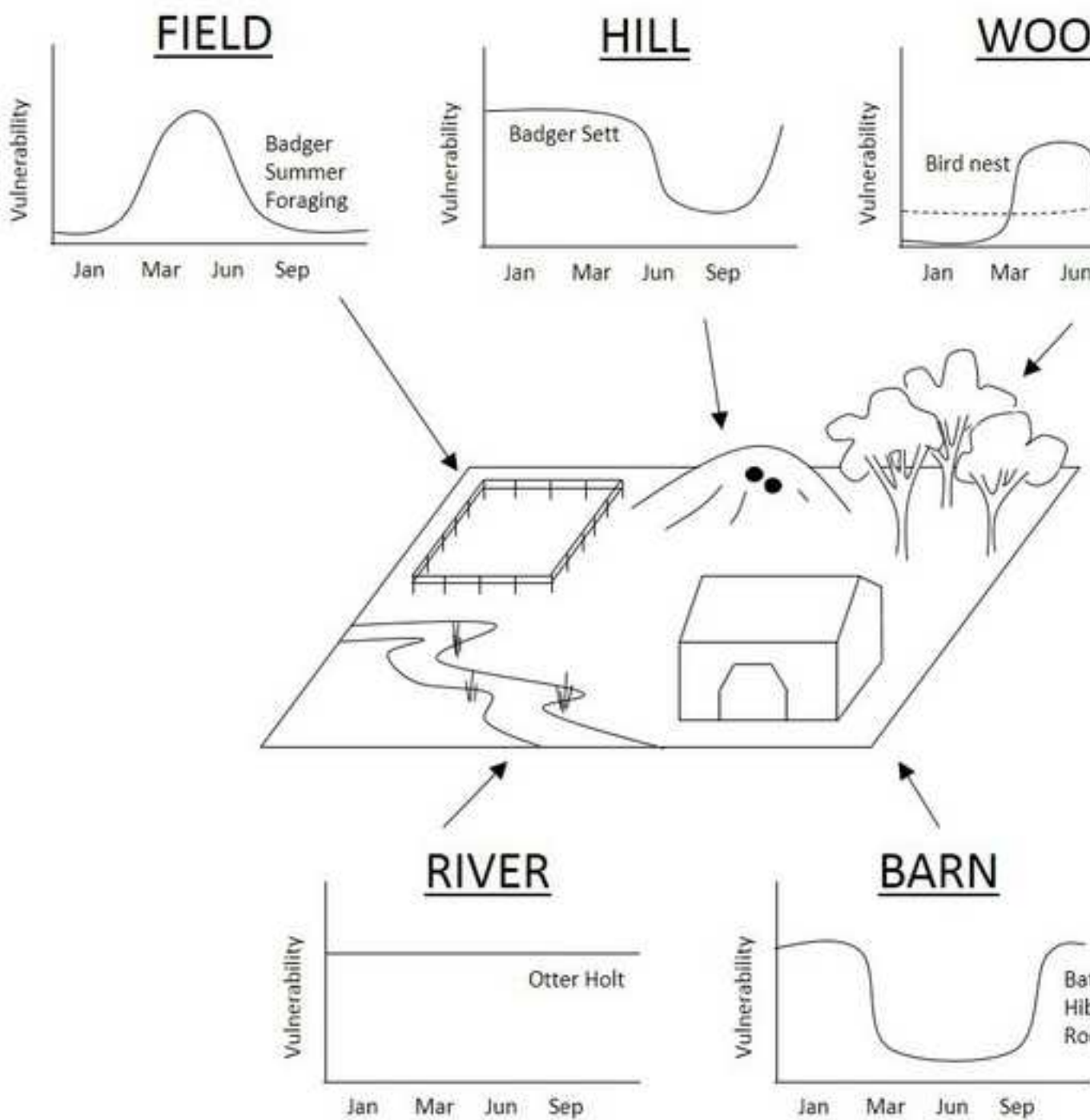Real world   Vector   Count raster

○ Undiscovered holt   ● Discovered holt

Figure 2
[Click here to download high resolution image](#)

Figure 3