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***Highlights (for review)**

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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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ABSTRACT

Assessing the potential environmental impacts of disturbance on protected species during and after the development process is a legislative requirement in most nations. However, the restrictions that this legislation places on developers are often based on limited ecological understanding, over-simplified methodologies, less-than-robust data and the subjective interpretations of field ecologists. Consequently, constraints may be imposed with no transparent methodology behind them to the frustration of, and occasionally large expense to, developers. Additionally, protected species numbers continue to decline and biodiversity continues to be threatened. This paper describes a GIS conceptual model for assessing ecological disturbance vulnerability, based upon a case study development in Scotland. First, uncertainties in traditional methods of recording and representing ecological features with GIS are reviewed such that they may be better accounted for in the disturbance model. Second, by incorporating temporal fluctuations in ecological behaviour into the disturbance susceptibility concept, it is argued that it is possible to synchronise development with conservation requirements. Finally, a method is presented to account for disturbance tolerances at the scale of the individual animal. It is anticipated that this model will enable environmental impact assessors to produce more robust analyses of wildlife disturbance risk and facilitate synchronisation between development and wildlife vulnerability to minimise disturbance and better avoid delays to the works programme.

KEYWORDS

Ecological Disturbance; Development sites; Protected species; GIS representation; Ecological networks; Temporal ecology

1 INTRODUCTION

Increasing demand for housing, commerce and industry, driven by an expanding human population, is perpetuating global urban development (Millennium Ecosystem Assessment, 2005). However, as global landscapes become increasingly urbanised, space available for new development becomes ever more constrained. This constraint is compounded by the need to maintain multifunctional landscapes that promote prosperity for both humans and wildlife (Angold et al., 2006, Rudd et al., 2002) for intrinsic purposes, and to continue the delivery of ecosystem services (Millennium Ecosystem Assessment, 2005). For these reasons, a series of legislative measures have been

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introduced affording legal protection to selected species and habitats deemed to be under threat or of particular cultural significance. Central to this concept of protection is the notion of ‘wildlife disturbance’, such as that caused by excessive noise, vibration, loss of food sources or the introduction of new predators. Disturbance is prohibited for certain species under the Habitats Directive (European Commission, 1992) and Birds Directive (European Commission, 2009), and various national laws devolved from them. There is, however, no universally accepted definition of the term ‘wildlife disturbance’ leaving it open to interpretation in best practice guidance issued by statutory regulators. Consequently, environmental impact assessments with a wildlife disturbance component are open to subjectivity and lack a standardised approach.

A more in-depth model of wildlife susceptibility to disturbance is clearly needed to reduce levels of subjectivity and improve the means by which development constraints are integrated into development programmes. Given that the risk of disturbing an animal is largely subject to spatial criteria, Geographical Information Systems (GIS), offer a solid foundation upon which to achieve this task. GIS data are also easily displayed via a website or server system, facilitating communication, and may be viewed at multiple scales to better understand a sites context and broader landscape connections. GIS has already established its credentials as a planning tool in ecology, for example, in the design of wildlife corridors (Jenness et al., 2010), nature reserves (Ball et al., 2009) and habitat restoration schemes (Rempel, 2008).

Goodchild (2007) defines three levels of abstraction between real world phenomena and GIS representation - conceptualisation of the processes and interactions inherit to the studied phenomenon, recording of the variables of interest and representation of the variables in appropriate digital form. In the context of wildlife disturbance on a development site there are challenges at each level of abstraction.

Difficulties at the conceptual level are illustrated through a lack of legislative clarity in the definition of wildlife disturbance. Such ambiguity has led to differing requirements for the treatment of badger

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(*Meles meles*) setts for example, protected from disturbance in the UK by the Protection of Badgers Act 1992. In Scotland, the current requirements for compliance with this legislation involves creating a protection zone of 30 m around the sett, within which potentially disturbing activities are prohibited (Scottish Natural Heritage, 2013a). However, the English approach (English Nature, 2002) leaves the interpretation of disturbance to the field ecologist. Whilst the English approach can facilitate a more complex conceptualisation of wildlife disturbance, its application could be biased by social or cultural values and research specialisations of the individual ecologist. Conversely, the problem with the Scottish methodology is twofold: first, the discrete representation undermines the obvious distance decay in disturbance probability with respect to proximity to the sett entrance; second, the protection zone radius of 30 m, based on tunnel lengths of excavated setts (Raynor, 2012) is designed to protect the sett structure, giving little consideration to adverse effects of noise or vibration upon the badger inside. In more extreme cases of physical disturbance such as pile driving or blasting, the protection zone is increased to 100 m radius (Scottish Natural Heritage, 2013a), although the justification for this distance is not given in any of the reviewed literature.

At the recording level, challenges arise from not being able to monitor wildlife completely and directly in the field. For example, whilst animals are frequently GPS or radio tagged, giving insight into their spatio-temporal positions, which in turn allows the derivation of home range (Powell, 2000), interaction patterns (Handcock et al., 2009) and travelling routes (Nams, 2005), the capture and tagging of every animal on a large development site is impractical. Thus, ecological knowledge is often derived from field signs including faeces, hair, prints and scratches, which are easy to miss in the field (Parry et al., 2013). Although such uncertainties are reducing with the introduction of video technology (Moll et al., 2007), surveillance generally covers only a small area of a given site and a sub-selection of individual animals.

At the representational level, challenges chiefly arise from the temporally-dynamic nature of ecosystem functioning, thus affecting the severity and likelihood of wildlife disturbance. Bats for example, are extremely reliant upon undisturbed hibernation in order to maintain sufficient fat supplies to last the winter (Thomas, 1995). Similarly, the Forestry Commission (1995) advocates the cessation of works in close proximity to badger setts at dawn and dusk to allow its occupants to move in and out, illustrating the species' dependence upon daylight cycles. The lack of an innate temporal query language within most GIS applications means that answering questions regarding when a particular operation (e.g. pile driving) should be conducted to coincide with periods of low disturbance likelihood is difficult.

This paper offers a detailed assessment of the conceptual, recording and representational challenges faced in communicating wildlife disturbance constraints within a GIS for a case study site in the Central Lowlands of Scotland. To begin with, potential receptors to disturbance are discussed, along with uncertainties in their traditional GIS representations. Temporal fluctuations in disturbance susceptibility are then considered, and insights offered into how anthropogenic disturbance may be synchronised with cyclic variations in wildlife activity to minimise disturbance. Finally, in the light of these discussions, a conceptual model for wildlife disturbance vulnerability is proposed. The model accounts for a more detailed understanding of wildlife ecology and encompasses spatio-temporal uncertainties in ecological knowledge.

2 METHODOLOGY

Challenges are illustrated using ecological examples from a large (10 km^2) brownfield site, located 20 km west of Glasgow, Scotland. The site has proposals for 2500 units of housing, a 150,000m^2 business park, related infrastructure and a community woodland park (Renfrewshire Council, 2014). During the sites' extended period of phased decommissioning, public access has been heterogeneously restricted, meaning that some areas of the site have remained undisturbed for over 75 years, whilst others were in use until 2002. Such conditions have yielded an ecological mosaic exhibiting a wide variety of successional traits, and have encouraged a number of protected species to occupy the site including 12 badger social groups (average 3-4 adults per group plus cubs),

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3 otters (plus cubs), 2 breeding pairs of barn owls, small populations of common pipistrelle, brown long-eared and Daubenton's bats and 2 breeding pairs of little ringed plover (results of monitoring by the author since 2012).

Surveys for a number of species were undertaken using methods described in the Chartered Institute of Ecology and Environmental Management's (CIEEM) Technical Guidance Series (Chartered Institute of Ecology and Environmental Management, 2014). These surveys are designed to be conducted at fine spatial scales, to determine specific landscape features, such as a particular hedgerow or building, considered important for conservation of an individual animal or social group. This scale is equivalent to Johnson's (1980) fourth order of hierarchical habitat selection; the first being the geographical distribution of the species, the second being the selection of a home range within that distribution, and the third being the usage of generalised habitat patches within the home range. By adopting a fine spatial scale we constrain our study to disturbance concepts affecting individual animals and social groups, rather than wider populations or meta-populations. Conceptual, recording and representation challenges were then identified by comparing the practical requirements of field survey and GIS design with ecological disturbance concepts highlighted throughout the literature. These concepts were augmented with our own critical thinking and then consolidated into a conceptual model that encapsulates a more complete idea of the potential for wildlife disturbance due to different activities.

3 RESULTS AND DISCUSSION

3.1 INFERRING THE UNKNOWN FROM THE KNOWN

Initially, an understanding of how protected species use their available space is necessary in order to determine both the projected impact of a disturbance event across the ecological network, and the likely position of an individual animal within it at any given time. However, each of the features that comprise the ecological network, such as foraging resources, pathways and shelters (collectively referred to hereafter as ecological network components (ENCs)) are often difficult to observe, record

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and map comprehensively (Parry et al., 2013). In seeking to improve communication of wildlife disturbance constraints, it is important to understand the limitations of the underlying ecological data upon which higher level knowledge can be derived. By integrating associated uncertainties that will be inherent within the disturbance model, decision makers can be better informed as to the reliability of disturbance projections, and field ecologists can learn where best to target their surveys as part of any monitoring efforts. The following section highlights some of these uncertainties and generalisations for different ENCs observed at the case study site. It is not intended to be an exhaustive list but rather to draw attention to nature of uncertainties in ecological data and to stimulate discussion and critical thinking.

3.1.1 Foraging resources

Food and water act as vital nodes within the ecological network and their quantities play a significant role in the estimation of carrying capacities (for a review see McLeod, 1997). However, given the spatial scale of the case study site, it was not possible to record the availability of food directly or completely. For example it was not feasible to count the number of berries available across all woodland and scrub patches, nor was it possible to account for movements of each and every live prey individual between territories of carnivores. Habitat types are therefore often used as a surrogate for food availability (e.g. Anderson et al., 2005, Scottish Natural Heritage, n.d.). Whilst this generalisation forms a convenient measure by which to estimate resource availability, it assumes a homogeneous distribution of resource biomass throughout each mapped habitat parcel. Biomass can however vary significantly between microhabitats (Shakir and Dindal, 1997, Shevtsova et al., 1995).

Moreover, the habitat categorisation processes typically used by commercial ecologists (e.g. Phase 1 Habitat Classification (Nature Conservancy Council, 1990) and the National Vegetation Classification (NVC) (Rodwell, 2006)) required for a vector-based GIS representation, often renders habitat boundaries that are not always agreed upon by ecologists (Stevens et al., 2004). Consequently, a discrete approach to mapping habitats may yield highly uncertain estimates of foraging resource availability. Whilst there are alternative habitat representations available that use a raster model, such as the Land Cover Map (Morton et al., 2011), these do not reflect the continuous nature of habitat and food availability transitions through space. Rather, they depict the same discrete land cover classes as the vector models, but are spatially generalised over grid cells of 25 m or 1 km (Morton et al., 2011).

3.1.2 Home range and territory

The amount of foraging resources available to an individual animal is constrained by its home range or territory. Individuals with a home range or territory categorised by poor foraging opportunities are more vulnerable to loss of resources (Kitaysky et al., 1999). By using such data to influence land use change, under some scenarios it may be possible to dilute the effects of disturbance across multiple territories or home ranges to the point where it is no longer significant to any one individual. Multiple methodologies have been developed to derive home ranges from field data with varying degrees of complexity and realism. The simplest approach, and the one usually applied in commercial UK ecology, is to utilise a uniform radius buffer around shelters with a distance extrapolated from literature or ‘expert’ opinion (English Nature, 2002, Scottish Natural Heritage, 2013a, b). However, this approach is also the most generalised as it does not account for factors such as habitat composition, which can cause variations in home range size from individual to individual (Börger et al., 2006, Anderson et al., 2005), Consequently, the home range in reality may be markedly different to that formulated using this methodology.

Slightly more representative of the real world, Minimum Convex Polygon analysis (MCP) (Nilsen et al., 2008, Börger et al., 2006) can be utilised to derive home ranges and territories from a variety of spatial data (depending on the species) including field signs, direct observations, location tracking, camera trap data and echo-sounding. This has the advantage over the buffering approach in that the analysis is based on data taken directly from the study area, but is nevertheless subject to a significant number of limitations highlighted by Worton (1987) including bias caused by sample size and a tendency to overestimate range sizes. The MCP methodology has, however, been

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demonstrated to be effective for delineating badger territories from bait marking data (Delahay et al., 2000) as latrines are highly characteristic of badger territory edges (Roper et al., 2001). Conversely, territories of avian species derived from MCP analysis of field signs such as sightings, droppings, down, and in particular, subjective interpretation of song (Bibby et al., 2000) are likely to yield home range boundaries of much lower certainty.

A further limitation to MCP analysis is that the derived territories and home ranges are discrete, thus giving no representation of spatial usage within them. Greater insight into the internal structure of home ranges and territories can be gained through the application of utilisation distribution methodologies including kernel density (Worton, 1989, Seaman and Powell, 1996, Powell, 2000, Börger et al., 2006), time-geographic density estimation (Downs et al., 2011) and random walk models (Horne et al., 2007) to generate a representation of probability of usage of space within each home range or territory. GIS representation of this internal 'probability-of-use' gradient then becomes more suited to a raster approach (continuous variation), rather than a vector approach (discrete objects). Whilst these types of analyses are also subject to uncertainties, such as that caused by spatial-autocorrelation (Blundell et al., 2001), the chief concern here is that they require considerable geo-statistical skill to compute and interpret, and also tend to rely upon telemetry data. In the UK protected species require a licence to trap and fit with a GPS device. Combined with the complexity of utilisation distribution methodologies this may lead practicing ecologists to implement the less robust MCP methodologies as at the case study site.

3.1.3 Pathways

Pathways link areas used for shelter and foraging together, thus are an essential component of the ecological network. As with home ranges and territories, pathways are often inferred from field signs yielding different levels of certainty depending upon the species studied. Some species on the case study site, such as badgers, follow well-defined paths and these can often be recognised in the field by the trained ecologist (Kyne et al., 1990, Neal and Cheeseman, 1996). Badger paths also tended to follow prominent linear features, such as hedgerows, woodland edges and walls, which is

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consistent with other observations made by Feore and Montgomery (1999), and Hutchings and Harris (Hutchings and Harris, 2001). Badgers are also known to show little variation in traveling routes, which in turn facilitates the mapping of these features as discrete lines, but also signifies greater sensitivity of the animals to path disturbance. In contrast, it is more difficult to define an exact flight path for birds and bats since they show more spatio-temporal variability in their travelling routes, their movements leave no field signs such as footprints or flattened grass to observe and their movements must be considered in three dimensions.

The inference of such ‘fuzzy’ pathway locations may be improved by additional observation, location tracking or camera traps to help improve data reliability. Additionally, a least cost path can be inferred, based on known shelters, foraging locations and movement impedances or preferences. Davies et al. (Davies et al., 2012) for example inferred travelling routes for pipistrelle bats *(Pipistrellus pipistrellus)* constrained by lighting sources (known features of avoidance) and hedgerows, which are known to be preferred travelling routes for the species (Mitchell-Jones, 2004). Nonetheless, given the ‘fuzzy’ nature of these features, vector representation as discrete lines can imply greater accuracy in the data than is appropriate. Murdock and Potts (Murdock and Potts, 2009) interviewed ornithologists, and found the estimated positional accuracy of such surveys to be ±50 m. Hence, it may be more appropriate to represent these paths as rasters with a cells size of 50 m or generate 25 m buffer zones around the vector lines to reflect this spatial uncertainty.

3.1.4 Shelters

As with other ENCs, acquiring a comprehensive dataset of all protected shelters in a large study area can be difficult. Otter *(Lutra lutra)* holts in particular are notoriously difficult to locate due to their often secluded position, camouflage and underwater entrances (Parry et al., 2013). Even when shelters are discovered, the attribution of a species to that shelter can be problematic. Since many shelters are subterranean, both occupation and species are typically inferred from field signs, such as the shape of the tunnel, foot prints, faeces, hair and scratches found in close proximity to the shelter. Although ecological surveyors are trained in using these field signs to find shelters and infer

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species occupation, the process is not infallible, meaning that some shelters may be missed or that the wrong species may be assigned to it. The latter is particularly problematic when considering legislative-driven constraints on wildlife disturbance. Consider, for example, that female foxes are known to use unoccupied badger setts to rear their young (Trewby et al., 2008), but that they are not protected from disturbance by wildlife legislation. Moreover, it is difficult for the ecologists to know if the sett is likely to be used by a badger in the future, thus the decision as to whether to continue to control nearby anthropogenic disturbance, is equally problematic from a legislative perspective.

Similar legal uncertainties also surround the protection status of otter resting places and bat feeding perches. UK legislation and best practice guidance (e.g. Mitchell-Jones, 2004, Scottish Natural Heritage, 2013b) offers no insight into the frequency of use necessitated for these features to receive protection, thus their explicit inclusion within a disturbance susceptibility model may place unnecessary restrictions on development works if alternative areas are available. Perhaps a more meaningful disturbance analysis would consider areas with the potential to be used by wildlife as resting places or perches, rather than simply attempting to establish those in current use. A similar methodology to that used in habitat suitability modelling could be used (for reviews see Fielding and Bell, 1997, Hirzel et al., 2002, Hirzel and Le Lay, 2008); however, further research would be required to produce such a model for each relevant feature.

Uncertainty is not only introduced at the recording level but at the digital representation level too. Shelters are usually represented within a GIS as vector points or generalised into raster cells for larger scale studies (e.g. National Biodiversity Network, 2012). In the latter case the raster can either indicate a count of shelters present, or simply indicate presence in a Boolean format. Whilst the Boolean representation is subject to the most spatial and thematic generalisation, it is less prone to error (see Figure 1). However, in the case of disturbance modelling at the site scale, the locational precision of shelter data offered by the vector representation is imperative to formulating the most

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robust strategies of avoidance and mitigation. The issue with this, as with pathway representation as vectors lines, is that a degree of precision and accuracy is instilled within the data that may lead to false conclusions regarding its reliability.

Figure 1.

3.2 THE ISSUE OF TIME

Development at the case study site is being conducted over a multi-phased programme, lasting in excess of a decade. Consequently, temporal ecological factors, which may be overlooked in development projects conducted over shorter timescales, warrant significant attention here. As illustrated in section 3.1.4, GIS models of representation can be used to generalise the spatial dimension; in much the same way, a single GIS layer, in the absence of other thematically related layers, can be considered a generalisation of the temporal dimension. Many species exhibit spatial variation in their home ranges through time including otters (Erlinge, 1967), deer (Börger et al., 2006) and badgers (Cresswell and Harris, 1988). For hibernating species such as bats, the spatial extent of their home ranges will decrease to nothing during winter. Thus, any GIS layer depicting a home range for a single instance in time must, in the absence of supplementary data, be taken by the researcher to be representative of all instances of time. Further, seasonal generalisations occur in modelling food resource availability, since the abundance of nuts and berries for instance are also dependent upon seasonality. This in turn may affect the temporal frequency of pathway use since a community may use different feeding areas in different seasons. Finally, shelter usage may also be dependent on seasonality. A main badger sett, for example, is likely to be active all year round; an outlier however will be more frequently used in the breeding season when territorial behaviour peaks (Cresswell et al., 1992). These concepts are illustrated for different species in Figure 2. Seasonality is not the only driver for temporal generalisation, daily cycles also play a significant role in governing ecological behaviour, and subsequently in determining the reliability of the GIS data representing it. Diurnal shelter usage is chiefly governed by the sleeping cycle of the animal, so for a

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nocturnal animal, their shelter is likely to be in use during daytime. Thus, in a temporally-enabled GIS, the frequency at which data is recorded, and the granularity at which it is modelled, will govern the generalisation of the time dimension.

Temporal generalisation has particular significance when using ecological GIS data to influence landscape management decisions. If the protection of the ENCs described above are considered to be of primary importance in species conservation, then a failure to account for their temporal fluctuations, both spatially and thematically, may result in under or over protection at a given time. Under protection may result in development which will impact negatively upon local wildlife, whereas over protection will place unnecessary strain on developers' time and resources. This has particular relevance for constraining site operations such as cable laying, safety lighting, blasting and pile-driving where wildlife disturbance may only be temporary. Undertaking these works near a winter food source, travelling route or shelter may be considered acceptable in the summer for example, provided alternative resources are available.

Unfortunately, the representation of the temporal dimension within GIS can be problematic and is a primary focus of past and current GIS research (Armstrong, 1988, Erwig et al., 1999, Huang and Claramunt, 2005, Pelekis et al., 2004, Raper, 2012). Although a full discussion on approaches and methods of time representation in a GIS is beyond the scope of this paper, it is important to note here that the vast majority of solutions require customised software, bespoke query languages and considerable GIS skill to implement and maintain. In the case of commercial ecology this becomes unfeasible since data sharing between stakeholders, whose specialisms are likely to lie beyond the domain of GIScience, is of paramount importance. We therefore argue that the extension of GIS into the temporal dimension needs to be conducted in conjunction with existing propriety or open source software, making solutions accessible, intuitive and interoperable with other data sources.

Figure 2.

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4 A CONCEPTUAL MODEL FOR REPRESENTING DISTURBANCE VULNERABILITY

The previous sections have illustrated that ecological networks are spatially and temporally dynamic, and that failure to represent these complexities within ecological disturbance models is an over simplification of reality. By recognising these complexities and integrating them into ecological network models, a more solid foundation for disturbance modelling can be formulated. The final conceptual model for wildlife disturbance vulnerability is constructed from variables that fall into one of eight categories discussed below.

Spatial accuracy – Given that disturbance potential for an ENC warrants analysis at a fine spatial scale, the accurate representation of disturbance source and receptor positions is of paramount importance. In creating a 30 m protection zone for an otter holt, recorded using GPS for example, a positional accuracy of 10 m has the potential for significant misalignment of the mapped and real world protection zones. Whilst differential GPS can act to reduce inaccuracy problems (Rempel and Rodgers, 1997), such resources are not always available in low to medium budget projects. Therefore, positional accuracy, which is often reported by the GPS device itself, can be used to buffer the source or receptor to yield a polygon containing possible positions. In cases where positions have been inferred, such as MCP analysis to yield a home range, accuracy may not be so easily ascertained, and must therefore be interpreted by the ecologist who made the field observations, or calculated statistically.

Two dimensional proximity – This is the main focus of the current disturbance protection method advocated by Scottish Natural Heritage (2013a), which involves buffering disturbance protection zones around shelters. It should, however, be noted that classification of the study area into binary categories of ‘susceptible to disturbance’ and ‘not susceptible to disturbance’ is not reflective of a disturbance magnitude that dissipates continuously with respect to distance from the source (see Reed et al., 2012). It should also be recognised that the distances currently used to draw these protection zones around ENCs, although based on limited ecological knowledge, have already been

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accepted into best practice guidance and policy. Thus, the conceptual model proposed here seeks to represent this uncertainty rather than encourage a potentially contrary methodology. A disturbance susceptibility surface could be employed here to represent the graduation in disturbance susceptibility with respect to distance from the ENC. This could be interpolated by assigning a 100% susceptibility to the ENC and a 5% susceptibility at the protection zone radius proposed under best practice guidance.

Proximity in the third dimension – Extending the two dimensional proximity concept to incorporate relative differences in height between disturbance source and receptor can reflect multiple processes and forces. This promotes greater consideration for elevated ENCs such as bat roosts atop tall buildings, which should be considered less susceptible to ground based disturbances such as digging. This can be represented as a modification to the two dimensional susceptibility raster by reducing the susceptibility value proportional to the vertical separation. The rate at which this susceptibility is reduced in this way may also depend on whether the source is above or below the receptor. Where a badger sett is dug into a slope for example, disturbance susceptibility is less significant below the sett tunnels than above, since the compaction force only operates above and not below the structure.

Medium composition – This factor can be used to represent how resilient an ENC is to disturbance. A shelter dug into sand is more likely to collapse when compared to a shelter dug under concrete slabs when exposed to vibration for example. Equally, pippistrelle bat roosting activity has been shown to vary with light intensity (Downs et al., 2003); thus, introduction of new light sources may need to be more stringently controlled near roosts in buildings with windows, or in trees, than in other types of roost. This would be represented in the same way as foraging resources, i.e. each parcel in a GIS layer depicting the modelled medium can be assigned a score of resistivity, in much the same way as is done in groundwater flow modelling. Separate resistivity values can be given for different types of disturbance. A building for example may have a high resistivity to light pollution but a low resistivity

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to noise. It is however, unlikely that validated models will exist for all required resistances and as such may need to be given subjective weightings by appropriate experts.

Tolerance – Some species are more tolerant of disturbance than others. This has been particularly noted through the study of bird nest proximities to main roads (Hockin et al., 1992). It may therefore be beneficial to not only apply resistivity scores to the medium within which the ENC is situated, but also to the ENC itself, based on the species that utilises it. Again, this may need a subjective weighting devised by experienced ecologists. In addition, this parameter can be modified to demonstrate community resilience. An urban otter family, established near a railway embankment for example, may be less susceptible to vibration and noise disturbance than an isolated rural community. This can be found by proximity analysis of existing ENCs to existing disturbance sources, and by considering the magnitude of the disturbance.

Position in breeding cycle – Breeding periods signify particularly vulnerable times for wildlife, however only partial consideration for this aspect is undertaken in the current disturbance protection guidance. Although statutory regulators may authorise unavoidable disturbance outside of the breeding season, this concept of temporally variable disturbance risk is generally not reflected in the criteria for determining whether disturbance will occur. One exception to this rule is the protection zone of an otter holt which increases from 30 m to 100 m if the holt is found to be natal (Scottish Natural Heritage, 2013b). As previously discussed in section 3.2, the representation of temporal data within a GIS can be problematic. One solution to this may be to utilise custom GIS coding to compare the system clock to known breeding times. The disturbance risk can then be intensified or relaxed around the modelled ENC accordingly.

Position in sleep cycle – This is also used to represent temporal fluctuations in disturbance susceptibility, but unlike breeding cycles, the period of the fluctuation is variable. Badgers for example are nocturnal creatures and Forestry Commission guidelines (The Forestry Commission, 1995) suggest that works in close proximity to setts should not be undertaken two hours either side

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of dawn and dusk. Thus, two four hour windows of increased disturbance risk are proposed in a 24 hour period. However, this is complicated by the fact that dawn and dusk times vary according to the time of year. Additionally, this concept can also be used to highlight changes to disturbance susceptibility during hibernation. These can be modelled and represented in much the same way as is suggested for the position in the breeding cycle parameter.

Disturbance index – Although best practice guidance tends to account for the intensity of a disturbance activity (Scottish Natural Heritage, 2013a and English Nature 2002), it does not consider the more complex relationship between activity intensity and duration. It is reasonable to assume that a moderate intensity, long duration disturbance event is as likely to disturb wildlife as a high intensity, short duration event. Equally, the frequency of disturbance events may also play a significant role in determining the overall disturbance magnitude. Thus, a disturbance index can be conceived to be a function of disturbance intensity, duration and frequency. Additionally, there may be a rate of disturbance increase where the animal becomes accustomed to the activity rather than experiencing disturbance, meaning that higher levels of noise or light, for instance, could be permitted. Although units for duration are standard for each different disturbance type, units for the magnitude for each will differ. Light for example is measured in candela, noise is measured in decibels, and seismic vibration is measured as a displacement acceleration in metres per second squared. This means that separate disturbance indices must be modelled for each type of disturbance.

The concept of the ecological disturbance model builds upon the utilisation distribution methodologies discussed in section 3.1.2 by representing a probability of use gradient within the home range (at a scale that is fine enough to show preferred traveling routes and shelters) for discrete time periods. The granularity of these periods will depend upon the variability of habitat usage for the modelled species at the studied site, and would need to be agreed by ecological experts on a case by case basis. Additional modelling can be conducted on these (raster) surfaces to

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infer the likelihood of disturbance at a point in time, based on knowledge of the species ecology such as breeding seasons and periods of scarce resource availability. Once the vulnerability gradient map has been created, a disturbance index for a disturbance event can be calculated and compared to the map to finally assess the likelihood of disturbance for animals occupying the site. These steps are illustrated in Figure 3 for a single badger territory at the case study site, during the breeding season.

Figure 3.

5 CONCLUSION

This paper has outlined a new approach to wildlife disturbance susceptibility conceptualisation, recording and representation within a GIS which accounts for ecological complexities and uncertainties in both space and time. By building up a digital representation of the spatio-temporal relationships between species and their foraging resources, home ranges, territories, pathways and shelters, the effects of disruption to any particular instance of these, at a given time, can be modelled. In addition, by considering spatio-temporal uncertainties in both data recording and GIS representation, insight can be given into the reliability of such generated data. Ultimately, an improved representation of development constraints due to wildlife disturbance considerations will facilitate better synchronisation between development activity and periods of heightened disturbance susceptibility. This may well pave the way for developments that would otherwise have been rejected under static modelling of the worst case scenario, whilst simultaneously acting to protect wildlife when they require it the most.

Although adoption of the proposed methodology will necessitate an increase in time, computing resources and data to undertake, these disadvantages are likely to be outweighed by a number of additional advantages. First, the proposed methodology can transparently demonstrate the ecological importance of ENCs, and could alleviate current developer frustrations emanating from a

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perception that the ecological industry is overly bureaucratic (see Coleridge, 2013), confusing (Department for Environment Food and Rural Affairs, 2013) and inflexible. Second, a GIS approach facilitates not only a more robust approach to disturbance modelling, but also provides a platform upon which modelling results can be shared. This may stimulate a greater adoption of ecological concerns into development planning and ease data integration between other development stakeholders. Third, representing spatio-temporal dynamics of wildlife vulnerability (as opposed to the static, discrete parcels proposed under some current approaches) would mark a move toward data that more accurately reflect the real world processes it represents. Calls for such data are not novel to the ecology industry but have been a criticism of GIS-based approaches to spatial modelling in a wide variety of applications for some time (Goodchild et al., 2007).

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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.

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Figure 1

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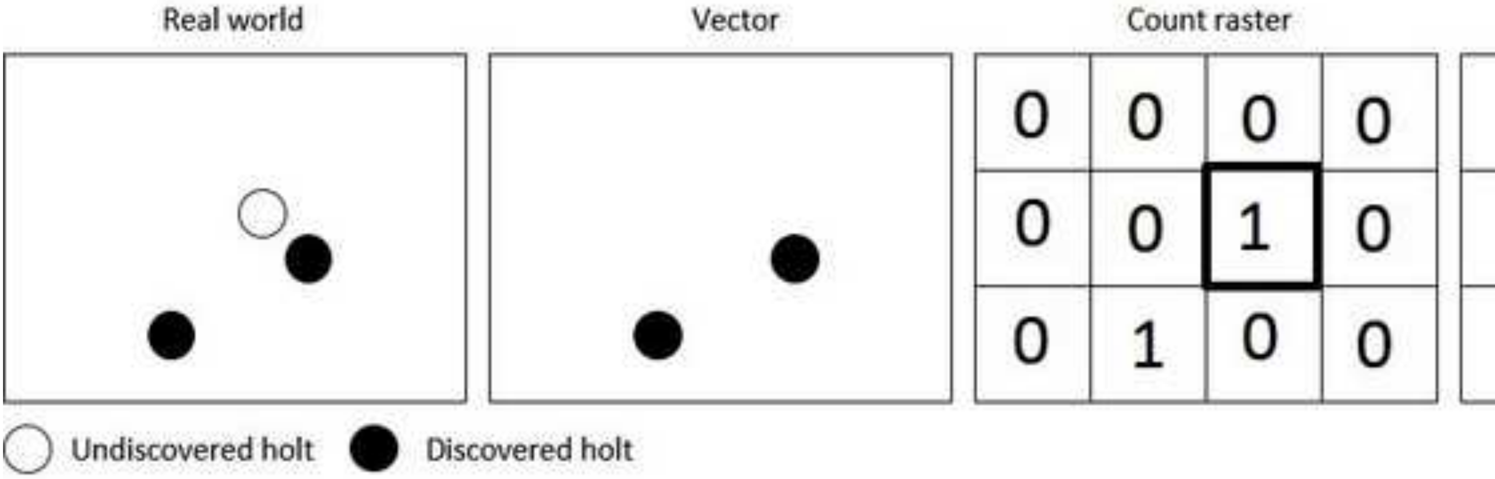

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Figure 2

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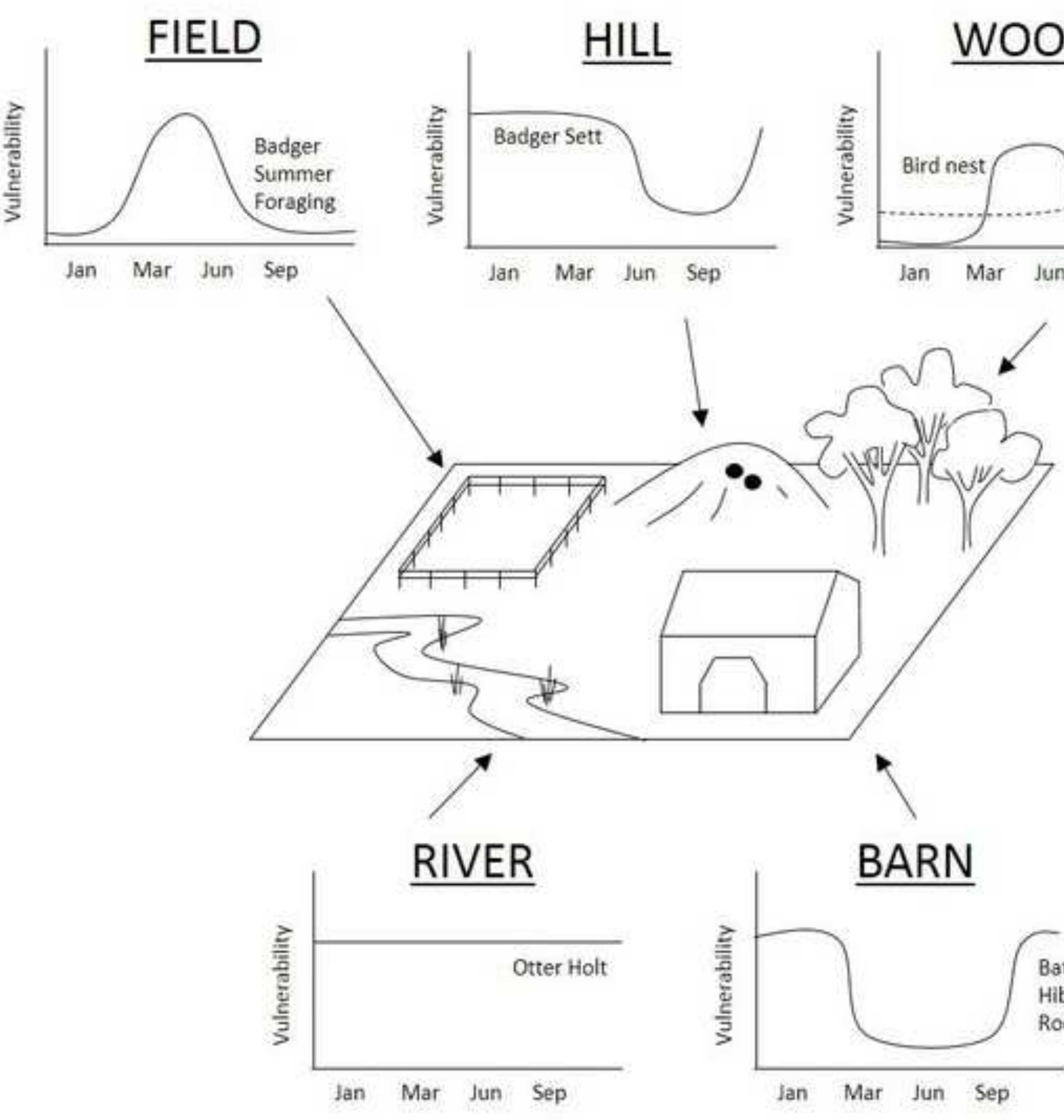

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Figure 3

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