Golden eagles and wind farms

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Summary

1. Wind farms may impact on the Scottish golden eagle population via a combination of displacement/disturbance and additional mortality which may apply to both range holding birds and more free-ranging non-breeding adults and sub-adults. Displacement effects can be equivalent to habitat loss.

Displacement/disturbance around wind farms

2. If all current schemes (approved, awaiting a decision and in scoping) are built, 31 golden eagle ranges would be within 6 km of a wind farm. Eight of these are within 3 km of a planned wind farm and may be particularly susceptible to displacement or collision mortality.
3. To date, there has been no recorded golden eagle fatality at a Scottish wind farm. However, there is growing evidence of displacement of birds from wind farm sites.
4. Monitoring results from the Beinn an Tuirc wind farm reveal evidence for golden eagle displacement. However, this interpretation is confounded by habitat management which may have, by itself, resulted in a shift in range use. Few flights were reported over or through the Beinn an Tuirc wind farm after it became operational, and no foraging flights were recorded within the wind farm's footprint.
5. The Ben Aketil and Edinbane wind farms on Skye have been monitored regularly since January 2007. This includes construction and operational periods. Results from more than 40 months of monitoring support the displacement hypothesis. Construction has only recently finished at Edinbane so it is not yet possible to separate out effects related to construction from those related the wind farm's operation. However, there have been a small number of foraging flights within the wind farm's footprint.
6. Evidence from Beinn Ghlas is largely absent from the public domain, but the available evidence points to the wind farm preventing re-occupation of the golden eagle range, i.e. displacement is operating there as well.
7. Evidence from the impacts of conifer afforestation on Scottish golden eagle productivity and range abandonment suggests that the effects of displacement by wind farms are likely to be site-specific.

The PAT (Predicting *Aquila* Territories) model

8. The PAT model is summarised. An example is used to explain how predictions from a PAT model can be used to assess the extent of habitat loss from a wind farm. Possible improvements to the PAT model are discussed, some of which are scientifically challenging.
9. Some qualitative tests of PAT range-use predictions were illustrated using data from Skye and Lewis. Despite numerous caveats about the nature of the empirical flight data, the fit between the recorded flights and the PAT predictions was reasonably good.

10. The most detailed eagle flight data set was used to illustrate how decisions about the implementation of the PAT model can influence the quality of the fit of the model. In particular, the range centre and exclusion zones can alter the predictions significantly[1].

Population models

11. A simple population model is described, with the lack of real life history data highlighted.
12. The model is used to illustrate how additional eagle mortality can be incorporated to measure the population level impacts of collision mortality.
13. The role of floaters, and the impact of additional floater mortality, is explored. In particular, the effect of adjusting the population cap (carrying capacity) is explored and shown not to have a major impact on conclusions about mortality effects.

Collision Risk Models (CRM)

14. Displacement of eagles by wind farms is partial, so any impact assessment needs to address both displacement and collision mortality. The imprecision of the current Collision Risk Model (CRM) is emphasised, particularly with regard to collision avoidance rates; problems related to an assumption of random flights are explored.

Habitat management/prey enhancements

15. Based on limited evidence available so far, prey enhancement via habitat management (mitigation) is unlikely to show rapid benefits for golden eagles.
16. Supplementary feeding, as an aid or alternative to habitat management, is currently largely untested for golden eagles in Scotland. Supplementary feeding has been used at Beinn an Tuirc and a trial is ongoing in Morven, but evidence from other attempts, combined with a knowledge of eagle ecology, suggest uncertainties over the outcome.

Further work

17. There is an urgent need for co-operative work across wind farm developments to: a) measure in detail post-construction impacts on golden eagle mortality and displacement/disturbance (ideally comparing pre- and post development locations); b) assess the impacts of mitigation measures on golden eagle range use and productivity; and c) determine cumulative impacts of developments on the favourable conservation status of golden eagles.

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[1] The words ‘significantly’ and ‘significant’ where relating to effects or impacts are used throughout the report in a general ecological sense rather than technical definitions under EIA and Habitats Regulations.

1. Possible effects of wind farms on golden eagles

Many wind farms across upland Scotland have the potential to adversely affect either territorial pairs or free-ranging sub-adults and floaters (non-territorial adults). The effects of wind farms can be summarised as displacement/disturbance and/or direct cause of death. This review examines the evidence for both of these effects and the methods used to detect and assess them plus some discussion of possible mitigation methods.

Displacement

Displacement prevents or reduces the probability of using certain habitats. The amount of lost habitat depends on the likely displacement distance and any 'barrier' effects. If displacement happens it is possible to ask what are the consequences and are displacement effects the same for sub-adults, floaters and range holding birds? At the extreme it might be expected that a range will be abandoned, either immediately following construction or after the death of the range-holding birds. If displacement happens, is it possible to mitigate against the loss by techniques such as prey enhancement?

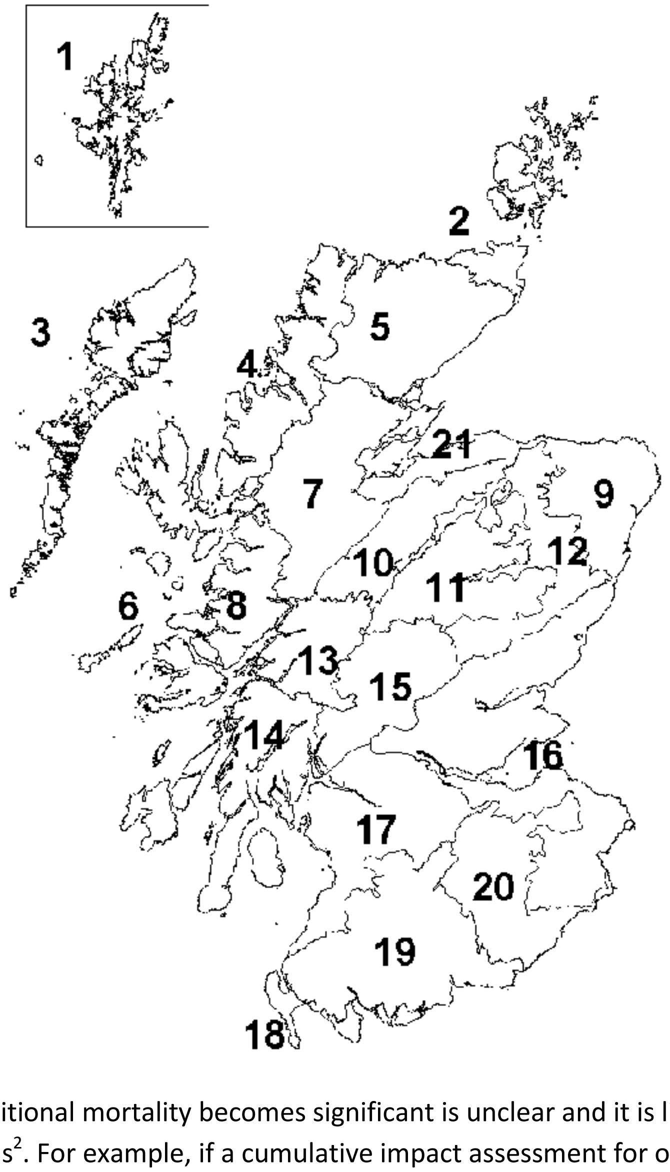
The population-level effects of range abandonment through displacement depend on productivity of the range and its neighbours. The effects can be counter-intuitive and are best illustrated by two extreme examples. In both examples there are 10 pairs, seven of which are unproductive while the remaining three pairs fledge 1 young each per year. The fledging rate (FR) is 3/10 or 0.3 per year. Below this level of productivity golden eagle populations are not viable at normal levels of survival. In the first scenario three of the seven unproductive ranges are abandoned. In the second scenario the three productive ranges are abandoned.

In scenario 1 there are seven pairs after the three unproductive ranges have been abandoned. However, because the three productive ranges remain occupied the FR increases to 0.43 (3/7) and the population is in a more favourable status. This has wider effects because less new recruits are needed each year and the population of seven birds is almost certainly a source population exporting young birds to other populations. However, in the second scenario there are still seven ranges but the three productive ranges are abandoned. This means that the seven pairs now fledge no young and the FR is 0. As a consequence the population is in a less favourable condition; indeed it is now a significant sink population which draws in birds from elsewhere to join an unproductive population. This extreme example emphasises the need for good data on which to make judgements.

Mortality

Mortality could arise directly through collisions with a wind farm's infrastructure, particularly the rotating turbine blades and any above ground power cables. It is also possible that displacement could reduce the life span of a bird if resources, such as prey, are reduced. There are two important questions. What is the magnitude of any additional mortality and how will the additional mortality affect the viability of the population? The first question is assessed using a collision model which is susceptible to sampling bias and errors during the collection of flight data and, more significantly, the magnitude of any avoidance rate used in the calculations. The second question can be answered simply if the additional mortality is trivial, for example one extra death every 100 years but as the additional mortality increases it is necessary to use some type of population modelling. It is also important that any additional mortality is placed in a wider context by carrying out some type of cumulative assessment. The spatial and temporal extent of a cumulative assessment is less clear and there is evidence for inconsistencies across assessments, even within the same Scottish region.

Figure. 1. Biogeographic zones of Scotland, termed Natural Heritage Zones (NHZs), developed by Scottish Natural Heritage (SNH, 1998, 2000). 1 = Shetland, 2 = North Caithness and Orkney, 3 = Western Isles, 4 = North West Seaboard, 5 = The Peatlands of Caithness and Sutherland, 6 = Western Seaboard, 7 = Northern Highlands, 8 = Western Highlands, 9 = North East Coastal Plain, 10 = Central Highlands, 11 = Cairngorms Massif, 12 = North East Glens, 13 = Lochaber, 14 = Argyll West and Islands, 15 = Breadalbane and East Argyll, 16 = Eastern Lowlands, 17 = West Central Belt, 18 = Wigtown Machairs and Outer Solway, 19 = Western Southern Uplands and Inner Solway, 20 = Border Hills, 21 = Moray Firth.



The point at which additional mortality becomes significant is unclear and it is likely to vary across species and populations[2]. For example, if a cumulative impact assessment for one NHZ (Natural Heritage Zone, see Fig. 1[3]) predicts one additional eagle death over 300 years (1 between 2010 and

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[3] SNH has identified 21 NHZs that reflect the variation in biological and landscape qualities across Scotland (SNH, 1998, 2000)

2310) then the impact is obviously insignificant. However, one additional death every year is clearly significant. So, the threshold for significance is somewhere between one per year and one every 300 years. SNH (2009) have stated that a “biologically sound” approach is needed. This suggests that there is no reason to believe that the threshold will be the same for all species or, indeed, all populations of the same species. There is nothing unusual about the need to identify a threshold; it applies to every designed experiment in which the statistical power is largely dependent on a desired effect size. If the threshold (effect size) is too small then biologically insignificant losses will be judged significant and a wind farm development will be stopped unnecessarily. Similarly, if the threshold is set to be too large then wind farms may be constructed that result in biologically significant population declines. There is, therefore, a need to balance biological and financial criteria which include: socio-economic factors; wider biodiversity issues (e.g. climate change, land management); the conservation status of a species and species trends (is it already “committed to extinction”?).

Detecting effects

The general extent of any real displacement or mortality will only become apparent if constructed wind farms are monitored and the data made generally available. In this way real data could be used to inform the cumulative impact assessments. During the writing of this review and related work (e.g. Fielding and Haworth 2010a), it has become clear that there is a dearth of such wind farm monitoring information, at least in the public domain.

Population modelling

If additional wind farm mortality or range loss is significant it becomes important to undertake some population modelling. A population model uses the values for parameters such as reproductive output, survival rates and immigration/emigration rates to model the trajectory of a population’s size. A population growth rate of 1 is a stable population with fixed size over time. If the growth rate is above or below 1 the population will either grow or retract at a rate determined by the magnitude of the growth rate.

2. What is the possible scale of impact?

The SNH wind farm database (April 2010 with some updating) was combined with national census data to assess the possible scale of interaction between golden eagles and wind farms. An earlier analysis by Fielding *et al* (2006) concluded that there were few grounds for concern at that time.

The minimum distances from golden eagle range centres to all wind farm proposals in the database were obtained using a GIS and all of those within 6 km of a wind farm were selected for further analysis (Table 1). Six kilometres was used because this is the distance threshold in the PAT model (McLeod *et al* (2007), Fielding *et al* (2003)). The spatial relationships between wind farm proposals and non-breeding golden eagles are covered in section 3.

Table 1. Relationship between wind farm planning status and golden eagle range status. Figures in parentheses are the known or proposed number of turbines. The information in this table is subject to change as schemes pass through the development and planning process and new ones are proposed.

Wind farm status	Wind farm	Range status	Turbine distance (km)
Operational	Ben Aketil (12)	no evidence of pair	1.04
	Braes of Doune (50)	range vacant > 20 years	1.16
	Edinbane (18)	no evidence of pair	2.04
	Tom nan Clach (17)	range vacant > 20 years	2.50
	Millenium (6)	range vacant > 20 years	2.59 and 3.26
	Edinbane (18)	no evidence of pair	2.85
	Arnish Moor (3)	active	3.01
	Novar (34 plus ext of 16 approved)	range vacant > 20 years	3.22
	Beinn Ghlas (14)	range vacant > 10 years	3.27
	Kilbraur (19 ext of 8 planned)	range vacant > 20 years	3.36
	Cruach Mhor (37)	range vacant > 20 years	3.38
	Beinn an Tuirc (46)	active	3.39
	Paul's Hill (28)	range vacant >10 years	4.35
	Beinn Tharsuinn (20 plus ext of 2 planned)	range vacant > 20 years	4.54
Approved	Monan (abandoned)	active	0.58
	Mid Hill (25 plus ext planned)	range vacant > 20 years	0.68
	Muaitheabhal (33)	active, range losses predicted	1.01 and 1.18
	A'Cruach (24)	active	1.79
	Griffin (68)	range vacant > 10 years	1.80
	Carraig Gheal (20)	active	2.31
	Loch Luichart (17)	active	3.09
	Pentland Rd (6)	active	3.16
	Gordonbush (35)	active	3.30
	Corriegarth (20)	range vacant > 20 years	3.36
	An Suidhe (24)	active	3.58

	Muaitheabhal (33)	active	3.80 and 4.80
	Achany (23)	active	4.10
	Carcant (3)	active	4.44
	Clachan Flats (9)	active	4.78
Application	Calliachar (14)	range vacant > 20 years	1.13
	Dunbeath (Braemore) (23)	active	1.75
	Shira (22)	active	2.13
	Raera (16)	active	2.26
	Raera	range vacant > 20 years	2.88
	Srathy North (35 & 77)	active	3.02
	Cour (13)	range vacant > 20 years	3.08
	Lairg Estate (3)	range vacant > 20 years	3.62
	Shira (22)	active	3.64
	Dunmaglass Estate (50)	active	4.19
	Glenkirk (31)	range vacant > 20 years	4.19
	Druim Fada (4)	active	4.38
	Dunbeath (Braemore)	active	4.72
	Pairc (26)	range vacant > 20 years	4.76
	Dunmaglass Estate	range vacant > 20 years	4.82 and 4.84
	Corriegarth	range vacant >10 years	4.96
	Cambusmore (33)	active	4.96
	Pairc	active	5.12
	Earlshaugh (39)	active	5.15
Scoping	Durness (2)	range vacant > 20 years	1.62
	Allt Duine (31)	Active	3.00
	North Tolsta (3)	active	3.41
	Ballygroggan Mark II (12)	active	3.90
	Loch Arkaig	active	3.91

Currently, Arnish Moor on Lewis and Beinn an Tuirc in Argyll are the only operational wind farms with an active golden eagle range within six kilometres. Arnish Moor is thought to have been reoccupied in recent years and bred successfully in 2009 (Reid *pers comm.*). The Beinn an Tuirc scheme is discussed in more detail in the next section. The remaining 12 ranges do not appear to be active, seven have no evidence of occupation for more than 20 years, two have no evidence for more than 10 years (including Beinn Ghlas – see next section for more details) and three named ranges on Skye, in the vicinity of the Ben Aketil and Edinbane wind farms, have no apparent history of occupancy.

The situation with approved wind farms looks potentially more problematic for golden eagles. Twelve of the fourteen approved schemes have one or more active range centres within 6 km of the wind farm, although nine of these are more than three kilometres from a wind farm. The remaining three ranges appear to have been inactive for a considerable number of years. The Monan wind farm scheme has now been abandoned for reasons unconnected with golden eagles. Two ranges on

Lewis are predicted to be abandoned as a consequence of the construction of the Muaitheabhal wind farm. It is essential that post construction these windfarms is subject to considerable monitoring with results being made available to the general public. This information will then inform future assessments.

There are 14 schemes currently awaiting a decision and nine of these have one or more active golden eagle range within six kilometres, although eight ranges are more than three kilometres and one of the others, Raera, is currently a commercial conifer forest that is rarely used by golden eagles (Haworth *pers comm.*). The remaining golden eagle ranges appear to have been inactive for a considerable number of years.

Finally, there are five schemes in scoping, four of which have an active golden eagle range within six kilometres. All four of these ranges are at least three kilometres from the wind farm.

No additional analyses have been undertaken with the data in Table 1 because we currently do not have access to all of the required information, particularly site layouts, turbine sizes and Environmental Statements. The wind farms in Table 1 cover a wide range of scenarios including single, relatively small turbines to multi-turbine schemes with modern large turbines. It is clear that the impact of a wind farm will not be independent of the size and number of turbines. As more wind farms are constructed, and flight data are collected, it should be possible to factor these variables into a robust meta-analysis of the effects of wind farms on golden eagles and other species.

In summary, there are currently very few golden eagles affected by operational wind farms and, as shown later, there is no evidence that this very small number of wind farms is having any significant negative effects on golden eagles. However, if all of the approved wind farms are constructed this may change, particularly for range holding birds via a combination of displacement leading to a small number of lost ranges and some additional mortality that, at predicted levels, should not have a detrimental impact on the Scottish population. If all current schemes (approved, awaiting a decision and in scoping) are built, 31 golden eagle ranges would be within 6 km of a wind farm. Eight of these 30 ranges are within 3 km of a planned wind farm and may be particularly susceptible to displacement or collision mortality.

3. Displacement of golden eagles following wind farm construction: has this occurred?

Given the information provided in Table 1 there are few examples from which to draw conclusions. There are only three wind farms that can provide any evidence for the scale of effects.

Beinn an Tuirc

Unusually, there are two peer-reviewed publications dealing with the potential conflict between the wind farm and resident golden eagles (Madders and Walker (2002) and Walker *et al* (2005). Beinn an Tuirc wind farm has 46 turbines erected during 2001 within an occupied eagle territory in Argyll. However, the range had a relatively poor productivity history with only one fledged young between 1988 and the start of construction. This is thought to be largely a consequence of the extensive conifer plantations that are widespread in this part of upland Argyll.

Pre-construction survey work indicated that the eagles preferentially used the higher, unplanted parts of their range to the north and east of the proposed wind farm. The only part of the conifer plantations used by the eagles were either recently planted (<8 years old) or forest rides. One explanation for this pattern of use is the distribution of red grouse, their main live prey item. Red grouse were closely associated with the amount of heather cover which was greater at higher altitudes because there was less sheep grazing. The survey work also suggested that the eagles only used the north eastern part of the proposed wind farm, so some turbines from this area were moved to the little used south western part of the proposed wind farm.

Madders and Walker (2002) describe a management strategy that aimed to improve the range quality by enhancing the amount of live prey, which were mainly red grouse. Prey surveys suggested that the proposed wind farm contained about 25 grouse territories. The management plan, which included forest clearance and management of existing heather, aimed to increase the abundance of potential prey species such as grouse. In addition, it was hoped that the creation of new areas of foraging habitat, in felled forest, away from the wind farm would reduce the risk of eagle collisions with the turbines by altering the spatial pattern of their range use.

Walker *et al* (2005) use several lines of evidence to support a hypothesis of avoidance of the operational wind farm, i.e. displacement. First, only one of 811 recorded eagle flight paths was recorded at collision altitude within the wind farm and this passed between the two discrete turbine clusters. They suggest that this single flight was in response to the presence of an intruding eagle. Similarly, two of the three eagle flights over the wind farm were also associated with intruding eagles. They recorded no flights within either of the turbine clusters. They exclude accessibility as an explanation for this exclusion since the turbines are spaced at wider distances that the widths of the forest rides used by hunting eagles in the pre-construction surveys. Secondly, the presence of prey within the wind farm, particularly relatively high grouse availability in later summer and regular use by corvids, precludes lack of prey as explanation for the lack of use. Sheep carrion is an important part of the eagles' diet and most sheep carcasses are removed from the wind farm area to reduce the chance that eagles will enter the wind farm. However, not all sheep carcasses were removed but there was no evidence that eagles exploited carrion within the wind farm. Although no data are provided, Walker *et al* (2005) do not think that rotor noise is responsible for the avoidance because there was no indication that the wind farm was used more when turbines where inactive. Finally, spatial analyses of post-construction activity provided some support for greater use of the area where trees had been felled to improve foraging potential. In the 2005 paper Walker *et al* suggest that grouse numbers have increased since the felling, and use of the area by eagles should increase further as prey numbers recover further. However, in 2009 grouse density was lower in the main grouse management area than in 1999, possibly because of problems with heather beetle infestations and excessive grazing (Robson 2009). It is difficult to determine the contribution that the management plan made to the successful breeding in 2008, when the site fledged twins. This is because any interpretation is confounded by the use of supplementary feeding. In addition to the failure to increase grouse numbers there is now less open habitat available within the territory than there was before windfarm construction (Robson 2009), mainly because of continued conifer planting. In 2009, supplementary feeding continued but with no breeding success. This is discussed further in section 6.

The extensive habitat management plan makes it difficult to separate out avoidance of the wind farm from attraction to the newly open habitat in the cleared areas. Walker *et al* (2005) argue that the continued avoidance of the open habitat within the wind farm suggests that the existence of relatively open areas *per se* is not sufficient motivation to attract eagles for foraging. Although their analyses of spatial use indicated an overall shift to the newly felled areas in the north east, they did not find a significant shift in the location of the core areas, possibly because of the combination of previously unplanted areas and nest sites. They raise the possibility that, if the location of the core area is inflexible for most eagles, identifying the core area and protecting it may be particularly important for all wind farm developments. While the PAT model (Section 4) can be used to identify core areas it is always preferable to use empirical data, but this must come from observing a very large part of the range and not just the area associated with a proposed wind farm.

Edinbane

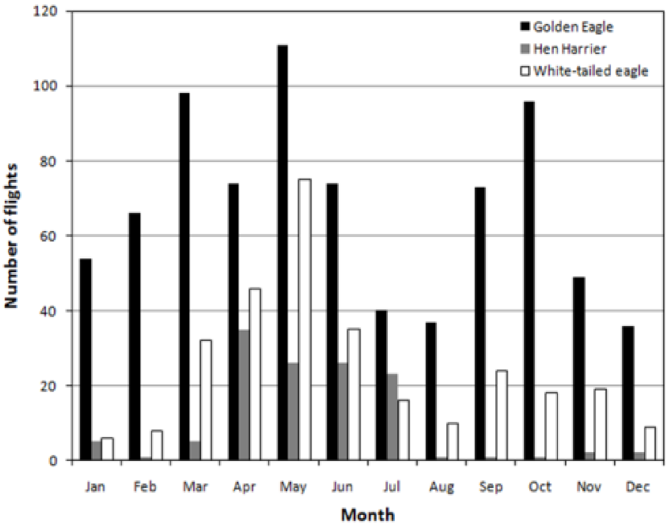
Unlike Beinn an Tuirc and Beinn Ghlas, the Edinbane (18 turbines) and Ben Aketil (12 turbines) wind farms do not impact on golden eagle ranges but there is evidence of consistent historic and continuing use by non-breeding birds. These wind farms have been subjected to considerable monitoring over a period which includes construction and operation (Fielding and Haworth 2010b).

The first two Ben Aketil turbines were erected by 19th September 2007, rising to five by the 2nd October and nine by the 20th October. On the 3rd November 2007 all ten were complete with eight functioning. At Edinbane all turbine bases had been completed by 20th June 2009, with the first turbine complete by July 15th. Construction was slower than Ben Aketil and six were complete, with four more under construction, by the 6th October. By the 28th November 13 were complete and construction had started on the five bases for the later extension.

Observations were gathered from four Vantage Points (VPs) between January 2007 and December 2009. Three main VPs were used throughout and a fourth was added to specifically target a potential white-tailed eagle roost. Observations from the three main VPs comprised more than 100 hours of survey effort per VP per annum. The amount of bird activity varied between months (Figure 2). Both golden and white-tailed eagles showed obvious peaks in activity in spring and autumn with considerably less activity in winter and summer. These annual trends have been relatively consistent between the years. There are obvious implications of such patterns for survey protocols.

Spatial and temporal patterns of habitat usage were investigated using a grid of four hectare cells. The total lengths of digitised flight lines that intersected each cell were calculated separately for each month. Similarly, the total number of surveying hours was calculated for each grid cell by overlapping the viewsheds on to the grid. Usage is then expressed as kilometres of flight per 100 hours of observation (Figures 3a,b,c). These data provide evidence of a clear shift in spatial use by golden eagles. Usage of the now operational Ben Aketil wind farm has declined to less than a third of the 2007 levels and there has also been a marked, but later, decline within the Edinbane wind farm area. More importantly, the changes in use by golden eagles at different stages of construction and operation show some consistency across the two wind farms

Figure 2. Number of flights per month (2007-2009 combined) at the Ben Aketil and Edinbane wind farms for three species. Survey effort was the same each month.

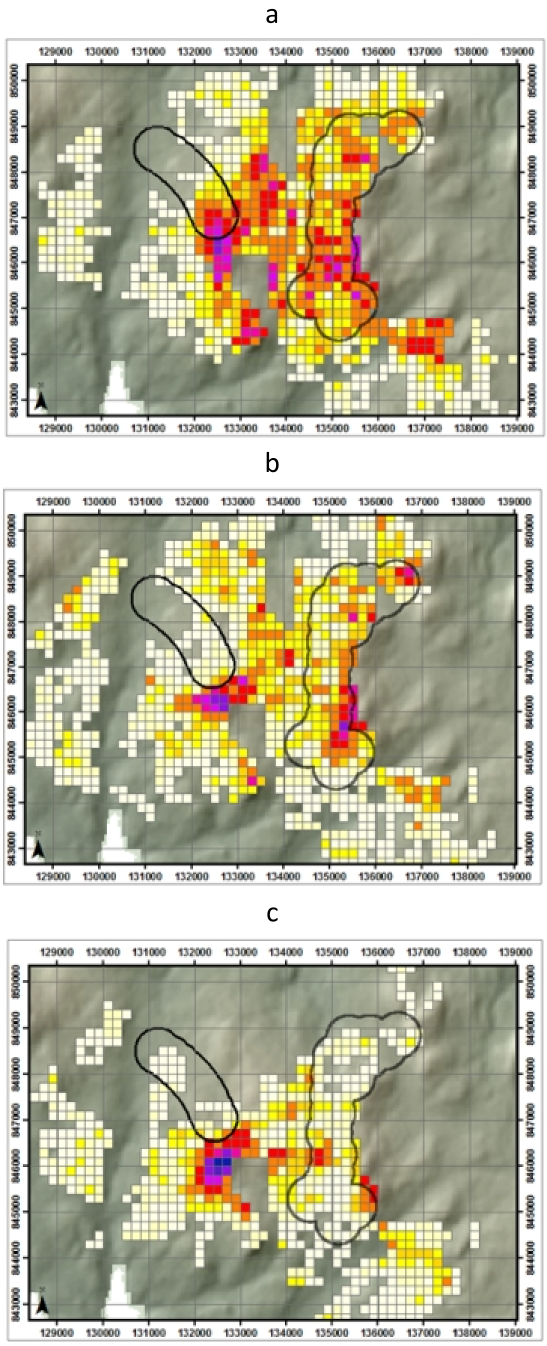


Although it is impossible to assign definitive explanations for these shifts in use it is possible that golden eagles are now avoiding the operational wind farm. However, it is important to note that there was a considerable amount of construction activity associated with the development of the Edinbane wind farm throughout 2009. This may have been a transient influence on the behaviour of golden eagles. Further monitoring over the next two to five years will clarify if the apparent avoidance was a result of construction or the presence of turbines.

Although no results are presented here data from 2010 (January to August) is consistent with those collected between 2007 and 2009.

In addition to mapping the movements of eagles around the wind farm the productivity of all golden eagle ranges on Skye have been monitored. In 2007 29 ranges were occupied by golden eagles but only nine young eagles successfully fledged. In 2008 29 ranges were occupied by golden eagles and 17 young eagles fledged. In 2009 29 ranges were occupied by golden eagles and 13 young eagles fledged. In 2010 15 young were fledged from 29 ranges. There is no evidence from these data that the presence of wind farms on Skye is impacting negatively on the local breeding golden eagle population.

Figure 3 Golden eagle usage (km flight / 100 hr observation): a 2007; b 2008; c 2009. Wind farm footprints are shown as black lines.



Others

The other main operational wind farm with golden eagle interest is Beinn Ghlas (14 turbines) in Argyll. Unfortunately it is very difficult to obtain monitoring reports for this scheme. Moss and Walker (2008) suggest that the eagles avoid the windfarm and that the wind farm was deserted because there was less food. The management aimed at increasing the number of grouse by reducing the sheep grazing failed probably because of a combination of heather beetle infestations, increased deer grazing and additional loss of young grouse to corvids. Gregory (2009) provided some useful background data. It would be helpful if more details of the monitoring undertaken by, or for, the operators were made public. This remainder of this summary is based entirely on material in Gregory's report.

Beinn Ghlas is an historic eagle territory that was reasonably productive between 1970 and 1991, when the last known chick was fledged. Since then, and prior to the start of wind farm construction in 1998, the pair seems to have had a few problems after eggs were laid in 1992. The turbines became operational by the autumn of 1999. During the wind farm's operation there has only been one suggestion of a (failed) breeding attempt. Gregory (2009) suggests that this failed breeding attempt occurred when both adults had gone missing from a neighbouring range and new birds were exploring the vacant territories. It is now thought that subsequent eagles visiting the Beinn Ghlas nest site have been from a neighbouring range and that the Beinn Ghlas range is now vacant even though five neighbouring territories continue to hold resident breeding pairs. Gregory (2009) thinks that there is evidence for some detrimental effects of wind farms on golden eagles, not least on their ability to fledge young. Given the detailed range history provided by Gregory (2009), it is impossible to deduce that it is only the wind farm which is preventing range re-occupation. However, it remains a possibility that displacement by the wind farm is a factor in maintaining this range vacancy. It may also be relevant that there is little scope for range adjustment because of constraints created by closed canopy forests and neighbouring ranges. This problem is considered further in section 5.

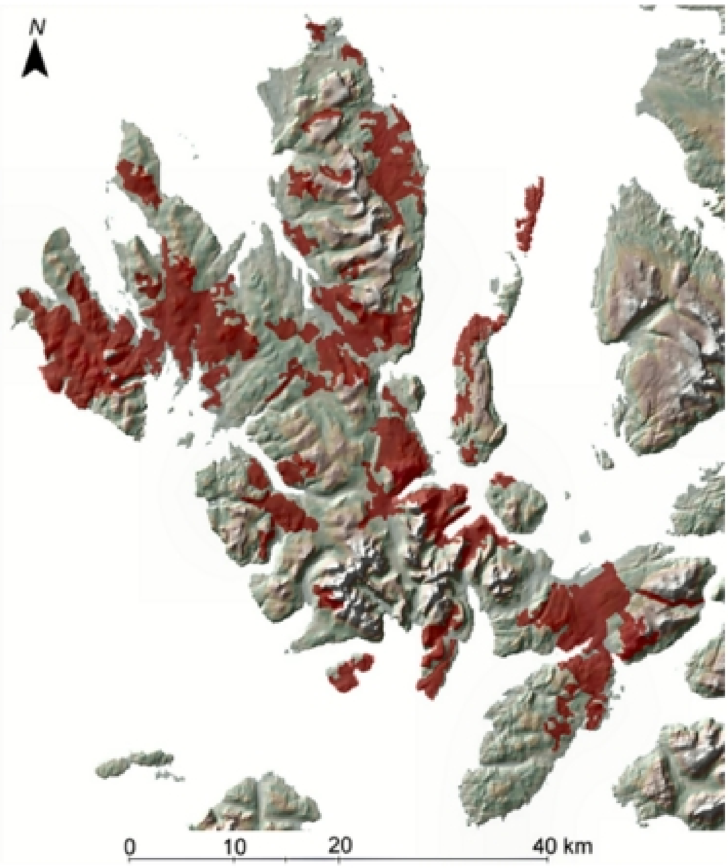
There has been an attempted mitigation in form of sheep removal from a fenced area. It was hoped that this would result in more live prey such as grouse and hares. While there is no evidence that this mitigation has been successful in helping to get the range reoccupied it is unknown if it has succeeded in increasing the amount of live prey.

Loss of non-breeding habitat

Generally, golden eagles do not breed until they are least five years old. Young birds disperse and become nomadic to a lesser or greater extent (Soutullo *et al* 2006ab). During this nomadic phase they appear to avoid occupied ranges and certain habitats such as conifer plantations, lochs and suburban areas. Consequently, any remaining habitat is potentially habitat for non-breeding golden eagles. If there is a significant regional or national loss of such habitat it could lead to a reduction in survival rates and a decline in the number of floaters. Such declines would eventually lead to a population decline as new vacant ranges remain unfilled. This means that wind farms, even away from occupied ranges, may impact on the Scottish golden eagle population. Since persecution appears to be significant in some regions (Whitfield *et al* 2003, 2006ab) the loss of non-breeding habitat, in regions with little persecution, could increase the threat to the national population by encouraging emigration to regions with significant persecution. Fielding *et al* (2006) described a method to assess the amount of sub-adult (non-breeding) habitat. Briefly, this habitat was in upland regions >6 km from occupied golden eagle nests but excluding conifer forests and lochs. Upland habitat was defined as a series of habitat types (Fielding *et al* 2006). In 2006 they identified 20,475 km^2 of sub-adult habitat in Scotland. Figure 4 shows the distribution of non-breeding habitat on Skye.

Using SNH wind farm data for 2006 they found that 378 km^2 (1.85%) was overlapped by 232 of the 527 potential wind farm schemes. This was not considered to be a significant problem at that scale and Fielding *et al* (2006) concluded that, nationally, wind farm developments did not give grounds for serious concern because the amount of non-breeding eagle habitat lost to wind farms, by displacement, was minimal, totalling about 2% if all schemes were installed. There was some evidence that birds would not be displaced from regions free from persecution but particular care was needed in planning wind farm schemes and in assessing cumulative impacts was needed in some regions (NHZs).

Figure 4 Areas of potential non-breeding habitat (red shading) idenfied on the Island of Sky. (Contains Ordnance Survey data © Crown copyright and database)

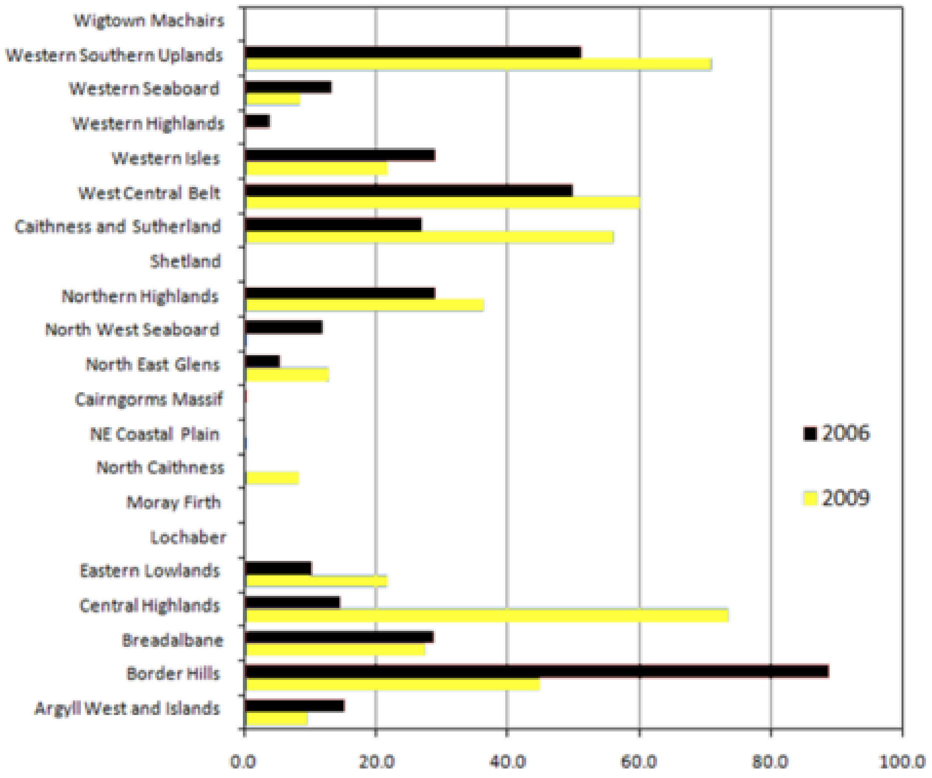


Fielding and Haworth (2009a) recalculated the predicted losses using the September 2009 SNH wind farm database (Figure 5). Overall, there was little change but there were increases and decreases across the NHZs.

It is possible to take these non-breeding habitat loss calculations a little further and attempt to quantify the losses in prey terms as well as habitat area. For example, the Argyll West and Islands NHZ has a mean fledging rate 0.46 young/pair/year from 44 pairs. Therefore the mean is 20.2 young fledged per year. If it is assumed that the annual survival for subadults is 80% this leads to an estimate of 16.2 1st yr, 13.0 2nd year, 10.4 3rd year and 8.3 4th year birds, or a sub-adult population of 47.8 birds. This estimate assumes no immigration or emigration of sub-adults from the NHZ. Fielding *et al* (2006) estimated that there was 1007 km^2 of sub-adult habitat in the NHZ. This equates to an estimated 21.1 km^2/bird. Fielding *et al* (2009) quoted an estimate from Brown and Watson (1964) which suggested that a pair of eagles needed 174 kg of live prey and carrion per year. Brown

(1969) expanded on this by suggesting that another 54 kg was required if an average of 0.8 young per year were reared plus an additional 43 kg for sub-adult birds using the territory intermittently. Being conservative, assume a young bird needs 100 kg of prey per year. If the NHZ has 1007 km^2 of sub-adult habitat this means there must be 4.75 kg of food per km^2 per year or 0.396 kg per km^2 per month. Now, assume a 5% loss of sub-adult habitat in the NHZ to wind farms, the remaining area becomes 957 km^2 and, if the number of sub-adults is unchanged, the area per bird reduces from 21.1 to 20 km^2 and the required prey per km^2 becomes 5.00 kg/year - or a total of 0.416 kg per month. Therefore, there is an increased requirement for 21 g of prey per month per km^2. It seems unlikely that this would be a problem and, even it was, it would be very easy to compensate for by a tiny amount of supplementary feeding. If the habitat loss is 2% the additional prey requirement is 8 g/month/km^2.

Figure 5 Areas of non-breeding golden eagle habitat overlapped by wind farm footprints (from the SNH database) in NHZs in 2006 and 2009

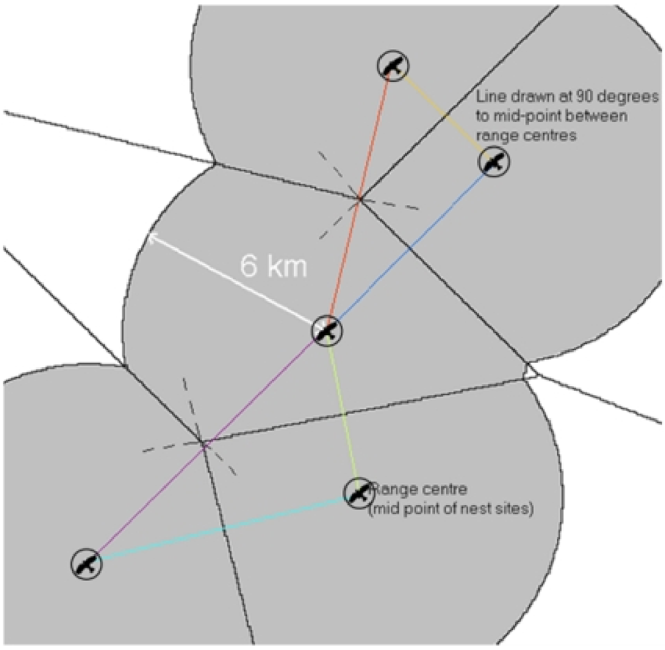


4. Modelling and golden eagles

Predicting range use: the PAT model

The PAT (Predicting *Aquila* Territories) model is based on the RIN (Forestry Commission Research Information Note, McGrady *et al* (1997)). If there are no neighbours a 6 km cut-off from the range centre is used as the notional range boundary. Within the notional territory a 'core area', within which 50% of eagle activity occurs, is further delimited by a circle of 2-3 km radius. If there are neighbours between-range boundaries are modelled by Thiessen polygons (Figure 6). The area within a Thiessen polygon is closer to the point on which the polygon is centred than it is to any other point in the dataset.

Figure 6. Territory boundary allocations using Thiessen polygons



Enhancing the RIN

The PAT model (McLeod *et al* (2007) and Fielding *et al* (2003)) begins with the RIN's Thiessen polygons but incorporates a combination of excluded areas and categorised variables. For example, golden eagles are sensitive to human disturbance and tend to avoid areas such as settlements and roads. But, in the absence of specific information on disturbance distances, buffer zones around human settlements, within which eagles are assumed not to range, are created as follows: single building 250m, cluster of buildings 400m, village 600m, and town 800m. Water bodies and the sea are treated as exclusion areas because they provide few air currents that golden eagles can exploit, and they provide few prey sources. However, it recognised that eagles do take wildfowl and seabirds (see Haworth *et al (*2009) for a full list of prey items in the Hebrides). The PAT model assumes that golden eagles avoid commercial forestry (or other closed canopy woodland) once trees reach 12 years as eagles are unable to access the open ground between trees.

Eagle behaviour

Breeding behaviour is included in the PAT via two mechanisms. First, because breeding birds are central place foragers and they should spend most time close to the nest (modelled as the range centre). Earlier modelling, and empirical data, confirmed that eagles tended to use central parts of their home range most frequently, and that this feature is incorporated into the PAT by weighting predicted range use towards the range centre. The density of empirical ranging observations underwent a sigmoidal decay with increasing distance away from the centre. In the model 500 m wide annuli are used for computational efficiency, with decreasing P(use) in each annulus. In addition, in a species such as the golden eagle, that is adapted for soaring flight, it is likely that terrain features will affect range use. In the cool Scottish climate thermal air currents are rare but wind deflected upwards off terrain features is probably an important aid for flight. Chalmers (1997) found a strong relationship between eagle activity and ridge features, which is consistent with the use of deflected wind currents on slopes.

Terrain recognition

In order to incorporate terrain features into the PAT an automated recognition method, using digital elevation data (DEM), was developed and applied to every pixel within the predicted range limits. In a raster DEM each pixel has an elevation value that can be compared with its neighbours, using a set of rules, to determine which type of terrain feature it represents.

If a pixel's neighbours all have lower elevations it is a candidate 'peak' pixel. Ridges and plateau/cliff edges, are recognized by comparing a source pixel's elevation with those of five neighbouring pixels along four opposing radial arms. For example, each source pixel is a ridge if, in any one of its opposing radials, the elevation angle relative to the source pixel is less than 168°. Figure 7a shows part of a DEM (50 m Digital Elevation Model). The actual altitudes of each pixel are shown in Figure 7b. Also shown are the comparisons between a source pixel and those in the four radial arms (Figure 7b).

Incorporating terrain features into a model.

Observed ranging locations were more frequently within 200m of a terrain feature (i.e. ridge or convex feature) than would be expected if they were evenly distributed within a 1200m buffer from terrain features (this buffer distance was chosen as almost every point in a home range was within 1200m of a terrain feature). Golden eagles were three times more likely to be seen close to ridge or convex feature pixels than expected if observations were randomly distributed. The distribution of observed ranging points relative to terrain features was used within the model for assigning predicted use relative to terrain features by weighting use by proximity to a ridge feature.

Figure 7 a partial digital elevation model; b digital elevations and search areas used to identify the terrain type for the central pixel

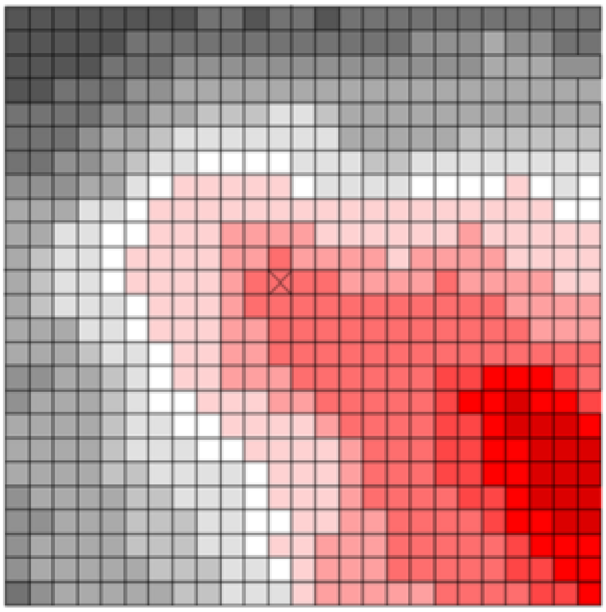
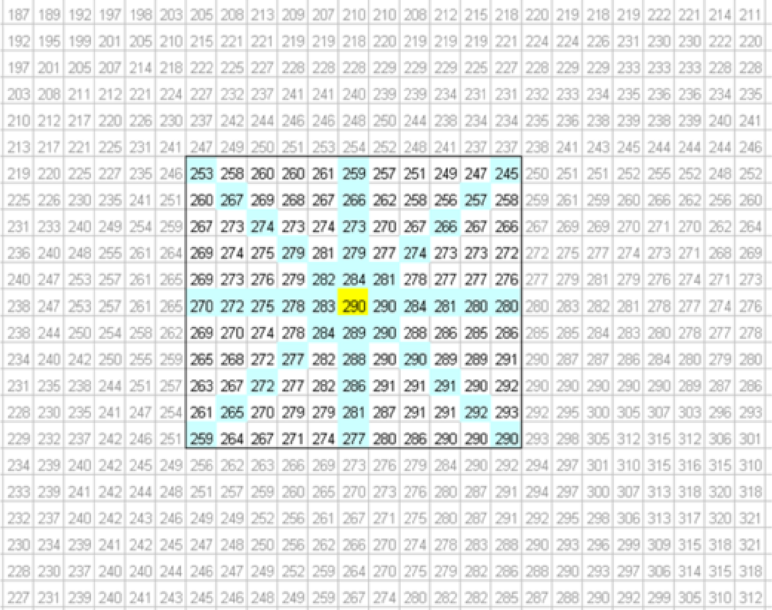


Figure 7b



187	189	192	197	198	203	205	208	213	209	207	210	210	208	212	215	218	220	219	218	219	222	221	214	211
192	195	199	201	205	210	215	221	221	219	219	218	220	219	219	219	221	224	224	226	231	230	230	222	220
197	201	205	207	214	218	222	225	227	228	228	228	229	229	229	225	227	228	229	229	233	233	233	228	228
203	208	211	212	221	224	227	232	237	241	241	240	239	239	234	231	231	232	233	234	235	236	236	234	235
210	212	217	220	226	230	237	242	244	246	246	248	250	244	238	234	234	235	236	238	239	238	239	240	241
213	217	221	225	231	241	247	249	250	251	253	254	252	248	241	237	237	238	241	243	245	244	244	244	246
219	220	225	227	235	246	253	258	260	260	261	259	257	251	249	247	245	250	251	251	252	255	252	248	252
225	226	230	235	241	251	260	267	269	268	267	266	262	258	256	257	258	259	261	259	260	266	262	256	260
231	233	240	249	254	259	267	273	274	273	274	273	270	267	266	267	266	267	269	269	270	271	270	262	264
236	240	248	255	261	264	269	274	275	279	281	279	277	274	273	273	272	272	275	277	274	273	271	268	269
240	247	253	257	261	265	269	273	276	279	282	284	281	278	277	277	276	277	279	281	279	276	274	271	273
238	247	253	257	261	265	270	272	275	278	283	290	290	284	281	280	280	280	283	282	281	278	277	274	276
238	244	250	254	258	262	269	270	274	278	284	289	290	288	286	285	286	285	285	284	283	280	278	277	278
234	240	242	250	255	259	265	268	272	277	282	288	290	290	289	289	291	290	287	287	286	284	280	279	280
231	235	238	244	251	257	263	267	272	277	282	286	291	291	291	290	292	290	290	290	290	290	289	287	286
228	230	235	241	247	254	261	265	270	279	279	281	287	291	291	292	293	292	295	300	305	307	303	296	293
229	232	237	242	246	251	259	264	267	271	274	277	280	286	290	290	290	293	298	305	312	315	312	306	301
234	239	240	242	245	249	256	262	263	266	269	273	276	279	284	290	292	294	297	301	310	315	316	315	310
233	239	241	242	244	248	251	257	259	260	265	270	273	276	280	287	291	294	297	300	307	313	318	320	318
232	237	240	242	243	246	249	249	252	256	261	267	271	275	280	287	291	292	295	298	306	313	317	320	321
230	234	239	241	242	245	247	248	250	256	262	266	270	274	278	283	288	290	293	296	299	309	315	318	321
228	230	237	240	240	244	246	247	249	252	259	264	270	275	279	282	286	288	290	293	297	306	314	315	318
227	231	239	240	241	243	245	246	248	249	259	267	274	280	282	282	285	287	288	290	292	299	305	310	312

Model Output

The output of the PAT model is a raster representation of predicted range use: each 50 x 50 m pixel having a predicted 'use value' constrained so that they sum to 100 for each range. Inevitably this means that each pixel has a very small use value. Pixels with higher use values are located near the range centre and around terrain features, and pixels with the lowest values are further away from the centre and terrain features (Figure 8). A 'use surface' can be generated if pixels, predicted as being used, are ordered in decreasing use value and then sequentially summed. Isolines can then be fitted to this surface, encompassing notional percentages of predicted ranging. For example, the 80% isoline encompasses the 80% highest use value pixels, and represents the geographic area required to encompass 80% of a pair's predicted ranging. Figure 8 is an example showing a 6 km range limit and exclusion zones (road with houses, a loch and a small woodland). Colour intensity is predicted probability of use.

Figure 8 Example PAT model showing exclusion zones and a 6 km threshold

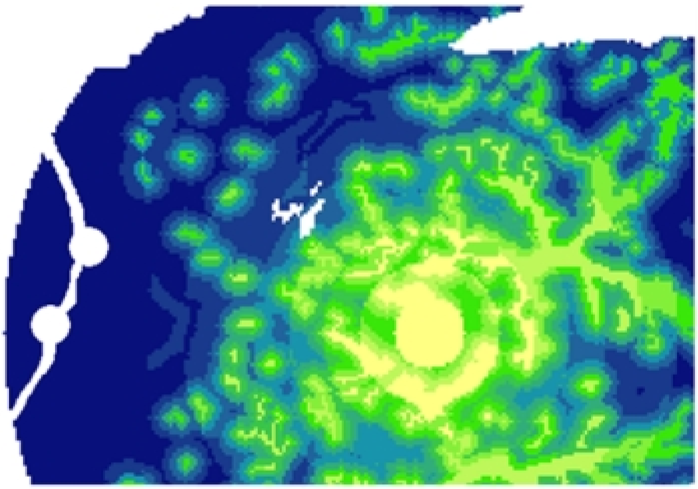


Figure 9a shows the major ridge features of the Fig 8 model highlighted, whilst Figure 9b emphasizes the 500 m annuli used to decrease range use with distance from the range centre.

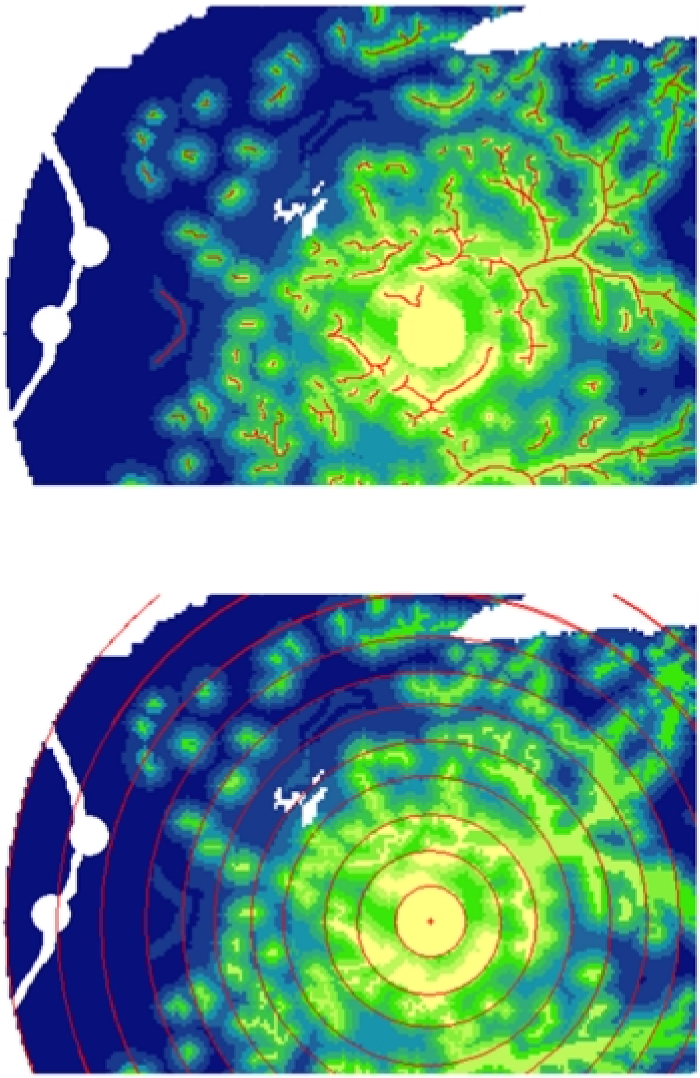
Summary of the PAT model

- Golden eagle ranges can be approximated by Thiessen polygons with a 6 km cut-off.
- Golden eagles are more likely to use habitat close to the range centre and close to a ridge.
- They are unlikely to use some habitats (water, closed canopy woodland, etc).
- These features were combined to produce an 'expected use' for each 50 m pixel within an eagle's range.
- More comprehensive descriptions of the PAT model can be found in McGrady *et al* (1997), McLeod *et al* (2007) and Fielding *et al* (2003).

Using the PAT in wind farm impact modelling

The predicted range loss resulting from a wind farm is the sum of 'use values' for pixels covered by (usually) a 500 m buffer drawn around the turbine locations.

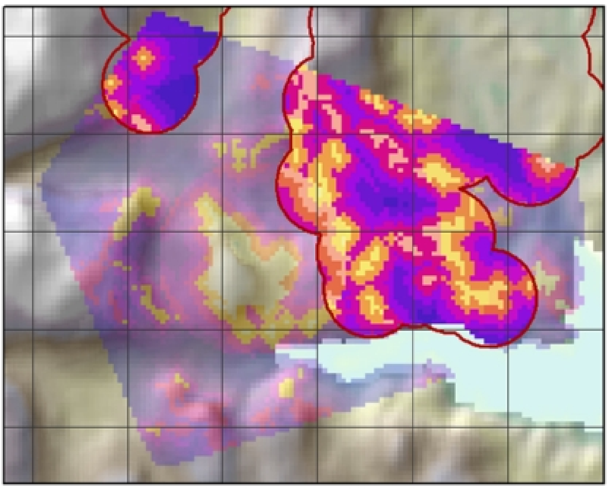
Figure 9 a Ridge lines overlaid on the PAT model in Figure 8; b 500 m annuli overlaid on the PAT model in Figure 8.



Example Range Loss Calculation

Figure 10 shows the PAT for a range that is predicted to lose just over 30% of its range (by use). The range size is 1,649 ha and the area within the wind farm footprint (500 m buffer drawn around the turbines) is 644.6 ha (39.1% of the range area). Therefore, in this example, the lost foraging habitat is less than the simple area loss. There are situations in which the reverse is true. The effective range loss is calculated by summing the use-values for all of the pixels covered by the wind farm footprint (Table 2). For example, there are 272 pixels with a ‘use-value’ of 0.0103%. Therefore, the combined habitat loss for these pixels is 2.79% of the predicted range (272 x 0.0103). Summing the cumulative use-value losses gives a total of 30.02% of the range. Since this is a large figure it is unlikely that the remaining 70% of the foraging habitat would be able to sustain a range. In this example, the pair of eagles are also constrained by neighbouring ranges on all sides. There is, therefore, little or no scope for any plasticity in the range use.

Figure 10. Predicted PAT model (with Thiessen boundaries constructed with respect to neighbouring ranges). The full PAT model is lighter than the affected area (within the red line buffer).



It is, however, important not to over-interpret predicted range losses. This is because the predictions lack the precision needed to, for example, compare losses of 5% or 6% from alternative wind farm footprints. In such cases it is probably wiser to look at barrier effects that may isolate part of the range

Table 2. Raw data used to estimate the range loss in Figure XX. Number is the number of pixels for each use-value.

Number	Use value (%)	Lost habitat (%)
1	0.0018	0.0018
13	0.0024	0.0309
53	0.0026	0.1394
18	0.0027	0.0481
6	0.0031	0.0188
23	0.0035	0.0794
104	0.0035	0.3598
59	0.0038	0.2248
104	0.0040	0.4150
26	0.0040	0.1040
88	0.0053	0.4620
16	0.0057	0.0910
111	0.0058	0.6416
184	0.0059	1.0930
72	0.0061	0.4370
74	0.0078	0.5772
216	0.0086	1.8554
34	0.0086	0.2938
161	0.0090	1.4522
34	0.0090	0.3074
272	0.0103	2.7907
20	0.0128	0.2566
46	0.0135	0.6205
67	0.0137	0.9186
203	0.0149	3.0146
163	0.0156	2.5428
65	0.0204	1.3234
6	0.0222	0.1331
117	0.0239	2.7928
17	0.0314	0.5336
93	0.0345	3.2122
30	0.0352	1.0560
50	0.0363	1.8145
1	0.0516	0.0516
4	0.0819	0.3275
Cumulative loss		30.021

Improving the PAT model

The current PAT model is essentially a topographic model in which probability of use is proportional to the distance from the range centre and ridge lines. In addition, territory boundaries are identified using simple geometric rules. Consequently, there is scope to improve the model by adding more 'biology'. However, before undertaking any more significant, and almost certainly very expensive, development work the benefits (model improvement as measured by accuracy and precision) need to assessed against the cost of such work.

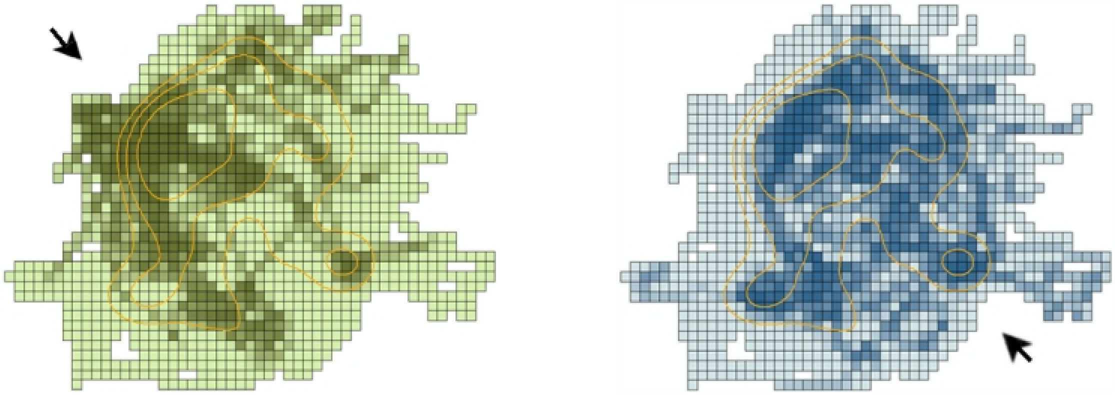
Before identifying possible improvements it is also important to understand what the current PAT model does not do. Currently it does not:

- model how sub-adults and adult 'floaters' use habitat, this is because it is based on an assumption of central place foraging;
- incorporate changes in use associated with changes in aspect, wind direction or wind speed;
- incorporate any information about prey distribution or prey transport costs;
- use topographic features such as ridges and rivers to delineate range boundaries (there is some empirical evidence that such features are used rather than simple mid-distances).
- make adjustments for a coastal nest site (other than excluding the sea from the model).

The golden eagle framework authors have, at various times, looked at the role of aspect in range use, including an unpublished analysis of some very detailed habitat use for data on Mull. They concluded that there was little evidence for any relationship with aspect. In places like California it is possible that aspect would play a role via thermals but in Scotland such an effect is unlikely to occur very often. Our limited analyses suggest that the interaction between wind direction and topography is more important in creating the uplift from ridges etc. There has been one analysis of the distribution on golden eagle flights with respect to wind direction (King *et al* 2006). Almost 2500 km of eagle flights were digitised from more than 1000 hrs observations over a two year period on a range with relatively simple and low level topography. Usage, corrected for survey effort, was compared qualitatively for days with prevailing north westerly and south easterly winds respectively and compared with usage for all flight data (Figure 11). Although the differences are not large there are subtle differences in patterns of use that are probably related to wind flow over ridges.

Unfortunately, wind flow in regions with complex topography is not simple and generally it can't be modelled from simple wind direction. The framework authors were of the opinion that the distribution of prey, and its context with respect to topographic features, was more likely to influence habitat usage. It seems very unlikely that it would be possible to refine habitat usage in the absence of a great deal of simultaneous data about position, weather, habitat and prey.

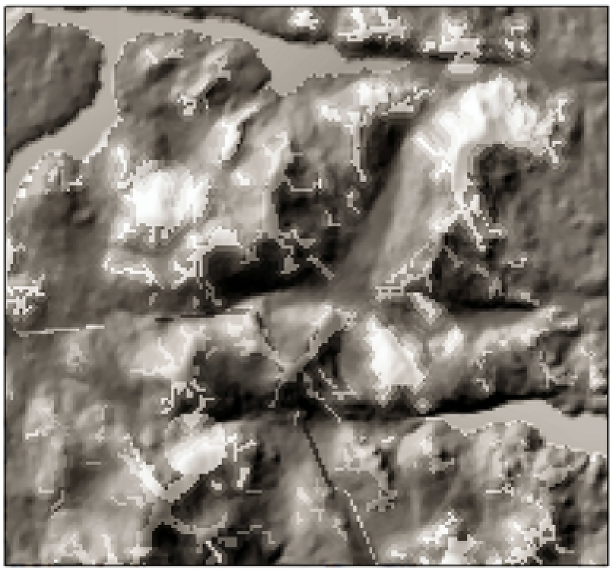
Figure 11. Spatial use by golden eagles with respect to wind direction, darker colours are associated with greater use. Wind direction is shown by the arrow. The contours are range-use kernels (25, 50 and 75%) of all flight activity.



Testing the PAT model predictions

Currently there have been no planned tests of the PAT model's predictions. Instead some data collected in association with wind farm impact assessments has been used for *ad hoc* qualitative assessments. One of the difficulties is that golden eagle ranges can be large and difficult to monitor visually because of restrictions created by the topography. For example, a non-coastal, isolated range is assumed to have an area of 113 km^2, based on the 6 km radius limit. If flight data were to be collected visually, and the current SNH 2 km visibility threshold was applied, then a very large number of vantage points would be needed to ensure complete coverage of the range. Obviously, this number is reduced if the visibility threshold is increased. Flight data for wind farm assessments are collected from vantage points that are designed to maximise coverage of the wind farm's footprint and are, therefore, unlikely to cover the entire eagle range. If the entire range is not covered then any assessment of relative use must be biased. Remotely sensed data (e.g. using GPS backpacks) could give much improved range use data compared with those from vantage points. However, the PAT model models range use for range holding birds only so this necessitates capturing and fitting GPS transmitters to range holding birds. Apart from the practical difficulties and cost implications of such captures there are ethical issues related to the effect that capture has on the bird. Gregory *et al* (2003) found that capturing birds for the original RIN study resulted in reduced reproductive success and less frequent use of the original nests after trapping.

Figure 12 PAT models for five adjacent ranges, lighter shades indicate greater predicted use. (Contains Ordnance Survey data © Crown copyright and database)



Nonetheless, there are some restricted data sets that can be used as general tests of the PAT model's predictions. All of these data come from the Hebridean islands of Skye and Lewis. It is important to understand that these are raw data and take no account of surveying effort. There is an inevitable link between the amount of survey effort and the number of recorded flights. However, it is still possible to look at the gross fit between recorded flights and the predicted use. In particular, are there are over-used regions which were not predicted and *vice-versa*? Examining Figures 12-15 leads to the conclusion that, at a minimum, the PAT correctly captures the most and least used parts of the range quite well.

Figure 13. Predicted range use overlain with flight lines for adult golden eagles. Note that these flights are derived from unequal survey efforts. Some areas of the map had more survey effort and others relatively little or none. (Contains Ordnance Survey data © Crown copyright and database)

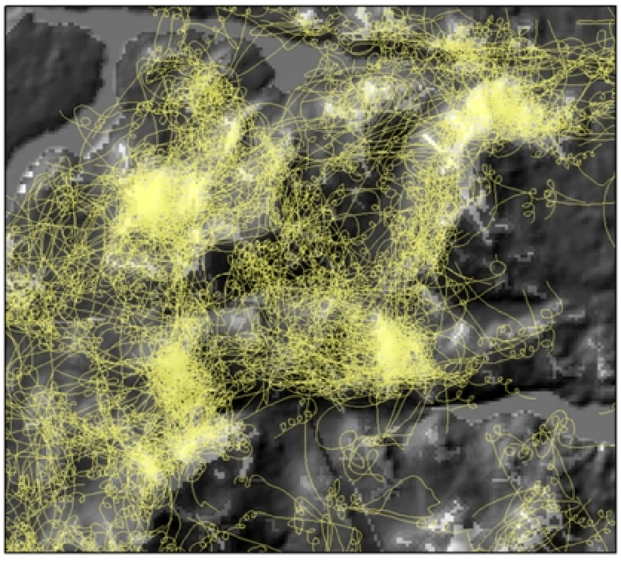


Figure 14. Enlarged section of Figure 13. (Contains Ordnance Survey data © Crown copyright and database)

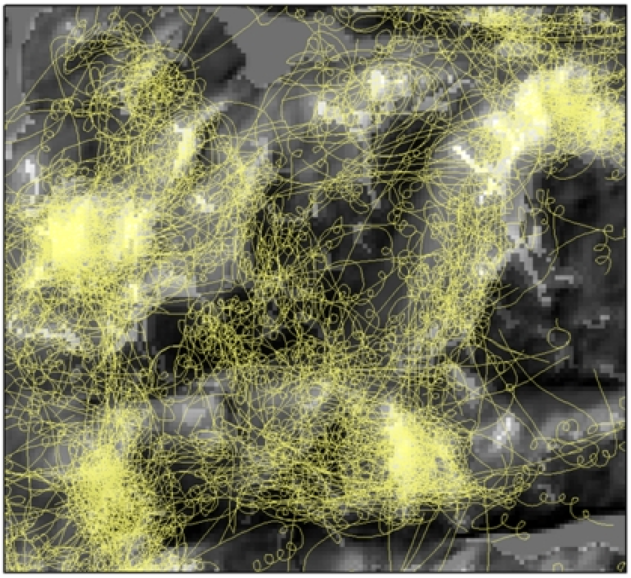
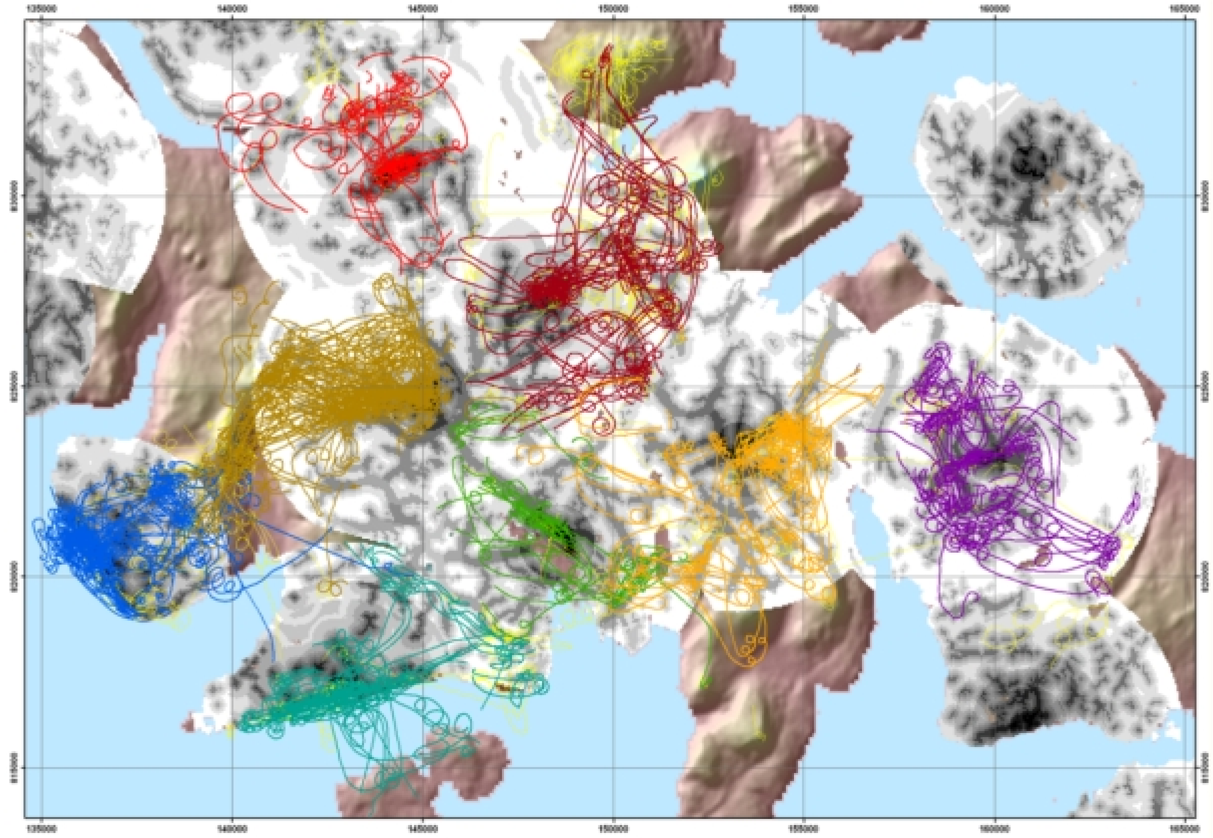


Figure 15 Flights recorded opportunistically on the Isle of Skye. (Contains Ordnance Survey data © Crown copyright and database)



The most intensive test of the PAT was undertaken by King *et al* (2006). This is from a range on Skye that had over 1000 hours of observation over a two year period. Unusually, almost all of the range was included in the vantage point viewsheds. Figure 16 shows the PAT predictions overlain with use isoline contours derived from effort corrected flight data. Also shown are the locations of 'looped' flights, i.e. locations where birds are gaining height by spiralling.

There are several observations from Figure 16. First, the standard PAT road exclusion buffer was used (splits the PAT predictions approximately SE – NW). It is clear from the loops and use kernel contours that this area was not avoided. However, the forestry exclusion (middle of right side of the image) had little use. Also, the PAT predictions used the standard 'range-centre' algorithm. The actual nest during these years was approximately 1 km north of the range centre. The distribution of loops fits the PAT model quite well and reflects the use of terrain features as an aid to movement around the range. The effort adjusted flight data has a less good fit to the PAT predictions. However, if the PAT is re-run but without exclusions and with the correct range centre (the nest site) the fit between empirical data and the PAT predictions is better (Figure 17). The flight use contours also illustrate the possible effects of prey resources since there was a large rabbit warren on a small hill in the SE corner of the range.

Figure 16 PAT predictions overlaid by the location of looped flights and range usage kernels (25, 50 and 75%) of all flight activity adjusted for survey effort. These are the same data as Figure 11. (Contains Ordnance Survey data © Crown copyright and database)

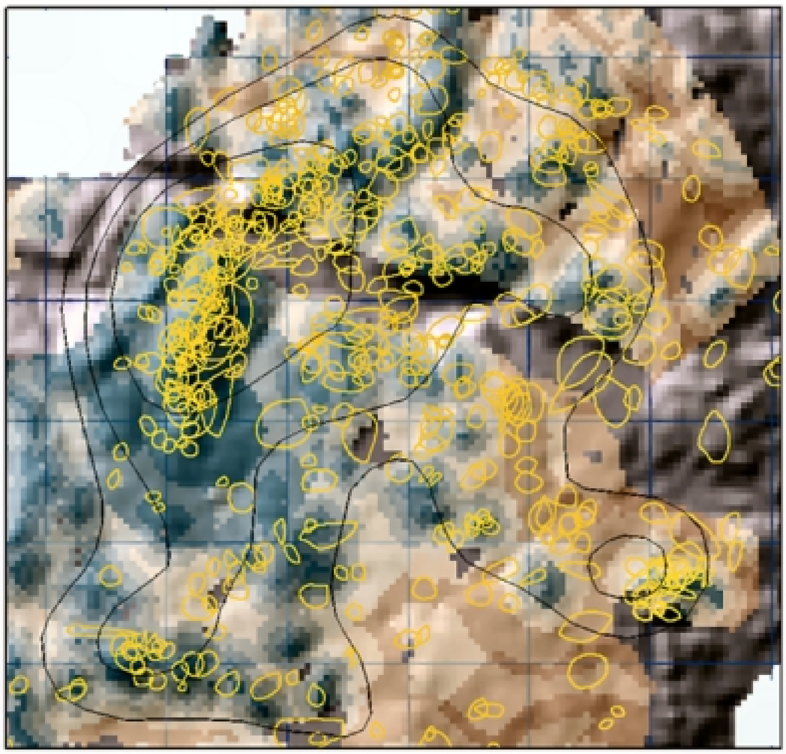
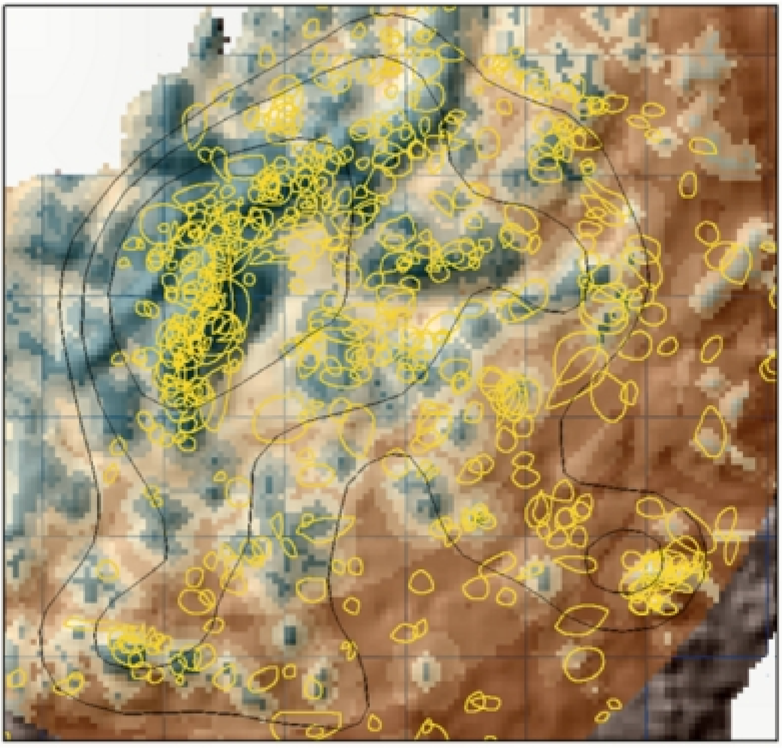


Figure 17 PAT predictions without exclusions overlaid by the location of looped flights and range usage kernels (25, 50 and 75%) of flight activity adjusted for survey effort. (Contains Ordnance Survey data © Crown copyright and database)



Population models

Population models, such as the Leslie matrix, are relatively simple matrix algebra models. The difficulties arise when assigning values to the parameters and, to a lesser extent, deciding on the model complexity. For example, should survivorship and reproductive output be assessed for each year in the life of an eagle or should it be modelled for a class of ages such as 'less than 5 years old'? The trade off is between model 'reality' and the number of estimated parameters. In general, simpler models with fewer parameters tend to be more robust.

The matrix algebra approach is essentially a computational short cut to preclude the need for repetitive calculations. Below, an example is shown diagrammatically (Figure 18), as a Leslie matrix and as a series of sequential calculations in Excel. The Leslie matrix for the model in Figure 17 is 5 x 5, made of four sub-adult classes plus an adult class. Only the adults (5th column) reproduce at a rate f. In these models f is 0.187 which is 50% of the mean rate from Eaton *et al* 2003. Only half of the productivity is used because this is a female-only model which assumes a 1:1 fledging ratio. S0, s1, s2 and s3 are the sub-adult survival rates (0.8, or 40% for sub-adults combined across the four years). The adult survival rate, v, is 0.952 which is the lowest rate for FCS in Whitfield *et al* 2006a. The initial population sizes were 79 (1st year), 63 (2nd year), 51 (3rd year), 41 (4th year) and 424 adults. These figures were derived from the reproductive rate (424 x 0.187 = 79.3) and subsequent 80% survival rates (2nd year = 79.3 * 0.8 = 63.4). This model does not include any density dependent processes.

Potentially, this omission allows a population to expand without limits. Although this is obviously unrealistic the simplification is not considered a problem when examining scenarios that predict declines.

Figure 18. Summary of a population model. Number of young fledged each year is shaded light yellow and the number of breeding pairs light pink. The fate of one cohort is highlighted in bold with arrows. 424 pairs at T0 fledged 79.0 young, of these 63.2 survived into their 2nd year, 50.6 into their 3rd year and 40.4 into their 4th year. 32.2 (40.4 x 0.8) of these survived to enter the adult population. At T3 there were 458.5 adults but only 95.2% of these (436.5) would survive into T4. These would be joined by the 32.2 cohort survivors from T0 to produce the new adult population of 468.9 pairs.

	T0	**T1**	**T2**	**T3**	**T4**	**T5**
Young	**79.0**	79.3	81.6	83.7	85.7	87.7
Year 2	63.0	**63.2**	63.4	65.3	67.0	
Year 3	51.0	50.4	**50.6**	50.7	52.2	
Year 4	41.0	40.8	40.3	**40.4**	40.6	
Adults	**424.0**	**436.3**	447.6	458.5	**468.9**	
Adult Survivors	**404.0**	415.3	426.1	436.5	**446.4**	

The model was implemented in ULM (Unified Life Models) software (Version 4.4, May 2006) originally developed by Legendre and Clobert (1995) and developed further by Stéphane Legendre (http://www.biologie.ens.fr/~legendre/ulm/ulm.html). The ULM software models a wide range of population dynamics using a model description text file. The parameters specified can also be modified interactively within the ULM environment. The mean growth rate was 1.0221.

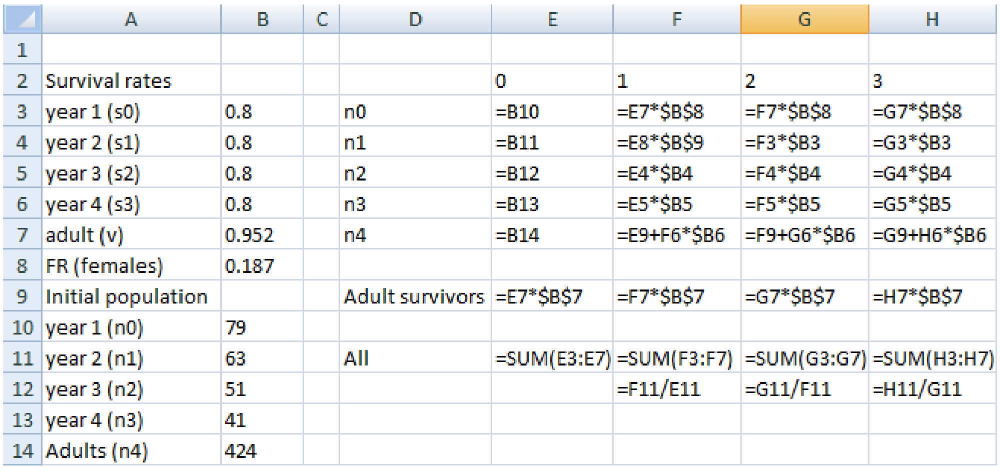
The Leslie matrix

```
0,   0,   0,   0,   f
s0,  0,   0,   0,   0
0,   s1,  0,   0,   0
0,   0,   s2,  0,   0
0,   0,   0,   s3,  v
```

The same model was implemented as a series of sequential calculations in Excel. There are minor differences in the predicted growth curves which can be explained by differences in the number of significant figures used in the models. The Excel model has a mean growth rate of 1.0230 and the correlation between the predicted population sizes was greater than 0.99. The mean scaled population structures are also identical to 2 dp: 0.12; 0.09; 0.07; 0.06; 0.66. Figure 19a shows part of the output and 18b shows the equations. The big advantage of the Excel model is that it is relatively simple to add extra mortality using simple subtraction (see below). For example, an additional bird could be subtracted from row 7 to reflect the loss of one bird per year. Similarly, the model can be made stochastic by adding noise to each instance of a parameter's use in the model. Poptools (Hood 2009) is particularly useful for this purpose.

Figure 19. (a) Excel implementation of the Leslie matrix model. Initial starting values are shaded light yellow. T0 – T5 are years; (b) Equations used in the model shown in a.

Survival rates				T0	T1	T2	T3	T4	T5
year 1 (s0)	0.8		n0	79.0	79.3	81.6	83.7	85.7	87.7
year 2 (s1)	0.8		n1	63.0	63.2	63.4	65.3	67.0	68.6
year 3 (s2)	0.8		n2	51.0	50.4	50.6	50.7	52.2	53.6
year 4 (s3)	0.8		n3	41.0	40.8	40.3	40.4	40.6	41.8
adult (v)	0.952		n4	424.0	436.3	447.6	458.5	468.9	479.9
FR (females)	0.187								
Initial population			Adult survivors	404	415.3	426.1	436.5	446.4	456.8
year 1 (n0)	79								
year 2 (n1)	63		All	658	670	683.5	698.6	714.5	731.5
year 3 (n2)	51		Growth rate		1.018	1.020	1.022	1.023	1.024
year 4 (n3)	41								
Adults (n4)	424								



	A	B	C	D	E	F	G	H
1								
2	Survival rates				0	1	2	3
3	year 1 (s0)	0.8		n0	=B10	=E7*B8	=F7*B8	=G7*B8
4	year 2 (s1)	0.8		n1	=B11	=E8*B9	=F3*$B3	=G3*$B3
5	year 3 (s2)	0.8		n2	=B12	=E4*$B4	=F4*$B4	=G4*$B4
6	year 4 (s3)	0.8		n3	=B13	=E5*$B5	=F5*$B5	=G5*$B5
7	adult (v)	0.952		n4	=B14	=E9+F6*$B6	=F9+G6*$B6	=G9+H6*$B6
8	FR (females)	0.187						
9	Initial population			Adult survivors	=E7*B7	=F7*B7	=G7*B7	=H7*B7
10	year 1 (n0)	79						
11	year 2 (n1)	63		All	=SUM(E3:E7)	=SUM(F3:F7)	=SUM(G3:G7)	=SUM(H3:H7)
12	year 3 (n2)	51				=F11/E11	=G11/F11	=H11/G11
13	year 4 (n3)	41						
14	Adults (n4)	424						

The following example uses data from the Argyll West and Islands NHZ as described in Whitfield *et al* (2006a). It assumes a starting population of 44 pairs with an annual productivity of 0.46 young per year (0.23 females fledged per pair). Survivorship rates are as above. The model is deterministic, i.e. no noise is added, and there is no density dependence. The equations were modified to allow for additional mortality. For example the equation in cell F6 in Figure 19b becomes "=(E5*$B5)-F16", where F16 contains the number of sub-adults killed that year. Similarly, the cell F7 equation becomes "=(E9+F6*$B6)-F15". In the results below a number of scenarios are modelled. In any year there are either 0 or 1+ birds killed in each or both age classes (year 4 sub-adults and adults). For simplicity deaths are assumed to occur at regular intervals: each year; every two years; every three years; every four years and every five years. The population growth rate, the number of ranges in year 25 and the time to reach notional population cap of 59 ranges are recorded for each simulation (Table 3).

Table 3. Results from population models incorporating additional mortality at five time intervals. The first row has no additional mortality. The number of adult and sub-adults 'killed' in each time period is shown in columns 2 and 3.

Scenario	Adult	Sub-Adult	Growth rate	Ranges after 25 years	Time (years) to reach 59 ranges
None	0	0	1.036	110	8
5 years	1	0	1.034	104	9
	0	1	1.034	105	9
	1	1	1.032	99	9
4 years	1	0	1.033	102	9
	0	1	1.034	104	9
	1	1	1.031	96	10
3 years	1	0	1.032	100	10
	0	1	1.033	102	10
	1	1	1.029	91	11
2 years	1	0	1.030	94	10
	0	1	1.031	97	10
	1	1	1.024	82	13
1 year	1	0	1.022	77	14
	0	1	1.025	84	13
	1	1	1.006	52	Not reached
	2	0	1.001	45	Not reached
	0	2	1.010	58	Not reached
	2	2	0.863	0	Not reached

As expected the additional mortality has a negative effect on both outcomes (growth rate and number of ranges). However, it is not until the last four scenarios that this becomes a significant problem. In all of the other scenarios it is likely that density dependence and range capacity would have resulted in the same final range total. The only effect of the additional mortality is the time taken to arrive at the population cap of 59 ranges. Even with 1 adult death each year (25 in total) the cap is only delayed by six years. There are some important caveats about the above model. First it only applies to the parameter values used. If any of them are reduced the effects will become significant at a reduced mortality level. Secondly, all parameter values are fixed and mortality is applied at a regular intervals. It would be wise to run the model over hundreds of simulations with noise added to the parameter values and the time interval between deaths.

Because population models are built around the same underlying mathematics it should make no difference which software is used. The main difference in software such as Vortex (Lacy *et al.,* 2005), RAMAS (http://www.ramas.com/index.htm), Poptools (Hood 2009), ULM (Legendre and Clobert 1995) and an Excel spreadsheet is largely related to level of complexity that can be modelled. For example, Vortex can be used to model the genetic consequences of inbreeding while RAMAS is essentially a spatial modelling tool. Unfortunately it is often difficult to justify a complex golden eagle population model because there is usually insufficient information on parameter values to justify anything other than relatively simple Leslie matrices. Generally it is more important to provide the model description files that would provide the transparency needed for others to repeat the models in same or different software, while allowing further tests of the model's sensitivity to the particular parameter values adopted in the model.

The original GEPM (Golden Eagle Productivity Model) was developed for a specific request to model the effect of removing young birds for the Irish re-introduction (O'Toole *et al,* 2002). The model was subsequently modified for the framework analyses (Whitfield *et al* 2006a) and later for wind farm impact assessments. As with the PAT model, it is important not to over-interpret the output from the GEPM. Rather, the model was used in the framework analyses and persecution papers (Whitfield *et al* 2003, 2004ab, 2006a) to investigate general patterns rather than a single scenario. Consequently, framework results are presented for a range of survival values because we had little knowledge about actual values and how they vary across regions. The regional NHZ models were run with the aim of discovering how robust the populations were so that their conservation status could be assessed, i.e. it was overall patterns which were important. Basically, should a population be expanding given its estimated productivity?

The GEPM has also been modified to provide an estimate of the number of floaters in the population. The number of floaters is the difference between the number of birds surviving to age 4 and the number of vacant breeding territories. Floaters provide an alternative way of understanding the population level impacts of additional mortality by providing an estimate of how much 'spare capacity' exists in the population. The ability of a golden eagle population to buffer itself against changes in mortality is partly dependent on the number of 'floaters'. Populations of large raptors such as eagles, which are in a favourable status, usually contain a large number of non-breeding adults, or floaters (Hunt, 1998; Kenward *et al*., 2000). These adult floaters can rapidly replace adult breeders when vacancies arise in breeding territories. Consequently the population will not begin to decline if ranges become vacant. However, if there are few floaters breeding territories may remain vacant, leading to a possible decline in productivity.

There has been some discussion about the impact that the population cap has on impact assessments involving floaters. The population cap is the number of potential ranges. If a population has a cap adult birds must either enter the floater population or migrate to search for a vacant range elsewhere when all local potential ranges are occupied. It is worth noting that currently the GEPM does not incorporate immigration or emigration, i.e. a closed population is assumed.

The population cap is important because if the cap is changed so will the number of floaters. Unfortunately, as with many aspects of golden eagle ecology in Scotland, there is little real information about actual range capacities. For example, in 2003 the Argyll West and Islands NHZ had 15 vacant ranges out of 59. It is also worth noting that the 59 does not include the ranges lost during the early afforestation of this NHZ. Therefore, in the framework FCS analyses (Whitfield *et al* 2006b) this NHZ was given an initial population of 44 pairs (59 potential – 15 vacant). However, there was evidence of birds in some of these vacant ranges, but insufficient evidence to call them occupied (Fielding *pers comm.*). Even though this NHZ is well surveyed it is very probable that some of the vacant ranges were occupied or have been subsequently occupied (Haworth *pers comm.*). The cap could have been 64 because 10 historic ranges were merged into 5 extant ranges during the framework analyses, largely on the basis of loses to forestry. It is possible that some of these merged ranges will subsequently split, as happened on Mull in 2008 (Haworth *pers comm.*). So, although 60 may be a little high for the current population the cap should not be much lower than 55.

Consequently, the population cap must always be viewed as approximate because range occupancy is dynamic and we only have two or three national survey snapshots for most ranges. It is unlikely that any of the NHZs are saturated in any year but it is not always the same ranges which are vacant. For example, one of the Mull ranges, which has been empty for many years (thought to be a result of afforestation), fledged one young in 2008. Also, the Eisgein population on Lewis, which has been monitored since 2003, expanded again in 2008 despite its already high density. The expansion across the western isles seems to be a simple increase in density, i.e. range packing rather than occupation of new habitat. It is certainly possible for the Argyll West and Islands NHZ population to expand in this way since many ranges do not have immediate neighbours on all sides.

If the cap is lowered this will have a consequence for floaters but this will be effective for different periods depending on the number of vacant ranges. For example, a cap of 50 for the Argyll West and Islands NHZ population leaves six empty ranges (44 start value) so these might be filled quite quickly, conversely a cap of 60 leaves 16 empty ranges which would take longer to fill. However, in an expanding population these will also be filled over a longer time period. Comparing GEPM results for 50 and 60 caps results in a great deal of similarity with the possible exception of some of the extreme values. Over the range of realistic population parameter values the differences, with respect to filling all ranges, are negligible.

Despite all of the caveats above, the most important outcome of a population model, with respect to an impact assessment, is “does the population have the continued potential to expand” after any additional mortality is added. However, as noted earlier, the only parameters that we have much confidence in are the number of occupied ranges and the fledging rates. Even then these are largely historic data, for example for many ranges it is the 2003 national survey. In much of western Scotland productivity was poor 2005-07 but recovered in 2008. For example, on Skye, the annual productivity (young per pair) between 2001 and 2010 was: 0.45; 0.55; 0.50; 0.55; 0.42; 0.35; 0.31; 0.59; 0.45 and 0.52. A conservative and consistent approach would be to take the last known population size and fledging rates and then assume the minimum survival rates for a region which is in FCS (Whitfield *et al*, 2006a). These survival rates came from an intensive literature survey and certainly seem to be conservative in Ireland where there is better data. Indeed they are minimum rather than optimal rates for FCS.

The use of population modelling, including effects on floaters is illustrated below using extracts from a report by Haworth Conservation Ltd (2006) in connection with the proposed Edinbane wind farm on Skye. It is worth noting that this report subsequently went through a judicial review.

Extract begins

Whitfield *et al* (2006a) suggested a minimum PAS of 40% survival from fledging to age four was required to achieve favourable conservation status. Because the Western Seaboard NHZ was assessed to be in favourable conservation status (Whitfield *et al*, 2006) it is reasonable to assume that the PAS rate is at least 0.4 for both the Skye and larger NHZ populations. Whitfield *et al* (2004b) also estimated that, in the absence of interference, the Scottish golden eagle annual adult survival rate should be 94.2% – 95.8% (TR = 0.042 – 0.058). Consequently, Whitfield *et al* (2006a) proposed an annual adult survival rate of 95.12% (20 years of territory occupation) as a minimum limit for favourable conservation status (TR >0.049 in Table 6.1 - 6.3). Crane and Nellist (in Watson 1997) suggested that, on Skye, the adult survival rate was 97.5%, indicating a TR which is less than 0.042. In these simulations we assume the more conservative, minimum rate of Whitfield *et al* (2006a).

However, given the uncertainty in the parameter values, and a need to assess the impact of additional mortality on the population's potential trajectory, the models were run over a combination of values. The Skye population models used six levels of adult mortality (TR = 0.04 – 0.07, 15 – 25 years of range occupancy), four levels of sub-adult survival (0.25 to 0.40) and eight fledging rates (0.35 to 0.53). The Western Seaboard NHZ population models used six levels of adult mortality (TR = 0.04 – 0.07, 15 – 25 years of range occupancy), four levels of sub-adult survival (0.30 to 0.45) and eight fledging rates (0.25 to 0.425).

The stochastic models were run 100 times for each combination of parameter values and the size of the predicted populations, after 30 years, was measured. When birds have such potentially long life expectancies it is important to run the model for long periods. Tables 4 and 5 present the results of the simulations, using the mean number of pairs from the simulations over the last 10 years of a 30-year simulation.

The simulation results in Table 4 and 5 suggest that, even with lower than observed fledging rates, both regional populations are expected to be stable or capable of expansion as long as the pre-adult survival rate is at least 35% (PAS>=0.35 in Table 4). This is not too surprising since, as with most long-lived animals with delayed reproduction, it is adult survival that has the largest effect on population trajectories. Both Skye and Mull, strongholds of the Western Seaboard population, are thought to have adult survival rates towards the top of the range estimated by Whitfield *et al* (2004b).

Table 4 Mean golden eagle population size for years 21-30 in a 30 year population simulation of the Skye population. The model starts with a population of 31 pairs and a population cap of 34 pairs. PAS – Pre-Adult Survival rate, TR – proportion of ranges vacated each year, LE – Life Expectancy or the mean number of years that a pair occupies a range.

			Fledging Rate							
PAS	**TR**	**LE**	**0.35**	**0.38**	**0.40**	**0.43**	**0.45**	**0.48**	**0.50**	**0.53**
0.25	0.04	25.0	**33.3**	**33.6**	**33.8**	**34.0**	**34.0**	**34.1**	**34.2**	**34.1**
	0.05	22.2	**30.6**	**31.5**	**32.3**	**32.5**	**33.2**	**33.5**	**33.8**	**34.1**
	0.05	20.0	27.0	28.0	29.1	**30.1**	**30.8**	**31.9**	**32.6**	**32.9**
	0.06	18.2	23.8	25.0	26.1	27.0	27.9	29.4	**30.8**	**32.4**
	0.06	16.7	21.2	22.7	23.7	24.6	25.9	27.4	29.0	**31.0**
	0.07	15.4	19.5	20.5	21.7	22.8	24.1	25.5	27.3	29.4
0.30	0.04	25.0	**33.9**	**34.1**	**34.1**	**34.2**	**34.2**	**34.1**	**34.2**	**34.0**
	0.05	22.2	**32.5**	**33.0**	**33.5**	**33.9**	**34.0**	**34.2**	**34.2**	**34.2**
	0.05	20.0	29.3	**30.7**	**31.6**	**32.8**	**33.4**	**33.9**	**34.1**	**34.3**
	0.06	18.2	26.4	27.9	29.2	**31.0**	**32.7**	**33.5**	**34.1**	**34.2**
	0.06	16.7	24.0	25.4	27.2	29.7	**31.3**	**33.0**	**33.7**	**34.0**
	0.07	15.4	22.3	23.7	25.4	28.2	30.0	**31.9**	**32.5**	**33.3**
0.35	0.04	25.0	**34.0**	**34.1**	**34.1**	**34.1**	**34.0**	**34.0**	**33.9**	**33.9**
	0.05	22.2	**33.6**	**33.9**	**34.0**	**34.1**	**34.2**	**34.2**	**34.1**	**34.1**
	0.05	20.0	**31.9**	**32.6**	**33.6**	**33.9**	**34.2**	**34.3**	**34.3**	**34.2**
	0.06	18.2	29.4	**31.2**	**32.8**	**33.7**	**34.1**	**34.3**	**34.4**	**34.3**
	0.06	16.7	27.2	29.7	**31.9**	**33.5**	**33.9**	**34.2**	**34.3**	**34.3**
	0.07	15.4	25.3	28.6	**30.5**	**32.1**	**33.1**	**33.8**	**34.0**	**34.2**
0.40	0.04	25.0	**34.2**	**34.2**	**34.1**	**34.0**	**34.0**	**33.9**	**33.8**	**33.8**
	0.05	22.2	**34.0**	**34.1**	**34.2**	**34.1**	**34.1**	**34.1**	**34.0**	**33.9**
	0.05	20.0	**33.7**	**33.9**	**34.2**	**34.3**	**34.3**	**34.3**	**34.1**	**34.1**
	0.06	18.2	**32.6**	**33.7**	**34.2**	**34.3**	**34.4**	**34.3**	**34.2**	**34.1**
	0.06	16.7	**32.1**	**33.3**	**34.0**	**34.1**	**34.3**	**34.4**	**34.2**	**34.2**
	0.07	15.4	**30.2**	**32.3**	**33.3**	**33.9**	**34.1**	**34.2**	**34.1**	**34.2**

Note: Figures are in bold if they are not significantly below the starting population of 31 occupied ranges. A population cap was set at 34 for these simulations. FR is the fledging rate (0.35 – 0.53 young per occupied range per year); PAS (Pre-adult survival) is the proportion of sub-adults surviving to adult age (0.25 – 0.40) and TR is the range turn-over rate (0.04 – 0.07 or a range of adult life expectancies (LE) from 15.4 to 25 years). The cells with the lighter shading highlight the fledging rates that are consistent with the empirical data for the Western Seaboard golden eagle population. The darker shaded row marks the combination of PAS and TR which is consistent with the minimum requirements for favourable conservation status (Whitfield et al, 2006a).

applies, i.e. TR = 0.05 or 5% of adults die each year of natural causes.

Table 5: Mean golden eagle population size for years 21-30 in a 30 year population simulation of the Western Seaboard NHZ population. The model starts with a population of 74 pairs and a population cap of 100 pairs. Legend as for Table 4.

			Fledging Rate							
PAS	**TR**	**LE**	**0.300**	**0.325**	**0.350**	**0.375**	**0.400**	**0.425**	**0.450**	**0.475**
0.30	0.040	25.0	**75.2**	**82.5**	**89.0**	**94.4**	**97.9**	**99.7**	**100.0**	**100.0**
	0.045	22.2	66.9	72.1	**77.6**	**83.9**	**91.3**	**95.3**	**99.2**	**100.0**
	0.050	20.0	58.4	63.6	68.7	**74.7**	**80.4**	**86.3**	**93.8**	**97.3**
	0.055	18.2	51.5	56.0	60.3	66.5	71.6	**78.0**	**84.6**	**91.7**
	0.060	16.7	45.7	49.9	53.3	58.8	62.9	69.3	**73.9**	**79.7**
	0.065	15.4	40.1	43.2	47.4	52.0	56.6	59.9	66.9	71.7
0.35	0.040	25.0	**85.7**	**93.6**	**98.1**	**99.8**	**100.1**	**99.9**	**100.1**	**100.2**
	0.045	22.2	**76.2**	**83.3**	**91.3**	**97.6**	**99.3**	**99.9**	**99.9**	**100.0**
	0.050	20.0	67.8	72.6	**80.1**	**86.6**	**95.6**	**99.1**	**99.9**	**100.0**
	0.055	18.2	58.7	64.4	72.1	**77.9**	**86.6**	**92.1**	**96.8**	**99.6**
	0.060	16.7	51.9	56.7	63.4	70.8	**75.4**	**81.7**	**89.3**	**95.0**
	0.065	15.4	46.1	51.1	56.0	61.4	67.2	**74.9**	**80.1**	**87.5**
0.40	0.040	25.0	**96.0**	**99.6**	**99.9**	**100.0**	**100.0**	**100.1**	**100.0**	**99.9**
	0.045	22.2	**86.9**	**95.2**	**99.3**	**100.0**	**100.0**	**100.1**	**100.1**	**100.0**
	0.050	20.0	**75.3**	**86.2**	**94.0**	**98.3**	**99.6**	**99.9**	**100.1**	**100.0**
	0.055	18.2	68.2	**75.4**	**82.7**	**90.3**	**97.5**	**99.7**	**100.1**	**100.0**
	0.060	16.7	59.3	66.0	**73.9**	**82.0**	**90.0**	**95.8**	**99.3**	**100.0**
	0.065	15.4	52.8	58.0	65.9	72.6	**79.6**	**85.9**	**92.8**	**98.1**
0.45	0.040	25.0	**99.5**	**100.0**	**100.1**	**100.0**	**100.1**	**99.9**	**99.9**	**99.8**
	0.045	22.2	**96.6**	**99.7**	**100.1**	**100.0**	**100.1**	**100.1**	**100.1**	**99.9**
	0.050	20.0	**87.1**	**95.9**	**99.4**	**99.9**	**99.9**	**100.0**	**100.2**	**100.0**
	0.055	18.2	**76.4**	**88.2**	**94.7**	**99.1**	**100.0**	**99.9**	**99.8**	**100.1**
	0.060	16.7	68.0	**76.9**	**87.1**	**94.1**	**98.0**	**99.9**	**100.0**	**100.2**
	0.065	15.4	59.7	67.9	**75.7**	**83.9**	**91.2**	**98.3**	**99.8**	**100.2**

Note: Figures are in bold if they are not significantly below the starting population of 74 occupied ranges. A population cap was set at 100 for these simulations. FR is the fledging rate (0.25 – 0.425) young per occupied range per year); The fledging rates from Whitfield et al (2006a) were 0.46 (2003 National Census Data) and 0.44 (mean of the 1982, 1992, 2003 National Census records).PAS (Pre-adult survival) is the proportion of sub-adults surviving to adult age (0.25 – 0.40) and TR is the range turn-over rate (0.04 – 0.07 or a range of adult life expectancies (LE) from 15.4 to 25 years). Shading is as described for the previous table.

The GEPM was modified to estimate the number of floaters and the results are shown in Table 6. It is clear from these results that, under even conservative estimates of the three main population parameters both populations have significant buffering potential. For example, on Skye with 40% pre-adult survival and 20-year range occupancy there are predicted to be 4 -5 new floaters available each year, even when the fledging rate is only 0.375. This is consistent with the predictions in Table 4 which suggests that under these conditions the population has reached its cap of 34 ranges, meaning that any additional adults must enter the floating population.

Table 6 Mean numbers of additional floaters available each year under a range of population scenarios. Legend as for Table 6.1. Results are shown for the Skye and Western Seaboard NHZ populations. Shading of cells is as described in Table 6.1.

			Skye					**Western Seaboard NHZ**				
			Fledging rate									
PAS	**TR**	**LE**	**0.375**	**0.400**	**0.425**	**0.450**	**0.475**	**0.375**	**0.400**	**0.425**	**0.450**	**0.475**
0.30	0.040	25.0	4.1	4.5	5.4	6.0	6.5	9.7	11.7	15.8	19.2	22.6
	0.045	22.2	3.4	3.7	4.3	4.7	5.3	8.4	9.4	11.7	14.4	18.6
	0.050	20.0	2.9	3.2	3.6	4.0	4.4	7.4	8.5	9.6	11.2	14.0
	0.055	18.2	2.6	2.8	3.2	3.6	3.9	6.5	7.5	8.8	9.6	11.4
	0.060	16.7	2.3	2.6	2.9	3.3	3.8	5.7	6.8	7.8	8.8	10.2
	0.065	15.4	2.1	2.3	2.7	3.2	3.6	5.2	6.1	6.8	7.8	8.9
0.35	0.040	25.0	5.5	6.3	6.8	7.8	8.9	16.2	20.0	23.3	26.5	31.6
	0.045	22.2	4.3	4.8	5.5	6.3	7.1	12.3	16.5	20.9	23.9	26.6
	0.050	20.0	3.6	4.1	4.6	5.1	6.3	9.7	12.1	16.2	20.2	24.0
	0.055	18.2	3.3	3.8	4.2	4.7	5.7	8.9	10.4	11.7	14.7	20.1
	0.060	16.7	2.9	3.5	4.0	4.7	5.6	7.9	9.1	10.6	12.3	15.0
	0.065	15.4	2.7	3.2	3.8	4.4	5.1	6.9	8.2	9.4	10.5	12.3
0.40	0.040	25.0	7.3	7.8	8.9	9.8	10.9	23.2	26.4	31.1	35.3	38.7
	0.045	22.2	5.4	6.2	7.1	8.2	9.3	19.8	23.9	28.4	31.9	35.7
	0.050	20.0	4.5	5.4	6.2	6.9	7.8	14.8	19.6	24.1	27.6	32.1
	0.055	18.2	4.1	4.9	6.1	6.9	8.0	11.6	14.2	18.5	22.9	29.3
	0.060	16.7	3.8	4.7	5.7	6.6	7.9	10.1	12.0	15.1	19.4	24.2
	0.065	15.4	3.6	4.6	5.1	6.1	6.9	9.1	10.7	12.4	14.7	21.4
0.45	0.040	25.0	9.0	10.1	11.4	12.2	13.6	31.0	34.1	39.0	43.3	48.4
	0.045	22.2	6.6	8.3	9.0	9.6	11.1	26.7	31.5	35.4	40.4	44.5
	0.050	20.0	5.9	6.8	8.0	9.0	9.7	22.6	26.9	32.4	38.9	41.9
	0.055	18.2	5.3	6.8	7.6	8.4	9.8	17.9	23.4	27.0	31.5	37.4
	0.060	16.7	5.1	6.2	7.4	8.7	9.8	13.8	17.7	23.5	29.0	32.7
	0.065	15.4	5.1	5.8	7.1	8.1	9.1	11.7	14.3	19.8	24.1	29.4

Haworth Conservation (2006) also used a simple arithmetic approach to explain the effects of additional mortality. The Western Seaboard NHZ has at least 74 pairs with an estimated annual range turnover rate of 0.045. If floaters are ignored the population of 74 pairs is made up of 148 adults. If the range turnover is 0.045 this means that 6.66 adults (148 x 0.045) are expected to die each year from natural causes. Over 25 years this is 167 adult deaths. Although the region around the Edinbane and Ben Aketil wind farms appears to be little used by adult birds it is still possible to examine a range of scenarios. For example, if the wind farms were predicted to kill one adult every four years (0.25 per year) this would increase the annual adult mortality from 6.66 to 6.91 and the 25-year total would be 173. The range turnover would increase from 0.045 to 0.047 (LE reduced from 22.2 to 21.4). The effects of four adult mortality scenarios are shown below for the Skye and Western Seaboard NHZ populations.

Western Seaboard NHZ population (74 occupied ranges, TR = 0.045)

Additional, annual,	**Adult deaths**		**New values**	
wind farm mortality	**Yearly**	**25 year**	**TR**	**LE**
0.00	6.7	167	0.045	22.4
0.25	6.9	173	0.047	21.4
0.50	7.2	179	0.048	20.7
0.75	7.4	185	0.050	20.0
1.00	7.7	192	0.052	19.3

Skye population (31 occupied ranges, TR = 0.045)

Additional, annual,	**Adult deaths**		**New values**	
wind farm mortality	**Yearly**	**25 year**	**TR**	**LE**
0.00	2.8	70	0.045	22.4
0.25	3.0	76	0.049	20.4
0.50	3.3	82	0.053	18.8
0.75	3.5	89	0.057	17.5
1.00	3.8	95	0.061	16.4

Using the lower limit for TR set by Whitfield *et al* (2006) for favourable conservation status, the NHZ population could withstand an additional, annual adult loss of 0.5 adults. On Skye, the limit is an additional 0.25 annual, adult mortality. However, it is important to remember that the Whitfield *et al* (2006a) evaluations refer to NHZ populations. It is obvious that the same absolute adult mortality must have a greater effect when the size of the population is reduced.

Similar calculations can be carried out for productivity and pre-adult survival. For example, if the 74 pairs in the NHZ have a mean fledging rate of 0.45 the population should fledge, on average, 33.3 birds (74 x 0.45) per year or 833 over the 25 years expected life time for a wind farm. Assuming that Whitfield *et al's* (2006a) lower limit for favourable conservation status of 0.40 applies to pre-adult survival (PAS), this means that 13.3 of the fledged birds are expected to survive to age four. Over 25 years 500 of the 833 fledged birds are expected to die from 'natural' causes, leaving 333 survivors. Using a TR of 0.045 (see above) 167 adults from the NHZ are expected to die over the same period, leaving an excess of 166 adults that would enter the floating population. At a lower fledging rate of 0.4 the excess is 130 adults.

If any additional mortality to sub-adults, arising from the sub-adults, is cumulative to the natural rate the effects can be estimated by reducing the number of birds surviving to become adults. However, this is not a simple calculation because the impact of the death of a sub-adult bird depends on its age. This is explained using the Skye population as an example. In this calculation, a FR of 0.40 and a PAS of 0.40 are assumed. A PAS of 0.4 implies an annual sub-adult survival rate of 0.795. Finally, it is assumed, for simplicity, that the wind farms kill one sub-adult (less than 4 years old) each year. Predicted adult losses are shown in Table 7.

Table 7 Impact of one additional sub-adult death on the number of birds surviving to become adults. (31 pairs, PAS = 0.4, FR = 0.4). The last row shows the estimated reduction in the number of adults. Figures in bold show the predicted impact of the additional death of one bird in the age class.

Age	No wind farm mortality	Age class of bird killed by the wind farm			
		Yr1	**Yr2**	**Yr 3**	**Yr 4**
Fledged	12.4	12.4	12.4	12.4	12.4
0 to 1	9.9	**9.1**	9.9	9.9	9.9
1 to 2	7.8	7.2	**7.0**	7.8	7.8
2 to 3	6.2	5.7	5.6	**5.4**	6.2
3 to 4	5.0	4.6	4.5	4.3	**4.2**
Additional Loss	0.0	-0.4	-0.5	-0.6	-0.8

If 12.4 birds are fledged 9.9 should survive to age 1 (79.5% survival). However, if one of these is killed by the wind farm the number drops to 9.1, leading to an eventual loss of 0.4 adult birds. The number of year 1 birds surviving is not 8.9 because the bird that was killed had a 20.5% chance of dying from other causes. If the additional death is to a bird in its second year (1 to 2), the number surviving to age two decreases from 7.8 to 7.0 and the number of adults is 0.5 less. Although the effect of the additional mortality is the same for each transition between age classes the effect on the number of adults is magnified as birds get older because there are fewer, subsequent, natural deaths. A simple mean of these reductions leads to an average loss of 0.6 adults under these conditions. However, the simple mean is an overestimate because the death is more likely to apply to younger birds. At its simplest this just reflects the fact that there are more younger birds, although it also likely that the inexperience of younger birds could make them more susceptible. An adjustment, made solely on the numbers of birds in each age class, reduces the mean number of adult birds killed from 0.6 to 0.55.

Although the loss of future adults is independent of the fledging rates, the impact on the population is not. This is because there is a minimum requirement for new adults to replace natural adult mortality. On Skye, a TR of 0.045 implies 70 natural deaths of range holding birds over 25 years. As long as at least 70 new adults arrive in the population it should remain stable. However, it is probably wise to apply a safety margin of at least 1.5 additional adults (floaters) per year. Thus the minimum requirement is 108[4] new adults over 25 years. It is important to remember that birds enter an existing floater population and these birds can be expected to have a higher, natural, survival rate than younger birds. Therefore, the size of the floater population should be much larger than the 1.5 birds which enter it each year. Tables 8 and 9 shows the expected number of new adults entering the Skye and NHZ populations over 25 years given a wide range of fledging rates and additional wind farm mortalities.

A TR of 0.045 implies 167 natural deaths of range holding birds in the NHZ population over 25 years. Therefore, as long as at least 167 new adults arrive in the population it should remain stable.

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[4] 70 birds lost is 2.8 per year (70/25). So, at least 2.8 floaters are needed for stability. However, to give a region of error assume that the population needs 2.8 + 1.5 = 4.3 floaters per year. This is 107.5 over 25 years.

However, it is probably wise to apply a safety margin of at least three additional adults (floaters) per year. Thus the minimum requirement is 242 new adults over 25 years.

Table 8. Expected number of new adults entering the Skye population assuming a range of fledging rates and additional losses of sub-adult birds arising from collisions with turbines. Figures in bold are below the threshold suggested above for a range turnover rate of 0.045. If the TR is 0.05 the threshold increases to 115 new adults.

Additional annual sub-adult deaths	Annual loss of potential adults (adjusted for age structure)	Fledging rate				
		0.30	0.35	0.40	0.45	0.50
0.0	0.00	**93.0**	108.5	124.0	139.5	155.0
0.2	0.11	**90.3**	**105.8**	121.3	136.8	152.3
0.3	0.16	**88.9**	**104.4**	119.9	135.4	150.9
0.4	0.22	**87.6**	**103.1**	118.6	134.1	149.6
0.5	0.27	**86.2**	**101.7**	117.2	132.7	148.2
0.6	0.33	**84.8**	**100.3**	115.8	131.3	146.8
0.7	0.38	**83.5**	**99.0**	114.5	130.0	145.5
0.8	0.44	**82.1**	**97.6**	113.1	128.6	144.1
0.9	0.49	**80.7**	**96.2**	111.7	127.2	142.7
1.0	0.55	**79.4**	**94.9**	110.4	125.9	141.4
1.1	0.60	**78.0**	**93.5**	109.0	124.5	140.0
1.2	0.66	**76.6**	**92.1**	**107.6**	123.1	138.6
1.3	0.71	**75.3**	**90.8**	**106.3**	121.8	137.3
1.4	0.76	**73.9**	**89.4**	**104.9**	120.4	135.9
1.5	0.82	**72.5**	**88.0**	**103.5**	119.0	134.5

As with the adults, the NHZ population is able to tolerate a larger rate of additional sub-adult mortality, which is obviously linked to the relative sizes of the two populations. It is not until the additional sub-adult annual losses rise above 1.2 per year (annual loss of 0.66 potential adults) that a threat to either population is likely to arise. Using a wind farm mortality rate of 1.0 per year as a maximum (loss of 0.55 potential adults), is precautionary and leaves the population with a reasonable level of buffering against unforeseen increases in natural mortality.

Unusually, there has been an ‘experimental” test of these predictions because seven young birds were removed from nests on Skye, between 2001 and 2004[5], as part of the Irish golden eagle re-introduction programme (O’Toole *et al*, 2002). A further five were removed from the NHZ (Mull and Canna), giving a total of 12 birds over five years. There have been no apparent adverse consequences arising out of these removals, which are equivalent to an additional loss of at least two sub-adults per year.

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[5] http://www.goldeneagle.ie/info_centre/index.html Skye: 2001 (1), 2002 (2), 2003 (2), 2004 (2); Mull 2003 (2), 2004 (1); Canna 2003 (1), 2004 (1).

Table 9. Expected number of new adults entering the Western Seaboard NHZ population assuming a range of fledging rates and additional losses of sub-adult birds arising from collisions with turbines. Figures in bold are below the threshold suggested above for a range turnover rate of 0.045. If the TR is 0.05 the threshold increases to 260 new adults.

Additional annual sub-adult deaths	Annual loss of potential adults (adjusted for age structure)	Fledging rate				
		0.30	0.35	0.40	0.45	0.50
0.0	0.00	**222.0**	259.0	296.0	333.0	370.0
0.2	0.11	**219.3**	256.3	293.3	330.3	367.3
0.3	0.16	**217.9**	254.9	291.9	328.9	365.9
0.4	0.22	**216.6**	253.6	290.6	327.6	364.6
0.5	0.27	**215.2**	252.2	289.2	326.2	363.2
0.6	0.33	**213.8**	250.8	287.8	324.8	361.8
0.7	0.38	**212.5**	249.5	286.5	323.5	360.5
0.8	0.44	**211.1**	248.1	285.1	322.1	359.1
0.9	0.49	**209.7**	246.7	283.7	320.7	357.7
1.0	0.55	**208.4**	245.4	282.4	319.4	356.4
1.1	0.60	**207.0**	244.0	281.0	318.0	355.0
1.2	0.66	**205.6**	242.6	279.6	316.6	353.6
1.3	0.71	**204.3**	**241.3**	278.3	315.3	352.3
1.4	0.76	**202.9**	**239.9**	276.9	313.9	350.9
1.5	0.82	**201.5**	**238.5**	275.5	312.5	349.5

Extract ends

Collision models

The Band (Band *et al*., 2007) Collision Risk Model (CRM) has received considerable attention, particularly its sensitivity to the range of values used in the calculations (e.g. Chamberlain *et al.,* 2005, 2006). It is clear that the avoidance rate is one of its largest problems, indeed Chamberlain *et al* (2006) suggested that "the value of the current model in estimating actual mortality rates is questionable until such time as species-specific and state-specific (i.e. different bird activities and behaviours under a range of conditions, for example breeding birds, recently fledged or moulting birds) avoidance probabilities can be better established." Whitfield (2008) made a significant contribution to the avoidance rate problem for golden eagles but it is important to reinforce one of his conclusions "it should be borne in mind that CRMs are crude and should not be 'over-interpreted'" Whitfield (2008) makes this point for two reasons. First there are the computational difficulties described above but also because one the inevitable predictions of the CRM, that collision mortality increases with flight activity, does not appear to be supported at many wind farms (de Lucas *et al* 2008). Indeed, based on reported bird mortality, predictions from the CRM appear to be large over-estimates. Either, the predictions are biased or considerable bird mortality is going unrecorded and/or unreported. Under-recording cannot be ruled out since there appears to be so little monitoring information in the public domain.

Less attention appears to have been paid to an important CRM assumption, namely that under scenario 2 of stage 1 of the Band model birds fly randomly inside a volume enclosing the wind farm rotors. If this is valid the occupation of one part of the risk volume must be independent of use elsewhere. This is incorrect for several reasons. Firstly, there is an inevitable spatial autocorrelation in usage and, secondly, birds use air space in different ways. For example, kestrel flight is much more 'space-filling' than golden eagle flight so might be expected to be at greater risk, an assumption supported by the findings of Whitfield and Madders (2006). The assumption of random use leads to some unexpected results. For example, assume that a species only uses the eastern part of a wind farm footprint that is assessed using a western and an eastern vantage point. Flights are only recorded from the eastern vantage point. The CRM calculations spread the use across the whole footprint. Now assume that, because of landscape concerns, the western part of the wind farm is removed from the scheme and only the eastern vantage point and its recorded flights are relevant. If collision calculations are undertaken for these two schemes the amount of bird activity remains the same (all recorded from the retained eastern vantage point) but the wind farm is 50% smaller. It might be expected that because the number of turbines has been reduced the risk to the species should be reduced. In fact, under the Band CRM, the risk increases. This is because flights are not random across the wind farm. While this might be obvious in this extreme, but real, example it highlights that fact that flights do not fill space randomly. This leads to the conclusion that the CRM might be better employed to determine the *relative* risk associated with each turbine. Chamberlain *et al* (2006) suggested previously that "the CRM may be useful for comparative purposes, but this is dependent on sound evidence that potential sites being compared can be assumed to have equal avoidance rates". This recommendation refers to comparisons between wind farm schemes rather than its use within a wind farm. Arguably the within-wind farm comparison is more robust than the between-wind farm comparison but it still depends on an equal avoidance rate for all turbines within a scheme. There is clear evidence, e.g. Smallwood and Thelander (2008), that some turbines kill more birds than others so even this assumption may be invalid. The difficulties arise from separating out increased mortality associated with a simple increase in activity and that caused by different avoidance rates. It is unclear how this difficulty can be resolved. However, it is clear that the CRM is based on faulty assumptions and unreliable parameter values.

It is also clear that it is dangerous to extrapolate from studies elsewhere. For example, Smallwood and Thelander (2008) estimated that there are 0.11 golden eagle collisions per MW per year. The Beinn an Tuirc wind farm is 30.36 MW, so their estimate predicts 3.3 collisions per year, or almost 30 collisions for its operational period. There is no evidence for any collisions at Beinn an Tuirc.

5. Avoidance, Displacement and Collision Risk interaction

Apart from the Beinn an Tuirc (Walker *et al* 2005) and Edinbane (Fielding and Haworth 2010b) wind farms there is little accessible information on the actual responses of golden eagles to wind farms, in particular how do they respond to loss of habitat? Nonetheless, the general pattern from Beinn an Tuirc, Beinn Ghlas, Edinbane and Ben Aketil is apparently some, but incomplete, displacement from the area around the turbines. This means that, as long as birds remain in the area, there will still be some risk of collision. The difficulty arises when trying to assess the magnitude of the reduction in activity around the turbines.

In the collision risk calculations an avoidance rate is applied at the end of the calculations. Some adjustment to calculations is needed because it is certain that the unadjusted collision estimates are far too high. Deriving the appropriate avoidance rates usually involves trying to compare observed and predicted levels of mortality. However, if a better understanding of the interaction between displacement and collisions is required it may help to split the avoidance rate to reflect two different biological processes. Currently, the late application of an avoidance rate in the calculations means that there is no reduction in use in the initial stages, i.e. there is no displacement. A simple shift in the application of the avoidance rate to start of the calculations by reducing the flying time, as a better approximation to the reality, produces the same result but there are important differences in how this relates to the behaviour of the birds.

It seems reasonably clear from the four wind farms described above that golden eagles avoid the region around wind turbines. Although not calculated here, it may be possible to arrive at an estimate for the likely percentage reduction using data from these four schemes. Next it becomes necessary to decide if eagles also avoid rotating blades at a micro scale, i.e. if they enter the wind farm area do they take late avoiding action to reduce the chance of flying into the blades? It seems, therefore, that a more biologically realistic collision risk calculation might need two separate avoidance rates. The first operates at the landscape or macro-scale and could be derived from operational wind farms given sufficient pre and post construction monitoring data. It is harder to see how the second value could be obtained other than by detailed recording of actual flights within wind farms. Ultimately, it might be argued that splitting the avoidance rate will not make any difference to the collision estimates. Indeed, introducing another imprecise parameter may degrade the calculation even more. However, partitioning out avoidance into macro- and micro-scale values may provide a better understanding of the processes that lead to eagle and other deaths at wind farms rather than producing an imprecise estimate that is frequently over-interpreted.

Beinn an Tuirc

Walker *et al* (2005) has detailed descriptions and analyses of the changes in flight patterns following construction. Following construction they only recorded one flight out of 811 in the wind farm and at collision altitude. Even then there was evidence of avoidance on a micro-scale in that it passed between the two discrete turbine clusters. They suggest that three of the four eagle flights over or through the windfarm were also associated with intruding eagles rather than foraging in the open spaces between turbines, i.e. there was 100% displacement of foraging birds.

Edinbane and Ben Aketil

Although there is strong evidence of displacement at Edinbane and Ben Aketil (Section 3) it is not complete as shown by some flight descriptions from Crane and Nellist[6] which describe flights close to turbines and apparent indifference to construction activities. In these examples, responses to other eagles do not seem to be the explanation.

"13.06 imm EA 10-100 slow sweeping flight across hill. Looked to be hunting and oblivious to turbines which it was very close to and constantly flying at height within turbine blade sweep range. Slow flight followed line of turbines N to S, between turbines and my VP, but much closer to turbines than me. ....

9.51 Ad EA being chased 10-100 N by 2 Raven across the west face below row of turbines, about 50m from the most northerly turbine it began rising up, still pestered by Raven, but remained <100. ....

10.56 Ad EA 10-100 gliding from E around N end of Beinn a Chearcaill, glided around west side of hill, flying over crags on west side, then north to Glac na Brothaig Airde and landed on top of crag 10.59, immediately gathering 3HC around it. (flight took the bird very close over 2 excavators working at northern end of road works.)

13.51 Ad EA spotted perched on Cnoc Cruinn a Bhraighe Bhuide. It remained motionless when a large explosion moved a serious amount of the W side of Cruachan Beinn a Chearcaill at 14.10."

There is a partial surrogate for the effects of wind farm avoidance in the effects of large scale plantation and natural forests on golden eagles. For example, as with wind farms, the afforestation of previously open habitats is thought to reduce golden eagle foraging opportunities with a consequential negative impact on prey caught in open country. The possible effects in Scotland were investigated by Whitfield *et al* (2001, 2007). This work is relevant to the effects of wind farms because both potentially represent a loss of foraging habitat, albeit with no increased direct mortality effects from trees.

Using 31 years of golden eagle breeding and forestry data Whitfield *et al* (2001, 2007) examined relationships between forest cover and eagle ecology at two scales: landscape and individual territory scales (equivalent to a cumulative assessment and a single wind farm impact assessment). Several territories were abandoned during the earliest phases of forest planting, but relatively few were apparently lost to later plantings. Territories with poorer breeding productivity appeared more vulnerable to abandonment than territories with better breeding productivity. At the landscape scale, temporal differences in breeding productivity were negatively related to the extent of forest cover, although productivity of individual territories showed no clear relationship with forest cover.

The loss of foraging habitat was assessed using the PAT model. Each pixel (0.25 ha), within the PAT output, has a predicted use. Thus, if closed canopy woodland covered 200 pixels, each of which has 0.1% predicted use, then it covers 200 x 0.1 = 20% of the predicted range use. Several territories

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[6] Crane and Nellist are the surveyors for the monitoring programme. They have monitored Skye's golden eagles for more than 25 years.

with < 5% increase in forest cover experienced reduced productivity; however, territories least constrained by neighbouring pairs of eagles showed an increase in productivity. This is illustrated by some of the more detailed results. For example, territories experiencing the greatest increases in forest cover showed a greater use of spatially-separated nest sites by occupying pairs. Hence, pairs that were less constrained by neighbours appeared to compensate for loss of open habitat by shifting their territory use, whereas pairs which were more constrained could not compensate for open habitat loss and suffered reduced productivity (and, probably in some cases, abandoned the territory).

Consequently, it was impossible to predict the probability of range loss from a simple range loss calculation and Whitfield *et al* (2007) concluded that simple guidelines based on the extent and locations of habitat loss are inadequate when predicting effects on large territorial raptors such as golden eagles. Consideration must also be given to the 'quality' of a territory or occupying pair, as well as the extent to which territory use is constrained by neighbouring pairs or other 'unsuitable habitat' which may have been affected by previous episodes of open habitat loss.

Summary of conclusions from the effects of forestry on golden eagle range occupation

- afforestation can have significant impacts, but
- the amount of forest needed to detect a change in productivity was very variable
- this is thought to be related to the amount of potential range plasticity – e.g. does the range have any potential to move or is it 'hemmed in' by neighbours, the sea, forests?

6. Mitigation measures for golden eagles at wind farms

Habitat management

As described in Section 3 there has been an attempted mitigation at the Beinn Ghlas wind farm in the form of sheep removal from a fenced area. It was hoped that this would result in more live prey such as grouse and hares. While there is no evidence that this mitigation has been successful in helping to get the range reoccupied it is unknown if it has succeeded in increasing the amount of live prey.

There was no habitat mitigation at the Edinbane or Ben Aketil wind farms on Skye.

Beinn an Tuirc is the only wind farm, with a significant golden eagle interest, that currently has a habitat mitigation scheme implemented (see Madders and Walker 2002). This was outlined in Section 3 and there was some evidence that it was successful at both increasing the amount of live prey and helping to keep the eagles away from the turbines. However, the time between the start of management and the first breeding success in 2008 was seven years. Even then this coincided with one of the best years of golden eagle productivity for many years across the west of Scotland. The tree clearance was intended to mitigate against land lost to the wind farm. However, Walker (2010) suggests there is now less open habitat available within the territory than there was before windfarm construction. Consequently supplementary feeding has been used, including during the successful year in 2008, the unsuccessful year in 2009 and the successful year in 2010.

The lesson from Beinn an Tuirc is that mitigation aimed at increasing live prey may not become effective until the wind farm is well established, i.e. there are unlikely to be immediate improvements in productivity. There is little doubt that much of upland Scotland has a degraded prey resource associated with long term excessive burning and grazing and it may be necessary to begin habitat management well in advance of the start of construction. For example, on the Eisgein estate on Lewis, burning ceased over all 42,000 acres of the estate in 2003, six years before the current successful wind farm application, and well in advance of any construction activity.

Haworth *et al* (2009) and Whitfield *et al* (2009) provided evidence that the commonly held belief in the importance of a narrow prey breadth is false. Their studies showed that food abundance and availability were important factors. This may be important when designing both habitat mitigation measures and supplementary feeding since it may be sufficient to increase the amount of prey across a suite of species rather than concentrating on a single species such as grouse or hares.

Finally, it is worth repeating the four conclusions from the Madders and Walker (2002) paper describing the mitigation at Beinn an Tuirc.

1. Impact assessments should take account of the cumulative effects of existing and proposed land uses.
2. Field work needs to be site specific and extensive, taking account of the full area occupied by the potentially impacted individuals and not just the area of the wind farm.
3. Mitigation must be large scale and provide additional prey in the appropriate places, i.e. both prey abundance and availability should increase.
4. Developments provide an opportunity to enhance previously degraded landscapes for benefit of a suite of species.

Supplementary feeding

The Beinn an Tuirc example suggests that mitigation via habitat management may not provide tangible benefits until a quarter to a third of the way through a wind farm's operational period. Therefore, unless habitat mitigation begins many years in advance of construction, it may be desirable, at least in the short term, to use a more active and immediate mitigation technique. Supplementary feeding is the active and regular addition of suitable prey, usually close to a nest. It is currently being used a diversionary aid to reduce red grouse predation by hen harriers. In the case of wind farms the aim is different, rather it is to mitigate against the loss of foraging habitat with a consequential reduction in prey.

In the short term supplementary feeding at the nest might appear to be an attractive option to alleviate prey shortages. Robb *et al* (2008) is the most recent review on the effects of supplementary feeding on bird populations. However, reflecting the number of papers in different groups, relatively few of the examples are birds of prey. There are many studies which show that the effects of supplementary feeding depend on the amount of natural prey. It is unclear where the balance is between insufficient natural prey to make supplementary feeding beneficial and sufficient natural prey so that supplementary feeding is not cost effective. Consequently, it is very difficult to see how, *a priori*, judgments can be made about the need and likely success for such actions.

Both Wiehn and Korpimäki (1997) and Dawson and Bortolotti (2002) showed experimentally that when American and Eurasion kestrels, respectively, were provided with additional food at the nest this did not translate into heavier offspring. Instead, in both cases, female parents reduced their foraging activities so that control and experimental offspring received the same amount of food. However, female parents were heavier in the experimental nests and spent longer at the nest leading, in the work of Wiehn and Korpimäki (1997), to more offspring. Similarly, Brommer *et al* (2004) found the same effect (heavier female parent and no effect on the offspring) of supplementary feeding of Ural owls *Strix* uralensis. In this example, the better female condition resulted in pairs breeding one week earlier in the next year. These experiments, and the earlier work of Dawson and Bortolotti (2000), point to the more significant effects of weather on offspring condition and survival.

The best evidence for positive benefits of supplementary feeding on an eagle comes from Spain and the work of Gonzáles, Margalida, Sanchez, Oria and others (for example Gonzáles *et al* 2006 and Margalida *et al* 2007). Margalida *et al* (2007) concluded that sibling competition and aggression was the main cause of nestling mortality and that supplementary feeding resulted in an increase in the fledging rates. Gonzáles *et al* (2006) document the magnitude of this effect with an increase in the fledging rates for adult pairs from 0.72 to 1.56 and a similar increase (0.53 to 1.57) for sub-adult pairs. They say that sibling aggression stopped completely in supplementary fed nests and that this made the removal of chicks, for fostering, both unnecessary and unethical. However, Ferrer and Penteriani (2007) are quite critical of these papers and the conclusions. They point out that the original purpose of the supplementary feeding was to replace prey that was lost following a cull of waterfowl (to avoid a botulism outbreak) and they question the level of sibling aggression quoted in the previous papers. Ferrer and Penteriani (2007) used classical population dynamics theory to show that despite the earlier claims, and even under the most sympathetic of assumptions, supplementary feeding had relatively little impact at the population level and was largely irrelevant to the population recovery now taking place. It is also important to recognise that these Spanish studies were concerned with reduced siblicide. In Scotland, it is more likely that the aim would be to fledge single birds from ranges that are currently, or historically, unproductive. The effectiveness of supplementary feeding for this purpose is unclear but evidence from the Irish golden eagle re-introduction, particularly with respect to the logistics, may be beneficial.

However, supplementary feeding is not a simple matter of placing a few rabbits on the ground. The effects of food shortages on golden eagles are complex and poorly understood, even in the absence of wind farms. Supplementary feeding needs to recognise the importance of timing of the food delivery. For example, if supplementary feeding only takes place following laying the earlier consequences of female malnutrition may mask the benefits arising from the extra food for nestlings. This is described in more detail in the Haworth *et al* (2009) and Whitfield *et al* (2009) studies. In the Inner and Outer Hebrides they found that the failure of breeding attempts occurs most frequently during incubation or with small young. A small number fail with young at more than 6 weeks old or shortly after fledging. Marquiss *et al* (1985) examined the factors associated with a reduction in breeding in SW Scotland and suggested that good breeding performance was associated with spring (pre-breeding) diet with the most productive pairs consuming more live prey (large birds) and less carrion. Tjernberg (1981) suggested that, whether breeding occurs or not, is probably determined by prey abundance early in spring just before eggs are laid.

In addition to timing, quantity of prey is also important. There is no single estimate of the amount of prey needed in a productive golden eagle range. McGahan (1967) working in south-central Montana, recorded that one pair of eagles brought an estimated 490 g of edible food mass per eagle per day to a nest during a 39-day period. Each eagle took an estimated 40-49 prey individuals over a 100-day period, with lagomorphs being the most important species (white-tailed jackrabbits *Lepus townsendii* and cottontails *Sylvilagus audubonii* and *S. nuttallii*). Brown and Watson (1964) estimated that a pair of eagles needed 174 kg of live prey and carrion per year. Brown (1969) expanded on this by suggesting that another 54 kg was required if an average of 0.8 young per year were reared plus an additional 43 kg for sub-adult birds using the territory intermittently. Takeuchi *et al* (2006) had a larger figure for the young bird. Using video recordings at nests they showed that there was temporal change in prey selection during nestling periods, but with similarities in later deliveries of snakes and in total prey weights (83.7–89.9 kg) delivered to successfully fledged broods. This is more similar to the figure that can be extrapolated from Collopy (1984) who gives a mean delivery of 1.42 kg per day (approximately 100 kg). However, these are delivery and not consumption figures so a lower figure is presumably adequate. Indeed, Collopy (1984) quotes a much lower figure of 0.885 kg per day given by Lockhart. Even after making allowances for the assumed 0.8 fledging, the adjusted figure for Brown and Watson (1964) remains lower at 67.5 kg. Similarly, Fevold and Craighead (1958) arrived at a larger figure for adult birds. They fed captive golden eagles mainly venison during autumn and winter. Extrapolating from their figures gives annual requirements of 112 kg (female) and 96 kg (male) or 208 kg for a pair. If the average range areas per pair quoted by Brown and Watson (1964) are used (4613 ha – 7273 ha) their food requirement estimates equate to only 0.06 and 0.04 kg prey ha^{-1} year^{-1}. In context of a wind farm, the loss of 3 km^2 of foraging habitat might equate to the loss of 15 kg prey year^{-1} (assuming no loss of prey hotspots, such as a rabbit warren). 15 kg of prey is equivalent to approximately 60 rabbits.

In the immediate future Haworth *et al* (2009), in the context of the general health of the Scottish population, suggested that some experimental work involving supplementary feeding and the creation of additional food resources, for example rabbit warrens, is warranted in selected areas. In the west of Scotland rabbits can be a particularly important food resource for eagles and although sometimes considered an alien and/or pest species conserving and enhancing rabbit populations may be crucial (Lees and Bell, 2008).

Because of the loss of open habitat at Beinn an Tuirc (Walker 2010) supplementary food has been provided during the pre-laying period. Food has been supplied as deer carrion and mountain hares have been released into the management area. Even though it is known that both have been used by the adult eagles there was no breeding in 2009. Natural Research Ltd are also currently in the middle of a food supplementation experiment in Morven. It is a designed experiment aimed to inform mitigation for wind farm developments in a region that already suffers from very poor productivity. During the experiment extra food is provided to five pairs for three years and the results will be compared with a control group of a further five. After three years the treatments are switched. The supplementary food consists of butchered deer carcasses and dead rabbits/hares which are provided during incubation and chick-rearing. The supply of food equates to about 1 kg per territory per week (1 kg prey is approximately four rabbits). The results from this experiment will obviously help to inform mitigation proposals for other schemes.

Conclusions and future work

It is clear from the above review that there is considerable uncertainty about certain aspects of the interaction between golden eagles and wind farms and the measures that can be taken to mitigate against any negative impacts. It is also clear that some of the techniques used to measure potential effects lack precision and are susceptible to over-interpretation.

Because there are currently so few operational wind farms that have the potential for significant effects on golden eagles the interpretation of the general effect of wind farms on golden eagles in Scotland can only be anecdotal. However, the local effects on individual wind farms are more certain.

There is clear evidence that, at one scale, golden eagles avoid wind turbines in Scotland, i.e. there is strong evidence for displacement leading to loss of foraging habitat and possible consequences for nest location choices. However, there is also evidence that, at a larger scale, wind farms may not be detrimental to golden eagles. For example, the Beinn an Tuirc range has remained occupied with two successful breeding year and the range closest to the Arnish Moor wind farm on Lewis has been re-occupied and has become a relatively productive range. Similarly, on Skye, the Ben Aketil and Edinbane wind farms appear to have displaced non-breeding birds but there is still considerable activity within the general area. However, the Beinn Ghlas range remains vacant and the predictions for the yet to be constructed Muaitheabhal wind farm on Lewis are the possible loss of two golden eagle ranges due to range abandonment.

It is also clear that there is relatively little information in the public domain. We are only aware of the range status adjacent to the Arnish Moor wind farm because of unrelated work by Robin Reid. Consequently, there is an urgent need for co-operative work across wind farm developments that will enable the robust meta-analyses that could inform assessments. In particular, more robust data are needed to measure post-construction impacts on golden eagle mortality and displacement/disturbance. Ideally, there would be continuing vantage point surveys that would enable pre- and post development comparisons, such as that undertaken at Edinbane (Fielding and Haworth 2010b). The work of Whitfield *et al* (2009) and Haworth *et al* (2009) examined some of the relationships between eagle prey and productivity and highlighted the importance of overall abundance rather than super-abundance of single prey species. Fielding *et al* (2009b) discussed the role of supplementary feeding in connection with the Irish re-introduction programme and highlighted some of the complexities that are described above. However, it is clear from some of the practices in Ireland[7] that supplementary feeding can be beneficial, albeit in combination with fostering. Consequently, it is important that general lessons are learned from current supplementary feeding exercises with respect to mitigation measures on golden eagle range use and productivity. It is important that all good practice is spread across the industry and failed methods described. Finally, continuous monitoring of the wider golden eagle population (e.g. Fielding and Haworth 2010b) is needed to determine the cumulative impacts of developments on the favourable conservation status of golden eagles.

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[7] http://www.goldeneagle.ie/news_viewnews.php?x=1&z=41&f=1&news_id=8&start=0

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