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A conservation framework for golden eagles: implications for their conservation and management in Scotland

(ROAME No. F05AC306)

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Summary

A conservation framework for golden eagles: implications for their conservation and management in Scotland

Commissioned Report No. 193 (ROAME No. F05AC306)
Contractor: D P Whitfield, A H Fielding, D R A McLeod and P F Haworth
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Background

The golden eagle *Aquila chrysaetos* is a large bird of prey (raptor), which typically occupies open upland landscapes and nests on cliffs or in large trees. Food consists of medium sized prey such as grouse, hares or rabbits, and carrion such as dead sheep and deer. Each pair can produce up to two young per year which, after leaving their parents' territory, disperse over wide areas until, after about four years, they settle on a breeding territory. Once occupying a territory, golden eagles are long-lived, with many adults living for up to twenty-five years.

An effective conservation strategy for uncommon and widely distributed species, such as the golden eagle, needs to have three elements:
a) species protection;
b) site protection; and
c) conservation in the wider countryside (i.e. outwith protected sites).

Species protection typically involves legislative provisions against human interference such as killing, capture or disturbance. In Scotland, the golden eagle is protected against killing and intentional or reckless disturbance by its listing on Schedule 1 of the Wildlife and Countryside Act (1981) as amended. The main focus for **site protection** of Scotland's golden eagles is through the UK and Scottish governments' commitments to the European Commission (EC) Wild Birds Directive (79/409/EEC) and the listing of the species under Annex 1 of the Directive. A total of eight Special Protection Areas (SPAs) have been classified in Scotland for golden eagles under the Birds Directive. Purely site-based approaches to large raptor conservation have limitations, however, and several studies indicate that these alone are unlikely to successfully conserve eagle populations. Hence, the main challenge for the successful conservation of the golden eagle lies in developing a strategic approach to **conservation in the wider countryside** which complements existing site and species protection measures.

Watson & Whitfield (2002) proposed **a conservation framework for the golden eagle** in Scotland with the overall aim to maintain the population in favourable conservation status by implementing effective site and species protection measures, and by adopting and implementing conservation policies which are targeted at known constraints across the species' current range. Essentially, the proposed conservation framework has two elements:

- Set targets for favourable conservation status based on criteria of abundance, demography and distribution, and assess whether these targets are being met; and
- Identify those constraints acting on the population(s), assess their regional influence on favourable conservation status, and use these assessments to implement policies targeted at influential constraints.

Favourable conservation status targets were set as follows (Whitfield *et al*., 2006):

- Nationally, at least 500 golden eagle territories should be occupied by pairs;
- Regionally, at least 66% of known (Highlands and Islands) or potential (south of the Highlands) territories should be occupied by pairs; and
- Demographic parameter values (i.e. production of young, pre-breeding survival and adult survival) should allow the maintenance of a stable or expanding population.

Tests as to whether these targets were being met were carried out based on Scottish Natural Heritage's Natural Heritage Zones (NHZs) as regional divisions (Figure 1) and using the results of the three national surveys of golden eagles undertaken in 1982, 1992 and 2003. This report provides an overview of the tests made to ascertain if the golden eagle population in Scotland was in favourable conservation status. More details of this work are given in several scientific papers published from 2003 onwards (and cited in the References).

Main findings

1. Occupying approximately 440 territories in 2003, the national golden eagle population of Scotland failed to meet the abundance target for favourable conservation status. Only three of sixteen regions, where eagles have occupied territories since 1982, were considered to be in favourable conservation status (Table 6). These are all in western areas: the Western Isles (Zone 3), the Western Seaboard (6), and Argyll West and Islands (14; Figures 1 and 7). A fourth region, the Northwest Seaboard (7), would have passed all tests for favourable conservation status if recent mergers of traditional golden eagle territories were taken account of.

2. The most serious failures to meet favourable conservation status tests were in NHZs in the central and eastern Highlands (the Central Highlands (10), Cairngorms Massif (11), Breadalbane and East Argyll (15), and North East Glens (12); Figures 1 and 7), where less than half of all known territories were occupied (Table 1). Based on the production of young golden eagles (Table 2), the populations in these regions should be expanding markedly, but instead they continue to decline (there was a loss of 15 occupied territories between 1992 and 2003, and 86 vacant territories by 2003; Table 1). This indicates, in the absence of any evidence for emigration, that survival of subadult and/or adult birds is low.

3. In two regions of western Scotland, the Western Highlands (8) and Lochaber (13), unfavourable status arose because of insufficient young birds being produced (Table 2), probably because of a shortage of live prey through a possible combination of heavy grazing by deer and sheep, and excessive burning of vegetation.

4. The evidence for impacts of a number of potential constraints on golden eagles in Scotland has been assessed rigorously. These include: topography, vegetation, land cover, geology, commercial forestry, unintentional human disturbance, wind farms, expansion of white-tailed eagles, persecution, and grazing by sheep and deer. Studies of the influence of different constraints on favourable conservation status, which have been published recently in several scientific papers and reports, are summarised. **Current evidence indicates that illegal persecution and low food availability in parts of western Scotland are the two main constraints on the Scottish golden eagle population**.

5. A number of lines of evidence indicated that illegal persecution of eagles, principally associated with grouse moor management in the central and eastern Highlands, is the most severe constraint on Scottish golden eagles. These lines of evidence, based on population modelling and analyses using a Geographical Information System (GIS) are as follows:

a) As carrion feeders, golden eagles are particularly vulnerable to poisoned bait. Records of the illegal use of poisoned baits were significantly associated with grouse moors; both nationally and within those regions where grouse moors predominated as a land-use (Figure 5). There was no evidence of a decline in records of poisoning on grouse moors between 1981 and 2000, even though poisoning incidents had declined in upland areas away from grouse moors;

b) Records of illegal persecution of golden eagles (including poisoning, trapping, shooting) were also more common in those regions where grouse moor management predominated;

c) There was no consistent or strong evidence of associations between territory vacancies and constraints other than persecution in these regions;

d) Persecution (assessed on the basis of the distribution of poisoning incidents) was associated with a lowering in the age of first breeding, a greater number of territory vacancies, and the use of territories by non-breeding immature eagles. The evidence indicates that persecution was reducing survival, constraining the distribution, and was probably creating ecological ‘traps’ by attracting dispersing immature eagles into areas of apparently suitable habitat that were unoccupied because previous residents had been killed. As well as affecting young birds from local nests, subadult (pre-breeding) survival in golden eagles from persecution-free areas would be reduced if they entered regions with persecution;

e) Observations of the age of birds and breeding success of Scottish golden eagles occupying territories in the 1982 and 1992 surveys were used to estimate population parameters (survival and productivity) in the different regions of Scotland. Age structure estimates of adult survival suggested that regions where persecution was most common suffered the greatest shortfalls in the numbers of adults. Regional differences in age structure associated with differences in the intensity of persecution (density of poisoning incidents) were used to adjust demographic estimates to simulate the absence of persecution.

In the absence of the estimated 3 – 5% decrease in national adult survival rates associated with persecution, population modelling suggested the Scottish golden eagle population would increase. Estimates that are not adjusted to simulate a ‘no-persecution’ scenario, however, suggested that the Scottish golden eagle population was vulnerable to decline. In golden eagles, a species which is naturally long-lived, even slight changes in adult and subadult survival rates can have major population consequences;

f) Population modelling based on breeding productivity and recent population trends indicated that too many birds were dying in those regions where grouse moor predominated, and this would not maintain the breeding population. For the Central Highlands (Zone 10) the Cairngorms Massif (11) and Breadalbane and East Argyll (15), population simulations using the 1992 and 2003 surveys indicate that survival rates must be lower than other zones, otherwise the golden eagle populations in these areas would be expected to expand due to high productivity (Table 2). In the Cairngorms Massif (11), for example, as few as 10% of subadult eagles may survive to breed. These low survival rates are inconsistent with the apparently considerable resources of food and space for eagles in these regions, but are consistent with eagles being killed, as suggested by other lines of evidence; and

g) Analysis of change in occupied territories between the 1992 and 2003 national surveys in relation to a number of potential constraints (including grazing, recreation, incidental disturbance and afforestation) found no strong evidence for the influence of any constraints other than persecution. The four NHZs (10, 11, 12 and 15) where persecution indices had not changed or increased were also those where grouse moor management is most common. Eagles in these areas showed a 21% decline (70 pairs (active territories) in 1992 down to 55 in 2003; Table 1). The only two regions with marked increases in occupied territories in 2003 (3 and 5; Table 1) were two of the three regions which experienced a decline in persecution indices. Occupied golden eagle territories therefore tended to decline where persecution was probably still influential, but to increase where persecution had probably declined. Overall, had there not been population increases in two regions where persecution had declined, the national Scottish population would have declined in 2003.

6. The highest national priority for the conservation and management of golden eagles in Scotland is to tackle persecution in those areas where it still persists. A secondary national priority for restorative management is to promote greater availability of live prey in parts of the western Highlands, potentially through changes in the management of deer and sheep. A number of studies have shown a positive link between the abundance of live prey and breeding success, although further research on the interactions between deer and sheep grazing and golden eagle ecology would be beneficial due to their complexity.

7. Golden eagles, and the constraints which appear to influence them, should continue to be monitored. Potential constraints which may be deserving of more attention in the future would include the potential decreased availability of sheep and deer carrion and the extensive culling of mountain hares on some Highland grouse moor estates.

Throughout its breeding range, from the arctic to northern Africa, the golden eagle has successfully adapted to a wide range of climatic conditions, but further attention to the potential implications of climate change for the Scottish population would be beneficial. Key enhancements to monitoring will be gained by surveillance of adult survival through fingerprinting of DNA from cast feathers and monitoring dispersing, pre-breeding birds using remote telemetry.

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FOREWORD

This report presents a highly important contribution to the conservation of golden eagles in Europe. A penetrating analysis of data from all golden eagle territories in Scotland has yielded a clear picture of the constraints on this bird. In particular, the sustained persecution of golden eagles in some areas and the consequences of heavy grazing pressure in the west are significant issues which must be addressed to allow golden eagles to attain favourable conservation status.

The excellent field data derived from three comprehensive national golden eagle surveys, conducted by members of the Scottish Raptor Study Groups, with support from SNH and RSPB, are the foundation for this work. Using a biogeographical division of Scotland into Natural Heritage Zones, the authors have identified specific constraints impacting on golden eagle populations in different parts of their range. The research has developed into a powerful and novel tool which should be applied to other rare and endangered birds. Modelling of future trends in populations under differing constraints points to clear priorities for conservation action.

The authors have produced an important piece of research, founded on robust data sources and a rigorous scientific approach. The resulting analysis has powerful messages for the long-term conservation of one of Scotland's most emblematic species. The development of objective tests for favourable conservation status of Scotland's golden eagles, above and beyond the long-established use of protected areas as a conservation tool, is ground-breaking work that has clear relevance for other wide-ranging species in Scotland and elsewhere in Europe.

Undoubtedly the highest priority of all is the need to address the illegal persecution which continues to affect golden eagle populations in the eastern and southern parts of the species' Scottish range. There can be no more urgent task than to eliminate this blight on the population of this majestic bird which, perhaps more than any other creature, is valued as a symbol of wild Scotland

Dr Jeff Watson

Formerly Director of North Areas for Scottish Natural Heritage, and author of *The Golden Eagle*, published by T & AD Poyser in 1997.

Publisher's note: Shortly after writing this foreword Jeff died following a long battle with cancer. That Jeff was determined to contribute to this report, despite his illness, speaks volumes for his concern for the plight of the golden eagle.

CONTENTS

1. INTRODUCTION

1.1 Biology of the golden eagle

Throughout its extensive northern hemisphere breeding range the golden eagle *Aquila chrysaetos* is one of the largest avian predators and occupies nearly all mountain landscapes where there are suitable trees or cliffs for nesting and open areas for hunting. Food is principally medium-sized (typically in size class 0.5 kg – 3 kg) birds such as grouse and, in particular, similar sized mammals such as lagomorphs (hares/rabbits), supplemented by scavenging carcasses of dead ungulates (McGrady, 1997; Watson, 1997).

Golden eagles avoid intensively cultivated agricultural areas and areas heavily populated by people. In Scotland, they are virtually restricted to open, upland landscapes: most birds nest on cliffs and feed on prey caught mainly in montane habitats, open moorland and blanket bog plant communities, and on carcasses of sheep and red deer *Cervus elaphus* (Watson, 1997). In some other countries, golden eagle ranges can contain considerable areas of woodland, where nests are more usually in large old trees, and prey is predominantly woodland or woodland edge species (e.g. Tjernberg, 1981; Takeuchi *et al*., 2006). Breeding densities are considerably lower where woodland predominates, however, but breeding success tends to be higher in these habitats (Tjernberg, 1985; McGrady, 1997; Watson, 1997).

Like other large raptors, golden eagles are long-lived, show delayed maturity and have low annual reproductive rates (e.g. Real & Mañosa, 1997; Ferrer, 2001), with a proportion of territorial pairs often not breeding every year and two fledglings being the typical maximum annual productivity for a pair (see reviews by McGrady, 1997; Watson, 1997; Whitfield, 2000). After fledging, juvenile eagles spend a variable period of continued dependency on their parents' territory, extending from weeks to several months, before dispersing up to several hundred kilometres. Although the period between fledging and settling to breed is one of the poorest known phases of the golden eagle life cycle, it is likely that an initial exploratory dispersal phase is followed by young birds settling in one or a number of areas rich in food away from or at the periphery of defended territories. As maturity approaches, subadult eagles probably move progressively closer to their natal area and attempt to obtain a territory: typically golden eagles first breed at 4-5 years old and at a site close to their natal area. Consequently the natal dispersal distance (between natal and breeding sites) may be relatively short compared to the dispersal movements in the years preceding breeding.

Probably around 40% of birds which fledge reach breeding age, although survival rates are likely to be variable both between populations and across years. Breeding pairs occupy a more-or-less exclusive home range which is defended as a territory against intruders and is occupied year-round in Scotland (in more northerly countries, territories are abandoned at the end of the breeding season and birds migrate south)[1]. Once birds reach an age when they hold a territory their survival rates are higher with over 90% annual survival probably not atypical, so some adults may live for decades. As for all long-lived species with low annual productivity and delayed maturity, populations of golden eagles are most influenced by changes in subadult survival and, especially, adult

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[1] In this report we use the terms 'territory' and 'range' (or home range) interchangeably.

survival, although even a population with high survival rates can experience difficulties if reproductive output is insufficient to meet losses through mortality.

Further information on golden eagle biology may be found in McGrady (1997), Watson (1997), Whitfield (2000) and Whitfield *et al.* (2004 a, b)

1.2 History of the golden eagle in Britain

Before the mid-18th century the range of the golden eagle in Britain and Ireland extended beyond its Scottish heartland into northern England as far south as Derbyshire, into the mountains of north Wales, and into much of western Ireland (Holloway, 1996). The 19th and early 20th centuries saw intense persecution of birds of prey in Britain, and coincided with an increase in the use of upland areas for sport shooting, especially for red grouse *Lagopus lagopus scoticus*. Because raptors were perceived as competitors with man for game species, and also because predators like eagles were considered threats to domestic stock such as sheep, large numbers of birds of prey were killed during this time (Brown, 1976). This led to substantial range contractions for most raptors and the extinction of several species (D'Arcy, 1999; UK Raptor Working Group, 2000).

The low point of the golden eagle population in the British Isles was probably reached in the years preceding World War II. By then, the golden eagle had been extinct as a breeding species in Ireland for decades (O'Toole *et al*., 2002) and in Britain perhaps as few as 100-150 pairs were confined to the remoter parts of the Scottish Highlands and Islands, in areas such as deer forests, where they were not seen as a threat (Love, 1983; Watson, 1997). Numbers gradually recovered in Britain following statutory protection of golden eagles under the Protection of Birds Act in 1954 (Watson, 1997) and the first national surveys in Britain conducted in 1982 and repeated in 1992 revealed similar results of about 420 occupied territories, with only one or two pairs in England and the remainder in Scotland (Dennis *et al*., 1984; Green, 1996). The results of the most recent national survey, in 2003, which included a re-analysis of the 1992 survey using a slightly modified definition of occupancy, suggested continued stability of the British population, at about 440 occupied territories (Eaton *et al*., 2007; Whitfield *et al*., 2007b). Golden eagles have recently been reintroduced to Ireland (O'Toole *et al*., 2002) and reintroduced birds laid eggs for the first time in 2005 and successfully fledged a chick in 2007 (L. O'Toole, pers. comm.).

1.3 Conservation of the golden eagle in Scotland

An effective conservation strategy for uncommon and widely distributed species, such as the golden eagle, needs to have three elements: species protection, site protection and conservation in the wider environment (i.e. away from protected sites) (Watson & Whitfield, 2002). Species protection usually involves legislative provisions against human interference such as killing, capture or disturbance. In Scotland, the golden eagle is protected against killing and intentional or reckless disturbance by its listing on Schedule 1 of the Wildlife and Countryside Act (1981) as amended. Site protection is also usually backed by legislation and can include designating relatively large areas or small sites (around nest locations, for example), which are subject to additional levels of protection over and above those available to non-designated areas. In Scotland the main focus on site protection of golden eagles is through the UK government commitment to the European Union (EU) Wild Birds Directive (79/409/EEC; golden eagle is listed on Annex 1 of the Directive as a species considered vulnerable in Europe) and the Habitats and Species Directive (92/43/EEC).

These Directives require (Birds Directive: Article 4.1) that “Member States shall classify in particular the most suitable territories in number and size as special protection areas for the conservation of [Annex 1 and regularly occurring migratory] species”; and that (Habitats Directive: Article 6.2) “Member States shall take appropriate steps to avoid, in the special areas of conservation, the deterioration of natural habitats and the habitats of species as well as disturbance of the species for which the areas have been designated, in so far as such disturbance could be significant in relation to the objectives of this Directive”. Several Special Protection Areas (SPAs) have been classified in Scotland for golden eagles (Watson & Whitfield, 2002), but purely site-based approaches to large raptor conservation have limitations and are unlikely to have successful outcomes (Pienkowski, 1991; Real & Mañosa, 1997; Watson & Whitfield, 2002). It is also becoming increasingly evident that conservation strategies based solely on protection of breeding pairs may also have limited effectiveness (Ferrer, 1993; Real & Mañosa, 1997; Penteriani *et al*., 2005a, b). Indeed, the Birds Directive stipulates that “outside these protection areas, Member States shall also strive to avoid pollution or deterioration of habitats” (Birds Directive: second part of Article 4.4). Hence, the main challenge to successful conservation of wide-ranging uncommon species such as the golden eagle lies in developing a strategic approach to conservation in the wider environment which complements existing site and species protection measures.

1.4 A conservation framework for the golden eagle

In response to the challenge of implementing effective conservation for golden eagles in the wider environment, Watson & Whitfield (2002) proposed a conservation framework for the golden eagle in Scotland with the overall aim to maintain the population in favourable conservation status by implementing effective site and species protection measures, and by adopting and implementing conservation policies that are targeted at known constraints across the species’ current range. Geographical targeting of conservation policies was considered possible due to relatively good biological information on eagle numbers, range and breeding success, good understanding of current constraints affecting eagle populations, and the existence of a geographical or zonal framework that was able to accommodate information on population 'conservation status' and land use 'constraints'.

Watson & Whitfield (2002) made a first attempt at defining the criteria for favourable conservation status and provided an initial outline of likely constraints and their potential regional importance. Emphasis was placed on the necessity for more rigorous analyses to serve as a foundation for the framework and a number of analyses have been undertaken subsequently.

Several publications describe the results of studies resulting directly or indirectly from the golden eagle conservation framework. These include: Whitfield *et al*. (2001, 2002, 2003, 2004a, b, 2006, 2007a, b), Fielding *et al*. (2003a, b, c, 2006), McLeod *et al*. (2002a, b), Watson *et al*. (2003), Haworth *et al*. (2006) and McGrady *et al*. (2003, 2004).

The present report provides a summary of the methods and results to date, discusses the implications of these studies for golden eagle conservation, and identifies further research requirements.

2. REPORT OBJECTIVES AND FORMAT

The objectives of this report are as follows:

1. To provide an outline of the methods used to develop the golden eagle framework analyses.
2. To provide details of results to date.
3. To discuss the conservation and management implications for golden eagles in Scotland
4. To identify opportunities and/or requirements for further work to support the framework.

The framework has two principal elements:

- Setting and testing of favourable conservation status targets, and;
- Identifying and assessing the influence of constraints.

Many of the analyses associated with the golden eagle framework have used a common set of analytical tools and similar methods, notably a Geographic Information System (GIS). To avoid undue repetition when describing methods across several sections in this report, we have placed a description of the principal methods employed in Annex 1 (General Methods) and many of the analyses originally presented by Fielding *et al*. (2003a) have also been included as Annexes.

3. FAVOURABLE CONSERVATION STATUS TARGETS: CRITERIA

An overarching concept of ‘favourable conservation status’ was introduced by Watson & Whitfield (2002). This concept was based on the principles set out in the Wild Birds Directive (79/409/EEC) and in particular the Habitats and Species Directive (92/43/EEC) which indicates that “conservation status of a species means the sum of the influences acting on the species concerned that may affect the long-term distribution and abundance of its populations” and that “the conservation status will be taken as ‘favourable’ when:

- Population dynamics data on the species concerned indicate that it is maintaining itself on a long-term basis as a viable component of its natural habitats, and
- The natural range of the species is neither being reduced nor is likely to be reduced for the foreseeable future, and
- There is, and will probably continue to be, a sufficiently large habitat to maintain its populations on a long-term basis.”

Watson & Whitfield (2002) proposed three criteria to assess favourable conservation status for golden eagles: the number of occupied territories, breeding performance, and the proportion of suitable habitat which is occupied. Essentially these describe three ecological features of a population: abundance, demography and distribution.

Breeding performance was suggested as a demographic measure for a conservation status target by Whitfield & Watson (2002) due to the availability of data for a large number of Scottish golden eagle territories. As pointed out by Whitfield *et al.* (2004a, b), however, breeding performance is much less influential in large raptor population dynamics than subadult survival and, especially, adult survival. The problem is that measures of these more influential parameters are difficult to obtain and monitor, especially across many territories and regions as would be required under a national conservation framework. Golden eagles acquire ‘adult’ plumage around four years old and can usually be distinguished from younger subadults on plumage and moult (Bloom & Clark, 2001). Birds occupying territories can be aged as subadult and adult, therefore, and ageing of birds can be undertaken across several territories and years. The proportion of breeding pairs which are non-adult can reflect age of first breeding and so can provide an index of population status (a higher turnover or a shortage of breeding adults is more likely to lead to younger birds being recruited earlier in to the breeding population, for example) (e.g. Newton, 1979; Balbontín *et al*., 2003; Whitfield *et al*., 2004a, b). Hence, Whitfield *et al.* (2004a) suggested an additional criterion for favourable status should be that in at least 95% of breeding pairs both partners should be adult (birds at least 4 years old) as a crude surrogate of adult survival rate when there was a risk of decline.

Watson & Whitfield (2002) acknowledged both the need for the conservation framework to be flexible and responsive to new information, and that the analyses underlying the favourable conservation status criteria were preliminary. Whitfield *et al.* (2006) subsequently provided further refinements to the criteria, and followed Watson & Whitfield (2002) in basing regional considerations on the Natural Heritage Zone (NHZ) approach developed by Scottish Natural Heritage (SNH, 1998, 2000: Fig. 1), as follows:

- **Nationally at least 500 territories should be occupied by pairs**. By proposing a criterion for a national target of 450-500 pairs Watson & Whitfield (2002) effectively set a target at the lower limit of 450 pairs. This target was set on the basis of contemporary population levels and, in a small minority of cases, an assessment of likely suitable though unoccupied habitat (Watson & Whitfield, 2002). It is possible that eagle populations may have been higher historically, or that habitat currently considered unsuitable, could be occupied in the future. Thus, it was a pragmatic and conservative approach contemporary with the EU Directives. It was apparent (Whitfield *et al.*, 2006), however, that on this basis the national population could potentially be in favourable conservation status while some regions were in unfavourable conservation status, and this may weaken management impetus to meet regional targets. Hence, to remove this possibility Whitfield *et al.* (2006) suggested that 500 occupied territories was a more appropriate national target i.e. the upper limit proposed by Watson & Whitfield (2002). A result of this recommendation, however, is that regional abundance targets (66% occupation rate) could be met but the national target could be failed, but this is deemed to be a more acceptable outcome, since the priority should be on regional targets.
- **Regionally, at least 66% of known (Highlands and Islands) or potential (south of the Highlands) territories should be occupied by pairs**. In 2003 there were 696 known territories with 687 of these in the Highlands and Islands (see next section). There were few known territories south of the Highlands (zones 16, 19, 20; Fig.1), largely because there is a longer history of golden

eagles being absent from these regions (Holloway, 1996) although several authors have concluded that there is potentially suitable habitat for additional pairs (e.g. Newton, 1994; Green, 1996; Watson, 1997; Fielding *et al.*, 2003a). Based on the extent of suitable habitat (Watson & Whitfield, 2002; Fielding *et al.*, 2006; Annex 5) Whitfield *et al.* (2006) assumed that there is capacity for an additional 20 golden eagle territories in these regions (1, 2, 8 and 9 territories in zones 16, 17, 19 and 20 respectively: see also Fielding *et al.*, 2003a), giving a conservative total of 716 known and potential territories in Scotland. From this total, to obtain a national target of 450 or 500 occupied territories required an occupation rate of 62.9% or 69.8% respectively. Taking the mid-point of these values (66%) gave an occupancy rate which also implicitly incorporated distribution targets by being applied regionally, and thus allowed the vague distribution target of Watson & Whitfield (2002: “substantially all of the apparently suitable habitat is occupied”) to be dispensed with.

- **Demographic parameter values should allow the maintenance of a stable or expanding population**. Following Whitfield *et al.* (2004b), where adult annual survival rate was estimated at 94.2% and 95.8% in the absence of persecution, Whitfield *et al.* (2006) adopted an **annual adult survival of 95.12%** as the lower limit for a favourable conservation status classification. This equates to an expected 20 years of territory occupation by an adult. Hunt (2002) estimated that at a Californian wind farm, the subadult survival rate of golden eagles was around 40% (to age 5) but could be as high as 61% in the absence of wind turbine strike mortality. Since the 40% survival was associated with additional wind farm mortality it was unreasonable to adopt a lower figure. Adopting a higher rate would have a marked effect on conclusions (tending to make it more difficult for regions to pass demographic tests) and did not appear to be justified given the apparent differences in food availability between California and Scotland (Hunt 2002, Watson 1997). Consequently, Whitfield *et al.* (2006) conservatively adopted **a minimum acceptable rate for subadult survival of 40%** (first four years of life) which equates to an annual survival rate of 0.795. Under these survival rates an **average reproductive rate of about 0.28 fledglings per pair per year** is required to maintain a stable population. It follows, however, that if these parameter values varied regionally then lower rates in one parameter could be compensated for, to a degree, by higher rates in another parameter. Due to survival rates having a greater influence on population dynamics of eagles than reproductive rates, a reduction in survival rate, for example, requires a disproportionately large compensatory increase in fledging rate.

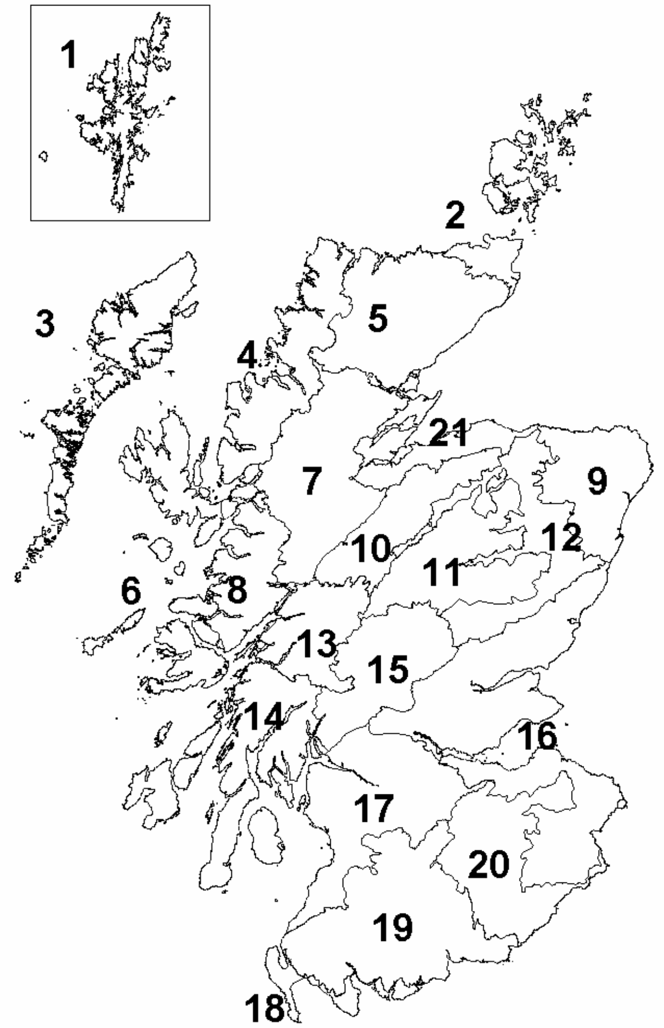

Fig. 1. Biogeographic zones of Scotland, termed Natural Heritage Zones (NHZs), developed by Scottish Natural Heritage (SNH, 1998, 2000). 1 = Shetland, 2 = North Caithness and Orkney, 3 = Western Isles, 4 = North West Seaboard, 5 = The Peatlands of Caithness and Sutherland, 6 = Western Seaboard, 7 = Northern 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.

4. FAVOURABLE CONSERVATION STATUS TARGETS: TESTS

4.1 Methods

To test whether Scottish golden eagle populations were in favourable conservation status Whitfield *et al*. (2006) entered the location of all occupied and vacant territories from the national surveys into the GIS as territory centres and eyrie locations (Annex 1).

SNH has identified 21 NHZs that reflect the variation in biological and landscape qualities across Scotland (SNH, 1998, 2000; Fig.1). The bulk of the golden eagle population is found within 12 of these zones and in four others, away from the Highlands, eagles have occurred in small numbers in the recent past but were probably more numerous historically and could recover in the future. NHZ boundaries were created as an additional layer in the GIS. Whitfield *et al.* (2006) assigned territories to a NHZ based on the location of the majority of predicted range use, taken as simple Thiessen polygons around territory centres using Dirichlet tessellation with a maximum ranging distance of 6 km in the absence of territorial neighbours (McGrady *et al*., 1997, 2002; McLeod *et al*., 2002a, b).

Level 1 test

Whitfield *et al*. (2006) used a set of layered tests which assessed regional measures against favourable conservation status targets. The Level 1 test considered whether territory occupancy was favourable, so that if < 66% of known (Highlands and Islands) or potential (south of the Highlands) territories were occupied in 2003, the NHZ was in an unfavourable conservation status irrespective of other criteria.

Level 2 test

NHZ data were then subjected to a Level 2 test using demographic parameters to judge whether the current regional population was likely to remain stable, expand (favourable conservation status), or decline (unfavourable status). This was based on three key parameters: adult survival, subadult survival and reproductive output (Whitfield *et al*., 2004b). Of these, only the latter could be measured with any confidence. However, using the Golden Eagle Population Model (GEPM) (O'Toole *et al*., 2002; Fielding *et al*., 2003a; Whitfield *et al*., 2004b)[2], it was possible to identify the combinations of adult and subadult survival rates which predicted stability or expansion for any measure of productivity. Hence, if acceptable lower values were set for annual adult and subadult survival (95% and 40% respectively, see above), it was possible to assess whether a NHZ was in favourable or unfavourable status with respect to reproductive output. Whitfield *et al*. (2006) examined this in two parts, by setting a Level 2a test utilising a mean fledging rate from 1982, 1992 and 2003, and a Level 2b test utilising the most recent fledging rate available (2003). Overall fledging rates in 2003 appeared to be below average, with 1982 especially good but 1992 poor (Green, 1996; Watson *et al*., 2003; Eaton *et al*., 2007): mean fledging rates from the three national census years gave a good approximation of long-term fledging rates across the same period for over 100 territories for which productivity data were available for 20 years (Fielding *et al*., unpublished; Annex 2).

The GEPM was run for each NHZ with a starting population set at the 2003 level, and with a capped population set at the same number of pairs as there were known territories. The output was a mean predicted number of occupied territories after 21- 30 years taken from means of 100 replicates. Further details of the GEPM are given elsewhere (O'Toole *et al*., 2002; Fielding *et al*., 2003a; Whitfield *et al*., 2004b, Annex 6). A NHZ was deemed to fail a Level 2 test if the GEPM did not predict stability or increase for the given fledging rates.

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[2] For full details of the GEPM, see O'Toole *et al*. (2002); a summary is given in Annex 6.

Level 3 test

As a Level 3 test, Whitfield *et al*. (2006) examined the predicted population projections from the GEPM against the observed trends in the number of occupied territories from the three national censuses (1982, 1992 and 2003). If the GEPM predicted stability or increase for the given fledging rates under the Level 2 tests but the observed population trend failed to match predictions then the survival limits employed in the Level 2 test were probably not being met. Hence, for example, if stability or increase was predicted but decline was observed, then a NHZ was deemed to have failed the Level 3 test. Whitfield *et al*. (2006) also used the observed proportions of non-adult pairs occupying territories as an additional means of assessing population status (e.g. Balbontín *et al*., 2003; Whitfield *et al*., 2004a, b).

Level 2 and 3 tests assumed that NHZ populations were closed and there was no net immigration or emigration, which is unlikely to be realistic (Whitfield *et al*., 2004b), but regional information on eagle dispersal in Scotland was absent and so immigration and emigration could not be formally incorporated. 'Source' and 'sink' population processes could not, therefore, be readily recognised. Golden eagles appear to show strong natal philopatry and limited breeding dispersal (e.g. Steenhof *et al*., 1984; Haller, 1982, 1994; Watson, 1997; Grant & McGrady, 1999), however, and so geographical range expansion is likely to be conservative, as in several other raptors (e.g. Lensink, 1997; Kenward *et al*., 2001). Consequently, it was probably reasonable to assume that a regional population would probably only be a net exporter of recruits if demography allowed and if regional carrying capacity (approximated by a high occupancy level) was met or approached and, conversely, a region would probably be a net importer of recruits when demography was inadequate for 'self-sufficiency' and vacant territories existed. The regional analyses of reproduction and survival by Whitfield *et al*. (2006) may not necessarily match population trend, therefore, but when a match was not evident it may have indicated potential immigration or emigration, or that assumed survival levels were incorrect. Information on regional territory occupation levels was then used to discriminate between the two alternatives.

In essence, therefore, the Level 3 test examined whether survival rates appeared to match the assumed 'acceptable' levels, and so was a test of regionally favourable survival rates, to complement the Level 2 tests of favourable breeding productivity. But because survival rates within most zones were not known but assumed, a deviation from expectations of population trend with the given demographic rates may have indicated that immigration or emigration had occurred rather than deviation from the minimum acceptable survival rates. To discern between these alternative explanations of deviation from expectations (and hence, implicate either deviation from survival expectation or immigration/emigration), additional information was employed on, for example, the availability of vacant territories within a zone, on the reasonable assumption that immigration would primarily be possible if, within a zone, the availability of vacant territorial opportunities could not be filled by locally-generated recruits (based on predictions from that zone's demographics), and that emigration would only occur if there was an excess of predicted local recruits over available vacant territorial opportunities.

For example, if the GEPM had indicated that observed reproductive rates in combination with the accepted survival rates predicted that a regional population should be expanding, but it was declining, then this would indicate a shortage of breeding adult recruits within the zone. This, in turn, could mean that the shortage of adults was because either that assumed ('acceptable') survival rates were too low (i.e. low recruitment was due to more eagles dying than was assumed to be acceptable) or that sufficient eagles were produced within the zone but that they were emigrating to other zones. In this scenario, if there were a persistently large number of unoccupied territories, then it would strongly infer low survival rates as the explanation of the disparity because the unoccupied territories should be occupied before emigration would occur, due to natal philopatry. Hence low survival rates, and thus failure of the Level 3 test, would be the reasonable conclusion.

Territory mergers

Several observers during the 2003 national census noted that vacant territories had been 'amalgamated' within occupied (active) territories. In other words, a former territory had been incorporated within a remaining territory by range expansion. As vacant territories which have been amalgamated within occupied territories may be more difficult for new pairs to re-occupy, Whitfield *et al.* (2006) considered it appropriate to examine the extent to which merger had occurred as it may influence the capacity for expansion of the breeding population. The proportion of known territories which are 'occupied' post-merger hence may also give a more realistic estimate of the extent of potential suitable breeding habitat which was occupied (noting that this was a favourable conservation status criterion of Watson & Whitfield (2002)).

It was apparent however that not all census observers had noted the occurrence of territory amalgamation. To obtain a national overview, Whitfield *et al.* (2006) compared a prediction of territorial boundaries (Thiessen polygons) for all known territories within the GIS with predicted territorial boundaries based only on occupied territories active in 2003. Where at least 80% of a vacant territory was overlapped by an occupied territory, Whitfield *et al.* (2006) considered the vacant territory to have been amalgamated within an active territory. This exercise incorporated all those amalgamations noted by field observers. If vacant ranges which had been amalgamated were assumed to be 'lost', this allowed Whitfield *et al.* (2006) to estimate the proportions of known territories which remained vacant post-merger.

4.2 Results

National target for occupancy

In 2003, 443 territories were occupied by pairs (Table 1), so the favourable conservation status target of 500 pairs for the national population was not met.

Level 1 test: Regional targets for occupancy, distribution and abundance

The number of occupied territories in NHZs was tested at Level 1 against target levels of 66% occupancy. Twelve NHZ were tested against numbers of known territories and seven failed (Table 1). For two NHZ (zones 4 and 5) failure was only two or three territories below target but for five NHZ the occupation level was substantially below the target, with vacancy shortfalls of 33 - 60% of known territories, depending on NHZ (Table 1). Whitfield *et al.* (2006) did not apply this type of test to the Southern Uplands and Border Hills NHZs (19 and 20) where less than 10 territories were known. However, on the potential number of territories expected from the extent of suitable habitat

(Methods), these NHZ would also fail the Level 1 test (Table 1). Whitfield *et al.* (2006) also did not test other NHZ where the numbers of potential territories were negligible (zone 2: 1 pair in 1982, none in 1992 and 2003; zone 16: no pairs in 1982 and 1992, 1 pair in 2003).

Level 2 and 3 tests: Regional demographic targets

Results from the GEPM for the Level 2 and 3 tests are given in Annex 3. Six of the eleven NHZs which were tested passed all demographic tests (Table 2). Three NHZs (zones 10, 11 and 15) passed the Level 2 reproductive output test but failed the Level 3 survival test. Based on fledging rates and assumed survival rates, all three of these regional populations (Central Highlands, Cairngorms Massif, Breadalbane & East Argyll) should be expanding rapidly but trends over the recent national surveys illustrated stability or continued decline, giving substantial numbers of vacant territories (Tables 1 and 2). Hence survival rates were lower (or emigration rates higher) than required for population stability or expansion and Level 3 tests were failed. Given the high vacancy levels of (apparently) productive territories in these regions, and no evidence of immigration in neighbouring zones, it was highly likely that poor survival, rather than emigration, was the cause of test failures.

Table 1. Results of the Level 1 tests for conservation status with respect to observed territory occupation in 2003 against target occupation based on 66% occupation of known (or potential: in brackets) territories. NHZs that failed the Level 1 test for known territories are in bold. The 'active – target' column gives the number of 2003 occupied territories with respect to the target. For NHZ that failed the test the 'shortfall' column gives the percentage of vacant known territories that would require to be occupied for the NHZ to pass the test. - = value not considered, NT = not tested. (From Whitfield et al., 2006).

NHZ	NHZ name	Territories					
		Known	Active 1992	Active 2003	Target	Active - target	Short-fall (% vacant)
3	Western Isles	93	62	81	61	+20	-
4	**North West Seaboard**	**71**	**45**	**46**	**48**	**-2**	**6**
5	**Peatlands of Caithness & Sutherland**	**31**	**13**	**18**	**21**	**-3**	**21**
6	Western Seaboard	98	74	74	66	+8	-
7	**Northern Highlands**	**90**	**45**	**43**	**60**	**-17**	**37**
8	Western Highlands	67	54	51	45	+6	-
10	**Central Highlands**	**26**	**12**	**12**	**17**	**-5**	**39**
11	**Cairngorms Massif**	**71**	**32**	**28**	**48**	**-20**	**46**
12	**North East Glens**	**17**	**6**	**3**	**11**	**-8**	**60**
13	Lochaber	36	28	25	24	+1	-
14	Argyll West & Islands	59	44	44	40	+4	-
15	**Breadalbane and East Argyll**	**27**	**20**	**12**	**18**	**-6**	**41**
19	**Western Southern Uplands & Inner Solway**	**4 (12)**	**3**	**2**	**8**	**-6**	**-**
20	**Border Hills**	**4 (13)**	**1**	**3**	**9**	**-6**	**-**

The Western Highlands (zone 8) failed both Level 2a and 2b tests as both mean and 2003 fledging rates were very low. Since there was little evidence of a decline across the three censuses (Table 1 and 51 pairs in 1982), this suggested that recruitment in this NHZ must be supplemented by immigrants from other NHZ or that survival was particularly high. However, the percentage of pairs in which both partners were adult in 2003 was 84.3% which suggests low adult survival compared to other NHZs (national value was about 90%; see also Whitfield *et al*., 2004b). The most likely source of immigrants was the neighbouring NHZ 6 (Western Seaboard: Fig. 1) where occupancy has been high for decades and all demographic tests were passed (Tables 1 and 2). The likelihood of immigration into NHZ 8 confounded a judgement on whether the Western Highlands passed the Level 3 test. Lochaber (zone 13) passed the Level 2a test but failed the Level 2b test. In this NHZ there was evidence for a recent decline in productivity and a small but long term decline in occupancy (Table 1 and 30 pairs in 1982). The percentage of pairs which were adult in 2003 was 84.0%, compared with 100% in 1982 and 1992, which was a further indication of recent difficulties. On balance, Whitfield *et al.* (2006) considered that Lochaber should be classed as unfavourable in status.

Table 2. Results of the Level 2 and 3 tests for conservation status with respect to demography. Fledging rate = number of fledged young per occupied territory per year. Mean fledging rate was calculated from 1982, 1992 and 2003 censuses. The Level 2a test used the mean fledging rate, and the Level 2b test used the 2003 fledging rate. NHZs that failed at least one test are in bold. NT = not tested. - = no data (typically because no territories were occupied), ? = a judgement on the test result was not clear (see text and Whitfield et al. (2006) for details). (From Whitfield et al., 2006).

NHZ	NHZ name	Fledging rate		Level tests		
		Mean	2003	2a	2b	3
3	Western Isles	0.33	0.35	Pass	Pass	Pass
4	North West Seaboard	0.39	0.33	Pass	Pass	Pass
5	Peatlands of Caithness & Sutherland	0.32	0.39	Pass	Pass	Pass
6	Western Seaboard	0.44	0.46	Pass	Pass	Pass
7	Northern Highlands	0.37	0.28	Pass	Pass	Pass?
8	**Western Highlands**	**0.20**	**0.16**	**Fail**	**Fail**	**Pass?**
10	**Central Highlands**	**0.47**	**0.83**	**Pass**	**Pass**	**Fail**
11	**Cairngorms Massif**	**0.78**	**0.68**	**Pass**	**Pass**	**Fail**
12	North East Glens	0.26	0.33	NT	NT	NT
13	**Lochaber**	**0.30**	**0.16**	**Pass**	**Fail**	**Pass?**
14	Argyll West & Islands	0.46	0.55	Pass	Pass	Pass
15	**Breadalbane & East Argyll**	**0.50**	**0.50**	**Pass**	**Pass**	**Fail**
19	Western Southern Uplands & Inner Solway	0.19	0.00	NT	NT	NT
20	Border Hills	-	0.33	NT	NT	NT

The Northern Highlands (zone 7) just passed the Level 2b test using the 2003 fledging rate, suggesting that this NHZ may have been close to the lower limits of favourable conservation status in recent years. The low ‘starting’ 2003 population of the North East Glens (zone 12) precluded population modelling using the GEPM, but the low productivity and decline in occupancy suggested that this NHZ would have failed the Level 2 and/or Level 3 tests. Similarly, the two zones in the Southern Uplands (zones 19 and 20) could not be formally tested, but do not appear to be in favourable demographic ‘health’ given the low reproductive rates (zone 19) and low population expansion rate (zones 19 and 20).

Territory mergers

After accounting for vacant territories which had been ‘lost’ due to territorial mergers, over 75% of known territories were occupied in all western Highlands and Islands NHZs, with amalgamations being particularly prevalent in the Northwest Seaboard, Western Seaboard, Northern Highlands and Western Highlands (Table 3). Amalgamations of territories had relatively little impact in eastern Highland NHZs with occupancy remaining below 50%. Estimates of occupancy of known territories, both with and without accounting for mergers, were strongly related to a measure of occupation of suitable habitat by breeding birds as described by Watson & Whitfield (2002) and are suggested as a means of assessing favourable condition based on distribution (Fig. 2).

Table 3. The numbers of known and vacant territories in the 2003 national census, the percentage of known territories which were occupied, and the number of vacant territories ‘lost’ to territorial merger (i.e. vacant territories which had been amalgamated within occupied active territories). The final column shows the estimated percentage occupation of known territories after accounting for vacant territories which had been amalgamated within occupied territories. (From Whitfield et al., 2006).

NHZ	NHZ name	2003 territories				
		Known	Vacant	% occupation	Lost to mergers	Post-merger % occupation
3	Western Isles	93	12	87.1	4	91.0
4	North West Seaboard	71	25	64.8	10	75.4
5	Peatlands of Caithness & Sutherland	31	13	58.1	1	60.0
6	Western Seaboard	98	24	75.5	11	85.1
7	Northern Highlands	90	47	47.8	20	61.4
8	Western Highlands	67	16	76.1	10	89.5
10	Central Highlands	26	14	46.2	1	48.0
11	Cairngorms Massif	71	43	39.4	5	42.4
12	North East Glens	17	14	17.6	0	17.6
13	Lochaber	36	11	69.4	3	75.8
14	Argyll West & Islands	59	15	74.6	5	81.5
15	Breadalbane & East Argyll	27	15	44.4	1	46.2

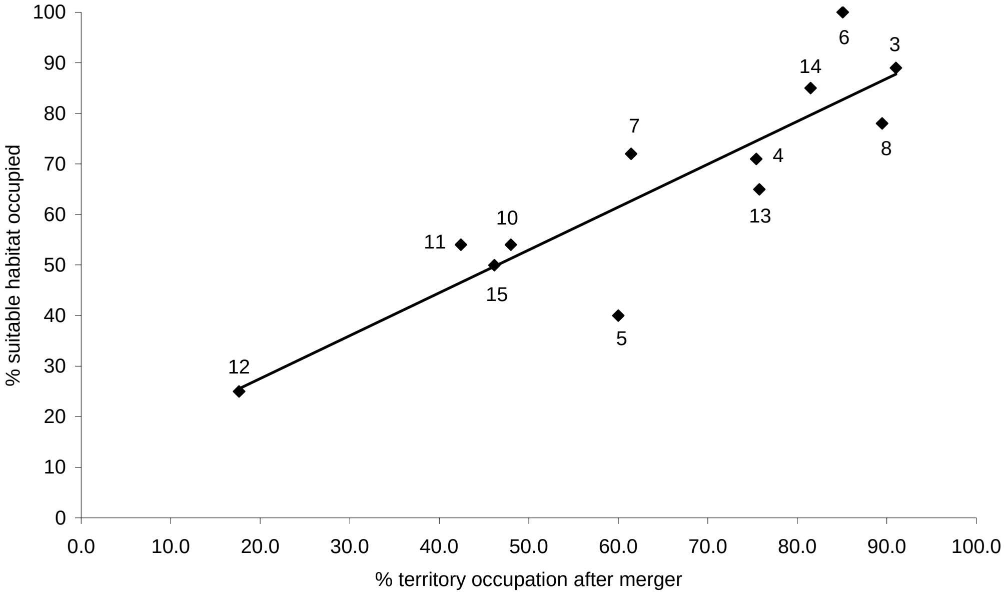

Fig. 2. The percentage of suitable habitat occupied by golden eagle pairs estimated by the percentage of 10 x 10 km grid squares with > 50% upland habitat which contained at least one occupied territory centre in 2003 (after Watson & Whitfield (2002): Table 1) plotted against the percentage of known territories occupied in 2003 after accounting for territorial mergers (present study: Table 3). NHZ number is given beside data points. The good agreement between measures (r = 0.81 after arcsine transformations) was a slight improvement over the relationship between the percentage of suitable habitat occupied (from Watson & Whitfield's measure) and the percentage of known territories occupied before accounting for territorial mergers (r = 0.77 after arcsine transformations). Note that zone 5 contains extensive areas of upland habitat without suitable eagle nest sites, lowering the apparent extent of occupancy as estimated by the method of Watson & Whitfield (2002). (From Whitfield et al., 2006).

4.3 Discussion

Only three of sixteen NHZ where golden eagles have occupied territories since 1982 were considered to be in favourable condition (Western Isles, Western Seaboard, Argyll West and Islands: 3, 6 and 14 respectively) (Table 4). The failure of the Level 1 test by the Northwest Seaboard (zone 4) was marginal and does not present a serious concern, especially as occupancy was 75% after accounting for territorial mergers (Table 3). Although failing the Level 1 test, the condition of the eagle population in the Peatlands of Caithness and Sutherland (zone 5) is recovering to an encouraging degree, apparently coincident with a relaxation in persecution (Whitfield *et al*., 2007b). The marked recovery of eagles in the Western Isles (zone 3) was also coincident with a relaxation in persecution (Whitfield *et al*., 2007b).

Two neighbouring NHZs in western Scotland (Western Highlands and Lochaber: 8 and 13 respectively) failed Level 2 tests on productivity. In other words, insufficient young birds are being produced for potential self-sufficiency and the generation of a stable population from within the respective zones. The Western Highlands has supported a stable population since at least 1982, however, and the implication is therefore that this region is a net importer of recruits to the breeding population. The stable, highly productive Western Seaboard region is the most likely source of these recruits. In Lochaber, productivity appears to have been ‘border line’ for self-sufficiency for a long time. The main difficulty in these two zones is probably a shortage of live prey, because numerous studies have shown a positive relationship between productivity and live prey availability (e.g. review by Watson, 1997).

*Table 4. Summary of the results of the Level 1 – 3 tests of favourable conservation status (FCS) and their resultant effect on regional classification of FCS in Scottish golden eagles. * = not formally tested, but result assumed on the basis of demographic parameter values and population trends; ** = failure to meet FCS was marginal and may be revised if territorial amalgamations were incorporated formally into the conservation framework (Table 3); - = insufficient data to draw a conclusion.*

NHZ	NHZ name	Level tests				FCS?
		1	2a	2b	3	
3	Western Isles	Pass	Pass	Pass	Pass	Yes
4	North West Seaboard	Fail	Pass	Pass	Pass	No**
5	Peatlands of Caithness & Sutherland	Fail	Pass	Pass	Pass	No
6	Western Seaboard	Pass	Pass	Pass	Pass	Yes
7	Northern Highlands	Fail	Pass	Pass	Pass	No
8	Western Highlands	Pass	Fail	Fail	Pass	No
10	Central Highlands	Fail	Pass	Pass	Fail	No
11	Cairngorms Massif	Fail	Pass	Pass	Fail	No
12	North East Glens	Fail	Fail*	Pass*	Fail*	No
13	Lochaber	Pass	Pass	Fail	Pass	No
14	Argyll West & Islands	Pass	Pass	Pass	Pass	Yes
15	Breadalbane & East Argyll	Fail	Pass	Pass	Fail	No
19	W Southern Uplands & Inner Solway	Fail*	Fail*	Fail*	-	No
20	Border Hills	Fail*	-	Pass*	Fail*	No

The failure of the Central Highlands, Cairngorms Massif, North East Glens and Breadalbane & East Argyll (zones 10, 11, 12 and 15 respectively) to pass all tests was probably linked: levels of occupancy are very low (even after accounting for territory mergers) and the results strongly indicated low survival was the problem. Low occupancy has been a long-term feature of these regions and many unoccupied territories are in areas of apparently suitable habitat; suggesting there is a shortage of recruits to fill these many territory vacancies. The shortage of recruits can not be due to insufficient young birds being produced; indeed the eagle population across this region should be expanding due to the high production of young birds in several territories (a long-term feature of these regions: Annex 2). Hence, in the absence of any evidence of emigration from these regions (and that such emigration would not be expected given natal philopatry and the likely presence of several territories with high food supplies) low survival has to be the cause of insufficient recruits, with signs of continuing deterioration rather than improvement.

5. IDENTIFYING AND ASSESSING THE INFLUENCE OF CONSTRAINTS

5.1 Background

Constraints, in the context of a conservation framework, may be defined as factors acting on a population which impinge on meeting or improving favourable conservation status. Several constraints have been proposed or illustrated to be influential on golden eagles in Scotland (e.g. Watson, 1997; McGrady, 1997; Whitfield, 2000; Watson & Whitfield, 2002; Whitfield *et al*., 2006). A key feature of constraint influence is that they differ regionally, which requires policies and actions to address constraints also to be targeted regionally. At a national scale, the priority which should be attached to addressing constraints should be driven by the extent or scale of influence e.g. if constraint A impacts 30 territories and constraint B impacts five territories, addressing constraint A should be the priority, all else being equal.

All constraints ultimately have an adverse effect on the abundance and distribution of eagles via their demography. Contemporary constraints fall into two broad categories, those that have a direct and immediate effect such as persecution and disturbance at nesting sites, and those that operate indirectly and more subtly such as changes in land use and management (Watson & Whitfield, 2002).

Potential or known constraints at the time when we embarked on the framework analyses, not necessarily exhaustive and not listed in any order of influence, are as follows (Watson, 1997; McGrady, 1997; Whitfield, 2000; Watson & Whitfield, 2002; Pedrini & Sergio, 2001a, b, 2002):

- Commercial afforestation
- Nest site availability
- Agricultural encroachment
- Grazing animals
- Persecution
- Unintentional disturbance through recreation
- Wind farms
- Competition with white-tailed eagles *Haliaeetus albicilla*
- Native woodland expansion

Each constraint is considered further in subsequent sections of this report where we have summarised work carried out under the golden eagle framework together with the findings of previous studies of relevance to the particular constraint being discussed. Fielding *et al*. (2003a) includes a detailed examination of both specific and generic studies of constraint influence. To avoid undue replication within the main report we have summarised these studies, and the relevant analytical methods and results, in Annexes 1, 2, 4, 5 and 6. Our aim was to investigate, with rigorous and objective science, each of these constraints, as guided by previous research and our initial analyses. To this end, we have published at least one paper in the peer-reviewed literature on most of the above constraints. Some constraints, such as commercial afforestation and persecution, turned out to require particular research attention, but this attention was justified by the train of our results and the results of other authors' research. Constraints inevitably vary in their influence, and given the huge amounts of effort we expended on this project, there is little point in pursuing issues which did not appear to be especially serious. Occasionally, we have presented novel research, as yet unpublished, or referred to directions which may take understanding further. These future directions are then drawn together in a final report section which makes recommendations for further studies.

5.2 Commercial afforestation

Since 1945, huge tracts of open landscapes in the uplands of Britain have been converted to plantation forestry. Within the golden eagle range, the greatest extent of plantation forestry has occurred in southwest Scotland and in Argyll in the southwest Highlands (Watson, 1997). Until quite recently, most forestry in the uplands comprised exotic conifers such as sitka spruce *Picea sitchensis* and lodgepole pine *Pinus contorta*. After about ten to twelve years the canopy closes in these plantation forests and they become densely packed stands of fast growing trees with very little structural or species diversity. From the golden eagle's perspective this type of afforestation may in the short term affect the availability of sheep and red deer carrion (if the number of animals in an area and/or their foraging range is reduced) and leads, in due course, to habitat loss through the removal of virtually all hunting potential for live prey in the planted area once the tree canopy closes (Marquiss et al., 1985; Watson, 1992, 1997; McGrady *et al*., 1997; Whitfield *et al*., 2001; Pedrini & Sergio, 2001).

Commercial afforestation has been associated with reduced breeding success and territory abandonment in breeding golden eagles, but there are differences between studies. Marquiss *et al*. (1985) showed that large-scale conifer afforestation of Galloway, southwest Scotland, in the 1970s coincided with reduced breeding success of 3 of 4 pairs of golden eagles. Watson (1992) found that breeding success of eagles in Argyll, in the western Highlands of Scotland, was negatively related to the extent of commercial conifer plantations over ten years old. Pedrini & Sergio (2001), however, found no relationship between forest cover and breeding success for eagles in the Italian Alps, as did Whitfield *et al*. (2001) on the island of Mull, western Scotland. Nevertheless, Whitfield *et al*. (2001) did find that, in most territories, breeding success declined following an increase in forest cover. With productivity varying substantially between territories in the absence of forestry, Whitfield *et al*. (2001) suggested that simple correlation between breeding success and forest cover at the level of an individual territory is unlikely to produce a consistent result or reveal how forest expansion can affect breeding success.

In Sweden, golden eagle density in mountain areas above or close to the tree-line was more than double that in areas of extensive forest (Tjernberg, 1985). Similarly, Pedrini & Sergio (2001a) found that eagle density was lower in areas of the Italian Alps with more forest cover, and concluded that forest expansion would have an adverse effect on numbers of eagles. In Argyll, six of seven eagle territories vacated during the 1960s and 1970s contained over 40% plantation forest cover within 4 km of the territory centre, and only three of fifteen occupied territories had more than 40% forest cover (Watson *et al*., 1987). Using the 40% forest cover criterion, Watson *et al*. (1987) predicted that Argyll would lose a further five territories by 1997.

Previous research and attention focussed on this issue in Scotland (Marquiss *et al*., 1985; Watson, 1992, 1997; Harding *et al*., 1994; McGrady *et al*., 1997) gave the strong impression that commercial afforestation was a serious potential constraint which warranted more detailed attention. Thus, we expended considerable effort on this issue, following on from our earlier analyses of forestry impacts on eagles of the island of Mull

(Whitfield *et al*., 2001). As in most of the conservation framework research, we were also guided in research effort expenditure by initial analyses of relationships between land cover and territory occupancy using the 1992 national eagle survey (Annexes 4 & 5). We then embarked on a detailed study (Whitfield *et al*., 2007a) in a region (mainland Argyll) which was previously reported to be the worst-affected by commercial afforestation and where we could test earlier predictions (Watson *et al*., 1987; Watson, 1992, 1997). This was followed by returning to a national perspective (but employing regional data), using change between 1992 and 2003 eagle national surveys (Whitfield *et al*., 2007b).

Our analyses on this constraint utilised: the previously developed PAT (Predicting Aquila Territory) model (McLeod *et al*., 2002a, b; Annex 1) which predicts golden eagle range use; digital land cover data (LCS88, forest stock maps from the Forestry Commission National Inventory of Woodland and Trees, digital terrain models, satellite imagery) (Whitfield *et al*., 2001, 2007b; Annex 1); fieldwork to calibrate digital forest mapping and forest structure (Whitfield *et al*., 2001); golden eagle national survey data (Annex 1); and long-term data on eagle breeding productivity, territory occupation and nest site use for Mull and mainland Argyll (Whitfield *et al*., 2001, 2007b). The methods we employed offered several advantages over previous analyses, notably the use of better characterisation of eagle range use and, therefore, impacts of habitat loss through afforestation, and a longer-term perspective.

The framework analyses (Annex 4) indicated, as expected, that conifer forests were associated with vacant ranges (Table A4-3) and suggested that range loss and subsequent merger associated with afforestation has occurred in Argyll West and Islands, Western Seaboard, Lochaber, and Western Isles (zones 14, 6, 13 and 3 respectively) (Table A4-4). An unexpected finding, though with hindsight entirely consistent with the distribution of conifer plantings, was that some ranges had been lost in the Northern Highlands (zone 7) due to commercial afforestation (Table A4-4). Although not picked up by our analyses, due to small numbers of eagles, it is also apparent that afforestation has caused difficulties for the small number of territories in the Western Southern Uplands (zone 19) (Marquiss *et al*., 1985). There were few indications of breeding productivity being associated with forest cover (Tables A2-7, A2-8) but this is not surprising given the conclusions of Whitfield *et al*. (2001, 2007a) that such an association should not necessarily be expected, even if forest cover has affected breeding output.

Whitfield *et al*. (2001) showed that on Mull (in Western Seaboard NHZ 6) two ranges had probably been abandoned due to afforestation but these losses were balanced by new ranges forming elsewhere. Change in breeding productivity on individual ranges was highly variable in relation to increasing forest cover and so could not be predicted safely, but across all ranges breeding success declined when forest cover exceeded 10 – 15% of areas used by territorial birds.

Whitfield *et al*. (2007a) showed that the prediction of range losses in mainland Argyll (NHZ 14) (Watson *et al*., 1987) was not realised, however, and that the response of individual pairs of golden eagles to afforestation in Argyll was highly variable, with some pairs apparently abandoning their territories when forest covered less than 5% of their territory use, yet others showed enhanced breeding productivity when nearly a third of their territory use was lost to forestry. Hence, reiterating Whitfield *et al*. (2001), it was suggested that the response of eagles to forest expansion was more complex than previously appreciated and that using set criteria was not reliable when predicting whether a given amount of forest expansion would cause territory abandonment. This is because breeding success differs between eagle pairs before afforestation so there will not necessarily be a relationship between forest cover and breeding success after afforestation.

The results of Whitfield *et al*. (2007a) suggest that, in Argyll, the first phase of forest planting caused several territories to be abandoned. Other territories were not abandoned but probably experienced a reduction in breeding success. The extent and location of new forest was undoubtedly influential in this process, although territories that were most susceptible to abandonment were probably those with a history of low breeding success (see also Kochert *et al*., 1999) and those that were more constrained by neighbouring pairs of eagles. After the first phase of forest development had removed some pairs, some remaining pairs of eagles apparently responded positively to later plantings by shifting their territory use into the ‘gaps’ that had been created and experienced enhanced productivity (as predicted by Whitfield *et al*., 2001). Pairs that were still constrained by other neighbouring pairs of eagles (or other unsuitable habitat) had fewer opportunities for a compensatory response, and so were adversely affected by secondary plantings of new forest. It was also apparent that territories with low breeding productivity were more likely to be abandoned (Whitfield *et al*., 2001, 2007a).

Thus, while commercial afforestation has caused a reduction in breeding success and territory abandonment in those regions where it has been extensive, its effects are not necessarily as detrimental as was once thought. Clearly, however, several territories have been lost and breeding success depressed in some pairs as a result of afforestation in the recent past (despite some pairs having enhanced productivity).

It is unlikely that many of these ‘lost’ territories can be regained, due to the large cost involved in restoration set against the economic value of many of the forests. However, commercial afforestation does not appear to be an obvious cause of any regional failures to meet favourable conservation status (although it may have made a small contribution to the unfavourably low productivity in zone 13). Changes in policy and practice relating to commercial forestry have taken place in recent years. Planting of new conifer forests has declined substantially (Reid, 1997; SEDDESB, 2005) and only a small number of territory vacancies between the 1992 and 2003 surveys were associated with (though not necessarily due to) new plantings (Whitfield *et al*., 2007b). In addition, in at least some of the regions where territory losses may have occurred due to forestry, new ranges were recorded. The Forestry Commission is also actively pursuing sensitive forest management with potential benefits to golden eagles where appropriate (K. Wishart, pers. comm.).

Currently, therefore, commercial afforestation is not considered a marked constraint on golden eagles. Historically it has caused reductions in breeding success and territory abandonment in some regions, but apparently without obvious effects on favourable conservation status. The more recent trend for native woodland expansion is considered in a later section.

For further details see McLeod *et al*. (2002a, b), Whitfield *et al*. (2001, 2007a, b), Annex 1 (methods), Tables A4-3 & A4-4, and Annex 5.

5.3 Nest site availability

Golden eagles use both cliffs and trees for nest sites. Cliffs appear to be preferred (e.g. Watson, 1997; Halley, 1998) but trees are used frequently in some areas (e.g. Tjernberg, 1983a, 1985; Watson & Dennis, 1992), with a requirement for trees of sufficient maturity and structure to support the large nest. In Scotland the vast majority of nests are on cliffs and almost all tree nests are in old growth Caledonian pinewoods (Watson & Dennis, 1992; Watson, 1997). Hence, eagle abundance and distribution may be limited by the availability of suitable cliff sites and/or trees.

Our analyses on this potential constraint involved the use of LCS88 and digital terrain (elevation) data. As well as the full dataset of the 1992 national eagle survey we also used a subset of the 1992 national survey in which observers had recorded altitude and aspect of nest sites (Annex 1).

Analyses examining the potential for nest site availability to be a constraint on eagles were confounded by difficulties in generating measures of cliff site availability due to the (for this purpose) low resolution of nest site grid references coupled with a poor capability of land cover data to document all cliffs (Annex 4). When dealing with cliff nest sites a 6-figure grid reference, with a resolution of 100 m, can lead to errors in documenting the exact location, and hence in deriving attribute features of that location. This is because contour bands at cliffs are obviously very strongly compressed and so even small differences in a given location (i.e. within 100 m) can lead to large differences in, for example, altitude and aspect of the given location. Coupled with this, in LCS88 it was also apparent that the ‘rock and cliff’ class did not incorporate all areas that, through either personal experience or eagle nest site locations, we knew were cliffs. LCS88 was therefore probably more use as a relative index of cliff availability across wide areas than as an absolute measure of cliff availability in specific locations.

The finding that vacant territories had fewer areas of cliff than occupied territories (Table A4-3), based on LCS88, probably reflected the more general finding that vacant ranges in the west were at the periphery of the golden eagle’s geographical range, away from the most rugged ground, and closer to areas of human habitation (Annex 4). The same influence may have explained the result that vacant territories had fewer alternative nest sites than occupied territories (Annex 4: Table A4-15), although territories which are occupied over longer periods of time may generate greater potential for more alternative sites to be used and documented, regardless of availability. The latter explanation may be most appropriate because there were no differences in the nest site features of territories with only one or more than one alternative nest sites (Annex 4: Tables A4-15 & A4-16, Fig. A4-2).

In general, therefore, although analyses were confounded by difficulties in data resolution and predicting the availability of ‘suitable’ cliff nest sites, there was no convincing evidence for a shortage of nest sites for golden eagles in most regions (for example, in the Western Isles over 90% of known territories (after mergers) were occupied in 2003, yet the breeding golden eagle population continues to expand as does a cliff-nesting population of white-tailed eagles: see later section) (Whitfield *et al*., 2006). Watson & Whitfield (2002), using LCS88 and a digital inventory of Caledonian pinewoods held by SNH, identified two regions where nest sites may be constraining, based on the availability of cliff habitats and Caledonian pinewoods (with greater emphasis on the former given the relative rarity of the latter): the Border Hills (NHZ 20) and the Peatlands of Caithness & Sutherland (NHZ 5). Given the large expanse of blanket bog in the Peatlands, the absence of eagles from extensive areas in this zone, whilst potentially explicable by a shortage of nest sites, is arguably entirely ‘natural’. In the Border Hills, management to allow potentially suitable tree nest sites to develop may be appropriate. Although golden eagles are considered to require large old trees (Tjernberg, 1983a; Watson & Dennis, 1992; Watson, 1997), as are white-tailed eagles (Shiraki, 1994; Halley, 1998; Helander & Stjernberg, 2003), experience with the reintroduced white-tailed eagles in Scotland may suggest that at least some pairs of golden eagles may use trees which would not necessarily be predicted to be suitable based on age. Interventionist management to create artificial tree nest sites (Mecionis, 2003; DeLong, 2004) may prove fruitful should more pairs show signs of establishing territories in the Border Hills; currently, however, there appear to be too few birds in the population to allow a marked recovery of golden eagles south of the Highlands (Whitfield *et al*., 2004b, 2006).

For further details refer to Annex 1 (methods) and the ‘Cliff nest site availability’ section of Annex 4.

5.4 Agricultural encroachment

The initial framework analyses confirmed that active eagle ranges are more likely in rugged mountainous terrain and that vacant ranges are more likely at lower altitudes with flat terrain (Tables A4-1 & A4-2). Active ranges were also more likely in areas with upland vegetation types (Table A4-3). These findings were confirmed by the more complex predictive analyses involving decision trees and neural networks (Annex 5). As golden eagles in Scotland therefore appear largely to exploit higher altitude, rugged areas with upland vegetation types, any encroachment of agriculture, through creation of additional hill pasture, for example, could have a detrimental effect.

Our analyses on this potential constraint involved the use of LCS88 and digital terrain (elevation) data, and the 1992 national eagle survey data.

Overall and on balance, there was no indication that increasing expansion of hill farm agriculture was an issue for golden eagles. Indeed, given recent and ongoing changes in agricultural subsidy criteria and after the 2001 episode of Foot and Mouth Disease, further agricultural expansion in the uplands would seem unlikely in the near future There was evidence that some eagle territories in the west which were in closer proximity to agriculture and human activity were less likely to be occupied (Tables A4-4 & A4-6); however these vacancies might also have been explained by their peripheral location in respect to the geographical range of eagles, and a less-rugged topography (Annex 4, see also Watson & Dennis, 1992). Similar results have been found in other populations of golden eagles (e.g. Haller, 1982; Pedrini & Sergio, 2001b, 2002), in Spanish imperial eagles *Aquila adalberti* (González *et al*., 1990, 1992; Bisson *et al*., 2002) and other large raptors (e.g. Carrete *et al*., 2002).

It seems likely that low occupancy of such geographically peripheral territories is because they are of lower ‘quality’ (Sergio & Newton, 2003), and are rendered as such by reduced availability or abundance of food and/or greater disturbance from human activities either through direct effects at the nest (White & Thurow, 1985; González *et al*., 2006) or indirectly by habitat loss to disturbance or degradation (McGarigal *et al*., 1991; Fernández-Juricic, 2002). When raptor populations expand such peripheral territories are often re-occupied (Haller, 1982, 1994; González *et al*., 2006), which suggests that occupation may ultimately be influenced by the numbers of birds available to inhabit territories. The ultimate influence of population status on occupation (and the relative importance of proximate factors like human activity) is illustrated by the Western Isles golden eagle population. For these birds, analysis of the 1992 survey data suggested reduced occupation of peripheral territories (Annex 4) yet expansion of the population by 2003 saw many of these territories re-occupied (Table 1, above; Whitfield *et al*., 2006). This theme is considered in more detail later under the 'unintentional disturbance' constraint.

We should probably always expect that some territories will be of lower quality and that such territories will often be at the limits of suitability, and so less likely to be occupied. Given the strong influence of mountainous areas of topographical complexity on the distribution of golden eagles in Scotland (Annex 4) we should expect reduced occupation at the edges of the uplands. In the absence of any likelihood of agricultural expansion and of any evidence for unfavourable conservation status being brought about by such encroachment (and with occupation of territories in close proximity to agricultural areas being at least partly conditional on the availability of birds), it seems highly unlikely that this constraint is of major influence, even in the west where an association with occupation was noted.

For more details refer to Annex 1 (methods), the 'Altitude and slope' and 'Land cover' sections of Annex 4, and Annex 5.

5.5 Grazing animals

The relationship between golden eagles in Scotland and large grazing animals, notably ungulates such as sheep and red deer, is complex (e.g. Watson, 1997; Watson & Whitfield, 2002). On the one hand eagles depend on carrion in the form of dead sheep and deer, especially for food in winter. On the other hand, over-grazing by large numbers of red deer and both past and present high sheep numbers, combined with a very wet climate, can result in loss of much of the 'natural' cover of heather *Calluna vulgaris* and other dwarf shrubs. These plants provide food supplies for important herbivorous prey species of golden eagles: red grouse *Lagopus lagopus scoticus* and, to a lesser degree, mountain hare *Lepus timidus*. Hence, the abundance of live prey for eagles can be effectively reduced through competition with larger grazers (Watson *et al.*, 1987, 1992a; Watson, 1997; Watson & Whitfield, 2002). This could potentially reduce breeding success and induce territory abandonment by preventing a territory being suitable for breeding, although carrion has been considered to be more influential on breeding density than live prey (Watson *et al.*, 1987, 1992a; but see Harding *et al.*, 1994, Fielding *et al.*, 2003a).

Eagles in several regions of Scotland have a large component of carrion (sheep and/or red deer) in their winter diet (Watson *et al*., 1993), especially in some western areas, and Watson *et al*. (1992a) found that differences in breeding density of eagles across Scotland were positively correlated with differences in an index of carrion abundance. The highest densities of eagles occurred in the west mainland and on the islands where the abundance of deer and sheep carrion was greatest. However, the same study showed no such positive correlation between carrion availability and breeding performance. Instead, breeding success was positively correlated with an index of numbers of medium-sized wild herbivores such as red grouse, ptarmigan *Lagopus* *mutus*, mountain hare and rabbit *Oryctolagus cuniculus*. In regions where breeding success was highest (in the eastern Highlands) grouse and hares were abundant. The poorest eagle breeding performance was in the western mainland where grouse and hare numbers were exceptionally low and where there were very few rabbits as alternative prey. In this area, over-grazing by large numbers of red deer and sheep, combined with a very wet climate, was proposed to have resulted in loss of much of the 'natural' heather cover, which in turn explained the low grouse and hare numbers.

Hence, abundant winter carrion, which occurs where large ungulates are present in high numbers, can lead to high densities of eagles. However, heavy grazing pressure by large ungulates, notably in the wet west of Scotland, can also result in the loss of medium-sized wild herbivores (principally red grouse and mountain hare: Watson, 1997) that are critical summer food for eagles if they are to breed successfully (Watson *et al*., 1992a; Watson, 1997; Watson & Whitfield, 2002).

The relationship between live prey abundance and breeding success suggested by Watson *et al*.'s (1992a) studies have considerable support from many other studies both of golden eagles and other raptors (Newton, 1979). Tjernberg (1983b) showed that breeding success in Swedish golden eagles was related to prey abundance. Elsewhere, when numbers of primary prey species declined either through prey population cycling (Gjershaug, 1996; Steenhof *et al.*, 1997) or disease (Fernández, 1993), golden eagle breeding success also declined. Watson (1997) found that across their global range, breeding success in golden eagles appears to be related to the degree of dietary specialisation (reflecting the abundance of preferred prey items) with breeding groups concentrating on few or single prey species (typically lagomorphs, marmots, grouse or pheasants) having higher breeding success than groups with a greater breadth of prey species. This finding was confirmed by Katzner *et al*. (2005) for imperial eagles *Aquila heliaca* in a region of Kazakhstan.

Visually, the impact of large ungulate herbivores on vegetation composition and structure, and thereby potentially on smaller herbivores which are important live prey for golden eagles, is readily apparent from comparisons of areas subject to different grazing intensities (Fig. 3: see also, for example, plates in Thompson *et al*., 2003). The loss of heather and its replacement by grasses, sedges and rushes through frequent burning and grazing and trampling by red deer and, notably, sheep is also well documented (Rawes, 1981; Grant *et al*., 1981; Hobbs & Gimingham, 1987; Sydes & Miller, 1988; Grant & Armstrong, 1993; Clarke *et al*., 1995; Hope *et al*., 1996; Hester *et al*., 1999; Palmer *et al*., 2003; although see Virtanen *et al*., 2002). Red grouse, with a diet primarily composed of heather, will clearly be badly affected by a loss of heather through grazing by large ungulates, and the decline in red grouse during the 20th century seems most likely related primarily to increases in large ungulate grazing pressure and associated land management (e.g. Sydes & Miller, 1988; Hudson, 1992; Redpath & Thirgood, 1997; Fuller & Gough, 1999). Although there are studies suggesting mountain hares can be out-competed by larger ungulate grazers (Hewson, 1989; Hulbert & Anderson, 1991; see also Hope *et al*., 1996), the effects of the loss of heather on mountain hares is less obvious. Scottish mountain hares browse heather, mainly in winter, but prefer graminoids (Hewson, 1989; Hulbert *et al*., 1996; Hulbert & Iason, 1996). Despite the common perception that mountain hares in Britain are associated with heather moorland, Hulbert *et al*. (1996) suggested that the presence of moorland was not a prerequisite for the presence of mountain hares. Loss of heather per se therefore may have less detrimental impact on hares than on red grouse.

Fig. 3. Photograph taken in central Scotland in June 2005 illustrating the effect of large ungulate grazing pressure on open landscape vegetation, with heavier grazing below the fenceline, dominated by grasses, and lighter grazing above the fenceline, dominated by heather Calluna vulgaris*.*

The purported lack of influence of carrion on breeding success (Watson *et al*., 1992a) was confirmed by Pout (1998) whose analyses of golden eagle pellets on the Isle of Harris indicated that live prey was the most important resource for breeding (although see Agafanov *et al*. (1957) in Watson (1997) for an account of carrion possibly affecting reproductive parameters in steppe eagles *Aquila nipalensis*). Pout's (1998) study suggested that the amount of carrion available was likely to be well in excess of the winter requirements of golden eagles so it was potentially available as a food resource during the breeding season. Moreover, while Watson *et al*.'s (1992a) choice of study subjects tried to avoid areas where persecution occurred, a low territory density in several eastern and central Highland regions can be influenced by persecution away from estates where eagles are tolerated (Whitfield *et al*., 2004a, b) and so the low density in eastern and central regions in Watson *et al*.'s (1992a) relationship may have been confounded by persecution (see also Harding *et al*. (1994) for additional critiques). Conversely, Watson *et al*. (1989, in prep.), comparing temporal trends in territory occupation supported Watson *et al*.'s (1992a) spatial analysis by illustrating that deer carrion was influential (interestingly, however, these studies were in northeast Scotland where carrion was implicitly assumed to be only weakly influential in Watson *et al*.'s (1992a) regional comparisons).

Our main analyses on this issue used several datasets obtained through SNH: land cover (LCS88), digital terrain, numbers of sheep and cattle in each parish (based on June agricultural census data from 1998), and deer numbers taken from Deer Commission for Scotland (DCS) counts 1993-1998. Measures by parish of vegetation biomass offtake by deer and livestock, vegetation biomass production and grazing 'equity' (production – offtake) were also obtained from SNH and Dr Helen Armstrong (a summary is given below, and more details are in Annex 1 'Grazing, livestock and large herbivores'). Caveats associated with deer and livestock data are given in Annex 1 ('Grazing, livestock and large herbivores' section) along with further details of the analysis methods: 1982 and 1992 national eagle surveys were also used in these analyses. We also examined changes in sheep and deer numbers between the two most recent eagle surveys using estimated deer numbers in 1992 and 2003 from DCS data and June parish census data for sheep. Whitfield *et al*. (2007b) and Annex 1 ('Grazing, livestock and large herbivores' section) provide further information on these 'change' analyses.

Armstrong *et al.* (unpublished) calculated, for each parish, the biomass offtake (kg of dry matter (DM)) by grazers (sheep, red deer and cattle) in relation to biomass production (kg DM) by the vegetation (see Annex 1, 'Grazing, livestock and large herbivores' section, for further details of this and other relevant methods). The difference between production and offtake gave a grazing equity figure. A high equity value represents a high production by vegetation relative to offtake, which will leave more vegetation biomass available to grazers such as red grouse and mountain hare that are important prey for eagles. On the other hand, a low equity value represents a low production by vegetation relative to offtake, leaving less food for the herbivorous prey of eagles. Equity values may therefore provide a useful surrogate for the availability of the key prey species of golden eagles in Scotland. Whereas live prey may be reduced when equity is low, regardless of the numbers of large grazers, carrion should be relatively high when there was a coincidence of low equity and high numbers of grazers because if available vegetation biomass is low when numbers of grazers are high then there will be less food per capita and so more sheep and/or deer fatalities.

Overall, analyses suggested that active territories had higher grazing equity values than vacant territories (Annex 4: Table A4-8) but there was little apparent coincidence between high numbers of grazers and low equity (Annex 4: Tables A4-8 & A4-9). These results gave only limited support to the notion that territory occupancy was related to carrion availability, but gave a stronger indication that the availability of live prey (as influenced by the competitive effects of larger herbivores) may have an influence on occupation of territories in some regions. Indications were that low equity was more of a problem in the west than in the east following earlier suggestions (Watson, 1997). There was evidence that in the Western Highlands and Northern Highlands range vacancy was associated with a high offtake of vegetation production by large grazers (Western Highlands only), lower vegetation production and a low grazing 'equity' (Annex 4: Table A4-9). Given the densities of animals involved, the major contributory grazing mammals to these patterns were likely to be red deer, although sheep made some contribution. Whitfield *et al*. (2007b) also rejected a hypothesis that regional changes in territory occupation between 1992 and 2003 national golden eagle censuses were influenced by change in carrion abundance because there were no clear patterns relating change in territory occupation and changes in sheep and deer.

The effects of heavy grazing and over-frequent burning of vegetation on breeding success and territory occupation are probably widespread, especially in the west (Watson *et al*., 1992a; Watson, 1997; Watson & Whitfield, 2002) but are likely to be more important in some regions than others. Previous research (Watson *et al*., 1987, 1992a; Watson, 1997) emphasised detrimental effects in the area roughly equivalent to two neighbouring NHZs (Western Highlands and Lochaber: 8 and 13 respectively), which as noted earlier, failed Level 2 favourable conservation tests on productivity. The effects of sheep and red deer grazing and over-frequent burning of vegetation on live prey may be the most likely constraint which is deserving of attention in these regions (Watson & Whitfield, 2002; Fielding *et al*., 2003a). Our analyses only found a link in Western Highlands, however. Deficiencies in breeding productivity data (Annex 2) meant that many of our analyses in this regard were equivocal, and that it would be very useful to re-visit this work with long-term productivity data across many territories.

The hypothesis that in Scotland, golden eagle breeding density (and thus territory occupation) is influenced by carrion and not live prey, and that breeding success is influenced by live prey and not carrion, is likely to be an overly simplistic interpretation. While supported by some studies, it is also challenged by several other lines of evidence which suggest more complex linkages, for example:

- the possibility of a direct link between live prey, breeding success and territory occupation: if, through inadequate live prey, breeding attempts are not possible in a territory then occupying and defending it is not an effective strategy;
- if the density of resident pairs is increased by carrion then the live prey available to each pair is reduced, as is per capita productivity, but productivity of the population as a whole per unit area may increase (analyses illustrate how productivity of golden eagles per unit area is higher in the high density west than in the low density east: Annex 2)

The findings of Fielding *et al*. (2003a), repeated in Annex 4 (see Tables A4-8 & A4-9), suggested that live prey do influence territory occupation, and studies of habitat loss on eagles (Whitfield *et al*., 2001, 2007a) illustrate how territory abandonment can occur when the capability for breeding is compromised.

Major reductions in sheep and deer numbers and more controlled burning may reduce carrion availability and, hence, the number of occupied eagle territories in a given area (Watson, 1997). However, high densities of eagles sustained by carrion may not produce sufficient young to replace losses of breeding pairs, which will lead to a decline in regional breeding density in the absence of immigrant recruits (Whitfield *et al*., 2006). In several areas in the western Highlands and Islands (notably the Lochaber and Western Highlands zones), high numbers of red deer and/or sheep may be contributing to a ‘moribund’ eagle population. A reduction in carrion availability, for example, as a result of management to reduce sheep and deer numbers, might reduce the number of breeding pairs, but such a reduction may also occur due to low breeding productivity unless recruits are available from neighbouring populations (and drawing recruits away from more productive regions arguably reduces the capability of these host populations to be buffered against population decline by non-breeding adults: Hunt, 1998).

As well as reducing carrion, sheep stock reductions and deer culls may also increase the abundance of live prey which could enhance local breeding success of eagles. It is possible, therefore, that reductions in the number of occupied territories resulting from less carrion might be offset by enhanced breeding success and the availability of more recruits to a breeding population. Indeed, productivity gains may potentially be high enough to allow numbers of non-breeding adults to develop, thereby providing a buffer for the population against periods of demographic adversity, and resulting overall in a larger population of eagles.

The preceding discussion highlights that the commonest method of monitoring eagle population status, counting resident pairs, does not necessarily provide a good indication of the 'health' of the population or even the number of eagles in the population (e.g. Hunt, 1998; Kenward *et al*., 2001; Whitfield *et al*., 2004a, b; Penteriani *et al*., 2005b). There are however practical difficulties associated with the monitoring of non-breeding birds which, for example, may range over extensive areas.

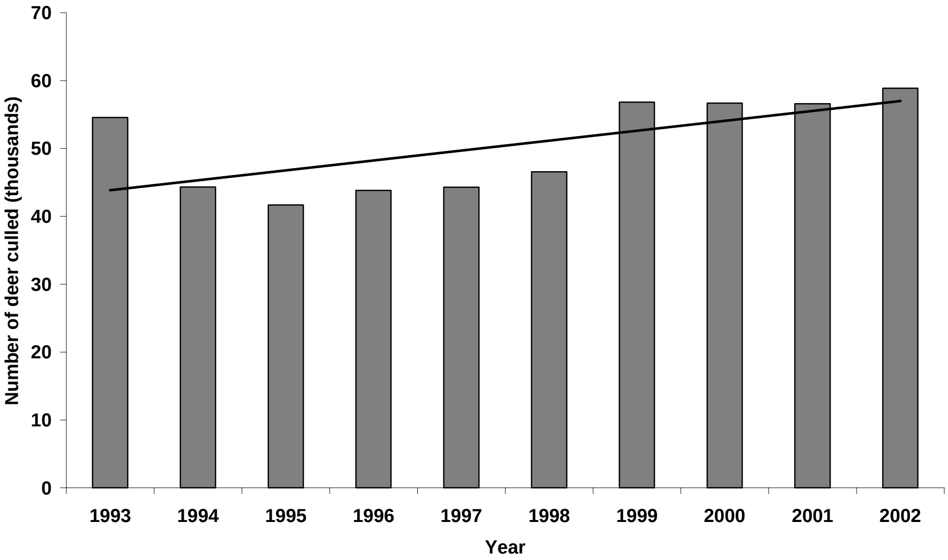

Fig. 4. The numbers of red deer culled in Scotland 1993 to 2002 according to statistics collated by the Deer Commission for Scotland (1997, 2002, 2003), shown with a linear trend line.

After a long period of increase (e.g. Sydes & Miller, 1988; Fuller & Gough, 1999) sheep numbers in Scotland appear to be in decline, probably reflecting the changing economics and subsidies available to sheep farming and, on grouse moors, their likely removal as reservoirs of diseases which also affect red grouse (Whitfield *et al*., 2007b; Laurenson *et al*., 2003). Attempts to decrease deer numbers, for a number of biodiversity benefits, have been ongoing for several years (SNH, 1994; Hunt, 2003) but although culls of deer have increased in recent years (Fig. 4) there are few signs of widespread reductions in red deer numbers; indeed, numbers continue to increase in many regions (Hunt, 2003; Whitfield *et al*., 2007b). As noted by Hope *et al*. (1996), any benefits to vegetation and smaller herbivores through reductions in sheep are unlikely to be realised in the Highlands without concomitant changes in red deer and burning management.

Given the obvious practical difficulties in reducing numbers of large grazers across large areas, the most cost-effective option in those regions where such reductions are most obviously needed (Lochaber and West Highlands) would be to concentrate attempts to increase productivity in selected territories where such management would incur the most benefit (Sergio & Newton, 2003): currently in Lochaber and West Highlands a disproportionately small number of territories contribute to the regional production of young (Fielding *et al.*, unpublished). We emphasise that any management, however, would benefit from a thorough preceding analysis of long-term changes in breeding productivity, territory occupation, diet, and grazer densities, on a territory-by-territory basis.

For further details see Annex 1 (for methods) and Annex 4 (for results), and Whitfield *et al*. (2007b).

5.6 Persecution

Despite more than 60 years of statutory protection, some golden eagles are still killed illegally each year in Britain (e.g. RSPB, 2001, 2003). The commonest method of persecution is by poisoning which may or may not be targeted at raptors, including golden eagles. Deliberate persecution by shooting and trapping of eagles can still also occur, but is probably more difficult to detect. Destruction of nests or nest contents, or other interference with breeding attempts through deliberate disturbance may also occur (e.g. Watson & Dennis, 1992; Watson, 1997; Scottish Raptor Study Groups, 1997). For many years persecution has been considered a factor limiting the distribution and abundance of golden eagles in Scotland, with grouse moor management strongly implicated in its perpetration (e.g. Sandeman, 1957; Brown, 1976; Love, 1983; Newton, 1994; Green, 1996; Scottish Raptor Study Groups, 1997; Watson, 1997; UK Raptor Working Group, 2000; Thompson *et al*., 2003). Despite such previous work, rigorous and geographically explicit quantification of the potential links between persecution and golden eagle ecology had not been conducted prior to our research.

Our analyses of this potential constraint involved the use of records of the illegal use of poisoned baits to control predators from the years 1981-2000. These records may or may not have involved humans targeting raptors (including golden eagles) as their victims; but like other carrion feeders, some raptors such as golden eagles, if present, were vulnerable to being killed by such acts, and were either recorded as victims or were indicative of attempts to illegally control predators, which could include golden eagles, if present. Our efforts to remove potential spatial pseudo-replication and lack of spatial specificity in the data are described by Whitfield *et al*. (2003). These data were used in most of our analyses on direct inferences of persecution, including our initial explorations of potential constraint influence (Annexes 1 & 4). A second dataset, restricted purely to records of persecution of golden eagles (shot or trapped, poisoned, or apparently deliberate destruction of nests and/or nest contents e.g. burnt nest sites or smashed nest contents) between 1981 and 2003 was used in analysis of change in eagle territory occupation in 1992 and 2003 national surveys (Whitfield *et al*., 2007b). The same exercise on accounting for pseudo-replication as applied to the poisoning data was also repeated on these data, termed eagle persecution data. We also used land cover data in LCS88, in particular the muirburn class, as a unique surrogate for the distribution of areas under grouse moor management. Eagle data involved those from the 1982 and 1992 national surveys (Annexes 4 & 5; Whitfield *et al*. 2003, 2004a, b) and the 1992 and

2003 national surveys (Whitfield *et al*., 2007b). Please refer to Annex 1 and Whitfield *et al*. (2003, 2004a, b, 2007b) for further details on methods and validation of the datasets.

It is worth pointing out in relation to the emphasis we have eventually placed on the influence of this constraint that initially, on the basis of previous work, and before any of our analyses, we did not realise that persecution could prove to be remarkable in its influence, beyond potential localised effects. Our analyses on this issue began, as for other constraints, by overlapping land cover data with the distribution of active (occupied) and vacant (unoccupied) eagle territories. This revealed that dry heather and strip muirburn were the most common vegetation types underlying vacant territories (Table A4-3), a somewhat unexpected result given the likely suitability of these vegetation types for eagles, although both are heavily associated with grouse moors. Nevertheless, the result was confirmed by the predictive modelling using decision tree and neural networks, which showed dry heather to be powerful negative predictor of occupied eagle territories (Annex 5). Simple examination of the distribution of eagle territories showed that many were vacant in the eastern and central Highlands, where dry heather and strip muirburn often predominated. Next, we found that in most of these regions there were more poisoning incidents than expected (Table A4-14) and recalled that in these regions none of the other potential constraints (e.g. conifer forests, human habitation, centres of recreation activity), appeared to have much, or any apparent associations with territory occupancy (Annex 4). This prompted an examination of a possible relationship between strip muirburn, a surrogate of grouse moor distribution, and poisoning incidents (Whitfield *et al*., 2003) which found a significant association between strip muirburn and poisoning, showing that poisoning incidents in the uplands were more likely to occur on grouse moors. Poisoning incidents were also widespread, having been recorded from northeast Sutherland down to the southern limits of the Highlands, and across the uplands of southern Scotland. There was an indication that poisoning had declined in the west since 1981, but no indication of a decline in the east and south (Whitfield *et al*., 2003).

As poisoning is potentially lethal for golden eagles, this could explain why we found initially that strip muirburn was associated with vacant territories, and so we undertook more detailed analyses of the distribution of vacant territories, the occurrence of subadult birds in pairs (a sign of a shortage of adults) and the occurrence of subadults which were not apparently occupying a territory (Whitfield *et al*., 2004a). The results showed that vacant territories were significantly associated with poisoning records and strip muirburn, as were territories which were occupied by pairs in which at least one bird was not an adult. Widespread territory vacancies and a shortage of adult birds (indicating low survival) could thus be explained by persecution on some grouse moors. There were also more records of non-breeding subadult (immature) birds in these eastern regions where grouse moor and poisoning predominated than could have been produced by the breeding pairs, suggesting that young birds from further west were being attracted into the area, probably by the good food supplies and the widespread absence of territorial pairs. As such movements were placing young birds at risk of dying through persecution there was probably a ‘black hole’ or ‘ecological trap’ at work (Grant & McGrady, 1999; Delibes *et al*., 2001) where dispersing birds are killed in otherwise attractive habitat before they have a chance to return closer to their natal area to breed. Thus, the effects of persecution could act on eagle populations beyond the limits of its occurrence. With several indications of adverse effects, and a widespread influence, Whitfield *et al*. (2004a) concluded that persecution appeared to be having a major impact on the demography of eagles. These analyses were pointing to a serious influence but did not tell us, however, how *much* this constraint may be affecting the demography of eagles, and so we carried out a further study which examined this (Whitfield *et al*., 2004b).

In essence, this next stage of the work showed that persecution in the eastern and central Highlands appeared to be putting the Scottish eagle population at risk of decline through reducing eagle survival rates and that the shortage of occupied ranges in these areas was consistent with the estimated effect of persecution on survival (Whitfield *et al*., 2004b). That persecution was apparently having a marked effect on eagle survival in eastern and central regions was echoed via a different analytical route, involving the 2003 national survey (earlier work had involved only the 1982 and 1992 national surveys) when we were testing all regions for favourable condition (see section 5, above: Whitfield *et al*., 2006).

Finally, we looked at the possible influence of several constraints on the change in occupied territories between the 1992 and 2003 national surveys (Whitfield *et al*., 2007b). We included eagle persecution data as well as poisoning data in this study (see above, and Annex 1) and found that the two datasets were positively correlated in their distribution and in their change over time. There was little indication that indices of recreation and carrion were related to change in occupied territories between the two surveys, and only a small number of vacated territories (at worst, and largely away from the east) had been recently planted with conifers. Regional changes in occupied territory numbers were negatively associated with persecution. The decline in the Scottish breeding population predicted by Whitfield *et al*. (2004b) had not happened, on face value, but this was largely the result of an increase in the Western Isles (where recorded persecution had decreased) and the population *had* declined in the regions highlighted by the earlier studies as being most likely to decrease (Whitfield *et al*., 2007b).

Hence, there were several lines of evidence from analyses underpinning the conservation framework to indicate that persecution was a major constraint on golden eagles in Scotland, with the influence being primarily centred on (but its effects not exclusively limited to) regions in central and eastern Highlands where grouse moor management prevailed (Central Highlands, Cairngorms Massif, Northeast Glens, Breadalbane & East Argyll). Reiterating the chronological summary above, this evidence was as follows:

- Nationally, dry heather and muirburn vegetation types (both strongly associated with grouse moors) were associated with more territory vacancies than any other vegetation types (Annex 4: Table A4-3). Counter-intuitively, in view of its potential high suitability as a habitat for golden eagles, dry heather was such a strong influence that it was a negative predictor of territory occupancy in predictive modelling of eagle distribution using neural network and decision tree models (Annex 5).
- There was no consistent or strong evidence of associations between territory vacancies and non-persecution constraints in the regions where dry heather and muirburn predominated (Annex 4).
- Apparently deliberate interference on breeding attempts appeared to have marked effects on golden eagle productivity in several regions (Annex 2: Table A2-6; see also Watson & Dennis, 1992).
- Mapping a form of land management unique to grouse moors ('strip muirburn'), Whitfield *et al*. (2003) used a GIS analysis to show that records of illegal poison

use from 1981 to 2000 were disproportionately associated with grouse moors in Scotland (Fig. 5). The association between poisoning incidents in the uplands and grouse moors was stronger in later years of the study period. Whitfield *et al*. (2003) suggested that this was at least partly due to a decline in the illegal use of poisons away from grouse moors. There was no evidence of any temporal decline in poisoning incidents on grouse moors over the study period. This research indicated that illegal methods for controlling predators (including eagles and other protected birds of prey) were associated with grouse moor management.

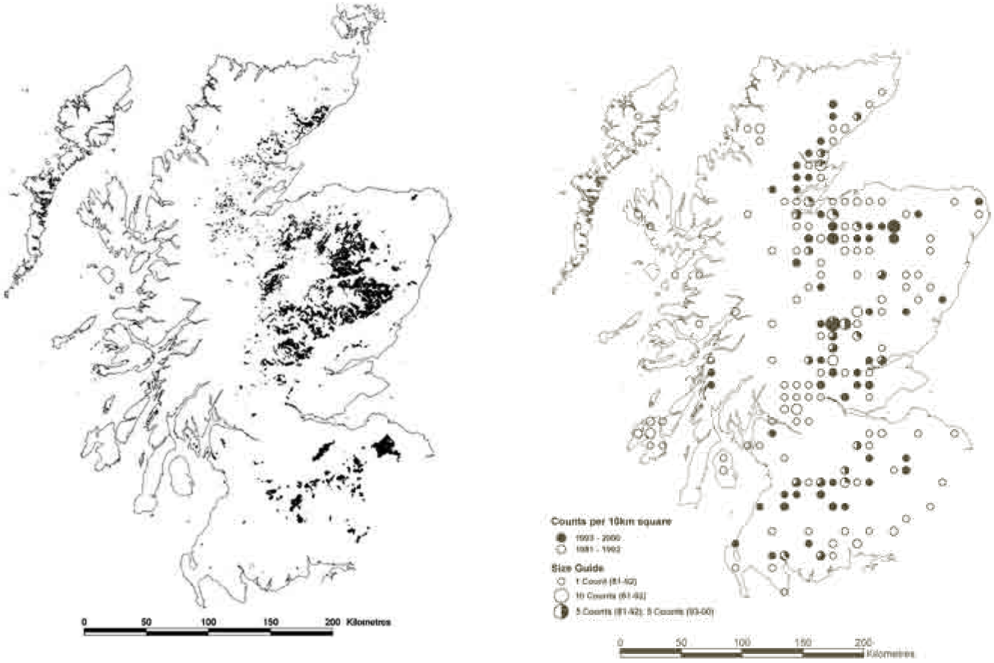

Fig. 5. The distribution of strip muirburn from LCS88, a vegetation type exclusively limited to grouse moor management (left), and the distribution of recorded illegal poisoning incidents in upland vegetation classes 1981-2000 (right) (see Whitfield et al., 2003, for further details).

- In a GIS analysis, utilising data collected during the 1982 and 1992 national censuses of the golden eagle in Scotland and contemporary data on the distribution of poisoning incidents (a common method of raptor persecution in Scotland), Whitfield *et al*. (2004a) showed that persecution was associated with a reduction in the age of first breeding, territory vacancies, and the use of territories by non-breeding immatures. Persecution was therefore reducing adult survival and distribution and was probably creating ecological traps that attract mobile immatures, increasing subadult mortality in birds that may originate from persecution-free areas. Mapping ‘strip muirburn’, a form of land management unique to moors managed for red grouse shooting, where most poisoning occurred, Whitfield *et al*. (2004a) also showed similar associations between golden eagle demography and grouse moor management and concluded that persecution may be having a major impact on the golden eagle population of

Scotland, since it occurs over a wide area and at least in some areas has persisted over many decades.

- Whitfield *et al*. (2004b) used observations of the age structure and breeding productivity of the Scottish population of golden eagles together with the classic theory of population dynamics to derive estimates of life history parameters. Regional differences in age structure associated with differences in persecution intensity were then used to derive estimates of prospective 'persecution-free' life history parameters. The different parameter combinations were entered into a population model to simulate their effects on the number of occupied territories over time. Most simulations suggested that with demographic parameters including persecution effects the population should decline. The disparity between these predictions and the observed apparent stability in occupied territories was ascribed to the buffering effect of a lowering in age of breeding in areas where persecution is most intense and that more favourable parameter estimates within the estimated limits may be more realistic. The results indicated, nevertheless, that currently the population is vulnerable to decline as also suggested by the apparent lack of adults to occupy vacant territories. In the absence of the estimated 3 – 5% annual adult mortality through persecution, modelling suggested the population would increase. Removing estimated effects of persecution on reproductive rate and preadult survival were on their own insufficient to reverse the declines predicted from parameter values which included persecution effects. This may indicate that the effects of persecution on preadult survival are more severe than estimated. In the absence of persecution Whitfield *et al*. (2004b) concluded that the population could expand to fill currently vacant but apparently suitable habitat and have a more secure long-term status.
- Population simulations using a range of demographic parameter values (Whitfield *et al*., 2006) suggested that populations in those zones where grouse moor management and persecution predominated and where there were still sufficient resident pairs to allow modelling (zones 10, 11, and 15) should be expanding based on the observed productivity values. Instead, the populations in these regions remain low or are in decline (see also Annexes 3 & 6). To explain the observed recent trends in the numbers of resident pairs in these regions the survival rates of subadults (= preadults = immatures) and/or adults must be dramatically below those in other zones, and below those considered as acceptable for favourable conservation status, consistent with the analyses and predictions of Whitfield *et al*. (2004a, b). For example, while a 40% survival rate for subadults was considered to be an acceptable 'minimum' by Whitfield *et al*. (2006) in the Cairngorms Massif (zone 11) it may as low as 10% (Annex 3). The low survival rates in zones where grouse moor and persecution predominates are inconsistent with the apparently considerable food and space resources available in these zones but are consistent with eagles being killed, as suggested by other lines of evidence.
- Whitfield *et al*. (2007b) analysed change in the number of occupied territories between the 1992 and 2003 national eagle surveys against a number of potential constraints, including grazing, recreation, conifer afforestation and persecution. They found little evidence to suggest that recreational disturbance was influential on the occupation of golden eagle territories, although some local effects may have occurred and further analyses are warranted. Similarly, evidence suggested that only a limited number of territories had been abandoned recently due to the planting of commercial conifer forests, although several territories have been lost

to forestry in the past in western Scotland. Whitfield *et al*. (2007b) also rejected a hypothesis that changes in territory occupation between national golden eagle censuses were influenced by change in carrion abundance. By contrast, results were consistent with the hypothesis that persecution was influential in the observed change in territory occupation between censuses, so that the number of occupied eagle territories tended to decline where persecution was probably still influential and tended to increase where persecution had probably declined. The four regions (zones 10, 11, 12 and 15) where persecution indices had not changed or had increased were the four Highland regions where grouse moor management is most common (Whitfield *et al.*, 2003). Earlier analyses had suggested that there was no evidence that poisoning incidents had recently decreased in grouse moor areas (Whitfield *et al.*, 2003), and Whitfield *et al*. (2007b) confirmed that there was also no evidence that golden eagle persecution has declined recently in these four regions. The marked decline of eagle pairs in these four regions where there has been no apparent decrease in the intensity of persecution (70 pairs in 1992 down to 55 in 2003: a 21% decline) bears out the predictions of Whitfield *et al.* (2004b). The two regions with the only marked increases in occupied territories in the 2003 census (zones 3 and 5) were two of the three regions which experienced a decline in persecution indices. The third region with a decline in persecution index (zone 14) showed no change in occupied territories between 1992 and 2003. In this region golden eagle persecution (and poisoning: Whitfield *et al.*, 2003) was largely confined to the island of Islay and seemed to decline in the mid-1980s so any positive response in occupied territories would have been more likely between the 1982 and 1992 censuses: this expectation was met, with four and eight occupied territories in 1982 and 1992 respectively.

- Records of illegal persecution of golden eagles (including poisoning, trapping, shooting) were more common in those regions where grouse moor management predominated (Whitfield *et al*., 2007b).

The failure of the Scottish golden eagle population to meet favourable conservation status targets is largely due to the marked failures to meet favourable status in those regions where grouse moor management prevails, and the failure to meet favourable conservation status in these ‘grouse moor’ regions is largely through the continued illegal killing of eagles (Whitfield *et al*. 2006). Persecution is the most serious constraint facing the Scottish golden eagle population, mainly though its effects on adult and subadult survival. Evidence of persecution spans many decades (Whitfield *et al*; 2004 a, b and references therein). Historical (Watson, 1997) and recent (Whitfield *et al*., 2007b) reductions of persecution in some areas of Scotland indicate that golden eagle populations respond positively to policies which are effective in tackling this issue. Not only is the conservation status of the national population being compromised by persecution, but in several eastern Highland and Southern Upland regions there are far fewer eagles than would be expected given the apparently suitable habitats and continued declines and absences from such areas should be a source of great concern. Regional extinctions are even possible. It is also worth highlighting that the rarity of golden eagles in England is probably a result of a combination of the shortage of potential recruits coming from Scotland, due largely to persecution in areas where potential recruits are most likely to originate, and to raptor persecution in upland areas of England (e.g. Brown, 1976; Gibbons *et al*., 1994; Holmes *et al*., 2003)

Persecution should also be viewed with concern because it has many other pervasive effects that may not be immediately obvious. A key concern associated with this constraint is the loss of post-fledgling birds. Once they have left the nest, golden eagles have very few natural predators and so their life history and behavioural traits have largely evolved in the absence of any selection by predators. Killing by humans introduces a novel selective influence and can therefore not only have direct effects on abundance, distribution and dynamics of populations (Novaro *et al*., 2005) but can also influence their genetic makeup (e.g. Coltman *et al*., 2003; Balbontín *et al*., 2005).

The behaviour of animals in populations subjected to persecution can also change; notably they adopt behaviours which minimise the risk of contact with humans (Kitchen *et al*., 2000). This change in behaviour may be either learned or be genetic (e.g. an eagle which flees from a human attempting to kill it is more likely to pass on its genes to subsequent generations than an eagle which does not flee). The golden eagle is well known as being sensitive to disturbance (Watson, 1997) and it is very likely that this is at least partly a result of a long history of persecution. González *et al*. (2006) suggested that the sensitivity of Spanish imperial eagles to incidental disturbance is probably largely because of persecution, and Ferrer *et al*. (1990) have shown that Spanish imperial eagles became less likely to fly away from their nests and more likely to defend their nests when approached by observers after they were protected from persecution. Several species of large raptors which are tolerated or welcomed by humans may nest in close proximity to areas with high levels of human activity when free from persecution (e.g. Bird *et al*., 1996; Millsap *et al*., 2004).

Golden eagles are often ‘top’ predators in their environment and their removal by persecution may also have effects on biodiversity (Sergio *et al*., 2005) and on the guild of smaller predators (Fielding *et al*., 2003c). Reflecting several recent studies on the impacts of large raptor species on smaller species (e.g. Petty *et al*., 2003; Sergio *et al*., 2003) increasing evidence is accumulating that golden eagles can restrict the distribution and abundance of smaller raptors (Poole & Bromley, 1988; Ratcliffe, 1993; Gainzarain *et al*., 2000; Fielding *et al*., 2003c; Sergio *et al*., 2004). Golden eagles are persecuted because of their perceived effects on red grouse shooting bags but Brown & Watson (1964) concluded that “predation by eagles alone can never have a crucial effect on the total numbers of their prey”. A small direct impact of golden eagles on grouse coupled with a suppressive effect on smaller raptors that have a higher impact on grouse (e.g. Redpath & Thirgood, 1997) is a possible example of an effect more thoroughly researched in mammalian carnivore communities (e.g. Palomares *et al*., 1995), whereby prey species benefit numerically from the presence of a top predator. On grouse moor areas of Scotland, the impacts of smaller raptors on red grouse numbers and overall losses of red grouse to raptors may be less in the presence of golden eagles. This is however difficult to study in situ, because most grouse moors do not have an intact raptor guild. Nevertheless, this is an issue worthy of further study, not least because predictions of raptor impacts on red grouse (e.g. Redpath & Thirgood, 1997) and predictions of potential raptor numbers on grouse moor habitats (Potts, 1998) should ideally take into account the potential influence of golden eagles (Thirgood *et al*., 2000; Whitfield *et al*., 2004b).

For further details please refer to Annex 1 (for initial methods and datasets), Annex 2 (notably Table A2-6), Annex 4 (notably Tables A4-3 & A4-14), Annex 5 (for decision tree and neural network modelling), Annexes 6 & 3 (for population model and outputs respectively), and Whitfield *et al*. (2003, 2004a, b, 2006, 2007b).

5.7 Unintentional disturbance

Anderson *et al*. (1990), studying golden eagles in Colorado, have shown eagles may avoid areas of human activity with major effects on home range use and size. Holmes *et al*. (1993) found that 90% of perched wintering birds flushed at 300m distance and Schueck & Marzluff (2001) have also documented that golden eagles may avoid periods or locations of human activity.

Several authors working on large eagles have suggested that to avoid adverse disturbance effects, various human activities, from recreation to forestry operations, should be restricted or avoided completely at 400 - 1500 m around nest sites (disturbance free zones: Stalmaster & Newman, 1978; Fraser *et al.*, 1985; and see Table 5).

Table 5. Some examples of disturbance-free zones around nest sites suggested to avoid breeding failure and territory loss in large eagles due to unintentional human disturbance.

Species	Disturbance-free zone	Reference
Golden eagle	1000 m	Petty (1998)
Golden eagle	750 – 1500 m	McGrady *et al.* (1997)
Golden eagle	800 m	Richardson & Miller (1997)
Spanish imperial eagle	500 m	González *et al.* (1992, 2006)
Bald eagle	400 – 800 m	Anthony & Isaacs (1989)

Most studies stress that disturbance is greatest when activities are in direct line-of-sight from the nest, and this is probably why the suggested disturbance-free zones are greatest for the golden eagle as this species typically breeds in more open areas and has more elevated nest sites than the other two eagle species considered here. When potential disturbance is in direct line-of-sight of nests the suggested mitigation has been to increase the disturbance-free zone (Anthony & Isaacs, 1989; McGrady *et al.*, 1997) or the shielding of human activities, by tree-planting along tracks, for example (Andrew & Mosher, 1982; Gonzalez *et al.*, 1992). Brendel *et al*. (2002) describe a more sophisticated approach to avoiding recreational disturbance, which may be more successful than simply attempting to exclude people from buffer zones.

Watson & Dennis (1992) analysed data from 335 nest sites collected during the 1982 UK national survey and found that sites classified as having easy access were more likely to fail than sites without easy access. Failure may have been affected by factors other than unintentional disturbance, but the authors considered that unintentional disturbance was at least a contributory influence. Also, nest sites with some evidence of disturbance (including unintentional disturbance) were more likely to fail than nests with no evidence of disturbance. By contrast, however, Watson & Dennis (1992) found no evidence that proximity to public roads increased the chance of nest failure, but suggested that all but the most inaccessible nests close to roads may be abandoned relatively quickly, precluding the detection of depressed breeding success. Several nest sites close to public roads have been abandoned in Scotland since the 1950s (Watson & Dennis,

1992) and the abandonment of some eagle territories in the northern Cairngorms has been attributed to increased recreational activity in the area following construction of a skiing development (Watson, 1981).

Golden eagles therefore are potentially sensitive to unintentional disturbance at nesting sites and might be excluded from foraging areas if they avoid areas of human activity. Any expansion of human activity in areas used by eagles may thus unintentionally create problems and act as a constraint. Recreation, primarily involving hillwalkers, is the most likely source of such disturbance, although the activities of land managers, such as farmers, may also unintentionally create problems.

Our analyses on this potential constraint involved the use of digital data on locations of human habitation, roads/tracks and mountain peaks which are popular with hillwalkers.

The difficulty with an analysis (e.g. Watson & Dennis, 1992) examining territory occupation and breeding success in relation to proximity to roads is that it does not necessarily differentiate between the different types of disturbance which may be associated with road proximity, and that proximity to a road may also reflect other features of a territory, for example topography and proximity to agriculture. Our studies found that nationally, vacant territories encompassed a greater road length than occupied territories (Annex 4: Table A4-7) but this could have been a further reflection of a more generic finding documenting greater occurrence of vacancies in territories at the periphery of the geographical range of golden eagles (see section on 'agricultural encroachment'). Also, in zone 14 (Argyll West and Islands) which made a major contribution to the national result on roads (Table A4-7), the greater occurrence of roads in vacant territories could have been due to the road infrastructure necessary to service commercial forestry, and it could have been the effects of the forest plantations and not the roads which was more influential.

Due to the difficulties in interpreting the use of road proximity or extent as surrogates for disturbance, we also examined features of territories in relation to the distribution of 'Munros'. In Scotland most recreational activity which could lead to casual disturbance of eagles is due to hillwalkers (e.g. Hall, 2002). Much hillwalking in Scotland is due to people visiting a series of over 280 mountain peaks colloquially called 'Munros' (summits above 3000 ft ASL) (Bennet *et al.*, 1991).

At a national scale the evidence was equivocal for an effect of disturbance arising from the recreational use of Munro mountains (Annex 4: Tables A4-10 to A4-12). This is unsurprising given the simplicity of the analyses and that any effect of recreation is likely to be highly locally specific and therefore difficult to analyse and detect at higher spatial scales. More critical analyses therefore need additional data about relative levels of Munro usage, access routes and eagle nest site use. The results do tend to indicate, however, that any effect of recreation, at least in relation to the most popular sources of hillwalking, probably does not constitute a serious national issue. In regional analyses, there was statistically significant evidence that range vacancy was associated with the presence of Munros in only on region, the Northern Highlands (zone 7; Annex 4: Table A4-13)[3].

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[3] This was also the only region where there was a significant negative relationship between proximity to human habitation and range occupancy (Table A4-6).

Whitfield *et al*. (2007b) examined the potential influence of several constraints on change in territory occupancy between the 1992 and 2003 national surveys and found no consistent evidence for eagle territory abandonment to be associated significantly with centres of recreational activity (i.e. the locations of Munro mountains). Local influences on a small number of territories are quite likely to have occurred, however (e.g. Watson & Dennis, 1992). Whitfield *et al*. (2007b) suggested that such influences are probably better examined on a different scale and with more specific methods (for example, using measures of visibility from eagle nests to paths or tracks used for recreation, rather than simple proximity to centres of recreation). Examination of territory abandonment between the 1982 and 1992 national eagle censuses in relation to recreation activity may also deserve further research because most indices of recreational activity suggest that there was a marked increase in the number of visitors in the 1960s with a further sharp upturn in the 1980s before a levelling off in activity in the 1990s, a pattern confirmed by the few site specific studies (e.g. Hall, 2002).

McLeod *et al*. (unpublished) have made preliminary analyses of site-specific patterns of territory abandonment in relation to hillwalking path proximity and again found no consistent evidence of recreation effects: whereas several cases of abandonment appeared to be associated with the close proximity of a hillwalking path, there were also several examples in the same area where close path proximity was associated with a prolonged period of eagle occupancy and high breeding productivity. Proximity to ‘Corbetts’ (hill tops between 2500 and 3000 ft ASL) was also examined with no consistent evidence for effects. Whitfield *et al*. (2006) suggested, on the basis of these analyses, that recreation may be an issue in some regions (notably in zone 4, North West Seaboard, where there were some non-significant indications of effects), but it is worth emphasising that in only zone 7 (Northern Highlands) was there significant evidence of some effect (Table A4-13). A. Watson *et al*. (in prep.) have noted that in northeast Scotland, where numbers of occupied territories have continued to decline, increasing recreational pressure associated with the Cairngorms National Park may present difficulties for the relatively few occupied territories that remain. Similarly, Fielding *et al*. (2006) and Whitfield *et al*. (2006) have indicated how the effects of persecution in eastern Highland regions may place those remaining occupied territories under disproportionate pressure from other constraints. It is important to bear in mind, nevertheless, that in those regions where persecution is influential, even when recreation or other constraints may cause difficulties for a small number of the relatively few territories which remain (and such difficulties therefore warrant appropriate management), attention should not deflect from the ultimate regional problem - that of too few eagles to occupy more territories (see also a similar discussion of prioritising management in another eagle species by Ferrer & Hiraldo, 1991).

Although further analyses are clearly justified, the difficulties in analysing the effects of unintentional disturbance on golden eagles in Scotland are several and include the difficulties in obtaining data on potentially relevant factors. The form, frequency, proximity and intensity of disturbance events and the sensitivity of the birds will all potentially contribute to the response of birds and hence the impact of disturbance (e.g. González *et al*., 2006). González *et al*. (2006), for example, found that hikers caused far less disturbance to nesting Spanish imperial eagles than most other forms of human activity (such as hunters, shepherds and ecotourists) which is of interest in the context of the majority potential influence on Scottish golden eagles. Any effects may also be strongly locally specific, and there are always difficulties in obtaining sufficient sample sizes to demonstrate disturbance influences on populations of large vertebrates like eagles (Delaney *et al*., 1999).

Unintentional disturbance is likely to be viewed by animals as a form of predation risk (Frid & Dill, 2002). The sensitivity of eagles to disturbance will thus be a function of their perception of humans as potential predators and, as noted in the previous section, a history and continued occurrence of persecution in Scottish golden eagles has almost certainly contributed to their sensitivity to disturbance (see Ferrer *et al*., 1990). In the long-term absence of persecution, therefore, we might reasonably expect two changes in Scottish eagles which would mitigate against any effects of unintentional disturbance.

First, in the absence of any selection or experience of humans having a detrimental effect, golden eagles should become more tolerant of human activity. Bald eagles *Haliaeetus leucocephalus* are sensitive to human disturbance in several areas such as northwest USA (e.g. Anthony & Isaacs, 1989), yet in other areas such as eastern USA they can be relatively insensitive and nest in close proximity to high levels of human activity (e.g. Millsap *et al*., 2004). Second, territories subject to incidental disturbance may be viewed by eagles as 'sub-optimal' and so when a population is low or in decline such sub-optimal territories may be abandoned before 'optimal' territories i.e. there are insufficient birds to fill all territories and so birds occupy the best territories. However, when the population is high or expands we should expect such sub-optimal territories to be re-occupied as for some birds occupying a sub-optimal territory is better than not occupying any territory. When a lack of sufficient birds is caused by persecution, as appears to be the case across much of Scotland, then the key influence on territory occupation is persecution, and the influence on territory optimality/sub-optimality (incidental disturbance, in this example) is secondary. Haller (1982, 1994) studied the Alpine golden eagle population which was seriously depleted by persecution at the beginning of the 20th century. Increasing protection and tolerance, however, led to all high altitude areas being occupied by the 1980s and as the population continued to expand, low altitude areas in the foothills, close to areas of human activity and disturbance, were then occupied. The recent increases in Scottish golden eagle pairs in the Western Isles documented in the 2003 national survey have also illustrated how 'peripheral' territories can be re-occupied when a population expands after a relaxation in persecution (Whitfield *et al*., 2007b).

For further details refer to Annex 1 on datasets, Annex 4 (notably Tables A4-6, A4-7, A4-10 – A4-13), and Whitfield *et al*. (2007b).

5.8 Wind farms

Despite their environmental benefits in generating electricity without emission of 'greenhouse' gases, wind farms have attracted controversy with regard to their impacts on birds, especially golden eagles. The two main potential negative effects of wind farms on birds are displacement from the wind farm area through disturbance (effectively equivalent to habitat loss) and fatality through strikes with turbine blades. Evidence from studies in the USA suggests that golden eagle fatalities through collision with turbines may be the main potential impact (e.g. Erickson *et al*., 2001; Smallwood & Thelander, 2004) whereas for breeding golden eagles in Scotland, displacement from wind farm areas (indirect habitat loss) may be the primary impact (Walker *et al*., 2005). The recent expansion of onshore wind energy developments in Scotland could therefore be a potential constraint for golden eagles.

Our studies on this constraint required the digitisation of all known wind farm developments in Scotland in all stages of development, from pre-scoping proposals to constructed facilities, using methods summarised in Annex 1 (‘Wind farms’) and detailed in Fielding *et al*. (2006). We used the 2003 national survey to document the distribution of occupied eagle territories and several techniques, building on previous work, to document areas that were likely to be occupied by non-breeding golden eagles (Fielding *et al*., 2006).

In our study, taking into account all known wind farm proposals to January 2005, Fielding *et al*. (2006) examined the potential for co-occurrence of golden eagles and wind farms in Scotland by documenting the spatial association between wind farm proposals and breeding eagle territories and areas potentially suitable for non-breeding eagles. Although there were records for over 500 wind farm proposals at various stages of development, relatively few coincided with eagle territories (only *c.* 4% of territories had a proposal within 3 km of territory centre). Similarly, only 2% of habitat predicted to be suitable for non-breeding eagles overlapped with proposed or installed wind farm areas. Moreover, estimates of the potential for electricity generation from all wind farm proposals, with respect to government targets for renewable energy supplies, suggested most proposals were unlikely to be constructed. Fielding *et al*. (2006) concluded that in comparison with other constraints on Scotland’s golden eagles, notably persecution, wind farms should not represent a serious concern if best practice in planning their location and minimising their impact are maintained. Potential future regional pressures on breeding eagles from wind farms were highlighted (in zone 5 and the mainland part of zone 14), however, and because of the uncertainties of the impacts of wind farms on eagles with respect to displacement or collision fatalities, the situation requires continued scrutiny and appropriate monitoring to be put in place at wind farm developments where there is a risk of adverse effects on golden eagles.

For further details see Annex 1 (‘Wind farms) and Fielding *et al*. (2006).

5.9 Competition with white-tailed eagles

Fig. 6. The skull of a female white-tailed eagle recovered in south Lewis with talon puncture holes whose form matched those of golden eagle talons. Prior to her death, this female was repeatedly seen interacting with a pair of golden eagles.

Following a number of phases of reintroduction, beginning in 1975, the white-tailed eagle has been successfully reintroduced into Scotland (Love, 1983; Evans *et al*., 2002; Bainbridge *et al*., 2003). Currently the population is over 30 breeding pairs (Etheridge *et al.* 2006) and in a phase of marked expansion (Bainbridge *et al*., 2003). It has been suggested that the continued spread of the white-tailed eagle in Scotland will lead to displacement of golden eagles through competitive effects as coastal ranges are reclaimed (Watson *et al*., 1992b; Watson, 1997; Halley & Gjershaug, 1998; Halley, 1998). On the Isle of Mull, however, where both species occur at relatively high density there was little evidence of competitive effects (Whitfield *et al*., 2002) and other work has recently challenged the basis for earlier predictions of competitive effects, such as dietary overlap (Madders & Marquiss, 2003), purported historical evidence for competition and a greater digestive efficiency of white-tailed eagles (Whitfield *et al*., 2002). Both species occupy the nest sites of the other species (Willgohs, 1961; Bergo, 1987; Watson, 1997; Crane & Nellist, 1999; Whitfield *et al*., 2002; Love, 2003) and Fielding *et al*. (2003c) characterised the relationship between the two species in Scotland as one of 'armed neutrality' in that whilst both species may occasionally kill each other (for example, Fig. 6), this did not appear to have a predatory basis.

Comparisons of the 'food niche' of white-tailed and golden eagles in Scotland have tended to concentrate on examining the extent of dietary overlap, with no overt reference to spatial partitioning, for example (Watson *et al*., 1992b; although see Madders & Marquiss, 2003). In other words, the assumption has been made that a high dietary overlap implies a high level of competition. This is probably too simplistic (Wiens, 1989) and even when apparent resource use is very similar, competition may be minimal (Katzner *et al*., 2003). Whitfield *et al*. (2002) suggested that whilst there may be some dietary overlap between white-tailed and golden eagles in Scotland, if the common dietary components are obtained in different locations then this would further serve to mediate any competitive effects. In western Norway, golden eagles tend to be found more often at higher elevation cliff nest sites whereas white-tailed eagles are more closely tied to coastal sites and tree nests (Halley, 1998) and across Europe the white-tailed eagle is more frequently found in lowland areas whereas the golden eagle is characteristically an upland species (e.g. McGrady, 1997; Helander *et al*., 2003).

Not surprisingly, therefore, Evans *et al*. (unpublished) have shown that in Scotland white-tailed eagles occur at lower elevations and nearer open water than golden eagles and are more likely to use tree nest sites (see also Nellist & Crane, 2001). Unpublished observations of a single white-tailed eagle range and several neighbouring golden eagle ranges on the Isle of Lewis (in Western Isles) confirm that the white-tailed eagle pair had very different range use to golden eagles, with a strong attachment to the coast whereas most golden eagle ranging was terrestrial and at higher elevations (R. Reid & P. Haworth, unpublished data). To a degree, recent dietary studies have tended to confirm this differentiation between the species, with a stronger emphasis on aquatic food sources and a broader dietary spectrum in the white-tailed eagle (perhaps reflecting the availability of a greater range of food sources at lower elevations at the aquatic/terrestrial boundary; Madders & Marquiss, 2003). Hence, there is probably a fairly large degree of niche separation and any interactions between the two species, at least currently, seem to have little more than a local short-term effect.

The absence of any discernible effect of the expansion of white-tailed eagle numbers on golden eagles on the Isle of Mull (Whitfield *et al*., 2002) seems to have been reflected by the more recent expansions of white-tails on the Isle of Skye and the Western Isles: areas where numbers of golden eagles have remained stable or expanded, respectively (Eaton *et al*., 2007). Consequently there is little basis, at least currently, for concluding that the continued expansion of the white-tailed eagle represents a potential constraint for the golden eagle. As noted by Whitfield *et al*. (2002), however, continued studies need to be alert to this situation changing as the white-tailed eagle population expands, especially regarding competition over nest sites where tree nest sites for white-tails may be limited.

5.10 Native woodland expansion

While management for the expansion of native woodland has clear biodiversity benefits in areas like the uplands of Scotland where historical removal of native woodlands has occurred by redressing these historical losses of more extensive woodland cover, native woodland expansion may have similar effects on golden eagles to those described for exotic woodlands (Whitfield, 2000). Although golden eagles can be found breeding successfully and productively in areas where native woodlands predominate, densities tend to be lower and unwooded areas or areas with an open woodland structure are primarily used for feeding (Tjernberg, 1985; Pedrini & Sergio, 2001; McGrady *et al*., 2003, 2004; Takeuchi *et al*., 2006).

There are aspirations to expand native woodland cover in most regions of Scotland where golden eagles breed (e.g. SNH, 2002). The potential effect of new native woodland on golden eagles has been considered at length by McGrady *et al*. (2003, 2004) who conclude essentially that the effects of new native woodland are likely to be similar to those of commercial woodland (see also Whitfield, 2000), although some additional opportunities are probably available under native woodland grant schemes. Golden eagles require open habitat so the Forestry Commission's Scottish Forestry

Grant Scheme (SFGS), which only allows up to 20 % of the area to be managed as open habitat is not considered 'eagle-friendly' (McGrady *et al*., 2003, 2004).

As for new coniferous plantations, the influence of new native woodland will depend on its extent and location, the characteristics of the eagle territory (or non-breeding area), and on the management, both of the remaining open areas and of the woodland-open vegetation interface. McGrady *et al*. (1997, 2003, 2004) and Petty (1998) give guidelines on designing and managing new woodlands to benefit eagles and minimise adverse effects: notably the retention of substantial open areas to maximise the abundance/ availability of prey. Additional insights from other studies (McLeod *et al*., 2002b; Whitfield *et al*., 2001, 2007a) suggest that to minimise adverse impacts and maximise the potential benefits of new woodland for territorial eagles, it should be planned at lower elevations and away from slopes, ridges and the centre of territories (simple rules such as assuming that a particular extent of tree cover within set distances of the territory centre will or will not have an adverse effect are inadequate). Detrimental effects are most likely in territories where there is little capability for the resident pair to respond by shifting range use; where ranging is potentially constrained by surrounding unsuitable land, which would include neighbouring pairs, and there are limited opportunities for establishing alternative nest sites. Territory abandonment is most likely in pairs with a low breeding success, as inappropriate woodland planting can result in reduced breeding success in all pairs. If a pair already has low breeding success then a further reduction in the capability for successful breeding can mean it is not worth the pair occupying the territory. At the population level, the most 'valuable' pairs or territories are those which contribute most to the population's productivity (Sergio & Newton, 2003), and in areas where the numbers of pairs are well below the potential (e.g. eastern Highlands) particular care needs to be taken if new forests are planned in the remaining territories. For example, as noted by A. Watson *et al*. (in prep.) the expansion of native pinewoods in the Cairngorms needs to take careful account of the needs of the few pairs of eagles which remain in the area.

Due to the thorough review provided by McGrady *et al*. (2003, 2004) and the current low occurrence of native woodlands in areas which may also be suitable for golden eagles, we did not examine this issue beyond simple exploratory analyses which did not reveal any substantial difficulties (Table A4-3).

5.11 Other constraints

The conservation framework for the golden eagle was always intended to be a dynamic process (Watson & Whitfield, 2002), and consequently revisions and updates should be considered as both desirable and inevitable. In this light, the relative influence and importance of different constraints may change over time (hopefully, in some cases, because policies are enacted to tackle and reduce their influence) and new constraints may become evident.

Potential 'novel' constraints which may deserve more attention in the future would include increased removal of carrion and the extensive culling of large numbers of mountain hares on several Highland grouse moor estates (P. Stirling-Aird, B. Etheridge, N. McDonald, pers. comm.). Increased removal of carrion in the form of sheep and/or deer carcasses may result from: management aimed at indirect control over numbers of carrion feeders such as ravens *Corvus corax*, in response to complaints from the public; or as a result of the need for improved standards in carcass removal following EC

Regulation 1774/2002 on Animal By-Products, introduced to reduce the risk of pathogen transmission from dead livestock. The latter is also causing concern in Spain, for example, where traditional carcass dumping pits - 'muladares' - used by several scavenging birds of prey, are threatened. The reason for culling of hares seems to be research suggesting that mountain hares are a potential reservoir for the tick-borne louping-ill disease which can infect red grouse and affect their numbers (Laurenson *et al*., 2003); the management response crudely attempts to remove the hare 'reservoir'. With hares being such an important prey species, any serious reductions will obviously have an impact on golden eagles and/or deflect more eagle predation on to alternative prey, notably grouse. It is not clear, however, whether golden eagles are still present on grouse moors where hare culls have taken place: an immediate requirement would appear to be documentation of the areas where culls have been undertaken.

Although the golden eagle has successfully adapted to a wide range of climatic conditions, breeding from the Arctic to northern Africa (Watson, 1997), further attention to the possible effects of climate change may be beneficial to interpretation of potential future versions of the conservation framework. Effects of weather on golden eagle reproduction have been documented in the USA (Steenhof *et al*., 1997) and western Scotland (Watson *et al*., 2003). Whitfield et al. (2007b) and A. Fielding (unpublished) have suggested that climate change is most likely to affect Scottish eagles in the oceanic-influenced western Highlands and Islands, but further research on how weather variables may influence golden eagles in Scotland would be beneficial.

Death through collision with power transmission wires is not generally considered a problem in Scotland but limited evidence suggests that it may be a greater source of post-fledging mortality than is usually acknowledged (Watson, 1997). Certainly it is an issue deserving of further study, and initial attention should focus in low altitude eagle ranges, such as on western coastlines or Hebridean islands, where transmission wires may be more frequent. Proposals for new transmission wires in areas heavily used by eagles should be viewed with caution.

Similarly, whilst electrocution of large eagles through perching on power poles has been found to be a major source of mortality in several countries, notably the USA and Spain (e.g. Boeker & Nickerson, 1975; Ferrer *et al.*, 1991), in Scotland such deaths appear to be rare (Watson, 1997). Spanish power poles are more likely to be metal and so electrocution risk is greater. Nevertheless, pole and transmission wire modifications and designs that reduce electrocution risk (Nelson & Nelson, 1977; Olendorff *et al.*, 1981; Ferrer *et al.*, 1991) are available so that this problem can be tackled or avoided in the future. The apparent rarity of electrocution mortality in Scotland may be because there are few power lines in eagle territories (although see earlier) and/or a high availability of natural perches where lines occur (Watson, 1997) and/or the predominant designs of poles in Scotland are safe for raptors. As golden eagles appear to be vulnerable to electrocution, however, this is an issue that deserves further research in Scotland (Watson, 1997).

5.12 Constraints: conclusions

Hopefully, it should be apparent that we did not unduly or subjectively concentrate attention on some constraints over others which were not justified by initial exploratory analyses. As well as the many research findings presented in the six Annexes to this report, and the peer-reviewed publications of the conservation framework (Watson &

Whitfield, 2002; Whitfield *et al*., 2006), there have been several publications in the peer-reviewed literature on constraints: commercial afforestation (Whitfield *et al*., 2001, 2007a, b; Fielding *et al*., 2003b), grazing animals (Whitfield *et al*., 2007b), persecution (Whitfield *et al*., 2003, 2004a, b, 2007b), unintentional disturbance through recreation (Whitfield *et al*., 2007b), wind farms (Fielding *et al*., 2006), competition with white-tailed eagles (Whitfield *et al*., 2002), and climate (Watson *et al*., 2003).

As a reminder, we have summarised the results of the FCS tests (section 5) in Fig. 7.

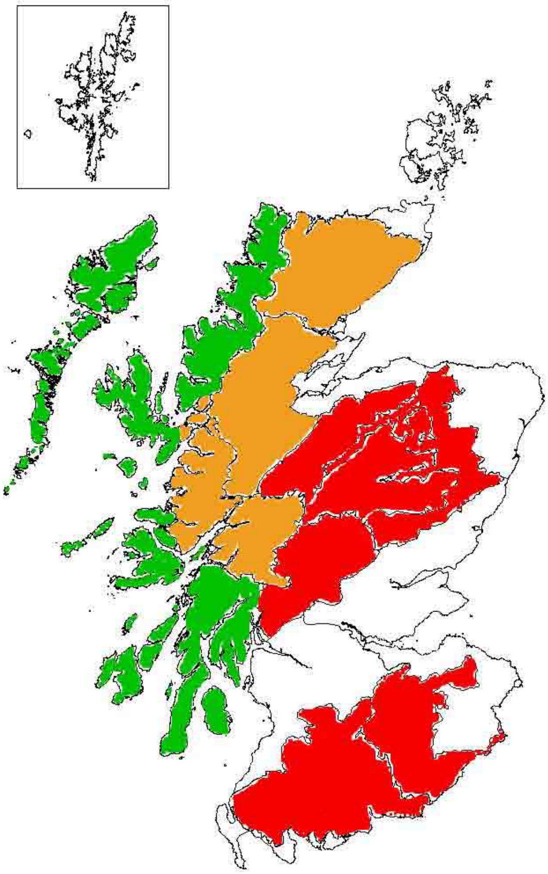

Fig. 7. Summary of the conservation status of the golden eagle in Scotland in 2003 (see also Table 4). Green = region in favourable conservation status (note North West Seaboard classed as favourable here, because Level 1 test failure was marginal), Amber = region in unfavourable conservation status, but failure in only one test, Red = region in unfavourable conservation status, with failure in more than one test.

The results of various studies on constraints are summarised by Whitfield *et al*. (2006) who give a regional breakdown of their likely importance (Table 6). Though not necessarily exhaustive, or complete, the results are similar to the likely regional constraint influences predicted by Watson & Whitfield (2002).

Linking the analyses of constraint influence with the analyses and assessment of favourable conservation status, it is apparent that the most serious constraint is persecution because of the scale, severity and form of its influence (Table 6 & Fig. 7). A lower priority nationally, but of importance in some western regions, is a shortage of live prey. This issue, which is likely to be the result of heavy grazing by red deer and sheep, combined with excessive burning regimes, deserves further research.

Table 6. Summary of favourable conservation status (FCS) and likely constraints on FCS in Scottish golden eagles according to region (from Whitfield et al., 2006).

NHZ	NHZ name	FCS?	Constraints
3	Western Isles	Yes	Sheep > deer > persecution
4	North West Seaboard	No	Deer > recreation > persecution
5	Peatlands of Caithness & Sutherland	No	Persecution > sheep > wind farms > nest sites
6	Western Seaboard	Yes	Sheep > deer > afforestation
7	Northern Highlands	No	Deer > persecution > recreation > afforestation
8	Western Highlands	No	Deer > sheep > afforestation
10	Central Highlands	No	Persecution > deer
11	Cairngorms Massif	No	Persecution > deer
12	North East Glens	No	Persecution > deer > sheep
13	Lochaber	No	Deer > sheep > afforestation > recreation
14	Argyll West & Islands	Yes	Sheep > afforestation > wind farms
15	Breadalbane & East Argyll	No	Persecution > deer > sheep
19	Western Southern Uplands & Inner Solway	No	Persecution > sheep > afforestation
20	Border Hills	No	Persecution > sheep > nest sites

The differences between regions in the frequency of territorial mergers (Table 3) is potentially revealing of differences in constraint influence. Although merger may occur simply because a neighbouring territory is vacant due to insufficient birds being available to occupy all territories, there may be spatial limits to its frequency because breeding eagles are ‘central place foragers’ (Fielding *et al*., 2003b). In eastern regions, where poor survival through persecution appears to have created a shortfall in adult availability, territorial mergers were infrequent. On the other hand, mergers may occur because of a reduced carrying capacity of the landscape either through habitat loss or degradation of prey supplies through extractive land-use practices (Whitfield *et al*., 2007a). The greater prevalence of mergers in western regions where such constraints were implicated is consistent with this explanation and that constraints differ in their influence between eastern and western regions.

Although the causes of failing to reach favourable conservation status targets, and thus constraint influence, apparently varies between eastern (and southern) and some western regions, the relative importance of addressing these constraints is clear, given the relative disparities between current status and target status, and regional population trends. If the low level of territory occupancies and low survival in regions of southern Scotland and, especially, eastern Highlands are not reversed then the golden eagle population of Scotland will continue to fail national favourable conservation status targets. Indeed, in some regions, extinction as a breeding species threatens. The poor status in the eastern Highlands, and the cause of this poor status - persecution, has been prevalent for many decades. The ability of golden eagle populations to respond favourably to a reduction or removal of persecution has been shown historically (e.g. Watson, 1997) and recently (Whitfield *et al*., 2007b) and so suggests that recovery in the eastern Highlands can occur should the constraint be successfully addressed. The scale of persecution, however, as shown by the geographical distribution of persecution incidents (Whitfield *et al*., 2003), the geographical extent of vacant territories (Whitfield *et al*., 2004a) and (reflecting this) the geographical extent of the regions failing to be in favourable status, is potentially considerable in relation to the relatively few active territories in the affected areas. Hence, given the potentially wide dispersal of young eagles prior to settling on breeding territories, so that birds fledged considerable distance away may be drawn into ‘black holes’ or ‘ecological traps’ (Grant & McGrady, 1999; Delibes *et al*., 2001; Whitfield *et al*., 2004a) in otherwise attractive habitat, even if persecution is reduced to relatively restricted areas, recovery may still take a prolonged period.

6. IMPLICATIONS FOR MANAGEMENT

The highest priority for management and political initiatives is tackling persecution. As noted previously, the failure of the national golden eagle population to meet favourable conservation status targets is largely due to the marked failures to meet favourable status in several eastern Highland regions: the Central Highlands, Cairngorms Massif, Northeast Glens, Breadalbane & East Argyll. Several lines of evidence indicate that the failure of golden eagle populations to meet favourable conservation status in these regions is largely due to the continued illegal killing of eagles which is associated with grouse moor management. Not only is the conservation status of the national population being compromised by persecution, but in several eastern Highlands and Southern Upland regions there are far fewer eagles than would be expected based on the availability of apparently suitable habitat. Continued declines and absences from such areas should be a source of great concern.

There are welcome recent signs of a reduction in persecution in some regions (Western Isles, eastern Caithness & Sutherland) which have been reflected in recovery of resident pairs of eagles. There is however no evidence of a reduction in persecution in the Central Highlands, Cairngorms Massif, Northeast Glens and Breadalbane & East Argyll. In these regions, illegal persecution associated with grouse moor management has been highlighted as problematic for golden eagles over many years by many studies. Here, the golden eagle population continues to decline and even if the remaining territories are now on ground where eagles are tolerated or welcomed, there is concern that young eagles dispersing away from these areas are being killed. In the absence of sufficient recruits even those territories on ‘safe’ ground are liable to disappear.

A secondary national priority should be management to encourage greater abundance and availability of live prey in two western regions: Western Highlands and Lochaber. The most likely constraint here is heavy grazing by red deer and sheep combined with excessive burning. The most practical solution, at least initially, would be targeted reductions in deer and/or sheep in areas where the most benefit to the regional eagle populations would accrue (i.e. in sufficient numbers of territories with relatively good productivity but where there is potential for productivity to be increased further). Existing studies would suggest that such reductions may create a more or less immediate reduction in carrion availability but vegetation recovery and live prey increases would only occur after a greater time lag. Because of the importance of carrion as a food source for eagles, depending on the areas selected for management, there may be a need for short-term food supplementation, to bridge the ‘gap’. Management might include the creation of rabbit warrens (which may also bring long-term benefits) or the introduction of other live prey species. Introduction of mountain hares or red grouse may not be successful if vegetation has not yet recovered from unfavourable conditions induced by large ungulate grazing and burning; but as noted in section 6.5 the competitive relationship between hares and sheep/deer may be different to the competitive relationship between grouse and sheep/deer. But we would strongly recommend (see below) that due to the complexity of relationships which may be influencing food availability in western regions, further research to underpin future management options should be carried out.

While these two key constraints and the respective regions where their influence is greatest are the most pressing issues requiring action (because favourable conservation status is most obviously being compromised), this does not mean that other constraints and other regions should be ignored. For example, the Northern Highlands region may be on the verge of unfavourable demographic status due to a number of factors which potentially require a range of management initiatives (Table 6). More generally, as a further example, there should also be continued scrutiny of wind farm proposals which may affect eagles, and appropriate monitoring of existing and new developments, especially in mainland Argyll and Caithness & Sutherland, where pressures on eagles are liable to build. Better and more explicit linkage between policies for native woodland expansion and those for golden eagle populations (and other open country species) would also undoubtedly bring benefits.

7. OPPORTUNITIES AND RECOMMENDATIONS FOR FURTHER WORK

There are several areas where further work would be highly desirable, many of which have been highlighted in previous sections. Further studies of the impacts and role of unintentional disturbance are needed (as are, arguably, a greater degree of monitoring and documentation of temporal and spatial patterns in recreational activity, such as those described by Taylor & MacGregor, 1999).

The relationships between grazing, burning, carrion, live prey, eagle diet, breeding density and breeding success need further exploration to refine management prescriptions: relevant data are probably already being collected in relation to the management of the Cuillins SPA and on an estate on south Lewis (K. Crane, K. Nellist, R. Reid, P. Haworth & A. Fielding, unpublished). Pilot field demonstrations of the consequences of changes in management prescriptions would be invaluable in situations where background data have already been collected.

Studies of territory occupation, breeding success, timing of breeding failure and the relative contributions to regional productivity, together with (ideally) pilot experimental work on food provisioning, would also be beneficial to guide restorative processes in those regions where a shortage of live prey appears to be an issue. As weather and prey can interact to affect golden eagle reproduction (Steenhof *et al*., 1997) combining research on prey with weather influences would also probably assist understanding.

Studies of the effects of golden eagles on other raptors and on the effects of raptor predation on prey supplies in relation to different raptor guild compositions would also be extremely useful.

The existing golden eagle conservation framework is primarily based on data on territorial adults and the outcome of breeding attempts, as these are easiest to locate and monitor. Subadults are also important components of eagle populations (e.g. Hunt, 2002) and essential targets for conservation (Ferrer, 1993; Real & Mañosa, 1997; Penteriani *et al*., 2005a, b). Knowledge of subadult ecology is scant in Scotland (Grant & McGrady, 1999) and the framework would be enhanced with more explicit consideration of this life history phase based on improved knowledge of movements and survival rates.

Considerable benefits would also result from a programme which monitors adult survival, so that direct measures of all three of the main demographic parameters used in population modelling are available. The most cost-effective method would probably involve genetic profiling of cast feathers using microsatellite markers (e.g. Marsden *et al*., 2003; Rudnick *et al*., 2005). This, in turn, would allow more explicit incorporation of source-sink population processes in the framework.

Monitoring is vital to the success of any conservation framework and Scotland is fortunate in having a network of dedicated and skilled volunteer observers who are typically members of the Scottish Raptor Study Groups (SRSG). The formation of a Scottish Raptor Monitoring Group (Galbraith *et al*., 2003; Anon., 2002) to share raptor information and make it available for policy purposes and practical action on the ground is also an important recent development which helps the golden eagle conservation framework. In this context, it would be helpful to integrate analyses of monitoring requirements (Fielding *et al*. 2003a) with the availability and coverage of annual monitoring efforts from SRSG members.

Clearly, continued monitoring of the influence of known constraints and examination of ‘novel’ constraints is also highly desirable. Of the ‘known’ constraints: persecution, because of the extent and scale of its influences; wind farms, because of the scale and speed of their spread; and new native woodlands, because of the requirement to reconcile the need for more native woodland cover with the need to minimise effects on species that rely on open ground, are perhaps most noteworthy in this respect.

Monitoring is also desirable in order to provide feedback on any policy initiatives which attempt to tackle the role of constraints: a vital component of the conservation framework is the link between the results of research and policy initiatives with the objective being a prioritised policy framework. Watson & Whitfield (2002) suggested that 'conservation policies' should be taken to include the agreed set of advice, prescriptions and incentives to be followed during the process of decision-making by government and its agents. They proposed that such policies are generally amenable to targeting on a geographical basis across Scotland, giving the opportunity to adjust the priority attached to a range of prescriptions or incentives, dependent on the anticipated effect on the overall goal of achieving or maintaining favourable conservation status for the golden eagle population. Information is now available to allow such a prioritised policy framework to be put in place and acted upon.

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ANNEX 1: GENERAL METHODS

NATIONAL CENSUSES AND GOLDEN EAGLE DATA

Field methods

National censuses

We used the results of the three national censuses of golden eagles in Scotland, from 1982 (Dennis *et al.*, 1984), 1992 (Green, 1996) and 2003 (Eaton *et al.*, 2007; Whitfield *et al.*, 2007b). These censuses attempted to visit at least twice in the same season every known golden eagle territory in Scotland. A first visit to a territory (home range) involved checking if it was occupied by a pair prior to egg laying (i.e. direct observation of a pair, or of a built up nest), with a second visit during incubation to check for any evidence of birds having laid eggs or, if a territory was not found to be occupied on the first visit, to provide a second check for occupation. For occupied territories, a third visit was made later in the breeding season to record the number of any young that had fledged or had reached an age where fledging could safely be assumed (Steenhof, 1987; Steenhof & Kochert, 1992). Productivity was estimated as the number of fledglings produced per occupied territory per year. Birds were aged as subadult or adult on the basis of plumage (Watson, 1997; Bloom & Clark, 2001) whenever possible.

Other surveys

The same methods as used by the national censuses were also used to produce three additional sources of information which allowed the derivation of productivity estimates. These sources and the results are described in detail in Annex 2 (golden eagle productivity in Scotland).

Analysis and utility

For each known territory recognised as a contemporary or former breeding territory, we calculated a territory centre that was the mean location of all used alternative nest sites (those used during a maximum period of 11 years, 1982-1992) or, if information on nest site use was not available, the mean location of all alternative nest sites (McGrady *et al.*, 1997, 2002; McLeod *et al.*, 2002a, b). For the 2003 census, we employed the eyrie locations used in 2003 or, if an eyrie was not used in 2003, the most recent eyrie used or territory centres from previous censuses. In several analyses involving the national survey data, following extensive checks and cross-validation of nest site coding, territory centres and eyrie locations were entered into a Geographical Information System (GIS) (ArcView®). Classification of territory status (e.g. occupied/active or vacant, age of pair) and breeding parameters (e.g. productivity) were included as attributes attached to the relevant territory. Data were also extracted from the 1992 national survey database for nest aspect and altitude.

PAT MODEL

The PAT (Predicting Aquila Territory) model was used in several analyses as an estimation of eagle range/territory use. Details of the model, its development and the precursor ‘RIN’ model are given by McGrady *et al.* (1997, 2002) and McLeod *et al.*

(2002a, b). A brief summary of the PAT model, taken from McLeod *et al.* (2002b), is given below.

The conservation and management of golden eagles requires information on home range, which is expensive and time-consuming to collect. McLeod *et al.* (2002b) describe modelling techniques for predicting golden eagle ranging behaviour within a GIS. The model, called PAT, used data on ranging behaviour and geospatial factors from two areas of western Scotland (Fig. A1-1). Essentially, when nest sites are known, the model attempts to predict how the eagles utilising those nest sites use the surrounding area: it does not predict where eagles will nest. A range centre was estimated from the weighted mean nest site location in the past ten years. Range boundaries were estimated from Thiessen polygons[1], in the presence of neighbouring ranges, and a maximum ranging distance generated from parameters responsive to local range density, in the absence of neighbouring ranges. The model assumed that eagles did not use the sea or freshwater bodies, and avoided areas of human activity and closed canopy forests. The model also assumed, based on empirical data, that golden eagles preferred areas close to ridges (and other convex terrain features) and close to the centre of the range. The model output, at 50 x 50 m resolution, was three-dimensional with geographical location as *x* and *y* co-ordinates and use as a percentage of total home range use as the *z* co-ordinate (Fig. A1-2). Comparison of the model's predictions against range use observations in the two study areas of western Scotland and a third area in southwest Scotland (Fig. A1-1) suggested that it provided a good fit to observed range use (Fig. A1-3).

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[1] Straight lines are drawn mid-way between neighbouring range centres to produce a series of polygons (known as Thiessen polygons) whereby each range contains all the space that is closer to its range centre than to any other.

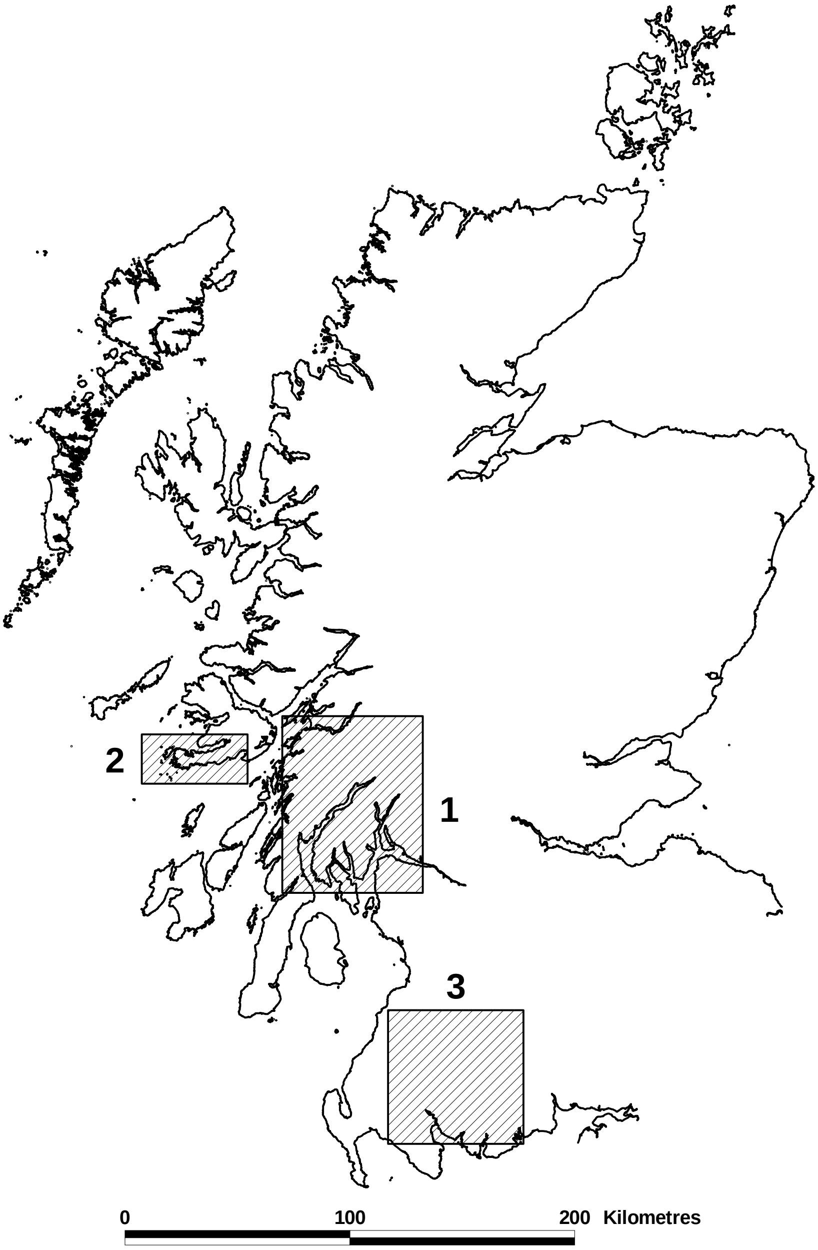
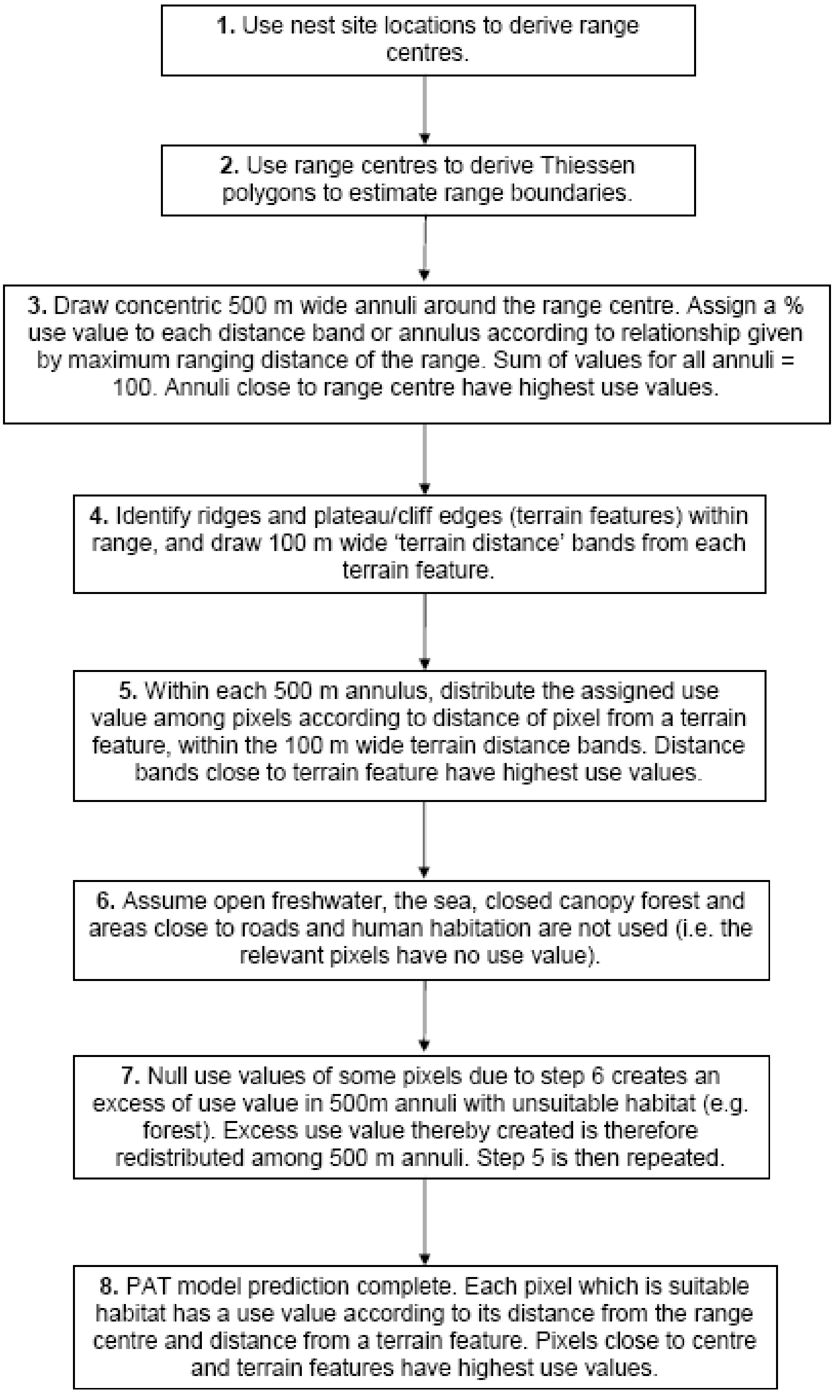


Fig. A1-1. The study areas used to develop the PAT model: 1. mainland Argyll, 2. Ross of Mull, 3. Galloway. Individual ranges are not illustrated to retain confidentiality. The model was developed and tested on ranges in Mull and mainland Argyll and independently tested on ranges in Galloway.



1. Use nest site locations to derive range centres.

2. Use range centres to derive Thiessen polygons to estimate range boundaries.

3. Draw concentric 500 m wide annuli around the range centre. Assign a % use value to each distance band or annulus according to relationship given by maximum ranging distance of the range. Sum of values for all annuli = 100. Annuli close to range centre have highest use values.

4. Identify ridges and plateau/cliff edges (terrain features) within range, and draw 100 m wide 'terrain distance' bands from each terrain feature.

5. Within each 500 m annulus, distribute the assigned use value among pixels according to distance of pixel from a terrain feature, within the 100 m wide terrain distance bands. Distance bands close to terrain feature have highest use values.

6. Assume open freshwater, the sea, closed canopy forest and areas close to roads and human habitation are not used (i.e. the relevant pixels have no use value).

7. Null use values of some pixels due to step 6 creates an excess of use value in 500m annuli with unsuitable habitat (e.g. forest). Excess use value thereby created is therefore redistributed among 500 m annuli. Step 5 is then repeated.

8. PAT model prediction complete. Each pixel which is suitable habitat has a use value according to its distance from the range centre and distance from a terrain feature. Pixels close to centre and terrain features have highest use values.

Fig. A1-2. Flow chart illustrating the steps in the prediction of range use by golden eagles using the PAT model (from McLeod et al. 2002b).

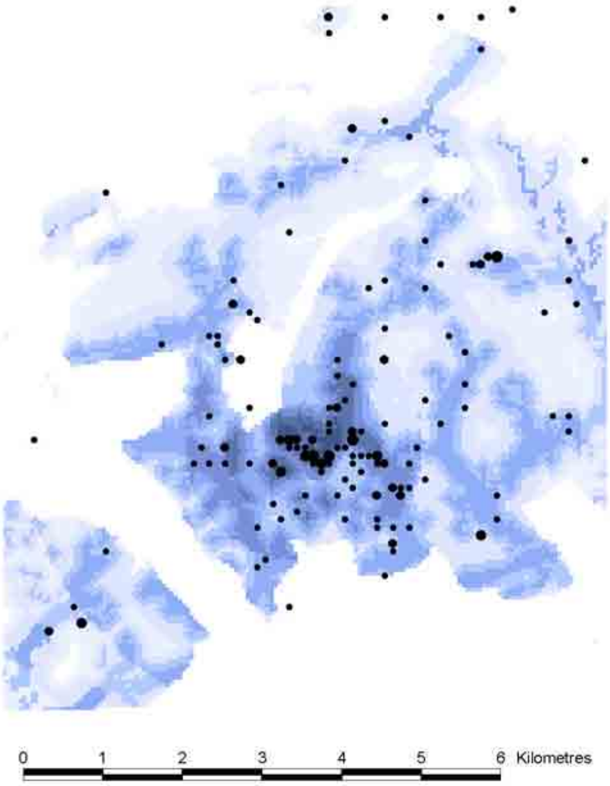


Fig. A1-3. Two dimensional representation of predicted range use for a golden eagle territory according to the PAT model (shaded areas) and observations of range use (solid circles – size proportional to number of observations) for a study range in mainland Argyll. For the PAT predictions different intensities of shading represent different classes of predicted range use with darker shading representing greater predicted use of an area (number of classes kept low in this example for clarity of presentation). Note also that range use observations are not necessary to generate the PAT predictions, but in this case the PAT model was run for a study range to give an indication of 'observed' use (from McLeod et al., 2002b).

Since the model was published (McLeod *et al.*, 2002b), range use observations have been collected for a number of additional eagle ranges in western Scotland (Haworth *et al.*, unpublished). Subjectively, comparison of PAT predictions against these observations also suggested a reasonably good fit of predicted to observed range use (Fig. A1-4).

The PAT model can not be considered ideal and numerous discrepancies with actual range use are and will be evident, however, and it does not provide a substitute for actual range use observations; a point made repeatedly in its development (McLeod *et al.*, 2002a, b). However, in the absence of range use observations, when detailed analyses of individual range use are required, it clearly can offer advantages over other less novel and more simple options (McLeod *et al.*, 2002a, b). Outstanding issues which require resolution and further analysis include alteration to range use where the centre is close to a boundary (e.g. when pairs nest on sea cliffs), accounting for split range centres (when alternative nest sites are widely apart), effects on range use of ‘barriers’, such as large forestry blocks or lochs, including distance of ‘barrier’ to range centre, and accounting for the possibility that golden eagles use terrain features as territorial boundaries.

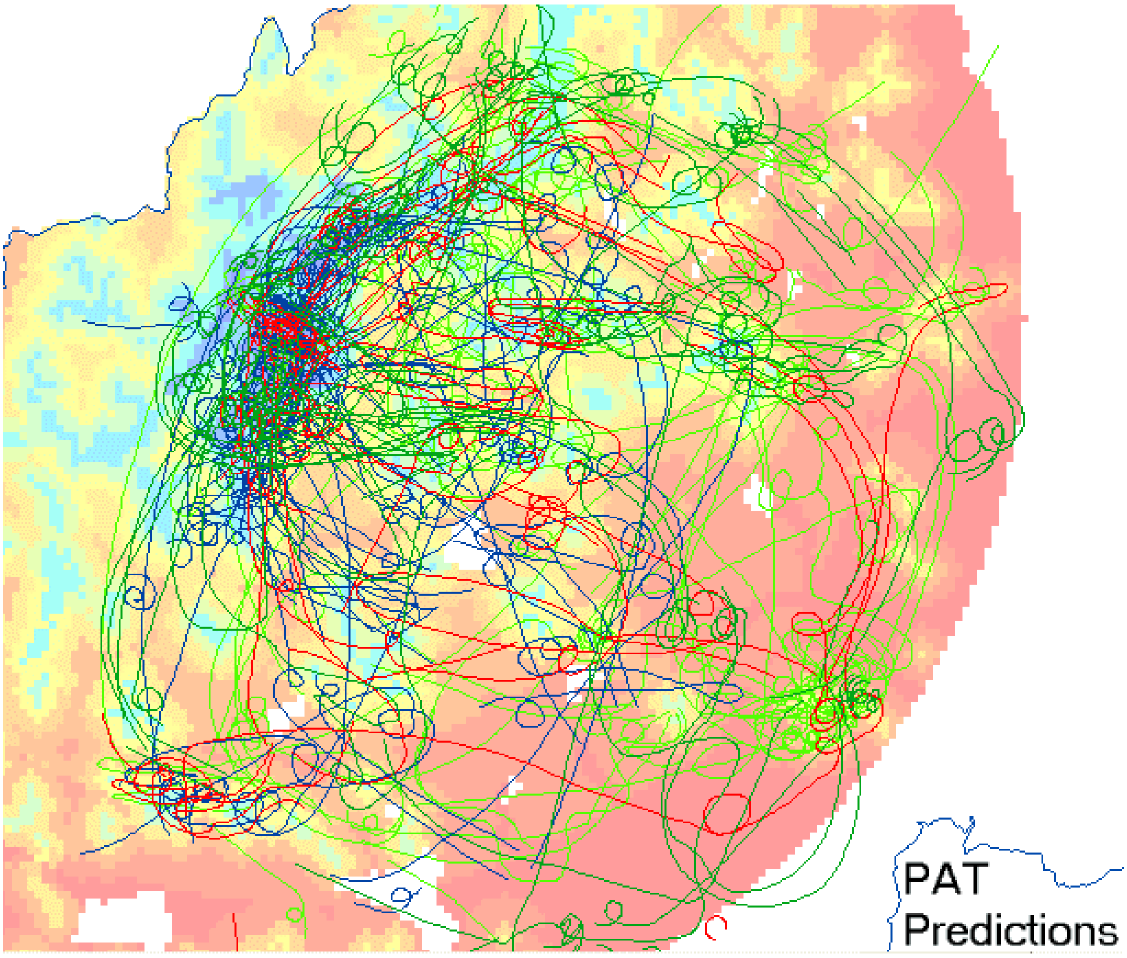


Fig. A1-4. Range use observations (observed flight lines from selected vantage points – colours represent different months, loops represent upward circling) superimposed on PAT model predictions (pink- low use, yellow – moderate use, pale blue – high use, dark blue – very high use) for a golden eagle range in western Scotland (site and scale details not given for confidentiality). PAT predictions in this example made no allowance for potential avoidance of roads, human habitation or woodland: the apparent discrepancy between predicted and observed in the west-northwest of the predicted range is liable to be due to eagles’ avoidance of a large swathe of woodland/scrub in this area.

LAND COVER AND ENVIRONMENTAL DATA

The majority of our studies involved at least initial analysis using a GIS. As noted earlier, eagle range (=territory) centres and associated attributes were entered in the GIS, and various ‘environmental’ data layers were then entered and measures of spatial association (e.g. overlap) or disassociation were extracted and tested. Here we describe the environmental datasets. Further information may be found in the several peer-reviewed papers which have been produced and which are referred to in the main report text.

NHZ boundaries

NHZ boundaries were supplied, by SNH, as an Arcview shapefile.

LCS88

Vegetation type descriptions were obtained from land cover data generated from the LCS88 data set (Land Cover of Scotland 1988: MLURI, 1993) provided by SNH in a digital format (Arcview shapefiles). The National Inventory for Woodlands and Trees (Forestry Commission, 1:25,000 data capture) is an update of the LCS88 to include Forestry Commission (FC) new planting, New Woodland Grant Schemes, woodland in urban and woodland beneath cloud/shadow, at 31 March 1995. (Woodland areas are considered to have greater than 50% cover by tree crowns.) Data were used for eight classes:

- Coniferous plantation - Conifers occur mainly as large plantations with well-defined edges and with trees of even age and height in regular rows.
- Coniferous (semi-natural) - not generally extensive, consisting principally of mature Scots Pine as in remnants of the former Caledonian Forest. Mostly confined to Highland and upland areas.
- Broadleaved - comprises a wide range of woodland types form managed policy woodlands to dense birch scrub. Tall scrub comprising hazel, alder and willow or dominated by birch is also included within this category.
- Mixed woodland - woods comprising a mixture of broadleaved and coniferous trees with at least 20% of each type.
- Scrub - occurring mainly on steep slopes or in rugged terrain, gorse, broom, or occasionally juniper giving more than 50% ground cover.
- Recent (unknown tree type) - land recently ploughed in preparation for tree planting.
- Felled - (unknown tree type) - areas where woodland has been felled and evidence of replanted trees cannot be seen. Areas of windblow will be included in this category.
- Open canopy (young plantation) - woodland, mainly coniferous plantation, between the stages of ground preparation by ploughing or ripping when no trees are evident and when the canopy of the developing trees closes.

For the purposes of any future analyses employing forest cover data, we would emphasise the considerable efforts which we had to use to characterise, adapt and cross-validate the stage and nature of forest plantations beyond those apparent from most digital datasets. For example, FC stock maps may give a date of planting, but if these refer to privately-owned forests they may not represent the actual date of planting or even if planting occurred, and through ground and climatic conditions, tree growth (and hence canopy closure) may not reflect planting dates. Some areas classified as part of a forest scheme may also actually remain unplanted.

We classified the vegetation of Scotland using the summary classes of LCS88. We included the following LCS88 summary classes as upland ‘open’ vegetation polygons in a layer within the GIS: bog, bracken, heather with muirburn, dry heather moorland, wet heather moorland, undifferentiated heather moorland, coarse grassland, and montane. A subclass of the ‘heather with muirburn’ class, termed 'strip muirburn' habitat results from land managers periodically burning heather moorland to stimulate new growth in heather, the main food plant of red grouse. By burning strips, a mosaic of short (for feeding) and long heather (for cover) is created (e.g. Muirburn Working Party, 1977; Hudson, 1992; Gimingham, 1995). This management aims to create good habitat for red grouse (Hudson, 1992; Moorland Working Group, 1999) and is uniquely associated with areas where shooting of red grouse occurs (e.g. Moorland Working Group, 2002). Hence, we took the distribution of the strip muirburn habitat as a surrogate for the distribution of moorland actively managed for red grouse (cf. Thompson *et al.*, 1997). In LCS88 about 20% of heather moorland in Scotland appears as strip muirburn or about 3% of the total land area of Scotland (Wrightham & Armstrong, 1999).

LCS88 was created by interpretation of aerial photographs supplemented and validated by ground survey. Mapping of strip muirburn included areas of burned and unburned heather. Unlike some habitats, strip muirburn is easily identified from aerial photographs due to sharp boundaries arising from the effect of burning on vegetation which remain visible for many years. Indeed, analysis of potential errors in habitat identification revealed that ‘validation’ ground survey was more likely not to record strip muirburn mistakenly than was photographic interpretation. Errors where strip muirburn was not recorded where it should have been (‘omission’ errors) were comparatively low, and so this habitat was well-suited to the mapping technique (MLURI, 1993). Other errors in LCS88 (e.g. during digitisation of habitat boundaries) were also insignificant in the context of the scale of the current analyses (MLURI, 1993). On some moorland sporting estates or parts of estates where red grouse are shot as game, strip muirburn may not be practised. Therefore, our surrogate estimate of the extent of moorland managed for red grouse using the distribution of strip muirburn was conservative.

Digital terrain data

Terrain data were extracted from the Ordnance Survey's (OS) 1:50,000 raster digital elevation model. These Ordnance Survey data were provided under licence to SNH.

Solid geology

Version 3.1 of a digital database of the solid onshore geology was compiled by the British Geological Survey to process and manipulate the map data within a GIS. The geological coding is by lithostratigraphy and lithology. The data capture scale is 1:250000 and all data are copyright British Geological Survey, supplied under licence to SNH.

Potential prey

Data were obtained for the national distributions of several potential prey species: red grouse (BTO Breeding Bird Atlas), mountain hare (JNCC), seabirds (JNCC Seabirds

Team ‘Seabirds 2000’). However, none of these were used in analyses because, following preliminary examination, it was decided that problems with the data outweighed their usefulness. The main difficulty with the hare data was the likely sparse coverage of survey effort, as hares were purportedly ‘absent’ from large areas where we knew from several years of experience in the uplands that they were present. The lack of any survey effort details with the data therefore rendered the data too unreliable for use. There was a similar problem with the seabird data, although gaps in coverage were clearly not as extensive as with the hare data as judged by the apparent ‘absence’ of seabirds from parts of the western coasts of Scotland. However, the clustered distribution of several seabird species rendered the ‘filling in’ of gaps in coverage through analytical interpolation procedures extremely difficult and potentially unreliable. Although there were several golden eagle territories which were coastal in western Scotland (and so where seabirds potentially form a large dietary component for eagles), nationally such territories were relatively infrequent and so unreliability in the coverage of seabird surveys could potentially incur a large bias. Hence, the data were not used.

The red grouse data were far less problematic, because whilst survey coverage was not complete they included measures of survey coverage, although they were essentially ‘presence/absence’ data on a tetrad basis. Thus it may be possible to generate estimates of occurrence in unsurveyed areas by interpolation procedures (e.g. Kriging) guided by the presence of heather, or to utilise the data at a scale appropriate to the relative coarseness of the data resolution. Either way, such options were beyond the time and resources available for the project, given other priorities, although the data may prove useful in any future analyses which may focus more closely on relationships between golden eagles and live prey (a difficulty here, however, may be that mountain hares are potentially equally or more important to eagles than red grouse as a ‘key’ prey species and an absence of data on hares may confound any relationships between live prey and eagle biology).

Grazing, livestock and large herbivores

Armstrong *et al.* (unpublished) calculated, for each parish, the biomass offtake (kg of dry matter (DM)) by grazers (sheep, red deer and cattle) in relation to biomass production (kg DM) by the vegetation. This followed procedures described by Armstrong *et al.* (1997a, b). They assumed that twelve of the LCS88 vegetation types could be equated to one of the vegetation classes used by Armstrong *et al.* (1997a) The dry matter (kg DM) biomass production of the vegetation in each parish was calculated for each combination of six altitude classes and seven temperature zones, as described in Armstrong *et al.* (1997a). The DM production of each vegetation class in each temperature and altitude zone was then multiplied by the maximum proportion of the DM production of each vegetation class likely to be available to grazing animals. It was also multiplied by the average digestibility of the vegetation class to give the maximum amount of digestible DM available to large, grazing animals per unit area. The total available, digestible DM was calculated for each agricultural parish in Scotland. The potential biomass offtake was then modelled by assuming intake from a given vegetation type depended on the area covered by the vegetation type and the potential daily intake of digestible dry matter (DM) available from it, in relation to those for all vegetation types present. Potential daily intake of digestible DM was calculated as the product of diet digestibility and daily DM intake. Daily DM intake is assumed to be limited either by diet digestibility or by a maximum grazing time, whichever is the lower. If the latter, estimates of bite weight and bite rate were used to predict the maximum achievable DM intake

(Armstrong *et al.*, 1997b). Using species-specific information on potential daily intake, digestibility and bite parameters, and multiplying these individual measures by estimates of the number of each large grazing species per agricultural parish, parish estimates of DM offtake were calculated.

The difference between production and offtake provided by Armstrong *et al.* (unpublished) gave a grazing equity figure. A high equity value represents a high production by vegetation relative to offtake, which will leave more vegetation biomass available to grazers such as red grouse and mountain hare that are important prey for eagles. On the other hand, a low equity value represents a low production by vegetation relative to offtake, leaving less food for the herbivorous prey of eagles. Equity values may therefore provide a useful surrogate for the availability of the key prey species of golden eagles in Scotland, which are thought to have an important influence on the breeding success of eagles (Watson, 1997). Data on production and offtake were supplied by SNH.

In addition to the productivity and consumption estimates, Armstrong *et al.* (unpublished) also provided (through SNH) estimates of the numbers of sheep, cattle and deer in each parish (based largely on census data from or up to 1998). Sheep densities were based on 1998 census returns by agricultural parish to the appropriate UK government department (SEERAD - Scottish Executive Environment and Rural Affairs Department, formerly SOAFD - Scottish Office Agriculture and Fisheries Department). Parishes vary in size across regions and do not coincide exactly with NHZ boundaries so where NHZ boundaries cut across a parish the assumption was made that grazing density and equity were the same across a parish. Density of sheep was taken as number of sheep per unit area of all land in a zone rather than per unit area of upland habitat in initial analyses involving the 1982 and 1992 survey data (this was revised to incorporate more specific reference to upland habitats in analyses of change between 1992 and 2003 national eagle surveys, as described later). In many areas sheep may not be shepherded on to the open upland habitats used by eagles, but may graze on improved pastures below the altitudes typically used by eagles. Deer data were taken from Deer Commission for Scotland (DCS) counts 1993-1998. Counts across the whole of Scotland were not taken in the same year and even within the same zone counts were always conducted across several years. Hence, there may be a degree of error if deer moved between count areas between years, and deer numbers will have changed across the counting period.

The grazing data have to be treated with some caution because parishes can be very large, particularly in regions with many eagles. Secondly there are some temporal mismatches between grazing data sets and the golden eagle data. Therefore the allocation of grazing data to individual ranges may be unreliable. There are also some missing data for confidentiality reasons: in some parishes there are very few land owners and so it is easy to work out patterns of livestock ownership. Whitfield *et al*. (2007b) give further discussion on the spatial limitations of these data.

For analyses of change in grazing and numbers of large herbivores between the 1992 and 2003 national eagle surveys (Whitfield *et al.*, 2007b) counts of red deer were obtained from Deer Commission for Scotland (1997, 2002, 2003). These data were counts of stags, hinds and calves within geographical count areas spanning the years 1963 – 2003. Boundaries of count areas were obtained from Deer Commission for Scotland and entered with associated count data as a layer in the GIS. Count areas did not encompass all of the geographical distribution of red deer in Scotland but covered the substantial majority (Deer Commission for Scotland, 2002). The number and timing of years when counts took place varied between count areas. For each count area we took total counts (all age and sex classes) from a year as close as possible to 1992 and 2003, and interpolated or extrapolated counts to estimates for 1992 and 2003 from simple linear regressions of count against year, or took counts from the years themselves if available. For each territory (either active or vacant) within deer count areas we derived Thiessen polygons around each eagle territory centre using Dirichlet tessellation (e.g. McGrady *et al.,* 2002; McLeod *et al.,* 2002a, b), set at a maximum distance of 3 km from the centre. We also assumed that deer would only be found within upland habitats (upland habitats defined and used in the GIS according to Whitfield *et al.* (2003)) and used the resulting area to derive an estimate of deer density within each territory. Using Thiessen polygons has an advantage over simple circles around a territory centre in this context by being responsive to differences in nesting density and does not produce any overlap in estimated territory use (McLeod *et al.,* 2002b). As a measure of short-term change we took (density of deer in 2003 – density of deer in 1992).

Data on change in sheep numbers (including total sheep, rams, ewes for breeding, lambs) were taken from the annual agricultural June parish census (Fuller & Gough (1999) give details), obtained from the University of Edinburgh Data Library through SNH after conversion to a 5 km square resolution, for the years 1972, 1982, 1992 and 2000 (2000 being the closest year to the 2003 national eagle census for which data were available). These data were entered as a layer in the GIS. As for deer, we estimated the density of sheep within upland habitats within 3 km Thiessen polygons for each active and vacant eagle territory. Short-term change in sheep density was estimated by (density of all sheep in 2000 – density of all sheep in 1992). These ‘change’ data were used by Whitfield *et al*. (2007b).

Recreation: Munros

Golden eagles appear to be intolerant of repeated and frequent human presence (e.g. Watson & Dennis, 1992; Watson, 1997) and territory abandonment can potentially occur when nesting eagles are repeatedly disturbed at the eyrie by recreational activity (e.g. Brendel *et al.*, 2002). In Scotland most recreational activity which could lead to casual disturbance of eagles is due to hillwalkers (e.g. Hall, 2002). Much hillwalking in Scotland is due to people visiting a series of over 280 mountain peaks colloquially called ‘Munros’ (summits above 3000 ft ASL) (Bennet *et al.*, 1991). The locations of Munros were entered as an additional layer in the GIS as a basis for examining potential spatial associations between Munro distribution and eagle distribution.

It is recognised that this is a simplistic approach that would be improved if data were available on the relative numbers of visitors to each Munro and the access route through a golden eagle territory. Initial explorations using the latter parameter were thwarted by insufficient time but did not reveal any clear adverse patterns and could be worth pursuing.

Persecution

We used two data sets relating to incidence of persecution. The first encompassed all records of illegal poisoning in Scotland collated annually by the Royal Society for the

Protection of Birds (e.g. RSPB, 2001) for the years 1981-2003 inclusive. Whitfield *et al.* (2003, 2004a) describe how these data were prepared to remove spatially coarse records and potential pseudo-replication, before the geographical grid references for the poisoning incidents were entered as a layer into the GIS. These data may be referred to as poisoning data to distinguish them from the second data set involving golden eagle persecution (see next paragraph) and were used in most analyses (e.g. Whitfield *et al.*, 2003, 2004a, b). Poisoning data referred to all records involving evidence of the use of agricultural pesticides which were inconsistent with their prescribed (more recently, legal) use, and included incidents of dead birds (often, raptors), mammals, and poisoned baits, which had been independently confirmed for pesticide presence and levels.

The second data set involved all records of illegal persecution of golden eagles collated by the RSPB (RSPB, 2003), which involved a wider range of type of incidents than the poisoning data, including trapping, shooting, poisoning, and destruction of nests or eggs, but exclusively involved golden eagles. The records were assigned to one of three categories: 'confirmed' cases, incidents where definite illegal acts were disclosed e.g. the substantive evidence included a shot bird or a trapped bird; 'probable' cases, where the available evidence pointed to persecution as by far the most likely explanation but where the proof of an offence was not categorical; 'possible' cases, where persecution was a possible explanation but where another explanation would also fit the known facts (RSPB, 2001, 2003). For the purposes of our analyses we only considered confirmed and probable cases, and we excluded any incidents with a minimum spatial resolution greater than 10 km (Whitfield *et al.*, 2003). These data may be referred to as eagle persecution data, and their geographical grid references were entered as a layer in the GIS (Whitfield *et al.*, 2007b). Whitfield *et al*. (2007b) showed that poisoning data and eagle persecution data, and their change over time, were strongly correlated spatially.

Wind farms

We obtained data on wind farm schemes from two main sources. The first was the Casework Recording System (CRS) of Scottish Natural Heritage (SNH). The development of renewable energy schemes is governed by UK legislation and the government planning system. Proposals for wind farms are determined by local government authorities or, if a proposal is over 50 MW, by Scottish government ministers. SNH is a statutory consultee for schemes in Scotland in its position as the advisor to government on natural heritage issues. SNH records all cases of development and potential development as 'casework' items in the CRS in order to maintain an audit trail of responses and to monitor progress (further details are given by SNH (2003, 2004)). Potential schemes on which SNH is consulted before formally entering the planning system (and therefore the public domain) are required to be treated as 'commercial in confidence' if so requested by the scheme developer (hence all analyses involving such schemes were conducted within SNH), although several are in the public domain by virtue of, for example, consultation with local communities, other publicity or because wind farm proposals are often preceded by planning applications for anemometry masts. Cases are recorded and updated according to the following stages in the planning process: pre-application (proposals at the earliest stage of development which have not entered the planning system), scoping (schemes registered with the planning authority in order to seek a direction or 'scoping opinion', about the nature of any environmental assessment needed), application (schemes for which planning permission has been sought), approved (schemes with planning consent), refused (rejected by the planning authority) and installed (operational). Data were collated as of the end of December 2004.

The second data source was the Gazetteer of Wind Power in Scotland (SWAP, 2005). Data in SWAP had a number of origins, similar to those behind CRS and were collated to 9 January 2005. The primary purpose of utilising both data sets was to obtain as much information on potential wind farms as possible: there is no requirement for developers to consult SNH prior to formal entry of the planning system, and there was sometimes a lag in the recorded status of schemes in CRS because authorities do not necessarily communicate planning decisions to SNH. The primary contribution of SWAP to our data collation was to pre-application schemes. Care was taken to avoid duplication of recorded schemes (which was possible due to scheme name changes) by cross-checking other details such as location and developer. If there was a doubt about possible duplication, only one case, from CRS, was considered. It is possible that some pre-application schemes were duplicated in our dataset, but we do not believe that this duplication was more than a handful of cases across the whole of Scotland, at most.

Scheme parameters recorded by CRS and SWAP included ‘installed capacity’ (the energy output capacity in MW if or when the scheme is installed), the number of turbines, and the dimensions of turbines (CRS only). Geographical grid location was also recorded, although for schemes in the earliest stages of development these could be crude which should temper any conclusions reached.

ANNEX 2: GOLDEN EAGLE PRODUCTIVITY IN SCOTLAND

SOURCES OF INFORMATION

National surveys

Estimates of productivity were available through the national censuses for the years 1982, 1992 and 2003. The 1992 national survey collated productivity data covering the period 1982-1992, but the number of years for which these data were available varied between territories. Unfortunately, even for those territories for which data were presented there was inconsistency in the years for which data were available. This is perhaps inevitable, however, when annual coverage of eagle territories is not complete nationally due to the reliance on voluntary efforts of the Scottish Raptor Study Groups (SRSGs) and the difficulties in accessing many eagle territories. Under the circumstances, the coverage achieved by the SRSGs, both spatially and temporally, provides a remarkable resource backed by many years of skilled experience and familiarity with the species.

SRSG reports

Most annual monitoring of golden eagles in Scotland is undertaken by volunteers, particularly by members of the SRSGs. The SRSG network is incomplete in its geographic coverage and the consequent impact of this on data availability is primarily responsible for zonal differences in data availability. More extensive SRSG productivity data are summarized in various publications, particularly in the ‘Raptor Roundups’ (annual publications as supplements to Scottish Birds, the journal of the Scottish Ornithologists’ Club). Because inconsistent summary data are given for the SRSG regions and SRSG areas differ from NHZs it was impossible to allocate their productivity data to NHZ. However, it was still possible to examine broad historical and regional trends. Although some of these SRSG data were included in the 1992 national survey database referred to above they also include data up to and including 1999.

Other surveys

Data from 110 ranges (mainland Argyll, Mull, Skye and Lochaber) were also collated between 1982 and 1999 inclusive (e.g. Watson *et al.*, 2003). On average data were obtained from 99 ranges each year (minimum 86, maximum 106).

More comprehensive regional data were also available in a variety of internal SNH reports and other publications. These have the advantage of documented consistent survey effort on known ranges, often over 18 years.

RESULTS AND DISCUSSION

National surveys

Using the GIS it was possible to allocate territories recorded under the national surveys to NHZ and thereby obtain an estimate of productivity for each NHZ (Table A2-1: productivity data for 2003 are shown in the main text on testing NHZ for favourable conservation status). These data produced a national estimate of 0.42 young fledged per occupied territory (excluding single adults) for 1982-1992. This was higher than the national estimate of 0.32 for 1992 only, but lower than the national estimate of 0.52 for 1982 only (Green, 1996) because it covered an 11-year period. 1992 seems to have been a relatively poor year and 1982 a relatively good year for golden eagle productivity. Productivity across Scotland in 2003 was 0.36 young fledged per occupied territory (Eaton *et al*., 2007).

Table A2-1. Mean productivity for each NHZ for the 1982 and 1992 national surveys. These means exclude unoccupied territories and those in which only immature singles or groups were seen. Total is number of young fledged in each NHZ.*

	1982				**1992**				**1982/92 mean**			**1982/92 sum**			
NHZ	**Mean**	**n**	**se**	**total**	**Mean**	**n**	**se**	**total**	**Mean**	**n**	**se**	**Mean**	**n**	**se**	**total**
2	1.00	1		1		0			0.50	1		1.00	1		1
3	0.45	58	0.07	26	0.20	60	0.06	12	0.27	71	0.04	0.54	71	0.08	38
4	0.50	46	0.09	23	0.22	49	0.06	11	0.29	59	0.04	0.58	59	0.09	34
5	0.22	18	0.10	4	0.36	14	0.13	5	0.20	22	0.07	0.41	22	0.14	9
6	0.47	77	0.07	36	0.33	75	0.06	25	0.37	83	0.04	0.73	83	0.09	61
7	0.42	79	0.07	33	0.29	55	0.07	16	0.28	86	0.04	0.57	86	0.08	49
8	0.29	55	0.07	16	0.14	57	0.05	8	0.19	63	0.04	0.38	63	0.07	24
10	0.24	17	0.11	4	0.29	14	0.16	4	0.20	20	0.09	0.40	20	0.18	8
11	0.60	35	0.12	21	0.63	27	0.15	17	0.45	42	0.07	0.90	42	0.14	38
12	0.30	10	0.21	3	0.13	8	0.13	1	0.17	12	0.09	0.33	12	0.19	4
13	0.39	31	0.11	12	0.32	28	0.10	9	0.32	33	0.08	0.64	33	0.16	21
14	0.50	36	0.11	18	0.37	43	0.08	16	0.35	48	0.06	0.71	48	0.12	34
15	0.53	17	0.17	9	0.32	22	0.14	7	0.33	24	0.08	0.67	24	0.17	16
19	0.67	3	0.33	2	0.00	3	0.00	0	0.33	3	0.17	0.67	3	0.33	2
20		0			0.00	1		0	0.00	1		0.00	1		0

**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,10 = Central Highlands, 11 = Cairngorms Massif, 12 = North East Glens, 13 = Lochaber, 14 = Argyll West and Islands, 15 = Breadalbane and East Argyll, 19 = Western Southern Uplands and Inner Solway, 20 = Border Hills.*

There was little evidence for a between-survey correlation in mean productivity (Fig. A2-1). The mean figures for 1982 and 1992 combined were generally lower than the individual surveys because they included some territories that were only occupied during one of the survey years, or were only surveyed once.

Following various studies such as Watson *et al*. (1987), the typical view of golden eagle productivity in Scotland is that it tends to be high in the east and low in the west. Whilst the present results illustrate that this view is not unreasonable, it is apparent that productivity is not universally high in the east and universally low in the west (Table A2-1). The relative importance of the west also became more apparent when regional contributions to the number of fledglings produced nationally were examined, with only a minority of national output of young being contributed by eastern regions (Table A2-2). In large part this was probably due to a combination of fewer territories in the east and that productivity there was often not markedly greater than in many western territories.

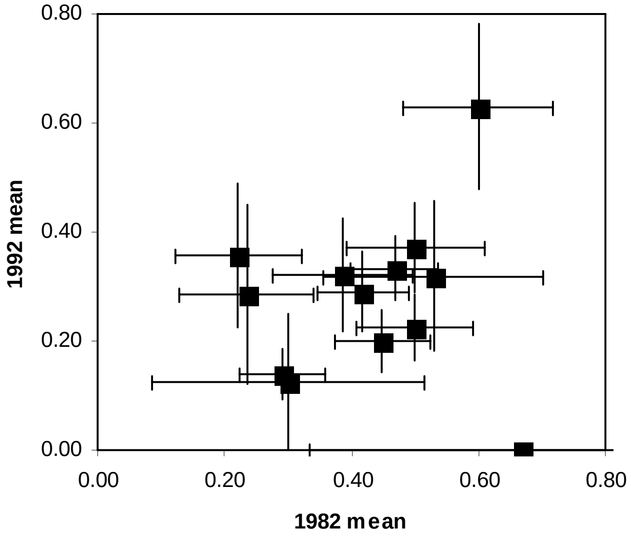


Fig. A2-1. Mean productivity (±s.e.) for each NHZ in 1982 and 1992 (excluding unoccupied ranges).

Table A2-2. Frequency of young fledged in 1982 and 1992 combined. The total fledged in each NHZ is expressed as a percentage of the national total.

	Number fledged frequency							
NHZ	0	1	2	3	4	Total	% National	Mean of successful ranges (combined years)
North Caithness and Orkney	0	1	0	0	0	1	0.3	1.00
Western Isles	50	25	5	1	0	38	11.2	1.23
North West Seaboard	40	22	6	0	0	34	10.1	1.21
The Peatlands of Caithness and Sutherland	24	5	2	0	0	9	2.7	1.29
Western Seaboard	50	33	11	2	0	61	18.0	1.33
Northern Highlands	57	22	12	1	0	49	14.5	1.40
Western Highlands	45	18	3	0	0	24	7.1	1.14
Central Highlands	21	3	1	1	0	8	2.4	1.60
Cairngorm Massif	44	10	12	1	0	37	10.9	1.61
North East Glens	16	2	1	0	0	4	1.2	1.33
Lochaber	21	12	1	1	1	21	6.2	1.40
Argyll West and Islands	31	15	8	1	0	34	10.1	1.42
Breadalbane and East Argyll	16	6	5	0	0	16	4.7	1.45
Western Southern Uplands and Inner Solway	2	2	0	0	0	2	0.6	1.00
Border Hills	2	0	0	0	0	0	0.0	

While mean fledging rates are useful, a better indicator of NHZ productivity would assess productivity per unit area. In order to examine productivity at the NHZ level the extent of area included in the productivity density calculations must first be defined. In the subsequent analyses 'fledgling density' was based on the area of 'eagle habitat' within each NHZ. Eagle habitat was defined as that within 6 km radius Thiessen polygons of all known territory centres. The results summarising fledgling density by NHZ are summarised in Table A2-3.

Table A2-3. Number of young fledged in each NHZ and the fledge density (number fledged per 100 km^2 of eagle habitat) in 1982 and 1992. The area of eagle habitat is defined as the area within 6 km radius Thiessen polygons from territory centres. Area is km^2. Combined is pooled data from 1982 and 1992.

		Total fledged			fledged per 100 km^2 of eagle habitat		
NHZ	Area	1982	1992	combined	1982	1992	combined
North Caithness and Orkney	89.5	1	0	1	1.117	0.000	1.117
Western Isles	2559.8	26	12	38	1.016	0.469	1.484
North West Seaboard	3085.1	23	11	34	0.746	0.357	1.102
The Peatlands of Caithness and Sutherland	2556.1	4	5	9	0.156	0.196	0.352
Western Seaboard	2959.8	36	25	61	1.216	0.845	2.061
Northern Highlands	4634.3	33	16	49	0.712	0.345	1.057
Western Highlands	2607.0	16	8	24	0.614	0.307	0.921
Central Highlands	1868.4	4	4	8	0.214	0.214	0.428
Cairngorm Massif	3385.8	21	17	38	0.620	0.502	1.122
North East Glens	1421.7	3	1	4	0.211	0.070	0.281
Lochaber	2120.8	12	9	21	0.566	0.424	0.990
Argyll West and Islands	3208.1	18	16	34	0.561	0.499	1.060
Breadalbane and East Argyll	2389.6	9	7	16	0.377	0.293	0.670
Western Southern Uplands and Inner Solway	383.6	2	0	2	0.521	0.000	0.521
Border Hills	180.2	0	0	0	0.000	0.000	0.000

Two western NHZs, the Western Isles and the Western Seaboard stand out as being much more productive than other regions by this analysis (Table A2-3). They are even more important given their large counts of territories. Three eastern regions, North East Glens, the Peatlands of Caithness and Sutherland and the Central Highlands, all have poor productivity per unit area. These observations are further supported by the contribution made by each NHZ to the national production of young (Table A2-2). The western NHZ are again highlighted as very important sources of young birds. Only the Cairngorm Massif in the east makes a major contribution to the national total, mainly because of its large mean (frequency of twins) for successful ranges.

With numbers of territories in the 2003 national survey having declined in several eastern regions, but having expanded dramatically in the Western Isles (Eaton *et al*., 2007; Whitfield *et al*., 2007b; see main text in present report on testing favourable conservation status), the disparity between western and eastern regions in contributions of young birds and in productivity per unit area will have become even more evident.

SRSG reports

The estimates of 0.47 and 0.46 young fledged per occupied range from SRSG data (1982-1999 and 1982-1992 respectively) were slightly higher than the estimates derived from the national surveys (Tables A2-4, A2-5). However, it was possible that at least some SRSG data were biased such that ranges with a poor success may not be checked or reported as frequently. Also, there were few data from some regions (e.g. Western Isles) which may have decreased the overall estimate had data been available. On the other hand, the data refer to some years not covered by the national survey data. Overall, the results suggested that any coverage or reporting bias was not severe.

Table A2-4. Scottish Raptor Study Group productivity data based on a maximum of 18 years of data (1982-1999). The percentage surveyed is derived from the sum of ranges checked or occupied divided by the maximum number of ranges for which data were available in any single year. [1]There is inconsistency in how data are presented for different years in the source material, for example sometimes only the number of checked ranges is given. 'Ranges' is either the sum of occupied ranges or the sum of occupied ranges plus the sum of surveyed ranges for those years without occupancy data. This is likely to reduce the estimated success since it may include unoccupied ranges. The mean number of young fledged is the total fledged divided by ranges.

Region	Checked	Occupy	Ranges[1]	% surveyed	Success	Fledged	Twins	Mean fledged per occupied range	Mean fledged per successful range
Argyll	610	933	996	55.1	383	465	72	0.467	1.214
Borders		9	9	25.0	3	3	0	0.333	1.000
Central	151	193	193	42.9	64	81	10	0.420	1.266
England	36	20	20	100.0	11	11	0	0.550	1.000
Highland	990	582	1488	69.3	618	722	104	0.485	1.168
Lewis-Harris	75	94	94	9.9	33	38	5	0.404	1.152
N.E.	273	433	433	53.5	207	252	69	0.582	1.217
Orkney		1	1	5.6	0	0	0	0.000	
S.W.		51	51	70.8	19	19	0	0.373	1.000
Tayside	149	240	240	40.4	83	118	35	0.492	1.422
Uists	46	53	53	16.4	18	19	1	0.358	1.056
W. Isles	174	214	214	16.7	58	69	11	0.322	1.190
All	2504	2823	3792	44.9	1497	1797	307	0.474	1.20

Table A2-5. Scottish Raptor Study Group productivity data based on 1982-1992 data. Data were obtained using the methods described for Table A1-1.

Region	Checked	Occupy	Ranges	% surveyed	Success	Fledged	Twins	Mean fledged per occupied range	Mean fledged per successful range
Argyll	379	573	573	55.4	223	270	47	0.471	1.211
Borders		1	1	9.1	0	0	0	0.000	
Central	113	141	141	51.3	43	48	5	0.340	1.116
England	22	12	22	100.0	9	9	0	0.409	1.000
Highland	470	486	843	35.1	359	416	57	0.493	1.159
Lewis-Harris	53	36	53	9.1	11	12	1	0.226	1.091
N.E.	238	283	283	57.2	136	160	48	0.565	1.176
Orkney		1	1	9.1	0	0	0	0.000	
S.W.		37	37	84.1	13	13	0	0.351	1.000
Tayside	57	80	80	22.0	30	47	17	0.588	1.567
Uists	28	28	28	18.2	5	5	0	0.179	1.000
W. Isles	174	149	149	22.3	33	41	8	0.275	1.242
All	1534	1827	2211	35.4	862	1021	183	0.462	1.184

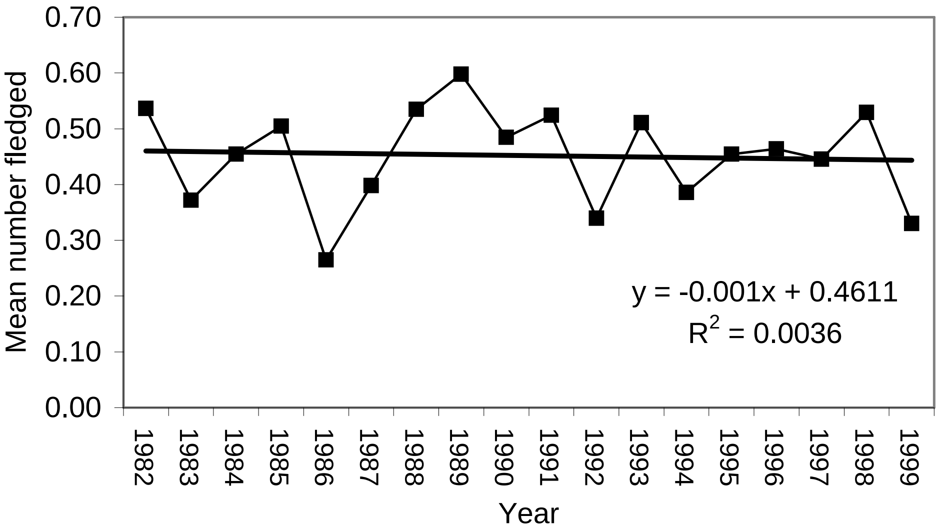


Figure A2-2. Annual productivity (fledged per occupied range) between 1982 and 1999 for 110 ranges in Argyll, Mull, Lochaber and Skye.

Other sources

The long-term data from mainland Argyll, Mull, Lochaber and Skye revealed an overall average number of young fledged per occupied range of 0.45. There was no evidence of a trend over time (Fig. A2-2). These data tended to confirm that the national survey years were relatively good (1982) and poor (1992, 2003) for productivity.

Using these data it was possible to test if the national survey years provided a reasonable relative approximation of productivity in the intervening years, by matching productivity from the national surveys to the same territories monitored over the greater number of years. The relationships between actual productivity and that predicted from the 1982 and 1992 national survey data are shown in Figs. A2-3 & A2-4.

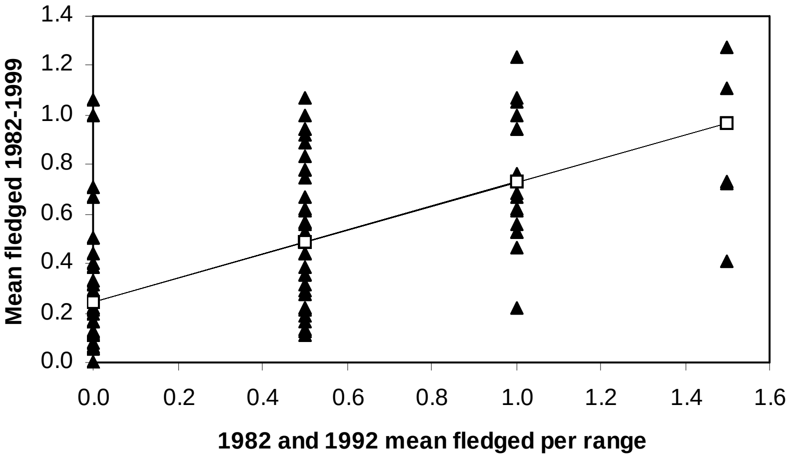


Fig. A2-3 Relationship between actual productivity over the period 1982 and 1999 inclusive and that predicted by the 1982 and 1992 productivity data (data from Fig. A2-1). Fitted line: mean = 0.246 + 0.482 (82and92mean) (weighted by number of range observations), R^2 = 38.0%, p<0.0001, n = 106 ranges.

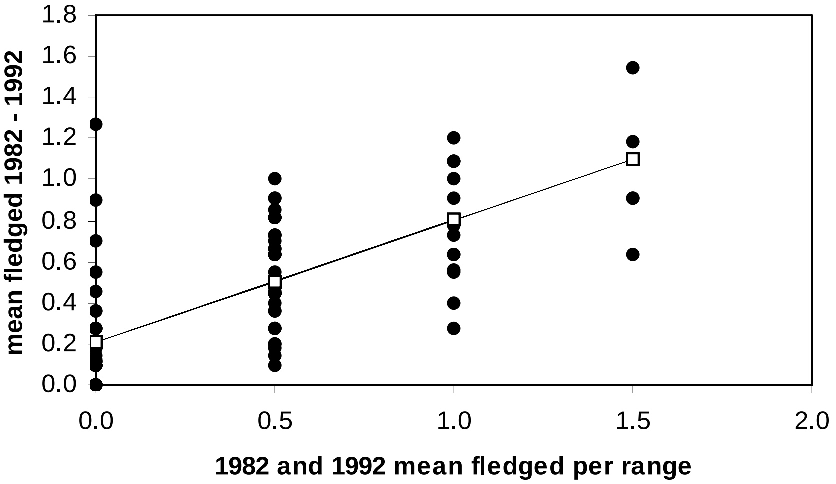


Fig. A2-4 Relationship between actual productivity over the period 1982 and 1992 inclusive and that predicted by the 1982 and 1992 productivity data (data from Figure 3.1). Fitted line: mean = 0.206 + 0.598 (82and92mean) (weighted by number of range observations), R^2 = 50.7%, p<0.0001, n = 106 ranges.

The results suggested that productivity data from the national survey years provided a reasonable approximation to relative productivity over the longer term (Fig. A2-3 & A2-4). It is interesting that the R^2 declined markedly when the prediction was extended to the end of the century (Fig. A2-3), inferring that 1982 and 1992 national survey data become less representative when taken out of the temporal context to which they referred. Further analyses using the 2003 national survey results confirm these findings from the two earlier surveys (Fielding *et al*., unpublished).

Table A2-6. Productivity data from other sources. 'Ranges' is the sum of ranges checked during the period of the study. Interference, where available, is a count of the number of ranges that were unsuccessful due to intentional and unintentional disturbance. When interference data are available a revised productivity estimate is presented after excluding ranges experiencing interference.

Region	Source	Period	Ranges	Occupied	Success	Fledge	Interference	Fledged per pair	Fledged per success	Adjusted Fledged per pair
Caithness & E. Sutherland	SNH (1990)	1990	5	5	1	1		0.20	1.00	
Deeside	Everett (1971)	1964-1968	61	61	29	38		0.62	1.31	
East Inverness	Watson *et al* (1992)	1982-1985						0.63		
Easter Ross & East Inverness	SNH (1990)	1990	16	16	4	6		0.38	1.50	
Eastern Highlands	Watson *et al* (1992)	1982-1985						0.69		
Galloway	Everett (1971)	1964-1968	18	18	3	3		0.17	1.00	
Harris	Haworth (pers comm.)	1999-2001	36	34	9	9	0	0.26	1.00	0.26
Highland region	SNH (1990)	1981-1990	500					0.49		
Lewis	Haworth (pers comm.)	1999-2001	60	47	13	14	7	0.30	1.08	0.35
Lochaber	Watson *et al* (1992)	1982-1985						0.28		
Monadhliath	Everett (1971)	1964-1968	20	20	6	7		0.35	1.17	
Mull	Watson *et al* (1992)	1982-1985						0.60		
North East	Watson (1957)	1945-1957	65	65	40	52	12	0.80	1.30	0.98
North Lochaber	SNH (1990)	1990	13	13	6	7		0.54	1.17	
North Sutherland	SNH (1990)	1990	16	16	8	8		0.50	1.00	
North Sutherland	Watson *et al* (1992)	1982-1985						0.52		
NW Sutherland	Brown (1969)	1967	20	20	7	8	3	0.40	1.14	0.47
Perthshire	Everett (1971)	1964-1968	40	40	16	20		0.50	1.25	
Perthshire	Watson *et al* (1992)	1982-1985						0.80		
Scotland	SNH (1990)	1990	154	154	77	95		0.62	1.23	
Scotland	Everett (1971)	1964-1968	243	243	114	138		0.57	1.21	
Skye	Watson *et al* (1992)	1982-1985						0.57		
Skye+Lochalsh	SNH (1990)	1990	32	32	15	20		0.63	1.33	
South Argyll	Everett (1971)	1964-1968	64	64	19	19		0.30	1.00	
South Argyll	Watson *et al* (1992)	1982-1985						0.62		
South Lochaber	SNH (1990)	1990	13	13	3	3		0.23	1.00	
Speyside	Everett (1971)	1964-1968	54	54	19	24		0.44	1.26	
Speyside & Moray	SNH (1990)	1990	10	10	4	6		0.60	1.50	
Uists	Bird reports	1997-2001	85	85	27	34	4	0.40	1.26	0.42
Wester Ross	SNH (1990)	1990	10	10	2	2		0.20	1.00	
Wester Ross	Everett (1971)	1964-1968	59	59	34	35		0.59	1.03	
Wester Ross	Watson *et al* (1992)	1982-1985						0.57		
All (excluding double counting)			697	682	265	316		0.46	1.19	

The final source of productivity data was a range of published and unpublished reports that differed in the detail reported (Table A2-6). The average number fledged per pair over these studies was 0.46 (range 0.17 – 0.80). In some regional reports, likely causes of breeding failure were presented, which included human interference (e.g. nest site destroyed, nest contents destroyed, failure associated with signs of humans at the nest). In those regions where interference was quantified it seems that the mean number fledged was reduced by around 20%. These data tend to confirm, again, that any bias in reports of the SRSG data was negligible.

Constraints on productivity

Two simplistic preliminary approaches were adopted to search for potential constraints on regional productivity. The first used ‘habitat’ measurements at the NHZ level to search for correlations with productivity. The second used single factor analysis of variance, undertaken at national and NHZ scales, to compare the habitat in ranges that fledged 0, 1, 2, 3 or 4 young in 1982 and 1992 combined.

Table A2-7. Correlation coefficients between areas of 34 LCS88 single and mosaic habitat classes and three measures of range productivity. Data are presented separately for the 1982 and 1992 national surveys and for 1982 and 1992 combined. Successful is the mean number fledged by successful ranges, pooled across the two survey years. Coefficients in bold are significant at p<0.05, bold italics are significant at p<0.01. In order to avoid any double counting eagle habitat was measured within 6 km radius Thiessen polygons centred on all known range centres. No multiple testing adjustments were made to significance levels.

	Mean fledged				Total fledged			fledged per 100 km^2 eagle habitat		
	1982	1992	combined	successful	1982	1992	combined	1982	1992	combined
1. Arable	-0.103	-0.435	-0.284	**-0.605**	-0.382	-0.439	-0.411	-0.305	-0.447	-0.409
2. Improved Grassland	0.306	**-0.569**	0.021	-0.215	**-0.570**	**-0.594**	**-0.589**	-0.125	***-0.655***	-0.380
3. Good Rough Grassland	-0.011	-0.340	-0.124	-0.438	-0.432	-0.304	-0.389	-0.313	-0.343	-0.367
4. Poor Rough Grassland	-0.240	**-0.549**	-0.433	-0.438	-0.456	-0.471	-0.470	-0.408	-0.510	-0.508
5. Bracken	0.156	-0.305	0.080	-0.286	-0.140	-0.210	-0.170	-0.084	-0.307	-0.196
6. Heather Moorland	-0.332	0.279	-0.268	**0.615**	0.151	0.117	0.140	-0.279	-0.039	-0.208
7. Peatland	-0.193	0.064	-0.205	0.112	-0.145	-0.081	-0.122	-0.256	-0.076	-0.209
8. Montane	-0.116	**0.570**	0.130	**0.543**	0.396	0.407	0.407	0.039	0.341	0.180
9. Rocks and Cliffs	0.102	0.205	0.234	-0.069	***0.683***	***0.645***	***0.680***	**0.593**	***0.694***	***0.717***
10. Felled Woodland	-0.109	-0.434	-0.210	-0.160	-0.292	-0.287	-0.295	-0.358	-0.366	-0.410
11. Recent Planting	-0.106	-0.165	-0.072	-0.029	-0.139	0.010	-0.083	-0.294	-0.064	-0.229
12. Coniferous Plantation	-0.209	-0.468	-0.321	-0.133	-0.341	-0.312	-0.336	-0.450	-0.408	-0.490
13. Semi-Natural Coniferous	-0.180	**0.545**	-0.106	**0.570**	0.058	0.145	0.093	-0.163	0.076	-0.077
14. Mixed Woodland	-0.146	-0.167	-0.104	0.111	-0.111	-0.106	-0.111	-0.326	-0.175	-0.301
15. Broadleaved	-0.092	-0.005	0.037	0.139	0.121	0.174	0.144	-0.167	0.103	-0.067
16. Scrub	-0.181	-0.457	-0.307	-0.029	-0.283	-0.324	-0.304	-0.409	-0.425	-0.470

	Mean fledged				Total fledged			fledged per 100 km^2 eagle habitat		
	1982	1992	combined	successful	1982	1992	combined	1982	1992	combined
17. Freshwaters	0.058	-0.055	0.121	-0.080	0.433	0.225	0.359	0.168	0.148	0.181
18. Marsh	0.519	-0.417	0.279	**-0.639**	-0.325	-0.406	-0.363	0.226	-0.417	-0.034
19. Saltmarsh	0.204	-0.355	0.079	-0.512	-0.100	-0.146	-0.120	0.106	-0.165	-0.002
20. Dunes	0.351	-0.033	0.252	-0.296	0.200	0.064	0.151	0.491	0.119	0.388
21. Tidal Waters	0.034	-0.306	-0.082	-0.410	-0.353	-0.298	-0.338	-0.236	-0.319	-0.304
22. Rural Development	0.473	-0.369	0.260	**-0.594**	-0.424	-0.434	-0.436	0.051	-0.481	-0.181
23. Urban	0.224	-0.328	0.090	-0.406	-0.332	-0.275	-0.315	-0.097	-0.317	-0.209
24. Missing or Obscured	-0.078	0.436	0.159	0.391	0.087	0.190	0.129	-0.114	0.247	0.034
25. Heather Moorland / Peatland	-0.081	0.260	0.072	0.113	***0.679***	**0.608**	***0.663***	0.283	**0.552**	0.441
26. Poor Rough Grass / Heather Moorland	-0.244	-0.175	-0.191	-0.104	-0.003	-0.001	-0.002	-0.180	-0.012	-0.128
27. Good Rough Grass / Heather Moorland	-0.059	0.421	0.062	0.216	0.498	**0.571**	**0.536**	0.310	0.520	0.445
28. Peatland / Montane	-0.060	**0.569**	0.198	**0.605**	0.353	0.370	0.366	0.021	0.317	0.157
29. Good Rough Grass / Poor Rough Grass	0.051	-0.468	-0.110	-0.466	-0.349	-0.371	-0.364	-0.182	-0.405	-0.306
30. Improved Grassland / Good Rough Grass	0.046	-0.161	0.029	-0.261	0.234	0.232	0.238	0.223	0.247	0.263
31. Good Rough Grass / Bracken	0.065	0.295	0.282	0.223	0.177	0.361	0.252	0.054	0.379	0.208
32. Poor Rough Grassland / Peatland	0.133	0.104	0.261	0.046	-0.127	-0.012	-0.084	-0.093	0.033	-0.049
33. Heather Moorland / Montane	0.022	**0.558**	0.131	0.331	0.418	0.397	0.417	0.146	0.278	0.224
34. Remaining Mosaics	-0.013	-0.018	0.047	-0.085	0.400	0.373	0.396	0.215	0.325	0.292

The patterns of correlations in Table A2-7 broadly matched what was expected from the known ecology of golden eagles. It would appear that the highest productivity was achieved where there is less arable land and improved grassland, but an abundance of rocks and cliffs combined with larger areas of heather moorland. The similarity of these findings to those for range occupancy (Annex 4) suggests that the higher quality ranges, where productivity is greater, are also likely to be those which are occupied more frequently (see also Sergio & Newton, 2003). If the analysis was restricted to those ranges that were successful a different pattern of variables emerged. These may considered to be factors associated with ‘twin’ frequency. Thus, twins were more likely when there was less arable, marsh and urban land but more heather moorland and montane habitats.

In order to obtain reasonable sample sizes for productivity differences three production classes, using data pooled from 1982 and 1992, were established:

1. 0 young fledged,
2. 1 young fledged
3. 2 or more fledged.

Results are summarized below.

Even in those NHZ where a feature's means differed significantly between production classes there were few general trends apparent (Table A2-8). Features that appeared important in one NHZ were insignificant in others. While this may be partly due to differences in statistical power, detailed examination of the mean values does not provide evidence for general, across-NHZ, trends and may result from regional differences in the importance of different food types, for example (Watson, 1997). Consequently, analyses of factors affecting productivity undertaken at the national scale were unlikely to identify important regional influences. It is also important to realize that causal relationships cannot be inferred from any of these analyses. It is probable that some significant relationships arose because of correlations with other single or multifactorial processes that were not directly measured. For example, the relationships with the topographic and solid geology variables probably related to the effects that topography and geology have on land use rather than their direct effects on golden eagle productivity.

*Table A2-8. Habitat features in which mean values for different production classes (0, 1 and 2+ young fledged in 1982 & 1992 combined) differed significantly in at least one NHZ . P values are indicated by asterisks: * = p<0.05, ** = p<0.01, *** = p<0.001. Data are presented for two data extraction scales: 3 km and 6 km radius circles.*

Feature	General trend with increasing production (only for those NHZ marked by an asterisk)	Natural Heritage Zone[1]											
3 km		3	4	5	6	7	8	10	11	12	14	15	All
Topographic													
Min. altitude	0 and 2+ have same mean												**
Max.altitude	Higher altitudes			**	*								
Altitude sd	Greater variability			*									*
Mean slope	Greater slopes			*									**
Max. slope	Greater slopes			*	*				*				***
Slope sd	Greater variability			*							*		***
Forest													
Closed	Increasing cover		**										
Broadleaf	Increasing cover		***										
Other													
Bracken	Increasing cover									**	*		
Human	No simple pattern						*						
Montane	Increasing cover				**								
Wetland	No simple pattern	*											
Heathland													
Burnt	Decreasing cover												**
Dry	No simple pattern		***		*								
Undifferent.	No simple pattern							*					
Grassland													
Improved	Decreasing cover												*

	Feature	General trend with increasing production (only for those NHZ marked by an asterisk)	Natural Heritage Zone[1]											
3 km			3	4	5	6	7	8	10	11	12	14	15	All
	Grazing													
	Calves	Increasing density	*											
	Sheep	Increasing density									**			
	Offtake	No simple trend				*								*
	Solid geology													
	Basic igneous	Increasing area					**							**
	Gneisses	Increasing area				*								
	Limestones	Increasing area						**						
	Meta. igneous	Increasing area									**			
	Meta. sedimen.	No simple trend												*
	Mixed metam.	Increasing area						***						
6km			3	4	5	6	7	8	10	11	12	14	15	all
	Topographic													
	Mean altitude	0 and 2+ have same mean			*									
	Min. altitude	0 and 2+ have same mean							*					**
	Max.altitude	Increasing area			**									
	Mean slope	Greater slopes			**									**
	Max. slope	Greater slopes			*									***
	Slope sd	Greater variability			*							*		**
	Forest													
	Open	No simple trend							**					
	Other													
	Arable	Decreasing cover												*
	Bracken	Increasing cover									**			
	Cliff	Increasing amount			*									*
	Montane	No simple trend			*									
	Wetland	No simple trend	*											
	Heathland													
	Burnt	No simple trend									**			**
	Dry	No simple trend		***										**
	Grassland													
	Coarse	No simple trend			**									
	Improved	No simple trend				*	*							**
	Grazing													
	Cattle	Decreasing density												*
	Sheep	Increasing density									*			
	Offtake	No simple trend												*

[1] NHZ key: 3 - Western Isles; 4 - NW Seaboard; 5 - Caithness & Sutherland; 6 - Western Seaboard; 7 - Northern Highlands; 8 - Western Highlands; 10 - Central Highlands; 11 -Cairngorm Massif; 12 - North East Glens; 14 - Argyll West and Islands; 15 - Breadalbane and East Argyll

The habitat data extracted at the wider 6 km scale failed to identify as many significant variables and many of the significant variables lacked clear trends. For example, the means for montane habitat in the Caithness and Sutherland NHZ were 50.5 (0 fledged), 167.2 (1 fledged) and 3.6 (2 or more fledged). This failure to pick up consistent trends at the broader scale was perhaps not too surprising since it is known that successful pairs tend to stay closer to the nest (Haworth *et al*., 2006). Thus, the 6 km buffers will have included more areas that were not used by foraging birds.

CONCLUSIONS

- Using data from a number of sources across a large number of years, it was apparent that average productivity of golden eagles in Scotland was typically about 0.46 young per occupied territory per year.

- Data from the national surveys could be used as a reasonable approximation of relative productivity over the intervening years.

- There was no significant correlation between regional productivity in the 1982 and 1992 survey years.

- Previous studies have concluded that golden eagle productivity is higher in the east of Scotland. Although several territories in the eastern Highlands were highly productive, high productivity was by no means universal in eastern regions and numerous territories in the western Highlands and Islands regions were also relatively productive. The majority of regional contributions to the national output of young birds came from western regions: the disparity between eastern and western regions in this regard has become even more marked as numbers of occupied territories continue to decline in the east. If productivity was measured per unit area, rather than on a per territory basis, the most productive areas were in the west.

- Nationally, it appeared that the highest productivity was achieved where there was less arable land and improved grassland, but an abundance of rocks and cliffs combined with larger areas of heather moorland. If the analysis was restricted to those ranges that were successful a different pattern of variables emerged. Thus, ‘twins’ were more likely when there was less arable, marsh and urban land but more heather moorland and montane habitats. The most productive territories nationally had similar land cover types to those which were occupied, suggesting that ‘higher quality’ territories were more likely to be occupied.

- Simple analyses failed to find any consistent influences of land cover on productivity at regional levels: features that appeared important in one NHZ were insignificant in others. Consequently, analyses of factors affecting productivity undertaken at the national scale were unlikely to identify important regional influences and more sophisticated approaches would be beneficial to identify those factors influencing productivity at regional and local scales.

ANNEX 3: POPULATION SIMULATIONS FOR DEMOGRAPHIC TESTS OF FAVOURABLE CONSERVATION STATUS

Population simulations were run with the GEPM for 11 NHZ. Each simulation produced a mean population size for years 21 – 30 of a 30 year simulation. Populations with less than 10 active ranges were not simulated. Each combination of subadult survival, adult survival and fledging rate was simulated 100 times. Results are summarised as the mean population sizes.

Figures in bold show conditions under which the population is predicted to remain stable or expand. Boxed results show those that most closely match the 2003 and the mean fledging rates (1982, 1992 and 2003).

FR = Fledging rate (young fledged / occupied territories / year), TR = Adult turnover rate which is also given as LE (Life expectancy of an adult bird as the number of years on territory), JS = Juvenile (subadult) survival rate, which is the proportion of fledged birds that survive to their fourth year).

Western Isles NHZ

Starting population = 81 pairs, population cap = 95 pairs. The fledging rates were 0.35 (2003) and 0.33 (mean 1982, 1992, 2003).

			FR							
JS	TR	LE	0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.30	0.040	25.0	68.2	74.3	80.3	**86.7**	**92.1**	**94.5**	**95.1**	**95.2**
	0.045	22.2	59.9	65.5	70.8	75.7	**82.0**	**88.2**	**93.0**	**94.9**
	0.050	20.0	52.4	57.1	62.5	67.3	73.0	78.5	**84.7**	**91.1**
	0.055	18.2	46.4	50.2	54.9	59.6	64.9	70.0	75.5	**81.5**
	0.060	16.7	40.5	44.1	48.0	52.1	57.4	62.2	67.1	72.3
	0.065	15.4	35.7	38.5	42.1	46.3	50.0	54.7	59.5	64.4
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.35	0.040	25.0	76.9	**84.1**	**90.6**	**94.4**	**95.1**	**95.2**	**95.1**	**95.0**
	0.045	22.2	67.4	73.3	80.2	**87.8**	**93.6**	**94.9**	**95.2**	**95.1**
	0.050	20.0	58.7	64.9	70.8	78.3	**84.9**	**91.7**	**94.6**	**95.1**
	0.055	18.2	51.5	57.0	62.2	69.2	75.6	**82.9**	**88.9**	**93.7**
	0.060	16.7	45.2	50.7	54.8	60.5	67.3	72.9	79.5	**86.6**
	0.065	15.4	39.3	43.9	48.5	53.8	59.1	64.8	70.6	76.9
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.40	0.040	25.0	**85.7**	**92.3**	**94.9**	**95.2**	**95.1**	**95.1**	**95.1**	**95.1**
	0.045	22.2	22.2	**81.5**	**90.6**	**94.5**	**95.2**	**95.1**	**95.2**	**95.2**
	0.050	20.0	20.0	73.2	**81.4**	**89.1**	**93.7**	**95.1**	**95.1**	**95.1**
	0.055	18.2	18.2	64.1	71.6	78.6	**87.2**	**93.2**	**94.7**	**95.2**
	0.060	16.7	16.7	57.0	62.8	69.8	77.5	**85.1**	**91.9**	**94.7**
	0.065	15.4	15.4	49.3	55.3	61.9	69.1	75.7	**83.4**	**90.5**
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.45	0.040	25.0	**92.6**	**95.1**	**95.2**	**95.1**	**95.1**	**95.2**	**95.0**	**95.0**
	0.045	22.2	**83.4**	**91.5**	**94.9**	**95.1**	**95.1**	**95.1**	**95.2**	**95.1**
	0.050	20.0	72.8	**82.3**	**91.3**	**94.4**	**95.3**	**95.1**	**95.1**	**95.1**
	0.055	18.2	63.8	72.9	**81.3**	**90.0**	**94.3**	**95.3**	**95.1**	**95.0**
	0.060	16.7	56.5	62.9	71.4	80.0	**88.8**	**93.6**	**95.1**	**95.3**
	0.065	15.4	48.7	55.9	63.5	71.1	78.5	**87.7**	**93.6**	**95.0**
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.50	0.040	25.0	**95.0**	**95.2**	**95.1**	**95.1**	**95.2**	**95.1**	**94.9**	**94.9**
	0.045	22.2	**91.2**	**94.9**	**95.2**	**95.1**	**95.2**	**95.1**	**95.1**	**95.0**
	0.050	20.0	**81.3**	**91.0**	**94.7**	**95.2**	**95.1**	**95.1**	**95.1**	**95.0**
	0.055	18.2	71.6	**80.6**	**90.9**	**94.8**	**95.1**	**95.0**	**95.1**	**95.1**
	0.060	16.7	62.6	71.2	80.6	**89.8**	**94.6**	**95.2**	**95.1**	**95.1**
	0.065	15.4	54.4	62.1	71.6	80.6	**89.6**	**94.3**	**95.1**	**95.2**

North West Seaboard NHZ

Starting population = 46 pairs, population cap = 72 pairs. The fledging rates were 0.33 (2003) and 0.39 (mean 1982, 1992, 2003).

			FR							
JS	TR	LE	0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.30	0.040	25.0	39.3	43.1	**48.8**	**54.3**	**58.4**	**63.0**	**66.7**	**70.0**
	0.045	22.2	35.9	40.6	44.9	**49.4**	**52.5**	**55.8**	**58.8**	**62.2**
	0.050	20.0	32.0	35.5	39.6	42.7	45.8	**48.4**	**51.5**	**54.7**
	0.055	18.2	28.4	30.7	33.9	37.0	40.1	42.8	45.4	**48.4**
	0.060	16.7	24.6	27.2	30.0	32.5	35.3	37.8	40.2	42.9
	0.065	15.4	21.9	24.0	26.1	28.5	31.2	33.5	36.0	38.7
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.35	0.040	25.0	45.4	**51.6**	**57.6**	**62.6**	**66.9**	**70.5**	**71.6**	**71.8**
	0.045	22.2	42.2	**47.2**	**51.6**	**55.1**	**59.5**	**63.7**	**68.1**	**70.6**
	0.050	20.0	36.7	40.9	44.8	**47.9**	**50.8**	**55.3**	**60.5**	**65.6**
	0.055	18.2	31.6	35.1	39.4	41.7	44.9	**48.9**	**54.0**	**58.7**
	0.060	16.7	27.6	31.4	33.8	37.2	40.2	43.2	**47.7**	**52.9**
	0.065	15.4	24.4	27.4	30.1	32.9	35.7	38.8	42.8	**47.0**
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.40	0.040	25.0	**52.8**	**59.7**	**64.7**	**69.1**	**71.4**	**71.9**	**72.0**	**72.1**
	0.045	22.2	22.2	**52.8**	**56.8**	**61.6**	**67.4**	**70.4**	**71.7**	**72.0**
	0.050	20.0	20.0	45.9	**49.3**	**53.7**	**58.1**	**65.1**	**69.5**	**71.5**
	0.055	18.2	18.2	39.9	43.5	**46.9**	**51.6**	**57.8**	**63.8**	**68.8**
	0.060	16.7	16.7	35.0	38.2	41.5	45.7	**51.5**	**57.3**	**63.0**
	0.065	15.4	15.4	30.6	33.9	37.3	40.8	**46.1**	**50.8**	**56.0**
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.45	0.040	25.0	**59.3**	**65.6**	**70.1**	**71.7**	**71.9**	**72.0**	**72.1**	**72.0**
	0.045	22.2	**52.7**	**57.3**	**62.4**	**68.8**	**71.3**	**71.8**	**72.0**	**72.1**
	0.050	20.0	45.8	**50.2**	**54.9**	**60.7**	**67.8**	**70.9**	**71.8**	**72.0**
	0.055	18.2	39.7	43.8	**47.8**	**53.9**	**60.7**	**67.6**	**70.9**	**71.8**
	0.060	16.7	34.6	38.4	42.7	**48.3**	**54.8**	**60.6**	**66.6**	**70.5**
	0.065	15.4	30.7	34.1	37.3	42.2	**48.1**	**53.7**	**59.7**	**65.2**
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.50	0.040	25.0	**65.1**	**70.0**	**71.8**	**72.0**	**72.1**	**71.9**	**71.9**	**72.0**
	0.045	22.2	**57.0**	**63.8**	**69.3**	**71.7**	**72.0**	**72.1**	**72.1**	**72.0**
	0.050	20.0	**49.1**	**54.6**	**61.8**	**69.0**	**71.5**	**71.9**	**72.0**	**72.0**
	0.055	18.2	43.5	**48.3**	**54.5**	**62.1**	**68.8**	**71.3**	**71.9**	**72.0**
	0.060	16.7	37.9	42.0	**47.9**	**55.2**	**62.1**	**68.4**	**71.1**	**71.9**
	0.065	15.4	33.3	37.5	42.8	**48.8**	**55.2**	**61.7**	**68.3**	**70.8**

The Peatlands of Caithness and Sutherland NHZ

Starting population = 18 pairs, population cap = 31 pairs. The fledging rates were 0.39 (2003) and 0.32 (mean 1982, 1992, 2003).

			FR							
JS	TR	LE	0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.30	0.040	25.0	11.1	12.3	13.2	14.6	17.1	**19.4**	**21.2**	**23.9**
	0.045	22.2	7.7	10.5	11.8	13.7	16.9	**19.3**	**21.5**	**24.0**
	0.050	20.0	5.7	8.1	11.4	13.6	16.7	**19.3**	**21.4**	**23.9**
	0.055	18.2	3.2	6.5	10.8	13.8	16.9	**19.5**	**21.4**	**23.5**
	0.060	16.7	1.8	6.2	9.8	13.4	17.0	**19.0**	**20.6**	**22.2**
	0.065	15.4	1.1	4.5	9.4	12.5	15.5	17.7	**18.9**	**20.0**
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.35	0.040	25.0	12.4	13.4	15.9	**19.2**	**21.3**	**24.7**	**27.6**	**29.9**
	0.045	22.2	10.3	12.7	15.5	**18.8**	**21.0**	**24.8**	**27.8**	**29.9**
	0.050	20.0	7.5	12.1	14.9	**19.0**	**21.3**	**24.1**	**27.5**	**29.3**
	0.055	18.2	5.8	11.8	15.9	**18.9**	**21.2**	**24.2**	**25.6**	**27.5**
	0.060	16.7	5.6	11.4	15.0	**18.3**	**20.5**	**22.4**	**24.0**	**24.8**
	0.065	15.4	3.9	10.2	13.8	16.6	**19.0**	**20.4**	**21.4**	**22.6**
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.40	0.040	25.0	14.2	17.2	**20.1**	**23.2**	**26.9**	**29.8**	**30.7**	**31.0**
	0.045	22.2	12.9	16.7	**19.9**	**22.8**	**27.1**	**29.7**	**30.6**	**31.0**
	0.050	20.0	12.4	17.1	**20.0**	**22.9**	**26.9**	**29.2**	**30.2**	**30.8**
	0.055	18.2	12.0	16.7	**20.1**	**22.7**	**25.5**	**27.2**	**28.6**	**29.8**
	0.060	16.7	12.0	16.4	**19.4**	**21.7**	**23.6**	**24.7**	**25.8**	**27.3**
	0.065	15.4	9.4	15.2	**18.1**	**19.9**	**21.0**	**22.3**	**23.1**	**24.0**
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.45	0.040	25.0	16.9	**20.3**	**24.4**	**28.3**	**30.5**	**30.9**	**31.0**	**30.9**
	0.045	22.2	15.7	**20.2**	**24.2**	**28.0**	**30.3**	**31.0**	**31.0**	**31.0**
	0.050	20.0	16.2	**20.3**	**24.0**	**27.7**	**29.8**	**30.6**	**31.0**	**31.1**
	0.055	18.2	16.1	**20.3**	**23.6**	**26.1**	**28.1**	**29.0**	**30.1**	**30.9**
	0.060	16.7	15.7	**19.6**	**21.7**	**24.0**	**25.4**	**26.7**	**28.0**	**29.3**
	0.065	15.4	13.8	**18.2**	**20.1**	**21.5**	**22.9**	**23.7**	**25.0**	**26.2**
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.50	0.040	25.0	**19.6**	**24.6**	**29.1**	**30.7**	**31.0**	**31.0**	**30.9**	**30.8**
	0.045	22.2	**19.4**	**24.6**	**28.6**	**30.5**	**31.1**	**31.0**	**31.0**	**30.9**
	0.050	20.0	**19.5**	**24.4**	**27.9**	**30.1**	**30.9**	**31.1**	**31.1**	**31.1**
	0.055	18.2	**19.6**	**23.6**	**26.4**	**28.2**	**29.6**	**30.6**	**31.1**	**31.2**
	0.060	16.7	**18.8**	**22.3**	**24.2**	**25.6**	**26.8**	**28.3**	**30.1**	**30.9**
	0.065	15.4	17.3	**20.1**	**21.6**	**22.9**	**23.8**	**25.1**	**27.5**	**29.0**

Western Seaboard NHZ

Starting population = 74 pairs, population cap = 100 pairs. The fledging rates were 0.46 (2003) and 0.44 (mean 1982, 1992, 2003).

			FR							
JS	TR	LE	0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.30	0.040	25.0	63.4	68.7	**75.4**	**82.6**	**88.7**	**95.2**	**98.7**	**99.8**
	0.045	22.2	55.8	60.6	66.3	71.8	**77.8**	**83.7**	**91.5**	**96.9**
	0.050	20.0	49.1	53.4	58.6	63.2	68.1	**74.4**	**81.2**	**87.3**
	0.055	18.2	43.4	47.1	51.4	55.5	60.2	66.2	71.4	**77.8**
	0.060	16.7	38.1	41.3	45.2	49.5	53.7	58.3	63.3	68.5
	0.065	15.4	33.7	36.6	40.4	43.0	47.1	51.5	55.9	60.6
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.35	0.040	25.0	71.5	**79.6**	**87.1**	**94.1**	**98.9**	**99.9**	**100.0**	**100.0**
	0.045	22.2	62.8	68.9	**76.5**	**83.8**	**90.7**	**97.4**	**99.5**	**100.0**
	0.050	20.0	55.0	61.1	66.6	73.5	**80.6**	**88.5**	**96.2**	**99.0**
	0.055	18.2	48.3	53.4	58.1	64.9	71.3	**78.5**	**85.5**	**92.5**
	0.060	16.7	42.4	46.7	52.3	57.3	62.9	69.8	**76.5**	**82.6**
	0.065	15.4	37.3	41.2	45.9	50.5	55.3	61.6	67.7	73.6
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.40	0.040	25.0	**80.6**	**89.4**	**96.7**	**99.6**	**99.9**	**100.1**	**100.1**	**100.2**
	0.045	22.2	22.2	**78.7**	**86.3**	**95.5**	**99.4**	**99.9**	**100.0**	**100.1**
	0.050	20.0	20.0	68.4	**76.0**	**85.4**	**93.7**	**98.7**	**100.0**	**100.1**
	0.055	18.2	18.2	60.1	67.5	**75.5**	**83.0**	**91.3**	**97.3**	**99.8**
	0.060	16.7	16.7	52.9	59.2	66.2	73.4	**81.6**	**89.9**	**96.1**
	0.065	15.4	15.4	46.9	52.0	58.7	64.7	72.7	**80.5**	**87.9**
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.45	0.040	25.0	**90.0**	**97.5**	**99.8**	**100.0**	**100.1**	**100.1**	**100.0**	**99.9**
	0.045	22.2	**78.7**	**88.2**	**96.5**	**99.7**	**100.0**	**100.0**	**100.1**	**100.1**
	0.050	20.0	68.3	**77.1**	**87.6**	**96.2**	**99.5**	**100.0**	**100.1**	**100.0**
	0.055	18.2	60.4	67.4	**77.6**	**86.3**	**94.8**	**99.4**	**100.0**	**100.1**
	0.060	16.7	52.7	59.5	67.4	**76.0**	**86.2**	**94.2**	**98.8**	**99.9**
	0.065	15.4	46.3	52.2	59.2	66.9	**75.8**	**84.3**	**92.4**	**98.1**
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.50	0.040	25.0	**97.5**	**99.8**	**100.0**	**100.1**	**100.2**	**100.1**	**99.9**	**99.8**
	0.045	22.2	**86.9**	**96.6**	**99.8**	**100.0**	**100.1**	**100.1**	**100.1**	**99.9**
	0.050	20.0	**76.4**	**87.4**	**96.3**	**99.7**	**100.0**	**99.9**	**100.0**	**100.1**
	0.055	18.2	67.1	**77.0**	**88.1**	**96.5**	**99.7**	**100.0**	**100.0**	**100.0**
	0.060	16.7	59.1	67.3	**76.6**	**87.4**	**95.6**	**99.5**	**100.1**	**100.0**
	0.065	15.4	51.1	58.9	66.3	**77.5**	**87.1**	**95.4**	**99.1**	**100.0**

Northern Highlands NHZ

Starting population = 43 pairs, population cap = 90 pairs. The fledging rates were 0.28 (2003) and 0.37 (mean 1982, 1992, 2003).

			FR							
JS	TR	LE	0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.30	0.040	25.0	38.0	41.7	**46.6**	**52.2**	**57.9**	**61.8**	**66.9**	**71.1**
	0.045	22.2	34.0	38.5	**43.0**	**47.9**	**52.1**	**55.2**	**58.7**	**62.2**
	0.050	20.0	30.6	34.1	37.7	42.0	**45.3**	**47.9**	**51.0**	**54.6**
	0.055	18.2	27.0	29.8	33.1	36.7	39.8	42.2	**44.9**	**47.7**
	0.060	16.7	23.5	25.8	28.6	31.7	34.8	37.4	40.0	42.1
	0.065	15.4	20.6	22.7	24.8	27.7	30.6	33.0	35.6	38.0
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.35	0.040	25.0	**43.2**	**49.1**	**56.7**	**61.7**	**66.7**	**71.8**	**77.4**	**82.8**
	0.045	22.2	40.0	**46.0**	**50.8**	**54.3**	**58.2**	**62.9**	**68.4**	**74.2**
	0.050	20.0	35.4	40.1	**44.6**	**47.9**	**50.9**	**55.0**	**59.9**	**64.4**
	0.055	18.2	30.3	35.0	38.4	41.8	**45.1**	**48.3**	**53.5**	**57.7**
	0.060	16.7	26.5	29.8	33.6	36.8	39.8	42.7	**47.0**	**51.9**
	0.065	15.4	23.2	26.1	29.2	32.5	35.1	38.0	41.8	**45.8**
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.40	0.040	25.0	**51.3**	**58.4**	**64.0**	**69.7**	**76.2**	**82.0**	**86.1**	**88.3**
	0.045	22.2	22.2	**52.3**	**56.7**	**61.0**	**66.7**	**72.9**	**79.0**	**84.6**
	0.050	20.0	20.0	**45.0**	**49.1**	**52.7**	**58.3**	**64.8**	**71.2**	**77.2**
	0.055	18.2	18.2	39.7	**43.1**	**46.4**	**51.5**	**57.2**	**63.0**	**68.8**
	0.060	16.7	16.7	34.4	37.7	41.4	**45.8**	**50.7**	**56.2**	**61.8**
	0.065	15.4	15.4	30.1	32.8	36.8	40.0	**44.8**	**50.4**	**54.5**
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.45	0.040	25.0	**58.0**	**65.3**	**71.8**	**79.3**	**84.6**	**87.7**	**89.1**	**89.5**
	0.045	22.2	**52.6**	**57.0**	**62.3**	**68.9**	**77.3**	**82.8**	**86.9**	**88.7**
	0.050	20.0	**45.4**	**49.9**	**54.2**	**59.7**	**67.2**	**75.1**	**81.4**	**86.4**
	0.055	18.2	39.3	42.9	**47.4**	**52.8**	**60.0**	**66.9**	**73.8**	**81.2**
	0.060	16.7	34.3	37.9	41.7	**46.9**	**53.7**	**59.5**	**67.0**	**73.2**
	0.065	15.4	29.9	33.8	37.3	41.6	**47.2**	**52.6**	**57.6**	**64.3**
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.50	0.040	25.0	**64.5**	**72.0**	**79.8**	**85.8**	**88.5**	**89.3**	**89.6**	**89.8**
	0.045	22.2	**56.4**	**62.8**	**70.1**	**78.6**	**84.7**	**88.1**	**89.3**	**89.6**
	0.050	20.0	**49.2**	**54.3**	**61.4**	**69.5**	**77.3**	**84.3**	**87.5**	**89.2**
	0.055	18.2	42.9	**47.6**	**53.9**	**61.0**	**68.8**	**76.9**	**84.3**	**87.8**
	0.060	16.7	37.5	42.0	**47.8**	**55.0**	**61.7**	**68.4**	**75.9**	**83.7**
	0.065	15.4	33.0	37.0	41.8	**48.4**	**54.5**	**60.1**	**68.1**	**76.4**

Western Highlands NHZ

Starting population = 51 pairs, population cap = 70 pairs. The fledging rates were 0.16 (2003) and 0.20 (mean 1982, 1992, 2003).

							FR				
JS	TR	LE	0.200	0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.30	0.040	25.0	37.1	45.2	50.4	**55.3**	**59.6**	**63.8**	**67.0**	**68.8**	**69.8**
	0.045	22.2	33.0	40.3	44.7	48.5	**52.1**	**55.3**	**58.7**	**62.4**	**66.7**
	0.050	20.0	28.8	34.7	38.1	42.0	44.9	48.2	**51.2**	**54.7**	**60.0**
	0.055	18.2	25.2	30.1	33.4	36.8	39.9	42.8	45.3	49.4	**54.1**
	0.060	16.7	22.5	26.5	29.1	32.5	35.2	37.9	41.0	44.0	47.8
	0.065	15.4	19.6	23.4	25.7	28.1	30.9	33.5	36.1	38.9	42.6
			0.200	0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.35	0.040	25.0	40.5	**52.4**	**58.0**	**62.2**	**66.1**	**69.0**	**70.1**	**70.0**	**70.1**
	0.045	22.2	36.2	46.4	50.5	**54.0**	**58.0**	**62.2**	**66.9**	**69.2**	**70.0**
	0.050	20.0	31.5	39.9	43.5	47.2	50.7	**55.1**	**60.4**	**65.7**	**68.9**
	0.055	18.2	27.6	34.5	38.4	41.8	44.7	49.2	**54.5**	**59.7**	**64.8**
	0.060	16.7	24.1	30.0	33.8	37.2	39.6	43.8	48.7	**53.1**	**57.5**
	0.065	15.4	21.1	26.4	29.5	32.8	35.5	38.5	43.0	47.0	**51.1**
			0.200	0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.40	0.040	25.0	45.9	**59.1**	**63.9**	**68.1**	**69.8**	**70.1**	**70.1**	**70.1**	**70.1**
	0.045	22.2	40.9	**51.2**	**55.3**	**59.7**	**65.0**	**69.1**	**69.9**	**70.1**	**70.0**
	0.050	20.0	34.9	44.3	48.2	**52.6**	**57.8**	**64.8**	**68.7**	**69.8**	**70.1**
	0.055	18.2	30.1	38.6	42.5	46.4	**51.7**	**57.8**	**64.1**	**67.7**	**69.8**
	0.060	16.7	26.0	34.1	37.4	41.5	46.1	**51.5**	**57.2**	**61.9**	**67.3**
	0.065	15.4	22.8	29.6	33.3	36.7	40.9	45.1	49.9	**55.3**	**59.8**
			0.200	0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.45	0.040	25.0	**51.8**	**64.2**	**68.3**	**70.0**	**70.1**	**70.0**	**70.1**	**70.1**	**70.0**
	0.045	22.2	44.8	**55.7**	**60.8**	**66.8**	**69.6**	**70.1**	**70.0**	**70.1**	**70.2**
	0.050	20.0	38.4	48.2	**53.0**	**59.6**	**66.5**	**69.5**	**70.0**	**70.1**	**70.1**
	0.055	18.2	33.4	42.1	46.7	**52.9**	**60.7**	**66.0**	**69.3**	**69.9**	**70.1**
	0.060	16.7	28.9	37.2	41.9	47.3	**53.1**	**59.2**	**64.9**	**68.8**	**70.0**
	0.065	15.4	24.7	32.9	36.7	41.3	46.9	**52.8**	**57.5**	**64.2**	**67.9**
			0.200	0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425
0.50	0.040	25.0	**57.0**	**68.1**	**69.9**	**70.1**	**70.1**	**70.1**	**70.1**	**69.9**	**69.9**
	0.045	22.2	49.0	**60.0**	**66.7**	**69.7**	**70.1**	**70.1**	**70.2**	**70.0**	**70.0**
	0.050	20.0	42.1	**52.3**	**59.9**	**67.3**	**69.6**	**70.1**	**70.1**	**70.1**	**70.1**
	0.055	18.2	36.6	46.4	**52.6**	**61.2**	**66.8**	**69.7**	**70.1**	**70.1**	**70.1**
	0.060	16.7	31.6	41.0	46.9	**53.1**	**60.4**	**66.5**	**69.6**	**70.0**	**70.1**
	0.065	15.4	27.3	36.2	40.7	47.3	**53.3**	**59.4**	**66.2**	**69.3**	**70.1**

Central Highlands NHZ

Starting population = 12 pairs, population cap = 26 pairs. The fledging rates were 0.83 (2003) and 0.47 (mean 1982, 1992, 2003).

			FR									
JS	TR	LE	0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425	0.450	0.475
0.30	0.040	25.0	9.0	10.4	11.6	11.8	**12.3**	**12.6**	**12.7**	**13.0**	**13.9**	**14.5**
	0.045	22.2	3.6	7.4	9.5	9.6	11.0	11.2	11.6	11.7	**12.1**	**12.7**
	0.050	20.0	0.5	3.1	6.9	8.2	9.2	9.6	10.2	10.2	10.5	10.9
	0.055	18.2	0.0	0.5	3.7	6.0	7.6	8.6	9.2	9.3	9.5	10.0
	0.060	16.7	0.0	0.1	0.8	3.7	5.1	6.9	7.5	8.0	8.5	9.1
	0.065	15.4	0.0	0.0	0.1	1.0	3.4	5.4	6.6	7.1	7.8	8.9
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425	0.450	0.475
0.35	0.040	25.0	10.3	11.4	11.9	**12.1**	**12.8**	**13.0**	**14.2**	**14.9**	**15.8**	**17.8**
	0.045	22.2	4.9	8.9	11.1	11.5	11.3	11.6	**12.2**	**12.7**	**14.1**	**15.3**
	0.050	20.0	0.8	5.4	7.5	9.5	10.4	10.4	10.6	11.1	**12.3**	**13.6**
	0.055	18.2	0.1	1.1	4.8	8.0	8.3	9.5	9.4	10.0	11.3	**13.2**
	0.060	16.7	0.0	0.5	1.2	5.2	7.3	8.1	8.7	9.2	11.2	**13.3**
	0.065	15.4	0.0	0.1	0.2	2.1	4.3	6.6	7.8	8.9	10.9	**13.1**
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425	0.450	0.475
0.40	0.040	25.0	10.2	11.7	**12.4**	**13.4**	**14.0**	**14.8**	**15.2**	**17.4**	**21.0**	**22.6**
	0.045	22.2	6.0	9.2	11.2	11.7	**12.1**	**13.0**	**13.7**	**15.6**	**17.4**	**20.4**
	0.050	20.0	1.1	5.2	8.8	10.1	10.6	11.1	**12.2**	**13.5**	**16.3**	**19.2**
	0.055	18.2	0.0	2.0	5.3	7.3	9.4	10.0	11.9	**13.5**	**15.7**	**18.5**
	0.060	16.7	0.1	0.2	2.2	4.3	7.7	9.2	11.2	**13.4**	**15.6**	**18.5**
	0.065	15.4	0.0	0.0	0.2	3.0	5.2	8.1	11.1	**13.2**	**15.8**	**18.5**
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425	0.450	0.475
0.45	0.040	25.0	11.0	**12.6**	**12.6**	**14.0**	**15.3**	**17.1**	**19.7**	**22.5**	**24.1**	**24.7**
	0.045	22.2	5.7	10.1	11.4	11.9	**13.1**	**14.4**	**17.2**	**19.5**	**22.8**	**24.5**
	0.050	20.0	1.1	5.7	8.9	10.0	11.0	**12.7**	**15.2**	**18.4**	**22.0**	**23.9**
	0.055	18.2	0.6	2.5	5.8	8.7	10.0	11.8	**14.7**	**18.0**	**21.8**	**24.3**
	0.060	16.7	0.0	0.9	2.7	5.7	8.9	11.5	**14.9**	**17.7**	**21.5**	**23.8**
	0.065	15.4	0.1	0.2	1.0	4.4	8.2	11.8	**14.6**	**18.2**	**21.0**	**23.2**
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425	0.450	0.475
0.50	0.040	25.0	11.7	**13.2**	**14.0**	**15.0**	**17.9**	**21.2**	**23.2**	**24.9**	**25.1**	**25.6**
	0.045	22.2	8.5	11.7	**12.3**	**12.9**	**15.0**	**18.6**	**21.6**	**23.6**	**25.4**	**25.5**
	0.050	20.0	4.3	8.2	10.8	11.5	**13.1**	**16.3**	**20.4**	**23.3**	**25.2**	**25.7**
	0.055	18.2	1.7	4.9	8.3	10.1	**12.5**	**15.8**	**19.6**	**23.5**	**24.9**	**25.7**
	0.060	16.7	0.6	3.0	6.2	9.6	**12.2**	**15.8**	**19.3**	**23.2**	**24.7**	**25.6**
	0.065	15.4	0.6	1.8	4.4	8.8	11.8	**15.6**	**19.5**	**22.5**	**24.3**	**25.2**

Cairngorms Massif NHZ

Starting population = 28 pairs, population cap = 71 pairs. The fledging rates were 0.68 (2003) and 0.78 (mean 1982, 1992, 2003).

			FR				
JS	TR	LE	0.60	0.65	0.70	0.75	0.80
0.10	0.05	20.0	20.9	23.5	25.3	26.3	26.5
	0.06	18.2	17.7	19.7	21.8	23.1	23.7
	0.06	16.7	15.4	17.0	18.6	20.3	21.1
	0.07	15.4	13.6	14.9	16.1	17.5	18.9
	0.07	14.3	12.2	13.1	14.4	15.5	16.5
	0.08	13.3	11.2	12.1	12.8	13.4	14.8
0.15	0.05	20.0	**28.0**	**29.2**	**30.8**	**32.4**	**35.1**
	0.06	18.2	24.8	25.8	27.4	**29.1**	**31.5**
	0.06	16.7	22.1	23.1	24.7	26.5	**29.2**
	0.07	15.4	20.1	20.9	22.4	24.0	27.1
	0.07	14.3	17.8	19.2	20.4	21.7	24.1
	0.08	13.3	15.9	17.4	18.3	19.7	21.7
0.20	0.05	20.0	**34.4**	**39.1**	**44.5**	**48.8**	**53.9**
	0.06	18.2	**30.4**	**35.1**	**40.6**	**45.4**	**50.0**
	0.06	16.7	27.9	**31.9**	**37.3**	**41.1**	**44.7**
	0.07	15.4	25.5	**29.5**	**33.9**	**37.2**	**40.1**
	0.07	14.3	22.7	26.5	**30.2**	**33.4**	**35.5**
	0.08	13.3	20.8	23.7	26.8	**29.6**	**31.7**
0.25	0.05	20.0	**48.5**	**54.5**	**60.1**	**65.2**	**67.9**
	0.06	18.2	**44.8**	**49.6**	**54.8**	**60.8**	**65.3**
	0.06	16.7	**40.3**	**44.7**	**48.5**	**54.8**	**59.9**
	0.07	15.4	**36.6**	**40.1**	**43.7**	**48.1**	**53.4**
	0.07	14.3	**32.2**	**35.0**	**38.7**	**43.3**	**47.8**
	0.08	13.3	**28.5**	**31.7**	**34.7**	**38.0**	**42.9**
0.30	0.05	20.0	**61.1**	**66.5**	**69.0**	**70.2**	**70.5**
	0.06	18.2	**55.8**	**62.5**	**67.4**	**69.2**	**70.2**
	0.06	16.7	**50.1**	**57.0**	**63.1**	**67.2**	**69.5**
	0.07	15.4	**44.3**	**50.1**	**57.0**	**62.4**	**67.4**
	0.07	14.3	**38.8**	**44.8**	**51.0**	**57.2**	**63.2**
	0.08	13.3	**34.8**	**40.0**	**45.7**	**51.6**	**58.3**

Lochaber NHZ

Starting population = 25 pairs, population cap = 36 pairs. The fledging rates were 0.16 (2003) and 0.30 (mean 1982, 1992, 2003).

			FR									
JS	TR	LE	0.150	0.175	0.200	0.250	0.275	0.300	0.325	0.350	0.375	0.400
0.30	0.040	25.0	4.2	13.4	15.9	23.0	**25.8**	**28.4**	**31.7**	**34.2**	**35.5**	**35.9**
	0.045	22.2	7.7	14.5	16.2	22.6	**25.6**	**27.6**	**30.0**	**31.9**	**33.0**	**34.1**
	0.050	20.0	7.5	13.9	15.5	20.6	23.5	**25.2**	**26.8**	**28.1**	**29.4**	**30.5**
	0.055	18.2	2.4	8.7	13.4	17.2	20.2	22.2	23.5	24.6	**25.7**	**26.6**
	0.060	16.7	0.0	4.3	9.8	13.7	16.3	19.1	20.6	21.8	22.5	23.5
	0.065	15.4	0.0	1.1	5.0	11.9	13.6	15.7	17.6	19.0	20.1	20.8
			0.150	0.175	0.200	0.250	0.275	0.300	0.325	0.350	0.375	0.400
0.35	0.040	25.0	6.8	16.2	18.9	**27.0**	**30.0**	**33.7**	**35.3**	**35.9**	**36.1**	**36.1**
	0.045	22.2	10.8	16.2	18.4	**26.5**	**29.1**	**31.6**	**32.9**	**34.2**	**35.0**	**35.7**
	0.050	20.0	9.1	15.2	17.3	24.3	**25.9**	**27.7**	**29.3**	**30.4**	**31.9**	**32.8**
	0.055	18.2	4.1	12.0	14.7	20.6	22.9	24.1	**25.4**	**26.5**	**27.7**	**28.8**
	0.060	16.7	1.1	6.7	11.3	16.7	19.6	21.4	22.3	23.5	24.4	**25.5**
	0.065	15.4	0.0	1.3	7.8	13.8	16.3	18.5	19.7	20.8	21.6	22.6
			0.150	0.175	0.200	0.250	0.275	0.300	0.325	0.350	0.375	0.400
0.40	0.040	25.0	8.5	18.5	23.3	**31.6**	**34.5**	**35.7**	**36.1**	**36.1**	**36.1**	**36.1**
	0.045	22.2	12.0	18.1	23.4	**29.5**	**32.0**	**33.7**	**34.8**	**35.5**	**36.0**	**36.2**
	0.050	20.0	11.2	16.5	21.1	**26.3**	**28.2**	**29.7**	**30.8**	**32.6**	**34.0**	**35.4**
	0.055	18.2	4.5	13.8	17.1	23.0	24.5	**26.1**	**27.1**	**28.5**	**30.1**	**32.0**
	0.060	16.7	1.2	6.7	14.0	20.2	21.5	22.9	24.0	24.9	**26.1**	**28.2**
	0.065	15.4	0.0	1.7	8.7	17.1	18.9	20.1	21.1	22.4	23.4	24.9
			0.150	0.175	0.200	0.250	0.275	0.300	0.325	0.350	0.375	0.400
0.45	0.040	25.0	11.8	20.9	**26.6**	**34.7**	**35.9**	**36.0**	**36.1**	**36.1**	**36.0**	**36.0**
	0.045	22.2	14.5	20.4	**26.6**	**32.1**	**33.8**	**35.0**	**35.7**	**36.0**	**36.2**	**36.1**
	0.050	20.0	13.0	18.2	24.1	**28.1**	**29.8**	**31.5**	**33.1**	**34.5**	**35.8**	**36.1**
	0.055	18.2	8.9	15.6	19.6	24.8	**25.9**	**27.5**	**28.8**	**30.7**	**33.3**	**35.3**
	0.060	16.7	3.0	10.4	15.8	21.6	23.0	24.3	**25.3**	**27.2**	**29.5**	**32.2**
	0.065	15.4	0.0	4.5	11.8	18.5	20.4	21.4	22.4	24.1	**26.0**	**29.9**
			0.150	0.175	0.200	0.250	0.275	0.300	0.325	0.350	0.375	0.400
0.50	0.040	25.0	19.4	**24.5**	**29.7**	**35.8**	**36.0**	**36.2**	**36.1**	**36.0**	**35.9**	**35.9**
	0.045	22.2	19.3	23.9	**28.8**	**33.6**	**35.0**	**35.9**	**36.1**	**36.2**	**36.0**	**36.0**
	0.050	20.0	17.7	21.8	**25.8**	**29.7**	**31.4**	**33.1**	**35.1**	**35.9**	**36.2**	**36.1**
	0.055	18.2	14.1	18.5	21.9	**26.0**	**27.6**	**29.0**	**31.2**	**33.8**	**35.6**	**36.1**
	0.060	16.7	8.4	14.8	18.3	22.9	24.2	**25.3**	**27.3**	**30.0**	**33.6**	**35.5**
	0.065	15.4	1.8	11.5	15.4	19.9	21.4	22.6	23.8	**26.7**	**31.2**	**34.7**

Argyll West and Islands NHZ

Starting population = 44 pairs, population cap = 60 pairs. The fledging rates were 0.54 (2003) and 0.46 (mean 1982, 1992, 2003).

			FR										
JS	TR	LE	0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425	0.450	0.475	0.500
0.30	0.040	25.0	36.6	39.4	43.0	**48.3**	**53.8**	**57.8**	**59.6**	**60.0**	**60.1**	**60.1**	**60.0**
	0.045	22.2	33.2	36.1	40.7	**45.2**	**50.0**	**53.6**	**56.6**	**58.3**	**59.5**	**60.1**	**60.1**
	0.050	20.0	30.0	32.4	36.6	40.6	**44.2**	**47.5**	**49.8**	**52.7**	**55.6**	**58.3**	**59.7**
	0.055	18.2	26.2	28.7	31.7	35.2	38.6	41.3	43.9	**46.2**	**49.1**	**52.8**	**56.0**
	0.060	16.7	23.2	25.1	27.8	30.3	33.6	36.6	38.2	41.5	**43.6**	**47.2**	**51.3**
	0.065	15.4	20.6	22.2	24.1	26.7	29.6	32.1	34.4	37.0	39.2	42.7	**46.9**
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425	0.450	0.475	0.500
0.35	0.040	25.0	40.7	**46.1**	**52.7**	**57.6**	**59.7**	**60.1**	**60.1**	**60.0**	**60.1**	**60.0**	**60.0**
	0.045	22.2	37.6	44.0	**49.0**	**52.6**	**56.4**	**58.9**	**59.8**	**60.1**	**60.2**	**60.1**	**60.1**
	0.050	20.0	33.9	38.5	43.2	**46.6**	**50.1**	**53.6**	**56.4**	**59.0**	**59.8**	**60.1**	**60.1**
	0.055	18.2	29.1	33.2	37.5	41.0	43.4	**46.5**	**49.9**	**54.2**	**57.8**	**59.6**	**60.1**
	0.060	16.7	25.9	28.8	32.5	35.6	38.6	41.2	**44.6**	**49.0**	**53.8**	**57.3**	**59.3**
	0.065	15.4	22.3	25.0	28.2	31.4	34.2	36.7	39.6	**44.5**	**48.3**	**52.4**	**56.0**
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425	0.450	0.475	0.500
0.40	0.040	25.0	**47.1**	**54.4**	**58.8**	**60.0**	**60.1**	**60.1**	**60.0**	**60.0**	**59.9**	**59.9**	**59.7**
	0.045	22.2	43.9	**49.7**	**54.7**	**58.0**	**59.7**	**60.1**	**60.1**	**60.1**	**60.1**	**60.1**	**60.1**
	0.050	20.0	39.8	**44.1**	**48.0**	**51.8**	**55.8**	**58.6**	**59.9**	**60.1**	**60.1**	**60.1**	**60.2**
	0.055	18.2	33.9	38.3	41.7	**45.1**	**49.0**	**54.1**	**57.6**	**59.7**	**60.1**	**60.1**	**60.2**
	0.060	16.7	28.8	33.2	36.7	39.9	43.5	**47.8**	**53.6**	**57.9**	**59.4**	**60.0**	**60.1**
	0.065	15.4	25.4	28.9	32.6	35.4	39.2	42.7	**47.8**	**52.8**	**56.9**	**59.0**	**60.0**
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425	0.450	0.475	0.500
0.45	0.040	25.0	**55.3**	**59.3**	**60.0**	**60.1**	**60.1**	**60.0**	**59.9**	**59.8**	**59.8**	**59.8**	**59.8**
	0.045	22.2	**50.5**	**55.2**	**58.5**	**60.0**	**60.2**	**60.1**	**60.1**	**60.0**	**60.1**	**60.1**	**60.0**
	0.050	20.0	**44.3**	**48.5**	**52.7**	**57.0**	**59.5**	**60.0**	**60.0**	**60.1**	**60.2**	**60.1**	**60.1**
	0.055	18.2	38.5	42.3	**46.0**	**50.5**	**55.7**	**59.3**	**60.0**	**60.1**	**60.1**	**60.1**	**60.1**
	0.060	16.7	32.9	37.3	40.9	**44.8**	**50.7**	**55.9**	**59.1**	**60.0**	**60.1**	**60.1**	**60.2**
	0.065	15.4	28.8	32.9	35.9	39.6	**45.1**	**50.9**	**55.5**	**58.6**	**59.9**	**60.1**	**60.1**
			0.250	0.275	0.300	0.325	0.350	0.375	0.400	0.425	0.450	0.475	0.500
0.50	0.040	25.0	**59.2**	**60.0**	**60.1**	**60.1**	**60.0**	**59.9**	**59.9**	**59.8**	**59.8**	**59.7**	**59.6**
	0.045	22.2	**55.0**	**58.5**	**60.0**	**60.1**	**60.0**	**60.1**	**60.1**	**60.0**	**60.0**	**59.9**	**59.8**
	0.050	20.0	**48.2**	**52.9**	**57.4**	**59.8**	**60.1**	**60.1**	**60.1**	**60.1**	**60.1**	**60.0**	**59.9**
	0.055	18.2	41.5	**45.8**	**50.6**	**56.5**	**59.5**	**60.1**	**60.1**	**60.1**	**60.2**	**60.1**	**60.1**
	0.060	16.7	36.6	40.1	**45.0**	**51.5**	**57.2**	**59.5**	**60.1**	**60.1**	**60.1**	**60.0**	**60.1**
	0.065	15.4	31.9	35.7	39.9	**45.7**	**51.6**	**56.6**	**59.3**	**60.0**	**60.1**	**60.0**	**60.1**

Breadalbane and East Argyll NHZ

Starting population = 12 pairs, population cap = 27 pairs. The fledging rates were 0.50 (2003) and 0.50 (mean 1982, 1992, 2003).

			FR								
JS	TR	LE	0.300	0.325	0.350	0.375	0.400	0.425	0.450	0.475	0.500
0.30	0.040	25.0	10.6	10.7	**12.6**	**12.4**	**12.3**	**13.5**	**13.5**	**14.5**	**15.8**
	0.045	22.2	9.1	10.4	10.9	11.2	11.7	11.4	11.8	**12.8**	**13.8**
	0.050	20.0	7.5	8.6	8.7	9.8	9.9	10.4	10.7	11.0	**11.6**
	0.055	18.2	3.1	5.9	7.7	8.1	8.8	9.3	9.2	9.9	10.9
	0.060	16.7	0.8	2.8	6.4	6.7	7.2	8.2	8.7	9.2	10.5
	0.065	15.4	0.0	1.1	3.2	4.9	5.5	6.8	7.9	8.6	10.2
			0.300	0.325	0.350	0.375	0.400	0.425	0.450	0.475	0.500
0.35	0.040	25.0	11.8	**12.4**	**12.5**	**13.2**	**13.6**	**15.0**	**15.9**	**18.1**	**19.3**
	0.045	22.2	10.7	11.4	11.6	11.8	**12.2**	**12.8**	**13.8**	**15.1**	**17.0**
	0.050	20.0	8.7	9.5	10.3	10.4	10.6	11.2	**12.4**	**14.3**	**15.6**
	0.055	18.2	4.1	7.6	8.6	9.4	9.6	10.1	**11.6**	**13.6**	**15.5**
	0.060	16.7	1.7	3.7	7.2	8.2	8.6	9.4	11.0	**13.4**	**15.8**
	0.065	15.4	0.1	2.2	4.9	6.6	7.9	8.8	11.1	**13.0**	**15.7**
			0.300	0.325	0.350	0.375	0.400	0.425	0.450	0.475	0.500
0.40	0.040	25.0	**12.2**	**12.6**	**13.7**	**14.9**	**16.4**	**17.9**	**20.1**	**22.8**	**25.3**
	0.045	22.2	11.0	11.7	**12.3**	**13.0**	**13.7**	**15.5**	**17.8**	**21.2**	**23.7**
	0.050	20.0	8.3	10.3	10.6	11.3	**12.2**	**14.0**	**16.5**	**19.5**	**22.5**
	0.055	18.2	5.0	7.8	9.4	10.0	11.5	**13.9**	**16.3**	**18.9**	**22.1**
	0.060	16.7	1.7	4.7	7.8	9.3	11.5	**13.2**	**15.9**	**19.2**	**21.9**
	0.065	15.4	0.2	1.8	5.8	8.4	10.8	**13.4**	**16.1**	**19.0**	**21.9**
			0.300	0.325	0.350	0.375	0.400	0.425	0.450	0.475	0.500
0.45	0.040	25.0	**12.6**	**14.3**	**15.2**	**17.3**	**19.8**	**22.7**	**25.1**	**26.1**	**26.3**
	0.045	22.2	11.3	**12.3**	**13.3**	**14.9**	**16.5**	**20.8**	**23.3**	**25.8**	**26.3**
	0.050	20.0	9.5	10.2	11.1	**13.0**	**15.5**	**18.7**	**22.3**	**24.6**	**26.1**
	0.055	18.2	6.1	8.6	9.8	**12.3**	**14.7**	**18.1**	**21.8**	**24.0**	**26.1**
	0.060	16.7	2.8	6.4	9.1	**12.1**	**14.7**	**18.5**	**22.1**	**24.5**	**25.8**
	0.065	15.4	0.9	3.5	7.4	11.7	**14.8**	**18.0**	**21.3**	**23.5**	**24.9**
			0.300	0.325	0.350	0.375	0.400	0.425	0.450	0.475	0.500
0.50	0.040	25.0	**13.8**	**15.7**	**18.4**	**21.8**	**24.6**	**26.2**	**26.0**	**26.9**	**26.8**
	0.045	22.2	**12.1**	**13.1**	**15.2**	**18.4**	**21.8**	**25.0**	**26.2**	**26.7**	**26.8**
	0.050	20.0	10.6	11.4	**13.0**	**16.3**	**20.5**	**23.9**	**26.0**	**26.6**	**26.8**
	0.055	18.2	8.3	10.1	**12.8**	**16.5**	**20.0**	**23.8**	**25.7**	**26.5**	**26.9**
	0.060	16.7	6.7	9.6	**12.4**	**15.9**	**19.7**	**23.4**	**25.4**	**26.4**	**26.9**
	0.065	15.4	3.8	8.8	**12.2**	**15.7**	**19.2**	**22.9**	**24.6**	**25.9**	**26.6**

ANNEX 4: EAGLE RANGE CHARACTERISTICS FROM 1992 NATIONAL SURVEY

Several analyses were conducted by Fielding *et al*. (2003a) in relation to characterising the vegetation, topography and land use for active and vacant ranges (=territories) from the 1992 national survey. These were done both nationally and by region (NHZ). Range (=territory) centres were entered in the GIS and from overlapping topography, land cover, grazing animal and Munro layers in the GIS features of ranges within 6 km and 3 km of the centre were extracted (Annex 1) and then tested for differences. As these analyses were largely exploratory, corrections for multiple testing were not undertaken, but a good indication of the strength and likelihood of any differences may be gained by the relevant statistics (typically from single factor ANOVAs). Here we have simply summarised these findings and placed the relevant tabulation at the end of the Annex.

Altitude and slope

Nationally, vacant ranges tended to be higher in mean altitude but with less altitudinal variation and less steep slopes. The conclusions were the same for both 3 and 6 km radius data extractions (Table A4-1). However, these conclusions need to be treated cautiously because they masked important differences between NHZs (Table A4-2). In general, within a NHZ, the vacant ranges tended to be at lower altitude (Table A4-2). The discrepancy between national and regional considerations arises because of differences in altitude limits, and the numbers of active and vacant ranges, between NHZ (notably that the majority of vacant ranges were in the east and here ranges are at higher altitudes than in the west).

These results are probably not surprising given golden eagle biology in that they show that active ranges are more likely in rugged mountainous terrain and that vacant ranges are more likely at lower altitudes, where presumably anthropogenic influences are more likely.

Land cover

Over 70% of the average golden eagle range (3 km buffer) in Scotland was composed of just four vegetation types: undifferentiated heath (29.4%); bog (17.4%); wet heath (13.2%) and montane (10.5%). The rank order was the same if only active ranges were considered, although the proportions change slightly: undifferentiated heath (31.7%); bog (15.9%); wet heath (14.4%) and montane (11.2%). If only vacant ranges were considered the rank order was again the same but the proportions were again changed: undifferentiated heath (24.9%); bog (20.3%); wet heath (10.8%) and montane (9.1%) (Table A4-3).

Nationally, most of the significant differences in the mean extent of vegetation types between vacant and active means were in the expected directions and were not dependent on the radius of data extraction. Thus vacant ranges had more closed canopy and mixed woodland, improved grasslands and bogs, and less cliffs. In other words, vacant ranges were associated with afforestation, lower altitudes closer to agriculture and flatter less rugged ground. However, the most marked national differences between vacant and active ranges concerned heathland land cover types. Vacant ranges had four times as much burnt heather cover and twice as much dry heather (burnt and dry heather are spatially linked), while active ranges had more undifferentiated heath habitat

(Table A4-3). Both burnt and dry heather vegetation types are strongly associated nationally with areas managed for grouse moors (e.g. Watson, 1997; Whitfield *et al*., 2003) and this finding, that the strongest national 'habitat' association with vacant ranges was for vegetation types typified by grouse moor management, echoes several other findings of the framework analyses and provided a further line of evidence that features of grouse moor management formed the most severe constraint on the national population.

In regional analyses, as expected, afforestation was associated with range vacancies in western NHZs (West Argyll, Lochaber, Western Highlands: zones 14, 13 and 8 respectively). There was also evidence that afforestation may have caused range vacancies in the Northern Highlands (zone 7). Interestingly, in the Cairngorms Massif (zone 11) open woodland (primarily native pinewood *Pinus sylvestris*) was associated with active ranges, probably because of a disproportionate occurrence of active ranges away from extensive areas of open moorland managed for grouse (Table A4-4).

There was also evidence, reflecting the greater prevalence of vacant ranges at lower altitudes in the west, of an association between agricultural 'improvement' (enclosed grassland pasture for sheep) and range vacancies in some western zones (Western Seaboard, West Argyll, Lochaber, Western Isles: zones 6, 14, 13 and 3 respectively) (Table A4-4). Whether these relatively few ranges have been abandoned because of agricultural encroachment or that low altitude ranges closer to areas of human influence are in general are of lower 'quality' and so occupied less frequently is unclear, although the latter explanation seems more likely.

Results for associations between range occupation and underlying geology were mixed and inconsistent and therefore more difficult to interpret than those for other types of potential 'land cover' influence (Table A4-5). Results for associations with areas of human activity (e.g. buildings, villages) and heathland and montane vegetation types tended to reflect the other findings for an association between vacancy and lower altitudes in some western regions (Table A4-6), and in two western regions (Western Highlands, Western Isles) vacant ranges were smaller than active ranges (Table A4-6). Again, reiterating other findings for range vacancy occurring at lower altitude areas in the west, when the extent of roads was considered, vacancy was associated with the amount of roads in the Western Seaboard (zone 6) and West Argyll (zone 14). In the Central Highlands (zone 10) there was a greater length of primary roads in vacant territories (Table A4-7).

Grazing

Nationally, probably thanks largely to the influence of the large number of ranges in western Scotland, there was a relationship between range vacancy and low vegetation production, high offtake of vegetation by large grazers and low equity (the balance between production and offtake – low equity meaning there was a relatively high removal of vegetation productivity in relation to its output) at both 3 km and 6 km scales around range centres (Table A4-8). Although high numbers of red deer was the only significant associative variable describing upland grazing animal abundance, at the 6 km buffer, the differences between active and vacant ranges (and numbers of animals) implicated both sheep and deer as the likely cause of the deficit in vegetation productivity available to other grazers.

Regionally, in the Peatlands of Caithness and Sutherland there were more sheep in vacant ranges (Table A4-9). There was evidence that in the Western Highlands, Western Isles and Northern Highlands range vacancy was associated with a high offtake of vegetation production by large grazers (Western Highlands only), lower vegetation production and a low grazing ‘equity’ (Table A4-9). Although in only the Western Isles were these results echoed by significant differences in the numbers of grazing animals, given the densities of animals involved, the major contributory grazing mammals to these patterns were likely to be red deer in the Western and Northern Highlands and sheep in the Western Isles (although here too red deer may have been implicated). As cattle are rarely found in open upland areas, but are more frequently hefted to areas closer to farms, we would interpret the association between higher cattle numbers in vacant ranges (Table A4-9) as yet another indication that in western regions, vacancies tended to be associated with lower altitude areas closer to areas of human activity and agriculture.

Overall, from both national and regional considerations, it might have been expected that if carrion availability was implicated in range occupation in the west there would be a coincidence between active ranges, low equity and high numbers of grazing animals (since higher mortality would be more likely when a high number of animals was associated with a low amount of available forage). In only the Western Isles as part of regional analyses did this coincidence hold and even here Pout (1998) has suggested that at least in parts of the Western Isles carrion availability is probably well in excess of eagle requirements. There is also the possibility that as persecution in Western Isles was primarily a result of perceived impacts on sheep (rather than perceived impacts on grouse, as prevails in the eastern Highlands and southern Scotland), an association between sheep numbers and range vacancy may be a consequence of persecution (especially when this seemed to be most likely in the more readily accessible, less remote ranges). In this respect, persecution in the Western Isles has recently declined and this has been associated with an increase in range occupation (Whitfield *et al*., 2007b).

The results therefore gave only limited support to the notion that range occupancy was related to carrion availability, but gave a stronger indication that the availability of live prey (as influenced by the competitive effects of larger herbivores) may have an influence on occupation of ranges.

Munros (recreation)

There were 69 active and 30 vacant golden eagle range centres within 3 km of a Munro top. The proportion of active ranges within 3 km of a Munro top (0.70) was not significantly different from the overall proportion of active and vacant ranges within 3 km of a Munro top (0.68). As expected the number of ranges in each Munro district (Table A4-10) was positively correlated with the district area ($r = 0.703$, $p<0.001$) and the Munro density ($r = 0.742$, $p<0.001$). Only the number of active ranges was correlated with the number of Munros ($r = 0.636$, $p < 0.005$) and Munro density ($r = 0.535$, $p=0.001$), presumably reflecting the association between mountain terrain and eagle habitat. The number of vacant ranges was only significantly correlated with the district area ($r = 0.382$, $p = 0.02$). There was no evidence for a general relationship between the number of Munros and the number of vacant ranges ($r = -0.01$, $p >0.90$).

The proportions of active and vacant ranges that were nearest to a Munro (within a 6 km threshold) did not differ significantly from the overall proportions and there was no evidence of a difference in mean distances to, or number of, Munros within 6 km of each range centre (Table A4-11). However, when the analysis was restricted to the range closest to a Munro there was some evidence that vacant ranges were significantly closer, particularly when the distance threshold was reduced from 6 km to 4 km (Table A4-12).

Hence, at a national scale the evidence was equivocal for an effect of disturbance arising from the recreational use of Munro mountains. This is unsurprising given the simplicity of the analyses and that any effect of recreation is likely to be highly location specific and therefore difficult to analyse and detect at larger spatial scales. More critical analyses therefore need additional data about the intensity of Munro usage, access routes and eagle nest site use. The results do tend to indicate, however, that any effect of recreation, at least in relation to the most popular sources of hillwalking, probably does not constitute a serious national issue.

In regional analyses, only in the Northern Highlands (zone 7) was there consistent and statistically significant evidence that range vacancy was associated with the presence of Munros (Table A4-13).

Persecution

Influences of persecution have been dealt with in greater detail by Whitfield *et al*. (2003, 2004a, b) and are summarised in the main text of this report. Here, we include the analysis of Fielding *et al*. (2003a) which examined the number of poisoning incidents in relation to expectations based on the extent of each NHZ. In order to produce an equitable comparison the number of poisoning incidents must be related to the NHZ area. This is possible if the observed number of incidents is related to an expected number derived from the area of a NHZ. Comparisons must then be standardized by taking account of the magnitude of the expected number; this is possible if standardized residuals are calculated.

All of the NHZ with positive residuals (more observed incidents than expected) were in the east (Table A4-14). Only two NHZ, Breadalbane & East Argyll and Central Highlands, had significantly many more incidents than expected from their areas (although an additional two, Northeast Glens and Cairngorms Massif also had more incidents than expected) while nine NHZ had significantly less incidents than expected. These results are in line with other analyses which indicated the areas from which the most persistent threats from persecution originate.

Cliff nest site availability

Vacant ranges had fewer nest sites but they did not differ in their altitude characteristics (Table A4-15). However, the results should be treated with caution because no data were available for over 40% of the ranges. The relationship between mean nest altitude and mean range altitude appeared to be relatively constant irrespective of the mean range altitude: most nests were at an altitude which was approximately 14% below the mean range altitude (Fig. A4-1).

Given the precision of recorded nest locations (±100 m) it was impossible to estimate nest aspect from slope models derived from a Digital Elevation Model. Instead data on the altitudes and aspects of a number of recorded nests were available for a restricted number of non-coastal active ranges (n = 285 nests) from the 1992 national survey.

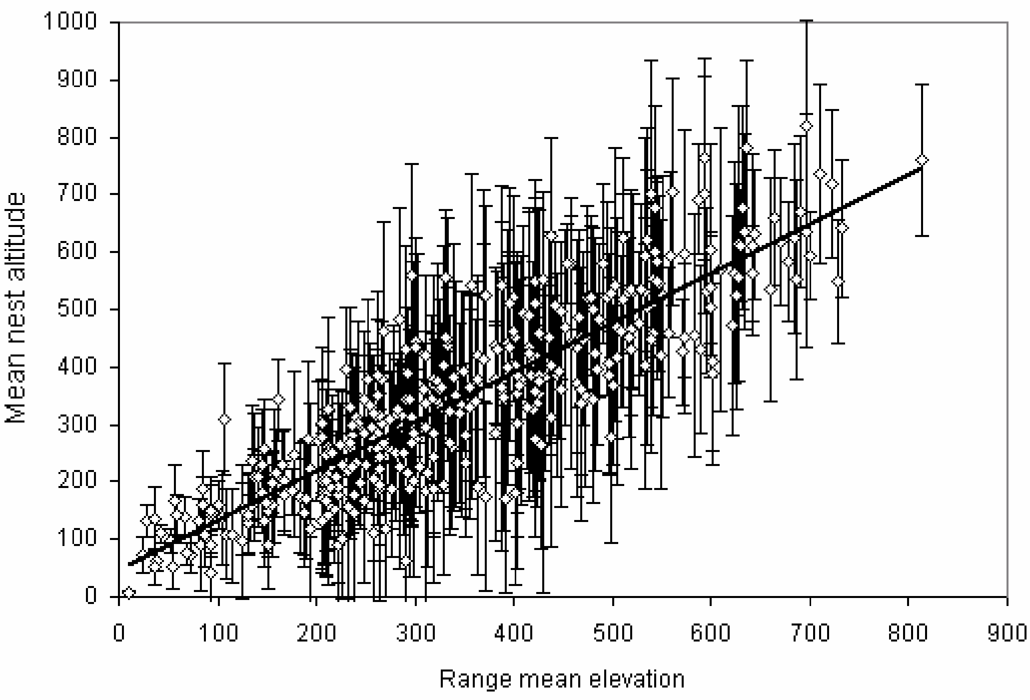
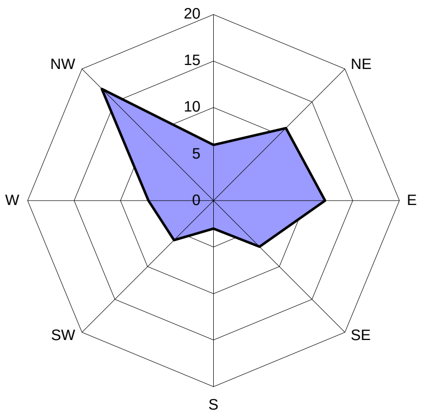


Fig. A4-1. Mean nest site elevation ± sd (m) against mean range elevation (3 km range buffer) around territory centre (m). Best fit line is shown: mean nest altitude = 48.223 + 0.8611.mean range altitude, R^2 = 0.756.

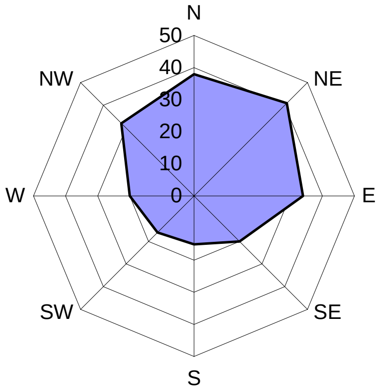
Most nest aspects were between NW and NE (Fig. A4-2). Despite the apparent differences in aspect profiles there was no evidence that nests in ranges with single nests differed in aspect from nests in ranges with multiple nests (Table A4-16). This suggested that eagles with limited nest site opportunities did not face different constraints.

The number of known and active ranges within a NHZ was strongly positively correlated with the abundance of the LCS88 Rocks and Cliffs habitat class (Annex 5: *r* = 0.66 for all ranges and *r* = 0.70 for active ranges). The data also suggested that vacant ranges had less cliff habitat than active ranges (Table A4-3). This finding was supported by analysis of 10 m pixel slope data: within 3 km of range centres active ranges had 1546 pixels with >= 50º of slope compared to 810 pixels for vacant ranges (t = 5.27, p <0.0001). Within 6 km of range centres the equivalent values were 4480 and 2573 pixels (t = 5.40, p <0.0001). The mean slope was greater for active ranges in all NHZ (Table A4-1).

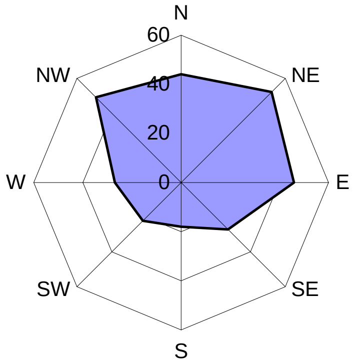
The topographic differences between vacant and active ranges, however, do not necessarily indicate that cliff nests sites were limiting, and more likely reflect the general finding that vacant ranges (especially in the west) were more likely to be 'peripheral' in relation to the typical eagle range, being at lower altitudes, in less rugged terrain and closer to areas of human activity. A better way of examining the issue of the potential limiting influence of cliff nest sites was to attempt to predict the availability of potential nest sites.



Ranges with one nest



Ranges with two or more nests



All ranges

Fig. A4-2. Aspect of nest sites in ranges with one nest site and more than one nest site (see Table A4-16 for sample sizes).

As a preliminary analysis with this aim we compared the nest count per range (from the 1992 survey only) to the number of 10 m pixels with a slope >=50%, within 3 km of a range centre. The hypothesis was that, if it is possible to predict potential nest sites from a slope map, there should be a relationship between the number of nests and the slope characteristics of ranges. Unfortunately, no such relationships were found in any NHZ, and the correlation between the mean number of 10 m pixels with >= 50º of slope and the mean number of nests per range, in each NHZ, was –0.009 (p>0.95). If the analysis was restricted to active or vacant ranges only, no significant relationships were again apparent. There were also no significant correlations between the areas of 12 solid geology classes and either the areas of the LCS88 Rocks and Cliffs habitat class within the NHZ or the mean number of nests per range. This is possibly unsurprising since the 'quality' of cliffs used appears to differ between NHZ and may be associated with multiple factors including geology, range density and absence of mammalian ground predators. For example, some golden eagles on Lewis are effectively ground nesting. Therefore, although the abundance of cliffs is a valuable resource, it seemed unlikely that a satisfactory model, based solely on the distribution of cliffs, could be generated to predict nest site availability.

Table A4-1. National altitude and slope characteristics of active and vacant ranges. Because these were exploratory, descriptive analyses, no multiple testing adjustments have been made to the p values from a comparison of the vacant and active means. The column 'Larger' indicates which range class had the largest mean value.

		vacant		active		All				
Buffer	Variable	mean	*sd*	mean	*sd*	mean	*sd*	F	Sig.	Larger
3 km	Area	2604.2	523.5	2570.4	514.9	2580.9	517.4	0.6	0.449	
	Altitude (n = 195, 430, 625)									
	Mean	356.9	197.4	324.0	170.6	334.3	179.9	4.5	0.034	vacant
	Minimum	151.6	149.9	83.8	117.5	105.0	132.2	37.3	0.000	vacant
	Maximum	641.8	284.5	655.9	268.2	651.5	273.2	0.4	0.549	
	Standard deviation	108.3	52.9	127.2	53.8	121.3	54.2	16.8	0.000	active
	Slope (n = 195, 430, 625)									
	Average	11.5	4.2	13.9	4.9	13.1	4.8	34.7	0.000	active
	Maximum	61.1	10.5	66.4	9.1	64.8	9.9	42.5	0.000	active
	Standard deviation	8.3	2.3	9.6	2.2	9.2	2.3	44.8	0.000	active
6 km	Area	10062.0	2351.4	9698.7	2444.3	9812.1	2419.7	3.0	0.082	
	Altitude (n = 195, 430, 625)									
	Mean	339.3	188.0	298.9	165.3	311.5	173.6	7.4	0.007	vacant
	Minimum	110.0	128.9	51.4	93.0	69.7	108.9	41.5	0.000	vacant
	Maximum	733.7	283.4	740.3	278.0	738.3	279.5	0.1	0.785	
	Standard deviation	126.7	51.5	140.0	53.2	135.9	53.0	8.5	0.004	active
	Slope (n = 195, 430, 625)									
	Average	10.9	3.9	12.8	4.5	12.2	4.4	25.7	0.000	active
	Maximum	66.7	9.8	70.3	8.2	69.2	8.9	23.5	0.000	active
	Standard deviation	8.3	2.1	9.4	2.1	9.1	2.2	36.2	0.000	active

Table A4-2. Topographic character of habitat within 3 km of active and vacant ranges in each NHZ. If mean values between vacant and active ranges differ significantly (p < 0.05) the p value is bold.

			Altitude			Slope		
	Status	n	Maximum	Mean	sd	Maximum	Mean	sd
Western Seaboard	vacant	20	336.0	141.3	75.8	64.5	9.8	8.5
	active	73	201.0	105.2	69.0	12.7	9.8	8.7
	p		**0.005**	**0.002**	**0.009**	0.070	**0.005**	**0.035**
Caithness & Sutherland	vacant	17	423.4	226.4	65.1	50.8	6.7	5.7
	active	14	590.1	265.5	109.7	66.1	9.4	8.5
	p		**0.023**	0.228	**0.008**	**0.001**	**0.021**	**0.004**
Breadalbane	vacant	4	739.8	455.8	119.2	57.3	11.3	7.4
	active	20	856.8	497.8	149.9	66.9	14.9	9.2
	p		0.232	0.517	0.168	**0.013**	0.127	**0.038**
West Argyll	vacant	11	475.0	229.0	100.1	64.1	12.0	8.0
	active	43	469.5	231.7	93.3	64.1	12.1	9.0
	p		0.937	0.943	0.622	0.985	0.927	0.083
Lochaber	vacant	9	810.0	410.6	145.2	62.7	13.3	9.1
	active	27	961.7	477.5	192.2	68.6	20.0	11.2
	p		**0.043**	0.192	**0.003**	**0.022**	**0.000**	**0.001**
NW Seaboard	vacant	19	675.5	310.9	131.2	69.9	13.2	10.2
	active	42	736.5	335.6	146.5	72.2	14.3	11.0
	p		0.322	0.411	0.244	0.258	0.282	0.168
NE Glens	vacant	11	656.1	397.5	90.5	53.6	10.1	6.7
	active	8	670.9	425.7	96.8	57.0	10.7	7.2
	p		0.746	0.334	0.49	0.564	0.498	0.449
Cairngorm	vacant	33	865.5	584.8	107.2	58.3	11.8	7.8
	active	28	870	591.2	119.1	63.3	12.7	9.0
	p		0.919	0.852	0.159	0.058	0.213	**0.015**
Central Highlands	vacant	10	778.9	539.5	106.7	57.4	10.5	8.0
	active	11	840.2	563.1	125.7	60.3	11.7	8.3
	p		0.336	0.663	0.162	0.375	0.348	0.649
Western Highlands	vacant	6	696.0	293.9	152.3	66.0	17.4	10.7
	active	55	758.3	322.6	171.2	68.1	18.5	10.9
	p		0.429	0.449	0.283	0.305	0.538	0.643
Northern Highlands	vacant	36	822.7	449.4	152.8	62.6	13.5	9.1
	active	47	856.1	457.2	158.5	65.8	15.2	9.9
	p		0.411	0.750	0.560	**0.033**	**0.039**	0.073
Western Isles	vacant	16	246.3	74.6	47	63.4	9.1	8.5
	active	58	371.8	126.5	75.6	63.6	10.3	8.3
	p		**0.028**	**0.019**	**0.026**	0.960	0.383	0.759

Table A4-3. National mean and standard deviations of the areas (km^2) of four major habitat categories. Sample sizes (vacant, active and all) are given after the category name. Significance tests follow the guidelines in Table A4-1.

		vacant		active		All				
Buffer	Variable	mean	*sd*	mean	*sd*	mean	*sd*	F	Sig.	Larger
3 km	**Woodland (n = 195, 430, 625)**									
	Closed canopy	144.7	250.4	78.3	195.5	80.7	200.05	11.7	0.001	vacant
	Broadleaved	23.8	43.5	24.3	46.0	25.4	47.01	0.1	0.702	
	Mixed	16.4	48.2	9.4	31.5	10.0	32.44	3.9	0.049	vacant
	Open canopy	95.5	210.3	81.8	221.9	85.8	228.22	0.3	0.614	
	Scrub	0.7	4.7	0.2	2.5	0.2	2.62	2.9	0.088	
	Habitat (n = 195, 430, 625)									
	Arable	2.6	18.5	1.9	26.93	2.0	24.0	0.1	0.760	
	Bog	515.9	537.8	411.7	492.46	446.0	512.6	5.7	0.018	vacant
	Bracken	7.8	27.4	12.6	59.33	10.6	50.4	1.2	0.273	
	Cliffs	24.3	79.3	53.4	172.48	46.0	151.1	5.1	0.024	active
	Human	2.1	9.5	1.2	9.50	1.5	9.3	1.2	0.272	
	Montane	230.9	341.3	275.9	344.62	270.6	352.2	2.3	0.130	
	Other	9.7	41.6	21.1	117.42	17.7	99.2	1.7	0.187	
	Wetland	59.9	115.3	62.0	113.43	59.6	111.8	0.0	0.827	
	Heathland (n = 195, 430, 625)									
	Burnt	207.0	425.6	59.2	229.45	102.5	306.1	31.6	0.000	vacant
	Dry	119.1	239.1	68.3	192.44	82.6	206.1	8.0	0.005	vacant
	Undifferentiated	660.5	577.7	809.2	674.46	765.8	646.6	7.1	0.008	active
	Wet	271.2	579.5	371.8	601.80	342.2	591.6	3.8	0.051	
	Grass (n = 195, 430, 625)									
	Coarse	62.1	154.6	87.3	194.60	80.5	189.0	2.5	0.112	
	Improved	57.2	104.9	33.4	78.23	39.9	87.9	9.9	0.002	vacant
	Smooth	92.9	146.2	99.1	145.68	95.0	143.9	0.2	0.620	
6 km	**Woodland (n = 195, 430, 625)**									
	Closed canopy	591.6	813.4	422.4	719.37	472.0	750.1	7.1	0.008	vacant
	Broadleaved	121.5	167.1	135.3	176.19	127.6	171.0	0.4	0.550	
	Mixed	67.2	127.0	56.3	116.88	58.2	117.8	1.6	0.203	
	Open canopy	379.6	571.6	355.6	610.95	359.7	591.5	0.3	0.575	
	Scrub	591.6	813.4	1.1	5.62	472.0	750.1	5.2	0.008	vacant
	Habitat (n = 195, 430, 625)									
	Arable	40.6	175.66	1.0	5.4	25.0	170.79	2.3	0.126	
	Bog	2071.3	1878.47	1640.6	1747.13	1775.0	1798.75	7.8	0.005	vacant
	Bracken	37.2	113.35	46.0	165.37	43.3	151.04	0.5	0.498	
	Cliffs	86.3	235.64	157.6	454.74	135.4	400.65	4.3	0.039	active
	Human	15.1	38.91	9.8	29.89	11.5	33.03	3.5	0.062	
	Montane	805.7	1064.67	900.1	1057.63	870.6	1059.88	1.1	0.303	
	Other	62.4	256.81	71.8	303.02	68.9	289.22	0.1	0.707	
	Wetland	259.1	338.81	293.3	386.71	282.6	372.50	1.1	0.288	

Buffer	Variable	vacant mean	 *sd*	active mean	 *sd*	All mean	 *sd*	F	Sig.	Larger
	Heathland (n = 195, 430, 625)									
	Burnt	746.6	1433.2	254.4	827.68	397.5	1059.0	29.2	0.000	vacant
	Dry	426.2	779.1	249.3	676.01	297.9	701.3	8.3	0.004	vacant
	Undifferentiated	2412.1	1958.5	2791.5	2290.18	2684.4	2187.2	4.0	0.045	active
	Wet	970.5	1897.7	1338.5	2076.20	1242.5	2027.8	4.4	0.035	active
	Grass (n = 195, 430, 625)									
	Coarse	243.3	526.9	327.3	629.40	304.4	615.1	2.6	0.105	
	Improved	305.6	373.2	212.9	303.63	237.2	327.0	10.8	0.001	vacant
	Smooth	417.0	453.7	416.6	464.06	413.4	464.7	0.0	0.991	

Table A4-4. Areas (km^2) of forest and grassland in each range category for each NHZ. If a habitat was absent from a NHZ the entry is blank. Sample sizes are given in Table A4-2.

		Forest		Grassland		
	Status	Mixed	Closed	Open	Smooth	Improved
Western Seaboard	vacant	9.6	72.6	35.0	254.4	105.9
	active	8.7	82.0	150.9	225.0	44.3
	p	0.865	0.873	0.061	0.578	**0.004**
Caithness & Sutherland	vacant		70.3	163.5	43.7	14.6
	active		42.8	7.6	21.7	5.9
	p		0.627	0.06	0.252	0.328
Breadalbane	vacant	0.9	54.9	110.6	64.1	15.4
	active	13.4	45.8	142.3	182.9	21.6
	p	0.484	0.814	0.886	0.159	0.722
West Argyll	vacant	32.2	438	464.2	152.2	115.7
	active	4.0	141.5	284.5	136.7	41.0
	p	**0.000**	**0.001**	0.244	0.814	**0.025**
Lochaber	vacant	18.0	214.3	159.3	89.5	63.4
	active	3.9	96.8	59.4	58.6	6.4
	p	**0.013**	0.159	0.120	0.479	**0.016**
NW Seaboard	vacant	10.0	17.5	38.3	16.2	19.6
	active	5.1	12.2	2.1	10.5	6.4
	p	0.477	0.615	0.082	0.588	0.105
NE Glens	vacant	41.8	388.7	169.3	214.2	204.7
	active	65.1	382.6	85.4	135.9	247.3
	p	0.509	0.971	0.305	0.340	0.703
Cairngorm	vacant	10.6	102	7.2	84.3	39.7
	active	8.9	153.6	42	118.8	55.5
	p	0.824	0.452	**0.025**	0.390	0.501
Central Highlands	vacant	3.6	71	95.8	81.8	64.5
	active	3.4	164.7	37.0	90.5	25.7
	p	0.969	0.366	0.225	0.876	0.291
Western Highlands	vacant	79.3	267.8	205	74.2	38.1
	active	16.2	55.9	82.5	118.2	16.9
	p	**0.004**	**0.000**	0.082	0.375	0.129
Northern Highlands	vacant	27.2	186.5	71.9	39.7	19.6
	active	25.0	70.4	29.6	32.3	13.9
	p	0.883	**0.014**	0.065	0.617	0.544
Western Isles	vacant		0	0	52.1	70.6
	active		0.1	1.0	13.7	48.4
	p		0.603	0.350	**0.006**	0.430

Table A4-5. Mean areas (km^2) of the principal solid geological class underlying each range class in each NHZ. If a class was absent from a NHZ the entry is blank. Sample sizes are given in Table A4-2.

		Solid Geology				
	Status	Argillaceous	Intermediate Igneous	Metamorphic Sedimentary	Miscellaneous Sedimentary	Sandstones
Western Seaboard	vacant	123.1	12.3	163	0	181.5
	active	49.9	12.8	135.7	0.7	182.4
	p	0.147	0.956	0.824	0.550	0.993
Caithness & Sutherland	vacant	89.2	84.9	976.2	290.7	325.9
	active	18.5	77.3	1666.8	80.9	300.2
	p	0.425	0.941	**0.048**	0.147	0.888
Breadalbane	vacant		4.3	1951.1	250.2	452.1
	active		13.5	2648.2	24.9	10.1
	p		0.466	**0.026**	0.065	**0.025**
West Argyll	vacant		9.8	1793.3		164.1
	active		22.6	1622.6		23.8
	p		0.450	0.563		0.121
Lochaber	vacant		43.3	1728.4	5.7	
	active		127.5	1561.1	0.0	
	p		0.207	0.697	0.077	
NW Seaboard	vacant	55.4	78.6	548.8	30.7	1094.1
	active	51.0	23.6	516.7	15.3	1254.3
	p	0.931	0.275	0.909	0.177	0.581
NE Glens	vacant		10.2	2031.8		4.4
	active		17.0	1575.4		0.0
	p		0.554	0.356		0.409
Cairngorm	vacant		48.7	1886.8		
	active		90.1	1904.7		
	p		0.338	0.943		
Central Highlands	vacant		10.2	2260.7	2.1	
	active		2.5	2285.3	0.0	
	p		0.085	0.940	0.306	
Western Highlands	vacant	93.2	41.7	1486.2	0.0	169.0
	active	5.4	36.3	1987.9	4.2	111.8
	p	**0.008**	0.893	0.213	0.744	0.751
Northern Highlands	vacant	33.5	22.7	2441.4	27.0	99.9
	active	3.7	11.1	2517.0	6.9	10.6
	p	0.188	0.260	0.511	0.161	0.141
Western Isles	vacant			46.9	0.0	
	active			74.9	0.3	
	p			0.707	0.603	

Table A4-6. Means areas (km^2) of human-related (e.g. habitation), montane and two heathland habitats within each range class. Area is the mean land area within the 3 km radius buffer (maximum possible is 2827.8 ha). If a habitat was absent from a NHZ the entry is blank. Sample sizes are given in Table A4-2.

		Habitat		Heathland		Area
	Status	Human	Montane	Burn	Heath (undiff)	
Western Seaboard	vacant	0.1	38.1	0	436.8	2025.6
	active	0.4	116.6	4.2	539.3	2159.2
	p	0.555	0.106	0.479	0.503	0.437
Caithness & Sutherland	vacant		16.5	81.5	990.8	2781.1
	active		22.6	69.2	603.8	2772
	p		0.654	0.872	0.079	0.901
Breadalbane	vacant	0	309.6	484.6	531.5	2827.8
	active	35.1	463.7	268.0	527.3	57.8
	p	0.665	0.125	0.070	0.727	0.665
West Argyll	vacant	2.3	53.5		683.7	2479.7
	active	0.6	99.6		1087.8	2304.2
	p	0.099	0.419		**0.025**	0.461
Lochaber	vacant	9.3	296.8		1200.7	2746.6
	active	0.3	509.2		1268.8	2809.0
	p	**0.018**	0.085		0.720	0.190
NW Seaboard	vacant	1.5	294	5.6	432.7	2709.6
	active	0.2	329.9	0.7	607.9	2746.1
	p	0.187	0.736	0.199	0.135	0.644
NE Glens	vacant	6	69.7	943.6	128.9	2827.8
	active	11.9	36.3	838.6	200.1	2827.8
	p	0.604	0.584	0.708	0.22	
Cairngorm	vacant	2.8	365.1	658	371.2	2827.8
	active	2.3	349.4	392.2	384.3	2827.8
	p	0.886	0.878	0.075	0.855	
Central Highlands	vacant	4.8	249	164.1	785.5	2827.8
	active	0.0	464.8	120.4	886.5	2827.8
	p	0.306	0.168	0.567	0.516	
Western Highlands	vacant	1.0	234.7		1183.8	2379.8
	active	0.9	277.3		1567.7	2693.9
	p	0.961	0.676		0.101	**0.013**
Northern Highlands	vacant	0.6	448.9	60.7	1180.2	2823.4
	active	0.0	500.6	53.6	1385.5	2827.3
	p	**0.049**	0.593	0.732	0.057	0.315
Western Isles	vacant	2.3	34.0		49.0	1759.2
	active	1.5	169.7		99.5	2380.8
	p	0.615	**0.030**		0.135	**0.002**

Table A4-7. Mean length (m) of each road class per NHZ and range class (unknown = range not checked, single = single eagle present). Unrepresented classes are excluded. If the mean lengths differed significantly (p<0.05) between range occupancy classes within a NHZ the results are in bold.

		vacant		active		unknown		single		all			
ALL		mean	sd	mean	sd	mean	sd	mean	sd	mean	sd	F	Sig.
	n	195		430		18		30		673			
A class	Narrow	284.5	1128.96	306.3	1243.5	29.7	125.87	133.5	731.2	284.8	1175.4	0.50	0.686
	Single	612.6	1728.02	439.0	1466.1	316.6	934.04	409.3	1527.9	484.7	1537.6	0.67	0.570
B class	Narrow	369.9	1294.61	324.2	1277.7	439.7	1358.85			326.1	1256.3	0.80	0.493
	Single	132.3	817.99	98.5	757.7			169.4	927.7	108.8	773.0	0.27	0.851
	Minor	1844.0	2977.68	1412.0	2420.7	1658.8	3772.41	974.4	2061.5	1524.2	2625.9	1.69	0.169
Primary	Dual	1.5	21.48							0.4	11.6	0.82	0.485
	Single	**407.3**	**1379.59**	**163.2**	**832.7**	**545.1**	**1615.09**	43.8	240.1	**238.8**	**1037.7**	**3.39**	**0.018**
	Total Road	**3652.2**	**4498.74**	**2743.1**	**3664.6**	**2989.8**	**4605.73**	1730.3	3424.6	**2968.0**	**3961.7**	**3.41**	**0.017**
						Western Isles							
	n	16		58		5		3		82			
A class	Narrow	300.9	1203.75	540.1	1792.6	106.8	238.81			447.3	1599.17	0.26	0.857
	Single	75.4	301.50	980.2	2247.2	480.2	1073.76			737.3	1944.31	1.10	0.356
B class	Narrow	1056.2	2319.50	338.2	1084.6	1058.0	2365.76			509.8	1486.19	1.34	0.268
	Single	8.8	35.00	174.5	825.4					125.1	696.90	0.32	0.808
	Minor	3722.2	5415.89	1830.6	3232.1	4134.0	6661.82			2273.2	3991.83	1.67	0.181
	Total Road	5163.4	6359.47	3863.7	5122.7	5779.0	7480.51			4092.7	5457.04	0.96	0.416
						North West Seaboard							
	n	19		42				7		68			
A class	Narrow	881.3	2060.34	411.2	1151.1					500.2	1422.55	1.20	0.307
	Single	911.4	1936.11	679.5	1799.2					674.4	1746.85	0.69	0.505
B class	Narrow			174.7	746.0					107.9	589.77	0.70	0.501
	Minor	1572.8	3191.04	710.6	1611.2			1568.7	2412.3	1039.9	2241.20	1.19	0.310
	Total Road	3365.5	4429.53	1976.0	2868.9			1568.7	2412.3	2322.3	3357.41	1.33	0.272
						The Peatlands of Caithness and Sutherland							
	n	17		14						31			
A class	Narrow	571.8	1219.38	142.6	533.45					377.9	981.56	1.49	0.232
	Single	645.5	1823.70	350.4	1310.92					512.2	1593.99	0.26	0.616
B class	Narrow	584.1	1775.64							320.3	1329.98	1.51	0.230
	Minor	1678.9	3047.79	1258.9	2368.49					1489.2	2725.84	0.18	0.677
	Total Road	3480.3	3884.90	1751.8	2621.13					2699.7	3433.81	2.01	0.167
						Western Seaboard							
	n	20		73		2		1		96			
A class	Narrow	742.0	1824.59	629.3	1744.2					633.1	1728.21	0.16	0.925
	Single	**542.4**	**1438.30**	**371.3**	**1324.8**	**1648.5**	**2331.33**	**7948.0**		**512.5**	**1557.47**	**10.64**	**0.000**
B class	Narrow	1010.9	2314.49	1065.8	2348.9	1312.0	1855.45			1048.4	2302.77	0.08	0.971
	Single			0.2	1.9					0.2	1.63	0.10	0.959
	Minor	**3023.5**	**3125.10**	**1593.4**	**2405.5**	**2980.0**	**552.96**	**7845.0**		**1985.4**	**2658.77**	**3.52**	**0.018**
Primary	Single	491.0	1867.98	13.1	112.0					112.3	863.44	1.65	0.182
	Total Road	**5809.7**	**5385.53**	**3673.2**	**4101.8**	**5940.5**	**77.07**	**15793.0**		**4291.8**	**4558.65**	**3.68**	**0.015**

		vacant		active		unknown		single		all			
ALL		mean	sd	mean	sd	mean	sd	mean	sd	mean	sd	F	Sig.
						Northern Highlands							
	n	36		47				9		92			
A class	Narrow	106.9	641.50	154.3	743.7			445.0	1335.0	164.2	776.97	0.68	0.507
	Single	217.8	910.98	79.5	444.5			207.0	621.0	146.1	676.52	0.46	0.632
B class	Narrow	92.4	554.50							36.2	346.86	0.77	0.464
	Single	246.8	1209.47							96.6	759.80	1.16	0.318
Primary	Minor	992.0	1790.10	1534.6	2515.9					1172.1	2156.08	2.17	0.120
	Single	705.8	1709.92	240.3	1032.3					398.9	1314.97	1.77	0.177
	Total Road	2361.8	3212.56	2008.7	2620.1			652.0	1956.0	2014.1	2829.70	1.32	0.271
						Western Highlands							
	n	6		55		3		2		66			
A class	Narrow			474.5	1661.9					395.4	1525.16	0.29	0.835
	Single			342.5	1213.0			1233.5	1744.4	322.8	1144.30	0.66	0.582
B class	Narrow	395.3	968.36	151.5	976.2					162.2	933.24	0.17	0.916
Primary	Minor	3595.5	2319.73	1483.6	2568.9	1076.0	1863.69	2257.0	3191.9	1680.5	2560.93	1.34	0.271
	Single	705.5	1317.59	591.8	1706.0					557.3	1605.71	0.22	0.883
	Total Road	4696.3	1353.52	3043.9	3722.2	1076.0	1863.69	3490.5	4936.3	3118.2	3544.62	0.73	0.535
						Central Highlands							
	n	10		11		2		3		26			
A class	Single	441.8	1397.09							169.9	866.44	0.50	0.685
B class	Narrow	366.2	1158.03							140.8	718.18	0.50	0.685
	Single	237.6	610.14					1693.7	2933.5	286.8	1050.58	2.51	0.085
Primary	Minor	1797.4	2110.25	1412.5	2431.1			1113.7	1928.9	1417.4	2120.03	0.39	0.758
	Single	**523.2**	**1654.50**	**393.5**	**889.8**	**4906.0**	**1247.34**			**745.1**	**1700.04**	**8.20**	**0.001**
	Total Road	3366.2	4874.43	1806.1	2340.8	4906.0	1247.34	2807.3	2582.2	2760.1	3501.00	0.59	0.627
						Cairngorm Massif							
	n	33		28		6				67			
A class	Narrow			50.4	266.65					21.1	172.38	0.69	0.505
	Single	875.0	2235.06	193.6	688.41					511.9	1657.90	1.62	0.205
B class	Narrow			157.9	835.30					66.0	539.99	0.69	0.505
	Single	207.8	1194.00	199.3	959.95					185.7	1035.21	0.10	0.902
Primary	Minor	817.9	1855.05	1294.1	2221.53					943.7	1956.62	1.22	0.301
	Dual	9.1	52.22							4.5	36.65	0.51	0.604
	Single	147.6	713.34							72.7	502.24	0.72	0.492
	Total Road	2057.5	3625.73	1895.3	2764.41					1805.4	3135.53	1.12	0.334
						North East Glens							
	n	11		8						19			
A class	Single	1376.6	3132.44	1725.0	3092.96					1523.3	3033.60	0.06	0.813
B class	Single			921.1	1709.16					387.8	1163.77	3.27	0.088
Primary	Minor	3265.6	2914.23	3240.3	3370.96					3254.9	3022.82	0.00	0.986
	Single	593.0	1966.76							343.3	1496.48	0.72	0.409
	Total Road	5235.3	4791.66	5886.4	4632.40					5509.4	4605.42	0.09	0.770
						Lochaber							
	n	9		27				1		37			
B class	Narrow	60.7	182.00	207.9	1100.07					172.1	958.68	0.10	0.906
	Single	467.7	1403.00	371.1	1963.90					394.6	1825.34	0.03	0.971
Primary	Minor	1758.1	3882.61	948.6	2028.75					1145.5	2561.40	0.40	0.675
	Single	1660.3	2661.31	445.1	1061.38					740.7	1642.61	2.01	0.150
	Total Road	3946.8	6710.71	1972.8	3008.19					2452.9	4187.07	0.87	0.430

		vacant		active		unknown		single		all			
ALL		mean	sd	mean	sd	mean	sd	mean	sd	mean	sd	F	Sig.
						Argyll West and Islands							
	n	11		43				1		55			
A class	Narrow	500.1	1658.61	9.3	60.7					107.3	742.70	1.99	0.146
	Single	1321.7	2114.57	583.1	1648.4					720.2	1743.48	0.87	0.425
B class	Narrow	1186.6	1562.39	373.9	1148.7					529.7	1261.24	1.98	0.149
	Single	303.4	1006.14	183.1	1048.6					203.8	1022.65	0.08	0.925
Primary	Minor	1763.8	2260.09	1081.2	1856.0			2550.0		1244.4	1932.07	0.77	0.467
	Single	698.3	1557.62	0.0	0.0					139.7	727.15	4.60	0.014
	Total Road	**5773.9**	**3973.35**	**2230.6**	**2841.6**			**2945.0**	**3353.15**			**5.76**	**0.005**
						Breadalbane and East Argyll							
	n	4		20				3		27			
A class	Single	493.0	986.00	224.7	1004.7					239.4	930.84	0.24	0.792
	Single			50.5	225.8					37.4	194.37	0.16	0.850
Primary	Minor	621.0	1242.00	1471.7	1880.8					1182.1	1742.79	1.19	0.321
	Single			429.3	1269.0			438.3	759.2	366.7	1116.01	0.24	0.789
	Total Road	1114.0	1303.21	2176.1	3285.6			438.3	759.2	1825.7	2919.53	0.58	0.566

Table A4-8. Numbers of large grazing animals per golden eagle range. Production, offtake and equity measures relate the amount of dry matter (kg) per range using the information provided by Armstrong et al. (unpublished) (after Armstrong et al., 1997a, b). Significance tests follow the guidelines in Table A4-1.

		Vacant		Active		All				
Buffer	Variable	mean	*sd*	mean	*sd*	mean	*sd*	F	Sig.	Larger
3 km	Calves	29.1	21.1	28.0	23.47	28.3	22.5	0.3	0.578	
	Cattle	122.2	155.8	95.3	140.84	101.9	144.2	4.6	0.033	vacant
	Hinds	94.0	72.0	88.6	72.02	90.1	71.1	0.7	0.388	
	Stags	58.1	60.7	50.2	44.57	52.5	49.4	3.3	0.069	
	Sheep	1533.7	1105.5	1483.2	1036.63	1489.5	1054.5	0.3	0.581	
	Production	3488441.3	1109845.9	3803779.2	1067540.85	3696534.4	1083386.7	11.4	0.001	active
	Offtake	221829.9	209217.2	180513.8	184009.39	190777.9	190563.0	6.2	0.013	vacant
	Equity	3266611.4	1068804.8	3623265.5	1045334.75	3505756.5	1058763.4	15.4	0.000	active
6 km	Calves	115.0	81.2	106.7	88.18	109.2	85.3	1.2	0.267	
	Cattle	476.0	574.1	347.4	499.27	381.1	520.0	8.1	0.005	vacant
	Hinds	371.9	276.9	337.4	270.44	347.3	269.6	2.2	0.143	
	Stags	226.6	225.0	193.3	170.93	203.1	187.1	4.2	0.042	vacant
	Sheep	5917.5	4077.3	5525.0	3701.05	5619.7	3823.7	1.4	0.235	
	Production	13369793.5	4038589.7	14134553.0	3958090.48	13875745.8	3981981.2	4.9	0.027	active
	Offtake	865963.1	780255.6	667450.2	660918.81	720281.0	696846.1	10.8	0.001	vacant
	Equity	12503830.4	3856514.8	13467102.8	3876927.40	13155464.8	3880389.5	8.3	0.004	active

Table A4-9. Estimated number of large grazing animals and overall vegetation productivity plus grazing offtake and equity. Sample sizes are given in Table A4-2.

		Grazers				Production		
	Status	Calves	Cattle	Sheep	Stags	Offtake	Produce	Equity
Western Seaboard	vacant	9.1	92.5	2064.2	15.7	139557.0	3968016.0	3828458.0
	active	13.9	76.9	1711.8	23.6	127767.0	3929881.0	3802113.0
	p	0.131	0.223	0.245	0.154	0.480	0.907	0.933
Caithness & Sutherland	vacant	30.5	101.2	1654.5	51.5	186561.0	4597094.0	4410533.0
	active	27.1	48.7	1215.8	43.2	109668.0	4303853.0	4194185.0
	p	0.439	0.151	**0.038**	0.265	0.121	0.146	0.239
Breadalbane	vacant	25.4	251.8	2679	38.1	378998.0	3651666.0	3272669.0
	active	32.0	171.1	2966.6	48.0	272773.0	3584962.0	3312189.0
	p	0.569	0.483	0.734	0.597	0.438	0.845	0.880
West Argyll	vacant	7	167	2288.8	16.7	239528.0	3958018.0	3718490.0
	active	14.9	216.3	1925.7	36.0	327030.0	4277924.0	3950894.0
	p	0.124	0.535	0.402	0.163	0.389	0.541	0.622
Lochaber	vacant	36.2	68.6	1750	53.2	151408.0	2983852.0	2832444.0
	active	27.7	75.4	1558.0	44.1	148075.0	3095491.0	2947416.0
	p	0.163	0.624	0.461	0.237	0.812	0.423	0.390
NW Seaboard	vacant	26.8	12.6	735.2	51.4	68748.0	4152076.0	4083328.0
	active	24.6	14.8	758.8	47.1	66835.0	4265504.0	4198669.0
	p	0.295	0.455	0.609	0.212	0.726	0.411	0.398
NE Glens	vacant	30.5	399.4	2545.9	134.8	631153.0	3876184.0	3245031.0
	active	34.6	318.3	1622.2	104.4	515344.0	3326506.0	2811161.0
	p	0.691	0.573	0.183	0.566	0.502	0.272	0.307
Cairngorm	vacant	46.2	162.3	1569.6	111.9	325155.0	3148832.0	2823677.0
	active	44.8	168	1673.7	108.7	332730.0	3349253.0	3016523.0
	p	0.812	0.884	0.705	0.855	0.871	0.385	0.350
Central Highlands	vacant	41.5	126.1	1366.3	19.6	217911.0	3146146.0	2928235.0
	active	45.6	117.2	1108.2	49.7	233030.0	2894828.0	2661799.0
	p	0.606	0.894	0.082	0.108	0.844	0.286	0.141
Western Highlands	vacant	53.5	25.5	773.3	76.4	119803.0	3005420.0	2885617.0
	active	60.3	37.2	830.3	88.6	147098.0	3782166.0	3635068.0
	p	0.493	0.092	0.626	0.380	**0.041**	**0.008**	**0.009**
Northern Highlands	vacant	36.5	108.6	929.5	67.6	216893.0	2744868.0	2527975.0
	active	41.5	76.5	804.1	76.4	179231.0	3028182.0	2848951.0
	p	0.120	0.075	0.066	0.214	0.084	**0.025**	**0.019**
Western Isles	vacant	2.9	37.2	1321.5	5.5	55995.0	3164829.0	3108834.0
	active	4.8	50.5	1842.0	9.4	74939.0	4253950.0	4179011.0
	p	**0.035**	0.483	**0.007**	**0.032**	0.430	**0.003**	**0.003**

Table A4-10. Number and density of Munros in different districts (after Bennet, 1991) and the number of active and vacant ranges (identified as 3 km contiguous Thiessen polygons around Munro tops).

District	Area km^2	Munros	Munro density (100 km^{-2})	Ranges	Active	Vacant	% vacant
Minginish and the Cuillin Hills	103.9	12	11.55	4	4	0	0
Fort William to Loch Treig and Loch Le	217.8	19	8.72	4	4	0	0
The Fannaichs	147.4	10	6.78	3	2	1	33
Glen Affric to Glen Moriston	74.5	5	6.71	4	4	0	0
Loch Duich to Cannich	254.9	17	6.67	9	5	4	44
Glen Shiel to Loch Hourn & Loch Quoich	197.3	13	6.59	5	3	2	40
Braemar to Montrose	214.8	14	6.52	3	2	1	33
Loch Lomond to Strathyre	124.2	8	6.44	2	1	1	50
Loch Maree to Loch Broom	144.4	9	6.23	4	4	0	0
Loch Linnhe to Loch Etive	178.4	11	6.16	5	5	0	0
Loch Broom to Strath Oykel	97.7	6	6.14	3	2	1	33
Glen Etive to Glen Lochy	218.8	13	5.94	4	4	0	0
Kyle of Lochalsh to Garve	187.4	11	5.87	5	2	3	60
Glen Lyon to Glen Dochart and Loch Tay	241.3	14	5.80	2	2	0	0
Loch Treig to Loch Ericht	244.4	14	5.73	2	1	1	50
Pitlochry to Braemar and Blairgowrie	192.3	11	5.72	3	1	2	67
The Cairngorms	315.1	18	5.71	6	3	3	50
Loch Leven to Rannoch Station	35.6	2	5.62	1	1	0	0
Loch Torridon to Loch Maree	107.1	6	5.60	2	1	1	50
Inveraray to Crianlarich	165.5	9	5.44	2	2	0	0
Scourie to Lairg	37.3	2	5.37	1	1	0	0
Applecross to Achnasheen	57.0	3	5.26	2	1	1	50
Killilan to Inverness	76.6	4	5.22	2	1	1	50
Strathyre to Strathallan	39.6	2	5.05	0			
Loch Lochy to Loch Laggan	100.5	5	4.97	1	0	1	100
Loch Rannoch to Glen Lyon	241.6	12	4.97	2	2	0	0
Loch Arkaig to Glen Moriston	40.7	2	4.92	1	0	1	100
Knoydart to Glen Kingie	166.9	8	4.79	3	3	0	0
Loch Ericht to Glen Tromie and Glen Ga	150.6	7	4.65	3	1	2	67
Glen Albyn and the Monadh Liath	92.4	4	4.33	2	1	1	50
Glen Tromie to Glen Tilt	99.9	4	4.01	2	0	2	100
Mallaig to Fort William	77.3	3	3.88	2	2	0	0
Altnaharra to Dornoch	28.3	1	3.54	0			
Durness to Loch Shin	28.3	1	3.54	1	1	0	0
Loch Tay to Perth	28.3	1	3.54	1	1	0	0
Loch Vaich to the Moray Firth	28.3	1	3.54	2	1	1	50
Mull and Nearby Islands	28.3	1	3.54	1	1	0	0

Table A4-11. Mean number and distance from each range class (active and vacant) to the nearest Munro (within a 6 km threshold).

	Vacant (52)		Active (132)			
	mean	s.d.	mean	s.d.	t	p
Number	3.2	2.1	3.3	2.4	0.34	0.738
Mean distance	3831.0	1250.0	3960.0	977.0	0.51	0.508

Table 4-12. Distance to the nearest golden eagle range (active or vacant) from each Munro at three thresholds (6, 5 and 4 km).

Threshold	status	n	mean	s.e.	95% C.L. for difference	t	p
6 km	vacant	39	2161	204			
	active	98	2617	121	-929, 18	-1.92	0.059
5 km	vacant	37	2003	181			
	active	95	2527	112	-949, -99	-2.46	0.017
4 km	vacant	35	1847	153			
	active	85	2295	103	-836, -99	-2.53	0.014

Table A4-13. Mean distance from each Munro (n = 274) to its nearest active or vacant golden eagle range centre.

	Range count		Mean Munro distance (km)				
NHZ	Active	Vacant	Active	Vacant	Combined	t	p
North West Seaboard	10	5	2.55	2.62	2.57	-0.24	0.82
Peatlands of Caithness & Sutherland	2		3.71		3.71		
Western Seaboard	13		2.28		2.28		
Northern Highlands	34	18	2.77	2.09	2.53	2.26	0.03
Western Highlands	18	7	2.98	2.75	2.92	0.35	0.74
Central Highlands	7	4	3.24	2.16	2.85	1.34	0.22
Cairngorm Massif	21	27	2.98	2.60	2.77	1.38	0.17
Lochaber	49	13	3.01	3.23	3.06	-0.52	0.61
Argyll West and Islands	2	2	4.34	5.11	4.72	-0.47	0.72
Breadalbane and East Argyll	22		3.07		3.07		
All	178	76	2.92	2.64	3.05	1.63	0.11

Table A4-14. Poisoning incidents (see Annex 1) by NHZ (excluding those with no golden eagle ranges). The expected numbers of incidents are derived from the NHZ area. NHZ are ranked by the magnitude of the standardized residual between the number of observed and expected incidents.

NHZ	Incidents	Area (km^2)	Expected	Residual
More than expected				
Breadalbane and East Argyll	34	3550	10.2	55.84
Central Highlands	23	2732	12.0	10.21
North East Glens	20	3766	16.5	0.74
Cairngorm Massif	20	4036	17.7	0.31
Less than expected				
North West Seaboard	0	3646	16.0	15.95
Northern Highlands	5	5491	24.0	15.06
Western Seaboard	0	3118	13.6	13.64
Western Highlands	1	2652	11.6	9.69
Western Isles	3	3246	14.2	8.84
Lochaber	1	2423	10.6	8.70
Peatlands of Caithness & Sutherland	9	5188	22.7	8.26
North Caithness and Orkney	0	1726	7.6	7.55
Argyll West and Islands	13	5198	22.7	4.17
Western Southern Uplands & Inner Solway	22	6697	29.3	1.82
Border Hills	14	4130	18.1	0.92

Table A4-15. Number and altitude of nests recorded in the 1992 national survey (sample size in brackets).

	Vacant (57)		Active (295)		All (357)			
	mean	sd	mean	sd	mean	sd	F	Sig.
Number	2.6	1.7	3.7	2.09	3.5	2.07	14.4	0.000
Mean altitude	375.2	190.5	342.2	156.21	347.5	162.41	2.0	0.160
Altitude s.d.	47.9	54.9	56.1	46.25	54.8	47.77	1.4	0.232

Table A4-16. Nest aspect for non-coastal ranges in the 1992 national survey (Watson's F test for two circular means indicated no significant difference (p>0.3) between the mean nest aspects for ranges with one nest and ranges with two or more nests).

	Range features		
Nest aspect	Single nest	Two or more nests	All
N	6	38	44
NE	11	41	52
E	12	34	46
SE	7	20	27
S	3	15	18
SW	6	16	22
W	7	20	27
NW	17	32	49
n	69	216	285
Mean angle	6.1°	25.3°	21.5°
Standard error	24.3°	10.8°	10.0°
Lower 95% C.I.	318.4°	4.1°	1.8°
Upper 95% C.I.	53.9°	46.4°	41.0°

ANNEX 5: FURTHER EXAMINATIONS OF LAND USE INFLUENCES AND PREDICTING THE POTENTIAL FOR RANGE EXPANSION

NHZ HABITAT AND RANGE FREQUENCY

If the number of golden eagle ranges is dependent on particular habitats that vary between NHZ there should be correlations between the number of ranges in a NHZ and the extent of certain habitats. Table A5-1 lists correlation coefficients with and without data from the NHZ without golden eagles in the 1992 survey. The pattern of correlations was as expected with more known ranges where there was more heather and cliffs, i.e. both foraging and nesting habitat were abundant. Conversely, there were fewer ranges where there is more improved and rough grazing.

Table A5-1. Correlation coefficients between the areas of 34 LCS88 habitat types and the numbers of active (excluding single adult ranges) and vacant golden eagle ranges in the NHZ. Correlations were calculated using all NHZ and after excluding NHZ that had no ranges. Significant correlations (p < 0.05, 2-tailed with no adjustment for multiple testing) are in bold.

	Regions without breeding eagles excluded			Regions without breeding eagles included		
LCS88 habitat class	**all**	**vacant**	**active**	**all**	**vacant**	**active**
1. Arable	*-0.351*	*-0.311*	*-0.331*	*-0.395*	*-0.215*	**-0.461**
2. Improved Grassland	**-0.554**	**-0.483**	**-0.517**	**-0.676**	**-0.548**	**-0.560**
3. Good Rough Grassland	*-0.381*	*-0.318*	*-0.341*	**-0.519**	*-0.403*	**-0.494**
4. Poor Rough Grassland	*-0.346*	*-0.324*	*-0.315*	**-0.545**	**-0.485**	**-0.547**
5. Bracken	*-0.089*	*-0.019*	*-0.120*	*-0.227*	*-0.122*	*-0.320*
6. Heather Moorland	*0.420*	**0.748**	*0.190*	*0.168*	**0.661**	*-0.227*
7. Peatland	*0.156*	*0.347*	*0.048*	*-0.028*	*0.242*	*-0.203*
8. Montane	**0.580**	**0.656**	**0.460**	*0.415*	**0.535**	*0.192*
9. Rocks and Cliffs	**0.658**	**0.441**	**0.699**	**0.621**	*0.337*	**0.654**
10. Felled Woodland	*-0.267*	*-0.225*	*-0.241*	*-0.348*	*-0.278*	*-0.421*
11. Recent Planting	*0.109*	*0.106*	*0.131*	*-0.171*	*-0.139*	*-0.223*
12. Coniferous Plantation	*-0.191*	*-0.103*	*-0.193*	*-0.415*	*-0.269*	**-0.538**
13. Semi-Natural Coniferous	*0.245*	**0.576**	*0.047*	*0.101*	**0.528**	*-0.204*
14. Mixed Woodland	*-0.287*	*-0.265*	*-0.263*	*-0.160*	*-0.148*	*-0.251*
15. Broadleaved	*0.124*	*0.050*	*0.162*	*0.115*	*0.013*	*0.064*
16. Scrub	*-0.412*	*-0.347*	*-0.397*	*-0.319*	*-0.088*	**-0.484**
17. Freshwaters	**0.559**	**0.487**	**0.505**	*0.450*	*0.361*	*0.346*
18. Marsh	*-0.386*	*-0.379*	*-0.347*	*-0.407*	*-0.390*	*-0.259*
19. Saltmarsh	*-0.264*	*-0.347*	*-0.189*	*-0.156*	*-0.293*	*-0.107*
20. Dunes	*0.107*	*-0.071*	*0.164*	*0.205*	*-0.032*	*0.390*
21. Tidal Waters	*-0.418*	*-0.362*	*-0.393*	*-0.329*	*-0.185*	*-0.362*
22. Rural Development	**-0.439**	*-0.409*	*-0.402*	**-0.522**	**-0.509**	*-0.361*
23. Urban	*-0.365*	*-0.340*	*-0.335*	*-0.403*	*-0.462*	*-0.279*
24. Missing or Obscured	*0.262*	*0.362*	*0.196*	*0.090*	*0.237*	*-0.053*
25. Heather Moorland / Peatland	**0.818**	**0.766**	**0.749**	**0.731**	**0.665**	**0.595**
26. Poor Rough Grass / Heather Moorland	*0.264*	*0.024*	*0.356*	*0.021*	*-0.270*	*0.081*
27. Good Rough Grass / Heather Moorland	**0.610**	**0.567**	**0.585**	**0.519**	*0.467*	**0.471**
28. Peatland / Montane	**0.487**	**0.560**	*0.370*	*0.330*	*0.440*	*0.118*

LCS88 habitat class	Regions without breeding eagles excluded			Regions without breeding eagles included		
	all	**vacant**	**active**	**all**	**vacant**	**active**
29. Good Rough Grass / Poor Rough Grass	*-0.267*	*-0.309*	*-0.211*	*-0.420*	*-0.447*	*-0.402*
30. Improved Grassland / Good Rough Grass	*0.126*	*-0.046*	*0.214*	*0.211*	*-0.047*	*0.257*
31. Good Rough Grass / Bracken	*0.235*	*0.008*	*0.344*	*0.122*	*-0.150*	*0.227*
32. Poor Rough Grassland / Peatland	*-0.001*	*-0.125*	*0.067*	*-0.210*	*-0.340*	*-0.185*
33. Heather Moorland / Montane	**0.482**	**0.705**	*0.298*	*0.409*	**0.688**	*0.136*
34. Remaining Mosaics	*0.382*	*0.136*	**0.461**	*0.363*	*0.033*	*0.395*

There were some interesting differences in the magnitude of the correlations between range frequency and habitat extent when active and vacant ranges were analysed separately (Table A5-1). For example, vacant range frequency was highly positively correlated with the extent of heather moorland and montane habitats. However, this probably reflected differences in the between-NHZ abundance of habitats and the frequency of vacant ranges (i.e. a high frequency of vacant ranges in the eastern Highlands where heather moorland and montane vegetation were more prevalent), and echoed other findings concerning the strong national influence of range vacancies in the eastern Highlands.

DENSITY AND 'FREE' SPACE

Estimated densities (relative to all land area within a NHZ) suggested that the highest densities (of only active ranges and both active and vacant ranges in the 1992 survey) were in the west, with lowest densities in the eastern Highlands and south of the Highlands (Table A5-2).

Table A5-2. Density (1000 km^{-2}) of all known and active ranges in each NHZ. [1]The area for 'All' excludes NHZ that have no eagle ranges.

NHZ	**Area (km^2)**	**All ranges**		**Active ranges**	
		n	**density**	**n**	**density**
North Caithness and Orkney	1726	1	0.6	0	0
Western Isles	3246	82	25.3	58	17.9
North West Seaboard	3646	68	18.6	42	11.5
The Peatlands of Caithness & Sutherland	5188	31	6.0	14	2.7
Western Seaboard	3118	96	30.8	73	23.4
Northern Highlands	5491	92	16.8	47	8.6
Western Highlands	2652	66	24.9	55	20.7
Central Highlands	2732	26	0.9	11	4.0
Cairngorm Massif	4036	67	16.6	28	6.9
North East Glens	3766	19	5.1	8	2.1
Lochaber	2423	37	15.3	27	11.1
Argyll West and Islands	5198	55	10.6	43	8.3
Breadalbane and East Argyll	3550	27	7.6	20	5.6
Western Southern Uplands & Inner Solway	6697	4	0.6	3	0.4
Border Hills	4130	2	0.5	1	0.2
All[1]	57597	673	11.7	430	7.5

In those NHZ with high golden eagle densities there appeared to be little room for further expansion beyond the areas which had been occupied at some point up to 1992 (Table A5-3). Most of the high density areas were in the Western Highlands and Islands, and even when large areas of these NHZs appeared not to be part of territory occupied at some recent time by resident golden eagles (i.e. outside of a 6 km Thiessen polygon), much of the remaining habitat appears unsuitable (Fielding *et al*., 2003a: Appendix 4). Thus, in the west there appeared to be little scope to extend the Scottish golden eagle population beyond the known ranges, particularly since much of the unused space is either unsuitable or present as fragmented blocks of upland habitat that may be too small to support an eagle range. In other NHZ, such as in the eastern Highlands and Borders, there would appear to be some room for expansion beyond those areas classed as active or vacant ranges.

Table A5-3. Area (km^2) of each NHZ that was enclosed and excluded by 3 km and 6 km Thiessen polygons constructed around all known ranges. Contiguous Thiessen polygons were used in preference to circular buffers to avoid overestimating the extent of known eagle range within a NHZ.

	3 km Thiessen			6 km Thiessen		
NHZ	outside	inside	% outside	outside	inside	% outside
Western Highlands	1148	1504	43.3	42	2610	1.6
Western Seaboard	1378	1740	44.2	158	2959	5.1
Lochaber	1528	894	63.1	307	2116	12.7
North West Seaboard	2028	1618	55.6	557	3088	15.3
Northern Highlands	3238	2252	59.0	849	4641	15.5
Cairngorm Massif	2393	1643	59.3	645	3391	16.0
Western Isles	1781	1465	54.9	688	2558	21.2
Central Highlands	2026	706	74.2	861	1871	31.5
Breadalbane and East Argyll	2779	771	78.3	1156	2394	32.6
Argyll West and Islands	3983	1214	76.6	1978	3220	38.1
Peatlands of Caithness and Sutherland	4355	833	83.9	2623	2565	50.6
North East Glens	3291	476	87.4	2342	1425	62.2
Western Southern Uplands & Inner Solway	6589	108	98.4	6311	385	94.3
North Caithness and Orkney	1699	28	98.4	1638	90	94.8
Border Hills	4084	46	98.9	3949	180	95.6
Moray Firth	1987	2	99.9	1961	27	98.6
Eastern Lowlands	8593	13	99.8	8528	78	99.1
Shetland	1468	0	100.0	1468	0	100.0
North East Coastal Plain	3227	0	100.0	3227	0	100.0
West Central Belt	5192	0	100.0	5192	0	100.0
Wigtown Machairs and Outer Solway	756	0	100.0	756	0	100.0

PREDICTIVE MODELLING

If the national golden eagle population is to expand three requirements must be met:

1. There should be sufficient unused suitable foraging habitat surrounding suitable potential nest sites.
2. Productivity and juvenile survival should be high enough to provide the recruits needed to fill new ranges.
3. Adult survival needs to be sufficiently high so that birds occupying new ranges are able to remain long enough to allow successful reproduction and minimize turnover in range occupation.

In this section the first of these requirements is investigated by attempting to identify areas of currently unoccupied habitat that have the potential to become golden eagle ranges. As a starting point there are over 200 known ranges that are apparently no longer in use. If suitable regions can be identified it may be possible to suggest appropriate local management schemes, including tree planting for nesting opportunities, that would increase the potential for occupancy.

If we assume that there is some relationship between habitat characteristics and the probability that a golden eagle will establish a range, i.e. p (range) = f (habitat), it may be possible to develop a predictive model to identify regions that could support additional ranges. The difficulty is identifying the most appropriate form for the model given the constraints of available predictors (habitat information).

Two approaches have been tested here. The first uses a series of hierarchical artificial neural networks (strictly multi-layer perceptrons) to derive the most appropriate form for the model. Artificial neural networks offer a number of advantages over other ‘standard’ methods:

- There is no presumption about the probability distribution of the model error term.
- In its most simple format (no hidden layer) a neural network produces a model that is indistinguishable from a generalized linear model (GLM).
- Unlike a GLM there is no need to use a variable selection algorithm.
- If each potential predictor has its own set of hidden layer neurons a large number of smooth data transformations will be automatically searched, thus there is no requirement for initial data transformations.
- If all predictors are linked via a common hidden layer then transformations and interactions are automatically searched, i.e. there is no need to pre-specify interactions.
- Finally, if separate networks are built for each predictor and their outputs used as the inputs to a second network it is possible to approximate a generalized additive model (GAM).

The software used (NEVPROP3; Goodman, 1998) has the additional advantage that cross-validation can be used to avoid overfitting and bootstrapping provides an estimate of the ‘relevance’ of each predictor (with or without interactions). Boddy & Morris (2000) and Lek & Geugan (1999) have written general introductions to the use of artificial networks with ecological problems.

The second modelling approach uses decision tree software to construct recursive, rule-based, partitions that are generally relatively easy to convert into ecological mechanisms (Bell, 1999). Decision trees are also advantageous because they model constraints rather than correlations. Huston (2002) argued that we should concentrate on an analysis of constraints rather than correlates because constraints can be ranked and management can then concentrate on the most severe local constraint operating for a particular set of ranges. Decision trees also have the advantage that outliers (ranges with unusual characteristics) will be isolated in their own branch and will not disrupt the covariance matrices used in many other methods (Bell, 1999).

In both approaches it is general range characteristics, rather than specific nest requirements, that are modelled.

Modelling Data

Analyses were restricted to the national scale because of the need for large sample sizes. However, in the decision tree analyses the NHZ was included as a potential categorical predictor. There were 47 potential predictor variables, including topographic, habitat and road frequencies. A full list is in Table A5-6. Grazing data were not used because they were obtained at the parish level and there is a very large discrepancy in the size of parishes between the main golden eagle regions and the rest of Scotland.

Habitat data were extracted from 3 km radius circles or buffers (excluding sea) centred on all known golden eagle ranges (n = 673). The same data were also obtained from locations with no known ranges, referred to as ‘holes’ or random ranges. Random range data (n = 3504) were extracted using 3 km radius circles centred on a regular grid (4 km centres). Data were not extracted if a grid point was less than 3 km from a known range centre. Although the combination of a 4 km grid with a 3 km extraction buffer created some overlap the alternative (a 6 km grid) resulted in a much smaller sample size and large ‘gaps’ between the extraction buffers.

In any complex modelling process there is always a danger that the model will be overfitted to the particular characteristics of the cases used to generate the model. In such circumstances the model is likely to lack generality and fail when presented with new cases. Consequently, we gain little if all that we predict is the distribution of the individuals used to produce the predictions (Beutel *et al*., 1999; Fielding, 1999, 2002; Verbyla & Litvaitis, 1989). It is essential that the model is tested on data that are independent of those used to generate the prediction rules. In other words it is important to have some idea about how well the model will perform with new data. This is needed because the accuracy achieved with the original data is often much greater than that achieved with new data (Henery, 1994). Consequently, two data sets are needed to develop and test predictions. The terms ‘training’ and ‘testing’ data are used here. The problem now becomes one of finding appropriate training and testing data. Ecologists seem to have paid little attention to the range of available methods, or how the choice may influence the estimated error rates. One exception is Verbyla & Litvaitis (1989), who briefly reviewed a range of partitioning methods in their assessment of resampling methods for evaluating classification accuracy. Resubstitution (reuse of the training data) is the simplest way of testing the performance of a model. Unfortunately, this provides a biased assessment of future performance, possibly because the form of the predictions have been determined by some model-selection algorithm (e.g. stepwise variable selection). An inevitable consequence of model selection processes is that the final model tends to ‘overfit’ the training data because it has been optimized to deal with the nuances in the training data. This bias may still apply if the same set of ‘independent’ testing data has been used to verify the model selection (Chatfield, 1995). The best assessment of a model's future value is to test it with some truly independent data; ideally a sample collected independently of the training data ('prospective sampling'). Because this is often difficult, a common practice is to split or partition available data to provide training and the 'independent' testing data. There is a trade-off between having a large test set that gives a good assessment of the model's performance and a small training set which is likely to result in a poor classifier. Huberty (1994) provided a heuristic for determining the ratio of training to testing cases. This heuristic suggests a minimum ratio of $[1 + (p - 1)^{½}]^{-1}$, where p is the number of predictors. For example, if p = 10 the testing set should be $1/[1 + \sqrt{9}]$, or 25% of the complete data set. In this study 26 % of cases were used for testing, almost twice the minimum proportion suggested by the heuristic.

The complete set of 4177 records was split randomly into a training set (n = 3504, 360 active ranges (including 23 occupied by a single adult) and 2904 random ranges) and a test set (n = 913, 100 active ranges (including 7 occupied by a single adult), 18 ranges with unknown status, 195 vacant ranges, 600 random ranges). Vacant and unknown status ranges were excluded from the training set because it was thought that they might introduce too much 'noise'. The distribution of training and test cases is shown in Fig. A5-1. This data partitioning strategy was recently shown to be the best when building large-scale distribution models (Osborne & Suárez-Seoane, 2002). The same training and testing data sets were used for both the neural and decision tree modelling processes.

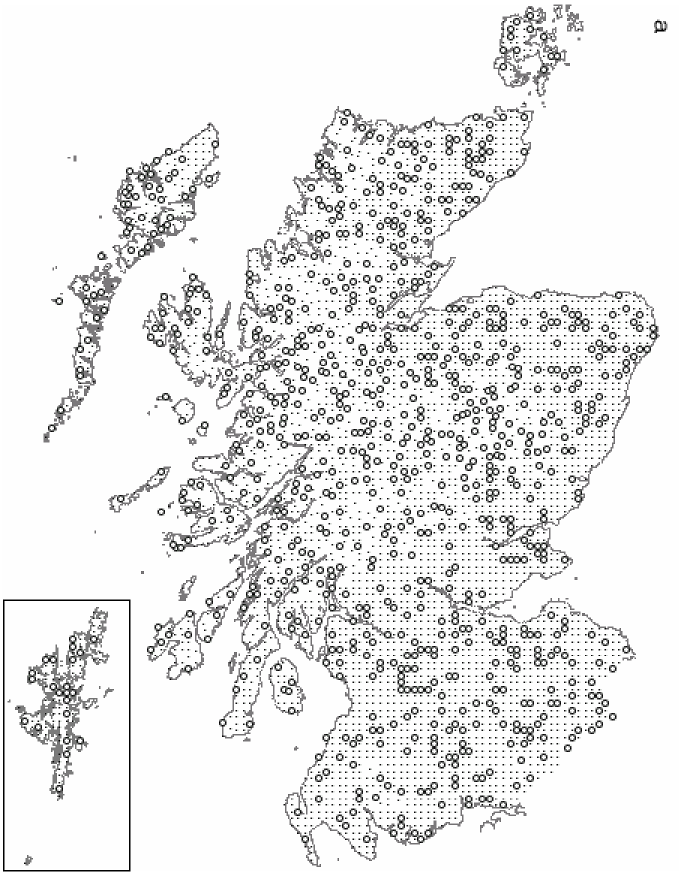


Fig. A5-1. (a) Distribution of training (dot) and testing locations (open circle) and (b) (available only in confidential version of Fielding et al., 2003a) distribution of active (filled circle), vacant (open circle) and random ranges (dot).

Neural network modelling

Six neural net architectures of increasing complexity were investigated (see Fig. A5-2 for schematic diagrams).

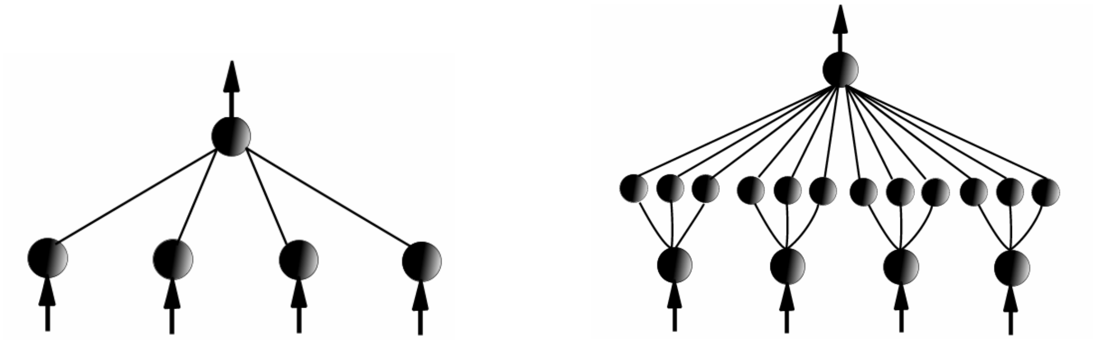
1. GLM_NN - simple with no hidden layer,
2. TRANS_NN – each predictor has a separate group of three hidden (transforming) neurons,
3. HID11 – multiplayer perceptron with a hidden layer of 11 neurons,
4. HID21 - multiplayer perceptron with a hidden layer of 21 neurons,
5. GAM_NN – simulated generalized additive model (output from 47 single predictor multiplayer perceptrons (4 hidden neurons) were used as inputs to a separate single layer network. 4 hidden neurons were selected because the approximate degrees of freedom for some trial loss (locally weighted regression) models were approximately 4.
6. GAMHID_NN – as for the GAM_NN but the upper network had a hidden layer of 11 neurons).

The best model was selected according to four criteria.

1. F statistics from an analysis of variance comparing the mean neural network output scores for active and random ranges. Larger F statistics indicate better separation.
2. The best model should be as simple as possible (rank order is GLM_NN, TRANS_NN, HID11, HID21, GAM_NN and GAMHID_NN).
3. Nevprop provides a C-index, which is the area under a ROC (Receiver-Operating Characteric) curve (AUC). The AUC (range 0.0 – 1.0) is usually taken as a performance index because it provides a single measure of overall accuracy that is independent of any particular threshold used to allocate cases to classes (Deleo, 1993). If the value is 0.5 the scores for two groups do not differ and a model would perform no better than a coin toss. Conversely an AUC of 1.0 indicates no overlap in the distributions of the group scores and the model would never misclassify. An AUC of 0.75 indicates that, on 75% of occasions, a random selection from the positive group will have a score greater (e.g. a neural network output score) than a random selection from the negative class (Deleo, 1993).
4. Accuracy of the predictions was assessed after applying a 0.5 threshold to the range predictions. Predicted values <0.5 were assigned a random status, while values >0.5 were allocated to the active class. The best model will have the greatest sensitivity (active ranges predicted correctly) and specificity (random ranges correctly predicted).

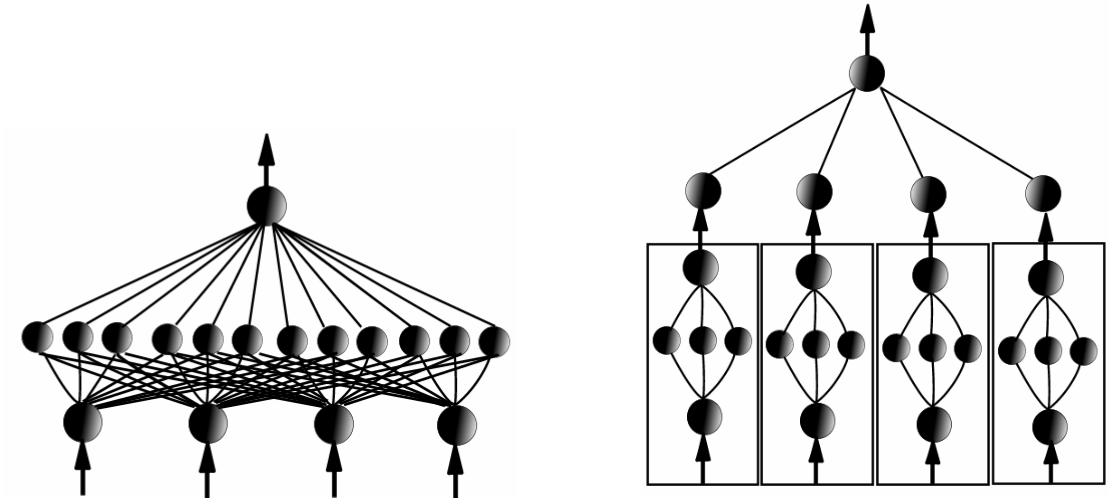
Summarised comparative results are given in Table A5-4. All models had high specificity and the C index was very large for both training and testing data. Using the criteria listed above the network with the smallest hidden layer (HID11) appears to be the best model. Although all subsequent analyses are restricted to its predictions and interpretation, there were only minor differences between the predictions of all models.

Because the aim was to work with binary rather than real valued predictions it is important that an appropriate threshold is applied to the neural network output (Fielding & Bell, 1997; Manel *et al*., 2001; Fielding, 2002). The default threshold for most neural and statistical classification techniques is the mid point of a continuous output variable (usually bounded 0 – 1). Thus a case with an output score <0.5 is usually assigned to class 0 while a case with a score >0.5 is assigned to class 1. Because this approach can be criticized on several grounds (Fielding & Bell, 1997; Manel *et al*., 2001; Fielding, 2002) a threshold was selected, for the best model only, by examining how sensitivity and specificity varied with threshold (Fig. A5-3). A value of 0.2 was used with the selected model because it is the threshold where the sensitivity and specificity curves cross, ensuring that both are maximized. In all subsequent analyses range status predictions for the HID11 were obtained after applying the 0.2 threshold (>0.2 = active).



Simulated Generalized Linear Model (GLM). The output is a weighted sum of the predictors.

Transformation layer (TRANS_NN). The output is a weighted sum of the predictors, but each predictor may have been transformed by its 3 hidden neurons.



Multilayer perceptron (HID11 and HID21). This similar to the TRANS_NN, except that links between the inputs allow for linear and non-linear interactions.

Simulated Generalized Additive Model (GAM). Outputs from 4 multilayer perceptrons (equivalent to a Loess regressions) form the inputs to a simulated GLM.

Fig. A5-3. Schematic diagrams of four neural network architectures. The actual number of neurons differs in the real networks. The lower arrows indicate inputs from the habitat predictors (4 in these examples). The upper arrow is the output from the neural network (a real value in the range 0 – 1). Shaded circles are nodes, or processing units. Lines indicate links between different nodes.

In Table A5-5 predictive accuracy is broken down by NHZ. Sensitivity is generally reasonable in the training data, but less so for the testing data. In particular three zones (Central Highlands, Argyll West & Islands and Breadalbane and East Argyll) are a concern, although the first and third have relatively few active ranges in the test data. However, with few exceptions, specificity is high in all NHZ for both training and testing data. Since this model is concerned with identifying potential ranges specificity is more important than sensitivity. This is because we would like to identify potential ranges that have a large similarity to existing active ranges (Fielding, 2002), i.e. a conservative model is preferred over an optimistic one that could identify large blocks of apparently suitable habitat. The predictions for active ranges, using the neural network model, are shown in comparison with actual ranges in Figure A5-6.

Table A5-4. Summary of performance indicators, using training data, for six neural models of golden eagle range distribution: (a) F statistics; (b) prediction accuracy (with 0.5 threshold). The C-index (area under the ROC curve, Fielding and Bell (1997)) is given for training and testing data.

	(a)	(b)		C-index	
Network	F	Sensitivity	Specificity	Training	Testing
HID11	6314.6	0.77	0.97	0.981	0.920
HID21	6141.6	0.75	0.98	0.980	0.923
GAMHID_NN	5089.2	0.66	0.98	0.973	0.920
TRANS_NN	4603.6	0.68	0.98	0.969	0.921
GLM_NN	4515.7	0.70	0.97	0.969	0.917
GAM_NN	3764.5	0.65	0.97	0.961	0.912

Table A5-5. Performance of the HID11 model within each NHZ. Results are shown for training and testing data. Number of predicted ranges is the number of ranges, within each category, that were predicted to be active after applying a 0.2 threshold (>0.2 = active).

Training data	Sample sizes			Ranges predicted active				
NHZ	active	random	all	active	random	All	Sensitivity	Specificity
Shetland	0	71	71	0	5	5		0.930
North Caithness and Orkney	0	85	85	0	1	1		0.988
Western Isles	46	65	111	45	4	49	0.978	0.938
North West Seaboard	40	74	114	39	21	60	0.975	0.716
Caithness & Sutherland Peatlands	13	183	196	9	8	17	0.692	0.956
Western Seaboard	61	46	107	58	11	69	0.951	0.761
Northern Highlands	46	113	159	45	24	69	0.978	0.788
Western Highlands	42	21	63	42	10	52	1.000	0.524
North East Coastal Plain	0	164	164	0	0	0		1.000
Central Highlands	11	73	84	9	10	19	0.818	0.863
Cairngorm Massif	19	85	104	15	20	35	0.789	0.765
North East Glens	8	153	161	6	3	9	0.750	0.980
Lochaber	18	62	80	17	19	36	0.944	0.694

Training data	Sample sizes			Ranges predicted active				
NHZ	active	random	all	active	random	All	Sensitivity	Specificity
Argyll West and Islands	34	166	200	34	29	63	1.000	0.825
Breadalbane and East Argyll	19	115	134	18	23	41	0.947	0.800
Eastern Lowlands	0	458	458	0	0	0		1.000
West Central Belt	0	269	269	0	4	4		0.985
Wigtown Machairs & Outer Solway	0	40	40	0	0	0		1.000
Western Southern Uplands	3	345	348	2	3	5	0.667	0.991
Border Hills	0	215	215	0	1	1		0.995
Moray Firth	0	101	101	0	0	0		1.000
All	360	2904	3264	339	196	535	0.942	0.930

Testing data	Sample sizes					Ranges predicted active					
NHZ	active	random	un-known	vacant	all	active	random	un-known	vacant	Sensitivity	Specificity
Shetland	0	22	0	0	22	0	1				0.955
North Caithness and Orkney	0	18	0	1	19	0	0		0		1.000
Western Isles	15	15	5	16	51	11	0	4	9	0.733	1.000
North West Seaboard	9	13	0	19	41	9	6		10	1.000	0.538
Caithness & Sutherland Peatlands	1	54	0	17	72	0	1		1	0.000	0.981
Western Seaboard	13	6	2	20	41	10	1	1	9	0.769	0.833
Northern Highlands	10	23	0	36	69	8	3		19	0.800	0.870
Western Highlands	15	11	3	6	35	12	5	3	4	0.800	0.545
North East Coastal Plain	0	35	0	0	35	0	0				1.000
Central Highlands	3	23	2	10	38	1	1	1	2	0.333	0.957
Cairngorm Massif	9	16	6	33	64	8	3	4	13	0.889	0.813
North East Glens	0	36	0	11	47	0	0		0		1.000
Lochaber	10	8	0	9	27	9	3		2	0.900	0.625
Argyll West and Islands	10	34	0	11	55	5	2		2	0.500	0.941
Breadalbane and East Argyll	4	23	0	4	31	2	3		0	0.500	0.870
Eastern Lowlands	0	77	0	0	77	0	0				1.000
West Central Belt	0	55	0	0	55	0	0				1.000
Wigtown Machairs & Outer Solway	0	7	0	0	7	0	0				1.000
Western Southern Uplands	0	61	0	1	62	0	0		0		1.000
Border Hills	1	38	0	1	40	0	1		0		0.974
Moray Firth	0	25	0	0	25	0	0				1.000
All	100	600	18	195	913	75	30	13	71	0.750	0.950

In addition to the model predictions it is also important to examine the model structure so that the contribution that each predictor makes can be estimated. In a GLM, with no interaction terms, this would normally be achieved by examining the magnitudes, and signs, of the standardized coefficients. In a neural network model similar interpretations are possible by using percentage relevance statistics (Neal, 1996; Goodman, 1998). Percentage relevance statistics suggest the overall relative importance of explanatory variables, *including* any nonlinear effects. The last point is important because there is no simple way of estimating the contribution made by individual variables in a GLM that includes interaction terms. Variables with a small relevance measure are assumed to have relatively little impact on the predictions.

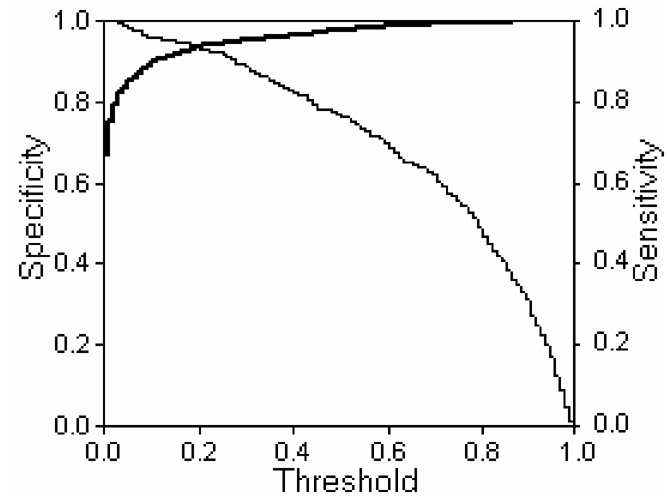


Fig. A5-4. Sensitivity (thin line) and specificity values for different threshold values applied to predictions from the neural network model for training data.

Six of the ten highest scoring relevance measures (Table A5-6) relate to topographic and solid geology variables, principally those associated with slope characteristics. These ten largest relevance measures account for almost 60% of the total. Only two vegetation measures feature in the ten largest relevance measures: bog and burnt heathland. Road length is also an important class of predictors. In general active ranges have greater and more variable slopes with fewer roads, more bogs and less areas of burnt heathland.

Only 21 of 360 active ranges were misclassified in the training data. In general these were thinly spread through the NHZ. Only the Peatlands of Caithness & Sutherland NHZ gave a cause for concern with 4 out 13 active ranges being misclassified. 75% (75 of 100) of the test active ranges were correctly identified. Sensitivity values were relatively low in two adjacent western NHZ: Western Isles and Argyll West and Islands. Between them they accounted for 9 of the 25 misclassified active ranges. Although two other NHZ (Central Highlands and Breadalbane & East Argyll) had lower sensitivity values, their sample sizes were very small.

Only 226 of the 3504 random ranges (196 in the training set and 30 in the test set) were predicted to be active (Table A5-5, Fig. A5-6). A further 104 (49%) of the 213 vacant and unknown status ranges were predicted to be active. Almost 70% (n = 136) of the random ranges from the training set, predicted to be active, were restricted to six NHZ (NW Seaboard, Northern Highlands, Cairngorm Massif, Lochaber, Argyll West and Islands and Breadalbane & East Argyll). The same NHZ accounted for 67% (n = 20) of the misclassified random ranges in the test set. Unsurprisingly, the number of misclassified random ranges per NHZ is correlated with the area of eagle habitat (land enclosed by 6 km Thiessen polygons drawn around all known range centres; Table A5-5, Fig. A5-5).

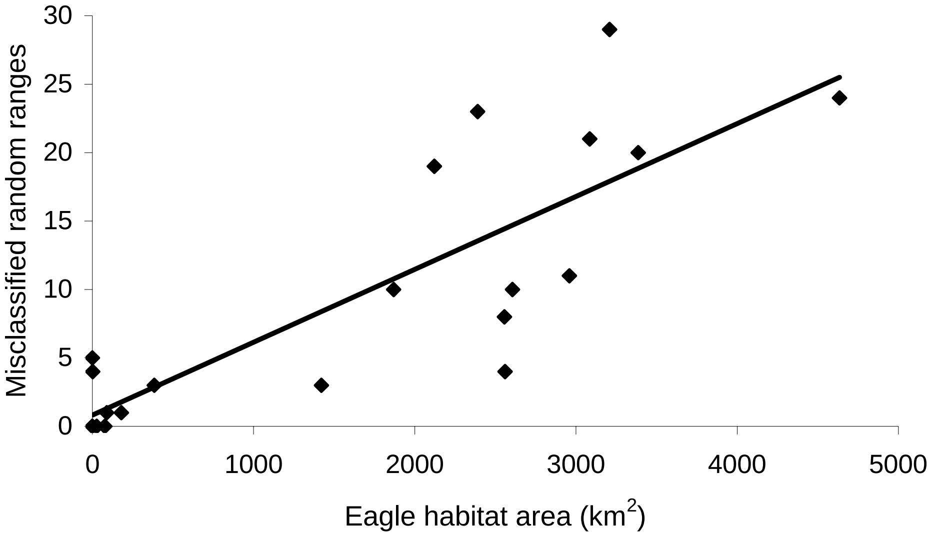


Fig. A5-5. Relationship between the number of misclassified random ranges and the area of eagle habitat (6 km Thiessens) within each NHZ. Regression equation: $y = 0.0053x + 0.8288$, $R^2 = 0.695$.

The residuals from Figure A5-5 indicate that five NHZ (Western Isles; Peatlands of Caithness and Sutherland; Western Seaboard; NE Glens and Western Highlands) had fewer predictions than expected, while the Breadalbane and East Argyll, Lochaber and Argyll West and Islands NHZ had considerably more than expected .

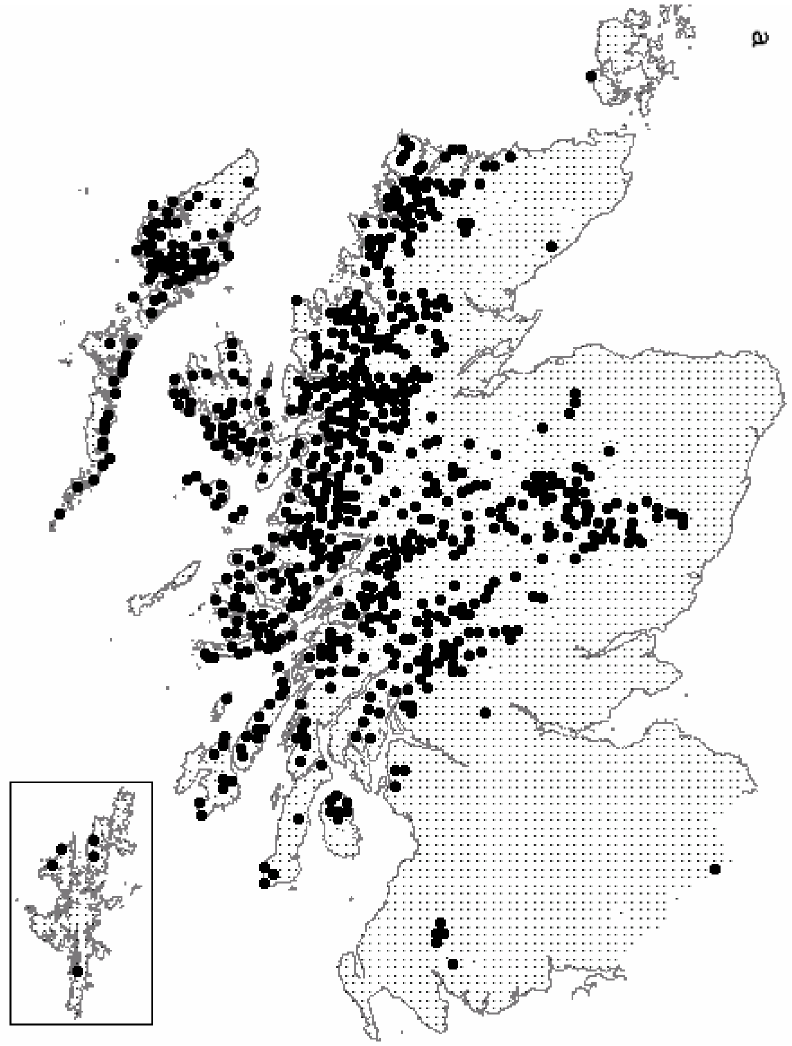


Fig. A5-6. (a) Predictions of range status using the HID11 network: active range (filled circle) and no range (dot). (b) (available only in confidential version of Fielding et al., 2003a) Distribution of actual active (filled circle), vacant (open circle), unknown (triangle) and random ranges (dot).

Table A5-6. Percentage relevance statistics for the neural network model.

[1] Slope variation measures were obtained by passing a 5 pixel radius kernel over a slope grid derived from a digital elevation model and returning the standard deviation of the slope for each kernel.

Predictor class and name		precentage relevance value	Sum	Mean
Slope	Maximum	6.46		
	Mean	0.69		
	Standard deviation	4.07	11.22	3.74
Slope variation[1]	Maximum	1.89		
	Mean	12.45		
	Standard deviation	6.34	20.68	6.89
Altitude	Minimum	2.31		
	Maximum	3.40		
	Mean	1.86		
	Standard deviation	0.88	8.45	2.11
Forest	Broadleaf	1.74		
	Closed	2.27		

Predictor class and name		precentage relevance		
Topography		value	Sum	Mean
	Mixed	0.73		
	Open	0.47	5.21	1.30
Grassland	Coarse	2.85		
	Smooth	1.98		
	Improved	0.51	5.34	1.78
Other	Arable	0.56		
	Bog	4.11		
	Bracken	0.95		
	Cliff	0.46		
	Human	0.15		
	Wetland	0.86		
	Montane	1.70		
	Range area	0.85	9.64	1.21
Heathland	Undifferentiated	0.13		
	Dry	1.66		
	Wet heath	0.78		
	Burnt	3.81	6.38	1.28
Solid geology	Acid igneous	3.07		
	Argillaceous	0.08		
	Basic igneous	1.04		
	Gneisses	5.95		
	Intermediate igneous	0.75		
	Lavas, etc	0.82		
	Limestone	0.62		
	Metamorphic igneous	0.52		
	Metamorphic sedimentary	0.89		
	Miscellaneous sedimentary	0.27		
	Mixed metamorphic	1.59		
	Sandstone	3.12	18.72	1.56
Road length	A Dual carriageway	0.03		
	A Single carriageway	1.86		
	Primary single carriageway	4.26		
	B Narrow	0.58		
	B Single	1.54		
	Minor	6.12	14.39	2.40

Decision tree models

Nine decision tree (six Quest and 3 Exhaustive CHAID) models were tested using SPSS AnswerTree (Release 2.0.1, SPSS Inc., 1998-1999). Quest (Quick, Unbiased, Efficient, Statistical Tree version 1.8.10: Loh & Shih, 1997), is a classification tree that was the best of the decision tree methods tested by Lim *et al.* (2000). Although CHAID (Chisquare-Automatic-Interaction-Detection) was designed originally to handle categorical variables only, SPSS extended it to include, ordinal, categorical and continuous dependent variables. Continuous independent variables are discretized (similar to ‘binning’ when drawing a histogram of a continuous variable) prior to evaluation. Because the basic CHAID algorithm is not guaranteed to find the ‘best’ split at each node it was later modified to perform an exhaustive search of all possible category subsets (Biggs *et al*., 1991). All models used 10-fold cross-validation to obtain an estimate of its potential performance with new data. The final trees were exported as SPSS syntax files to obtain range predictions for each case in the training and testing files. Nodes were only considered for a split if they contained at least 10 cases and either child node would have a minimum of two cases. Other conditions are described below.

1. Quest 1, equal prior probabilities and misclassification costs.
2. Quest 2, as above but prior probabilities were based on group sizes.
3. Quest 3, as 1 but misclassifying actual ranges as holes was ten times more expensive than the reverse misclassification.
4. Quest 4, as 3 but prior probabilities were based on group sizes.
5. Quest 5, as 4 but misclassification cost was halved to 5.
6. Quest 6, as 5 but prior probabilities were equal.
7. Chaid 1, default settings with equal costs
8. Chaid 2, as 7 but misclassifying actual ranges as holes was ten times more expensive than the reverse misclassification.
9. Chaid 3, as 8 but misclassification cost was halved to 5.

The best model was selected according to three criteria. The best model:

1. was as simple as possible (minimum number of nodes);
2. had the greatest sensitivity (active ranges predicted correctly) and specificity (random ranges correctly predicted);
3. had a simple ecological interpretation.

The summarised comparative results given in Table A5-7 suggest that the first Quest model was the best model and all subsequent analyses are restricted to its predictions.

Table A5-7. Comparison of the overall accuracy of nine decision trees with training and testing data. Cost refers to the relative cost of misclassifying an active range as random compared with the reverse error. Priors are the class prior probabilities for active and random ranges. When set to equal both are 0.5, otherwise they were determined by the relative class sizes. The actual number of members in each class, of the training data, were 360 (active) and 2904 (random).

Tree	Cost	Priors	active	sensitivity	random	specificity
Training data						
Quest	1	equal	300	0.83	2607	0.90
	1	group	157	0.44	2851	0.98
	10	equal	360	1.00	13	0.00
	10	group	302	0.84	2503	0.86
	5	equal	328	0.91	2187	0.75
	5	group	293	0.81	2554	0.88
Chaid	1	n.a.	147	0.41	2839	0.98
	10	n.a.	333	0.93	1320	0.45
	5	n.a.	176	0.49	2788	0.96
Testing data						
Quest	1	equal	83	0.83	533	0.89
	1	group	52	0.52	588	0.98
	10	equal	100	1.00	3	0.01
	10	group	84	0.84	513	0.86

Tree	Cost	Priors	active	sensitivity	random	specificity
	5	equal	94	0.94	433	0.72
	5	group	81	0.81	528	0.88
Chaid	1	n.a.	39	0.39	585	0.98
	10	n.a.	91	0.91	262	0.44
	5	n.a.	51	0.51	574	0.96
			Unknown ranges (n=18)		Vacant ranges (n = 195)	
				proportion predicted active		proportion predicted active
Quest	1	equal	11	0.61	117	0.60
	1	group	6	0.33	39	0.20
	10	equal	18	1.00	194	0.99
	10	group	10	0.56	116	0.59
	5	equal	16	0.89	155	0.79
	5	group	10	0.56	112	0.57
Chaid	1	n.a.	4	0.25	46	0.24
	10	n.a.	14	0.88	168	0.86
	5	n.a.	6	0.38	55	0.28

Sensitivity was good in most NHZ (Table A5-8). If the Western Southern Uplands and Border Hills NHZ are excluded (because of small numbers of active ranges), the greatest concerns relate to the NE Glens and Peatlands of Caithness and Sutherland NHZ. Both of these also had low sensitivity with the neural network model. Specificity was high in all but four NHZ: NW Seaboard; Western Highlands; Lochaber and Breadalbane & East Argyll). With the exception of the Western Seaboard all of the others also had low specificity with the neural network model.

364 of the random ranges were predicted to be active (Table A5-6, Fig. A5-7). A further 128 (60%) of the 213 vacant and unknown status ranges were predicted to be active. As in the neural network model, a large portion 74% (n = 268) of the random ranges, predicted to be active, were restricted to just six NHZ. Five of these were the same as those identified for the neural network model. The exception was that the Cairngorm Massif NHZ was replaced by the Western Seaboard NHZ. However, in this model the number of misclassified random ranges per NHZ was uncorrelated with the area of eagle habitat (land enclosed by 6 km Thiessen polygons drawn around all known range centres).

One of the main differences in the predictions from the decision tree model, compared with those from the neural network model, are the blocks of apparently suitable habitat in the south east and south west of Scotland. Apart from those differences the predictions are reasonably similar, including the prediction of coastal ranges for the Shetland Isles.

Table A5-8. Prediction accuracy of the Quest decision tree. Predictions are shown separately for each range class.

Natural Heritage zone	n	prediction active	 random	
Active				sensitivity
Western Isles	61	42	19	0.69
North West Seaboard	49	45	4	0.92
The Peatlands of Caithness and Sutherland	14	7	7	0.50
Western Seaboard	74	68	6	0.92
Northern Highlands	56	49	7	0.88
Western Highlands	57	56	1	0.98
Central Highlands	14	9	5	0.64
Cairngorm Massif	28	20	8	0.71
North East Glens	8	1	7	0.13
Lochaber	28	28	0	1.00
Argyll West and Islands	44	38	6	0.86
Breadalbane and East Argyll	23	19	4	0.83
Western Southern Uplands and Inner Solway	3	1	2	0.33
Border Hills	1	0	1	0.00
Random				specificity
Shetland	93	6	87	0.94
North Caithness and Orkney	103	1	102	0.99
Western Isles	80	2	78	0.98
North West Seaboard	87	45	42	0.48
The Peatlands of Caithness and Sutherland	237	3	234	0.99
Western Seaboard	52	15	37	0.71
Northern Highlands	136	33	103	0.76
Western Highlands	32	21	11	0.34
North East Coastal Plain	199	0	199	1.00
Central Highlands	96	9	87	0.91
Cairngorm Massif	101	18	83	0.82
North East Glens	189	2	187	0.99
Lochaber	70	40	30	0.43
Argyll West and Islands	200	59	141	0.71
Breadalbane and East Argyll	138	70	68	0.49
Eastern Lowlands	535	4	531	0.99
West Central Belt	324	3	321	0.99
Wigtown Machairs and Outer Solway	47	0	47	1.00
Western Southern Uplands and Inner Solway	406	13	393	0.97
Border Hills	253	20	233	0.92
Moray Firth	126	0	126	1.00

Natural Heritage zone	n	active	random	proportion predicted active
Unknown				
Western Isles	5	4	1	0.80
Western Seaboard	2	1	1	0.50
Western Highlands	3	3	0	1.00
Central Highlands	2	2	0	1.00
Cairngorm Massif	6	1	5	0.17
Vacant				
North Caithness and Orkney	1	1	0	1.00
Western Isles	16	12	4	0.75
North West Seaboard	19	16	3	0.84
The Peatlands of Caithness and Sutherland	17	2	15	0.12
Western Seaboard	20	15	5	0.75
Northern Highlands	36	27	9	0.75
Western Highlands	6	6	0	1.00
Central Highlands	10	6	4	0.60
Cairngorm Massif	33	16	17	0.48
North East Glens	11	2	9	0.18
Lochaber	9	6	3	0.67
Argyll West and Islands	11	6	5	0.55
Breadalbane and East Argyll	4	1	3	0.25
Western Southern Uplands and Inner Solway	1	1	0	1.00
Border Hills	1	0	1	0.00

It is again apparent that topographic features are the best predictor of range status. Only five predictors were included in the final Quest decision tree, the most important of which is AVESLPVAR (Fig. A5-8). This is the same predictor that had the highest relevance score in the neural network model. The first split separates out the majority of active ranges. Indeed a single trivial rule, using only this first split, produces a model that has 69% sensitivity and 92% specificity. The remaining 2799 cases (2687 random ranges and 112 active ranges) are first split on the area of montane habitat. Both of the subsequent child trees are mainly partitioned by topographic variables, in particular the slope.

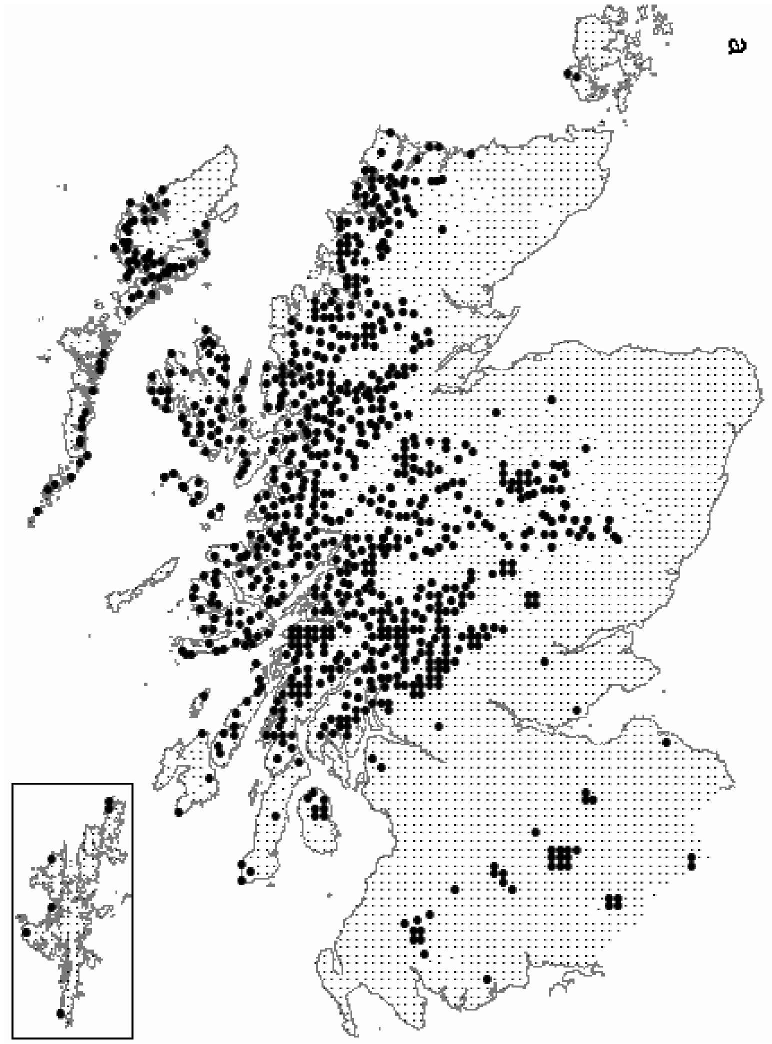


Fig. A5-7. (a) Predictions of range status using the Quest decision tree: active range (filled circle) and no range (dot). (b) Distribution of actual active (filled circle), vacant (open circle) and random ranges (dot) (available only in confidential version of Fielding et al., 2003a).

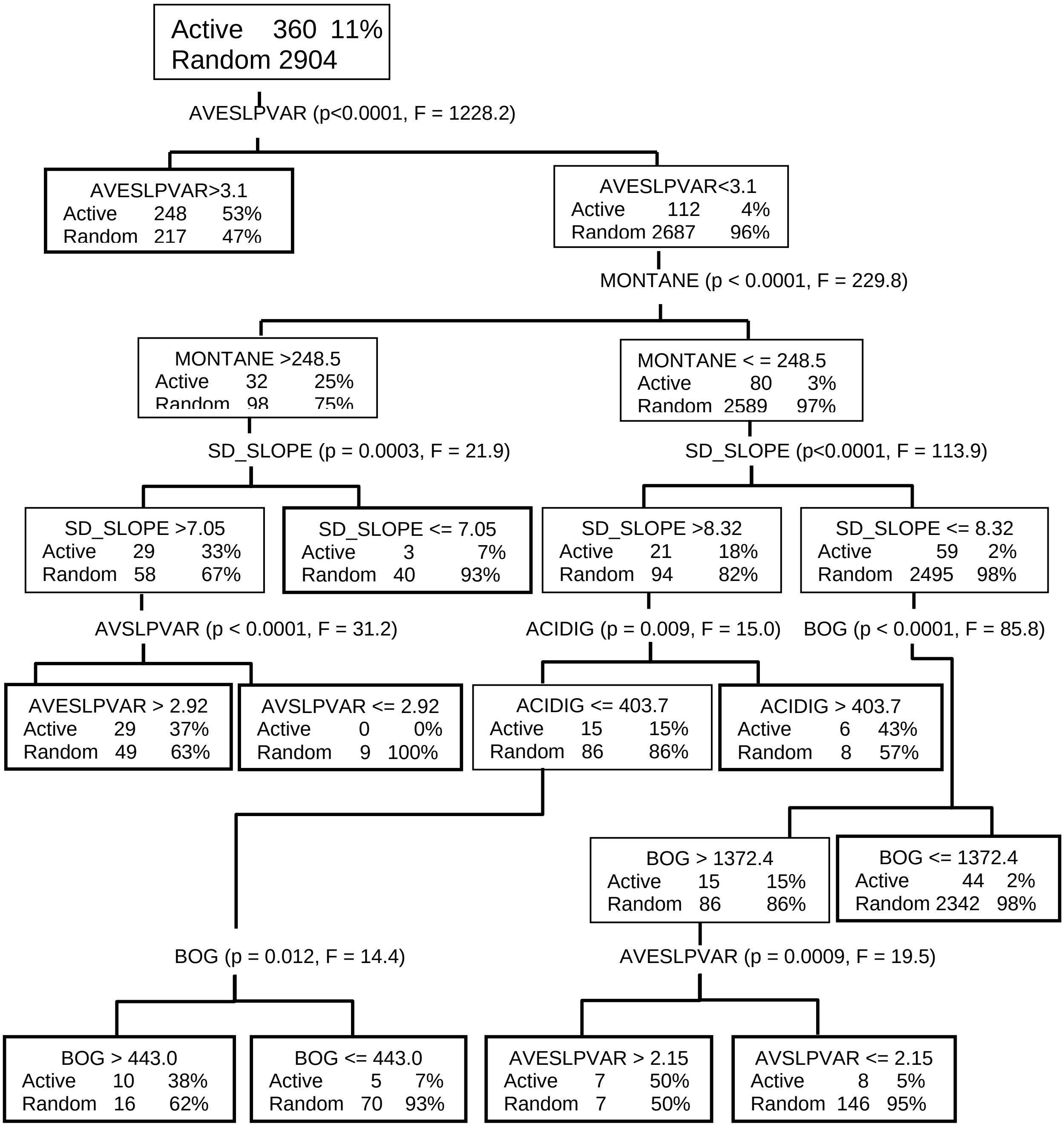


Fig. A5-8. The Quest decision tree for training data. Ranges are partitioned recursively using a threshold value of a particular predictor. The predictor and threshold are selected to maximize the 'purity' (high percentage of one class) of the resulting nodes. If a stopping rule is activated a node is not split. For example the parent node has 3264 cases (2904 random ranges and 360 active ranges). These cases are split into two groups depending on the value of AVESLPVAR. If this is >3.1 cases are assigned to the left node, otherwise they enter the right node. The left node cannot be split further but the right node is split on the area of montane habitat. Ranges with >248.5 ha of montane habitat are in the left node, those with <=248.5 ha are in the right node.

Comparison of the decision tree and neural network models

Both models produced similar predictions and identified similar predictors. Topographic predictors dominated both models, suggesting that active ranges have steeper and more variable slopes. While it is tempting to attach a biological explanation to these features, particularly relating to a link between broken terrain and foraging efficiency, it is important to remember that these important topographic features are also likely to constrain other land uses that may impact on possible usage by golden eagles.

We can have greater confidence in range predictions if they are repeated by both techniques. It is encouraging that, at least for active and random ranges, there is considerable overall agreement (Table A5-9). Even when results are broken down into NHZ classes only the Northeast Glens NHZ shows any major discrepancy (for active ranges).

Both models predict a large number of vacant and random ranges as active ranges. It is unlikely that all of these are inaccurate suggesting that the current Scottish population is considerably below its maximum, even in the absence of remedial management actions. Since we know that vacant ranges once supported resident eagles it perhaps wise to first concentrate on these ranges before examining possible ranges in ‘new’ locations. Identifying those vacant ranges which share a common prediction may provide important information about their future potential and current constraints.

It is interesting that most of the 65 historic vacant ranges, which both models predict as active, are geographically clustered (Table A5-10). Since the habitat, at least as represented by the current predictors, is apparently suitable for these ranges there must be other factors responsible for the current vacant status. In the absence of detailed range data it is difficult to provide explanations, although the clustering is suggestive of local influences which could include intentional and accidental disturbance. Indeed the closeness of Munros to many of these ranges is suggestive of accidental disturbance that is not a problem for most active ranges close to Munros. Detailed analyses of access routes with respect to nest locations and preferred topographic features may explain their particular susceptibility. Alternatively, estate-specific practices may also be responsible.

Table A5-9. Concordance (same prediction) between the predictions of the Quest decision tree and the artificial neural network (after applying a 0.5 threshold).

Range Status	Region		Concordance			
			agree	disagree	All	% agree
	National					
		active	401	59	460	87.2
		random	3250	254	3504	92.8
		unknown	12	6	18	66.7
		vacant	137	58	195	70.3
		All	3800	377	4177	91.0

Range Status	Region	Concordance agree	disagree	All	% agree
active	Natural Heritage Zone				
	Western Isles	45	16	61	73.8
	North West Seaboard	44	5	49	89.8
	The Peatlands of Caithness and Sutherland	12	2	14	85.7
	Western Seaboard	68	6	74	91.9
	Northern Highlands	50	6	56	89.3
	Western Highlands	53	4	57	93.0
	Central Highlands	11	3	14	78.6
	Cairngorm Massif	25	3	28	89.3
	North East Glens	3	5	8	37.5
	Lochaber	26	2	28	92.9
	Argyll West and Islands	39	5	44	88.6
	Breadalbane and East Argyll	22	1	23	95.7
	Western Southern Uplands and Inner Solway	2	1	3	66.7
	Border Hills	1	0	1	100.0
	All	401	59	460	87.2
random					
	Shetland	85	8	93	91.4
	North Caithness and Orkney	103	0	103	100.0
	Western Isles	78	2	80	97.5
	North West Seaboard	61	26	87	70.1
	The Peatlands of Caithness and Sutherland	229	8	237	96.6
	Western Seaboard	41	11	52	78.8
	Northern Highlands	114	22	136	83.8
	Western Highlands	24	8	32	75.0
	North East Coastal Plain	199	0	199	100.0
	Central Highlands	88	8	96	91.7
	Cairngorm Massif	90	11	101	89.1
	North East Glens	184	5	189	97.4
	Lochaber	50	20	70	71.4
	Argyll West and Islands	160	40	200	80.0
	Breadalbane and East Argyll	90	48	138	65.2
	Eastern Lowlands	531	4	535	99.3
	West Central Belt	319	5	324	98.5
	Wigtown Machairs and Outer Solway	47	0	47	100.0
	Western Southern Uplands and Inner Solway	396	10	406	97.5
	Border Hills	235	18	253	92.9
	Moray Firth	126	0	126	100.0
	All	3250	254	3504	92.8
unknown					
	Western Isles	3	2	5	60.0
	Western Seaboard	2	0	2	100.0
	Western Highlands	3	0	3	100.0
	Central Highlands	1	1	2	50.0
	Cairngorm Massif	3	3	6	50.0
	All	12	6	18	66.7

Perhaps more interesting are the historic vacant ranges that both models predict as non-ranges. It is apparent that most of these ranges are in the east, many being close to strip muirburn heather moor (Fielding *et al*., 2003a, confidential version), indicating a very powerful influence of strip muirburn (a surrogate for grouse moor management) on the predictive modelling. None of them are adjacent to Munros. In an attempt to understand the differences between vacant ranges that both models predict as either active or random, comparisons were made between the mean values for all predictors within 3 km of the 'range' centre (Table A5-11). Vacant ranges predicted as active had steeper, more variable slopes and tended to be higher. They also had more cliffs, montane and wet heath habitats. All of these features are consistent with their general position in the more rugged western mountain regions. Vacant ranges that are not predicted to be active had features associated with active land management. Thus they had more domestic grazing animals, more improved and smooth grassland and coniferous woodland. They also had more bog and burnt heather moor.

Table A5-10. Concordant predictions by the Quest and neural network models for vacant ranges.

		Prediction			Concordant predictions	
Natural Heritage Zone	All	agree	disagree	% agree	active	random
North Caithness and Orkney	1	0	1	0.0	0	0
Western Isles	16	13	3	81.3	9	4
North West Seaboard	19	13	6	68.4	10	3
The Peatlands of Caithness and Sutherland	17	14	3	82.4	0	14
Western Seaboard	20	14	6	70.0	9	5
Northern Highlands	36	26	10	72.2	18	8
Western Highlands	6	4	2	66.7	4	0
Central Highlands	10	6	4	60.0	2	4
Cairngorm Massif	33	22	11	66.7	9	13
North East Glens	11	9	2	81.8	0	9
Lochaber	9	5	4	55.6	2	3
Argyll West and Islands	11	7	4	63.6	2	5
Breadalbane and East Argyll	4	3	1	75.0	0	3
Western Southern Uplands and Inner Solway	1	0	1	0.0	0	0
Border Hills	1	1	0	100.0	0	1
All	195	137	58	70.3	65	72

Table A5-11. Comparison of mean values (3 km radius circle) for vacant ranges that both models predicted as either active (n = 65) or random (n = 72). Only those habitat features with a p value < 0.05 are shown. Variables are ranked by the F value.

Habitat feature	F	p	Mean	sd	Mean	sd	Largest mean
Slope sd	215.7	0.000	6.2	1.45	10.1	1.69	Active
Mean slope	105.5	0.000	8.3	2.89	14.5	4.05	Active
Max. slope	93.1	0.000	53.1	9.70	67.6	7.55	Active
Montane	41.8	0.000	67.8	139.85	407.0	419.70	Active
Sheep	31.5	0.000	2017.6	1319.62	1014.5	582.03	Random
Cattle	28.9	0.000	205.3	210.76	58.7	60.36	Random
Offtake	27.3	0.000	328599.4	277055.04	138272.5	95410.41	Random
Production	25.7	0.000	3983390.3	1142541.92	3054464.1	972849.30	Random
Altitude sd	23.3	0.000	85.6	37.79	125.7	58.04	Active
Burnt heath	19.5	0.000	405.0	591.10	63.5	194.05	Random
Improved grassland	18.8	0.000	89.8	134.84	14.7	33.82	Random
Grazing equity	16.8	0.000	3654791.0	1093960.16	2916191.6	995243.91	Random
Open forest	13.5	0.000	143.3	259.35	20.1	70.16	Random
Bog	13.4	0.000	702.9	658.90	360.7	377.36	Random
Max. altitude	12.7	0.001	553.4	217.08	723.1	332.02	Active
Closed canopy forest	11.6	0.001	184.2	266.01	58.3	136.83	Random
Range area	10.8	0.001	2732.8	343.96	2425.5	706.78	Random
Cliffs	9.7	0.002	4.7	21.62	51.5	125.26	Active
Smooth grassland	6.5	0.012	114.6	144.92	57.6	111.48	Random
Wet heath	6.3	0.013	141.2	444.21	377.1	643.05	Active
Misc. sedimentary rocks	5.2	0.024	85.2	288.71	2.9	16.13	Random

Conclusions

Both of the range prediction models suggest that there is considerable scope for future range expansion or re-occupancy of currently vacant ranges. The neural network predicted a further 330 active ranges (226 random ranges and 104 historic ranges whose current status is unknown or vacant) while the Quest decision tree predicted almost 500 additional ranges (364 random ranges and 128 historic ranges whose current status is unknown or vacant). An interesting difference between the two sets of predictions is the inclusion of blocks of predicted ranges in the Southern Uplands by the decision tree. These differences arose because the decision tree used a much reduced set of predictors. While it is unlikely that all of the identified ranges would support golden eagles it is reasonable to assume that a significant proportion is suitable. This raises the likelihood that population expansion is being restricted by a shortage of birds, which is confirmed by other forms of analysis (Whitfield *et al*., 2004a, b, 2007b).

A more conservative estimate of the number of vacant, suitable ranges is 65. These are the historic, but currently vacant, ranges that both models predict as suitable, i.e. predicted to be active. If all of these were filled it would represent a 15% population increase. Primarily this potential habitat is found in the more westerly and northern NHZ. Although some of these ranges may be constrained by intentional or unintentional disturbance (e.g. by people climbing Munros) there are others which, although apparently suitable and in the absence of apparent constraints, remain unoccupied. For example, range ML02, which is well known to the authors, has adequate prey, good nest sites and no disturbance, but is currently unoccupied.

If we accept that there are a significant number of suitable but unoccupied ranges then there must be a shortage of suitable young birds that are capable of occupying the potential ranges. Population modelling (Fielding *et al*., 2003a; Whitfield *et al*., 2004b) identified a wide range of conditions under which the population would not expand, particularly given the relatively low fledging rate over the last ten to twenty years (Annex 2). However, at observed fledging rates and with reasonable estimates of pre-breeding and adult survival some population expansion should be possible. That by 2003 such expansion had only occurred in some areas (with declines in other regions) was consistent with regional differences in survival estimates (Whitfield *et al*., 2004b) and with apparent changes in the intensity of persecution (Whitfield *et al*., 2007b). Consequently it seems that a large part of the explanation for an apparent shortage of young birds is related to the intentional disruption of breeding and killing of young birds that occurs in some areas (e.g. Whitfield *et al.*, 2004b).

ANNEX 6: SUMMARY OF THE STRUCTURE OF THE GOLDEN EAGLE POPULATION MODEL (GEPM)

The GEPM was used to simulate the response of the NHZ ‘populations’ to estimates of demographic parameters (O’Toole *et al*., 2002). Hence, the GEPM can also provide an assessment of the possible effects of changing demography on the future occupancy of golden eagle territories within NHZs. It incorporates four assumptions.

1. All parameter estimates apply equally to males and females.
2. All adult (>= 4 years old) birds are equally likely to occupy vacant home ranges.
3. Birds do not occupy ranges until they are at least four years old.
4. No regional trends are incorporated inherently.

An assumption of the GEPM as originally developed (O’Toole *et al*., 2002), that the number of occupied territories cannot rise above 110% of the initial value, was removed as we wished to have the option of examining conditions that could lead to occupation of vacant territories. Instead, the number of occupied territories was capped at the number of known territories in each NHZ. Capping was appropriate as it incorporated the concept of a limit to the availability of suitable habitat.

The GEPM is constructed in an Excel 2000 spreadsheet. This allows the model to be flexible and accessible. There are four user-determined parameters (default values used in all simulations are shown).

Home range count (HR)	Estimate of the number of occupied home ranges (territories) (default start point for each NHZ, see Whitfield *et al*. 2006).
Turnover rate (TR)	Proportion of home ranges that become vacant each year, or the annual adult mortality rate. TR^{-1} = average occupancy (adult life expectancy) in years (e.g. 0.05 is 20 years).
Fledging rate (FR)	Total number of fledged offspring divided by number of occupied territories.
Age-specific survivorship rates SR_i	These rates apply to birds not occupying territories.
SR_4	Proportion of fledged birds surviving to age 4.
SR_5	Proportion of year 4 birds surviving to age 5.
etc.	
SR_{12}	All birds not occupying a territory are 'killed' (hence default rate is 0.00); extending this has little effect on the model.

A fifth parameter used by O’Toole *et al*. (2002), harvest rate, created to estimate the impact of removing nestlings to donate to a re-introduction scheme was not employed. In all simulations it was assumed that adults (birds over 4 years old) not occupying territories had a similar annual survival rate to subadults (Hunt, 2002) but that the rate declined with increasing age (i.e. $SR_5 = 0.80$, $SR_6 = 0.80$, $SR_7 = 0.80$, $SR_8 = 0.80$, $SR_9 = 0.75$, $SR_{10} = 0.60$, $SR_{11} = 0.50$, and $SR_{12} = 0$).

The model tracks the fate of cohorts. Individual birds may die, or if they survive to four years old they may occupy a vacant territory. Range turnover applies to the range (= territory) rather than an individual. Therefore, when a range becomes vacant it is assumed that both previous birds are replaced by a new pair: this is a consequence of a cohort- rather than an individual-based model. Since the model works on the assumption that floating adults fill vacant ranges, then if number of vacant ranges > number of floaters there will be no floaters, and when the capped population limit is reached but survival rates and/or fledging rate produces an excess of adults then the number of occupied territories remains stable but the number of floaters increases.

The GEPM is a stochastic model in that user supplied parameters are perturbed by the addition of random ‘noise’ drawn from a normal distribution. The level of perturbation is under the user’s control, and may be set to 0 to produce a deterministic model. The default perturbation is obtained by setting the standard deviation to 10% of the mean for each demographic parameter (as in the present model simulations). Because the model follows national cohorts it does not discriminate between demographic and environmental stochasticity. The perturbation makes use of random data generators from Poptools (Hood, 2000).

Details of the structure of the model are given by O’Toole *et al*. (2002). GEPM is principally set up to take an initial value of HR (= number of occupied territories in 2003, in the case of analyses presented in this report) and model the effects of the other parameters on this value over time. The effect of varying the different parameters in the model is viewed by following the behaviour of the time course and by examining how HR has changed at the end of the time course. Because the model is stochastic it must be run many times for any particular combination of model parameters. The parameter values are not fixed for a single simulation; they vary between cohorts within a simulation. Hence, each parameter combination was replicated 100 times and simulations were run over 30 years. Examples of population trajectories are given in O’Toole *et al*. (2002). Results for specific parameter combinations are presented as mean values for HR for the years 21-30 taken from means of 100 replicates.