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Research Report

The National Gamebag Census as a Tool for Monitoring Mammal Abundance in the UK

A report to JNCC

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Summary

1. The GCT's National Gamebag Census (NGC) collates information from shooting and gamekeeping bag data across the UK since 1961. Historical data have been integrated with the more recent data, extending the time series back to 1900 for many species.

2. Such long runs of data on many predatory and game species make gamebag returns a valuable monitoring tool, which is envisaged by JNCC as a potentially important component of a planned network for the surveillance and monitoring of UK mammals. Independent assessments by Oxford University and BTO concluded that the NGC had a number of advantages and disadvantages for this purpose.

3. This project is a pilot study to assess the value of the NGC as part of a mammal surveillance scheme, particularly for mammals that are difficult to survey using other methods. The study considers NGC data for six mammal species: stoat, weasel, red fox, brown rat, rabbit and grey squirrel.

4. Between 1961 and 2001, the total number of estates contributing data to the NGC was 1602 for game species and 1222 for predatory species, with annual averages of 636 and 364 respectively. In comparison with other monitoring schemes, the sample sizes are large for species that are difficult to detect by conventional survey methods.

5. Geographical coverage of the NGC is wide. Data can be extracted at country, regional or county scales, or mapped using British National Grid references. Coverage was highest in south-east England (14% of estates), north-east England (11%), East Anglia (15%) and Scotland (25%); it was lowest in Northern Ireland (1%) and Wales (5%). By area, NGC estates cover 5% of the UK, varying regionally from 15% in eastern Scotland to under 1% in the Midlands and Northern Ireland.

6. The rate of data-entry error for bags is extremely low (0.2% for 1961-2001 data), and unlikely to affect trend analyses. The error rate for the number of gamekeepers was higher, at 3%. Overall, 16% of estates lacked grid references. For estates with grid references, 15% mapped outside their county boundaries, although at the country level the mismatch was under 1%. Such errors affect the potential for fine-scale measurement of distributional change.

7. NGC data from 1961-2001 were analysed using loglinear regression with year and site effects to take into account estate turnover. A generalised additive model was fitted to the resulting year coefficients to determine regional percentage changes in bags for each of the six species over the 25-year period 1975-2000, using bootstrapping to construct 95% confidence intervals for the estimated percentage changes.

8. Red fox bags increased significantly in 1975-2000 in seven out of ten NGC regions covering mainland Britain. Rabbit bags increased significantly in nine regions. Brown rat bags declined significantly in one region, and increased significantly in one other. Grey squirrel bags increased significantly in five out of the ten regions. Stoat bags declined significantly in three regions and increased significantly in two. Weasel bags declined significantly in six regions.

9. The effect of estate turnover was examined by comparing the results of loglinear regression incorporating year and site effects with annual means calculated from all estates contributing data each year. In general, regression indices fluctuated less than annual means, and in some cases, especially for red fox, showed a reduced magnitude in the long-term temporal change.

10. The statistical power of the NGC to detect temporal change was estimated for notional declines over 25 years of 10%, 25% and 50% in weasel bags from south-east England. The power to detect a 50% decline was over 98% from sample sizes of 40 estates or more. For a 25% decline, power exceeded 80% for sample sizes greater than 130. There was no power to detect a decline of 10% within the range of sample sizes considered (1-133). This compares favourably with the power of other mammal monitoring schemes.

11. Bag data are potentially as much a reflection of shooting or trapping effort as of species abundance. Gamekeeper density is one measure of effort that is part of the NGC dataset. Although gamekeeper density was strongly related to bag size for all species, the estimates of percentage change obtained with and without adjusting for gamekeeper density were similar across regions and across species.

12. For stoat bags, another proxy for effort is weasel bags because gamekeepers use the same traps for both species, leading to identical trapping effort. The inclusion of weasel bags in the analysis of stoat bags led to no detectable differences in the estimated regional percentage changes.

13. Given that the conclusions drawn concerning estimated changes in bag indices were the same with and without adjustment in both these cases, the lack of any explicit record of effort in the NGC may be less of a problem than hitherto believed.

14. Bag indices for red fox, rabbit and grey squirrel were compared with abundance indices available for 1995-2000 from the BTO's Breeding Bird Survey. There was a good level of concurrence, especially for grey squirrel. This gives some confidence that the NGC bag data for these species provide indices that track abundance.

15. Recommendations for improving the value of the NGC as a surveillance tool for mammals include data correction, recruitment of estates in under-represented areas, ongoing comparisons with Breeding Bird Survey indices, collection of some additional optional measure of effort, consideration of released gamebird numbers as a measure of effort, and small-scale intensive studies of the relationship between species abundance and bag density.

Table of contents Page

List of tables

List of figures

1 Introduction

1.1 The case for monitoring mammals

Management measures for conservation or population control rely upon accurate records of a species' distribution and abundance. Monitoring refers to the collection of data on various aspects of a species' ecology or biology; the specific parameters subject to monitoring will depend on its objectives. Monitoring changes in the size of animal populations is an important yet difficult problem for wildlife biologists and managers. In order to manage a population successfully for its conservation it is necessary to have information about its status, i.e. absolute abundance and distribution, at any point in time and trends in abundance over time. Complete censuses of populations are not feasible for virtually any species of animal, since every individual would have to be detected and counted in order to get a precise figure. For most species, the expense of mark-recapture or mark-resight estimation programs is prohibitive. The use of count data as indices of abundance and indicators of trends in abundance is therefore the only practical means of monitoring most animal populations, provided that the indices are related in a constant way to abundance over time.

Implicit in the concept of monitoring is an ongoing and regular effort to collect data allowing changes in the parameters of interest to be studied over time. In addition to establishing whether a population has been stable, increasing or decreasing over a given time-scale, long-term monitoring allows the *status quo* to be compared with historical distribution and abundance. This can provide further insight into current population status and act as a starting point for research into the causes of any observed decline in abundance or contraction of a species' geographical range. Repeated monitoring over time also has an important role in setting criteria and thresholds for action and determining when a downward trend constitutes a true decline.

Thresholds for defining conservation status are usually based on a critical abundance, e.g. number of breeding individuals, percentage change in abundance or range over a given time frame, or limited geographical distribution. Information collected through monitoring can thus be used in conjunction with predefined criteria to assign a species to a particular conservation category. The most widely used listing criteria are those adopted by the International Union for the Conservation of Nature and Natural Resources (IUCN, 2001), which provide a framework for the classification of species according to their extinction risk. However these listings are based solely on the threat of global extinction for each species and the IUCN is developing Red-list criteria for use at a national or regional scale. Red data lists exist for birds at a number of levels, ranging from global (e.g. Birdlife International, 2000) to national. The most recent red data list for UK birds was published in 2002 (Gregory *et al*., 2002). The criteria used to judge the UK conservation status of each bird species included global conservation status, recent population decline (both in numbers and geographical range), historical population decline, European conservation status, the rarity of breeding species, localised distribution and international importance. 'Recent decline' was taken to refer to the last 25 years. Species experiencing population declines of over 50% over this time period were assigned to the Red list, whilst those showing declines of 25-49.9% were assigned to the Amber list (Gregory *et al*., 2002). Similar criteria and listings are found in the UK Biodiversity Action Plan (BAP) (Department of Environment, 1994).

Monitoring wildlife populations is also a legal requirement for a number of species. This is stated explicitly in Article 7 of the 1992 Convention on Biological Diversity, which requires contracting parties to 'monitor through sampling and other techniques the components of biological diversity, paying particular attention to those requiring urgent conservation measures and those offering the potential for sustainable use'. As a signatory to the Convention, the UK published the UK Biodiversity Action Plan (BAP) in January 1995, incorporating separate Action Plans for priority species and habitats, recognizing the importance of the conservation of biological diversity and sustainable use of biological resources. Following its publication, a steering group was established to advise the government on implementation. Proposals were made for the monitoring of 1,252 long-list species or 'Species of Conservation Concern'. These were identified on the basis of the following criteria: globally threatened species, species for which the UK has less than 25% of the world population, species whose abundance or range has declined by more than 25% in the last 25 years, species found in less than 15 10x10-km squares in the UK and species currently listed under national or international legislation. Of these long-list species, Action Plans are being developed for 391 short-list 'Priority Species' identified as being either globally threatened or rapidly declining in the UK (by more than 50% in the last 25 years). Monitoring changes in the status of species is required as part of a long-term strategy for a Local Biodiversity Action Plan (LBAP).

Under the 1979 Bern Convention, ratified by the UK in 1982 (whose remit is to ensure the conservation of European wildlife and natural habitats by means of cooperation between States), signatories are required to adopt appropriate legislative and administrative measures to conserve wild fauna species listed in Appendix II (strictly protected animals) and regulate exploitation of wild fauna species listed in Appendix III (protected animals).

Under UK legislation, the Wildlife and Countryside Act (1981) pertains to the management of a number of animal species listed under schedules 5, 6 or 9. Under section 9 of the Act it is an offence to intentionally kill, injure or take a schedule 5 animal that is living wild at the time. In addition it is illegal to damage, destroy or obstruct access to the place or structure used by a schedule 5 mammal for protection or shelter. Section 11 restricts the use of certain methods of killing or taking animals regardless of their protected status. Use of self-locking snares, bows, crossbows and explosives is prohibited, although free-running snares are permissible. It also prohibits the use of traps, snares, nets, poisons, electrical devices, dazzling devices, automatic weapons, night-shooting devices, gas or smoke for killing, injuring or taking animals listed on Schedule 6.

1.2 The state of mammal monitoring in the UK

Despite the common aim of monitoring schemes of identifying population status changes and focusing finite resources on the most pressing conservation priorities, currently no structured national system exists in the UK for monitoring mammals and there is no unified protocol for data collection. Instead, a number of organisations operate independent mammal surveys, using a range of methods and often with a focus on a particular species or group of species. In an attempt to facilitate coordination between these organisations, The Joint Nature Conservation Committee (JNCC) has been seeking to develop a national programme for the surveillance and monitoring of UK mammals and launched the Tracking Mammals Partnership in May 2003. The Tracking Mammals Partnership is a collaborative project involving 23 organisations including The Game Conservancy Trust, with the goal of establishing a coordinated framework for the systematic monitoring of all British mammals. The Tracking Mammals Partnership aims to set up coordinated nationwide networks of volunteers to collect data on UK mammals, with a view to providing distribution and population trend information. This should be achieved through the use of standardised survey designs and methods to facilitate information exchange. The Partnership should also improve data availability between participating organisations as well as to the public. The publication of two scoping reports commissioned by the JNCC and DETR (Macdonald *et al.*, 1998, *Proposals for the future monitoring of British mammals* and Toms *et al*., 1999, *Developing a mammal monitoring programme for the UK*) to investigate how such a monitoring scheme might be set up culminated in a proposal issued jointly by DETR and JNCC in 1999 for a new national Mammal Monitoring Network (MaMoNet) to collect and analyse data on the life history of mammal species in addition to providing data on their abundance and distribution.

The main aims of the Tracking Mammals Partnership are:

- To develop a coordinated approach to the surveillance of all resident British mammal species.
- To raise awareness of species in need of more detailed monitoring and management.
- To raise awareness of the general public to the status of British mammals and the importance of surveillance and monitoring.
- To provide a cohesive framework for the various mammal organisations in the UK.

The main objectives of the Tracking Mammals Partnership are:

- To assess the distribution and abundance of all resident British mammal species to gain baseline data on populations.
- To assess trends (+/–) in the distribution and abundance of all resident British mammals to defined levels of precision.
- To provide standardised survey design to allow for comparison of data collected in different surveys.
- To co-ordinate collection of survey data to agreed data standards, with analysis and interpretation of the results to provide information on changes in abundance of UK mammals.
- To provide the ability to link the surveillance information obtained with that obtained from other surveys, e.g. habitat, climate, pollutants etc. This will provide an overall integrated monitoring network across the UK giving not only distribution, abundance and population trend information on British mammals but also links, at a UK level, to possible causal factors in changes in distribution and abundance.
- To report regularly on the changing status of UK mammals, via publications and the Internet, on behalf of the Partnership. Partnership reporting will concentrate upon giving summary population and trend statistics nationally or regionally, with suitable contextual information to aid understanding. Detailed reporting in relation to particular issues or problems will remain the responsibility of individual participating organisations.

1.3 The role of the National Gamebag Census in mammal monitoring

The Game Conservancy Trust (GCT) is a registered charity that researches the ecology of game and other associated species. In so doing it also studies their habitat needs, and factors limiting survival such as food supplies, predation and disease.

The objects of the Trust are:

- To promote for the public benefit the conservation and study of game species, their habitats and the other species associated with those habitats.
- To conduct research into the biology and ecology of game species and their environmental requirements and to publish the useful results of such research.

- To advance the education of the public in game biology and in the conservation of game (especially, but not exclusively, in the conservation of game as a sustainable resource).

The National Gamebag Census, hereafter NGC, was established by the GCT in 1961 to provide a central repository of records from shooting estates across the UK comprising numbers of game animals and predators killed annually - "bag data" (Tapper, 1992). Through the inclusion of data from historical game books, series for several species extend back to the early 20th century and for a few game species the early 19th century. Data on predatory species allowed by law to be controlled have been collected systematically since 1961. Gamebag records from shooting estates provide long-term data for a number of mammal species, from estates in England, Wales, Scotland and Northern Ireland. Because a substantial proportion of the countryside is managed for shooting, the data potentially reflect trends in national and regional abundance for many mammal species. Mammal species for which bag data are collated in the NGC include brown hare (*Lepus europaeus*), mountain hare (*L. timidus*), roe deer (*Capreolus capreolus*), red deer (*Cervus elaphus*), sika deer (*C. nippon*), fallow deer (*Dama dama*), muntjac deer (*Muntiacus reevesi*), Chinese water deer (*Hydropotes inermis*), rabbit (*Oryctolagus cuniculus*), hedgehog (*Erinaceus europaeus*), weasel (*Mustela nivalis*), stoat (*M. erminea*), polecat (*M. putorius*), mink (*M. vison*), red fox (*Vulpes vulpes*), grey squirrel (*Sciurus carolinensis*) and brown rat (*Rattus norvegicus*).

It is envisaged that the NGC will form an important part of the national programme for the surveillance and monitoring of UK mammals. GCT and JNCC seek to collaborate in developing and publicising the role and value of the NGC in the fields of mammal surveillance, conservation and ecology. This report explores the potential of the use of bag data from the NGC to these ends.

Project aims

- To assess the quality of the NGC mammal data in terms of their potential for surveillance and monitoring of UK mammals.
- To investigate the advantages and disadvantages (biases) associated with the NGC data.
- To make recommendations based on findings regarding the above as to how the NGC might be improved as a mammal monitoring tool.

Project objectives

1) To explore the coverage of the NGC. This will be addressed by determining the scale at which trend and range information can be obtained from NGC data for the six species concerned and the geographical coverage of the NGC.
2) To assess the representativeness of bag data from shooting estates in relation to national mammal population trends.
3) To investigate biases in the NGC arising from the method of data collection and whether these can be overcome.
4) To establish whether the turnover of estates within the scheme has a significant impact on population trend.
5) To determine whether changes in political, legislative and commercial considerations have affected sampling effort in a way that devalues the monitoring potential of the NGC.
6) To assess the magnitude of change that can be detected using trends obtained from NGC data, thereby demonstrating the alert potential of the scheme.

1.4 Survey design considerations

Spatial allocation of sampling units: coverage, randomisation and stratification

The spatial allocation of sampling units will ultimately depend on the objectives of monitoring and the constraints, budgetary or otherwise thereon. For example, a monitoring scheme designed to record the geographical spread of a species' range, of interest for species such as the otter *(Lutra lutra)* and pine marten *(Martes martes)*, will be different to one designed to detect trends in abundance within the current range of a species. The scale at which trends are detected is another important consideration; denser sampling would be required to detect trends on a regional as opposed to a national scale. Allocation of sampling units should be done so as to minimize bias since trends detected on the basis of a biased sample may lead to spurious conclusions being drawn about the species of interest. Bias can arise from a number of sources. One particular source is through unconscious selection of a sample that includes the best sites, which may occur when volunteers are asked to record the abundance of a species in their locality: they will tend to select areas where they know the species to be present. Site selection by habitat type or some other feature correlated with the animal of interest can also lead to bias.

The most widely used way to avoid bias is the by the random allocation of sampling units, whereby all potential sample sites should have an equal probability of selection. Random selection of sites is a desirable feature of any wildlife monitoring survey, since such monitoring on a national scale could be used to monitor geographical range expansion, and the location of sampling sites will not be associated with human population density or accessibility. Depending on the aims of the survey, it may be desirable to introduce some kind of stratification. If there is considerable variation between regions or habitat types that is likely to make them differ in terms of abundance of the target species or sampling effort, these can be treated as discrete strata. If the variation between strata is greater than that within, stratification increases the accuracy of population estimates. Sample units are usually allocated to strata in numbers proportional to the size of the stratum so that the sample is self-weighting. The BBS is based on a formal randomized sampling design with stratification by observer density, in which 1-km squares from the British National Grid are assigned randomly within BTO regions (Noble *et al*., 2001).

An alternative to random sampling is systematic sampling, whereby sampling units have a pre-defined spatial distribution, e.g. samples along transects or at intersections on grids. Systematic sampling is frequently used for studies designed to monitor temporal changes or trends as results from the same site are directly comparable between time intervals. Toms *et al*. (1999) identify three advantages of using the same sites in consecutive surveys. Firstly, doing so may avoid bias if there is a subjective element in the protocol for choosing sampling sites and this subjective element does not remain constant over time. Secondly, historical continuity improves the precision of estimates of change as it removes the component of error variance associated with site (variation between sites in average numbers per site), allowing the temporal rate of change in numbers averaged across sites to be estimated with increased precision. Thirdly, less time is spent seeking access permission prior to each survey. The NGC benefits from historical continuity, although there is turnover of estates over time, the effect of which will be addressed in this report.

NGC sampling units (estates) are not randomly allocated but are characterised by the presence of a shoot and the willingness of the shoot manager to contribute to the NGC, so biases may arise because land managed for shooting may differ in terms of habitat type to unmanaged land. Shooting estates are unlikely to be typical of the British countryside in general, as landowners participating in field sports such as fox hunting and gamebird shooting were found to maintain more established woodland and plant more new woodland and hedgerows than non-participants (Leader-Williams *et al.*, 2003). In the uplands, moors managed for grouse shooting retained considerably more heather than non-managed moors (Robertson *et al.*, 2001). Owing to these management practices, estates are also likely to be atypical with respect to the densities of game and non-game species they support, as they offer good-quality habitat in terms of food resources, shelter and breeding requirements. Predator control on shooting estates also reduces predator densities, which may affect prey species such as rabbits and small predators like weasels and stoats.

Sampling frequency and seasonality

In order to collect population trend information, surveys need to be carried out at regular intervals, over long periods of time to allow comparison of results across years. Sampling at distant time intervals may lead to short-term population dynamics being overlooked. Sampling at very close intervals will pick up seasonal fluctuations in the population; this may or may not be desirable depending on the purpose of the study. The frequency of regular sampling is likely to be determined by budgetary considerations. For the purpose of long-term monitoring requiring annual indices of population abundance, surveys might be carried once or twice each year. The seasonality of surveys may vary depending on the target species and times of maximum diurnal activity and visibility. The seasonality of monitoring should remain constant across all years as the abundance of many species will vary considerably within each year. Data collection by volunteers in the BBS involves an initial visit to set up transect routes and record habitat data, followed by two visits to the site during the bird breeding season (roughly April to June) when sightings and signs of mammals are recorded on two 1-km transects within the square. The Winter Mammal Monitoring Survey is similar in design to the BBS, the principal difference being the seasonal timing of the surveys.

Statistical power

Statistical power is a measure of the likelihood that a particular sampling protocol will be able to detect a trend of specified magnitude in monitoring data. If the statistical power of a proposed monitoring scheme is insufficient to give an acceptable likelihood of detecting a trend, then the resources invested in the implementation of the sampling protocol will be wasted, since little confidence can be placed in conclusions drawn from the data. Power analysis allows estimation of the probability of detecting upwards or downwards trends in abundance using regression techniques, based on a given sample size, estimate of sample variability, effect size and probability threshold α for rejection of the null hypothesis. By definition, a trend is detected when the regression has a slope that is significantly different from zero at the threshold α. The conclusion that a trend in abundance is occurring when in fact it is not, is termed a type I error, whilst a conclusion that no trend is occurring when in fact it is, constitutes a type II error. Power is defined as $1 – ß$, where ß denotes the probability of failing to reject the null hypothesis when it is false.

The statistical power of monitoring schemes is of foremost importance. Making inferences based on schemes with low power may result in erroneous acceptance of a null hypothesis of no change or no decline, resulting in failure to take appropriate action. Since monitoring schemes are resource intensive and thus costly, power analyses should be undertaken at an early stage in their design to identify an appropriate sample size and acceptable level of error to detect significant changes in population size. The effect size of interest is defined on biological grounds and according to the specific objectives of monitoring. A statistical power analysis may be carried out for a specific effect size. In terms of population change, 25% and 50% declines over a 25-year period were used as criteria to assign British bird species to Amber and Red lists (Gregory *et al.*, 2002) and would thus seem sensible effect sizes for power analyses.

1.5 Potential advantages and disadvantages of using bag data for monitoring

A specific set of biases are associated with mortality data methods, where data represent the number of animals killed rather than counted. These data collection biases are of particular concern, because a severe bias could obscure an underlying trend or create the appearance of a change in abundance when none has occurred. Identification of possible sources of error and bias in the National Gamebag Census is important in order to understand how, and with what caveats, changes in indices calculated from bags should be interpreted. It can also highlight ways in which current protocols might be improved upon in the future through reducing or controlling for known sources of bias.

A major assumption in the interpretation of mortality data is that the number of animals killed is a reliable index of the population density in a particular area. The number of animals killed is a function of both abundance and the amount of effort invested in harvesting; these factors are therefore confounded in bag data. On an annual basis there will be changes in the number of shooting days on an estate, number of participants per day, type of firearm used etc. that result in variations in effort over time. For predators, the number of gamekeepers on an estate, number of traps set, type of trap and duration of trapping will be included in the range of variables that underlie variations in effort.

Legislation, which can vary in its implementation on a regional basis, often initiates changes in shooting and culling practices. The variability in pest control methods over time resulting from either legislative stipulation or improvements in efficiency is problematic: changes in legislation have progressively outlawed a number of fox control methods over the last century primarily because they have been judged inhumane. Developments in predator control practice can also effect a change in the seasonality of culling. In recent decades, there has been a shift away from culling of foxes in the spring and summer using snares and terriers to autumn and winter culling by lamping with a rifle. This is expected to have an effect on fox bags since the culling period now coincides with an annual peak in fox density and dispersal before high natural mortality occurs over the winter (Reynolds, 2000). Legislation is also perceived to be an obstacle to the use of NGC data for those species afforded statutory protection, for example protection of the wildcat and partial protection of the hedgehog under the 1981 Wildlife and Countryside Act. This is not relevant to species in the current analysis but may affect the reliability of NGC data for those species concerned by causing a systematic decrease in the numbers recorded on predator sheets.

Macdonald *et al*. (1998) cite the lack of data on hunting or shooting effort in the NGC as a limitation to the interpretation of trends in bags since they may be a result of variations in the intensity of effort or in the proportion of the population susceptible to hunting as a result of methodological changes. They suggest that at the least a monotonic relationship should exist between effort and the number of animals killed. Thus it is not prerequisite that effort should remain constant over the period of monitoring but that the direction and magnitude of any changes are known. Toms *et al*. (1999) object to the use of mortality data on the grounds that shooting bags may not offer an accurate reflection of true population size, as effort (shooting days etc.) is not measured thus changes in bags cannot readily be ascribed to changes in abundance. In practice, this means that any apparent trend in a time series of bag data could reflect changes in sampling methods or intensity as well as changes in the abundance of the species concerned. A record of a measure of effort could resolve this problem. The NGC already records the number of gamekeepers on each shooting estate; this may, however, be insufficient since any changes in other measures of effort remain unknown. In addition, gamekeepers are unlikely to be wholly impartial with respect to their reporting of gamebags or the success of predator control operations. It is possible that there is an incentive for gamekeepers to exaggerate numbers of predators killed, introducing a systematic bias to the data. This might arise from a desire to avoid reporting consistently decreasing bags over time or to compensate for poor years, especially with regard to predator control, by reporting higher mortalities than have actually occurred.

Even if the direction and magnitude of changes in effort were known, the assumption that there is a linear relationship between yield and effort may not be applicable to some game species for which effort may be related to abundance. Non-linearity may prevail at low population densities if less effort is invested in shooting in years when low density is observed. This would lead to proportionally fewer individuals of a species being shot as it declines in number, and consequently a steeper decline in bags than in actual density. If hunting game species at low population densities becomes uneconomical, a nil count for a game species in one year could reflect a conscious decision to close a shoot rather than an empty bag. Hudson (1992) notes that grouse bags tend to under-emphasize the extent of any decline in grouse numbers, since the estates where grouse stocks fell to a level where grouse shooting ceased in any one year may not be included in the bag records. For predators on shooting estates, less effort may also be invested in control if abundance is perceived to be lower in a particular year. However, predator control tends to be prophylactic, i.e. independent of predator density in a particular year. At high population densities trap saturation may be possible, especially for single-catch traps. This is thought to be unlikely given daily checking of traps by keepers and typical trapping rates (J. C. Reynolds, personal communication).

Hunting can itself be the cause of changes in the abundance of a species; this is problematic since the method of data collection has a direct impact on the quantity it intends to measure. It is impossible to assess the implications of this for population dynamics without some *a priori* knowledge of population size and demography. The proportion of a population removed by hunting may be considerable; Hutchings and Harris (1996) estimated that around 60% of a local brown hare population may be removed by February culls. Hunting can also alter the age and sex structure of a population; this is particularly well documented in the fisheries literature, where gear selectivity has resulted in lower mean size and mean age of sexual maturity (Cook *et al.*, 1997, O'Brien *et al.*, 2000). However it is unlikely that such extreme effects would be observed in mammalian species in the UK, as hunting pressure is generally lower than that found in commercial fisheries. It is possible that hunting could affect the sex structure of mammal populations. For example, the catchability of mustelids using tunnel traps is dependent on their mass, which varies according to gender: females are the lighter sex and may not spring a trap unless it is lightly set (Game Conservancy Ltd, 1994).

For some pest species it is likely that the numbers killed over a year cannot be recorded with a high level of accuracy because mortality is unobserved and hence unquantifiable. In such cases numbers recorded on NGC forms are usually 'best guesses'. This applies to rats and squirrels, for which poisoning is legal, and to rabbits, where the use of cyanide gas is legal and commonly injected into burrows as a control measure. This could have the effect of increasing the variance of the bags and detract from the utility of NGC data as a surveillance tool for these species.

2 Methods

2.1 The National Gamebag Census

The GCT operates the census via postal questionnaires, mailed to members on an annual basis. Each estate contributing data has a unique identifier, based on its region and county within that region, with location recorded in six-digit British National Grid format. There are eleven NGC regions within the UK (south-west England, south-east England, East Anglia, east Midlands, west Midlands, Wales, north-east England, north-west England, east Scotland, west Scotland and northern Ireland), with boundaries given in the map in Figure 2.1.1. Information obtained from each estate includes shoot area, area of woodland and moorland, local ITE Land Classification (each 1-km square in Britain is allocated to one of 32 environmental land classes based on climate, topography, geology and other structural variables - Bunce *et al.*, 1996), number of gamekeepers employed and the number of each species killed. Quarry species and predatory species are recorded on separate parts of the form, and it is in these categories that mammalian species are recorded. The basic unit of the NGC is the annual bag density, or number of animals of a particular species killed per square kilometre, per year, per estate. This takes into account the total number of animals of that species killed on the estate during the 12-month period ending on 1 February (the end of the shooting season). The use of bag density gives equal weight to all estates, regardless of size, and provides an index of abundance that can be compared between areas and years. The number of gamekeepers reported in the NGC is the sum of the number of full-time employees plus half the number of part-time keepers.

The current analysis focuses on a subset of six species recorded by the NGC: one game species, rabbit, and five pest or predatory species, weasel, stoat, red fox, grey squirrel, and brown rat. Rabbits were chosen because there are data from other monitoring schemes (e.g. the BBS) with which results from analysis of NGC bag data may be compared. There have also been a number of studies on changes in the distribution of rabbits in the UK during the latter half of the 20th century as the population recovered from the myxomatosis epidemic (Lever, 1977; Trout *et al.*, 1986; Trout *et al.*, 2000). Grey squirrels have progressively expanded their distribution within the UK over the last 50 years and for this reason this species is also relatively well monitored by other schemes (Gurnell, 1991; Lloyd, 1983). Red foxes are of ecological importance, being the largest native terrestrial mammalian predator in the UK. Stoats and weasels are poorly monitored by other schemes as are brown rats, for which little data on UK abundance are available.

Extraction of data from the NGC

Bag data were extracted from the NGC for the six mammal species rabbit, weasel, stoat, red fox, grey squirrel and brown rat for all estates and over the complete time-span of the NGC. Data on estate location by region and county, estate area including areas of moorland and woodland and the number of gamekeepers employed were also extracted.

2.2 Assessment of NGC coverage

The numbers of estates contributing records for game and predatory species were determined at the following levels in increasing order of scale: county, NGC region, country and UK. The locations of estates contributing NGC records for rabbit and predator data were given by the British National Grid reference held in the database. They were overlaid on county, regional, country and UK boundaries using the Geographical Information System (GIS) MapInfo Professional version 7.0.

Figure 2.1.1: Map of the UK showing its subdivision into eleven NGC regions.

Data for rabbits and predatory mammals were imported into the GIS with their year of collection, so that the distribution of estates where any of the species was present in a particular year could be mapped.

2.3 Quantification of data-entry error

Two randomly selected sub-samples of the electronic data from the NGC, representing 1% of the extracted datasets for rabbits and predatory species respectively, were checked against hard copies of game census returns in order to quantify data-entry error. Two types of error were identified: an incorrect entry of bag size for one of the six species or an incorrect record of the number of gamekeepers on the estate. The level of error found in the sample was assumed be representative of that in the whole dataset. The number of estates for which a grid reference was lacking in the database was quantified on a regional basis. The remaining estates were mapped and the locations compared with the region and country attributions that were part of their identifiers to assess the number of anomalous grid references.

2.4 Log-linear Poisson regression with site and year effects

Preparation of the data for trend analyses

Prior to the application of any statistical analyses, the data were systematically checked in order to remove duplicate records and identify anomalies and obvious outliers. Estates contributing only one year's data and those reporting zero or blank records for their entire time series were omitted on the grounds of being uninformative.

Exploring the distribution of the data

The data for the five predatory species were separated into spreadsheets for each individual species, and the data were imported into GenStat^{TM} version 6.0 for Windows. The distribution of the bag data was explored in GenStat to determine whether the data conformed to a Poisson distribution with a mean equal to the variance or whether there was under- or overdispersion. The Poisson distribution arises as counts of animals that are distributed randomly and independently in time or space. The gamebag data were best approximated by an overdispersed Poisson distribution, i.e. the variance was greater than the mean.

Modelling site and year effects

The methods currently in widespread use for the analysis of count data can be classified as forms of Generalised Linear Model (GLM) (McCullagh and Nelder, 1989). These models are flexible, allowing regression and ANOVA-type models to be fitted, while taking into account the error distribution and data transformation (link function) required for linearity. The appropriate link function for bag (count) data is the logarithmic link used with a Poisson error distribution (McCullagh and Nelder, 1989). By including the logarithm of estate area as an offset in the model, the dependent variable effectively became the ratio of bag to area, i.e. bag density. Because of the overdispersion, statistical inference was based on F-statistics obtained as the ratios of deviances from nested models (McCullagh and Nelder, 1989).

Bag data from 1961 to 2001 were analysed for all species using a generalised linear model with a Poisson error distribution and logarithmic link function, with site (estate and region) and year effects declared as factors and the logarithm of estate area as an offset variable. Inclusion of estate effects allows for variations in bag size between estates whereas the year effect allows for variations in bag size over time. Declaring year as a factor rather than a variate gave a coefficient for each year, rather than a single coefficient for the linear effect of year. This is important since it is unreasonable to assume *a priori* that a trend is linear. The maximal model included the interaction term between region and year to test for the presence of differing temporal trends among regions. The effect of region corresponded to the average effect of estates within each region, after adjustment for year effects. This was compared with the output from an additive model containing region and year without the interaction term, and an F-test was used to assess the significance of the interaction. Having determined a significant interaction between region and year, all further analyses proceeded with discrete regional data sets restricted to data from 1961 to 2001. The coefficients for the factor year were saved, then exponentiated to give an index of bag size on the arithmetic scale. As the coefficients were usually estimated relative to 1961, all index values were relative to 1961, which had a value of 1. Where the reference year was not 1961, they were transformed to give an index of bag size relative to 1961 by subtracting the 1961 value from the log-scale estimates and then exponentiating. This allowed comparisons to be made between inter-annual patterns in abundance in different regions.

Investigation of the effect of estate turnover

For the purpose of assessing the effect of estate turnover on trend estimates, mean annual bags (calculated as the total bag divided by the number of estates contributing in that year) were calculated for each region for each year from 1961 to 2001 for all six species. These were then expressed as an index relative to the mean bag in 1961, whose value became 1, to enable a direct comparison with year coefficients from the log-linear regression with site and year effects. The annual mean number of gamekeepers per square kilometre in each region was also calculated. Annual indices from the log-linear regression are adjusted for the effect of turnover in the factor estate, whereas the annual mean bags do not take turnover into account. Annual mean bag indices and indices from the log-linear model were plotted on the same axes for regions 1-10 for all six species in order to determine the size of the effect of estate turnover.

2.5 Trend analysis

In order to determine percentage changes in abundance for a species between two points in time, it is first necessary to describe the observed temporal pattern by fitting a model to the data that describes the observed abundance as a function of time. This model could vary between two extremes. At one extreme, log counts could be treated as a linear function of time designed to estimate some average rate of population change over the whole period. However, the resulting average trend may mask fluctuations in the true population size if its trend is non-linear. At the other extreme, unconstrained annual abundance indices could be used. This would show every year-to-year fluctuation in population size, often obscuring the underlying long-term trend, so smoothing algorithms are usually applied by fitting a secondary model. Essentially, it is necessary to attempt to distinguish the genuine population trend from random noise.

Generalised additive models (GAMs) have been advocated as a suitable modeling framework for the analysis of population trends (Fewster *et al.*, 2000). They offer a more parsimonious approach to such analyses since smoothing procedures are incorporated into the model-fitting process itself, removing the need for a second smoothing model. The degree of smoothness of the curve is defined by its degrees of freedom (*df*), which can range between 1, yielding a linear curve, to n - 1, where n is the number of years in a time series, to give the annual estimates joined by straight lines. The choice of the *df* value for a given analysis depends on its purpose and the level of temporal resolution at which trends are studied. Inference of long-term trends requires a smooth line and corresponding low *df* whereas analysis of annual fluctuations would require the maximal number of *df*. The length of the time series is also pertinent to the choice of *df*, and a higher value may be required for longer time series to maintain flexibility in the trend curve. As a rule of thumb, Fewster *et al.* (2000) suggest a figure of 0.3 times the length of the time series. A GAM was run in GenStat taking year coefficients from regional analyses with site and year effects as the response variable and year as the explanatory variable. Graphs of fitted and observed values were plotted against year for a series of GAM *df* values (which determine the smoothness of the trend line) for the purpose of selecting an appropriate number of *df*. The optimal number of *df* was judged to occur at the point where further increases in *df* roughened the curve and did not modify the overall shape. A *df* value of 4 was found to be most appropriate and so a GAM with 4 degrees of freedom was used for all ten regions and six species (i.e. a total of 60 regression analyses were carried out). The fitted values from each regression were saved and used to determine the percentage change in abundance between 1961 and 2001 using the formula:

$$[(a_{2001} - a_{1961})/a_{1961}] \times 100$$

where a_{2001} is the fitted value for 2001 and a_{1961} is the fitted value for 1961. This was repeated for a 25-year period using 2000 and 1975 as the reference years in order to compare percentage declines with established Red and Amber data list criteria.

Estimation of confidence intervals for changes in bag indices

A bootstrapping method (sampling with replacement) was used to estimate the confidence interval associated with the estimated trend as well as the statistical power of detecting a trend of a certain magnitude from the NGC data. Monte Carlo confidence intervals are appropriate for estimating the standard error of a parameter that has complicated analytical properties (Buckland, 1984).

Confidence intervals for the percentage change in regional bags between 1975 and 2000 were estimated for all species using the bootstrap sampling method as follows: for 399 replicates, a random sample of sites with size equal to the actual number of estates in that region was selected with replacement using the full run of data from 1961 to 2001. Thus although the number of sites in each run was equal to the regional figure, within each sample each estate could be sampled once, more than once or not at all. A log-linear Poisson model with site and year effects and the natural logarithm of estate area as an offset variable was fitted to each sample, and used to derive annual indices. A GAM was fitted to the resulting year estimates, then percentage changes in bags were calculated from the 1975 and 2000 indices. After the original estimate and B bootstrapped replicates, there are B + 1 estimates for each of the 1975 and 2000 indices, thus B + 1 estimates of the percentage change between these years. These estimates were sorted in ascending order and the 100 $(1 - 2\alpha)$ % confidence intervals taken from the values with rank l and u, where $l = (B + 1)\,\alpha$ and $u = (B + 1)(1 - \alpha)$,afterBuckland (1984). The randomisations were carried out B = 399 times as this gives integer rank values for the upper and lower percentiles used to construct the confidence interval. Confidence intervals calculated in this way provide approximate hypothesis tests for significant population change. The hypothesis that the percentage population change derives from a distribution with zero mean is rejected at the 5% level if the 95% confidence interval for the change does not contain the value zero (Buckland 1984).

2.6 Investigating the alert potential of NGC data

Since the observed percentage changes in bags over a 25-year time period differed from the percentage changes defined by standard Amber and Red list criteria (i.e. 25% and 50% declines), a transformation was applied to the data to scale the observed trend to the percentage change of interest using the following formula:

$k_x = \ln[(1 - x/100)/(a_{2000}/a_{1975})]$

$b'_y = \exp[(y - 1975)/25*k_x + \ln(b_y)]$

where: x is the percentage decline of interest
a is the fitted value for abundance from the GAM
k_x is a constant whose value is determined by x, a_{1975} and a_{2000}
y is the year
b_y is the bag in year y on a given estate
b'_y is the transformed value for b_y

The GAM was applied to the transformed data for weasels in NGC regions 1, 2 and 3 to determine whether it yielded the desired percentage change over the 25 years and to establish whether or not the transformation affected the underlying distribution of the data using the estimated dispersion parameter as an indicator. The dispersion parameter was found to be unaffected by the transformation. Confidence intervals were estimated for weasel bag data in region 2 using the following values for the percentage declines (x): 10%, 25%, 50%. These were compared with confidence intervals constructed using the standard error of the difference for the fitted values from the GAM in 1975 and 2000 in order to ascertain whether randomisation carried out to calculate confidence intervals was necessary. If confidence intervals for the decline obtained by moving 1.96 standard errors away from the difference between 1975 and 2000 values are found to approximate those from the bootstrap method, significant savings can be made in computing time because the randomisation step used for estimation of confidence intervals can be omitted.

For each power analysis, confidence intervals for declines of 10%, 25% and 50% in the bag indices between 1975 and 2000 were estimated 100 times, and the statistical power calculated as the percentage of runs out of the total yielding a significant 95% confidence interval for the decline. In order to plot a power curve showing the effect of sample size on power for weasel data for region 2 (n=133), the analysis was repeated for the following sample sizes (number of estates): n=20, 40, 60, 80, 100, 120.

2.7 Investigating the effects of variations in sampling effort

Analysis of the effect of the number of gamekeepers

Log-linear Poisson regression was carried out on regional data for each species, incorporating site and year effects, the logarithm of estate area as an offset, and gamekeeper density (number of gamekeepers per square kilometre) as an additional explanatory variate. The significance of the additional variate was tested using an F-test of the difference in the regression deviance compared to the model without gamekeeper density.

Correlation analysis of stoat, weasel and rabbit bags

Since stoats and weasels are caught by the same means (tunnel trapping), it is reasonable to make the assumption that the same level of effort is applicable to both. This assumption would hold true providing that for both species catchability varied linearly with the amount of effort. Using weasel bags as a proxy for effort, it is possible to assess what proportion of the variation in stoat bags is attributable to variations in effort. If trends discernible from the NGC data were purely an artefact of fluctuations in trapping effort, one might expect stoat and weasel bags to covary in a given region over a specified time period, with a significant positive correlation between bag indices for the two species. However, if the bag indices are significantly negatively correlated, one might assert that changes in bags are indicative of underlying changes in abundance.

A second variable that might be expected to explain a large proportion of variation in stoat bags is rabbit abundance, represented by rabbit bags. Fluctuations in the rabbit population may have affected stoat abundance (and stoat bags), because rabbits constitute the major part of stoat diet: McDonald *et al*. (2000) found that lagomorphs constituted 65% of stoat diet by frequency of occurrence. Thus changes in stoat bags might be expected to be positively correlated with changes in rabbit bags over the years if changes in stoat bags are indicative of changes in stoat abundance.

Correlation analysis was applied to year coefficients from a log-linear Poisson regression including site and year effects, and the logarithm of estate area as an offset; stoat indices formed the response variable and weasel and rabbit indices the explanatory variables. This was done for stoats with either rabbits or weasels and stoats with rabbits and weasels to give correlation coefficients and partial correlation coefficients respectively.

2.8 Assessing the representativeness of NGC bag size indices as indices of abundance: comparisons with indices from other mammal monitoring schemes

The first six years of mammal data collected by BBS observers have now been analysed in order to assess the effectiveness of the BBS in monitoring UK mammal populations (Newson and Noble, 2003). Indices of relative abundance from the BBS are available for the years 1995-2000 for red fox, rabbit and grey squirrel. These indices were calculated using log-linear Poisson regression with site and year effects (Newson and Noble, 2003). Since an equivalent model has been applied to the NGC data, it is possible to compare the indices calculated from both regressions. In order to extract comparable annual indices from the NGC, the complete UK bag data for red fox, rabbit and grey squirrel were restricted to the years 1995-2000, and log-linear Poisson regression with site and year effects was used to calculate bag indices. The standard error for the difference between two year coefficients on a logarithmic scale were calculated as follows:

$$SE_{\mathrm{Diff}}=(SE_i^2 + SE_j^2)^{0.5}$$

where SE_i and SE_j correspond to the standard errors of the coefficients for years *i* and *j*. The standard errors for the year coefficients were used to construct confidence intervals on a logarithmic scale; these were then exponentiated to give asymmetric confidence intervals comparable to those for the BTO indices.

Data are available from various sources on changes in the distribution and abundance of foxes killed in Scotland since 1948 (Hewson and Kolb, 1973; Hewson, 1984). The 1954 Pests Act provided subsidy for the destruction of foxes in the form of bounties paid by Agriculture Executive Committees, and grants to Fox Clubs employing trappers or operating bonus schemes up to 50 % of their expenditure. Fox control was also undertaken by the Forestry Commission, whereby foxes were snared by Commission trappers (42%), shot (25%) or caught in gin traps (9%). The use of gin traps by the Forestry Commission ceased in 1958 when it became illegal in England and Wales. The North and West Forestry Commission Conservancies correspond roughly to the Game Conservancy's west Scotland region, whilst the South and East Conservancies correspond to the east Scotland region. Data on the number of foxes killed by the Forestry Commission are available for each Conservancy from 1961 until 1978 (Hewson, 1984).

3 Results

3.1 Assessment of NGC coverage

There is a greater availability of records for game species than predator species, reflected in lower annual mean receipt of predator sheets than of game sheets (Table 3.1.1). Between 1961 and 2001, the annual mean total of estates contributing bag records was 636 for game species and 364 for predatory species. On a regional basis, South-east England, East Anglia, North-east England, East Scotland and West Scotland have the highest numbers of estates contributing bag data to the NGC, whereas coverage is markedly lower in Northern Ireland. At the country level, England has the highest annual mean contribution of records, followed by Scotland, Wales and Northern Ireland.

Table 3.1.1: Overall number and average annual number of estates contributing records for game and predator sheets, by country and NGC region. Figures are for the period 1961 to 2001.

NGC region	Game sheets			Predator sheets		
	Overall number	Annual average	% in region	Overall number	Annual average	% in region
England	1096	410	68.4	855	264	70.0
South-west England	105	42	6.5	92	29	7.5
South-east England	235	84	14.7	175	59	14.3
East Anglia	237	95	14.8	189	63	15.5
East Midlands	134	39	8.4	113	31	9.2
West Midlands	128	38	8.0	98	27	8.0
North-west England	72	33	4.5	55	17	4.5
North-east England	185	79	11.5	133	39	10.9
Wales	83	25	5.2	50	9	4.1
Scotland	408	197	25.4	302	87	24.7
East Scotland	231	113	14.4	179	52	14.6
West Scotland	177	84	11.0	123	35	10.1
Northern Ireland	16	5	1.0	15	4	1.2
UK total	1603	636	100.0	1222	364	100.0

In terms of individual species, the average number of UK estates contributing bag data between 1961 and 2001 varied from 1294 for rabbit to 727 for grey squirrel (Table 3.1.2). The values are lower than the ones in Table 3.1.1 because not all species are caught on each estate. The regional pattern is similar to the one described for the dataset as a whole (Table 3.1.1), with a few differences linked to species distribution (low sample sizes for grey squirrel in Scotland). NGC species-specific coverage is lowest in Northern Ireland and in Wales.

Mean estate size ranges from 7 square kilometres in the west Midlands to 63 square kilometres in west Scotland (Table 3.1.3). There is a trend towards a larger mean estate area moving northwards through the UK. The proportion of the total land area in the UK for which bags are recorded in the NGC is considerable, averaging 5%. In terms of the area covered by contributing estates, the NGC has greatest coverage in the Scottish regions and North-east England, reflecting the large average size of estates in these regions. Coverage is also fairly high in East Anglia although mean estate size in this region is similar to that in other regions of the south of England.

Table 3.1.2: Number of estates contributing bag records for red fox, rabbit, brown rat, grey squirrel, stoat and weasel, by country and NGC region, during the period 1961 to 2001.

NGC region	Rabbit	Weasel	Stoat	Fox	Squirrel	Rat
England	908	641	659	576	624	580
South-west England	90	58	62	73	82	59
South-east England	196	133	135	119	137	124
East Anglia	207	147	153	126	127	132
East Midlands	121	89	88	65	93	76
West Midlands	99	71	73	58	77	65
North-west England	54	39	40	39	28	33
North-east England	141	104	108	96	80	91
Wales	58	27	28	32	31	25
Scotland	314	203	211	247	65	143
East Scotland	182	130	135	151	52	91
West Scotland	132	73	76	96	13	52
Northern Ireland	14	0	11	13	7	9
UK total	1294	871	909	868	727	757

Table 3.1.3: Number of estates submitting predator sheets to the NGC in 2000, by country and NGC region, with mean estate area, total regional area and percentage regional coverage. The number of game sheets submitted in 2000 was similar (432 in total).

NGC region	N	Mean estate area (km^2)	Total estate area (km^2)	Regional area (km^2)	Regional coverage (%)
England	234	14	3,361	130,379	2.6
South-west England	31	8	240	17,716	1.4
South-east England	35	10	334	19,677	1.7
East Anglia	55	12	668	22,168	3.0
East Midlands	17	8	136	16,365	0.8
West Midlands	17	7	111	13,689	0.8
North-west England	21	16	331	16,808	2.0
North-east England	58	27	1,541	23,956	6.4
Wales	15	22	331	20,759	1.6
Scotland	162	52	8,381	77,992	10.7
East Scotland	95	44	4,154	26,720	15.5
West Scotland	67	63	4,227	51,272	8.2
Northern Ireland	6	9	54	14,144	0.4
UK total	417	29	12,127	243,274	5.0

Location of estates contributing game and predator sheets

As can be seen from the maps of estate location for the game (Figure 3.1.1a) and predatory species (Figure 3.1.1b), there is good coverage throughout the UK with the exception of Northern Ireland.

Figure 3.1.1a: The distribution of estates contributing game sheets to the NGC in 1961-2001, excluding estates with no grid reference and ones mapping outside the boundary of the UK.

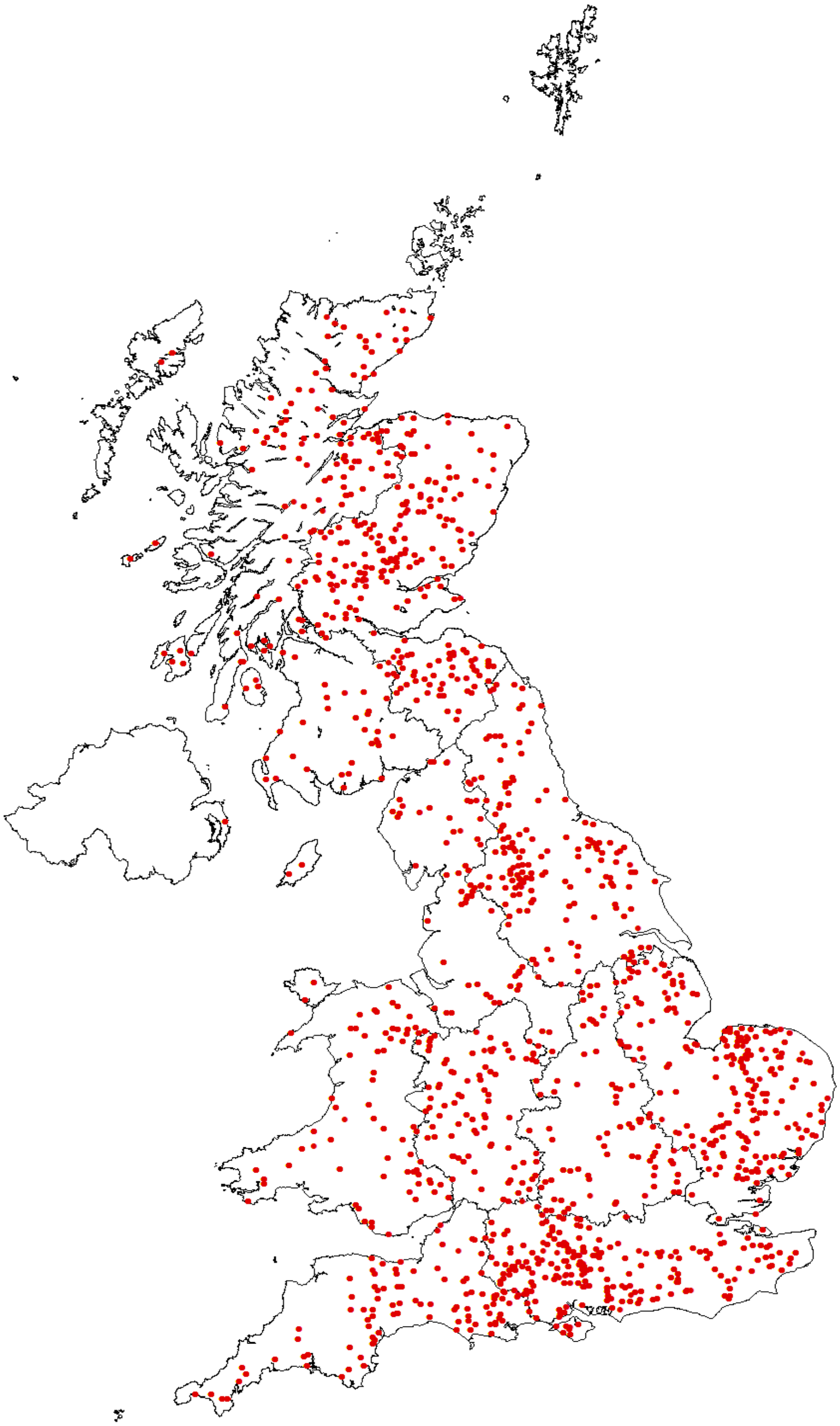
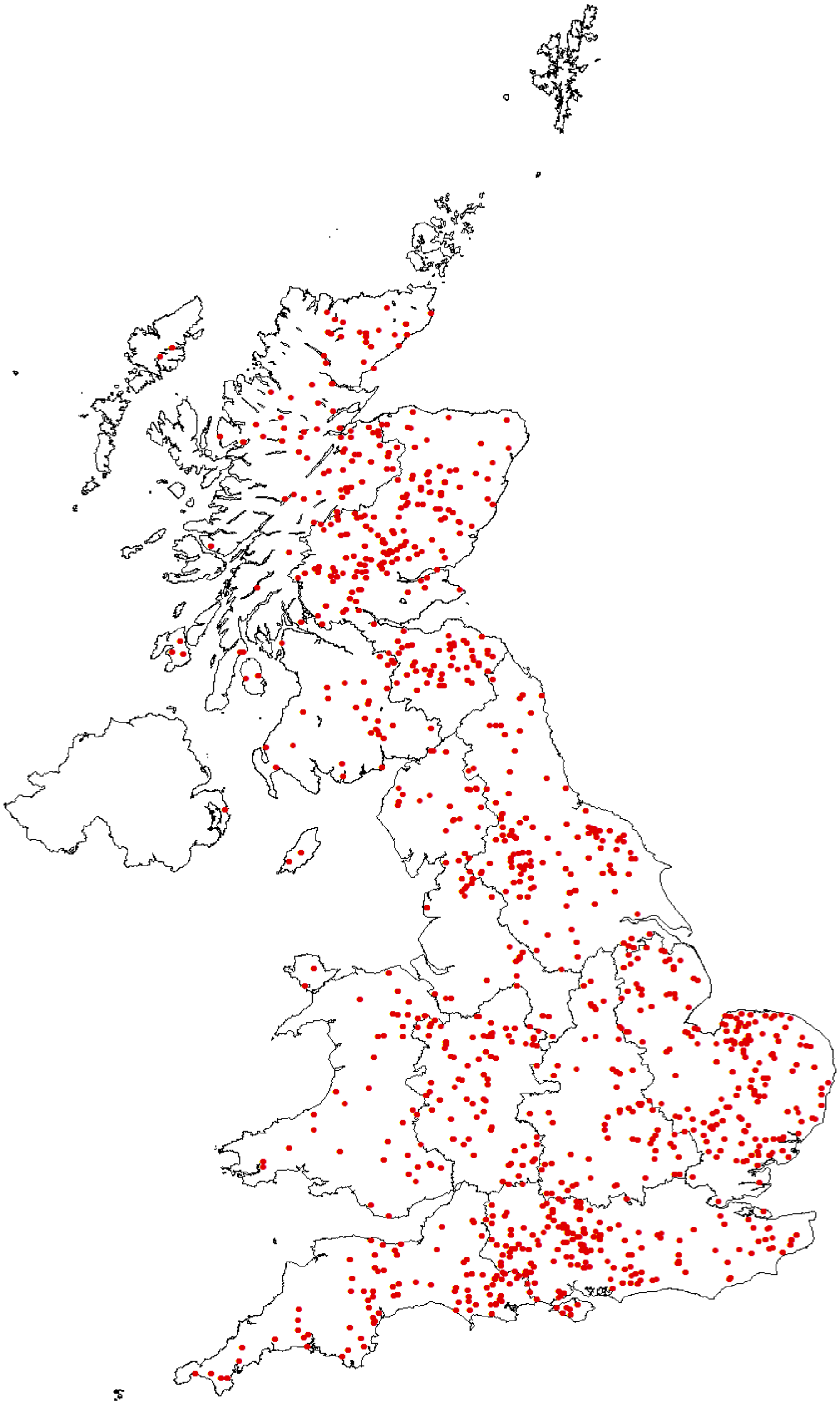


Figure 3.1.1b: The distribution of estates contributing predator sheets to the NGC in 1961-2001, excluding estates with no grid reference and ones mapping outside the boundary of the UK.

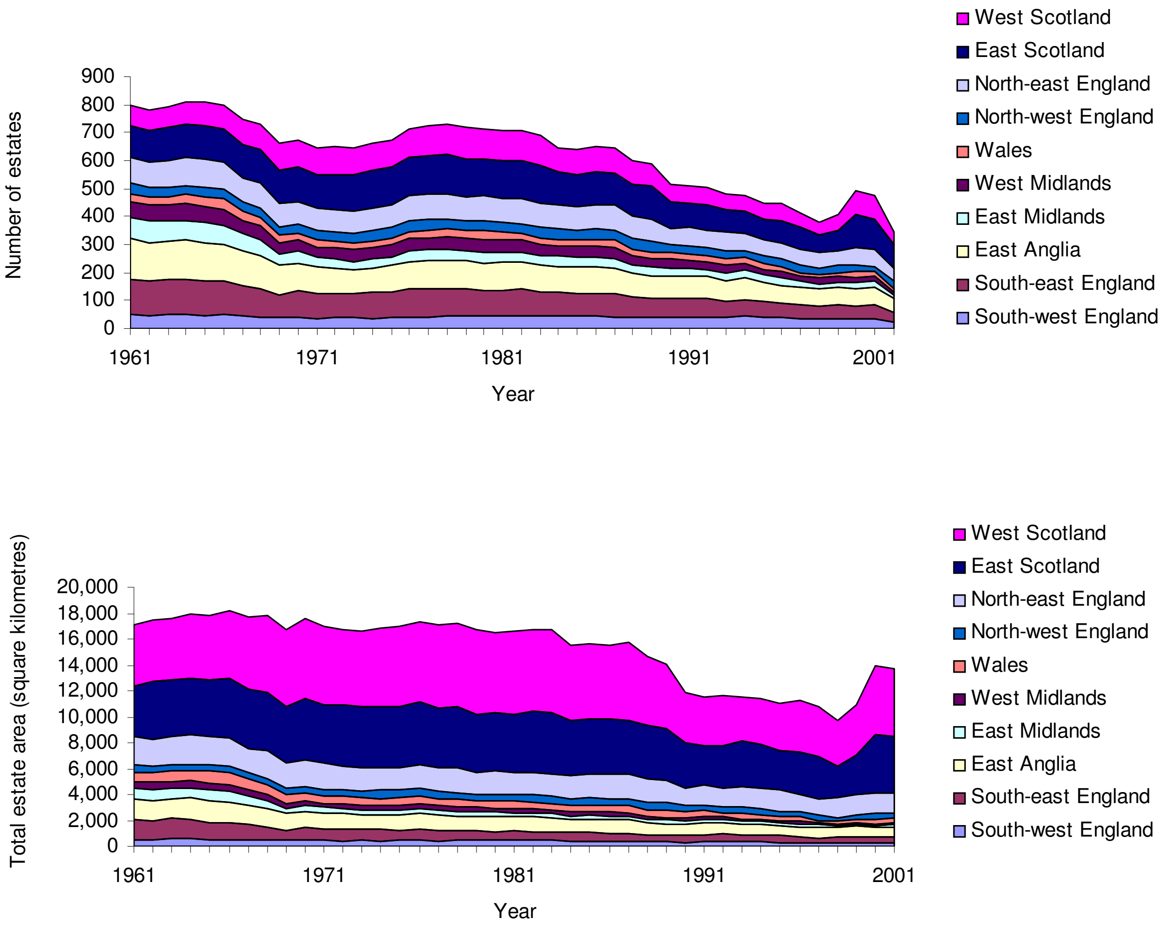


Within the UK, data can be extracted at the country, regional and county levels from the NGC, since information about the county and region code is contained within the estate code. The smallest scale available for study using the NGC data in their current format is the county level.

Temporal trends in coverage

There was a temporal shift in the regional composition of the sample, both in terms of the number of estates contributing to the NGC per region and their area, with declines occurring in both since 1961. Figure 3.1.2 clearly shows the geographical bias of the NGC sample, with most of the estate area covered being accounted for by the Scottish regions and north-east England. This does not however affect the data since the unit of bags (number of animals killed per square kilometre) is scale-invariant.

Figure 3.1.2: Number and area of estates contributing game sheets to the NGC, 1961-2001.



3.2 Quantification of data-entry error

Bags of the five predatory species are recorded on the same form, thus for each predator sheet five data items were checked. This results in a total of 805 data items for predator bags compared with 161 for the number of gamekeepers where a single datum was checked per form. Relatively fewer data were selected for predatory species prior to 1961; this is due to the reduced availability of records for these species before formalisation of the predator sheet. For rabbits and predatory species, the rate of errors in bags is higher in earlier time categories (Table 3.2.1), averaging 2.1% prior to 1961 with a corresponding figure of 0.2% after 1961 (χ^2_1=13.35, P<0.001). Overall, the average rate of error in predator bags did not differ significantly from that in the rabbit data (χ^2_1=3.52, ns). For records of the number of gamekeepers the error rate was consistent over the years (χ^2_1=0.30, ns) with comparable rates of error in both datasets (χ^2_1=0.76, ns); it averaged 3.3% (Table 3.2.1).

Table 3.2.1: Numbers of data items checked and rates of error in a) bags and b) the number of gamekeepers from a random 1% sample of the NGC.

a) Bags of five predatory mammals and of rabbits

Time period	Predatory species			Rabbits		
	Data checked	Number of errors	Error rate	Data checked	Number of errors	Error rate
1840-1860	0	-	-	2	0	0.0%
1861-1880	10	0	0.0%	4	1	25.0%
1881-1900	0	-	-	30	2	6.7%
1901-1920	0	-	-	48	0	0.0%
1921-1940	0	-	-	73	1	1.4%
1941-1960	10	1	10.0%	57	0	0.0%
1961-1980	440	1	0.2%	155	1	0.6%
1981-2001	345	0	0.0%	109	0	0.0%
Total	805	2	0.2%	478	5	1.0%

b) Number of gamekeepers

Time period	Predator sheets			Game sheets		
	Data checked	Number of errors	Error rate	Data checked	Number of errors	Error rate
1840-1860	0	-	-	2	0	0.0%
1861-1880	2	0	0.0%	4	0	0.0%
1881-1900	0	-	-	30	1	3.3%
1901-1920	0	-	-	48	0	0.0%
1921-1940	0	-	-	73	3	4.1%
1941-1960	2	0	0.0%	57	2	3.5%
1961-1980	88	4	4.5%	155	6	3.9%
1981-2001	69	3	4.3%	109	2	1.8%
Total	161	7	4.3%	478	14	2.9%

Estates with anomalous grid references were identified in order of decreasing scale, i.e. from UK down to the county level (Table 3.2.2). Anomalies identified at one level were excluded from the query used for identification of anomalies at the next scale, therefore the number of anomalies is cumulative going from the top to the bottom of the table. The level of error in the estate grid references is low at a UK and country level but increases moving towards finer scales. There is a higher rate of anomalies in estate grid references for game sheets than for predator sheets (χ^2_1=4.93, P<0.05): note that a large proportion of the errors for game and predator data result from anomalies for the same estates, which have incorrect grid references in both datasets.

Table 3.2.2: Anomalous estate grid references (number and percentage) in the NGC for game and predator sheets, for estates that contributed records between 1961 and 2001.

Scale	Game sheets (*n*=1603)		Predator sheets (*n*=1222)	
	Number	Percentage	Number	Percentage
Estates mapping outside UK boundary	10	0.6	7	0.6
Estates mapping outside country boundary	3	0.2	3	0.2
Estates mapping outside regional boundary	65	4.1	44	3.6
Estates mapping outside county boundary	167	10.4	97	7.9
Total number of anomalous grid references	245	15.3	151	12.4

The percentage of estates with no grid reference is not consistent among regions (χ^2_{10}=513.23, P<0.001), being particularly high in both Midlands regions and Northern Ireland and lowest in Scotland (Table 3.2.3). There was no significant difference in the percentage of estates with no grid reference between game sheets and predator sheets (χ^2_{10}=4.19, ns).

Table 3.2.3: Regional numbers and percentages of estates with no grid reference submitting data to the NGC between 1961 and 2001 (denominators for percentages are numbers of estates per region given in Table 3.2.1).

NGC region	Game sheets		Predator sheets	
	Number of estates	Percentage of estates	Number of estates	Percentage of estates
England	207	18.9	168	19.6
South-west England	7	6.7	7	7.6
South-east England	19	8.1	14	8.0
East Anglia	12	5.1	9	4.8
East Midlands	67	50.0	61	54.0
West Midlands	48	37.5	34	34.7
North-west England	13	18.1	9	16.4
North-east England	41	22.2	34	25.6
Wales	15	18.1	14	28.0
Scotland	17	4.2	7	2.3
East Scotland	10	4.3	5	2.8
West Scotland	7	4.0	2	1.6
Northern Ireland	15	93.8	14	93.3
UK total	254	15.8	203	16.6

3.3 Log-linear Poisson regression with site and year effects

Consistency of temporal changes in bag density across regions

The interaction between region and year was found to be highly significant for all species (Table 3.3.1), thus in all subsequent analyses data from each region were treated as discrete data sets. Region 11 (Northern Ireland) was omitted from further analyses, since the number of estates and consequently sample size was low. The significance of the interaction was initially tested for all the complete data series for each species. It was later tested for the restricted data from 1961-2001 as data for this period were consistently available for all species and regions. These time periods are presented in the upper and lower halves of Table 3.3.1 respectively.

Table 3.3.1: *F*-values testing for the significance of the interaction between region and year, calculated from the output of GLM models including and omitting the interaction term.

Species	Numerator df	Denominator df	*F*	*P*
All years				
Fox	477	9,519	2.56	<0.001
Rabbit	1,459	26,259	3.52	<0.001
Rat	440	6,414	3.85	<0.001
Squirrel	363	7,601	4.58	<0.001
Stoat	542	9,789	5.60	<0.001
Weasel	496	9,622	2.69	<0.001
1961-2001				
Fox	400	9,043	2.51	<0.001
Rabbit	408	14,515	4.37	<0.001
Rat	401	6,403	3.95	<0.001
Squirrel	353	7,601	4.70	<0.001
Stoat	406	9,672	4.28	<0.001
Weasel	371	9,132	2.29	<0.001

Estate turnover

Regional indices from the GLM with site and year effects show consistency with the indices calculated from annual mean bags in the directionality of the trend shown (Figure 3.3.1a-f). There there are notable differences, however, between the annual mean bags and the regression indices. The year estimates from the regression analyses tend to smooth out fluctuations between adjacent years and there is less noise in the indices. This is because the model includes estate as a site effect, thus the variation in the indices arising from a particular sample of estates contributing records for a species in a given year is removed from the estimates. Conversely this variation is a component of the residual error variance for the mean bags. The regression indices for brown rat bags in east Scotland (Figure 3.3.1c) and grey squirrel bags in north-west England (Figure 3.3.1d) do not conform to this observation as there appears to be more scatter in annual indices after accounting for site effects.

Figure 3.3.1a: Indices calculated from annual mean red fox bags (closed symbols) and year coefficients from the regression model (open symbols), by NGC region for the period 1961-2001.

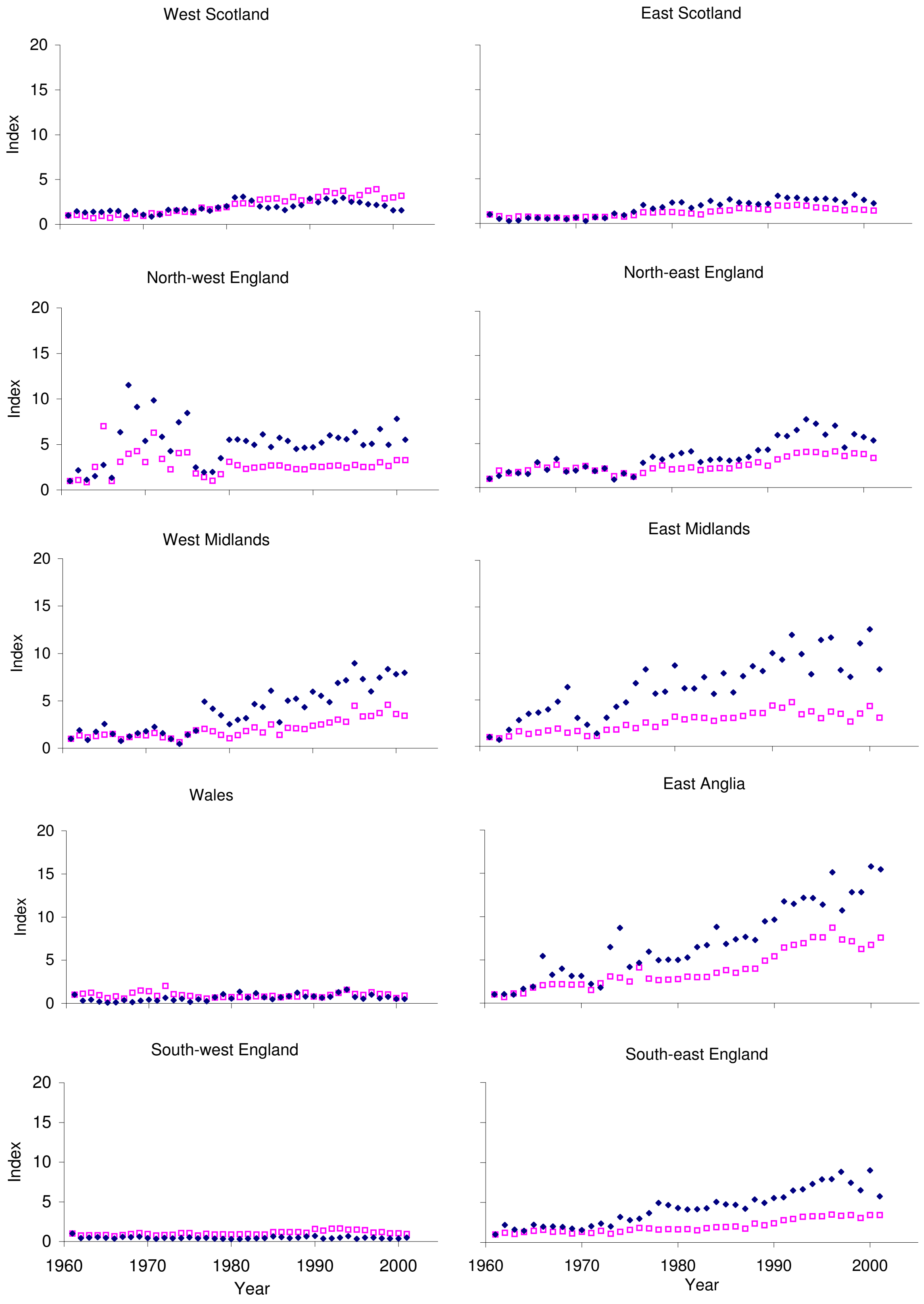


Figure 3.3.1b: Indices calculated from annual mean rabbit bags (closed symbols) and year coefficients from the regression model (open symbols), by NGC region for the period 1961-2001.

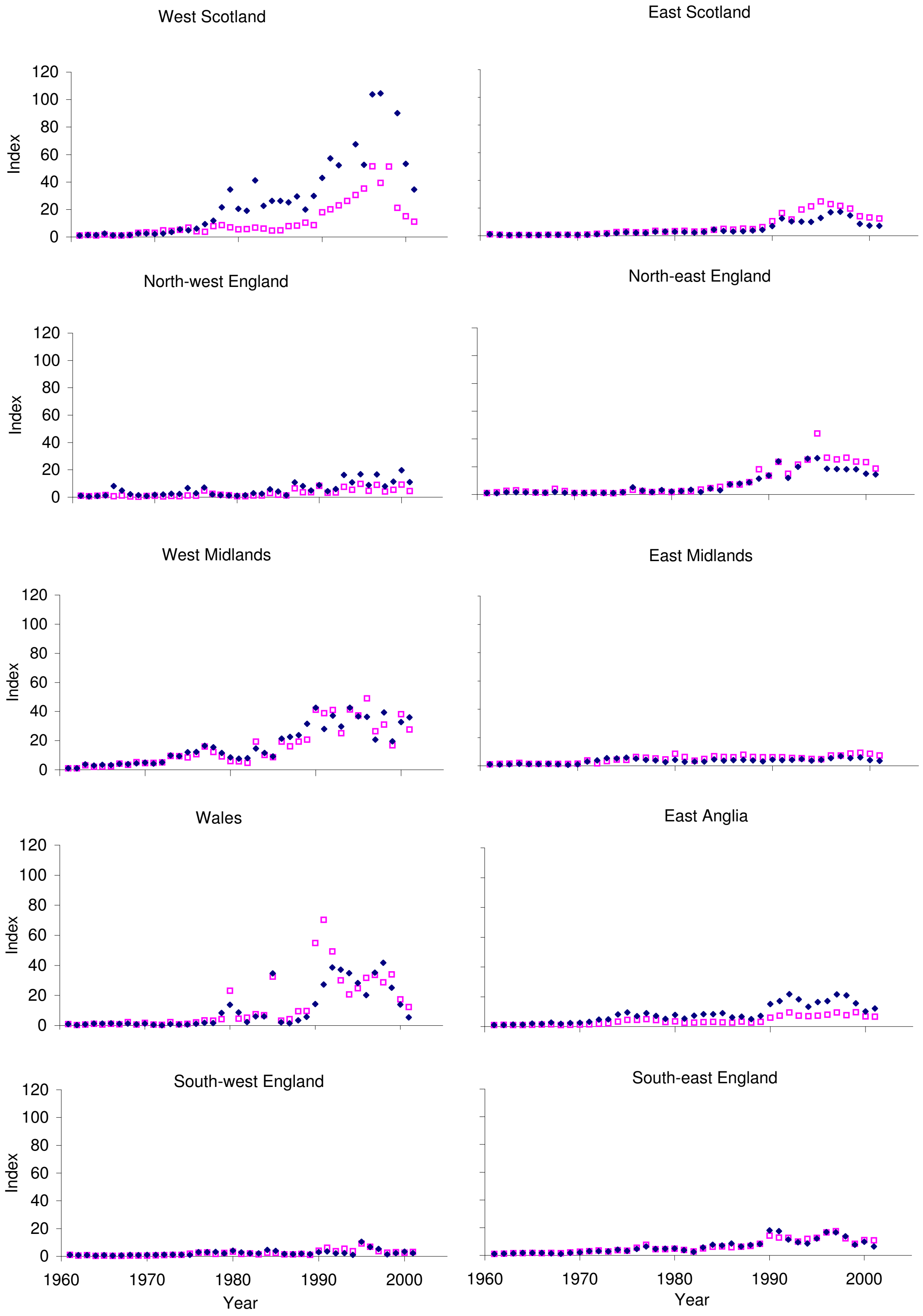


Figure 3.3.1c: Indices calculated from annual mean brown rat bags (closed symbols) and year coefficients from the regression model (open symbols), by NGC region for the period 1961-2001.

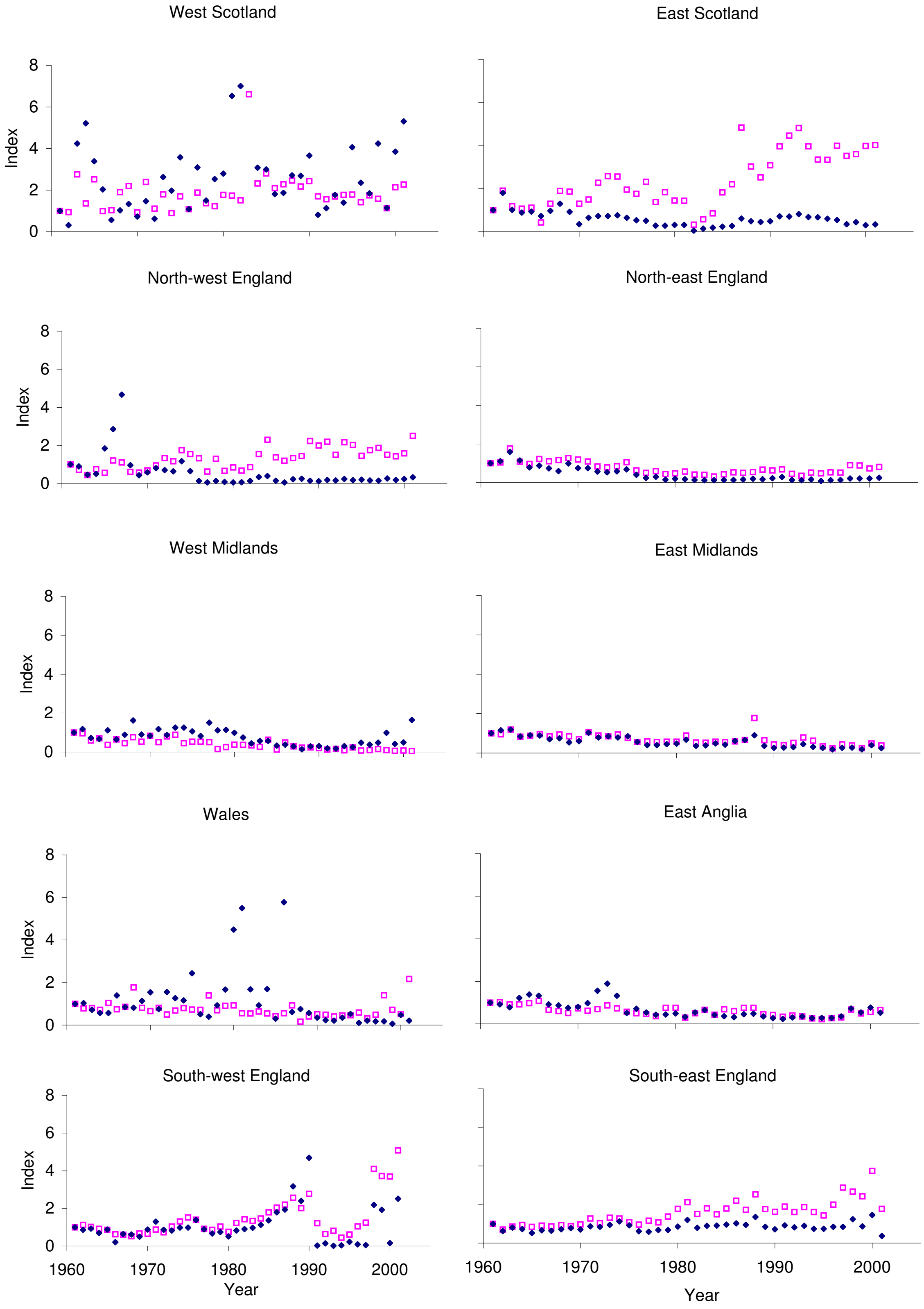


Figure 3.3.1d: Indices calculated from annual mean grey squirrel bags (closed symbols) and year coefficients from the regression model (open symbols), by NGC region for the period 1961-2001.

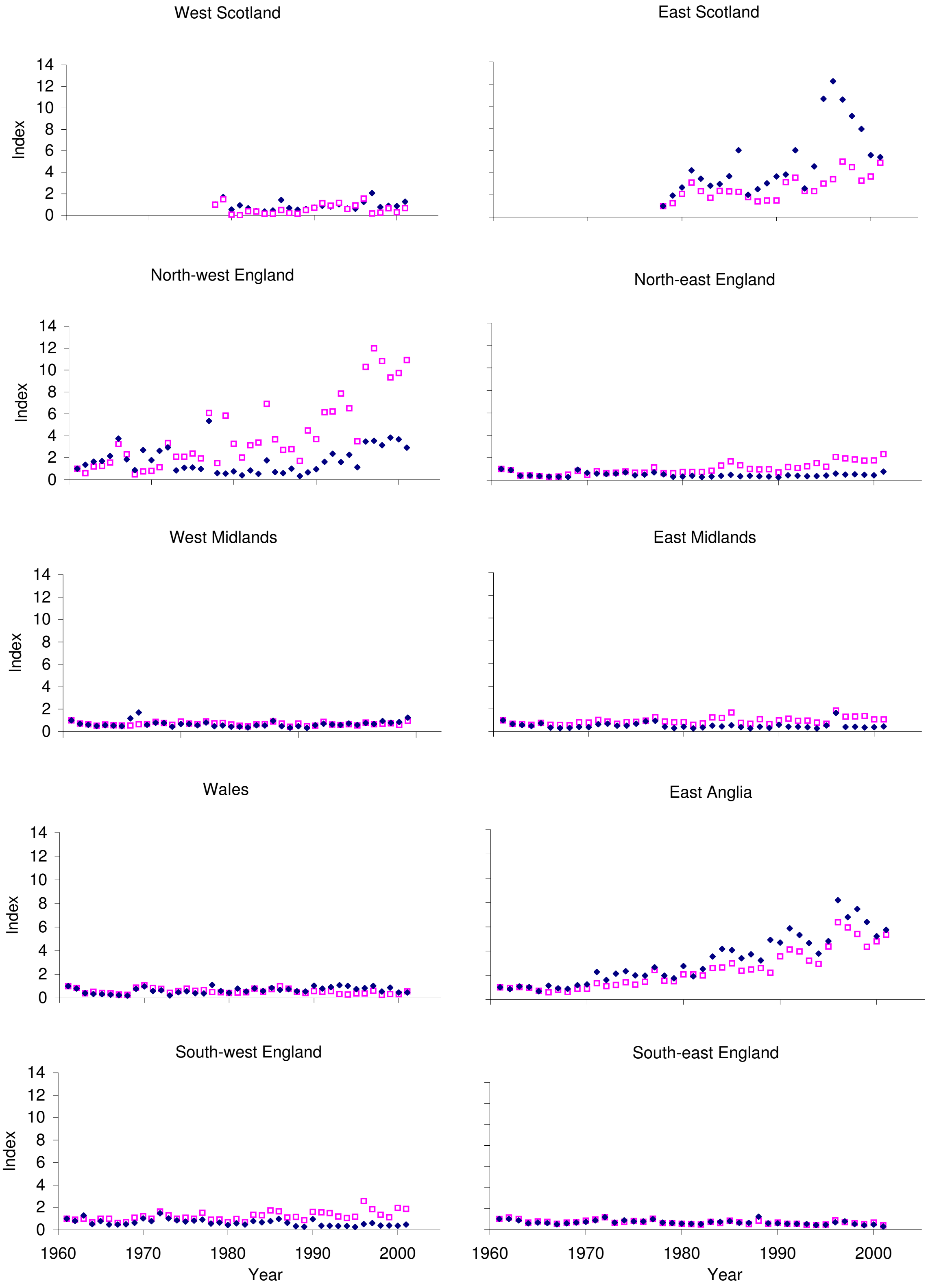


Figure 3.3.1e: Indices calculated from annual mean stoat bags (closed symbols) and year coefficients from the regression model (open symbols), by NGC region for the period 1961-2001.

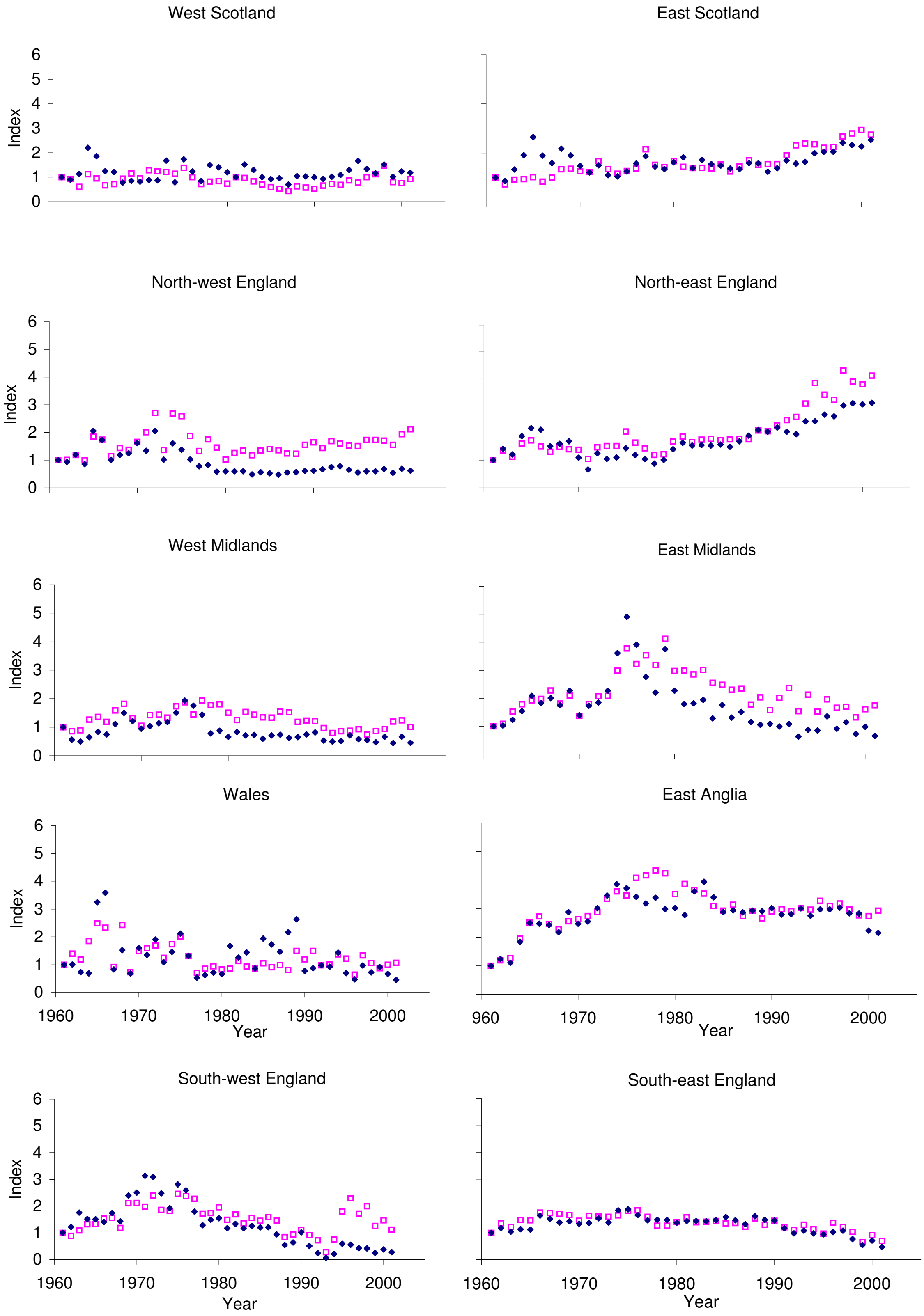
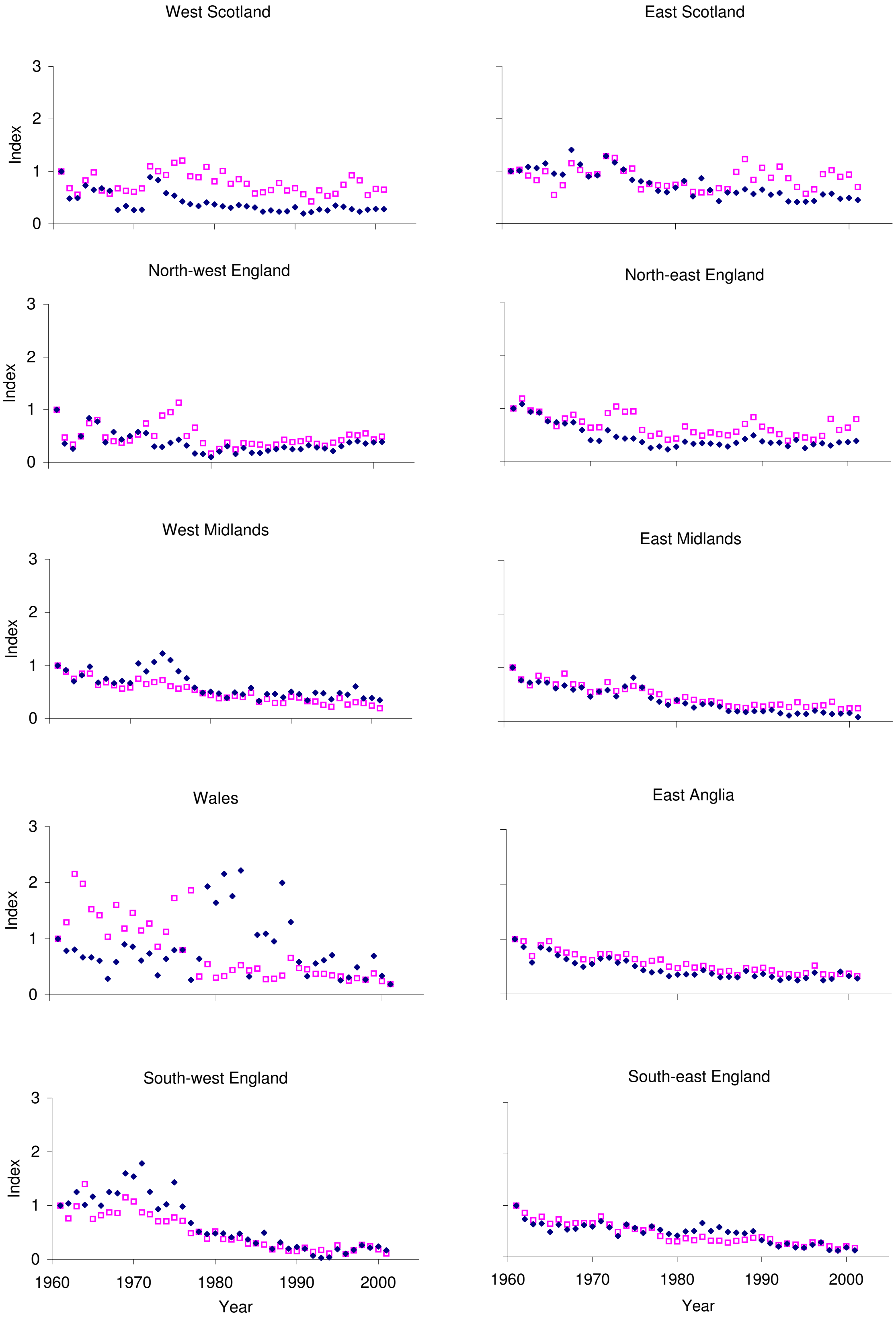


Figure 3.3.1f: Indices calculated from annual mean weasel bags (closed symbols) and year coefficients from the regression model (open symbols), by NGC region for the period 1961-2001.



West Scotland

East Scotland

North-west England

North-east England

West Midlands

East Midlands

Wales

East Anglia

South-west England

South-east England

The indices from Poisson regression with site and year effects for fox bags (Figure 3.2.3a) show a systematic decrease in the magnitude of the temporal trend relative to the annual mean bag indices. For most other species and regions they are of similar magnitudes, although in the regions where the regression indices show more variation they also over-estimate the trend relative to the mean bag indices. Indices from log-linear Poisson regression with site and year effect appear to show an increased upward trend in north-west England for stoat and rat bags (Figure 3.2.3c,e), relative to the mean bag indices.

3.4 Trend analysis

Red fox bags showed a significant increase between 1975 and 2000 in south-east England, East Anglia, the east Midlands, west Midlands, north-east England and east and west Scotland (Table 3.4.1). No significant change was detected in south-west England, Wales or north-west England over the same period, noting that the number of estates was lowest in the latter two regions (Table 3.1.2). Increases of the greatest magnitude occurred in East Anglia and the west Midlands.

Rabbit bags showed significant increases between 1975 and 2000 in all regions with the exception of north-west England. Increases in rabbit bags were most dramatic in Wales, the north-east of England and east and west Scotland. Note that the confidence intervals for the increases in rabbit bags are very wide in comparison to other species such as stoats and weasels.

There was a significant increase in brown rat bags in south-east England and a significant decrease in the east Midlands. In the other regions, no significant change in rat bags was detected between 1975 and 2000. Confidence intervals for percentage changes in rat bags were wide, although not to the extent seen with rabbit bags. There does not appear to be a strong regional pattern in the distribution of increases or decreases in rat bags.

Significant increases in bags of grey squirrels occurred in south-west England, East Anglia, east Midlands, north-east England, north-west England, east Scotland and west Scotland between 1975 and 2000, with the greatest percentage increases occurring in East Anglia, northern England and Scotland. There was no detectable change in squirrel bags in south-east England or Wales. With the exception of East Anglia, increases in squirrel bags were thus predominantly located in the north of England and Scotland.

Significant declines in stoat bags were observed in south-east England, the east Midlands and the west Midlands. Significant increases in stoat bags were observed in north-east England and east Scotland. No change was detected between 1975 and 2000 in the other more southerly and westerly regions (south-west England, East Anglia, Wales and north-west England). Confidence intervals for stoat bags tended to be narrow compared to those for other species such as rabbits and rats.

Significant declines in weasel bags between 1975 and 2000 were observed in all southern regions (west England, east England, East Anglia, Wales, east and west Midlands). In north-east and north-west England, east Scotland and west Scotland, the changes were not significant. The magnitude of the declines tended to be greatest in the western regions. Confidence intervals for percentage changes in weasel bags were fairly narrow, as with those for stoat bags.

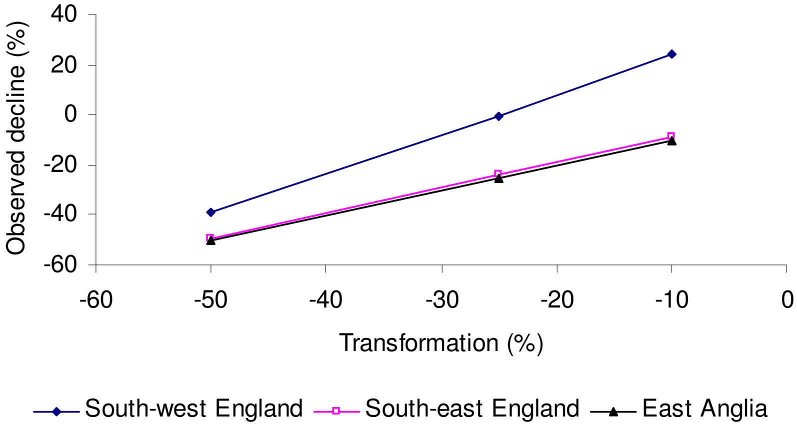
Table 3.4.1: Percentage changes in bags for red fox, rabbit, brown rat, grey squirrel, stoat and weasel over the 25-year period 1975-2000 (* indicates significance at the 5% level). The bootstrapped 95% confidence limits are given in parentheses. The numbers of estates in the analyses are the same as in Table 3.1.2.

NGC region	Fox	Rabbit	Rat	Squirrel	Stoat	Weasel
South-west England	23.7% (-5.1%, 59.8%)	140.3%* (30.7%, 523.2%)	237.4% (-19.9%, 443.7%)	57.2%* (12.2.%, 105.8%)	-27.1% (-65.3%, 62.1%)	-74.9%* (-83.7%, -32.0%)
South-east England	136.6%* (70.2%, 217.2%)	220.3%* (103.8%, 337.9%)	110.6%* (38.9%, 203.3%)	-26.2% (-49.6%, 3.4%)	-46.1%* (-60.8%, -28.8%)	-63.7%* (-75.3%, -53.1%)
East Anglia	180%* (92.9%, 273.7%)	175.8%* (73.2%, 640.3%)	-17.5% (-41.4%, 34.3%)	270.3%* (111.3%, 428.5%)	-19% (-34.1%, 4.2%)	-42.5%* (-57.2%, -27.3%)
East Midlands	70.1%* (7.10%, 161.1%)	83.7%* (4.7%, 329.1%)	-52.2%* (-72.2%, -31.4%)	41.2% (-3.8%, 89.8%)	-46.2%* (-65.3%, -23.9%)	-52.6%* (-72.9%, -26.9%)
West Midlands	178.6%* (94.6%, 244.3%)	301.1%* (48.3%, 473.5%)	34.2% (-56.9%, 199.1%)	3.9% (-32.7%, 65.9%)	-38.3%* (-63.0%, -3.0%)	-57%* (-78.9%, -16.7%)
Wales	5.6% (-61.4%, 50.1%)	896.8%* (2.0%, 4551%)	-87.8% (-128.8%, 332.3%)	-38.8% (-47.4%, 248.6%)	-21.9% (-65.3%, 43.6%)	-75.1%* (-84.6%, -3.0%)
North-west England	-1.2% (-48.0%, 104.9%)	371.1% (-39.0%, 1349%)	67.8% (-29.8%, 348.6%)	300.4%* (99.4%, 944.6%)	1.3% (-40.0%, 53.0%)	-19.9% (-46.9%, 18.9%)
North-east England	102.7%* (46.7%, 155.7%)	1827.2%* (805.0%, 4013%)	-1.3% (-34.3%, 72.8%)	181.2%* (112.1%, 278.2%)	170.6%* (79.5%, 260.2%)	-7.3% (-38.9%, 67.3%)
East Scotland	73.6%* (45%, 118.4%)	824%* (440.9%, 1151%)	123.5% (-36.6%, 343.3%)	429.8%* (99.3%, 531.3%)	94.3%* (29.5%, 138.8%)	-6.2% (-47.7%, 5.0%)
West Scotland	125.5%* (122.8%, 215.1%)	590.5%* (397.3%, 940.5%)	5.8% (-31.7%, 105.2%)	256.6% (-1675%, 1290%)	-4.6% (-24.1%, 34.1%)	-27.4% (-52.3%, 17.9%)

3.5 Alert potential of NGC data

Transformations were applied to weasel data for three regions in an attempt to adjust the actual change to the notional values of -10%, -25% and -50%. Log-linear Poisson regression was carried out to assess whether the modelled decline from the GAM approximated the target decline. The resulting declines for south-east England and East Anglia were similar to the ones sought from the transformation, i.e. the year index for 2000 differed from that for 1975 by the target value. However, for south-west England the observed decline was significantly different from the applied transformation (Figure 3.5.1).

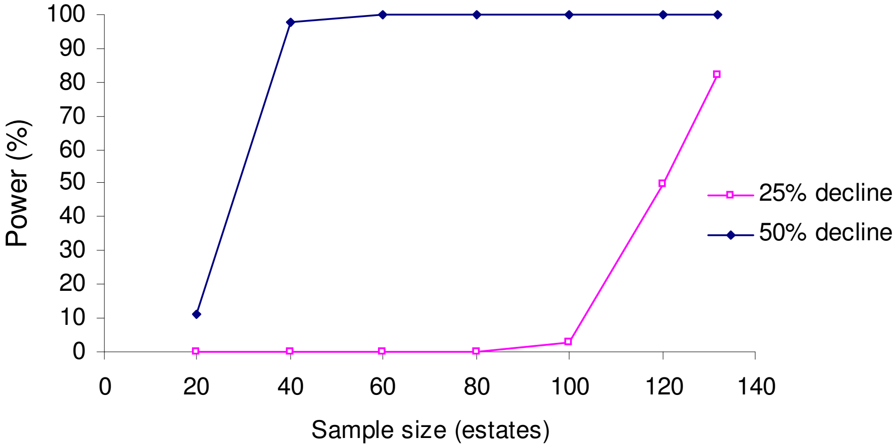
Figure 3.5.1: Applied transformations and 25-year declines modelled by GLM and GAM regression for weasel bags in three NGC regions.



South-east England was selected as a region for power analysis of NGC weasel bag data. This region has a sample size of 133 estates (Table 3.1.2). The choice of species and region was arbitrary, although there are likely to be differences between species and regions in terms of the variation in bag size. For the south-east of England, a change of -10% in weasel bags over 25 years could not be detected as significant, given that its confidence interval included 0: (-32.8%, 17.1%). Changes of -25% and -50% in weasel bags over the same time period were detected as significant, the confidence intervals for these values being (-47.5%, -1.1%) and (-66.0%, -32.4%) respectively. The width of the confidence interval was found to narrow progressively for declines of increasing magnitude.

The power of the NGC to detect a 10% decline in weasel bags over 25 years was nil within the range of sample sizes considered (1–133). Its power to detect a 25% decline was very low up to a sample size of over 100 estates, at which point power increases rapidly (Figure 3.5.2). With a sample size of 120 estates power is 50%; with 133 estates power rises to 82%. A 50% decline over 25 years can be detected with much higher power at low sample sizes: power is low (11%) for a sample size of 20 estates, but with 40 or more estates it is over 98%.

Figure 3.5.2: Power to detect a 25% and 50% decline in weasel bags over 25 years, in relation to the number of estates providing data.

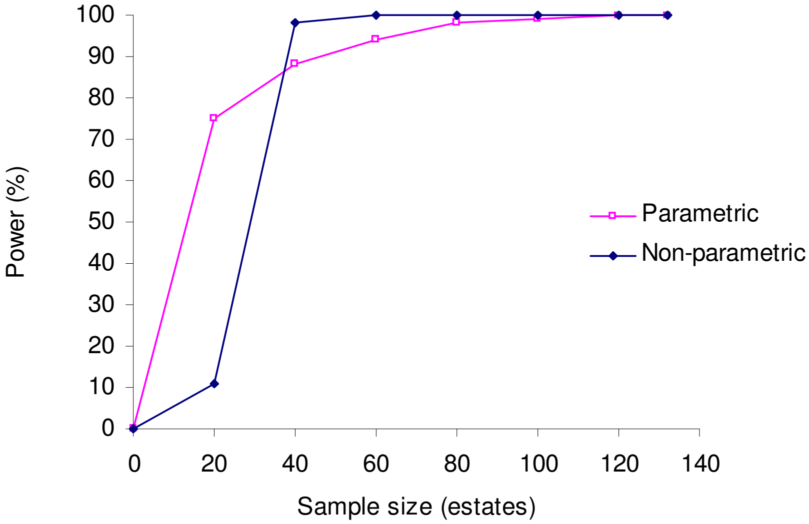


If the results of the statistical power analysis for weasel bags in south-east England are applicable to the rest of the NGC data, current regional sample sizes (Table 3.1.2) would allow declines in bags of 50% or more to be detected for all regions except Northern Ireland and possibly Wales. Declines of 25% could be detected in regions south-east England, East Anglia, north-east England and east Scotland with a statistical power of 80% or more for most species, but for other regions a greater sample size would be needed to detect a change of this magnitude. Sample sizes at a county level are too small in all cases to detect a 25% decline in bags, but may be sufficiently large in some cases that declines of 50% may be identifiable with adequate statistical power. At the country level, the sample sizes in both England and Scotland are considerable, although coverage is thinner in Wales. Thus it would be possible to identify trends at this level for 25% and 50% declines, and possibly 10% declines, in bags in England or Scotland.

The 95% confidence intervals calculated from standard errors for fitted values from the GAM using transformed data produced narrower confidence intervals than those constructed using the bootstrap method, although the confidence interval for the 50% decline approximated well to that obtained by randomisation. For completeness, statistical power was estimated anew using a single randomisation and parametric confidence intervals for this calculation.. For the south-east of England, a change of -10% in weasel bags over 25 years could not be detected as significant using parametric confidence intervals, given that its confidence interval included 0: (-25.4%, 7.8%). Changes of -25% and -50% in weasel bags over the same time period were detected as significant, the confidence intervals for these values being (-39.5%, -8.2%) and (-63.5%, -34.7%) respectively.

Although parametric confidence intervals provided a good approximation to non-parametric confidence intervals for a simulated decline of 50%, the use of parametric confidence intervals was shown to perform poorly and to over-estimate power severely at low sample sizes (Figure 3.5.3). The power estimates did however converge as sample size increased; for sample sizes of 60 and greater the estimates were all above 90%. This comparison demonstrates the need for bootstrapping in order to construct robust non-parametric confidence intervals and power estimates.

Figure 3.5.3: Power to detect a 50% decline in weasel bags in south-east England over 25 years, assessed by parametric and bootstrapping methods.



3.6 Investigating the effects of variations in sampling effort

Effect of the number of gamekeepers

The number of gamekeepers per square kilometre was found to be significantly related to bags for all species in at least some regions (Table 3.6.1). The number of gamekeepers per square kilometre had a significant effect on fox bags in all regions except for the west Midlands and north-west England. The number of gamekeepers per square kilometre had a significant effect on rabbit bags in East Anglia, the east and west Midlands, Wales, north-east and north-west England and west Scotland. For rat bags, the number of gamekeepers per square kilometre had a significant effect in south-west and south-east England, the east Midlands, north-east and north-west England and east Scotland. The number of gamekeepers per square kilometre had a significant effect on squirrel bags in all regions with the exceptions of north-east and north-west England and Scotland. For stoat and weasel bags, the number of gamekeepers per square kilometre had a significant effect in all regions.

The estimated percentage changes obtained after adjusting for the relationship with gamekeeper density (Table 3.6.2) were compared with those obtained without adjustment (Table 3.4.1), and were deemed to differ significantly if the 95% confidence intervals did not overlap. There were no significant differences between regional trends in bags for red fox, rabbit, brown rat, grey squirrel, stoat and weasel estimated with and without adjustment for gamekeeper density, except for fox bags in south-west England. There did not appear to be any systematic trend in the direction of deviations from percentage declines estimated from a model which did not include the gamekeeper variable.

Table 3.6.1: *F*-values (df in parentheses) for the effect of gamekeepers per unit area on bags for rabbit and five predatory species recorded in the NGC from 1961-2001, by region. * indicates significance at the 5% level, ** indicates significance at the 1% level and *** indicates significance at the 0.1% level.

NGC region	Fox	Rabbit	Rat	Squirrel	Stoat	Weasel
South-west England	68.18***	3.53	93.97***	32.28***	22.08***	6.35**
	(1, 714)	(1, 1185)	(1, 454)	(1, 815)	(1, 525)	(1, 389)
South-east England	46.71***	0.00	37.63***	85.04***	66.09***	111.46***
	(1, 1351)	(1, 3584)	(1, 1381)	(1, 1823)	(1, 1670)	(1, 1569)
East Anglia	12.62***	11.49 ***	0.58	56.3***	63.55***	30.78***
	(1, 1340)	(1, 4228)	(1, 1330)	(1, 1560)	(1, 1862)	(1, 1749)
East Midlands	9.74***	23.93***	36.51***	53.03***	39.80***	60.22***
	(1, 599)	(1, 1352)	(1, 695)	(1, 975)	(1, 851)	(1, 862)
West Midlands	2.95	13.77***	1.39	28.00***	17.06***	33.74***
	(1, 470)	(1, 1527)	(1, 523)	(1, 818)	(1, 734)	(1, 694)
Wales	10.31**	20.60***	1.31	31.41***	23.59***	36.82***
	(1, 210)	(1, 1276)	(1, 106)	(1, 204)	(1, 169)	(1, 163)
North-west England	0.00	25.48***	8.91**	51.04***	17.17***	39.3***
	(1, 458)	(1, 884)	(1, 200)	(1, 210)	(1, 459)	(1, 409)
North-east England	16.16***	31.61***	23.17***	22.9***	93.68***	38.16***
	(1, 973)	(1, 2367)	(1, 832)	(1, 833)	(1, 1264)	(1, 1242)
East Scotland	51.88***	0.19	16.78***	3.81	66.78***	30.47***
	(1, 2123)	(1, 5636)	(1, 528)	(1, 293)	(1, 1393)	(1, 1841)
West Scotland	14.98***	27.32***	0.13	0.01	21.89***	24.69***
	(1, 1179)	(1, 4100)	(1, 339)	(1, 59)	(1, 832)	(1, 694)

Correlation analysis of stoat, weasel and rabbit bags

There was a significant positive correlation between stoat and weasel year coefficients in south-east England, Wales, north-west England and west Scotland, and a significant negative correlation in East Anglia (Table 3.6.3). When a relationship with rabbit year coefficients was partialled out, the positive correlation between stoat and weasel year coefficients in south-east England became non-significant, and the positive correlation in south-west England became significant. There were significant positive correlations between rabbit year coefficients and stoat year coefficients in north-east England and east Scotland and significant negative correlations between these variables in south-east England and the west Midlands. Partialling out the relationship with weasel year coefficients made the negative correlation observed in south-east England non-significant.

After controlling for variation in stoat bags associated with variation in rabbit and weasel bags, the magnitude of most observed percentage changes became less (Table 3.6.4). In four out of five cases where the change in bag size had been significant, it remained so after recalculation. There were no significant differences between trends in stoat bags estimated from a model without the logarithm of rabbit bags and weasel bags and a model including these variables, based on the overlap between confidence intervals for the estimated percentage change for all regions.

Table 3.6.2: Percentage changes in bags for red fox, rabbit, brown rat, grey squirrel, stoat and weasel estimated from the GLM with the number of gamekeepers per square kilometre as an additional variable, over the 25-year period 1975-2000 (* indicates significance at the 5% level). The bootstrapped 95% confidence limits are given in parentheses.

NGC region	Fox	Rabbit	Rat	Squirrel	Stoat	Weasel
South-west England	124.5%	131.9%*	87.8%	55.4%*	-23.7%	-73.9%*
	(-8.7%, 198.8%)	(48.3%, 556.5%)	(-32.1%, 387.8%)	(10.4%, 117.6%)	(-78.5%, 43.6%)	(-100.4%, -28.4)
South-east England	136.4%*	221.4%*	111.2%*	-25.6%	-44.3%*	-61.9%*
	(54.3%, 235.4%)	(98.5%, 340.2%)	(35.4%, 202.5%)	(-38.1%, 24.4%)	(-89.0%, -18.4%)	(-83.6%, -64.7%)
East Anglia	187.3%*	181.1%*	-17%	296.9%*	-14.4%	-39.9%*
	(69.1%, 256.6%)	(72.6%, 639.9%)	(-43.7%, 46.3%)	(132.9%, 452.8%)	(-38.0%, 19.2%)	(-87.3%, -21.1%)
East Midlands	69.9%*	95.5%*	-51.6%*	44.4%	-46.6%*	-55.2%*
	(3.4%, 175.2%)	(3.8%, 357.0%)	(-97.6%, -11.2%)	(-12.1%, 176.5%)	(-75.7%, -24.3%)	(-73.9%, -25.4%)
West Midlands	180.4%*	264%*	30.8%	0.99%	-39.9%*	-58.1%*
	(91.9%, 275.0%)	(10.1%, 503..2%)	(-54.5%, 217.6%)	(-44.7%, 97.3%)	(-68.7%, -2.1%)	(-92.7%, -11.8%)
Wales	16.1%	5554.2%*	-87.1%	-23.6%	3.5%	-63.7%*
	(-47.8%, 46.7%)	(456.7%, 8954%)	(-133.3%, 351.9%)	(-98.4%, 265.2%)	(-78.4%, 88.0%)	(-88.2%,-2.7%)
North-west England	-1.15%	360.8%	92.8%	307.3%*	8.4%	-9.3%
	(-23.2%, 120.0%)	(-39.1%, 936.4%)	(-4.2%, 338.2%)	(102.8%. 1065.5%)	(-55.4%, 102.1%)	(-35.0%, 38.5%),
North-east England	109.6%*	1813.1%*	-9.2%	182.5%*	155.7%*	-11.7%
	(89.9%, 154.6%)	(773.5%, 3546%)	(-69.5%, 58.3%)	(98.5%, 301.4%)	(64.5%, 261.0%)	(-45.6%, 71.1%)
East Scotland	67.2%*	823.8%*	164.5%	431.2%*	89.9%*	-7.8%
	(17.8%, 111.4%)	(263.4%, 1251%)	(-73.2%, 389.7%)	(96.4%, 530.9%)	(16.6%, 141.2%)	(-52.3%, 10.9%)
West Scotland	124.5%*	522.9%*	3.9%	248.7%	-5.3%	-26.3%
	(83.5%, 227.9%)	(384.4%, 961.6%)	(-42.2%, 104.6%)	(-1808%, 1407%)	(-47.9%, 43.2%)	(-74.2%, 32.8%)

Table 3.6.3: Simple and partial correlation coefficients (*r*-values) from a correlation analysis of stoat and weasel and stoat and rabbit year coefficients from log-linear regression of stoat bag density with estate and year as factors, NGC 1961-2001, by region. Partial correlation coefficients are adjusted for the effect of the other variable in the analysis hence the degrees of freedom are reduced by one. * indicates significance at the 5% level, ** indicates significance at the 1% level and *** indicates significance at the 0.1% level.

NGC Region	Stoat v. weasel		Stoat v. rabbit	
	Simple r_{39}	Partial r_{38}	Simple r_{39}	Partial r_{38}
South-west England	0.301	0.334*	-0.069	0.167
South-east England	0.521***	0.254	-0.509***	-0.222
East Anglia	-0.460**	-0.358*	0.310*	-0.036
East Midlands	-0.046	0.156	0.160	0.218
West Midlands	0.247	-0.175	-0.491***	-0.466**
Wales	0.575***	0.555***	-0.215	0.121
North-west England	0.546***	0.571***	0.114	0.229
North-east England	-0.274	0.139	0.883***	0.876***
East Scotland	-0.071	0.099	0.795***	0.796***
West Scotland	0.541***	0.583***	0.119	0.283

Table 3.6.4: Percentage changes in stoat bags estimated from the GLM with the logarithm of rabbit bags and the logarithm of weasel bags as additional variables ('adjusted') and percentage changes estimated from the original model, over the 25-year period 1975-2000. The bootstrapped 95% confidence limits are given in parentheses. * indicates significance at the 5% level.

NGC region	Stoat original	Stoat adjusted
South-west England	-27.1% (-65.3%, 62.1%)	28.7% (-82.8%, 81.1%)
South-east England	-46.1%* (-60.8%, -28.8%)	-28.2% (-48.9%, -0.73%)
East Anglia	-19% (-34.1%, 4.2%)	-2.1% (-15.2%, 21.8%)
East Midlands	-46.2%* (-65.3%, -23.9%)	-32.5% (-53.0%, -12.1%)
West Midlands	-38.3%* (-63.0%, -3.0%)	-29.6% (-45.2%, 3.66%)
Wales	-21.9% (-65.3%, 43.6%)	25.7% (-80.3%, 806.2%)
North-west England	1.3% (-40.0%, 53.0%)	29.6% (-39.2%, 121.8%)
North-east England	170.6%* (79.5%, 260.2%)	106.5% (50.1%, 166.8%)
East Scotland	94.3%* (29.5%, 138.8%)	48.3% (4.3%, 84.8%)
West Scotland	-4.6% (-24.1%, 34.1%)	28.9% (-21.6%, 96.2%)

3.7 Assessing the representativeness of NGC bag size indices as indices of abundance: comparisons with indices from other mammal monitoring schemes

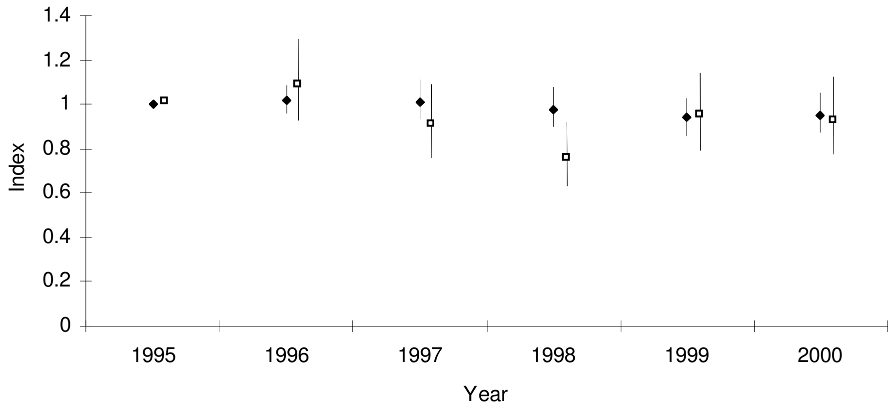
The NGC bag indices and BBS abundance indices for red fox, rabbit and grey squirrel (1995-2000) are given in Table 3.7.1. Based on the BBS analysis, none of these species showed evidence of an overall linear trend in numbers between the years 1995 and 2000 (Newson and Noble, 2003).

Table 3.7.1: NGC bag indices and BBS abundance indices for red fox, rabbit and grey squirrel, 1995-2000. The 95% confidence intervals are given for all years except 1995, the reference year (allocated a value of 1). BBS indices and 95% confidence intervals are taken from Newson and Noble (2003).

Species	Survey	1995	1996	1997	1998	1999	2000
Fox	NGC	1.00	1.02	1.01	0.98	0.94	0.95
			(0.96, 1.09)	(0.93, 1.11)	(0.90, 1.08)	(0.86, 1.03)	(0.87, 1.05)
	BBS	1.00	1.08	0.90	0.75	0.94	0.92
			(0.92, 1.28)	(0.75, 1.08)	(0.62, 0.91)	(0.78, 1.13)	(0.76, 1.11)
Rabbit	NGC	1.00	0.99	0.92	0.90	0.64	0.62
			(0.90, 1.09)	(0.80, 1.06)	(0.78, 1.04)	(0.55, 0.74)	(0.53, 0.73)
	BBS	1.00	0.99	1.24	0.99	0.85	0.93
			(0.92, 1.06)	(1.16, 1.32)	(0.93, 1.06)	(0.79, 0.91)	(0.87, 1.00)
Squirrel	NGC	1.00	1.64	1.42	1.32	1.12	1.26
			(1.51, 1.78)	(1.25, 1.6)	(1.17, 1.50)	(0.98, 1.27)	(1.11, 1.44)
	BBS	1.00	2.19	1.23	1.06	0.88	1.12
			(1.96, 2.43)	(1.08, 1.38)	(0.94, 1.21)	(0.78. 1.00)	(0.99, 1.26)

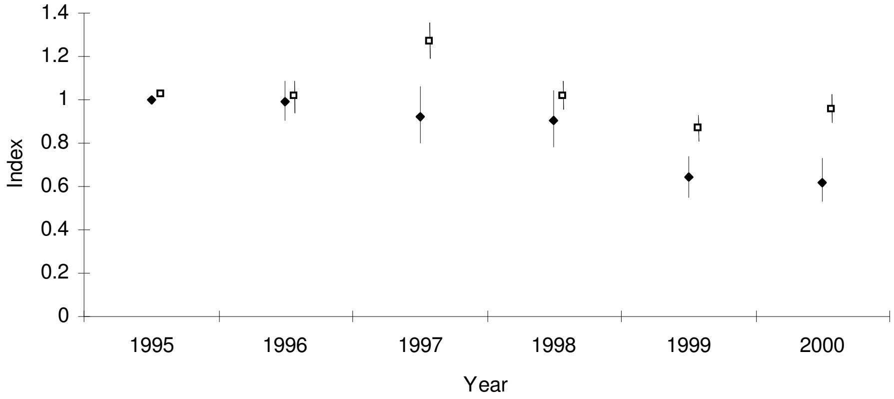
No significant differences are apparent in the annual indices for fox from either NGC bag data or BBS data (Figure 3.7.1). Note that the confidence intervals are narrower for NGC data for fox, as a result of a larger sample size.

Figure 3.7.1: NGC (closed symbols) and BBS (open symbols) indices for red fox, 1995-2000. The vertical bars around each point indicate 95% confidence intervals.



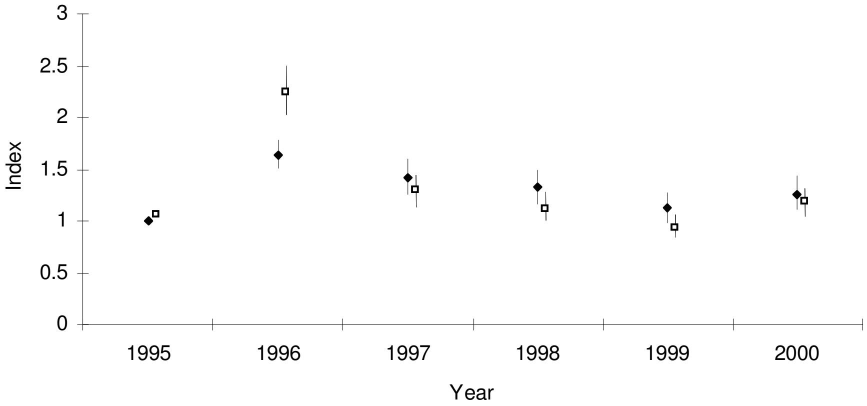
The indices show a similar pattern of variation for rabbits (Figure 3.7.2), although the rabbit index for BBS data is consistently higher in years from 1997 onwards. Error bars are wider for NGC data, probably due to the large variability in rabbit bags. There is agreement between BBS and NGC rabbit indices in 1999, where both rabbit bags recorded by the NGC and rabbit numbers recorded in the BBS were significantly lower than in preceding years.

Figure 3.7.2: NGC (closed symbols) and BBS (open symbols) indices for rabbit, 1995-2000. The vertical bars around each point indicate 95% confidence intervals.



Grey squirrel indices show good correspondence between NGC and BBS data, with a similar overall pattern of variation between years, and significantly higher values in 1996 relative to 1995 for both datasets (Figure 3.7.3). Error bars are comparable between the NGC and BBS although the 1996 index for the latter has a slightly wider error bar.

Figure 3.7.3: NGC (closed symbols) and BBS (open symbols) indices for grey squirrel, 1995-2000. The vertical bars around each point indicate 95% confidence intervals.



4 Discussion

4.1 Discussion of methods

Trend analysis

The question of whether species are increasing or declining in abundance is of central importance in species conservation and management. Surveillance or monitoring programmes should be designed to provide long-term trends in populations, and must be able to detect the difference between long-term population trends and short-term fluctuations in populations that are the result of natural between-year variation in breeding success and mortality. Use of a GLM framework incorporating the estimation of a set of site-specific factors in addition to year-specific factors allows the bags to vary between sites (estates); annual variation is then modelled in a parallel fashion across locations. In practice it is likely that annual trends differ between estates i.e. there is an interaction between estate and year. This source of variation was not modelled in this analysis owing to constraints on computing time, so was incorporated in the residual error term. It would be possible to explore this area further using Generalised Multilevel Modelling (Goldstein, 1995).

The use of a Generalised Additive Model avoids the fitting of a linear trend to the data, which is likely to be unrealistic. A consistent system was used to assess the fit of curves defined by varying degrees of freedom to the year coefficients in order to choose an appropriate *df* value. However, the distinction between a 'true' trend in bags and random noise remains somewhat subjective: erring on the side of a low *df* value has the added advantage of giving narrower confidence intervals for the fitted values from the GAM, since with high levels of smoothing, estimates for year effects are constrained by the estimates in adjacent years. Previous attempts to distinguish genuine patterns of nonlinear change from the 'noise' of annual fluctuations have applied smoothing algorithms to annual abundance estimates. Siriwardena *et al*. (1998) calculated trends from the CBC data using the Mountford method, which compares changes in abundance across all pairs of years, expresses indices of abundance as departures from the mean over the years in the analysis, and allows for serial correlation in counts. They then fitted a running-medians smoothing algorithm to the Mountford index series. Trend analysis in *The population status of birds in the UK* (Gregory *et al*., 2002) used a GAM approach, analysing absolute changes in population size for a fixed time period (i.e. 25 or 50 years) rather than using an annual rate of change. However, it is then simple to convert the rate of change over a fixed period into an annual rate to judge against predefined thresholds. For example, a 50% decline over 25 years is equivalent to an annual rate of -2.73%. Using a *per annum* criterion would be valuable for short-term data sets, for example the BBS data, however trends identified in this way should not be extrapolated over longer time spans because fluctuations in population size are unlikely to follow an exponential model.

4.2 NGC coverage

The lower availability of predator sheets relative to game sheets may be attributable to estate owners or gamekeepers perceiving that bag numbers are of greater interest than predator control efficiency. In terms of the utility of the data for trend analysis, this may have implications for the statistical power of the NGC for detecting trends in game and predator bags (discussed below). However, the number of estates contributing predator sheets has been increasing since 1961, when records of pest and predatory species were incorporated into the NGC. In contrast the number of estates contributing game sheets has been fairly consistent since 1961. Although the sample size of the NGC in terms of the average annual number of estates from which bag data are received may be lower than other mammal monitoring schemes (e.g. the BBS) for frequently observed species such as rabbits, sample size is likely to be comparable for species such as the red fox, and greater for species sighted rarely such as the stoat and the weasel.

4.3 Quantification of error in the NGC

The rate of error in bags for the species in this analysis between 1961 and 2001 is extremely low, with an error rate per datum of approximately 1 in 500 for rabbit bags and predator bags. This low rate of error is unlikely to have an appreciable effect on the results of analyses restricted to data from this period of time. For example, with an average of around 600 estates contributing data for game species per year, one might expect to find one error in rabbit bags every year and one error in any one of the predator species bags every year. The error rate for gamekeeper records was found to be considerably higher than that for bags. This may be caused by ambiguity in the layout of the census form and should be addressed if the gamekeeper data are to be used in further analyses.

The observed decline in the rate of errors in the bag data over time is probably due to the progression of changes that have been implemented in the layout of the game and predator sheets over time. Any changes have been designed with the aim of increasing the clarity of the sheet for those required to complete and return it, or else to improve the quantity of information that may be extracted from it. Prior to 1961 when the NGC was set up, records for game species were taken from the game books of individual estates and from data collected by Douglas Middleton at the University of Oxford's Bureau of Animal Populations (Tapper, 1992). A potential additional source of error for earlier bag data is the transfer of data from these records to the NGC.

Anomalous estate grid references are infrequent at large scales (UK and country), becoming more frequent at regional and county scales. The level at which grid reference errors become a concern in data analyses depends on the scale at which it is desirable to estimate population trends. For example, errors at county and regional scales are not relevant to an analysis on a country scale. The rate of anomalous estate grid references should not be regarded as a major problem; once such estates have been identified, it is a relatively easy matter to acquire a correct reference using GIS software, provided that there is a postal address or contact details for the estate.

The presence of estates with no grid reference in the datasets should not affect analysis of the data if the region codes are correct. However, for any study on species distribution (for example, it may be of interest to map the geographical spread of grey squirrels on Scottish estates over time), a grid reference would be required. The problem of estates for which there is no grid reference can be rectified in the same way as grid reference anomalies.

4.4 Loglinear Poisson regression with site and year effects

Turnover of estates in the NGC

There is more fluctuation between annual indices (where the effect of turnover of estates is not compensated for) calculated from mean bags than between indices estimated by log-linear Poisson regression with site and year effects, and thus more noise in any apparent trend. In order to estimate how a measure of population size changes with respect to time, it is desirable to remove site (estate effects) from annual effects since a few estates may have a disproportionate effect on annual mean bags if their bags are particularly high, creating outlying data. Removal of this variability is advantageous for the estimation of trends since it will allow narrower confidence intervals to be placed on the results. In some cases, the plots show a reduced magnitude of the temporal change in the indices from regression compared with those from mean bags. This may occur because the sample of estates from which annual means are calculated is not consistent with regard to the size of the bags reported over time. For example, if there is a net exit from the NGC of estates on which low bags are recorded and net entry of estates on which higher bags are recorded over the time-frame of analysis, the overall increase would appear as an annual effect for mean bags, whereas the variation would be attributed to the estates themselves in the regression model.

4.5 Trend analyses

Rabbit

The ubiquitous increases in rabbit bags almost certainly reflect increases in rabbit abundance following the widespread myxomatosis epidemic in the 1950s. It has been estimated that the virus destroyed over 99% of the UK rabbit population (Thompson, 1956), with pre- and post-myxomatosis estimates of its size standing at 100 million and 1 million respectively (Harris *et al*., 1995). Rabbit numbers have now recovered to the extent that rabbits are once more considered a major pest to agriculture, causing an estimated financial loss of £40 million per year in the UK through crop damage (DEFRA, 2002). Monitoring rabbit abundance is therefore of key importance to any initiatives designed to control rabbit numbers in the UK.

Data are available on the changes in the distribution of rabbits in the UK during the period for which trends in rabbit bags were analysed. The Mammal Society began collation of information regarding the distribution of mammal species on 10x10-km squares in the British Isles in 1965, although no data were collected on rabbit abundance on these squares. The survey recorded consistent increases in the proportion of surveyed squares where rabbits were present: 56% of squares with positive records of rabbits in 1970, 78% in 1976, and 87% by 1984. Similar expansions of the distribution of rabbits are recorded by the Forestry Commission, which has made four surveys of wildlife in forests at five-year intervals, starting in 1968. The survey unit, in common with the Mammal Society survey, was the 10x10-km square. The proportion of Forestry Commission sites where rabbits were present doubled, from 35-45% in 1968 to 80-98% in 1973. In addition, the Ministry of Agriculture, Fisheries and Food (MAFF) has operated three consecutive types of rabbit survey on farms in England and Wales since 1969 (whole-farm, 1969-1970; short transect, 1973-1978; long transect, 1980-1984). The adjusted data for the six MAFF regions show a gradual upwards trend in the presence of rabbits on farms in all regions. Although the MAFF surveys did not have the specific aim of monitoring rabbit abundance, there is a significant relationship between the proportion of sections with rabbit signs and rabbit counts, verified by a subsample of 90 sites. Records of presence or absence from the short transect and long transect surveys may thus provide a rough index of rabbit abundance. The indices for all six MAFF regions rose over the period 1973-1984; NGC rabbit bag indices also rose in all regions over this period.

In Scotland, the Department of Agriculture and Fisheries (DAFS) carried out surveys in 1969, 1970, 1973 and 1974 similar in design to MAFF's whole-farm surveys. These were superseded by a survey carried out in 1991 by the Scottish Office Agriculture and Fisheries Department (SOAFD) using the same method. These surveys were aimed at establishing the extent and financial cost of rabbit damage, and respondents were asked to classify rabbit damage as 'none', 'slight' or 'serious'. The results of the survey can therefore be used to monitor changes in the distribution of rabbits on Scottish farms, but owing to the subjective element involved in judging the severity of infestation, their interpretation with regard to changes in rabbit abundance remains problematic.

A government advice note on the subject of rabbits quotes an annual rate of increase in the UK rabbit population of 2%, and estimates that it is now at 35-40% of its pre-myxomatosis level (DEFRA, 2002). This annual rate of increase would equate to a 64% increase over 25 years and is of much smaller magnitude than that observed in NGC rabbit bag data. Trout *et al.* (1986) suggest that the discrepancy between rates of increase inferred from bag data and from other sources might be explained by faster rates of increase on land with a game interest and predator removal policy. This hypothesis is corroborated by the findings of Trout and Tittensor (1989), that significantly higher rabbit abundance indices are associated with sites where predator control has occurred than on ones with no predator control.

Red fox

The increases in fox bags in most regions agree with other studies of fox abundance, for example Hewson (1984). Increasing fox bags in East Anglia and Scotland are likely to reflect a real increase in fox density in East Anglia and coastal areas of east Scotland, where foxes were absent or rare prior to 1960 (Harris and Lloyd, 1991). A possible hypothesis to explain increases in other regions is that there has been an increase in food resources which has influenced fox breeding density or productivity. This is supported by the significant increases in rabbit bags in almost all NGC regions as a result of recovery following the myxomatosis epidemic. However, myxomatosis was introduced in 1953 and current fox bags are several times higher than pre-myxomatosis fox bags on the same estates (Tapper, 1992). The absence of a significant increase in fox bags in Wales runs counter to the hypothesis of increased prey availability underlying increasing fox numbers since large increases in rabbit bags were observed in this region. It is notable that other regions where there is no significant increase in fox bags are located on the west coast of England. It is thus plausible that there are broad-scale regional differences in fox population dynamics (demographic parameters, density-dependent population regulation) arising from variations in land use or regional differences in fox control methods (Reynolds and Heydon, 2000).

Data on the number of foxes killed in each of the four Forestry Commission Conservancies were analysed by Hewson (1984). Hewson notes that more foxes were killed annually by the Forestry Commission between 1971 and 1978 than in the previous decade in three of four Conservancies, and that there was a continuing increase in annual fox kills in the north and south Conservancies. The number of fox cubs killed also increased significantly in three Conservancies. Significant increases in fox bags were recorded by the NGC in east and west Scotland, with an increase of a greater magnitude occurring in west Scotland. Although contemporary data on fox kills by the Forestry Commission were not available, the data from both sources are in concordance for the period 1975 to 1978.

Brown rat

Introduced in the first half of the 18th century, brown rats are usually associated with farms, refuse tips, sewers, urban waterways and warehouses. Rat numbers are boosted by game-rearing practices (Harris 1995), since wide hedgerows, shelter belts and supplementary feeding of birds offer rats food and shelter. A study on three rural rubbish tips showed that estimates based on capture-mark-recapture data and on the rate of bait uptake underestimated abundance by factors ranging from 1.3-7.0 (Taylor *et al.*, 1981). Given that many of the bag data for rats are based on numbers poisoned, it is possible that these may also represent underestimates of the numbers killed. Rat densities are poorly monitored, if at all, by other monitoring schemes. Data from a MAFF survey in 1970 provide estimates of mean infestation rates on agricultural premises in England and Wales. However, there were no data for Scotland, and no data for other non-agricultural rural habitats. The NGC data therefore represent a potentially valuable source of information on the distribution of this species.

Control methods for rats have undergone major changes in the last 50 years. Attention was focused on rat control with the advent of the Second World War, when agricultural production was at a premium. Preliminary attempts at poisoning were unsuccessful because of the neophobic behaviour of rats. After the adoption of pre-baiting to accustom rats to feeding at certain sites before the introduction of poison, the effectiveness of this method of rat control dramatically improved. However, the widespread use of poisoning results in mortalities that cannot be counted, so the total number of kills that can be counted has declined (Tapper, 1992). Anti-coagulant poisons (e.g. warfarin) presented the next significant advance in rat control, allowing baits to be laid out continuously. The main period for rat control is the winter, when rats are concentrated around farm buildings or other places that provide food and shelter. Other than this, rats may be taken in the summer in Fenn traps or shot by gamekeepers.

Grey squirrel

Grey squirrels were introduced in the 19th and early 20th centuries, the earliest recorded introduction having taken place in 1876 in Macclesfield, Cheshire. Since then, their spread throughout the British Isles has been well documented. By the early 1960s, the only areas of England remaining uncolonised by grey squirrels were Cumberland, East Anglia, north Lancashire and Westmorland (Harris *et al.*, 1995). Usher *et al.* (1992) calculated indices from Forestry Commission Survey results and concluded that the distribution of squirrels in England and Wales is fairly stable, in contrast to a steady increase in Scotland. Lowe (1993) reported continued expansion of the range of the grey squirrel in north-west England with well-established populations in the Lake District up to Windemere and Ambleside. The increases in game bags reflect the findings of these studies of the change in the distribution of the grey squirrel. It is likely that the observed increases in East Anglia, northern England and Scotland result from the colonisation of a greater proportion of estates over time as well as increases in population density as squirrels have become established in newly-colonised sites.

Stoat

Stoat density and distribution are more closely related to prey availability than habitat *per se* (Harris, 1995), and stoat numbers may follow cycles in prey abundance causing marked variations in density between years. Stoat numbers are thought to have been severely reduced following the outbreak of myxomatosis. On one Suffolk estate, there was a ten-fold reduction in the number of stoats killed in 1960 compared to 1950 (Tapper, 1992). In the regions where significant declines in stoat bags were recorded, significant increases in rabbit bags were recorded, although in the regions where stoat bags increased, increases in rabbit bags were of a much greater magnitude. Fluctuations in prey abundance may thus explain some of the observed changes in stoat bags. Increases in fox bags may also be a factor contributing to observed declines in stoat bags; significant increases in fox bags occurred in regions where stoat bags declined and increased, although they were of greater magnitude in the regions where declines in stoat bags occurred. An antagonistic relationship between trends in fox and stoat bags might be expected for two reasons. Firstly, they may compete for prey resources such as rabbits and other small mammals. Secondly, through predation by foxes on stoats: increasing fox densities have been noted to cause a localised decline in stoat numbers through predation, sometimes even to zero (Mulder, 1990).

Weasel

The correlation between the local density and distribution of small rodents and weasels is well established (Delattre, 1984; King, 1980). The first outbreak of myxomatosis in the UK is thought to have increased the abundance of small rodents following a flush of vegetation released by the removal of grazing pressure by rabbits. This abundance of prey was accompanied by record catches of weasels on game estates (Sumption and Flowerdew, 1985) and a sudden reversal of the usual ratio (2-3:1) of stoats to weasels (Hewson, 1972). If increases in rabbit bags are indicative of increases in rabbit abundance, one might expect to see a reversal of this effect as rabbit grazing pressure has become more intense, suppressing the numbers of small rodents. The gradual recovery of the stoat, which accelerated after 1970, has been accompanied by a substantial decline in weasels, though not as yet to the pre-myxomatosis ratio (King, 1991).

4.6 Alert potential of NGC data

At a regional level, declines in weasel bags over 25 years could be detected with high statistical power (i.e. around 80% or more) for those regions with a large sample size of estates contributing bag data over the 25-year period. These include south-east England, East Anglia and north-east England for species recorded on predator sheets, and additionally north-west England and east Scotland for those recorded on game sheets.

Statistical power was estimated only for weasel in south-east England. The statistical power of the NGC is likely to vary considerably according to the species being analysed, so the power estimates for weasel may not be applicable to other species in this analysis. Differences in the variation in bags among species could arise because of underlying demographic differences: *r*-selected species will exhibit greater annual fluctuations in population size, which could be reflected in bags. They might also be caused by the variations in control methods or shooting that exist between species: species that are poisoned or killed by gassing (e.g. rats, rabbits) are likely to be recorded with a higher rate of error than species killed by trapping or shooting, because of uncertainty in the actual numbers killed. For example, the increased variation that is characteristic of bag data for rabbits and rats results in wider confidence intervals for these species and hence power is likely to be lower than for e.g. weasels and stoats.

Regional differences in bags may also exist because of differences in predator control or estate management policies. The predominance of different shoot types and their characteristic habitat may also affect bag variability, especially if some regions encompass a greater diversity of shoot types than others. The population status of particular species in different regions is another consideration: for those species whose UK distribution is currently expanding (e.g. grey squirrel), there is likely to be more heterogeneity in bag size in regions where the species is newly established and has only partially colonised them.

For some purposes it may be desirable to extract data from the NGC at a country rather than UK level. This can be easily achieved since the NGC regions do not transcend national boundaries. Pooling data from several regions in this way would increase the statistical power of estimating trends from the NGC data: further simulation analyses could be used to confirm whether this would allow smaller percentage changes, e.g. 10% over 25 years, to be detected at the country level. Power analysis could also be undertaken at this level for shorter time intervals than 25 years, for example to estimate the statistical power of the NGC to detect annual changes in bags. This may be useful where animals are being actively managed and it is necessary to have some measure of confidence in estimates of how numbers change from one year to the next.

4.7 Investigating the effects of variations in sampling effort

Log-linear Poisson regression adjusting for gamekeeper density

In general, trends for red fox, rabbit, brown rat, grey squirrel, stoat and weasel were similar whether or not an adjustment had been made for gamekeeper density. The sole exception was for fox in the south-west of England, which may be related to regional differences in land use and the prevalence of differing fox control methods. Reynolds and Heydon (2000) studied fox control methods and the impact of culling in three regions of the UK: mid-Wales, where hunting with hounds and gun packs are used to control perceived losses to foxes on sheep and poultry farms, the east Midlands, an area with mixed agriculture and intermediate culling, and west Norfolk (East Anglia), an arable area where culling by professional gamekeepers on large estates using a rifle and spot-lamp was predominant.

The general lack of a significant effect of gamekeeper density on bag size could be the result of a variety of factors that render it an uninformative measure of effort. In order to quantify the importance of variations in effort, it is necessary to have some measure of effort that has varied over the relevant period of time. The number of gamekeepers on an estate is likely to be relatively invariant over long periods of time, fluctuating only within a narrow range for a given estate and undergoing changes occasioned by contraction or expansion of the estate area, or by economic considerations. It is therefore a factor governed by decisions made at the level of individual estates, and it is unlikely that any systematic trend would be apparent at a regional scale.

Gamekeepers employed on adjacent estates may cooperate to share beats, especially to carry out activities requiring more than one person such as fox lamping. The number of gamekeepers per estate is therefore not necessarily a good indicator of the level of gamekeeping activity occurring there in terms of trapping or shooting effort. A large amount of variation is likely to exist between gamekeepers in terms of experience, site-specific knowledge, skill, and effort. These factors are also likely to vary over the employment duration of any individual keeper.

Effort is a composite of many different factors all of which are subject to variation. Concentrating on the number of keepers as a measure of effort fails to account for methodological changes and changes brought about by legislation, for example the trend towards increased prevalence of night shooting (lamping) for fox control over the last 35 years. A shift has occurred from spring and summer control using snares, cyanide gas and formerly gin-traps, to lamping using a rifle and spotlight predominantly in late autumn, winter and early spring when cover is reduced and visibility good (Harris *et al.*, 1995). Since this is the main dispersal period (Trewhella and Harris, 1988), shot foxes are likely to be rapidly replaced by itinerant animals immigrating from other areas, thereby allowing large numbers of foxes to be shot per unit area and creating the impression of a numerical increase. Taking all these caveats into account, a more satisfactory measure of effort would be the number of trap nights or lamping hours. Such a measure of effort would incorporate the method used to kill animals, number of persons employed for the task and total number of hours spent.

Log-linear Poisson regression of stoat adjusting for weasel and rabbit bags

The explicit consideration of effort in the analyses of large-scale trapping records has been emphasized by McDonald and Harris (1999). These authors suggest that the observed decline in the numbers of stoats and weasels trapped is equally consistent with a reduction in trapping effort as with a decline in stoat and weasel numbers. Based on stoat and weasel capture rates in 1997, they estimate that trapping effort would have had to have decreased by 45-75% to bring about the decline in stoat and weasel bags reported by Tapper (1992) between 1976 and 1996 (this is the decline in mean bags rather than bag indices from regression analysis). One cause for a downward trend in trapping effort over this period is the increasing reliance on artificially reared gamebirds for shooting rather than on wild birds.

McDonald and Harris (1999) found a significant difference in trapping effort between gamekeepers working on different shoot types, with a significantly greater effort made by gamekeepers on wild partridge and grouse shoots than by those working on reared pheasant shoots. Moreover, stoats and weasels were regarded as a more serious problem on wild grouse and partridge shoots than on reared pheasant shoots. Given the difference in effort between the two shoot types and the predominance of grouse moors in upland regions, one might expect to see greater declines in stoat and weasel bags in lowland regions of England than in northern England and Scotland. This is supported by the NGC data, with significant increases in stoat bags in north-east England and east Scotland, and significant declines in the Midlands and south-east England. Similarly, significant declines in weasel bags were recorded in all regions excluding the north of England and Scotland.

McDonald and Harris (1999) assert that there is no difference between the trapping effort made to catch stoats and weasels, since gamekeepers use the same type of trap for both species and set them in locations where all small predators are likely to be caught. Based on this assumption, and the regional differences in shoot type, one would expect the inclusion of weasel bags in the regression analysis to account for some of the declines observed in regions where the hand-rearing of game has become prevalent. McDonald and Harris (1999) conclude that there is little evidence from gamebag data to suggest that national stoat and weasel populations have changed more than expected on the basis of normal annual fluctuations. However, if the estimated changes in stoat bags remain significant after controlling for effort in the analysis using weasel bags, this may provide evidence for changes in the stoat population. It is more problematic to explain the significant increases in stoat bags in north-east England and Scotland by changes in gamekeeping practices.

When McDonald and Harris (1999) controlled for the effect of trapping effort, the number of weasels trapped by gamekeepers in 1997 was significantly lower in the south-west than in other regions of England. The estimated percentage decline in weasel bags from NGC data between 1975 and 2000 was also greatest in this region.

Of the five significant regional percentage changes in stoat bags estimated by the original log-linear Poisson regression model with site and year effects only, four were still significant when rabbit and weasel bags were included as additional explanatory variables; none of the differences in the estimated changes were significant. The directions of the changes that were observed were in agreement with the hypothesis proposed by McDonald and Harris (1999), that effort has progressively decreased in regions where there has been a shift towards increased reliance on hand-reared game, which has been more pronounced on pheasant shoots than on grouse or partridge shoots. If part of the estimated decline for these regions is attributable to a decrease in trapping effort, the effect of controlling for effort would be to decrease the magnitude of the declines or cause a reversal of the trend as has been observed for the regions in the south of the UK where pheasant and partridge shoots predominate (Tapper, 1992). However the results do not concur with the assertion of McDonald and Harris (1999) that there is little evidence for a decline in stoat bags, for when weasel bags were used as a proxy for effort, significant declines were observed in south-east England and the east Midlands.

In north-east England and east Scotland, a significant positive correlation was found between stoat and rabbit year coefficients after controlling for weasel year coefficients (Table 3.6.3). Adjusting the stoat year coefficients for variation in rabbit and weasel bags resulted in a decrease in the estimated percentage change in stoat bags. This suggests that a proportion of the observed increase in stoat bags is associated with an increase in rabbit bags, which is likely to indicate a numerical response by stoats to increasing rabbit density. If this increase were an artefact of increasing effort, one would not expect to see a significant increase in stoat bags when effort is controlled for.

4.8 Assessing the representativeness of NGC bag size indices as indices of abundance: comparisons with indices from other mammal monitoring schemes

The concordance of the indices from the NGC and BBS indicates that the annual variations in bags reflect changes in the abundance of red fox, rabbit and grey squirrel. The highest degree of concurrence between the indices from the two sources is observed for squirrel data. This is possibly because squirrels are conspicuous and hence easily sighted and counted by BBS observers. Where they are shot or captured by tunnel trapping the number of squirrels killed can be expected to be accurately reported in the NGC. The error bars around the annual indices provide a measure of the variability in the data; for BBS indices, these are considerably larger for fox than for rabbit and squirrel. This reflects the relative rarity with which foxes are sighted in the BBS. The sample size of BBS squares was 1040 for rabbit, 460 for squirrel and 227 for fox. Wide error bars for NGC rabbit bag indices are likely to be caused by inconsistency of reporting for this species due to the methods by which it is killed, since the sample size is larger than that in the BBS. Nevertheless, the numbers reported in the NGC will have been killed by the more opportunistic approaches of shooting and trapping rather than by the systematic one of gassing; as such, they probably provide a better reflection of abundance than would ones based on gassing. The indices for fox show the poorest correlation between NGC and BBS indices. This may result from the relatively low sample size for BBS fox indices. Where sample sizes are low for either the NGC or BBS, one would expect increased deviation of the indices from one another, owing to the geographical biases of the two schemes. The north of England and Scotland are under-represented in the BBS because of the reduced availability of volunteers in these regions; this could have an effect because of differences between population trends among regions, as evidenced by the significant interaction between site (region) and year effects.

Comparison of the statistical power of the NGC with other monitoring schemes

Macdonald *et al.* (1998) analysed data on brown hare abundance gathered in the Brown Hare Survey (Hutchings and Harris, 1996). For a total of 233 plots, the statistical power to detect an annual decline of 5% or more (equivalent to a decline of 72.3% over 25 years) was over 90%. Power to detect a decline of 2% annually was 49% and an annual decline of 1% was detected in 30% of simulations. These annual rates of decline equate to declines of 39.7% and 22.2% over 25 years respectively. The power estimates obtained for NGC weasel data over a 25-year period were comparable.

Although using numerical count data would extract the maximum amount of information on changes in mammal abundance from the BBS, considering the data as presence or absence data maximizes the available sample size. Statistical power analysis of BBS presence or absence data was carried out using the formula given in Sokal and Rohlf (1995, pp. 768-769) for the estimation of the sample size required to detect, with a given power, a specified difference (effect size) between two proportions at a predetermined level of significance. This approach was used for simulated declines of 25% and 50% (Newson and Noble, 2003). For an α value of 0.05, the current BBS sample size of around 1800 squares was found to have a power of between 60% and 90% to detect a 25% decline given a starting proportion of 0.15 or higher. This applies to brown hare, rabbit, grey squirrel, red fox, roe deer and mole. However, the BBS does not provide sufficient statistical power for cryptic species that are observed infrequently. The limited utility of BBS presence or absence data for the detection of shallow population trends from low probabilities of detection (below 0.2) is noted by Toms *et al* (1999). In contrast, the NGC would provide sufficient statistical power to detect declines of at least 25% for weasels, at country level and even at regional level in some cases, depending on the number of estates in a specific region. The power of the BBS to detect a 50% decline is 100% at the lowest starting proportion of 0.05; therefore a 50% decline could be detected by the current BBS mammal sample for 18 species, including rats, stoats and weasels in addition to those listed above (Newson and Noble, 2003). The power of the NGC to detect a decline of 50% in weasel bags was 100% for sample sizes of 60 or more estates.

5 Conclusions

The NGC has wide geographical coverage, especially in regions of the UK that are poorly covered by other monitoring schemes.

The NGC provides a valuable source of historic data that have been collected according to established protocols. The existence of an established network of contributors allows utilisation of data that are already available, thus greatly reducing costs.

The power for the NGC to detect changes in bag size is high in comparison with the power of many surveys designed to collect data on mammal abundance. Based on the premise that trends in bags reflect changes in abundance, the NGC represents a valuable tool for the surveillance of UK mammal populations.

The lack of a record of effort in the NGC is not perceived to be a major problem. Trends in bags were generally found to agree with indices of abundance from other sources; controlling for variations in effort did not alter the significance of estimated trends in stoat bags.

The NGC currently collates bag records for many species listed under national and international legislation including one UK BAP Priority Species, the brown hare. Species of Conservation Concern recorded by the NGC include roe deer, red deer, fallow deer, hedgehog, mountain hare, stoat, weasel and polecat. Hedgehog, mountain hare, stoat, weasel, polecat and deer species, all of which are recorded in the NGC, are listed under Appendix III of the Bern Convention (1979). Although no species listed in Schedule 5 of the Wildlife and Countryside Act (1981) is currently recorded by the NGC, hedgehog and polecat are listed under Schedule 6.

The NGC has a large sample size in comparison to other monitoring schemes for species that are difficult to detect by conventional survey methods such as sightings or sign searches. Macdonald *et al.* (1998) propose a role for the NGC in the monitoring of mammal species such as mustelids, which are better detected by game keepers than by direct sighting surveys because they are rarely seen and do not leave easily visible field signs (Corbet and Harris, 1991).

Recommendations

- Correction of anomalous grid references and effort to obtain references for estates that lack them. Use bag data to produce annual distribution maps for species whose geographical distribution is undergoing change.
- Correction of data on the number of gamekeepers to a level of error similar to that for game and predator bags.
- Recruitment of estates in NGC regions with comparatively small sample sizes and thus low statistical power.
- Continuation of comparisons between NGC bag indices and BBS indices on an annual basis; this could be facilitated by reassigning NGC counties to form the nine BBS regions, enabling direct comparisons to be made at a regional scale.
- Incorporation of an optional measure of effort into NGC forms. The number of shooting days might be used for game species, and the predator control method for predatory mammal species with the number of trap months, number of lamping nights etc. where applicable.
- Entry of rabbit data into the same database as used for predatory species, to allow easy extraction of rabbit data with data for predatory mammal species.
- Further investigation into the link between the number of hand-reared gamebirds on estates and trapping effort for stoat and weasel bags, using log-linear Poisson regression with site and year effects and the logarithm of the number of pheasants released on the estate (recorded in the NGC).
- Intensive studies on a small subsample of NGC estates would be valuable to establish how densities of mammalian species differ on land managed for shooting in contrast to the unmanaged countryside; estimates of density on estates could be used to quantify what proportion of the animals present is removed by shooting or trapping.

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