BTO Research Report No.223

Developing a mammal monitoring programme for the UK

M.P. Toms, G.M. Siriwardena & J.J.D. Greenwood
(Part III.A.1 written by S.N. Freeman & G.M. Siriwardena)

July 1999

A report by the British Trust for Ornithology under contract to the Joint Nature Conservation Committee (Contract No: F76-01-241).

British Trust for Ornithology,The Nunnery, Thetford, Norfolk, IP24 2PU.
Registered Charity No. 216652

M.P. Toms, G.M. Siriwardena & J.J.D. Greenwood
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Report to be cited as: Toms, M.P., Siriwardena, G.M. & Greenwood, J.J.D. (1999) *Developing a mammal monitoring programme for the UK*. BTO Research Report No.223. BTO, Thetford.

ISBN 1-902576-19-5

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*Talpa europaea**	Mole
Sorex araneus	Common Shrew
*Sorex minutus**	Pygmy Shrew
Neomys fodiens	Water Shrew
*Crocidura suaveolens**	Lesser White-toothed Shrew

Lagomorphs

Orcyctolagus cuniculus	Rabbit
Lepus europaeus	Brown Hare
Lepus timidus	Mountain Hare

Rodents

Sciurus vulgaris	Red Squirrel
Sciurus carolinensis	Grey Squirrel
*Clethrionomys glareolus**	Bank Vole
Microtus agrestis	Field Vole
*Microtus arvalis**	Orkney Vole
Arvicola terrestris	Water Vole
*Apodemus sylvaticus**	Wood Mouse
Apodemus flavicollis	Yellow-necked Mouse
Micromys minutus	Harvest Mouse
*Mus domesticus**	House Mouse
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Vulpes vulpes	Red Fox
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Mustela nivalis	Weasel
Mustela putorius	Polecat
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Mustela vison	American Mink

Meles meles	Badger
Lutra lutra	Otter
Felis silvestris	Wildcat
*Felis catus**	Feral Cat

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*Sus scrofa**	Wild Swine
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Cervus nippon	Sika Deer
Dama dama	Fallow Deer
Capreolus capreolus	Roe Deer
Muntiacus reevesi	Reeves' Muntjac
*Hydropotes inermis**	Chinese Water Deer
*Capra hircus**	Feral Goat
*Ovis ammon**	Feral Sheep

Marsupials

*Macropus rufogriseus**	Red-necked Wallaby

Key References

For convenience, we refer throughout the text to the following key references by short titles. The full references are given at the end of the report.

Short title	Reference
The Atlas	Arnold (1993)
The Handbook	Corbet & Harris (1991)
The Populations Review	Harris *et al*. (1995)
The Field Guide	Macdonald & Barrett (1993)
MMR	Macdonald *et al.* (1998)
Yalden	Yalden (1999)

PART I. AIMS AND OBJECTIVES OF UK MAMMAL MONITORING

The purpose of this part of the report is to set the scene for the rest. It does so in three somewhat disparate sections. The first begins with a discussion of what we mean by monitoring and of its role in conservation, which it is crucial to understand if one is to design good monitoring schemes. We go on briefly to review the status of British and Irish mammals, why we need to monitor them, and the broad aims of such monitoring.

The second section begins with a brief reminder of the current state of UK mammal monitoring. It summarises the proposals of MMR (see front of this report for main references) and then presents our responses to each of them, except for those for which more detailed discussion is presented in later parts of the report. Section 3 briefly considers how the monitoring is different from that of birds, to ensure that both the authors and the readers of this report do not make unwarranted assumptions. The authors, with largely ornithological rather than theriological experience, are particularly likely to do so. Furthermore, the methods of bird monitoring are so well-known and well-developed that even non-ornithologists may translate them to mammals unless they carefully consider the differences.

It is perhaps worth remarking here that, in Part I of the report in particular, we have generally used BTO examples to illustrate points being made rather than examples from other monitoring work. While we have no doubt that examples could have been found elsewhere, it has been more efficient for us to stick to the illustrative examples with which we are most familiar.

1. WHY SUCH MONITORING IS NEEDED

1.1 What is monitoring?

Monitoring is more than surveillance (the documentation of changes over time in the distribution or abundance of species). It has three additional features (Baillie 1990; Hellawell 1991; Furness *et al.* 1993; Greenwood *et al.* 1993; Greenwood 1999):

1. Objectives and targets
2. Contribution to understanding
3. Stimulating and underpinning action

Effective monitoring involves applying the surveillance to management objectives. For example, the UK Biodiversity Action Plan (Anon. 1995) has as its overall goal:

> *To conserve and enhance biological diversity within the UK and to contribute to the conservation of global biodiversity through all appropriate mechanisms.*

On the basis of such broad objectives, specific targets need to be defined - such as the UK Biodiversity Action Plan targets for mammals (Table I.1.1). We do not address such targets in this report but note the need not only for them to be established but also for that establishment to be based on wide consultation. This is because, for many species, there is likely to be a wide range of people with legitimate interests, such as farmers, foresters, naturalists and shooters; furthermore, both the extent of their interests and the balance that has to be struck between them are value judgements that should not be made in a democracy without open consultation. The difficulties in reaching decisions are perhaps illustrated by the fact that the Brown Hare Action

Plan envisages that the species' numbers are doubled by 2010; yet it is an introduced species that competes with a native species (the Mountain Hare), leading to an apparent conflict with both the Habitats and Species Directive and the Convention on Biodiversity (see below).

Surveillance provides the information to judge whether such targets are being attained. If they are not being met, the monitoring process should ideally: contribute to understanding of why not; contribute to the scientific advice underpinning remedial action; and provide alerts to those responsible for undertaking such action.

Fig I.1.1 identifies how monitoring supports management. It does so by contributing information and understanding that provide sound answers to the series of questions shown in Fig. I.1.1. Note that even if the answer is positive to the most basic question of all (which is whether the monitoring shows the target for the species to have been achieved), that is not the end of the matter; monitoring must be maintained (and the question constantly readdressed) in case of unforeseen changes. If the target is not being achieved, we need to ask whether we know why (question 2). There may be information external to the monitoring process that helps identify the reasons but if the monitoring programme has been well-designed, the monitoring alone may well provide the answer to that question. Whatever the answer, it will lead to a branching network of further questions and actions. Some of these must be informed by information from fields other than ecology or conservation (such as economics) or incorporate value judgements (such as the value people place on being able to watch Water Voles on their local brook) but all of them also require input of monitoring information. Note that the network is a closed one - all routes eventually lead back to the continuation of monitoring. This is essential, both to measure the effectiveness of deliberate management actions (such as legal protection) and to detect new problems (such as the impact of American Mink on Water Voles).

In the planning of a monitoring programme, it is important to remember that it is indeed a monitoring programme and not a research programme. Monitoring should, of course, contribute to our understanding of why targets are not being attained and of how management practices may be altered to improve the prospects of the population of interest. If too much ongoing research is built into the monitoring programme, however, this may divert resources from the more basic step of determining whether the population is being maintained at the target level and may result in much of the research not being focussed on the key issues. As with so many issues in respect of monitoring, striking the right balance is a matter of judgement.

1.2 The status of British and Irish mammals

The status and history of each species is summarised in Table I.1.2.

Further relevant information on ecology, life history, and behaviour is summarised for the 28 target species considered by MMR in their Appendix 1.1 and Table 1.4; for the other species it can be found in the Handbook and the Field Guide.

The conservation of a significant proportion of the UK's species of mammals is governed by domestic or European legislation (MMR, Chapter 1, Section 3). In more general terms, a very high proportion of species feature on the "long list" of the UK Biodiversity Action Plan (Anon 1995) (Table I.1.3), although, so far, there are species Action Plans for only a few of them (Table I.1.1).

1.3 Objectives of mammal monitoring

The Biodiversity Action Plan includes summary actions for monitoring the components of biodiversity, viz:

> *The Government and its agencies will:*
> - *examine and develop the integration of monitoring studies and seek to establish baselines for key components of biodiversity.*
> - *develop UK monitoring schemes to take account of threats and impacts on biodiversity.*
> - *develop thresholds for conservation action in relation to species population and habitat change.*

The UK Plan was a response to the Convention on Biological Diversity, Article 7 of 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 greatest potential for sustainable use.*
> - *maintain and organise by any mechanism data derived from such monitoring activities.*

There is thus an administrative and statutory requirement for monitoring key elements of biodiversity. The administrative requirement is, of course, only there to fulfil deeper objectives, which are hinted at by the Convention's stating that particular regard should be paid to species that are of social, cultural or scientific importance, that are threatened, or that are important for research into biodiversity and its sustainable use. Broadly speaking, there is an objective of conservation management, to which monitoring has important contributions to make. The species listed in the Action Plan (see Table I.1.3) must therefore be prime candidates for monitoring. Indeed the individual species Action Plans (see Table I.1.1) can only be undertaken sensibly if monitoring is in place for those species.

Conservation management also underlies the requirement under the Habitats and Species Directive to maintain or restore the favourable conservation status of nominated species (comprising, for the UK, Mountain Hare, Common Dormouse, Pine Marten, Polecat, Otter and Wildcat).

A high proportion of the UK mammal fauna is non-native (Table I.1.2). The Convention on Biological Diversity requires contracting parties to take action against alien species that threaten native species. Since all communities are invasible, since the success and impacts of invading species are unpredictable, and since those impacts on other organisms may be great (Williamson 1996), all aliens pose potential threats. Furthermore, the establishment of any alien reduces the component of biodiversity contributed by biogeographical differences (γ-level diversity in technical terms). Thus the conservation of biodiversity requires that aliens in particular are managed and hence (for the management to be soundly-based) that they are monitored, even if the management comprises doing nothing unless the alien population begins to grow to a threatening extent.

Mammals need to be managed (and therefore monitored) for reasons other than their own conservation. Many, at least sometimes, can cause significant economic damage. Some represent an economic resource to be exploited, such as the Red Deer that attract both tourists and hunters to the Highlands. Even more broadly, mammals are important because they may have ecological impacts that affect other species of animals and plants. Their impacts may tend to be greater than those of many other animals because mammals are individually large; compared with birds, they are, furthermore, numerous for their sizes (Greenwood *et al.* 1996).

The most developed wildlife monitoring in Britain and Ireland is that of birds (Greenwood *et al.* 1993; Greenwood 1999). One value of bird monitoring is that birds can be useful indicators of environmental conditions (Furness & Greenwood 1993). This is not so much because individual species indicate particular conditions but because whole suites of species can provide information on problems in particular habitats (such as the now well-known problems in farmland, Baillie *et al.* 1997) or about the effects broad-scale environmental changes (such as climate change, Crick *et al.* 1997; Crick and Sparks 1999). Mammals are less useful in this respect since there are fewer species and since it is impossible to apply a single monitoring system across more than a few of them. Furthermore, many mammal species are subject to direct control (or are recovering from past control), so their status reflects the direct impact of man rather than other environmental factors. Even so, some mammalian species may be useful indicators - Moles perhaps of ground invertebrates, Water Shrews and Otters of water quality, Dormice and Yellow-necked Mice of habitat fragmentation, and many species as monitors of climate change.

In contrast, some mammals are key prey for a range of predators (birds and other mammals) and some have major impacts on vegetation; they are significant components of ecosystems. Furthermore, many species are economically important as pests or as game.

Thus, the objectives of mammal monitoring are to provide management information in respect of individual species of conservation or economic significance and to provide a contribution to the ecological information that is needed for the conservation of biodiversity.

1.4 Aims of mammal monitoring

1.4.1 Information

Monitoring schemes should aim to provide information, not data. The data are simply the means to the end, which is the information about changes in status in relation to target and about the likely reasons for those changes. There are two reasons why we need to be clear on this point. The first is that the monitoring needs to include provision not just for the collection and curation of data but also for the interpretation of the data and the reporting of the results, i.e. turning the data into information.

The other reason for making clear the need for information rather than data is that the data can be used for purposes additional to the provision of monitoring information. They may be used for other conservation purposes. They may be used for scientific studies. They may be used for work of a commercial nature that not only provides employment for staff of the organisation doing the work but also produces surplus monies that can help support the other work of the organisation. They may simply be sold (though in BTO's experience, while some data are commercially valuable most are not and the overall cost of a data-provision service is greater than the income it generates). Mammal monitoring in the UK is likely to involve a number of different organisations (and very many different individuals). For their collaboration to be effective, they will need to recognise not only that their roles in the monitoring scheme are different but that so may be their interests in subsidiary uses of the data. Those who need the conservation information but not the data need not insist on access to the latter.

1.4.2 Geographical scope

Mammal monitoring must aim to provide information at UK level. It is also desirable that information is available at national level, i.e. separately for England, N. Ireland, Scotland and Wales, not only because this is the level of government that is mainly relevant but also because the ecological circumstances of the four countries are generally different. It would not, however, be cost-effective generally to run separate schemes in the four countries. Not only would the cost of doing far exceed the cost of running a UK scheme but insights would emerge less readily from separate analyses at the country level than from integrated analyses of the data that build on differences that there may be between the countries. (As examples of the latter approach from BTO studies, we can quote the understanding we are getting of changes in Welsh wintering wader populations from analyses at the UK level and insights into farmland bird problems that come from comparisons between east and west Britain). Thus, UK monitoring needs to cover each of the four countries adequately but through common rather than separate schemes.

The provision of information at the UK level is a broad aim that does not necessarily mean that the immediate focus of attention should be the UK population rather than local populations. Some mammals have such disjunct populations that it is not biologically meaningful to speak of a national population; greater insight is gained by focussing on the separate local populations. It may even be convenient and effective to use different methods for monitoring such separate populations. Nonetheless, for administrative and practical purposes it is important that the local information is drawn together at country or UK level. Furthermore, ecological insights may be gained from comparisons between populations in different regions. It is therefore important, even for these species, that the local monitoring fits into a broad national framework. It will generally be helpful if similar methods are used in different regions, though we acknowledge that there may be reasons for using different methods that override this general guideline.

Monitoring information is needed at a local level in some cases - for the management of deer populations in specific areas, for example. The uses of such information may be specific to each locality, so the information required may also be specific. For this reason and because the detailed monitoring required in these places is not required nationwide, it is not appropriate to conduct nationwide monitoring at the intensity sometimes needed locally. This is not to say that local intensive monitoring cannot feed into the national programme; it would be wasteful if it did not.

Mammal monitoring will produce site-based data. These data are valuable for local planning purposes. For example, although the whole country is not covered by Badger Groups, the National Federation of Badger Groups reports that records of Badger setts influence well over 100 planning cases a year in Britain (Pat Williams, in Sargent & Morris 1997). Although monitoring schemes should be set up in ways designed to best fulfil the monitoring aims, we recommend that consideration should be given to how the data may be made available for site-based work, the value of doing so, and the additional cost involved. This is likely to involve The National Biodiversity Network (see Part VI).

Northern Ireland is a special case. It is politically important for monitoring there to fit in with that in Great Britain, so that a UK overview is possible. Equally the management of Northern Ireland's animals will most effectively take place in an all-Ireland context. It would therefore be useful if the monitoring within the two administrations were comparable (just as, in the bird world, the Irish Common Birds Survey uses identical methods to the UK Breeding Birds Survey and there is close co-operation between the Irish Wetland Birds Survey (I-WeBS) and the Wetland Birds Survey (WeBS)). We recommend that contact be made with Irish agencies at an early stage, in the hope that at least some of the monitoring that will be established (or continue) in UK may be extended to (or parallelled within) the Republic of Ireland.

Northern Ireland also has its own landscape characteristics and culture that may mean that monitoring schemes that are effective in Great Britain are less appropriate there. Time has prevented us paying much attention to this issue but it needs to be addressed when plans for particular monitoring schemes are being drawn up.

1.4.3 Habitat information

The interpretation of population data may be considerably enhanced if relevant habitat data are available, especially if they are gathered at the same time as the population data. We recommend that this be considered for any mammal monitoring scheme, although the advantages of gathering habitat information need to be weighed against various possible disadvantages. These could include putting off volunteer data-collectors and the greater costs of gathering, collating, processing and analysing the data.

An important question is whether to use a common habitat classification for all species or to use different classifications, appropriate to each species. This issue must be explored when decisions have been taken about what mammal monitoring is to be undertaken. (To aid integration, BTO now uses a common habitat classification across most of its schemes (Crick 1992); however, because many bird species can often be covered by a single method, BTO runs relatively few schemes). Whatever new habitat classifications are used for mammal monitoring, it is important that they are as compatible as possible with existing classifications, especially those particularly relevant to the species in question (e.g. The Environment Agency's *River Habitat Survey* scheme, for riparian species). The key aspect of compatibility is the mapping of classifications on each other: one-to-one, one-to-many, and many-to-one mappings are useful; many-to-many mappings are not.

1.4.4 Monitoring range

The collation of distribution records in the past has been valuable in demonstrating changes in range, which have focused conservation attention on problem species. However, with important exceptions (see below) we agree with MMR that biological recording as currently practised provides only approximate information on range, especially on range change, mainly because there is little or no control over the amount of effort devoted to it. Even the breeding bird atlases, which have both managed to obtain almost complete coverage over just a few years, have to be used with caution as sources of information on change in range (Gibbons *et al.*1993; Greenwood *et al.* 1997; Donald & Fuller 1998). It will be possible to assess such changes better in future because the second atlas used a more formal protocol that can be used again. The success of this approach depends, however, on the ease with which birds may be detected and the great number of observers available to take part in the fieldwork. It seems unlikely that such complete coverage could be achieved for mammals, although useful distributional surveys of some species (especially the more apparent ones and those of more restricted distribution) could be achieved. There are some mammals for which it is important to monitor range but, in general, we do not believe that it would be wise to devote resources to the promotion of biological recording that could be used to mount focused schemes to monitor abundance.

Fortunately, monitoring of range generally adds little to monitoring of abundance (providing the latter is achievable). Indeed, the evidence from birds is that overall abundance is more sensitive to changing conditions than is range (Donald & Fuller 1998). This is a logical consequence of the tendency for there to be a correlation between mean abundance at the sites that species occupy and the number of sites occupied (see, e.g. Holt *et al.* 1997). Since national abundance is the product of local abundance and range size, range will necessarily be correlated with national abundance. The latter is thus generally a better, and largely sufficient, measure of status.

These considerations do not preclude all monitoring of ranges. Indeed, for species that are (or could be) rapidly expanding their range, monitoring the extent of the range may be significantly more sensitive, and therefore more important, than monitoring total abundance. For some other species, range may be so much easier to measure than abundance that it is better to monitor range. For yet others, useful information on range may be obtained at little extra cost during the monitoring of abundance. Furthermore, biological recording may provide a useful stop-gap if there are species for which formal monitoring schemes are unlikely be set up in the foreseeable future but this can only be assessed after decisions have been taken about what schemes should be set up.

Recent work by Chamberlain *et al.* (1999) is relevant to the interpretation of changes in range. They found that, while changes in the abundance of declining farmland birds had been most marked in regions of predominantly arable agriculture in England and Wales, local extinctions (losses of birds from 10x10km squares) had been most common in regions of predominantly pastoral agriculture. They proposed two explanations for this, both predicated on pastoral areas having been less suitable for the species in question, not just in recent years but form the start of the period that their study covered. The simpler explanation is that the abundance of these birds has declined in all areas but that this has led to local extinction more in the pastoral areas because the birds were initially less abundant there. Under this scenario, reductions in range would be a sensitive indicator of problems which affect entire ranges. (It would, however, be misleading to associate the range losses closely with the features of the squares from which species were lost: such features would at most be correlates of low abundance rather than a local extinction). The second possibility is that at the start of the study period the pastoral areas were already so poor for these species that their populations there acted as sinks, maintained only by immigration from the comparatively better arable areas. As conditions worsened in the latter the supply of immigrants to the pastoral areas dwindles and the species therefore disappeared. Under this scenario, the losses of range in the pastoral regions depended on conditions elsewhere, making them poor indicators of problems locally. Thus the interpretation of changes in range under either scenario is not straightforward, a further reason for generally concentrating on changes in abundance where possible.

1.4.5. Monitoring abundance

The prime aim of monitoring the status of UK mammals is thus to monitor their abundance. Whether one conducts a complete national census or estimates national abundance by taking samples, there are various measures of abundance (Sutherland 1996). Most obviously, one can count the animals directly. Alternatively, one can count things that are correlated with the number of animals, such as the number of nests, droppings, footprints, or calls, to obtain an index of abundance (Greenwood 1996). If one needs an estimate of the numbers of the animal, such indices must be calibrated (see Greenwood 1996 for further details). For monitoring, where one is mainly interested in relative change in numbers rather than absolute numbers, calibration is unnecessary; all that one requires is that the relationship between the index and numbers should not vary systematically with time, should be monotonic and, indeed, should be approximately linear. Linearity is the condition most likely to be broken. Non-linearity will only have a significant practical effect if it is marked. It can be checked and corrected for by calibration, though it is not necessary that the calibration is carried out before the monitoring is instituted. We recommend that monitoring programmes should be set up as a matter of urgency, with calibration conducted later if deemed necessary. Even without calibration, indices that are not strictly linearly related to absolute numbers can be useful, so long as one bears in mind their likely non-linearity (Greenwood 1996). A somewhat non-linear index that is affordable is preferable to a strictly linear one that is too expensive to measure with the resources available.

The relationship between an index and actual numbers may differ between habitats. This presents no problem if population changes are the same across habitats but if they are not then simply combining the data from all localities will give an estimate of national population change that is biased towards the change in the habitat that has the index that rises and falls more steeply in relation to population change. In these circumstances, it is better to calculate habitat-specific rather than national indices.

Indices are often preferable to absolute counts because they are much easier to obtain. Similarly, indices that are less accurately related to absolute numbers may be easier to obtain than more accurate indices. In practice, one must choose the method that provides the information at the level of accuracy required for the least expenditure of resources. Thus the Breeding Birds Survey will provide better monitoring of widespread species in the UK than the Common Birds Census has done, despite the greater accuracy of the mapping method used by the CBC relative to the line transect method used by the BBS. This is because this greater accuracy is achieved at the cost of much greater effort, so it is possible to sample only about one-tenth of the number of sites in the CBC than is possible in the BBS, resulting in lower precision of the national CBC index. An even more dramatic contrast is provided in the Republic of Ireland, where there were insufficient resources to mount a CBC-like scheme but where a BBS-like scheme is now successfully established.

Gibbons *et al.* (1993) used frequency of apparent occurrence (for example, the proportion of 2x2km squares within each 10x10km square) as a measure of abundance, having found good correlations between this index and direct counts for a wide range of bird species (Gibbons 1987). Such an index is not useful for species that are so common that they are found almost everywhere (though this problem can usually be overcome by using smaller units within which to record presence or apparent absence, like 1x1km rather than 2x2km squares). Frequency of occurrence is also not useful for highly aggregated species, since these may vary much in abundance while continuing to occupy the same number of sites. Nor is it useful for species that are so difficult to detect in a locality (because they are either scarce or cryptic) that they are detected in few places; this is because there will be considerable statistical error in measuring year-on-year change (Box 1). There is no such problem when the species only occurs in relatively few of the subunits but is readily detectable; in that case, the index of frequency of occurrence will be low but not subject to great sampling variance.

In using presence/absence records as indices of abundance it is important to bear in mind the problems of interpreting changes in range (subsection 1.4.4, above). These problems generally do not affect the interpretation of the proportion of occupied sites in an area as being an index of average abundance in that area (they may even strengthen it). They do, however, indicate the need for caution in interpreting losses at individual sites or comparisons of losses between suites of sites.

Box 1

Monitoring abundance by changes in proportion of subunits in which a species is detected: The importance of detectability

Consider a species that occurs in every one of 10 subunits in a study area. Suppose that its detectability is 50% of the subunits every year (on average). If in two successive years its abundance has not actually changed, there is a 47% chance that the ratio of the numbers of subunits in which it is recorded in those two years will lie between 0.75 and 1.33.

In contrast, if the detectability of the species were only 10%, then there is only a 12% chance that the ratio for two successive years will lie in the range 0.75-1.33. Indeed, there is a 22% chance that the species will be recorded in one or more subunits in the first year but none at all in the second (and conversely), suggesting substantial changes.

If counts of absolute numbers are attempted, the methods must be strictly defined in order to avoid bias. If bias is inevitable, then it should be kept constant by the adoption of equally strict protocols, so that the biased count obtained can be taken as a good index of actual numbers. Indeed, all index methods require strict protocols in order to avoid a drift in standards through time; such drift could create apparent trends in the population being monitored or mask real trends. For example, if animals are detected by spotlighting at night then it is important to control the specification of the spotlights; if detectability requires a particular level of skill, then attention must be given to observer skills. Problems of drift in standards are most likely when the monitoring is as crude as simply asking people to score the abundance of a species in their locality on some scale from absent to extremely common or (rather more useful in terms of monitoring) to score its recent population changes in some equally subjective way. Nonetheless, even such methods can be useful if more rigorous ones are unavailable, as shown by surveys of Rabbit infestation on Scottish farmland (Kolb 1994) or of the trends in bird populations across Europe (Tucker & Heath 1994).

The validity of crude indices may be tested by checking a subset of the data against parallel work by specialists, particularly if the specialists use more rigorous methods (and if they work in the same places). For example, nationwide monitoring of a species might be based on a combination of rigorous (but expensive) studies at a few sites and cruder (but cheap) surveys at many sites. Alternatively, one may use more than one simple index method; if the biases of the different methods are different and unlikely to drift in the same way, they provide checks on each other. These two forms of cross-checking are useful if the different types of survey suggest similar trends. Unfortunately, if they do not then one may have no way of deciding which to believe.

Cost aside, unbiased estimates of absolute abundance are preferable to indices. For this reason, those contemplating population monitoring may favour the use of Capture-Mark-Recapture methods. Unfortunately, estimates so obtained are extremely sensitive to the assumptions of the underlying statistical models and these assumptions are generally unrealistic. This means that, unless it can be proved that Capture-Mark-Recapture estimates for a particular population are accurate (by checking against intrinsically more reliable estimates), these estimates themselves must be treated just as indices. We are not aware of any reason to suppose that they are better indices than the simple total number of animals captured. Since the latter is more transparent and requires no calculation beyond counting, we recommend against the use of Capture-Mark-Recapture unless there are specific and demonstrated reasons for using it in particular cases.

1.4.6 Demographic rates

BTO's Integrated Population Monitoring programme involves the monitoring of demographic rates, not just numbers, particularly the productivity of individual nests and the survival rates of fully-grown birds; as a result, it provides deeper insights into the causes of population changes (Baillie *et al.* 1997; Crick *et al.* 1998; Greenwood *et al.* 1993). However, this depends on some of these rates being relatively easy to measure in birds, on a large and experienced network of nest-finders and bird-ringers, and on 60 or more years of development. We recommend that the monitoring of demographic rates is not initially taken as a primary aim of the mammal monitoring programme. Nonetheless, attention should be paid to such information when it arises from local monitoring work (as it often does in the case of deer, for example) and demographic data should be obtained where it is easy and cost-effective to do so in the course of monitoring of numbers (for example, the reproduction data gathered in the National Dormouse Monitoring Programme).

2. THE BACKGROUND TO THE CURRENT PROJECT

2.1 Current mammal monitoring in the UK

MMR give a full account of current monitoring. We do not even attempt to summarise it here, as the picture is so complex. Its chief features are:

1. *Many different organisations are involved.*

   There is very little national co-ordination, sometimes not even between bodies working on the same species or between projects funded by the same organisations.

2. *Not all species are covered*

   Even for species that are covered, cross-species co-ordination is not common.

3. *Some of it is not at the UK, GB or all-Ireland scales*

   Some projects are restricted to single countries of the UK or to only one administration in Ireland; others are restricted even more, to just a few much smaller areas.

4. *Much is not annual*

   Some is at longer than annual intervals but fairly regular but much is irregular - including some schemes that are apparently set up to provide regular monitoring, which may run for a few years but then close down.

5. *Level of refinement varies widely*

   At one extreme, irregular questionnaire surveys of respondents' opinions on status (which provide little hard information); at the other, careful estimates of national population size (which may provide more than is strictly necessary for monitoring).

6. *Reporting is disparate and scattered*

   Some surveys result in detailed and comprehensive reports; others in no publication at all. Each project reports separately. The reports appear in a variety of different publications.

7. *The role of volunteers varies widely*

   Given the diversity of methods appropriate to mammal monitoring, this is inevitable; but there are some species where volunteers could contribute but do not currently do so.

2.2 MMR proposals

Note that our responses to these proposals are given in Section 2.3, using parallel sub-subsections for ease of cross-reference.

2.2.1 What species?

MMR were asked to consider only a restricted list of species - those without asterisks in the table of species at the front of this report.

2.2.2 What is to be measured?

MMR propose particular methods suitable for individual species. Many of them involve the use of indices of relative change but there is also considerable emphasis on estimating absolute population sizes where possible.

2.2.3 Integration across species

MMR propose a scheme (the MaMoNet) in which all species are monitored in the same 10x10km and 1x1km squares of the OS grid.

2.2.4 The three-tiered structure and the Master Squares

To maximise effectiveness, MMR propose a three-tiered structure for the MaMoNet: Master Squares, Rich Interest Kilometre Squares, and Focus Zones.

The Master Squares form the core of the MaMoNet. They are 10x10km OS squares. On the basis that the sampling intensity used in the Water Vole and similar surveys has proved satisfactory, MMR propose that 18% of the squares should be sampled. The sample squares would be chosen at random.

Within each of the Master Squares, five 1x1km squares would be the basic survey units. Within each, methods appropriate to each species would be used to measure numbers or an index of abundance. For example, using a standard protocol, a 1x1km square could be completely searched for Badger setts, whereas one might search only a fixed length of river-bank for Water Voles. If other than a complete search is conducted, MMR recommend placing the actual survey location randomly within the square (or within suitable habitat).

To improve effectiveness, MMR recommend that the location of the 1x1km survey units is stratified. Stratifications suggested are: habitat (e.g. woodland, riparian, rough grassland, other), historical versus new survey sites and random versus systematic components of the Master Square sample (see 2.2.10).

2.2.5 The seven-year cycle

MMR recommend that the monitoring takes place on a seven-year cycle. In the first two years, surveys would be conducted of woodland squares (using methods appropriate to that habitat and to the species within it); the next two years would be devoted to riparian sites; the next two to other sites; the seventh to intensive analysis and writing up of major reports. “Mandatory rapid turn-around of customised feed-back material from the Central Office to the field workers and collaborating organisations” would take place throughout the cycle.

2.2.6 Rich Interest Kilometre Squares

These would be sites that have been surveyed in the past (particularly those covered in formal nationwide surveys) but which do not fall within the random sample of Master Squares. MMR propose that some at least of these should be surveyed as part of MaMoNet, to provide comparison with the past.

2.2.7 Focus Zones

The third tier of the MaMoNet would comprise sites where intensive investigations should be carried out. MMR suggest two sorts of Focus Zones:

intensive studies of local populations, providing detailed ecological understanding;

more intensive surveys than provided by the network of Master Squares in regions of particular significance, such as the region bordering the current main distribution of the Polecat (into which the species may be expected to spread).

As MMR remark, Rich Interest Kilometre Squares and Focus Zones grade into each other, especially in respect of the class into which historical study sites should be placed.

2.2.8 Manpower and costs

MMR envisage a professional team of staff, operating in two groups: a central office staff of three scientists and managers (plus a half-time secretary) and a peripatetic team of five full-time and five part-time fieldworkers (who would also do much of the data entry).

In addition, MMR envisage volunteer participation in the work of MaMoNet, suggesting two sources in particular: unemployed graduates, vacation students, and gap-year students; Mammal Society members and others recruited through 'The Look Out For Mammals' and similar projects. However, they conclude that: "for the foreseeable future professional Mammal Monitors comprise the metaphorical skeleton of the MaMoNet, to which ever more volunteers add increasing muscle bulk". They suggest that professionals would mainly cover the Master Squares, with volunteers being used mainly for RIKS.

Finally, MMR suggest an Advisory Panel to provide input from all the collaborating parties and from technical experts.

Box 2 shows the cost of the MaMoNet, based on figures provided in MMR.

2.2.9 Utilising existing and ancillary schemes

MMR consider how periodic national surveys (such as the Water Vole survey) could be moulded into the MaMoNet (see below). They appear to envisage, however, that existing monitoring effort will contribute only in so far as the personnel might be redeployed to participate in MaMoNet - e.g. stalkers who currently contribute to the various deer-monitoring programmes could carry out deer surveys in Master Squares.

MMR appear to see little role for data from ancillary schemes, such as the game bags surveys and the mammal records from the BTO/RSPB/JNCC Breeding Bird Survey.

Box 2

Costing the MaMoNet

We have used the figures quoted by MMR to come up with the following annual expenditure. We have inflated the MMR figures by 6%, to turn them from 1997 values into 1999/2000 values. All the figures are in £000s.

Capital Equipment[1]

Computing hardware	21	
Computing software	21	
3 caravans	22	
3 towing vehicles	48	
2 vans	15	
Field computers and software	11	
Other equipment	21	
Total per annum		**40**

Staff

Statistician	35	
Computer manager	29	
Outreach officer		29
Secretary (half-time)	8	
2 senior fieldworkers	57	
3 fieldworkers	70	
5 part-time fieldworkers	40	
Overheads[2]	25	
Total		**293**

Consumables (excl. vehicle costs)

Communications	1	
Reports[3]	3	
Total		**4**
Total of the above		**337**
Fuel and other running costs	28	
Advisory panel	5	
Grand total cost per annum	**370**	

Notes

1. The figures given for capital equipment are, except for the total, the initial expenditure; the total is a per annum figure, based on writing off the capital expenditure over 4 years, which is a realistic accounting procedure in our experience. (MMR do not give a write-off figure).
2. £8k per member of HQ staff (excluding secretary).
3. Average of three reports at £3k and one at £10k over 7-year cycle.

2.2.10 Setting up the MaMoNet

MMR propose that the first seven-year cycle be regarded as an Exploratory Cycle, in that results obtained during it should be used to modify the methodology.

They also propose that, to provide continuity with existing surveys, the Master Square sample is not randomised at the start of the MaMoNet scheme. Rather, the first cycle would use the "QQ grid" that was used in the Water Vole survey. (The QQ grid is a systematic grid of 10x10km squares, with randomly located 1x1km squares within it). Subsequently, a proportion (e.g. 20%) of the QQ squares would be randomly substituted in each cycle by the same number of squares drawn at random. Once the substitution had reached a certain stage (e.g. 60% of the squares being random), there would be no further substitution, so that long-term continuity would be conserved.

2.2.11 Sampling with partial replacement (SPR)

MMR recommend that the random Master Squares should neither all be reselected during each survey cycle nor all be retained in perpetuity. Rather, they recommend that most are retained from one cycle to the next but that there is a 20% (for example) turnover in each cycle, with 40% retained in perpetuity.

In respect of 1x1km survey squares within each Master Square, MMR recommend that the same ones are used throughout the period for which that Master Square is retained in the MaMoNet sample.

2.3 Aims of the current project

2.3.1 General aims and species to be considered

The MaMoNet is a carefully thought-out scheme that would deliver high quality information on Britain and Ireland's mammals were it to be implemented. It is, however, expensive and it is in the form of such an integrated package that it is not clear how it could be modified (say by dropping some components, by relaxing statistical rigour, or by making more use of ancillary data) to be less expensive but still provide effective monitoring. We have been asked to provide a series of costed options that would allow the conservation authorities to pick out a set that would provide them with what they believe to be the most cost-effective mammal monitoring that would satisfy their needs.

We have also been asked to consider all the mammals living at liberty in the UK, not just the restricted list that MMR has asked to address.

In the rest of Section 2.3, we briefly review the design issues raised in Section 2.2 (using corresponding subsection numbers, for ease of cross-reference). More detailed considerations of the statistically formal design are covered in Part II and of volunteer input in Part V.

2.3.2 What is to be measured?

The bedrock of the monitoring must be the surveillance of status - primarily abundance but range in certain circumstances (Sections 1.4.4 and 1.4.5. above). Surveillance does not, however, require absolute measurement of abundance, merely measurement of relative change, which may be easier (Section 1.4.5). As MMR demonstrate, measuring absolute abundance allows one to build spatial, habitat-related models of mammal distribution (using the powerful tool of GIS if appropriate). This is not the chief aim of monitoring but is a useful piece of associated research that may help illuminate some of the key questions that arise if management targets are not attained (Fig. I.1.1). If absolute abundance can be measured with little more effort than measuring relative change it is therefore worth measuring it. Otherwise, it may be better to concentrate on relative change and to mount special surveys or research projects when the monitoring reveals problems.

Absolute abundance estimates are also required for Population Viability Analysis, also considered by MMR. We are unconvinced of the value of PVA for the conservation of the native mammal fauna of the UK, given that long before a species had declined to such an extent that it was threatened by the stochastic processes to which PVA is relevant (demographic stochasticity, loss of genetic variation, environmental catastrophe) we would expect the conservation authorities not only to have been alerted (by the monitoring) but to have taken the necessary measures to have reversed the decline. Furthermore, the history of various mammals in Britain, which have recovered from low numbers or expanded from very small numbers of introduced individuals, leads to some doubt as to the applicability of the available PVA models. We note that MMR consider that one of the primary roles of PVA "will be for providing insight into the relative benefits of alternative population or habitat management strategies". We believe that direct studies of the effect of habitat management on the species in question are more likely to provide useful guidance.

2.3.3 Integration across species

It can be useful to monitor different species in the same sites, as one can then conduct site-by-site analyses of the influence of species on each other (e.g. the demonstration that Sparrowhawk *Accipiter nisus* and Magpie *Pica pica* predation is not generally the cause of declining songbird populations in Britain, Thomson *et al.* 1998). Indeed, developing a series of representative 1x1km squares to provide a general sampling framework for the UK has been a subject of much discussion. There are potential practical benefits to using the same squares for all species. For example, permission has only to be sought once; the corresponding disadvantage is that a landowner who is faced with (or fears being faced with) people coming back time and time again to monitor one thing after another may be more inclined to refuse access than one who is asked for permission simply to monitor one species. Another potential advantage is that the fieldworkers will become familiar with the area, which may be advantageous; conversely, they may get bored and either do the work less well or not do it at all.

Using the same places for different species has the particular disadvantage for British mammals that several species have already been subject to national surveys using different sample sites; to move them all onto a common series of sites would lose the considerable advantages of historical continuity of sample sites (see Part II). A more generally practical problem is that different species may require sampling on different scales (e.g. 40x40m trap grids for mice, 10x10km squares for deer) or in different habitats (e.g. riparian, reedbed and woodland specialists). Further, the optimal pattern of sampling over the UK will differ between species; Badgers, e.g., may need to be sampled everywhere, whereas Polecats need to be sampled intensively on the margins of their range, perhaps less intensively within their core range and scarcely at all in the rest of the country. Finally, landowners may change management practices if they perceive that their land is in some way special, thus rendering it no longer an unbiased sample of the countryside.

For these reasons, we would not generally support using the same sample locations for all mammals, though we acknowledge that there may be value in having common locations for species that are likely to interact directly.

2.3.4 The Master Squares

We take up issues of sample distribution, intensity of sampling and stratification in Part II.

2.3.5 The seven-year cycle

A seven-year cycle is too long for smaller mammals, which may show considerable short-term fluctuations. If the number of Weasels is 20% lower in 2010 than it was in 2000, how do we now that this is a long-term decline rather than a short-term hiccup, perhaps caused by peculiar weather? We need annual monitoring both to be able simply to distinguish long- from short-term changes and to be able to estimate the effects of weather and similar variables (Baillie 1990).

It is true that, if a change can be assumed to be constant, then the most effective way of measuring it is to study large samples of sites at long intervals rather than just a few sites every year, but the assumption of constancy flies in the face of our knowledge of the natural history of many British animals. We recommend that monitoring should be annual for species that are likely to show short-term fluctuations.

There may sometimes be other reasons (such as keeping a team of observers together) that favour annual surveys. Equally, for some species, practical reasons may make it desirable to undertake major surveys at infrequent intervals rather than to attempt annual surveys. If so, this has to be considered alongside the desirability of annual monitoring and a decision reached about the relative cost-effectiveness of different frequencies of monitoring. This will be a species-specific (or, at least, scheme-specific) decision. To impose the same frequency of survey on all species would be inefficient.

Note that the interval between surveys may be adjusted in the light of experience. Where there is doubt about the appropriate interval, it is perhaps best to err on the side of caution starting with a relatively short interval at first and extending it if experience shows a longer interval to be permissible.

In working on this project, we have been alert to the possibilities of hybrid approaches, i.e. major surveys at long intervals combined with smaller-scale annual surveys, the former to give long-term precision, the latter to provide information on short-term variation.

2.3.6 Rich Interest Kilometre Squares

Our considerations of sampling design (Part II) cover these.

2.3.7 Focus Zones

We recommend that special intensive study sites should be included in the monitoring scheme only if monitoring is the prime focus of the study. If the prime focus is demographic or other ecological investigation, the methods may not be appropriate for monitoring. It would not usually be cost-effective to constrain the methods of the intensive study in order to fit it into the monitoring scheme (or vice versa).

In contrast, we support the suggestion that monitoring may need to focus on particular geographical regions for some species.

2.3.8 Manpower and costs

We consider that the cost figure derived from MMR (Box 2) is only *c*.80% of the true cost of employing the number of staff involved, despite the salaries proposed by MMR for fieldworkers being higher than we believe is necessary. The reason is that MMR propose overhead rates that are unrealistically low - only 26% even if everything but the direct costs of running the vehicles and the Advisory Panel costs are excluded. This is substantially less than is charged by any organisation we know that has to cover its costs, as well as being substantially less than government guidelines to universities. In addition, we believe that MMR have underestimated the true cost of running the vehicles.

Note, furthermore, that we have made no attempt to estimate whether the man-power estimates listed by MMR are correct but we suspect that they are too low. We note, in particular, that their seven-year cycle involves two years' fieldwork on riparian habitats, two on woodlands and two on other habitats; yet the latter comprise the bulk of the country and may be expected to demand far more fieldwork.

We consider other ways of monitoring mammals, particularly the use of volunteers, later in this report. These have significant impacts on costs.

2.3.9 Utilising existing and ancillary schemes

Those conducting existing or ancillary schemes may have their own reasons for continuing to conduct them in much the same way as they do at present. For example, The Deer Commission for Scotland has a specific remit, for which it needs to conduct its current surveys of deer; it is presumably unlikely to abandon its current surveys in order to participate more directly in a UK monitoring scheme. The information from its work is, however, considerable and it would be unwise not to use it as part of the overall UK monitoring of deer. It may not be easy to combine it with information from other schemes, such as these conducted by the Forestry Commission, by The British Deer Society, by MoD, by BASC etc. (and then with information from any new scheme designed to cover the gaps); and the combination of such disparate sets of information may not provide a simple index of the state of the nation's deer. Nonetheless, bringing all this information together will be more cost-effective than to base our monitoring on some new scheme that ignores all these continuing sources of information.
Of course, even greater cost-effectiveness may be achieved if it is possible for existing schemes to modify their procedures with the aim of improving their contribution to the national monitoring programme.

2.3.10 Setting up the mammal monitoring programme

We agree with MMR that the early stages of any new programme should be regarded as exploratory. In terms of methodology for individual schemes, we urge that such exploration occurs as quickly as possible, so that the work can settle into long-term protocols that deliver reliable monitoring within a very few years. More broadly, we advise that it would be inadvisable to set up the entire suite of desired monitoring schemes all at once. There are three reasons for this. First, some new schemes (and, indeed, some continuing schemes that require substantial modifications to existing practices) will probably demand more resources at first than they need later, especially if volunteer-based (because of the effort needed to recruit participants). Second, some schemes may need to be preceded by pilot work to ensure that the best methods and design are chosen. Third, the extent to which volunteers are prepared to participate is likely to increase with experience. It would thus be wise to begin an overall programme with fewer schemes than it is intended eventually to run, building in new schemes as the original ones get established.

2.3.11 Sampling with partial replacement

The issue of Sampling with Partial Replacement is taken up in Part II.

3. HOW THE MONITORING OF MAMMALS IS DIFFERENT FROM THAT OF BIRDS

3.1 Mammals are less easily detectable than birds

Because mammals are less easily detectable than birds, simple counts may reveal only a small proportion of the population present in an area. Thus, they provide an index that may be less closely linked to the true population than would be a count of, say, singing Whitethroats *Sylvia communis* on a warm morning in early May.

If people go out to record mammals they may see none at all, resulting in a loss of motivation. To boost the frequency of observations, it may be necessary to trap mammals or to substitute observation of the animals themselves with observations of their signs. Trapping is generally more demanding of human and financial resources than is simple observation. The use of signs may demand special training and raises the problems of standardisation common to almost all index methods (see Section 1.5.6).

3.2 Identification

Apart from some groups of small mammals and mustelids, mammals are generally at least as easy to identify as birds. Indeed, a rather higher proportion may be identifiable by the non-specialist than is the case for birds, raising the possibility of drawing relatively inexperienced observers into the monitoring of some species.

3.3 More species-specific techniques are needed for mammals than for birds

A high proportion of terrestrial bird species can be covered by the Breeding Bird Survey (BBS), of freshwater and estuarine species by the Wetland Birds Survey (WeBS), and of seabirds by general counts at colonies. In contrast, most potential monitoring surveys for mammals cover only single species or a small group of species. In designing such surveys, one must try to find ways of including as many species as possible in each, without unduly compromising the quality of the data for individual species.

3.4 Availability of personnel

There are far fewer amateur mammal enthusiasts than there are birdwatchers, so there is a smaller pool of potential volunteer recruits for survey work. This problem may be rather less severe than one might imagine at first sight, however, because amateurs who regard themselves as interested in mammals tend to have a rather more scientific (or, at least, committed) interest than do many birdwatchers; "twitching" has not distracted mammal-watchers. Furthermore, it is clear that there is considerable potential for building up volunteer network for mammal monitoring (Part V). (It is true that Burnett, Copp and Harding (1994) reported that there were *c.*100 times more biological records referring to birds in the UK than those referring to mammals, suggesting a huge imbalance of observer effort. However, a large proportion of the bird records result form there having been three atlases of bird distribution and from bird-ringing. The latter alone produces almost a million records each year).

In contrast, a significant proportion of the admittedly small number of academic theriologists still feel able to engage in survey and monitoring as part of their academic work. Unfortunately, retirements and the relentless pressure to engage in theoretically more exciting work may quickly deplete the ranks of this group of scholars.

3.5 Diversity of existing inputs is greater for mammal monitoring

Most bird monitoring is organised by BTO, WWT, RSPB and JNCC, between whom there is close liaison. Thus bird monitoring is well integrated. As is made clear elsewhere in this report, recent and current mammal monitoring is conducted by a great variety of organisations: some do national work on one or a few species, others work only locally; individual bodies may conduct one-off surveys but not repeat them, or may mount schemes that last just a few years. Some effort may be needed to pull this diversity of interest together (as is essential if UK mammal monitoring is to be developed properly) but we do not believe that this will be a major problem.

3.6 There is no commitment to long-term monitoring of mammals

Whereas the ornithological bodies have managed to build up a level of commitment to long-term monitoring (including some commitment to the funding that makes it possible to keep schemes going in practice), this has not been the case for most mammal monitoring. Some guarantee of funding is essential if cost-effective monitoring is to take place. Otherwise, we will continue to waste resources (both human or financial) on a series of short-term initiatives that do not provide the long-term monitoring that is needed.

3.7 Mammals are generally less mobile than birds

Birds are sufficiently mobile for it to be useful to think of a “British population” for many species. For mammals it will often be more illuminating simply to draw together the monitoring information for different regional populations rather than to combine it into some sort of national index, since the regional populations are affected by regional management and environment and since they exchange few individuals. There may, of course, be an administrative need to report at UK (or country) level but for actually managing the animals we need the regional information. This presents no problems for species that are already well-monitored for management purposes, such as Red Deer. Indeed, given that such species may be monitored differently in different places (by different organisations!), it is an advantage, because formally combining the regional data into a national index may not be easy. For other species, it may be difficult to obtain enough data to provide reliable indices at scales less than the whole UK; if we have to accept that, then we must still remain alert to the likelihood of regional differences in population trends.

Lower mobility may mean that mammals treat the environment as more ‘fine-grained’ than do birds, so that populations may vary more from place to place within a habitat than do those of birds of corresponding size. This may be particularly important for surveys of small mammals based on trapping, where a sample area may typically be one hectare or less; larger sample areas would smooth out the spatial variation in small mammal numbers but may be practically inconvenient.

3.8 Many mammals occur in many different habitats

This has several consequences. First, it is especially important to ensure that all habitats are adequately represented in the national sample, since population changes may differ between habitats. Unfortunately, because mammals are difficult to detect, it may be necessary to use different field methods in different habitats. For example, deer in open country may be assessed best by direct observation but in woodland by signs, whereas birds can be directly observed (by sight or sound) in both habitats, using methods such as distance sampling to correct for the difference in detectability between habitats this may mean that monitoring information for deer in the different habitats cannot be formally combined.

If there are economic reasons for focusing attention on a species in some habitats (for example, Grey Squirrels in forestry but not in gardens), this may result in a disparity between habitats in levels of work and, therefore, in level of information.

PART II. CONSIDERATIONS OF DESIGN AND STATISTICS

1. INTRODUCTION

1.1 Why revisit these issues?

MMR cover many of these issues in some detail. Our purposes in revisiting them are: to remind readers of key points, to introduce some points not covered by MMR, and to make a closer connection between some of the matters of statistical nicety and those of practical possibility. Some of the points we make are elementary; we make no apologies for this, because getting the design of surveys right is so important.

1.2 Statistical nicety or practical pragmatism?

The purpose of scientific investigations is to draw conclusions about the natural world. The purposes of survey design and statistical analysis are to maximise the efficiency of that process and to minimise the possibility of the wrong conclusions being drawn. In simple situations, the statistical design and analysis may almost completely remove the need for the scientist to make subjective judgements. For example, if one were to count all the individuals of a highly conspicuous and easily identifiable species in a truly random sample of 1ha plots in Great Britain, one could arrive at an unbiased estimate of the total GB population (with reliable confidence limits) simply by applying standard statistical techniques. In most real cases, however, the investigator would have to make judgements about the extent to which individuals may have been missed in the counts (or counted twice), the extent to which such counting problems differed between plots, the extent to which randomly chosen plots that could not be counted because of access problems were likely to have more or fewer animals than the average, etc. Such judgements will determine how accurate the investigator judges the population estimate to be - i.e. how close it is likely to be to the true population size. Accuracy thus defined is affected by two consequences of the methods used - whether they give biased results (i.e. whether the estimates are systematically different from the true values in one direction or the other) and whether they give precise results (i.e. whether repeating the study would produce a closely similar estimate). In general, it is impossible simultaneously to minimise bias and maximise precision. We make these elementary points because so much attention is usually paid to minimising bias, both during the education of ecologists and in the design or interpretation of ecological fieldwork, that there is a tendency to neglect the simultaneous need to maximise precision; the overall need is to achieve the most accurate estimates possible (given the resources available), by striking the optimum balance between minimising bias and maximising precision.

The point can be illustrated by example. Suppose that one is monitoring a species and has decided that action should be taken to conserve it if its numbers fall by 25% over 25 years. Suppose that one adopts a monitoring method that estimates the population size after 25 years to be 90% of its original size and that this estimate is absolutely unbiased; but suppose further that the lack of bias has been purchased at the cost of low precision, so that the confidence limits of the estimate are 60% and 150%. This information is of limited practical use. In contrast, if one had adopted an admittedly biased method that gave confidence limits of 87% and 93% and had good reason to suppose that the bias was unlikely to be more than a few percent, then one could conclude that it was unlikely that the 75% alert limit had been exceeded, a useful conclusion. As Yates (1981) wrote: "The investigator must also avoid attaching exaggerated importance to minor sources of bias which, in fact, can only produce errors which are trivial relative to the random sampling error".

Unfortunately, bias is rarely measurable in practice, so the balance between bias and precision is a difficult one to strike and generally depends on subjective judgements. So long as the judgements about bias are based on the best information about the natural history of the subject species and are made explicit to those involved in using the results, one is justified in using methods that are not completely unbiased.

Recommendation:

* **Be prepared to abandon absolutely unbiased methods if practical considerations dictate that overall accuracy is higher for somewhat biased methods - but make explicit the likely sources and magnitudes of bias. To avoid post hoc controversy, seek wide agreement about the likely sources and magnitudes of bias before finalising plans for the monitoring scheme.**

2. THE SPATIAL DISTRIBUTION OF SAMPLING

2.1 Generalising from samples

To know the population size of an animal in Britain, we would ideally count every individual directly - i.e. conduct a census. This is usually impossible in practice, if for no other reason than cost. Instead, we count sample areas and generalise from them to the whole country. The process of generalisation is straightforward when the samples are taken through a formal, properly randomised design. If they are not, the reliability and extent of the generalisation are a matter of judgement. The worst case occurs when, for practical reasons, it is only possible to study a single site: not only may this site be atypical but one has no direct knowledge of how atypical it might be, having to rely on knowledge of the species, the habitat and of similar cases to judge the reliability of conclusions drawn from just the one site. Sometimes the conclusion may be that knowledge of one site is better than no knowledge at all; sometimes one may reach exactly the opposite conclusion, in that the cost of being misled by information derived from just one site is so great that it is better to have no knowledge at all.

The reliability of information derived from a single site depends on how representative it is of the whole country. This has to be a matter of careful judgement. Unfortunately, naive judgements about how typical is a site may be seriously awry: a “typical” oak woodland is more likely to resemble some sort of idealised oak woodland (rarely observed in real life) rather than an average oak woodland. Assumptions and assertions of typicality thus need particularly critical examination.

Even if a site is not truly representative, there may be good reasons for studying it. Thus the sites in the Environmental Change Network (ECN) are mostly not typical of the British countryside: some were chosen for study because they had special features or were ideal examples of a particular habitat; most are managed differently from the wider countryside, sometimes because they are long-term ecological study sites! Nonetheless, the view was taken that, for the purposes for which ECN was set up, the sites were sufficiently representative of the rest of Britain that the information derived from them would be sufficiently capable of generalisation. The benefit of establishing a set of truly representative sites for the ECN, rather than using the existing long-term study sites, was judged not to be sufficient to justify the cost of doing so. Similar practical considerations must illuminate the establishment of mammal monitoring.

Multiple sample sites, as in ECN, have the advantage that the differences between them provide information on how atypical each is likely to be of the whole population and therefore how reliable one’s generalisations are likely to be. Formally, provided the samples are truly random, one can place confidence intervals on statistical estimates derived from multiple samples. Thus, where possible, monitoring should be conducted at more than one site.

2.2 Sources of imprecision in sample surveys

Generally speaking, the precision with which population means are estimated will increase with the square root of the number of samples, but precision of the overall mean also depends on the precision with which measurements are made at each sample site. There is often a trade-off: more effort devoted to improving the precision of counts at individual sites means that there is less effort available for covering more sites. Texts on survey design provide methods for optimising the distribution of effort in such cases (e.g. Greenwood 1996). It has been suggested that such cost-benefit analysis in survey-design may often not be worth the effort, at least unless one pays careful attention to the design of the pilot studies that are used to provide information for the cost-benefit analysis (McArdle & Pawley 1994). However, the extent to which the recently-developed BTO/JNCC/RSPB Breeding Bird Survey (many sites, relatively undemanding work at each) provides better bird-monitoring at the national scale than the Common Birds Census (comparatively few sites, each studied intensively) shows the value of choosing the right design. For some mammal monitoring, rather little background information is available on which to base design judgements; it may be necessary to conduct pilot studies to provide the necessary information or to treat the first few years of a new monitoring scheme as a trial. (See also Part I, Subsection 2.3.10).

A great imponderable for surveys that use volunteers is the level of participation to be expected. One has little alternative but, having decided what level of participation is the minimum to make the planned survey effective, to use the experience of those involved in organising similar surveys in the past to judge whether that level of participation is likely to be attained.

2.3 What population is of interest?

Design texts lay great stress on the need to define what is the statistical population about which one is wishing to make inferences. Unless one does this, one's survey is likely to be both inefficient and misleading.

It is possible in principle to design surveys that will provide efficient and unbiased estimates of population change at the national level - i.e. the focus is on the national population. Paradoxically however, there are several reasons why having the national population as the immediate focus of the survey design may not be efficient for the monitoring of that population. The first we have covered in Part I, Subsection 1.4.2: where populations in different parts of the country are rather isolated from each other, greater insight is obtained by treating each as a separate entity than by aiming to obtain some measure that relates to the "national population"; the national perspective that is required can be obtained from drawing together the information about the individual populations.

There may be some geographical regions or habitats that contain so few of the species in question that the effort involved in surveying these places is not worthwhile as, for example, Badgers in urban areas (Cresswell, Harris & Jefferies, 1990). If so, it is best to omit those regions or habitats from the sampling programme. Given the extreme fragmentation of habitats in much of the UK and the fact that the smaller mammals in particular do not move far, selectivity of sampling may need to be fairly fine-grained. The information one gets from such selective sampling is, of course, relevant only to that part of the national population that lives in the parts of the country included in the survey; but if this is clearly most of the national population, that is enough for practical purposes. There is a proviso, however, in respect of surveys that are undertaken for monitoring purposes, since monitoring is concerned with change rather than with absolute numbers and change may, in some circumstances, be more marked in places where a species is scarce than where it is abundant. It is true that because such places hold only a small proportion of the national population the changes occurring in them (unless hugely different from the changes elsewhere) will have little impact on the national population index. However, knowing that there have been marked changes in such places may well be of much more relevance to conservation than knowing that the national population has changed rather little. For example, a species may be more sensitive to environmental change in marginal rather than in core areas, or we may need to assess how well a species, such as the Polecat, is re-occupying its former range, or we may need to assess how rapidly a species such as the Grey Squirrel is spreading and thus increasing its impact on other species. Thus whether or not to leave places with few of the species out of a survey must be a case-by-case decision as to whether such places are of special interest - Focus Zones, in MMR terms.

It may be necessary to leave out some places because they are difficult to work in or to access. This is acceptable for monitoring if one knows that the animal is relatively scarce in the excluded areas or if one can reasonably assume that the changes in those areas are unlikely to be markedly different from those in the rest of the country; otherwise, one simply has to admit ignorance of the excluded sector of the population. If the excluded places are left out according to clear criteria that can be related to habitat, the effect of leaving them out can be explicitly and clearly addressed. If they are left out on a more *ad hoc* basis, there is often a tendency to gloss over the possible effects; this should be avoided. For example, it may be reasonable to assume, in some circumstances, that places to which observers are refused access are representative of the countryside as a whole, so the exclusion does not bias the results. The refusal may, however, be linked to land-management practices that may affect the mammals being monitored (such as Pheasant-rearing) or, indeed, to the fact that illegal methods of control are being practised; if so, one has to accept that the population being studied is not that of the countryside as a whole but only a (rather ill-defined) part of it and that the results may not accurately reflect the population in the areas excluded from the survey.

Another situation in which one monitors only parts of the national population is when one is using existing schemes that are designed to provide monitoring of local populations (such as much deer monitoring) or of particular habitats (such as surveys of rodent infestation of houses). The extent to which it is appropriate to rely on such schemes depends on how representative are the populations they cover of the national population and on the expense of mounting surveys that would be more representative.

Appropriate monitoring techniques may differ between habitats. If so, it is sensible to design one's monitoring programme as a set of surveys, one for each of the habitats; each provides information on the population in that habitat. The way in which that information is brought together and used for national monitoring will depend both on statistical constraints and on needs; if the index in habitat A has gone up by X% and that in B by Y%, one can only produce an index of change in the national population if one knows the relative sizes of the populations in A and B - but that index is probably less revealing than are the separate indices for the two habitats.

For some species, useful information may arise from more than one monitoring scheme. For example, Grey Squirrels might be monitored by presence/absence in volunteers' gardens, through presence/absence in Woodland Trust reserves, through numbers seen on Breeding Bird Survey transects, and by frequency of occurrence of signs in hair tubes in plantations. None of these provides an index for the national population (or even that of a well-defined geographical region or habitat). Furthermore, the population to which each relates may be rather ill-defined (such as: "Grey Squirrels that use gardens at the times of day when volunteers are observing and that are bold enough to be observed") and may overlap with the populations covered by other schemes. Nonetheless, if the suite of schemes is taken as a whole and interpreted carefully, it will provide useful information for conservationists.

Finally, the interpretation of indices (rather than true counts) in terms of the population being studied is important. If Red Foxes in forests are monitored by counting scats along the edges of rides, one is monitoring not the Red Fox population in the forests but that proportion of the population that defecates along rides; the extent to which such Red Foxes are representative of the whole population must be explicitly considered not only when the survey is being designed, but also when the data are interpreted. The consideration must not only involve knowledge of Red Fox behaviour but also of the external factors that may be relevant, such as the likely impact of ride management on defecation behaviour. (In addition, as with all indices, one must consider how the relationship between the index and the population that is being monitored varies. For example, the number of ride-defecating Red Foxes may stay the same but the index may change because the defecation rate varies seasonally or because the detectability of scats by observers varies according to vegetation growth.)

Recommendations:

* **Treat isolated populations as separate targets for monitoring.**
* **Leave areas sparsely populated by the species in question out of the study, unless changes occurring there are likely to be of special interest.**
* **Be prepared to leave out areas that are difficult to cover but make as explicit as possible the criteria for omission and the likely resultant biases.**
* **Use existing local schemes as part of the national programme if this adds useful information, even if they use non-standard methods.**
* **Treat the populations in various habitats as separate targets for monitoring if different methods have to be used in the different habitats.**
* **Especially if no one scheme provides unbiased information on the national population, be prepared to use multiple schemes for monitoring a single species.**
* **Be particularly aware that indirect indices may refer to only part of the total population.**

2.4 Randomisation

It is well-known that the only way to obtain a sample that is truly representative of the population as a whole is to select which areas to sample by a formal randomisation process. Informal, haphazard processes are rarely truly random. Selecting “typical” sites is usually grossly non-random, suffering both from the misapprehensions about what is truly typical and from the failure to include a representative range of sites.

Unfortunately, many practical constraints make randomisation difficult or at least costly. Many of these have been considered in the last section - places or habitats where the species is scarce, difficulties of access, etc. If these practical difficulties interfere with full randomisation, one must carefully consider what exactly is the population that is being studied and then consider how typical it is likely to be of the wider population that is the true focus of interest. In other words, what bias is involved in using the non-random sample to estimate the changes going on in the wider population?

Recommendation:

* **If practical problems rule out proper randomisation, make the likely biases as explicit as possible.**

2.5 Species restricted to special habitats of limited extent

Riparian species present particular difficulties. It is not practically effective to choose study sites completely at random, since the majority of them will then not contain riparian habitat nor, therefore, the species in question. For example, the pilot Waterways Breeding Bird Survey achieved markedly better coverage than the Breeding Bird Survey of many riparian species, e.g. 36% of WBBS sites occupied by Dippers compared with 2% of BBS sites, 31% vs 4% for Common Sandpipers, and 41% vs 6% for Grey Wagtails (Marchant & Gregory 1999). There were similar differences for mammals, although the mammal data were collected incidentally to the bird data and are therefore perhaps less reliable: e.g. Otters on 14% of WBBS sites, Water Vole on 9% and American Mink on 8%, with almost none of these species being recorded on BBS sites (Marchant & Gregory 1999 and Part III).

It is easy enough to randomise at a coarse scale - for example, by choosing a sample of 10x10 km squares within which to survey the species in question. The difficulty lies in defining and identifying riparian habitat within such areas - when is a ditch too narrow or too dry to be included? Whatever method is used to pick which stretches of waterway to survey, one must consider carefully how this will bias the results. Another problem is how large a sample one should take within each 10x10km square; in principle, one should sample the same proportion of available habitat in each square, not the same absolute quantity. If one does not sample the same proportion and if the density of animals per km of habitat varies according to the amount of habitat available, then one's sample will provide a biased estimate of population size. Even if density does not vary according to the amount of habitat available, sampling that is not proportionate to the amount of habitat available may result in substantial inflation of the error variance of one's estimates of mean numbers or changes in numbers (Cochran 1977, section 11.2). In this respect, the methodology of Otter surveys, sampling at 5km intervals along every waterway in the selected 10x10km squares, has been superior to that of Water Vole surveys, in which exactly five samples were taken in each selected square.

Similar considerations apply to other special habitats, some of which may be of very limited extent. Small mammals that occupy field margins, but not the cropped area of farmland, may pose particular problems.

Recommendations:
* **For surveys limited to special habitats, make as explicit as possible both the criteria used to define the places to be sampled and the potential biases.**
* **The number of samples of a special habitat to be taken within higher level sampling units (such as 10x10km squares) should preferably be proportional to the amount of that habitat present in the higher units.**

2.6 Systematic surveys
Systematic surveys comprise those surveys in which the country is covered by a grid and cells of the grid are included in the sample not at random but in some sort of pattern. MMR consider the

problems resulting from systematic sampling at some length. Our aim is briefly to review and assess the magnitude of these.

The most obvious problem is that the pattern of the sampling may coincide with a pattern in the distribution of the organisms being monitored. This could easily happen, for example, if one studied soil invertebrates on a hillside that had been subject to periglacial sorting of the soil, producing some sort of patterned ground, but we believe that mammals are unlikely to show regular patterns in distribution or numbers at scales that would coincide with sampling patterns based on the national grid.

A more subtle problem, to which MMR pay much attention, is that estimates of the error variance of population means that are based on the assumption of random sampling are likely to be wrong if sampling is systematic. The direction and extent of the biases in the estimates of error and ways of obtaining unbiased estimates have been addressed in the technical literature (e.g. Bellhouse 1988; Murthy & Rao 1988). Unfortunately, they depend very much on the type of geographical trends and patterns present in the study population and how the pattern of the sampling relates to these (which determines the spatial correlation between samples), about which we are likely to know little in practice. However, both theory and simple simulations suggest that when the total number of potential sample units (e.g. 1x1km squares of the national grid) is large then the bias in estimating the error variance is likely to be small.

MMR also point out that the fact that the "false origin" of the Ordnance Survey grid was not randomly located raises difficulties in statistical theory, although these are unlikely to be other than trivial in practice.

To test out the effects of systematic versus random sampling, MMR used Countryside Survey data on habitat distribution taking the latter as an example of a spatial distribution comparable to that of mammals. They found no obvious biases in estimation of either means or standard errors when comparing systematic with random sampling. We agree with their conclusion that the theoretical reasons for preferring random sampling are unimportant in practice.

Advantages of systematic sampling pointed out by MMR are that it is easier for the layman to understand and that the sample units are easier to locate. Perhaps even more important, a regular grid makes the description and analysis of geographical patterns easier. Such patterns may be important in understanding population changes in distribution or abundance.

Note that, in refusing to follow statistical purists in condemning systematic sampling, we follow one of the masters of sampling methodology, who recognised that it would generally prove satisfactory except for material that had periodic features (Yates 1981).

Recommendations:

* **Generally choose random sampling but adopt systematic sampling if this is practically more convenient or if information about geographical patterns is required.**

2.7 Study sites chosen by the observers

Study sites chosen by observers are likely to be unrepresentative: they may be "typical" (i.e. atypical, see Section 2.1 above), chosen for reasons (such as accessibility) with which the abundance of mammals may be linked, or chosen because they have good populations of the species in question. In principle, such biases are undesirable. In practice, however, it may be better to get information from such sites than to get none at all.

Note that schemes in which observers choose where to go within a randomly chosen 10x10km square (for example) may be less biased than those in which there is absolutely free choice - but they cannot be bias-free.

Recommendation:
* **Be prepared to use study sites chosen by observers (rather than randomly allocated) if this is the only way to achieve enough sites but explicitly address the biases that may result.**

2.8 Sampling at more than one level

Suppose that traps are difficult to transport long distances but relatively easy to transport short distances. If one decides to set traps in a 40x40m grid at each sampling location, one could have such sites scattered across the country; but this would involve long distances between the sites. Alternatively, one might choose a set of 1x1km squares to which to take traps and then choose a set of 40x40m squares within each to be the sites at which one does the trapping; the 1x1km squares would still be at long distances apart but the small squares within each would be within easy distance of each other. This is an example of two-level sampling, which may be adopted for various practical reasons. If sampling is random at both levels, statistical interpretation is straightforward. The principle can be extended to more than two levels. The relative number of sample units to take at each level in order to maximise efficiency depends on relative costs (see e.g. Cochran 1977; Greenwood 1996).

If the sampling is non-random at any level, then the total sample is non-random - the randomisation at other levels does not compensate for the non-randomness at the one level. As always, judgement about likely bias must be made before one adopts non-random choice.

If non-random choice is at the lowest level, it may be circumvented simply by combining the data within each of the low-level groups, which are now treated as "clusters" rather than as sets of independent samples. In the trapping example for instance, moving the traps between five (say) random 40x40m squares within a 1x1km square may be too burdensome; as an alternative, one might choose one such 40x40m square as the starting point, then set the traps during the next four successive weeks in four squares 100m away from the first one, to the north, east, south and west. To analyse the data, one then lumps the results from each of these clusters of five. Such cluster-sampling allows more data to be gathered without expending the effort that full randomisation may require; the benefit is achieved at the cost of having rather broader confidence limits on one's results than if the design were fully randomised.

Recommendations:
* **If practical considerations suggest that it may be appropriate to sample at more than one level, pure random sampling or cluster sampling should be adopted if possible. Beware of bias entering at any level.**

2.9 Many small samples or fewer larger ones?

Since the samples in a cluster are lumped, each cluster is effectively one sample. Thus cluster sampling is an example of taking fewer larger samples than many small ones. Another example might be sampling 100 2x2km squares rather than 400 1x1km squares. To get the same area covered, taking fewer, larger samples (and clusters) is generally practically more convenient than taking many small ones - travel costs are likely to be lower, for example, but there is a cost to this, in that (because of the spatial autocorrelation between the parts of the larger samples, or the components of the clusters), the standard errors of the statistics that are estimated from the data will be higher. Cochran (1977) shows how to optimise cluster-sampling designs for a given cost (i.e. how many clusters of what size) and similar considerations apply when optimising the balance between number and size of samples.

For volunteer-based surveys, optimising the distribution of sampling effort will be more difficult because it will depend on predicting the responses of potential volunteers to various possible designs. For example, suppose that volunteers can easily walk several 1km transects. For the same total distance walked by a volunteer, precision will be greatest if his or her transects were part of a completely random sample; it would be less if they were randomly distributed within a fixed 10x10km square (itself randomly chosen, one hopes), even less if they were randomly distributed within a 5x5km square, and least if they were in adjacent 1x1km squares, but volunteers would rate these four designs in reverse order in terms of ease, so one would be able to recruit more people to the statistically less effective designs. Given that the likely reactions of volunteers to different designs are likely to be unknown in advance (unless some sort of opinion poll is undertaken), this aspect of survey design may depend largely on the judgement of those with good previous experience of organising such work and on their ability to balance theoretically probity with practicalities.

Recommendation:

* **Try to optimise the distribution of survey effort, in terms of number and size of samples, even though this may be difficult to predict for volunteer-based work.**

2.10 Stratification: general

In the most recent national Badger survey, OS grid squares were each assigned to one of seven land class groups; sampling was random within each group, not over the total set of squares (Wilson, Harris & McLaren 1997). A similar procedure has been used in surveys of Brown Hares, Red Foxes and bats. This is an example of stratified sampling: the potential sample units are divided up into sets (strata) and randomly sampled from within each set rather than from the entire population. The sets may be defined according to any relevant criteria, such as geography, land class, habitat, or observer availability, depending on the purpose of the stratification. Because stratified sampling adds to the complexity of organising a survey, it is important not only to be aware of its potential benefits but also to assess the likely magnitude of those benefits.

One purpose of stratification may be to ensure that all administrative divisions are adequately covered by the sample. For surveys involving hundreds of sample sites across the UK, adequate samples of each of the four countries (and of the English government regions) will be obtained without stratification. For smaller regions (should it ever be necessary to focus on such fine scales), stratification would be needed to ensure adequate coverage of each.
Stratification may also be needed to ensure adequate coverage of all habitats or land classes, which may be important if one wishes to use between-habitat differences to explain changes in abundance or distribution. Again, if the size of such strata is not too small, adequate sampling of

each will happen without stratification. Since the most efficient distribution of sampling effort for exploring differences between strata is similar to that for precise estimates of the national statistics, the guideline that Cochran (1977) provides for the latter can be used here: if non-stratified sampling is likely to provide samples of 20 or more in each habitat, stratification is not needed at the sampling stage; post-stratification (Section 4.3, below) will be sufficient. Thus it is only likely to be needed if some of the habitats cover less than 10% of the country.

Another reason for stratification arises from the need to replace some of the sample sites over the years because, for example, land-owners decide to withdraw permission. To keep the characteristics of the sample as constant as possible, one could replace sites that drop out with other sites from the same strata, as has been done in Badger surveys (S. Harris, pers. comm.). This is likely to be important if the turnover of sample sites is large or is concentrated in some strata.

Stratification also allows sampling to be more efficient as a means of assessing the national population size or population change. By concentrating sampling effort where variation is greatest (which for monitoring means spatial variation in changes in abundance over time), greater precision is obtained per unit effort; the same is achieved by concentrating it where sampling is easiest (cheapest). The ways of optimising the distribution of sampling between strata according to differences in variation and in cost are well known: sampling should be concentrated more on strata that have large between-locality, within-stratum variance in numbers and that are relatively cheap to sample (e.g. Cochran 1977; Yates 1981; Greenwood 1996). However, McArdle and Pawley (1994) point out that one needs good information on within-stratum variation to achieve the optimum design and they suggest that it may often not be worth the effort. Unless strata differ markedly in the magnitude of the between-locality variance in population size or change or in the cost of taking samples, optimum stratification produces relatively little gain compared either with including the same proportion of each stratum in the overall sample (as has been attempted with several recent mammal surveys) or with pure random sampling combined with post-stratification (Section 4.3, below).

If stratification is used to improve the efficiency with which the national mean is estimated, it is probably unnecessary to use more than about six strata (Cochran 1977). We suggest that a similar number is appropriate if the focus is on using between-stratum differences to understand a species' ecology: to use more would run the risk of obscuring major patterns with incoordinate detail, as well as of having sample sizes for some strata that were too small to provide precise enough statistics.

Decisions not to sample areas, habitats or land classes that hold few animals or that are particularly difficult of access (see 2.3, above) are an extreme case of stratification - they involve identifying strata that provide so little information relative to the cost of getting it that there is little point in sampling them at all.

If habitat-related stratification is to be used in mammal monitoring, it is important to use relevant strata. If stratification is being used because of major differences between strata in variance or cost (so that stratification is needed to optimise efficiency), then the distribution of samples between strata needs to be species-specific. Furthermore, different stratum divisions may be appropriate for different species. On the other hand, it would be practically convenient to use the same system for most species; this would also make the analysis of species interactions easier. Several surveys of mammals have already used the ITE Land Classes as the basis of stratification, generally on the four or seven aggregate landscape groups rather than the 32 basic

Land Classes, the latter rightly being considered to be too finely divided for the purpose. The ITE system underpins the GB Countryside Surveys, thus allowing monitoring based on these Land Classes to be linked to the data that they provide. Unless there are particular reasons for using other stratifications for other species, it would be appropriate to continue to stratify by aggregated Land Classes, such as the ten of ITE's Division 3 or the four of Division 4 (Bunce 1992).

Note that, for administrative reasons, these 32 Land Classes have now been modified somewhat differently in Scotland and in England and Wales, in that Land Classes that are scarce in Scotland have been combined with similar but commoner Land Classes for Scottish purposes and similarly for England and Wales (Barr 1998). Furthermore, the Northern Ireland Countryside Survey used a different landscape classification from that used in Great Britain. Thus stratification will need in future to be at two levels: first by geography (three - or perhaps four- strata), then by Land Classes that are somewhat different in the different countries. We understand that analysis of CS2000 may use three aggregate Land Classes in England and Wales (two lowland, one upland) and a different three in Scotland (one lowland, two upland). There are enough data from previous mammal surveys to test whether this level of aggregation is sufficient or whether it would be better to use more strata for mammal monitoring schemes.

Recommendations:

* **Adopt stratification for mammal monitoring if there are good reasons for doing so for the species in question (This may need some simulation work relevant to individual species, based on existing data and realistic assumptions). If there are no particular reasons for stratifying, and it is easier not to do so, then do not bother.**
* **Unless there are particular reasons for doing otherwise, stratify by aggregate ITE Land Classes.**

2.11 Focus Zones may be strata

A particular form of stratification arises for species where Focus Zones have been identified. To pursue the example in Part I, Subsection 2.3.3, Polecat sampling could be conducted on the basis of three strata: one, regions with no Polecats (sampling intensity zero); two, regions on the edges of the Polecat's range (Focus Zones - high intensity of sampling); three, core Polecat range (moderate sampling intensity). However, if it is appropriate to use different methods in different regions, the latter can no longer be considered to be strata in a single survey; rather, they are subdivisions of the national population for which different monitoring schemes are being operated.

2.12 Stratification by observer availability

Concentrating sampling effort where there are relatively more volunteer observers (see above) is another example of stratification. The costs of sampling in the different parts of the country may not be explicit in this case, since they are chiefly the indirect costs of drumming up support, but these certainly differ according to the availability of volunteers (with the complication that within any one region the costs rise disproportionately with the number of observers required - the 21st observer is more difficult to recruit than the first). If one were simply to choose potential sample sites at random across the whole country and then only use those for which it is possible to recruit observers, bias will creep in at two levels. First, whole areas of the country (e.g. the Scottish Highlands) will be under-represented; second, places within local areas that are closer to peoples' homes are likely to be over-represented. These problems have both been addressed in the BTO/JNCC/RSPB Breeding Bird Survey. First, the country is divided into regions and the sampling intensity in each adjusted to the availability of observers; by appropriately weighting the data from each region, an unbiased national index can be obtained. Second, potential sample sites within regions are randomly ordered and are allocated to observers in that order, so that a site that is close to potential observers cannot be included until all sites before it in the list have been included, even if some of these are distant. Difficulties are still encountered in large, sparsely-populated regions: it may be impossible to persuade anyone to cover a remote square that is early in the list, even though there are plenty who are prepared to cover less remote squares. Once again, if the patchy distribution of observers cannot be fully overcome by using methods such as used by BBS, one has to consider carefully the likely biases that will result.

It is, of course, important not just to take steps to overcome the problems arising from variation in availability of observers but to take steps to reduce that variation. Recruitment efforts should be particularly intense in poorly populated regions. Volunteers may also be supplemented with professionals in such regions but we regard this as a last resort if the scheme is meant to be volunteer-based: it adds significantly to costs and sets an undesirable precedent.

Recommendation:

* **In schemes that are largely volunteer-based, consider stratifying by observer availability as an effective means of avoiding bias while achieving large sample sizes in those regions where it is possible.**

2.13 Statistical power

Many of the recommendations above are designed to maximise statistical power (the ability to detect important changes in numbers). The matter should also be looked at the other way round: For a given cost and sampling design, what is the power of the scheme to detect changes in numbers of a certain magnitude?

MMR present considerable discussion of the issues of statistical power to detect trends and validation of survey methods. They generally fail, however, to justify the choice of sampling design and recommended sampling size for individual species in terms of the power that these will have to detect trends that are considered important to conservationists. This can only be done formally by using species-specific data on the scale of geographical variation in the magnitude of temporal changes and on the extent of short-term fluctuations, since both of these determine the variance of any estimate of trend. Where the requisite data are not available, informal estimates based on scraps of relevant data, on related species, and knowledge of the species' natural history can be made. We have not had time to conduct the necessary analyses. Instead, we have used our experience in avian monitoring, knowledge of species' natural history and population ecology, and expert advice to come up with suggestions for suitable sample size for each species. This will allow decisions to be made as to what schemes it would be useful to attempt to run. After that, it would be wise to undertake more formal statistical analyses to confirm the suggestions that we have made for the species that it has been decided to cover.

As a general guideline in discussions on the size of schemes to run, one should note that precision will generally increase according to the square root of sample size. Thus, to halve the width of one's confidence limits, one would need to quadruple the sample size. More optimistically, if one can tolerate confidence limits twice as wide as have been (or may be) obtained with an existing (or planned) survey, one can reduce the sample size to only 25% of the current (or planned) number.

3. THE TEMPORAL DIMENSION OF SAMPLING

3.1 The importance of historical continuity of sample sites

Monitoring necessarily entails taking samples repeatedly over time. There are two extreme possibilities, to choose the sample locations independently on each occasion (Fig. II.3.2A) or to use the same locations on each occasion (Fig. II.3.2B). (We consider intermediate cases in the next section). Using the same localities has three advantages. The first is concerned with avoiding bias. Suppose that, however carefully the protocol for choosing sample sites is defined, there is a subjective element in it. If different sites are chosen on each occasion, there is no guarantee that the subjective element will remain constant; thus a change in the sample mean over time may not, not because there has been a change in the population mean but because the bias is changing. This problem is circumvented if the same sample sites are used on every occasion. It can probably also be circumvented in most cases by using careful and consistent protocols for choosing study sites, so it is not a strong reason for sticking to the same sites on every occasion.

Much more important is the value of historical continuity for increasing the precision of one's estimates of change. This is because the error variance of the estimate of change contains two components if one uses independent samples - that due to differences between sites in average numbers and that due to differences between sites in the change in numbers. The former, which is typically large, is removed if one uses the same sites on each occasion, so the change in numbers is estimated more precisely. An illustrative example of the effect on precision is given by Greenwood (1996). The number of breeding territories of Chaffinches *Fringilla coelebs* were counted in each of two years on 65 English farms; in one simulation exercise, an independent sample of 20 was drawn from the population of 65 farms in each year (five squares falling in both samples by chance); in another, the same sample of 20 farms was used in the two years; in both cases, the sample data were used to estimate the change in mean numbers between the two years. The confidence limits of these estimates were five times wider for the independent samples than for the constant sample. In a national programme, it is likely that there would be much less (if any) overlap between independent samples drawn in different years, so the confidence limits of the estimate of change under that protocol would be even greater. If the Chaffinch result is typical, it means that historical continuity delivers an improvement in precision equivalent to more than doubling the sample size. This is a powerful advantage.

The third advantage of historical continuity is that (unless the sampling intervals are greater than a few years) it requires less time to be spent in seeking access permission. In our experience, the time taken in getting initial permission is an order of magnitude greater than the time taken in renewing that permission annually.

There could be a corresponding disadvantage to historical continuity: that land owners may be less prepared to grant access for a survey that will continue indefinitely into the future than for a one-off survey. We do not believe that this is generally the case.

Another disadvantage to long-term continuity is that land-owners may change their management as a result of the regular visits of those conducting the monitoring. They may, on one hand, manage the land more sympathetically for the wildlife; on the other hand, through having their attention drawn to a species that they may consider to be a problem, they may take steps to control its number or even to eliminate it from their land. If these steps are illegal (or likely to attract public opprobrium), they may then deny access to the land! We do not believe this problem to be a general one but it needs to be considered for a few species, of which Badger is perhaps the most obvious.

A final disadvantage of revisiting the same sites, especially for single-species surveys, is that observers know what to expect. They may be more persistent in looking for the species because they know it was seen there on a previous occasion. If there was no corresponding lessening of effort in sites from which the species had not been previously recorded, this would bias analyses of change.

Wilson *et al.* (1997) checked for both of the latter two potential biases by comparing the numbers of Badger setts in revisited and new squares (and found no evidence of bias). Given that some turnover of sites is almost inevitable, similar checks could be carried out in most repeat surveys even when the aim was to revisit the same sites

Drawing new samples on each sampling occasion gives more information on geographical variation in abundance. This provides better estimates of the mean abundance (or index of abundance) of animals, averaged over the whole country and over all the years of the study, but knowledge of actual abundance is of limited value for monitoring, which must focus on changes in abundance - which are better estimated using an historically continuous sample.

Geographical information is directly useful for modelling distribution in relation to environmental factors, which may help to understand what determines the distribution and abundance of the species. However, given the extent to which the distributions of mammals in Britain are dependent on direct management by man (current or historical), this is likely to be less illuminating than studying temporal changes. In any case, the increase in precision of the geographical models that is obtained by taking new samples every year is likely to be less than the corresponding decrease in precision of the temporal monitoring.

Geographical information is useful for planning and similar purposes: the presence of Common Dormice or Badgers in a wood may affect the decision as to whether it should be turned into a housing estate, but even if new monitoring samples are drawn every year for many years, the overall proportion of the country covered will be small. Site-based information is much more effectively delivered from Atlas surveys, local biological recording, and specific surveys of threatened sites.

Recommendation:

* **The general case for using the same sites from year to year is overwhelming, except possibly for species that may be persecuted by landowners.**

3.2 Partially replacing samples

It is not always possible to keep sampling the same sites: permission may be withdrawn, or fieldworkers may not be able to carry on. The available methods for analysing monitoring data can cope with the inexorable slow turn-over of sites that results (see 4.2, below).

MMR consider in detail more formal designs for Sampling with Partial Replacement (Fig. II.3.2C); their proposed MaMoNet incorporates these ideas. The rationale of such designs is that they provide information on both geographical and temporal variation, providing estimates of both overall means and temporal change. There are situations where this combination may be more useful than either the mainly temporal information provided by sampling the same localities on every occasion (Fig. II.3.2B) or the much greater spatial information provided by drawing independent samples on each occasion (Fig. II.3.2A) (see, e.g., Skalski 1990). We do not believe that these apply to the monitoring of mammals in the UK. Sampling with as little replacement as possible remains the method of choice.

There is one exception to this recommendation, which we see applying to cases where there is an individual or group of observers working in an area that is too large to be covered in one year but small enough to be covered over a few years. In this case, particularly if the species is one where trends in numbers are likely to be fairly slow, Rotational Sampling may be appropriate. This means that observers cover a different part of the area every year until they have covered the whole, when the same cycle of coverage is begun again (Fig. II.3.3). Such a strict cycle will give much more useful information for the level of effort expended than will the rather haphazard variation in coverage that generally happens in such studies. We do not see Rotational Sampling as being directly relevant at the national level but it may be relevant to local studies that feed into the national monitoring.

(Note that the protocol we present, in which each part of the population is sampled only once per cycle (Fig. II.3.3A), is a special case. More generally, Rotational Sampling involves each group of sample units staying in the sample for a certain number of years, then dropping out for the rest of the cycle (Fig. II.3.3B). The statistics of the general case have been considered by Rao & Graham, 1964).

Recommendation:
* **There is no advantage in sampling with Partial Replacement for mammal monitoring.**
* **Rotational Sampling may be useful for local studies (which may form part of the national programme) but is unlikely to be useful at the national level.**

3.3 Building on previous surveys

Setting up long-term mammal monitoring in the UK does not take place on a clean slate: several species have been the subject of previous surveys. Given the extent of the changes that have occurred in the countryside in recent decades, it is important to be able to compare the status of these species now and in the near future with what it has been in the past. Does this mean that the methods of previous surveys should be adopted without any changes for future monitoring? For methods that produce indices, the answer is clearly “Yes”, since the relationship between an index and absolute abundance depends on the methods used. Where past surveys have determined absolute abundance, the naive answer is “No”, at least if the future monitoring is also based on absolute abundance. Suppose, however, that either the past or the future estimates are biased (whether or not we realise it): unless the bias is identical, the comparison of future estimates with the past is undermined, so even in this case there is an advantage in continuing to use the methods used in previous surveys.

The advantage of sticking with the same methods does, however, need to be balanced against any advantage to be gained from the greater efficiency or greater accuracy of other possible methods. This is a different judgement to make, as it entails weighting the short-term advantages of continuing with the same methods against the long-term advantages of making a change. If such decisions have to be made, it is worth investing time in getting them right. It is also important to make them now, since the longer that suboptimal methods are used the more disruptive will be a shift to better methods. Where changes are to be made, it is important to calibrate old and new methods against each other if one wishes to retain long-term comparability of the data.

It is somewhat easier to avoid disruption if one wishes to change the sampling design rather than the field methods. MMR suggest moving from one design to another by gradual modification, which has the benefit of providing some historical continuity but allowing a better design to be used in the long-term. Where such changes are contemplated, it would be worth simulating alternative designs and ways of moving between them, to aid judging what is best.

An easy modification to design is to change the intensity of sampling (through it may need some ingenuity to do this in the case of systematic sampling unless one envisages changes in intensity) that are particularly large. If previous surveys indicate that the sample sizes used are providing estimates that are more precise than needed, then reducing the sample size may be justified to save resources. It is probably wise to err on the side of caution and to retain a larger sample size than appears theoretically necessary, making further downward adjustments in future if the evidence from the data confirms that this is justified. If one reduces the sample too much, sites can always be drawn back into the sample in future, since modern analytical techniques can cope with sites being temporarily missing; it is however, better to avoid this if possible. Note that changes in intensity may be stratified, whether or not previous surveys have been stratified. For example, regions or land classes judged unimportant may be sampled less intensively.

Recommendations:
* **Use the same field methods as have previous surveys of the same species, unless others are clearly more efficient.**
* **Use the same sampling design as have previous surveys of the same species or change gradually to better methods.**
* **Invest enough time in judging whether or not to change methods or design.**
* **If changes appear desirable, make them now.**

3.4 Within-year variation

Mammal numbers at a site typically vary (often markedly) during the year, generally showing season patterns. There are two ways of dealing with this problem (to avoid within-year variation being added to the short-term between-year variation that interferes with the detection of long-term trends). One is to sample repeatedly over a sufficient period (or on a sufficient number of occasions) throughout the year that seasonal variation is averaged out. The other is to sample at a fixed time each year, though this is not so useful if the seasonal pattern itself varies between years. If one chooses to sample at a fixed time of year, it should be a time when between-year differences are least influenced by year-to-year changes compared to long-term trends: one should not choose times when behaviour or numbers are likely to be highly influenced by weather, for example.

For multi-species schemes, the optimum sampling time may differ between species. An obvious way to overcome this problem is to sample repeatedly over a sufficient span of time that one includes the optima for every species. Generally speaking, it will not then be sensible to ignore the data for each species that come from the suboptimal times, though if these produce very variable data it might be. The decision is best made by analysing the data, to find whether the error variance is lower for analyses based on the whole data set or for those based just on data from the optimal season.

4. NOTES ON STATISTICAL INTERPRETATION

4.1 Estimation and hypothesis testing

Too little attention is generally paid to how monitoring information should be used to trigger conservation attention. In practice, decisions about the conservation status of species based on trends in their abundance or distribution are often simply based on the best estimate of the magnitude of the trends, with little discussion of the reliability of the trend data. If there is any discussion, it is often more concerned with bias rather than precision. This is satisfactory for better-known species, such as many birds, where confidence limits for long-term trends may lie only a few percent around the mean. For other species, concerns may justifiably be raised about the reliability of the estimate: if a species has declined by an estimated 40% but the confidence limits on that figure range from a decline of 63% to an increase of 7% (confidence limits will always be asymmetric on a proportional scale), are we justified in turning conservation concern on to that species?

There are two approaches to judging the significance of monitoring trends. The more traditional is to test them against the null hypothesis of No Trend. Conservation concern is only raised if the actual trend indicates a decline significantly stronger than this. (If the interest is only in declines, a one-tailed test would be applied.) The other approach follows the Precautionary Principle: assume that there is a problem unless the trend is either an increase or a decline that is significantly weaker than the value agreed to be critical - say 25% over 25 years. In this case, a one-tailed test of the null hypothesis that there has been such a decline (or greater) is appropriate. In practice, for species where the monitoring provides only rather imprecise estimates of trends (which will often include the rarer species), the traditional view will result in important declines being missed, the precautionary view in alarms being raised unnecessarily. The best way around this dilemma is to place confidence limits on one's trend estimates. These immediately indicate whether each of the two null hypotheses may be rejected. (90% limits are appropriate if one wishes to apply one-tailed tests at the traditional 5% significance level). If neither test is rejected, one then has a "most likely" scenario (the trend estimate) a "worst case" scenario (the lower confidence limit) and a "best case" scenario (the upper confidence limit); considering the costs and benefits of taking or not taking conservation action under each of these three then allows the conservation manager to decide on the best course.

Recommendation:

* **Rather than testing null hypothesis of No Trend unthinkingly, place confidence limits on trend estimates.**

4.2 Remarks on appropriate statistical models

Considerable attention has been given to methods for modelling animal abundance and presence/absence data over the last few decades, much of it based around data on birds held by organisations like the BTO. Clear recommendations can therefore now be made as to the best modelling approaches with which to analyse monitoring data on mammals.

In general, the methods currently in common use for analysing population trends can be classified as forms of Generalized Linear Model (GLM) (McCullagh & Nelder 1989). These models are flexible, allowing choices of error distribution and link function which are most appropriate for the data being tested. The framework also allows indexing, hypothesis testing and statistical controls to be implemented through the estimation of categorical or continuous predictor effects and tests akin to regression and analysis of variance. The approaches applied to bird count data are reviewed by ter Braak *et al.* (1994) and Thomas (1996); Fewster *et al.* (in press) describe how Generalized Additive Models (GAMs) can be used in similar ways to estimate smooth but non-linear population trends (GLMs are limited to the estimation of linear (on the scale used) or categorical effects). The appropriate link function for count (abundance) data is the logarithmic link, used with a Poisson error distribution, whereas a logit link with binomial errors is appropriate for presence/absence data (McCullagh & Nelder 1989).

4.3 Post-stratification, regression and other modelling as means of increasing precision and understanding.

Suppose that one estimates the change in national abundance over a period. The standard error of the estimate will reflect both sampling error and the differences between sites in the extent of population change. If the samples are divided into categories, such as land-classes or regions, after sampling has taken place (post-stratification) and if there are differences between the categories, then the variation ascribable to between-category differences can be removed using standard analysis-of-variance techniques, thus reducing the standard error of the national estimate. If the differences between categories are small, the removal of this component of the variation may be outweighed by the reduction in the number of degrees of freedom associated with the estimate, so post-stratification should not be undertaken blindly. However, sample sizes for national monitoring schemes are likely to be so large compared with the number of categories used that there is unlikely to be much effect on the degrees of freedom.

An alternative approach is to identify relevant continuous variables that can be measured at each sample site (or are already available in national databases). These may explain part of the between-sample variation in the population change. If so, this component can be removed using appropriate regression techniques.

Both stratification and regression have a further benefit: the differences between strata (or the correlation with the regression variables) may aid understanding of the causes of observed population changes. For example, if Wood Mouse populations remain stable in woods but decline in farms, this may suggest that it is changes in farmland management that are driving the population changes. At the very least, such pointers may guide further research, allowing the causes of decline to be confirmed more quickly than if the habitat difference had not been identified. Modelling may go beyond the simple approaches of stratification and regression to provide even deeper insights into the factors influencing the numbers and distribution of a species (Buckland & Elston, 1993).

Recommendation:

* **Sufficient resources should be available in monitoring programmes to allow refined statistical analyses to be conducted that extract the maximum information from the data.**

PART III MULTI-SPECIES SCHEMES

INTRODUCTION

In this section we review existing and potential multi-species monitoring schemes for British mammals. Schemes which provide data on large numbers of species are unlikely to be ideal for any single species but can nevertheless contribute useful information, especially if several are run in parallel. Multi-species schemes are particularly suitable for use by volunteers because they allow all interesting encounters to be recorded, adding to the personal rewards that volunteers perceive. An important benefit is that areas peripheral to a species' range or where its abundance is low can be monitored without a loss of volunteer motivation when other species are also included in the target group. Multi-species schemes also provide monitoring data cost-effectively (given that the data are of sufficient quality) because the costs of central organisation, data processing, newsletter production, etc., can be shared across budgets allocated to a range of species. Running several multi-species schemes in parallel would provide effective checks for each individual scheme and help guard against problems due to the inadequacies of any one approach for a given species.

Mammal monitoring data are already collected under two national schemes: the Breeding Bird Survey (run by the BTO under a consortium including the RSPB and JNCC) and the National Game Bag Census (run by the Game Conservancy Trust). We review the potential these schemes have as parts of a unified UK mammal monitoring programme in Part III.A. We then outline proposals (for discussion) for five new national multi-species programmes, each of which can add important information cost-effectively to the spectrum of current monitoring schemes, by broadening species and/or habitat coverage. We stress that the schemes we consider should not be taken as alternatives to one another: they would contribute complementary information (but are, equally, not inter-dependent and any number or combination could be selected for further development). Our accounts of the multi-species schemes should be read in conjunction with the individual species accounts (Part IV), in which they are referred to where they can contribute usefully to monitoring in individual cases. The ways in which we envisage single and multi-species schemes contributing to the monitoring of each species are summarised in Part VII.

A. EVALUATION OF EXISTING SOURCES OF DATA

1. MAMMAL MONITORING UNDER THE BREEDING BIRD SURVEY

1.1 Introduction

A trial mammal survey was instigated in 1995 as an adjunct to the BTO/RSPB/JNCC Breeding Bird Survey (BBS) as a result of interest from BBS volunteers in recording the mammals they saw while counting birds. This trial has continued into the 1999 field season and has produced encouraging detection rates and/or sample sizes for a range of mammal species, suggesting that the BBS has the potential to contribute significantly to the monitoring of mammal populations in the UK. Although it is clearly not designed specifically for the monitoring of mammals and thus is not organised as would be ideal for that purpose, the BBS has the benefit of being an extant, national scheme with a large sample size (over 2,200 sites currently surveyed annually). The scheme draws on a pool of volunteers which is likely to be largely different from that tapped by mammal organisations such as The Mammal Society. In this chapter, we describe the background of the BBS and its methods, and then assess its potential value for monitoring mammals.

1.2 The background of the Breeding Bird Survey

The British Trust for Ornithology (BTO) has organised standardised, volunteer-based and national surveys of British breeding bird abundance since the early 1960s. For more than 30 years, the principal national survey was the Common Birds Census (CBC), which has served both to highlight a range of important changes in the British avifauna (see, e.g., Marchant *et al.* 1990; Gregory & Marchant 1996; Siriwardena *et al.*1998) and to provide a basis for the development and testing of new and better statistical methods for the identification of population trends (see, e.g., Mountford 1985; Peach & Baillie 1994; Fewster *et al.* in press). Although the CBC provides good data on bird populations breeding in farmland and woodland, and the farmland plot sample (at least) is representative of the wider habitat in south and east Britain (Fuller *et al.* 1985), its methods are labour-intensive (limiting the likely total plot sample size) and it has large gaps, nationally, in its habitat and regional coverage.

The Breeding Bird Survey (BBS) was introduced in 1994 as a replacement for the CBC (although the two continue to run in parallel for calibration purposes) which avoids the sampling biases associated with the latter while incorporating simpler fieldwork methods and maintaining statistical power. The method used by the BBS is described in detail below, but basically consists of two counts of a 2km line transect, within each of a *random* sample of 1km grid squares, along which all birds seen or heard are recorded. The size of the sample of BBS survey squares has now exceeded that shown by simulation studies to be sufficient to provide the same statistical power as is provided by the CBC, and it continues to grow annually (Gregory & Baillie, *in press*). Another important feature of BBS is the habitat data that are collected along the survey transects according to standardised methods and which are computerised routinely. These data allow bird abundance to be related to habitat characteristics so that a range of environmental hypotheses can potentially be tested.

1.3 The methods of the BBS

BBS survey design and methods are described in full elsewhere (Gregory & Baillie *in press*; Gregory *et al. in press*), so we only summarise them here. The BBS is based on a formal sampling strategy under which 1km squares to be surveyed are selected as a random sample, stratified by regions defined by observer density. Observers are distributed with human population density and are therefore harder to recruit in more remote areas; the stratification allows for this without introducing bias. Survey squares are allocated to observers showing interest in the BBS through staff at BTO headquarters and the BTO's national network of volunteer regional representatives.

Volunteers visit their survey squares three times in each year: one visit to record habitat and (re-) establish a transect route and two visits on which birds are recorded. Bird-recording visits are made in each of an early (April to mid-May) and a late period (mid-May to June), with at least four weeks between them, and visits begin between 6.00 and 7.00am to coincide with maximum bird activity while avoiding the daily peak of birdsong. Transect routes ideally comprise two parallel lines within a square, each 1km long, although terrain and access mean that deviations from the ideal are common. Each 1km transect is divided into five 200m sections which form the units for bird and habitat recording. Visits each tend to take around 1½ hours.

All birds seen and heard while walking transects are recorded using three distance band categories (<25m, 25-100m and >100m on either side of the transect); birds in flight are assigned to a separate category. Habitat is recorded annually using an hierarchical coding scheme (Crick 1992) which is used (with minor modifications) in all BTO monitoring schemes. Space is provided on the recording form for both a primary and a secondary habitat to be recorded for each 200m transect section.

1.4 Mammal recording within the BBS

To date, mammal recording in the BBS has proceeded only on a trial basis: the approach in use is therefore subject to revision and there is scope for improvements to be incorporated. The BBS approach attempts to take account of the fact that mammals are generally harder to observe than birds (in particular, they tend to be more silent, are often more cryptic and are often less tolerant of the presence of human observers). Currently, counts are solicited for the 17 species which are most likely to be seen and for which useful data are most likely to be collected (see Table III.A.1.1). Space is also allowed on the recording form for additional species to be recorded (which we do not consider further here). Volunteers are asked to provide one of two forms of data: numbers of individuals seen during the early and late transect surveys and an additional qualitative assessment of whether a species is "known to be present in the square". The latter encompasses any identifications made which are not on the formal transects, for example, identifiable field signs seen during transect surveys and sightings made during the habitat recording visit to the square. Zero returns, where no evidence of mammals was found, are encouraged, because these are essential for unbiased monitoring, but their rates of submission could still be improved. More than 75% of BBS volunteers have provided information on the mammals in their survey squares since the inception of the trial survey.

1.5 The Mammal Species Recorded

Three years of BBS mammal data, from 1995-1997, were computerised and available for analysis as of spring 1999. Restricting ourselves to considering species' presence or absence for simplicity, we investigated the sample sizes provided by the survey. A square was assumed to have been surveyed for mammals if either at least one species of mammal was counted (or known to be present) or a return reporting no evidence of mammals was received. A species was treated as present if either a count was provided in either of the two survey visits or the species was recorded as "known to be present". A species was recorded as absent if mammal data had been submitted for a square (including zero returns but not including squares surveyed for birds from which no mammal forms were received) and the species was not reported. Note that "absence" here refers to non-detection rather than, necessarily, to true absence from a square (or from the habitat through which a transect passes).

The BBS is intended to survey the same squares year-by-year, but there is some inevitable turnover as volunteers are recruited, drop out or move. A total of 2,231 squares yielded some information on the presence or absence of mammals during the period 1995-1997, annual contributions rising from 1,334 in 1995 to 1,874 in 1997. The numbers and proportions of squares occupied in each of the three years are tabulated, for seventeen species, in Table III.A.1.1 To quantify the precision of the estimated proportions presented, we computed their standard errors (s.e.) using the following standard equation (assuming that the number of squares occupied is a binomial variable with probability p: Cochran 1977):

$$s.e. = \sqrt{\frac{p(1-p)}{(n-1)}}$$

where n is the total number of squares surveyed.

Rabbit was easily the commonest species, being reported as present in about 70% of squares for which information on mammals was given. Other species reported in more than 10% of the squares were Mole, Brown Hare, Grey Squirrel, Red Fox and Roe Deer. Small insectivores, rodents and mustelids, although likely to be widespread were reported only from a small proportion of squares, almost certainly because of their inconspicuous nature. The commonness with which the various species are detected within the BBS sample has important implications for the utility of the survey for the detection of changes in their populations: within the range of values found here, higher proportions of the total number of squares are better. These issues are explored in detail under "Statistical Power" below.

Geographical patterns in the occurrence of mammal species across Britain are often of interest, especially for species whose ranges are believed to be increasing (e.g. Reeves' Muntjac: Corbet & Harris 1991) or contracting (e.g. Red Squirrel: Corbet & Harris 1991). As a random sample of the landscape in Britain surveyed using standard methods, the BBS presents a powerful tool with which to examine the national ranges of British mammal species and a significant advance on the collation of opportunistic records used previously (Arnold 1993). Using DMAP For Windows mapping software (Morton 1993-1996), we plotted the spatial distribution of mammal records, contrasting examples of which are contained here. The Rabbit (Figure III.A.1.1) is shown to be almost ubiquitous, the Scottish highlands being the only large area in which most squares lack recorded presence. This is no doubt a major contribution to the significant spatial variation found in the generalized linear model fitted to the data for this species (see below). The scarcer Brown Hare was also not recorded from much of Northern Ireland, south-east and south-west England (Figure III.A.1.2). Reeves' Muntjac was first introduced to Bedfordshire but has since spread widely, both naturally and through many additional releases (Figure III.A.1.3). The distribution of the Red Squirrel records shows clearly that it is more widespread in eastern Scotland and northern England, but in all three years considered, only isolated locations elsewhere are found to contain the animal (Figure III.A.1.4).

1.6 Estimating Population Changes and Population Trends

Simple measures of changes in reported presence

Several possible options exist for identifying changes in BBS mammal data, each addressing subtly different questions with varying statistical sophistication, but all being generally definable within the framework of generalized linear models (GLMs: McCullagh & Nelder 1989). Using numerical count data would extract the maximum amount of information on mammal population changes from the BBS, provided that the number of animals counted is related to true abundance, but the sample size of squares available is maximized if we consider only the estimation of presence and absence. For such analyses, count records are thus considered to be simple "presences" and combined with the "known to be present" data. The proportions of squares where a species is present can be compared between years (or between regions or habitats) using a contingency table approach and referring the test statistic to χ^2 tables. Alternatively, in a GLM formulation, we can test whether the variation between years is significant by comparing a model with annual proportions and one with a single, time-invariant estimate derived by combining data across years, using a likelihood ratio test. Since the data to be modelled are proportions, we employ GLMs with a binomial error distribution and a logit link function.

Applying the likelihood-ratio test approach to data from 1995, 1996 and 1997, we found that proportions had varied significantly between years (at the 5% level) for nine of the 17 mammal species named on the recording form: Hedgehog, Mole, Common Shrew, Grey Squirrel, Brown Rat, Red Fox, Stoat, Weasel and Badger (Table III.A.1.2). Many of the smaller animals in particular were reported noticeably less frequently in 1995. This may be due to increasing effort by observers to record signs of mammals as the BBS has developed or to subtle changes in the recording instructions during the trial survey (see Discussion), so we repeated our analyses restricting our attention to 1996/1997. The analyses omitting the 1995 data gave rise to similar results for most species, but the inter-annual differences became non-significant for Mole, Rabbit, Brown Rat, Stoat, Weasel and Badger (Table III.A.1.2), suggesting that the effects identified previously for these species could have been artefacts of the change in survey methods. Note, however, that fewer significant results would be expected *a priori* from comparisons of two years as opposed to three.

The analyses of proportions described above are crude in the sense that they take no account of the matching of BBS survey squares between years: they assume that independent random samples of squares are drawn in each year. However, although there is some turnover in the BBS sample, the majority of squares *are* sampled in consecutive years. Analyses of the data can therefore make use of this to remove the need for the statistical consideration of square-specific effects (i.e. random biases caused by habitat-specific and geographical variations in density) which is inherent in the simple comparison of proportions. A method for doing this for pairs of years is presented by Cox (1970): data from any two years to be compared (not necessarily consecutive ones) are matched by square and squares where a species has been reported as either present or absent in *both* years are omitted. The remaining squares all therefore experienced a change in the reported status of the species, either from present to absent or from absent to present. The relative numbers of squares of these two types form sufficient statistics for the estimation of the difference in detection rates between years (Cox 1970, p. 56). A significant change in the overall proportion of squares reported to be occupied is indicated if significantly more than half of the squares where status has changed experienced change in one or other direction. Specifically, this test is based on the following observation: if the probabilities of detection at site j in the two years under consideration are p_j and $p_j+\Delta$, then N_{pa} has a binomial distribution (m, p) with binomial denominator $m=N_{pa}+N_{ap}$ and probability $p=e^{\Delta}/(1+e^{\Delta})$, where N_{pa} and N_{ap} are, respectively, the numbers of squares changing from present to absent and from absent to present (Cox 1970).

To evaluate the effects of the difference in approach, we used the matched squares method to conduct analogous analyses to those described above comparing the data from 1996 and 1997 for each species. The results are presented in Table III.A.1.2 . The results for most species were more significant than had been found in the unmatched analyses and inter-annual differences for seven (as opposed to four) species were significant at or around the 5% level (Mole, Common Shrew, Brown Hare, Grey Squirrel, Red Fox, Badger and Roe Deer: Table III.A.1.2). The variations between analyses in the species identified as having significant inter-annual differences in numbers of occupied squares show where a difference or a lack of difference identified by the unmatched analyses could reflect the influences of spatial biases in the sample of survey squares.

The number of squares in which changes in status had occurred varied from 30 (Red Squirrel) to 382 (Red Fox) (Table III.A.1.2). Clearly, the utility of the square-matching approach depends on the number of squares which have been surveyed in both of the years compared: survey square turnover means that this will fall as the years compared are moved further apart in time. Note, however, that long-term turnover should be less of a problem (notwithstanding any observer effects) than has been the case for the CBC because new surveyors for particular squares will be sought to replace any who drop out.

The potential of Generalized Linear Models for presence/absence data

Extending the methods applied within the GLM framework and using likelihood-based testing would permit the fitting of a range of more flexible models, and therefore the testing of various hypotheses of interest. A logical extension of the pairwise square-matching approach is the linking of data from several years via square identity (together with enough flexibility to allow squares not to have to have been surveyed in all years). This can be done in GLMs incorporating the estimation of a set of square-specific probabilities of species presence in addition to year-specific probabilities: the prediction of the model for a given square in a given year is then the sum of the "location effect" and the "time effect". Analogous models now routinely form the basis for analyses of population trends in CBC and Wetland Bird Survey data, as well as in other bird monitoring schemes worldwide (ter Braak *et al.* 1994; Thomas 1996). Under such models, the probability of detection of a species in a survey square varies between locations, but this proportion fluctuates between years in a parallel fashion in each location.

In the methods currently used to analyse CBC data, location or site effects are usually fitted as a categorical variable, i.e. survey sites are not assumed to be inter-related (Mountford 1985; Peach & Baillie 1994; Fewster *et al*. in press). An alternative is to characterise sites by means of a few biologically meaningful variables such as latitude, longitude, altitude, area of woods, etc.; i.e. to assume that site-specific variation derives from a combination of these influences, and therefore to estimate far fewer parameters during model fitting. This approach, using geographical and habitat covariates in place of categorical site effects, may be obligatory for the application of GLM-based indices for mammal monitoring under the BBS: with currently available computing resources, it would be prohibitively time-consuming routinely to conduct long-term analyses incorporating over 2,000 separate "location" effects. The covariate approach also provides greater numerical precision. It is also important to note that formal statistical methods such as likelihood-ratio tests allow the significance of any relationships with covariates in GLMs to be assessed. This permits both the identification of a parsimonious model to explain any spatial variation and the testing of covariate-based hypotheses.

As an example of the application of GLMs incorporating both spatial and temporal variation, we analysed BBS Rabbit data using the county in which a survey square was found (as a categorical variable) as the spatial component. Each presence/absence record was categorised both by year and by county: survey squares were located in a total of 74 counties. Both "year" and "county" were then fitted as categorical predictor variables in a Generalized Linear Model and the significance of their effects tested by comparing models omitting each in turn with the full model using likelihood-ratio tests. Spatial variation, expressed at the county level, explained a significant proportion of the variation in the probability of detection for Rabbits ($\chi^2_{73} = 773.27$, $p<0.01$). However, temporal variation was not significant at the 5% level ($\chi^2_2 = 4.84$, $p=0.09$), although the annual indices of rabbit presence suggest that a decline may have occurred: on a logit scale, the indices were 0.1807 (1995), 0.0542 (1996) and zero (1997: the point relative to which the other years are scaled).

The GLM framework allows a range of additional models to be fitted; for example, interactions between year and geographical category could be added, permitting a test of whether proportions of squares occupied vary differently between regions, rather than in parallel as assumed above. As implied earlier, relationships between distribution and climatic or habitat variation could also be explored using either continuous or categorical variables.

Analysing changes in abundance

The GLM approaches applied to BBS and CBC data have focused on the estimation of changes and trends in abundance (see, e.g., Field & Gregory 1999). Similar methods can be applied to mammal data from the BBS, although rather fewer species can be meaningfully approached in this way than was the case with analyses of presence/absence data. However, where the data *will* support models of abundance and counts of mammals from BBS transects are likely to be good measures of true abundance, these models would provide much more sensitive methods for revealing population changes than would the presence/absence method. GLMs used for modelling variations in abundance are generally formulated using a logarithmic link function and a Poisson error distribution (ter Braak *et al.* 1994; Pannekoek & van Strien 1996), but otherwise the principles are consistent with those applied to presence/absence data as described above. We have investigated models of the abundance data collected by BBS volunteers for Brown Hare: this is probably the species to which the BBS methods and sample of survey squares are best suited (see Part IV.5).

We fitted a GLM with log link and Poisson errors to BBS Brown Hare data for 1995-1997, allowing for categorical year and county effects (as with the model of presence/absence data for Rabbit, this approach was much less demanding of computer resources, reducing the number of independent location effects to be estimated from around 2,000 to around 70). Counts were taken as the total of early and late counts to avoid problems with non-integer mean values. Likelihood-ratio tests showed that both the year and county effects were highly significant, revealing both temporal and spatial variation on Brown Hare abundance. Abundance indices (antilogs of the fitted year effects) for the three years were as follows: 3.78 (95% confidence interval 3.41-4.18: 1995), 3.31 (2.99-3.67: 1996) and 3.86 (3.49-4.27: 1997); from the relevant likelihood-ratio test, χ^2_2=363.9, P<0.01. These results therefore show no sign of a trend in Brown Hare numbers, but that abundance underwent a transient fall in 1996. Note, however, that we have made no attempt here to compensate for any effects of changes in survey methods.

1.7 Statistical Power

A fundamental issue in the design of any monitoring scheme is the power with which it is able to detect the changes that are of interest: sample sizes of survey data points must be large enough for change of the magnitude required to be detectable in the appropriate period of time. Equally, an efficient monitoring scheme would not expend effort collecting more data than is required to meet its objectives.

Presence/absence data

We have used standard formulae and simulation analyses to investigate the power of the analyses of BBS presence/absence data outlined above, under scenarios with various postulated population trends and survey square sample sizes. Sokal & Rohlf (1995, p. 768-769) provide a formula for the estimation of the sample size required to detect, with a given power, a given difference (at a predetermined level of significance) between two proportions. The method applies to random samples of the proportions compared, so is applicable to the "unmatched" survey squares approach. We applied this formula to simulated changes (declines) of 10, 20 and 30% in the "true" proportion of squares where presence is detected, varying the postulated power of detection and investigating a range of starting proportions from 0.05 to 1.00. The results, given a required P-value of 0.05, are shown graphically in Figures III.A.1.5 - III.A.1.7. The current BBS sample size of around 2,000 squares (more have been surveyed for birds in recent years but fewer for mammals) would detect 10% declines in reported presence with a power of between 60% and 90% given starting proportions of (approximately) between 0.3 and 0.5 (Figure III.A.1.5). (Note that, for a given starting proportion, there is higher power associated with the detection of "true" increases than declines since power increases as a function of the magnitude of both proportions being compared, irrespective of their temporal order.) These figures correspond roughly to or are exceeded by the proportions of squares where Rabbit, Brown Hare, Grey Squirrel and Red Fox have been reported as present in recent years. A decline of 20% can be detected given the current BBS sample size and starting proportions (approximately) between 0.1 and 0.2 across the range of powers tested (Figure III.A.1.6). A postulated 30% decline can be detected with still smaller starting proportions (below 0.1 for powers of 90% or less) (Figure III.A.1.7). The observed BBS sample sizes for different mammal species (Table III.A.1.1) therefore suggest that the survey could provide useful monitoring data at least for Hedgehog, Mole, Badger and Roe Deer in addition to the four more frequently reported species. In addition, any increases in the BBS sample size in the future can only increase the power of the survey to detect smaller changes.

The above process considers the significance of change between two chosen years. To assess the power of BBS data (with squares unmatched) to detect a gradual, long-term decline, we used a simulation approach based on 10% declines over 10 years in the proportions of squares with reported presence from starting values 0.3, 0.1 and 0.03. The proportion 0.3 corresponds approximately to that in which Brown Hares were found, and the others represent values from which we would expect the detection of declines to be progressively more difficult *a priori*. We then extended the analyses to consider a 25% decline from a starting proportion of 0.03 over periods of 10 and 25 years, as described below.

We simulated artificial mammal presence/absence records for 2,000 BBS survey squares over periods of 10 or 25 years, as appropriate. Given a pre-determined linear decline over the one of these time periods, each square was recorded, at random, as having a species "present" or "absent" with the probability for each year which would provide the overall decline required. A GLM containing a linear trend in the probability of detection was then fitted to the simulated presence/absence data. This process was repeated 100 times, generating 100 data sets from the same hypothesized model. To investigate the power with which the pre-determined linear decline could be identified, we tested the significance of the linear time trend term in each replicate analysis. The proportion of the 100 replicates in which a decline significant at the 5% level was found provided a measure of the statistical power of this modelling approach. The results are summarised in Table III.A.1.3.

The results show that the power of the simple GLM approach to detect a 10% decline over 10 years is high, given a starting proportion of 30% (similar to that found in real BBS data for Brown Hare, Grey Squirrel and Red Fox). However, the power declines rapidly as the starting proportion is reduced (Table III.A.1.3). Nevertheless, a larger, 25% decline was successfully detected in the majority of cases even from a very low starting proportion (3%), especially over a longer time period (25 years: Table III.A.1.3). We have suggested elsewhere that this is a key level for the detection of trends, and it is one which has been adopted for the generation of lists of Birds of Conservation Concern (Gibbons *et al.* 1996): it is extremely encouraging that BBS data could have the power to detect such changes for almost all of the species listed in Table III.A.1.1.

However, BBS presence/absence data seem to have limited utility for the detection of shallow population trends from low probabilities of detection (below 0.2) such as have been found for many species (Table III.A.1.1). Clearly, however, the key issue is the magnitude of population change that is of interest, and therefore that which the BBS would be required to detect. In this context, a 10% decline over 10 years may not be a significant cause for conservation concern for common or widespread species.

Abundance data

We also carried out a rough assessment of the power of BBS data to detect changes in abundance, rather than presence, using a simulation approach. Our simulations were based on a log-linear Poisson regression model fitted to 1995-1997 BBS data for Brown Hare. This GLM included categorical county and year effects: we used the fitted county effects and the year effect for 1997 as the basis for our simulation. We assumed that the squares visited in 1997 were visited in each of the ten subsequent consecutive years, and generated randomised data (counts of hares) for each square in each year. To do this, we assumed that the county effects remained at their estimated values throughout, but made the year effect decline annually by a constant amount, such that it had declined by 10% at the end of the tenth year. We ran this simulation 100 times, producing 100 replications of a scenario in which the population varies spatially but declines at the same rate throughout. The results showed that we could detect the 10% decline at the 5% level (via a likelihood-ratio test) in all 100 of our replicates, showing that with a data set of the size of the Brown Hare one, the BBS has very high power to detect a population change of this size. We should note, however, that the scenario we modelled featured a linear decline, which is highly unlikely to occur in practice: one might be able to *fit* a linear trend to population change, but the real changes will not be smooth because of inter-annual variation due to sampling error and effects which are independent of those driving the long-term trend. This means that we have estimated power under the best of all conditions. Nevertheless, the 100% success rate is encouraging, and it suggests that more detailed simulation work would be worthwhile.

1.8 Waterways Breeding Bird Survey

The line transects used by the BBS, being random in orientation with respect to the landscape, are not optimal for the monitoring of birds in linear habitats such as rivers and canals. The recognition of this fact led to the organisation of a pilot Waterways Breeding Bird Survey (WBBS) in 1998, in which BBS methods were adapted for use on linear waterways (Marchant & Gregory 1999). The key differences between WBBS and BBS were that WBBS sites were selected at the tetrad (2x2km square) rather than the 1km square level, that transect sections were 500m rather than 200m in length (to match the Environment Agency's River Habitats Survey) and that total transect lengths were allowed to vary up to a maximum of 5km (ten 500m sections). Routes within tetrads followed the course of the waterway and habitat recording was extended to allow recording of the characteristics of the waterway itself as well as the surrounding terrestrial habitats. Mammal recording was conducted exactly as in the BBS: either presence/absence or a pair of counts (early/late).

Observers surveyed 103 random WBBS stretches in 1998, returning mammal data on 93 of these stretches. The mammals recorded on more than 1% of survey stretches are listed in Table III.A.1.4. The species found most commonly on WBBS stretches were similar to those reported most commonly under the BBS, but the comparatively small scale of WBBS means that it is unlikely to be able to contribute significantly to the monitoring of such species (except, perhaps, populations specific to riparian habitats). Of more interest is the potential for WBBS to monitor three key riparian species: Otter, American Mink and Water Vole. The proportions of squares from which these species were reported (Table III.A.1.4) were large enough to suggest that WBBS could contribute to their monitoring. To check this, we ran further simulation analyses of the kind summarized in Table III.A.1.2, investigating the power to detect a 25% decline over 25 years from a starting proportion of stretches with a species present of 0.1 (i.e. approximately the figure found for Otter, American Mink and Water Vole). Taking a significance level of 5% and putative WBBS sample sizes of 100, 200 and 400, the trend was detected in 8%, 17% and 29% respectively, of the simulations conducted. Therefore, even with a significantly increased WBBS sample size, trends of the magnitude which are likely to be of interest are unlikely to be detectable in a majority of cases.

1.9 Discussion

The suitability of BBS field methods

Our exploratory analyses show that mammal monitoring through the BBS has the potential to provide a wealth of data which are *statistically* adequate to contribute significantly to a national monitoring scheme. The principal asset of the BBS in this context is that a large number of randomised sites are being surveyed regularly already using standardised methods, allowing mammal data of a high statistical quality to be collected at minimal cost. However, as described above, the BBS was designed for bird monitoring and its methods do not represent the ideal for any mammal species. Transect methods, in general, represent efficient and readily standardised approaches to the sampling of many habitats for many taxa, and we have recommended them for British mammals elsewhere in this report (see Part III.B.3, III.B.4 and III.B.5). Visual counts of individuals are not, however, appropriate as a means of surveying many mammal species, so considering mammals, in general, effectively as supplementary bird species (in terms of field recording) is unlikely to be feasible. Doubts may also exist as to the reliability of birdwatchers as identifiers of mammals, although these might be eased by the provision of a simple identification guide. Nevertheless, more visible species which are commonly detected where they occur by BBS observers, such as Brown Hare, Grey Squirrel and Reeves' Muntjac, could yield useful abundance data from BBS counts. In addition, standardised recording of presence/absence through signs or off-transect sightings (see below) can add further information for the less visible species. The BBS clearly cannot provide count data for these species.

The principal weakness of the BBS for mammal monitoring, even for visual species such as Brown Hare (for which line transect sighting methods can be recommended as the best monitoring tool: see Part IV.5), is that the timing of the survey is sub-optimal. Vegetation growth is substantial by May and (especially) June, making the detection of cryptic and silent animals more difficult, both on the ground and in trees. A second, potentially more serious problem, is that mammal productivity can vary greatly between years, and distinguishing the adults and juveniles of species, including Brown Hare and the squirrels, can be difficult by May. This means that apparent changes in abundance from BBS counts could just represent fluctuating productivity, masking the true trends in the abundance of adults. For each of these reasons, winter or early spring transect surveys would be preferable (see Part III.B.3). However, neither problem makes BBS mammal data invalid. Provided that problems with detection caused by vegetation growth do not affect the form of the relationship between detection rates and true abundance, lower rates will have no effect on population indices: any inter-annual biases caused by weather can be controlled for in GLMs using variables such as temperature or rainfall. Differences between annual counts or probabilities of detection, which are due to variation in annual productivity, will essentially only produce noise around any long-term population changes, thus making these underlying changes in adult numbers harder to detect. This will, therefore, only be a problem if this noise prevents the detection of "true" trends of the magnitude and with the power required. The effects of variable productivity will vary from species to species and will be greatest for the less visible species which cannot be counted effectively. Winter Transect Surveys (see Part III.B.3) run in parallel with the BBS would help to show for which species the BBS provides data which are not severely influenced by the effects of annual productivity and vegetation growth.

Presence/absence monitoring

As we have already stated, surveys of animal abundance provide more powerful monitoring tools than do those of changes in status (presence/absence), all other factors being equal. However, direct counts of many species are not feasible and the BBS allows useful direct counts to be made for only a few species.

The utility of BBS presence/absence data clearly depends on how the data can be interpreted. An obvious, but important, point is that "absence" reflects non-detection, which is not necessarily true absence. It is then critical to what extent detection rates reflect abundance within BBS squares: ideally, detection of presence would occur at a certain, fixed density of a given species. In reality, visual detection will vary with vegetation and productivity as discussed above, and will also incorporate a strong stochastic component, especially for rarely seen species such as Stoats and Weasels. The extent to which presence/absence data are useful then depends on how much of the variation in detection rates is due to real changes in adult abundance. Given a sufficiently large sample size, these real changes should be detectable, but we have no data to determine whether the current and achievable BBS sample size approaches this goal. Calibration of the BBS against other survey schemes which can be considered to provide more reliable information for particular species could be used to indicate whether changes in the BBS data reflect real population changes. The winter and sign transect approaches we discuss elsewhere (see Parts B.3 and B.4) could provide the relevant information for many species. Rarely seen species might need calibration work based on intensive Capture-Mark-Recapture work in a sample of BBS squares.

Key issues in interpreting records of presence in BBS data are the criteria by which it is assessed and what it is that these criteria really mean. The wording of the instructions for volunteers changed after 1995, giving greater emphasis to the potential use of signs to assess presence, and this may explain the relatively low detection rates for many species in 1995 (Table III.A.1.3). However, the instructions have remained consistent in general, asking for records of species which were “present, but not on transect” (revised to “known to be present in square” for 1998 and 1999). These records can include animals which were “seen on a reconnaissance visit to the square” or for which the observer has seen “obvious signs of presence during fieldwork” (including tracks and signs). The principal problem with these methods is that there is considerable scope for variation in interpretation. For example, not all observers will be able to recognise Red Fox scats and others may not consider scats to be “obvious”. More seriously, “known to be present” could be taken not to require direct evidence within the current calendar year. The level of interest observers have in mammals will also influence the extent to which they look out for signs, although this will cause only noise, not bias, unless this interest changes with time. A further problem is that the methods for reconnaissance visits are not standardised in the same way as those for bird recording. Opportunities for mammal recording could therefore vary from a cursory visit to check for changes in habitat to an in-depth assessment of changes in, say, agricultural field uses, perhaps with special attention being directed towards looking for mammal signs.

All of these problems could be solved by improvements to the instructions for volunteers and/or the recording form used. Several non-exclusive courses of action are available. First, the data solicited could be restricted to sightings and signs which we can be confident volunteers can recognise (such as fresh molehills). Second, information on the type of evidence used to assess presence could be requested (sightings, scats, roadkills), ideally also incorporating an indication from the surveyor as to whether they looked for each type of evidence. Third, estimates of the time spent on habitat recording visits would assist interpretation. Fourth, tighter controls on the sources of evidence that are admissible, such as sightings during reconnaissance visits but not on other occasions that surveyors might (for example) drive through the square, would help to standardise the data. Whatever improvements are incorporated, it is important that they do not increase the complexity of the BBS unduly: its primary purpose will remain the monitoring of bird populations and any additions which would affect its efficiency in this role should be avoided.

Developing trend analysis methods

Substantial amounts of research have been conducted into methods of analysing trends in CBC and other bird abundance data, generally focusing on methods which are related to the GLM approaches described above (Mountford 1985; Peach & Baillie 1994; ter Braak *et al*. 1994, Thomas 1996; Field & Gregory 1999; Fewster *et al*. in press). These methods include sophisticated uses of covariate modelling to test hypotheses and the modelling of non-linear trends which allow realistic long-term population trajectories to be estimated (James *et al.* 1996, Fewster *et al.* in press). The wealth of options available and the flexibility of GLM-based approaches mean that the brief explorations of analyses of BBS mammal data presented above only scratch the surface of what is possible. If it is considered that BBS data can contribute significantly to UK mammal monitoring, its potential will be greatest if time and resources are made available for detailed investigations of the usefulness of the various modelling options to be conducted. Three months of staff time for a biostatistician to explore the possible methods ought to be sufficient.

Extensions to the BBS

We have already identified that the BBS has failings as a monitoring scheme for most (if not all) mammal species and that winter and sign transect surveys could be valuable, both in their own right and for the calibration of biases in the BBS data. These new survey options are discussed in detail in Parts B.3 and B.4, but it may be worth considering extending the BBS or building on the framework it provides to contribute to such new schemes. Three potential extensions are: (1) to add mammal-only survey squares to the BBS sample to improve sample sizes by recruiting volunteers who are not birdwatchers; (2) to ask volunteers to make an additional recording visit to their squares in late winter to record mammals; (3) to ask volunteers to make specific mammal sign-recording visits to their squares. The second and third potential extensions would depend on the willingness of bird volunteers to increase their efforts; the likelihood of them actually doing this might best be assessed by a questionnaire survey. Some winter visits to existing BBS squares are likely to be conducted in the future as a part of separate bird monitoring schemes, and so could be used to provide a limited amount of control mammal data.

Conclusions

We believe that the BBS has the potential to contribute significantly to the monitoring of many UK mammal species. It would have particular potential value if it were combined with other multi-species schemes run by a dedicated mammal monitoring organisation. The results of our analyses of its statistical power for mammal monitoring are encouraging. It is very likely to provide useful information for species such as Brown Hare, Grey Squirrel, Roe Deer and Reeves' Muntjac and may play a central role in their future monitoring (see individual species accounts). It should also at least provide ancillary information for many other species since the large sample size of survey sites and wide geographical coverage argue in its favour against any methodological failings. There are important issues in terms of the sensitivity of presence/absence monitoring to changes in abundance and in biases which the bird recording design might induce, and these should be addressed using alternative, more robust survey methods, but they do not invalidate BBS mammal monitoring in principle. The WBBS may also contribute data for key riparian species, but as yet it has no guaranteed long-term funding and the sample size it can generate is unlikely ever to be large, so it would be wise not to plan mammal monitoring schemes around the WBBS at present.

The additions or improvements to mammal monitoring under the BBS that we discuss above will need formal development, and the administration of data collection and collation additional to that already done for the bird data will need to be funded (see Part VII.2.1). Annual data management work ought to take no more than about 35 days of additional staff time; we would suggest that the development work ought to take no more than 55 days. We suggest that this work would best be done by the BTO staff who currently run the scheme, but that all interpretation of the data, which would be done in conjunction with data from other schemes, should be done by mammal specialists, hopefully within a dedicated mammal research organisation. The design of the final recording form for BBS mammal surveying and the details of the methods to be used, species coverage, etc., would also best be designed and organised in conjunction with the input of mammal specialists.

2. GAME BAGS

2.1 Introduction and background

Over the many years for which British country estates have been managed for shooting and hunting game, many of the gamekeepers and managers responsible have kept records of the animals killed both as quarry species and in the course of predator control. Game Bag records therefore are among the earliest long-term ecological time-series data, and as such their potential importance for monitoring contemporary environmental changes in an historical context has long been recognised by the game research community (Tapper 1992). In this section, we consider the contribution that bag return data can make to mammal monitoring in the UK, reviewing the pros and cons of the data in their current form and suggesting amendments which would increase the utility of the information collected.

A concise history of bag recording in the UK is given by Tapper (1992). The Game Conservancy Trust (and its forerunner organisations) masterminded the collation of shooting and gamekeeping bag data into a National Game Bag Census, which has been in operation since 1961. The census has been improved over time, with computerisation being introduced in the late 1970s, at which time the scheme was also expanded to provide better coverage of the uplands, especially in Scotland. Historical data from before 1961 have also been solicited from estates with long-term records of their own, and these have been integrated with the more recent data, extending the time series back to 1900 for many species. Such long runs of well-documented data on many predatory and game species clearly make Game Bag returns an important potential source of future mammal monitoring information.

Additional bag record data are collected by The British Association for Shooting and Conservation (BASC), primarily through one-off surveys of the organisation's gamekeeper and deer-stalker membership. These data do not provide long-term or continuing information, but could contribute to the meeting of monitoring objectives through repeat, "snapshot" surveys. BASC's membership also represents a valuable potential source of volunteers who might contribute to monitoring both through their bag records and through specific survey work.

2.2 The case for Game Bag data as a mammal monitoring tool

There are two principal reasons why Game Bag records could make a valuable contribution to mammal monitoring: the long runs of historical data and the fact that the data are already being collected according to established protocols (in the National Game Bag Census). The former will provide continuity and an historical context in which future monitoring information can be viewed. Such a context could be vital: historical population levels can provide us with targets for conservation action; they also allow the severity of recent population trends to be assessed in the light of conditions before the onset of any environmental changes causing contemporary concern. Integrating an established monitoring programme into future plans would allow data to feed directly into the consideration of policy and the development of conservation and management objectives without the need for pilot studies or trial surveys testing proposed methods. It might also be expected to be continued independently of any additional funding.

Mammal recording via Game Bags has several other key advantages. The National Game Bag Census already covers Great Britain effectively (but not Northern Ireland), including recording from estates in more remote areas such as the highlands of Scotland. Other forms of survey will always be more difficult in such areas, either because it is more expensive to survey them professionally or because volunteer densities are correspondingly low where human populations are sparse. Problems with access can occur for wildlife surveying if the surveyor does not have the confidence of the landowner or if the latter perceives that a conflict of interest exists between the survey and his or her shooting or agricultural interests. The use of records directly connected to game interests should avoid this problem and therefore maximize the likely cooperation from landowners. It is also notable that landowners who exploit game populations commercially have a vested interest in effective monitoring and so should, in principle, be cooperative.

The National Game Bag Census includes a record of the number of gamekeepers working on estates and it should be possible to enhance this recording of keepering effort. It is also possible that some of the concern about variable effort in the "sampling" of game populations (see below) does not represent a serious problem, at least for quarry species, because the numbers killed will tend to represent (more-or-less) a constant proportion of the population (40% has been suggested for Brown Hares: Tapper & Stoate 1992, but see Hutchings & Harris 1996) if a viable population is to be maintained by management. Game managers will use approximate judgements of game population sizes to determine the number of days' shooting that can be supported, improving the likely correlation between the sizes of Game Bags and abundance. However, improved recording of zero yields (no shooting) would be required if this approach were applied in population monitoring. Further expansion of the national census could perhaps be achieved by targeting individual shooters (for example, members of BASC) as well as gamekeepers and estates, and by developing coverage in Northern Ireland.

In summary, the following factors recommend the use of Game Bag data for mammal monitoring:

- long runs of historical data provide an important historical context that is unavailable elsewhere;
- data collection protocols are established and a national scheme is already funded, promising future continuity at low cost;
- the National Game Bag Census already covers much of Great Britain effectively, including some general problem areas;
- game and landowner interests are close to the data recording process, avoiding problems with cooperation and access which could affect other schemes;
- improvements to the recording of "sampling" effort and coverage (i.e. in Northern Ireland) may be possible.

2.3 The case against Game Bags

There are various problems with Game Bag data as they have been (and currently are) collected, some of which could be remedied easily and some which are more fundamental. The biggest general problem with the current bag record database is that the effort expended in shooting or control has not been recorded. 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. This is conceded tacitly by Tapper (1992), who wrote of the inclusion of predator records in the Game Bag Census that they "*...could provide a useful guide not only as to the efficiency of predator control, but also an insight into the changing abundance of predators*". In fact, these two types of information are confounded in the data. The significant changes in the methods used by gamekeepers to kill foxes documented by Reynolds & Tapper (1994) would be expected *a priori* to produce an upward trend and are probably sufficiently important to explain the increase in bag returns that these authors nevertheless ascribe to an increase in abundance. In addition, McDonald & Harris (in press) have recently shown that long-term declines in the bag returns for Stoat and Weasel can be explained as well by changes in trapping effort as by changes in abundance. It is highly likely that some changes in the sampling effort on which the Game Bag time-series are based have occurred for most species since 1961.

It is important to realise that "sampling effort" as we are referring to it here includes qualitative as well as quantitative factors. Thus, while a critical influence on shooting bags will be the number of man-days spent shooting, the efficiency of the guns and other equipment used (e.g. spotlights) and the skill levels of the shooters will also have important influences. Likewise, gamekeeper effort includes the nature of the method used for control (traps versus poison versus shooting), the efficiency of the method (which could vary with, say, type of poison) *and* the time expended by the keeper(s). The seasonal timing of control measures (which is likely to vary more than that of shooting harvests) is another feature of sampling effort which can affect kill rates via changes in the population sampled. For example, shifts in the timing of the peak effort in fox control have led to more dispersing animals being killed in addition to residents (Reynolds & Tapper 1994), a factor which has probably contributed to the rise in bag returns. Although it is unimportant for monitoring purposes whether sampling effort differs between estates (or gamekeepers, or whatever the sampling unit is chosen to be), it is critical either that it is constant over time or that the variation is quantified (see Part II). We would also note that amount or efficiency of effort may not be closely related to the numbers of individuals killed, because of interactions with features of species' social organisation. Once territory holders are removed, a virtually limitless pool of immigrant individuals may be tapped, such that the local abundance of breeding animals is not reflected well by the numbers of individuals killed. The extent to which this occurs is likely to depend on the time of year at which trapping is conducted, but it can certainly be a problem for foxes (Reynolds & Tapper 1994) and mustelids (McDonald & Harris, in press).

Even if attempts are made to standardise recording effort, a key problem is likely to stem from changing fashions in gamekeeping and game rearing. As the aims of game management change, for example from providing a shootable stock of wild Grey Partridges *Perdix perdix* to producing hand-reared Pheasants *Phasianus colchicus*, the gamekeeping measures thought to be necessary will change. As a result, the sampling effort underpinning the Game Bag record is likely to change qualitatively or quantitatively in response to strong commercial influences. It is difficult to see how gamekeeper effort could remain constant under such circumstances.

Another serious problem with Game Bag data is that they are, inevitably, drawn from shooting estates which are unlikely to be representative of the wider countryside. Management for game species will produce artificially high densities, especially where predator control is practised, and animals such as Brown Hares are sometimes moved to restock areas which have been hunted out or to create densities which are sufficiently high for driven shooting (Hutchings & Harris 1996). Within estates, shoot locations can also vary from year to year, if abundance is patchy, such that only areas with high densities are sampled to produce Game Bags, thus masking temporal variations in overall abundance (S. Harris, pers. comm.). These factors will limit the relevance of trends in Game Bag data to those of populations in the wider countryside. In addition, the removal of a large percentage of a species' population in the course of "sampling" means that the sampling method has a major impact on the population which, therefore, it cannot measure. For Brown Hares, as much as 69% of a local population can be removed in a year (Stoate & Tapper 1993): such a rate of killing probably represents the major mortality factor in the population, so an independent (preferably non-destructive) sampling regime would be required to measure its effects.

Changes in the sample of estates included in the Game Bag data set may also produce biases leading to the overestimation of abundance. Estates where shooting becomes poor over time are more likely to drop out of the survey altogether (if shooting ceases) than estates with better shooting, while new estates joining the scheme will probably be relatively good. In addition, zero records are not returned in the current National Game Bag Census, i.e. no data are obtained when no shooting has occurred in a given year because game numbers were too low to permit it. Each of these effects will bias abundance indices upwards. There is also a potential problem in that shooting and gamekeeping interests would have close control over the data collection for any Game Bag-based survey. Such individuals and organisations will have a commercial interest in having particular population levels of prey and predators perceived by the public at large, by conservation bodies and by government, especially if the results of the survey were to feed into policy. It would be preferable if monitoring could be conducted by an independent organisation.

A final problem with Game Bag data is specific to the information collected about predators. Such data are critically sensitive to changes in legislation: persecution of predators may continue after protection has been put in place, but gamekeepers and landowners are highly unlikely to report animals killed in such circumstances, even if anonymity is assured. This problem affects the data collected for Wildcat, for example: the Game Bag time-series ceases when full protection under the Wildlife and Countryside Act was granted to the species in 1988 (Tapper 1992). Partial protection for Hedgehog under the same Act has also reduced the number of estates reporting data on this species (Tapper 1992).

In summary, the following factors argue against the use of Game Bag data for mammal monitoring:

- probable historical changes in sampling effort and methods, both qualitative and quantitative, tend to devalue historical Game Bag data;
- some recording of effort or some commitment to standardisation is needed to make monitoring data from Game Bags reliable, but the extent to which the data collection can be extended is limited;
- the main influences on gamekeeping and shooting, such as commercial factors and game rearing fashions, will always be external to a monitoring scheme, so the need for standardisation under the latter is unlikely to be a guiding influence;
- shooting estates are unlikely to be representative of the wider countryside;
- shooting pressure will respond geographically to local variations in density, masking population variation;
- shooting can remove large proportions of local populations, thus becoming a major mortality factor, the effects of which shooting bags cannot measure;
- zero bag returns are not reported and estates entering into the Game Bag Census will tend to report larger bags than those leaving the scheme, creating bias;
- political and commercial considerations may influence the accuracy of the reporting of Game Bag statistics;
- legislation can change over time, leading to unavoidable changes in effort or to a species dropping out of the scheme altogether.

2.4 Conclusions and future priorities

It is important to remember that, whatever the problems with the historical Game Bag data set are, many of the historical weaknesses are irrelevant in terms of the potential utility of such data for future monitoring. Thus, in principle, efforts could be made to include the recording of shooting and gamekeeping effort and to improve the recording of zero returns in the National Game Bag Census. However, the principal asset of the Game Bag data is the historical context it provides, and any large scale changes would remove continuity from the data: the Game Conservancy would probably be unwilling to do more than include some ancillary recording forms in the census as an add-on, preserving the structure of the data (S.C. Tapper, pers. comm.). It is also the case that, once it is acknowledged that changes need to be made to the collection of bag data to make them more useful, the historical data have been devalued implicitly. In turn, this process weakens the case in favour of using bag returns in the first place.

Given the addition of sufficient recording of sampling effort (which will depend, ultimately, on the amount of information that contributors of Game Bag data are prepared to record) to the national census, Game Bag trend data could be modelled in the same way as CBC and other bird abundance data are modelled. This form of analysis would consist of models of bag sizes which incorporate year effects (estimates of the temporal variation) and categorical estate effects which change when sampling effort changes, i.e. treating estates where methods (say) change as new estates (see ter Braak *et al.* 1994; Thomas 1996). This would not work if changes in effort are too frequent, so standardisation over time would need to be encouraged: effort for a given species would have to remain stable over runs of several years (say, 5-10) at each site (estate) and then not change across all or most estates at the same time. Such developments of the Game Bag data would need to be funded as specific, contracted research projects.

Even without any improved recording of effort, the utility of bag data will vary between species because it will be much more reliable for some species than others. The data might be better for quarry species which are managed sustainably, representing a constant proportion of the population, and they might be particularly poor for generalist predators with only a peripheral impact on game, such as Brown Rats, Hedgehogs and Grey Squirrels, for which control measures might vary considerably in space and time. We have discussed the utility of bag data for each species group in the individual species accounts, but another general consideration is that sampling effort will be more variable for species for which it is easier for gamekeepers to assess abundance (S.C. Tapper, pers. comm.). For example, years with higher frequencies of Red Fox sightings are likely to lead to higher intensities of control, but such clues are not available as to variations in the abundance of small mustelids, so gamekeepers are more likely simply to expend the same effort (i.e. to set the same number of traps) every year. The diversity of possible methods for the control or hunting of a given species will also influence the variation in effective effort that occurs both spatially and temporally: more choice is likely to lead to more experimentation. Nevertheless, it is at least possible for trends in Game Bags apparently to reflect real changes in abundance: the trends revealed by the data for Grey Partridge match those shown by the CBC (Tapper 1992; Siriwardena *et al.* 1998), lending support to both surveys. (Unfortunately, no such national, potentially corroborative data yet exist for British mammals.)

We conclude that Game Bag data can provide a useful contribution to the monitoring of mammals in the UK, but that they should not be considered as a central part of future monitoring policy. The value of the data will depend, to some extent, on the other monitoring options available for a given species. Where other survey schemes can be established which will provide alternative sources of population trend information, similarities after 5-10 years between the changes over time shown by these schemes and the changes shown by the National Game Bag Census will give a clearer idea of the conclusions that we can draw from historical bag data. Meanwhile, we recommend that efforts should be made to encourage contributors of bag data to record as much ancillary information on sampling effort as is feasible. Such efforts would best be focused on the species for which Game Bag data are most likely to be valuable. We suggest that these species should be Stoat and Weasel: other schemes are unlikely to monitor them well and there is reason to believe that the bag data for these species are relatively reliable (see above).

The potential value of the long-term historical data sets that Game Bags provide also means that it would be worthwhile to investigate further how they might be interpreted. A key issue is the calibration of changes in sampling effort: monitoring the effects of future changes in effort via comparisons with independent measurements of abundance would calibrate not only that particular change but also shed light on the likely implications of past changes. Experiments mimicking past changes in effort which are regarded as being particularly important would also be valuable for calibration purposes. A further topic for investigation is the relationship between bag sizes and true abundance for many species: this is known to be non-linear for some gamebirds, but has not been investigated for mammals (S.C. Tapper, pers. comm.). Last, some data which are supplementary to the central Game Bag records are also currently collected and computerised, and explorations of these data could also clarify some of the biases which are believed to exist in the data (S.C. Tapper, pers. comm.). We recommend that the funding of studies such as these should be considered: the potential of Game Bag data to supply a valid historical context for mammal monitoring should be explored in detail.

B. FURTHER POTENTIAL MULTI-SPECIES SCHEMES

3. WINTER TRANSECT SURVEY

This scheme would use volunteers to carry out visual transects during the late winter period. The exact protocol to be adopted will require further discussion, although two approaches (the two extremes) are outlined here. In the more formal approach (based on the BBS protocol) volunteers would walk two predetermined 1km transects within a 1km square and record the number of individuals encountered of each selected species. The less-formal approach (used specifically in upland areas and primarily for Mountain Hares) would ask volunteers to walk a transect (with location and length selected by the volunteer) once each winter or early spring with the aim of counting the number of individuals of the selected species seen.

3.1 Advantages and disadvantages of this approach

Advantages

- It would cover several species for which other monitoring schemes were deemed less suitable and/or for which the species' is more visible in winter thereby maximising detection rates (e.g. Rabbit, Brown Hare, Mountain Hare, Red Fox and to a lesser extent Grey and Red Squirrels and deer).
- It would provide information on some species for which monitoring at other times of year would contain a significant productivity component masking long-term trends (e.g. Rabbit, Brown Hare, Mountain Hare). In this way it could act as a control for the BBS data.
- Volunteers might have fewer conflicting demands on their time during late winter.
- Visual transects would be better for volunteer involvement than more demanding techniques requiring greater identification skills.

Disadvantages

- The possibility of poor weather conditions.
- Difficulties in getting volunteers to participate in some areas (especially the more remote areas)
- Shorter day length, meaning that most fieldwork would have to be carried out at weekends, rather than early mornings on weekdays.

This scheme should be used in conjunction with other schemes (notably Sign Transect Survey and BBS) to determine its effectiveness and to help reveal patterns in long-term trends. Following examination of the data from concurrent schemes it may be possible to reduce the number of schemes being operated.

3.2 Which species could be monitored under this scheme?

All non-marine, non-Chiropteran species could potentially be recorded through this scheme where visual identification can be determined. This would allow volunteers to record everything they could see and identify, thus increasing their morale and providing some useful data on distribution. However, the target species for this scheme should be:

- for monitoring population trends (main spp): Rabbit, Brown Hare, Mountain Hare.

- for monitoring population trends (other spp.): Grey Squirrel, Red Squirrel, Stoat, Weasel, all deer species.
- for monitoring distribution: Pine Marten, Polecat, Feral Ferret, American Mink, Otter, Wildcat, Wild Swine, Chinese Water Deer, Reeves' Muntjac.

All the included species should be printed on recording forms. Observers should be asked to write in any other mammals that they come across, both to allow them the pleasure of doing so and to provide records of escaped aliens.

3.3 Examples of previous surveys using winter visual transects

National Brown Hare Survey - (see Part IV.5). The national population estimate for this species of 817,520±137,521 individuals was based on a winter transect survey of 738 1km squares over three winters. Counts of Brown Hares are best made between October and March, when detection rates are high and sward height is low. However, heavy persecution through driven shoots during February and March suggests that survey work should be completed by the end of January, thereby avoiding the influence of this important annual mortality factor Stephen Harris (pers. comm.).

Various Rabbit surveys - The five Rabbit surveys documented by Trout *et al. (*1986) were all carried out during the first third of the year (typically January to March). This is a time of year when the population is made up of overwinter survivors, thus reducing the influence of annual productivity on the monitoring estimate. It should be noted that visual transects for Rabbits carried out during the late winter may not provide as much information of value for monitoring (e.g. index of abundance) as transects involving the winter recording of field signs (see Part III.B.4).

Mountain Hare Walks - Several local natural history societies carry out less formalised walks along standard routes in upland areas, along which the numbers of Mountain Hares are recorded. (See Part IV.5).

3.4 What a Winter Transect Survey could offer

The use of volunteer recorders would allow the accurate visual recording of a small number of mammal species, although recorders should be allowed to gather information on all mammal species seen, even if this were not used in the same way for monitoring (e.g. records of Pine Martens could provide useful distributional data but would not contribute to any attempt to monitor population trends for this species). Allowing volunteers to record all species encountered should also improve participation rates and maintain motivation. An essential part of the management of the volunteer effort would be the provision of an annual newsletter, sent out prior to the fieldwork season and/or the inclusion of other taxa.

3.5 Alternative approaches

There is a continuum of possible methods with: (a) formalised transects based on the BBS protocol of observers following as closely as possible 'ideal' transect lines at one extreme and (b) less-formal 'walks' by volunteers along standard routes chosen by themselves at the other. The former adopts a more rigorous, statistically acceptable protocol while the latter is more likely to find favour with volunteers. The lack of a rigid protocol need not necessarily reduce the monitoring potential of winter 'walks'. It is felt that the winter 'walks' would be particularly suited to the monitoring of population trends in Mountain Hares. Such walks have been undertaken by local natural history societies in northern England specifically for this purpose, albeit at the local level. The choice of approach to adopt will depend on the required rigour and potential volunteer involvement. Consideration should also be given as to whether a particular approach should be adopted nationally. There could be good reasons for using different approaches in different regions (related to volunteer availability) or for different species/habitats.

3.6 The data recording forms

Form complexity will depend on the approach adopted and the type of data required.

Formal approach - We suggest that the forms can be read using an optical-mark-reader, allowing the rapid input of voluminous but fairly simple data. Recorders would simply count the number of individuals seen for each species, with no attempt to estimate density through distance sampling, and score through the appropriate box on the recording form. All the species likely to be encountered would appear on the form, with space for others to be entered by hand. Details on other species would have to be input by hand if these data are to be used, although they need not be used. Each transect should have a numeric code, as should each fieldworker. Fieldwork effort, as denoted by the amount of time spent in the field, should also be recorded. One form should be used for each of the two visits. A further form could be used to allow the recording of habitat details using a simple hierarchical approach.

Less formal approach - A single form would be required for this approach, with a reduced set of species and effort targeted towards recording Mountain Hares. Again the aim would be to use a form that can be optically-mark-read, containing information on the site, observer and habitat in addition to the species count data.

3.7 Organisation of the Winter Transect Survey

A single individual should be able to co-ordinate the scheme, producing fieldwork material, helping with the analysis of data and preparing regular newsletters. Additional inputs would be required at the graduate scientist, or more likely post-doctoral level, when data analysis takes place. It is suggested that this individual would also co-ordinate the Sign Transect Survey, working for the umbrella mammal monitoring body which would also run the other multi-species schemes and collate data from them and the mammal data generated by the BBS.

3.8 Other considerations

A discussion on the potential for the various forms of winter transect should cover the following points:

- Integration of components from sign transects may be appropriate for some species covered by the winter visual transects (e.g. Rabbit and Mole).

- Late winter may be a difficult time of year in which to recruit volunteers and carry out fieldwork, although fieldworkers may have fewer alternative commitments. The effects of short daylength and poor weather need to be considered.

- Feedback to volunteers would need to be an important component of the scheme, ensuring that continuation rates are maintained, with an annual newsletter.

- Consideration should be given to the points raised in Part III.A.2 on the power of BBS data to detect long-term trends in mammal species.

- Determination of appropriate transect lengths and survey design need further evaluation, using a modelling approach and existing datasets on mammal density and distribution within a range of habitats. 'Winter walks' may be best in upland areas (e.g. for Mountain Hare with the more rigid protocol used in other habitats.

3.9 Resource Requirements

Set-up costs - see Part VII.2.3

Ongoing costs - Were the scheme to be run centrally (apart from the initial recruitment of participants) and assuming co-ordination of this scheme with the Sign-Transect Survey and the Mammals on Roads Scheme, we estimate the annual requirements to be as outlined in Part VII.2.3.

We anticipate that the establishment and operation of a single scheme (in isolation) would require approximately half of the sums shown in Part VII.2.3: running all three schemes together would allow certain economies of scale

4. SIGN TRANSECT SURVEY

This scheme would use volunteers to carry out a search for easily recognisable mammal field signs along transects of standardised length and location. A protocol for the number of visits and their timing requires further consideration.

4.1 Advantages and disadvantages of this approach

Advantages

- It would permit the monitoring of a number of species that are difficult to observe directly in the field, but which have easily identifiable field signs (e.g. Hedgehog, Mole, Red Fox and Badger). Several of these species may already be covered by species-specific schemes or other multi-species schemes and data from the proposed scheme could provide an ancillary role.
- Where there is a known relationship between field sign density and population density it would be possible to monitor changes in abundance directly. Where the relationship is not clear, then it would still be possible to monitor population change through changes in the field sign index. Complications caused by any differences between habitats (see Rabbit species account) could be overcome by maintaining the same field sign transects over time, coupled with basic habitat recording.

Disadvantages

- Fieldworkers need basic skills in identifying particular field signs (but note the success of Look Out For Mammals: Part V.6).
- The field signs of some species cannot be readily separated in the field except by very experienced fieldworkers.
- Different searching may be required for different species' field signs, resulting especially from where they are located (e.g. squirrel dreys in trees, Mole hills on the ground).

This scheme could be used in conjunction with other schemes (notably Winter Transect Surveys and BBS) or in parallel to them (and to other single-species schemes) as a means of providing ancillary data to support those gathered by the different approaches. If several schemes are operated in parallel it should be possible to determine which one(s) offer the best power while being the most cost-effective and to make judgements about future funding and continuation of individual schemes.

4.2 What species could be monitored under this scheme?

Only a limited number of species should be covered by this scheme, so professional fieldworkers with experience of the more difficult species and their field signs would not be needed. The target species for this scheme and their acceptable field signs should be:

Hedgehog (droppings), **Mole** (mole-hills), **Rabbit** (warrens, droppings), **Red Squirrel** (dreys and cones in areas where the two species do not overlap), **Grey Squirrel** (dreys and cones in areas where the two species do not overlap), **Red Fox** (droppings), **Badger** (active setts, latrines, droppings)

4.3 Previous surveys using field sign transects

Field signs have been widely used in a number of monitoring and census schemes. These have been dealt with under the individual species accounts and so will not be further reported here. (See Rabbit, Water Vole, Harvest Mouse, Common Dormouse, Otter, Mink, Pine Marten, Badger, and various deer species.)

The National Fox Survey which is currently in progress (see Part IV.17) is a volunteer-based scheme involving searches for fieldsigns, so will potentially provide both a useful baseline for the monitoring of one key Sign Transect Survey species and useful clues as to effective methodologies. The latter will include successful field methods (transect routes and lengths, etc.), ideas as to volunteer skill and motivation and data on likely fieldsign encounter rates in different areas. Preliminary analyses following the Fox Survey's pilot year show encouraging levels of participation and of fox scat detection.

4.4 What a field sign transect scheme could offer

The use of volunteer recorders in this manner is likely to offer the best opportunity for monitoring Mole, Rabbit and Red Fox, as well as providing important ancillary data for a number of other species. Some additional information would also be generated for other species where the monitoring of distribution is considered important, although such records would typically be of individuals seen during fieldwork, rather than their signs. However, volunteers should be allowed to record field signs of all mammal species where they felt confident enough to do so. Allowing fieldworkers to record all mammal signs encountered should also improve participation and retention rates and allow them to 'opt out' of some species. An annual or seasonal newsletter should also be produced to further increase motivation.

4.5 The data recording forms

We suggest that the forms can be read using an optical-mark-reader (OMR) to allow the rapid input of voluminous but fairly simple data. Recorders should simply count the number of field signs encountered for a given species along the transect route, with no attempt to estimate field sign density from established methods - i.e. no consideration need be given to the accumulation rates for droppings and the effects of weathering, particularly if the same sites are used in future years. All the species likely to be encountered should appear on the form with counts scored through within boxes. Information on additional species would be added by hand within a section for other records, mainly to prevent observers being frustrated by being unable to submit records that they found to be interesting. These data would require computerisation by hand rather than by OMR if it was felt advantageous to analyse them. Habitat information should be entered on the same form along with details about transect location and observer code.

4.6 Organisation of Sign Transects

A single individual should be able to co-ordinate the scheme, producing fieldwork material, helping with the analysis of data and preparing regular newsletters. Additional inputs would be required at the graduate scientist, or more likely post-doctoral level, when data analysis takes place. It is likely that this individual would also co-ordinate the Winter Transect Survey, working for the umbrella mammal monitoring body which would also run the other multi-species schemes and collate data from them and the mammal data generated by the BBS. Transects should follow linear features (including hedges, roads, rivers, paths, etc) on the basis that higher encounter rates are likely to be encountered along linear features than random lines (see National Fox Survey account and Pine Marten account).

4.7 Other considerations

A discussion on the potential for this method should cover the following points:

- Volunteers will vary in their ability to identify different mammal field signs, with the level of ability likely to improve through training workshops.
- Support to volunteers should allow the provision of training workshops, guidance notes and verification of any field signs sent in.
- Volunteers should be allowed to record all mammal field signs encountered so long as they are confident of a correct identification.
- Sightings-based data should also be allowed to be submitted, providing useful distributional data and some supporting evidence for those field sign records submitted.
- Integration of Sign Transects with other transect schemes (notably Winter Transect Survey) may be appropriate (e.g. fieldworkers could walk a transect in one direction recording animals seen and on the return record field signs). This would provide the fieldworker with something to do on the return route and enable two sets of data to be gathered at the same time.
- Feedback to volunteers should take the form of a regular newsletter sent out prior to the commencement of fieldwork.
- Detailed consideration should be given to the points raised in Part III.A.1 on the power of the BBS transect approach to detect long-term trends in mammal populations.
- Determination of appropriate sampling protocols should be undertaken alongside those for other potential transect methods, thereby maintaining consistency in methods and allowing overlap in participation.
- Timing of transects is important and needs to be considered on the basis of temporal patterns in field sign occurrence for individual species and the timing of other studies. The number of visits each year, together with transect length and degree of stratification should be determined through pilot fieldwork and simulation studies.
- Fieldworkers from the National Fox Survey could be recruited into the scheme.

4.8 Resource Requirements

Set-up costs - see Part VII.2.3.

Ongoing costs - Were the scheme to be run centrally and assuming co-ordination of this scheme with the Winter Transect Survey scheme and the Mammals on Roads Scheme, we estimate the annual requirements to be as outline in Part VII.2.3.

We anticipate that the establishment and operation of a single scheme (in isolation) would require approximately half of the sums shown in Part VII.2.3: running all three schemes together would allow certain economies of scale.

5. MAMMALS ON ROADS

This scheme would monitor a small number of mammal species recorded (alive or dead) along routes regularly travelled by volunteers. The recording of wildlife road casualties in particular has been used to calculate national estimates of annual mortality (attributable to road traffic) for a range of bird and mammal species (Finnis 1960; Hodson 1965; Bourquin 1983; Morris (pers. comm.); Shawyer 1999). This approach can be modified to produce indices of occurrence based on the presence of individual species along standard routes. Alive or dead individuals would be recorded separately, to allow the production of either separate or combined indices.

5.1 Advantages and disadvantages of this approach

Advantages

- The scheme would permit the monitoring of a number of species that are difficult to observe directly in the field, but which are often encountered crossing roads or found dead on them.
- A similar method has been successfully applied to a range of bird and mammal species for monitoring purposes.

Disadvantages

- The species covered would be limited by those that are readily identifiable as they cross the road or when they have been hit by cars.
- The survey will need to be simple to carry out safely, thus reducing the range of species that can be covered and the amount of data that can be collected.
- Fieldwork would need to be restricted to the summer months when days are sufficiently long to allow successful recording.
- The frequency of carcass removal (particularly of large carcasses) along some roads may cause a problem, since this is likely to vary on an annual basis depending on local government spending priorities. Some roads are cleared every two days (Colin Shawyer, pers. comm.).
- Live sightings may not provide many data.

5.2 What species could be monitored under this scheme?

Only a limited number of species could be monitored under this scheme, given the practicalities of recording mammals while driving a predetermined route. The target species for monitoring population trends through this scheme should be: Hedgehog, Grey Squirrel, Red Fox, Stoat, Badger, Fallow Deer, Roe Deer and Reeves' Muntjac.

Several other species could perhaps also be included, namely: Rabbit, Brown Hare, Weasel and Red Deer.

Those species for which this approach could generate useful data on distribution include Polecat, Pine Marten, Feral Ferret, Otter, Wildcat, Reeves' Muntjac and Chinese Water Deer.

5.3 What a Mammals on Roads Survey could offer

The use of volunteer recorders in this manner is likely to offer a good opportunity for monitoring a number of mammal species, providing ancillary data to that gathered by other multi- and single species schemes. Additional information on distribution is also likely to be generated for several species as a result of this scheme.

5.4 The data recording forms

We suggest that the forms can be read using an optical-mark-reader (OMR) to allow the rapid input of voluminous but fairly simple data. Recorders should simply count the number of individuals encountered for a given species along the transect route (with live and dead recorded separately). All the target species should appear on the form with counts scored through within boxes. Information on additional species would be added by hand within a section for other records, mainly to prevent observers being frustrated by being unable to submit records that they found to be interesting. These data would require computerisation by hand rather than by OMR if it was felt advantageous to analyse them. The fieldwork instructions should also stress the value of collecting corpses of certain species: those for which genetic monitoring work is being undertaken (Polecat, Pine Marten, Feral Ferret), although the safety element should be stressed.

Information on road type, adjacent landscape, route length and traffic density should also be recorded, with the latter being derived through a simple count of oncoming traffic during a journey on which mammals were not being counted. This will allow for changes in traffic density over time to be examined, with the road death data being adjusted accordingly.

5.5 Organisation of Mammals on Roads

A single individual should be able to co-ordinate the scheme, producing fieldwork material, helping with the analysis of data and preparing regular newsletters. Additional inputs would be required at the graduate scientist, or more likely post-doctoral level, when data analysis takes place. This scheme could be co-ordinated alongside the Winter Transect Survey and the Sign Transect Survey, under an umbrella mammal monitoring body which would also run the other multi-species schemes and collate data from them and the mammal data generated by the BBS.

5.6 Other considerations

A detailed analysis of the advantages and disadvantages of this approach (when applied to road deaths) is provided in MMR. There are a number of other considerations:

- The safety of observers is paramount and fieldwork methods, choice of roads, etc. should all be evaluated against safety aspects. Motorways and dual-carriageways should not be used.
- Fieldwork should only take place during late spring through to early autumn.
- Details of traffic density, class of road and main habitats along the route all need to be recorded, though not during the monitoring visits
- Volunteers should be able to select the location and length of their route.

- Timing of recording should be ideally limited to the morning commuter period or some other consistent time of day.

- Feedback to volunteers would need to be an important component of the scheme, ensuring that continuation rates are maintained.

5.7 Resource requirements

Set-up costs - See Part VII.2.3.

Ongoing costs - Were the scheme to be run centrally and assuming co-ordination of this scheme with the Winter Transect Survey scheme and the Sign Transect Survey, we estimate the annual requirements to be as outlined in Part VII.2.3.

We anticipate that the establishment and operation of a single scheme (in isolation) would require approximately half of the same shown in Part VII.2.3: running all three schemes together would allow certain economies of scale.

6. MAMMALS ON NATURE RESERVES

In this scheme the wardens of nature reserves would send in records of having observed (or not) mammals on their reserves.

6.1 Advantages and disadvantages of this approach

Advantages
- There are a large number of reserves.
- They have a good geographical spread, especially being better spread than the human population.
- They cover a good range of semi-natural and natural habitats.
- Those wardening them are committed to nature conservation: some are paid, others have taken on the role in a voluntary capacity but in a sufficiently formal way for the commitment to be explicit.
- Most of those wardening reserves are highly competent field naturalists.
- Since reserves are mostly permanent, they provide better security for long-term continuity of observations.

Disadvantages
- Reserves tend not to include agricultural land (which makes up the majority of the UK) or 'human sites'.
- Reserves are usually places of particular nature conservation interest, making them unrepresentative of the countryside in general.
- Reserves are generally managed to maintain or enhance their nature conservation interest, so trends observed on them may not be representative of those in the wider countryside.

The last point is particularly important but may be less significant than might at first sight appear. Reserves are generally small, so that for medium and large mammals at least their populations are unlikely to be self-sustaining; they are generally so isolated that immigration from other reserves cannot be relied upon to re-establish mammalian populations that have become extinct on one reserve (an important difference between mammals and birds - the latter appear to be able to find almost any patch of suitable habitat). Thus, except for small species, mammal populations on reserves, while they may be denser, are likely to be correlated with those in the surrounding countryside.

We believe that the advantages of this approach outweigh the disadvantages and that such a scheme could make a valuable contribution to the monitoring of a number of species.

6.2 Which species could be monitored under this scheme?

All species of non-marine mammals (excluding bats) could be included. All could be the subject of reports of having been directly observed, alive or dead. These records could be supplemented by separate records of signs: these should only be signs that are fairly reliably identifiable. The records of signs should be separated from those of direct observations, to allow both finer analysis and inspection of the data for evidence that ability to detect signs is changing over time. To reduce the chances of observers being put off by the scheme appearing unduly difficult, the list of signs should be modest. We suggest that the following may be appropriate:

- Droppings and latrines: Hedgehog, Water Vole, Red Fox, American Mink, Badger, Otter.
- Nests: Harvest Mouse.
- Active setts: Badger.
- Feeding signs: Common Dormouse? (Should involve suspect nuts being submitted for confirmation, at least until the observer's reliability has been established).

Note that this scheme could be valuable in documenting the potential spread of several species whose distributions are currently restricted and for which any data at all may be difficult to get - i.e. Wild Swine and Chinese Water Deer. It could also be useful in picking up the spread of Wild Goats into new areas.

We have considered whether certain other species of very restricted distribution should be excluded from this scheme, as the number of sites providing positive records will be too small to be useful. The exclusions might be: Lesser White-toothed Shrew, Orkney Vole, Ship Rat, Fat Dormouse, Feral Sheep, Red-necked Wallaby. We have concluded that excluding them would frustrate those few observers who recorded them, so that it would be better to include them.

Note that we recommend that *Felis catus* be included in this scheme. Although there is no way in which observers can distinguish between Feral Cats and those that are attached to people, recording the occurrence of cats on reserves could be one of the few practicable ways of monitoring the occurrence of this widespread species.

All the included species (and signs) should be printed on the recording forms. Observers should be asked to write in any other mammals that they come across, both to allow them the pleasure of doing so and to provide records of escaped aliens. (Though, for the benefit of pedants, Dog and Man should perhaps be mentioned on the form as species not to be bothered with).

6.3 Participating organisations

It would be sensible to start with organisations that each currently manage large numbers of reserves. This will ease the setting up of the scheme. We suggest that likely candidates are:

- The statutory country conservation agencies
- Plantlife
- RSPB
- The National Trust
- The Wildlife Trusts
- The Woodland Trust

Between them, these bodies own thousands of reserves (and the number continues to increase). Not all would be recruited into the scheme. We suggest that the following criteria be used to prioritise candidate reserves initially:

- Is there a warden who is prepared to participate? Many reserves (perhaps an increasing proportion) have professional wardens (though a single person may have several reserves to care for); it may be that all the participating organisations would be prepared to include any of their reserves that had professional wardens. Other reserves are cared for by volunteer wardens, whose participation in the scheme would be a matter of persuasion, rather than instruction (we assume).

- Is the warden (or a colleague) able to identify a fair proportion of the mammals included in the scheme? So long as the observers were asked to indicate which species (and signs) they did not feel sufficiently reliable with, it would not matter if they could not deal with all the species on the list; but there would be no point in including a reserve if the observers were only capable of identifying a handful of species - unless these were species of especially high priority.
- How often is the reserve visited?
- Is the reserve in a region that is comparatively poorly covered for mammal monitoring?
- Does the reserve contain priority habitats?
- Is the reserve in a part of the country that is of special interest (a Focus Zone) for any species?

Note that the last three criteria do not mean that sites that do not fulfil them should not be accepted (since we need good geographical and habitat coverage) but that sites fulfilling these criteria would be particularly acceptable.

6.4 Setting the scheme up on each reserve

When a reserve is taken into the scheme, it should be formally registered and various information placed in a database. Relevant information would be:
- Name of reserve
- Organisation managing the reserve
- 10km square in which the reserve falls (if in more than one, perhaps the parts in the different squares should be recorded separately?)
- Area
- Habitat composition - some broad habitats should be used, with simple classes of the areas of each - such as 0, <5%, 5-33%, 33-67%, 67-95%, >95%
- Name and address of observer responsible for submitting the records for the reserve

Observers should be supplied with a form on which to record (and submit) any changes in the above details, so that the database can have the new information added to it. Observers should be reminded from time to time of the need to submit such changes.

When a reserve is registered in the scheme, it should be given a unique number, which can be used in future to identify it on recording forms. Since the latter should be optically readable forms (using marked boxes) this number will be the way in which the data on recording forms are identified to locality, so it is important that observers get it right. We suggest that the number should have a part that identifies the organisation managing the reserve and that the whole number should contain enough simple devices that most wrong numbers can be picked up automatically when record sheets are being read.

6.5 The data recording forms

We suggest that the forms are optical-mark-read forms. These have proved very useful in Garden BirdWatch, for the rapid input of voluminous but fairly simple data - though data that are more complex than what is being contemplated here.

We suggest that each form should cover 13 weeks of the year (it is difficult to fit more onto an A4 sheet), with a box for each week against each species (and each species-sign), to be marked if the species (sign) is observed in that week. There should also be a set of boxes to be filled in for each week, recording the amount of time spent on the reserve that week, in broad classes - say, 0, 1-4, 5-16, 17-64, >64 man-hours. This will provide a means for checking long-term drift in recording effort and, indeed, of correcting for it.

There will need to be a careful description of what man-hours to include. It should cover all the hours on the reserve when mammals might be seen, even if the person is doing some task not directly concerned with recording - e.g. making fences. It should cover everyone who has been asked specifically to look out for mammals, not just the warden (or nominated observer). Perhaps there might be value in having a separate row of boxes for each species for casual records - e.g. reports by members of the public; this would broaden the recorder base for the species that are more difficult to observe; on the other hand, it would raise serious problems about the reliability of the data.

It may be useful to ask observers to write in whether there has been any unusual level of effort put into recording any species that week - for example, a periodic survey to estimate deer numbers or an occasional trap-survey of small mammals.

6.6 Organisation of the scheme

Forms should be submitted quarterly. The data should be input quickly, routine analyses run, and a report produced for the observers. The report should be sent out during the subsequent quarter, together with the survey forms for the quarter after that. This will give the observers prompt feedback and remind them to continue with the work. This is a routine that works well with Garden BirdWatch and it fits in with the design of the recording forms (13 weeks).

Routine analyses should cover such things as seasonal patterns, long-term trends, and geographic patterns, as well as unusual records (the latter are useful for lightening the tone of the reports). The report should comprise four A4 sides and contain news about the scheme and announcements, some of the results from the routine analyses, etc.

Most of this work should be undertaken by one organisation, preferably one that can integrate the results with those of other schemes. Contact with the observers might, however, be better undertaken by the organisations managing the reserves, to whom the wardens owe their primary allegiance. So they might be responsible for sending out the newsletters and recording forms to observers and for collecting up all the data emanating from their reserves. There is always a danger of delay in such a two-tiered approach to contact with the observers but if it helps participation rates then that is a risk worth taking.

The participating organisations should perhaps receive summary data for their reserves quarterly and should certainly be provided with an electronic copy of the data from their own reserves annually.

6.7 Data analysis

The accuracy with which changes in populations would be indexed by this scheme could be affected both by differences between reserves (such as size and habitat) and changes in recording effort (number of weeks per year in which the reserve is visited and number of hours per week spent on the reserve). However, if these are recorded as we suggest, statistical analyses based on General Linear Models and General Additive Models can allow for them. Furthermore, the effect of differences between reserves is reduced by retaining the same reserves in the sample indefinitely.

6.8 Resource requirements

Were the scheme to be run centrally (apart from initial recruitment of participants), we estimate the annual requirements to be as outlined in Part VII.2.4.

Were the scheme to be run in a more devolved way, with participating organisations taking on more of the responsibility for communication with the reserve wardens, the central costs would be diminished but the total costs may be rather higher.

Set-up costs would be 1.5 man-months post-doctoral scientist, for design and programming

7. GARDEN MAMMAL WATCH

7.1 Introduction

A scheme in which volunteers recorded the occurrence (or apparent absence not) of mammals in their gardens would have the following benefits:

- It would cover significant parts of the populations of some species that would not be covered so well by other schemes (e.g. Hedgehog, Grey Squirrel, Red Fox - 14% of the British population of the latter live in urban areas (The Populations Review)).

- It would provide information on species that were difficult to cover by more formal surveys.

- Because large numbers of people might participate, it would provide coverage even in areas of sparse human populations, which may not be covered by other schemes.

- It would introduce people to monitoring who might go on to more demanding schemes.

- An associated newsletter would be a good medium for seeking casual records (outside peoples' gardens) of species for which these would be valuable - e.g. Polecat, Pine Marten.

- It would draw more citizens into direct participation in science-based conservation.

The disadvantage of garden-based recording is that gardens are a peculiar habitat, of marginal importance for most mammals. Urban and suburban gardens, in particular, are unlikely to reflect closely the status and trends of mammal populations in the country in general; rural gardens may do so, though some are so large that for smaller mammals they may form almost self-sufficient islands of habitat distinct from their surroundings. Thus, except for species with significant populations in gardens, such a scheme should only be used in conjunction with other schemes, to provide a suite of information on species that may be difficult to monitor otherwise.

7.2 Previous garden mammal surveys

Look what the cat's brought in - a one-off, five-month survey run during spring and summer 1997 by The Mammal Society. This produced capture records for 964 cats, comprising a wide range of mammals (up to the size of Grey Squirrels and Rabbits), birds, amphibians and reptiles.

The Garden Mammal Survey - a one-off survey run during the autumn and winter 1998-99, by The Mammal Society and The People's Trust for Endangered Species. The data have not yet been analysed but we understand that about 2,000 gardens were included and that these provided records of about 40 different species (bats included). A book on garden mammals was made available to participants at a modest price.

7.3 The BTO/CJ Garden BirdWatch (GBW)

The scheme we have in mind would be similar to GBW, instructions and recording forms for which are provided as an Appendix 2. This has run since 1995 and currently has 11,000 registered participants, of whom about half supply data. (The number of participants increased by *c*.40% during 1998). About 75% of the participants remain in the scheme from one year to the next.

Participants supply details of their gardens when they register. Thereafter, they record birds weekly, submitting their records quarterly. These records comprise: semi-quantitative data on 10 species, presence/absence records for 31 species, and records of the provision of water and various foods. Species other than the 41 printed on the main survey form may be recorded on a "Scarcer Species Form". Records are requested for each of the 13 weeks in the quarter; observers indicate in which weeks they were actually recording, so that weeks with active recording but nothing seen (which are theoretically possible) can be distinguished from weeks with no active recording.

The main survey form and the Site Registration Form are optically-read mark-sense cards, so can be processed quickly. (The main value of the Scarce Species Form, from which it is more laborious to extract data, is that is prevents participants being frustrated by being unable to submit records of these species).

Forms are sent out quarterly, in advance, together with an 8-page A4 newsletter.

Participants pay an annual fee which, with sponsorship from CJ Wildbird Foods Ltd., covers the cost of running the scheme - printing, postage, staff time and office overheads.

7.4 What Garden Mammal Watch could cover

Participants in Garden Mammal Watch are likely to vary considerably in their ability to identify mammals: some may be capable of identifying only a few species from direct observations; others may be able to identify almost all mammals they see and a range of signs. To maximize both participation rates and the value of the data, we suggest that a fairly long list of species is listed on the recording form, with each participant asked to indicate those that she or he wishes to cover (thus allowing observers to participate who do not feel able to identify all the species or their signs).

Species to be included should be decided after the Garden Mammal Watch data have been analysed. The following points are relevant to that decision.

- All obvious and easily identifiable species should be included, even if we do not really need the data; otherwise, participants may be frustrated at not being able to record them.

- Animals brought in by cats should be included, even if they may have originated outside the garden; this will not only avoid participants' frustration at not being able to record such animals but will greatly increase sample sizes of some species.

- Observers should be encouraged to record everything - even pests that have been killed or removed.

- More difficult species may be included, provided observers are reassured that they can opt out of recording them and provided that identification guidance is given.

- Species of restricted distribution should be included because all records of them outside their current range will be useful.

- Signs commonly left in gardens and relatively easily identifiable should be included.

- There should be opportunity to submit records of species additional to those listed.

- Observers should record the presence of non-resident cats and whether or not a cat is in residence - since these will influence whether some mammals occur in the garden and whether they are recorded if present. (Perhaps dogs should also be recorded).

- Houses should be included as part of the garden - to encourage records of House Mouse, Fat Dormouse, etc.

We suggest that observers are asked to record the presence of each species in each week, with a 13-week (i.e. quarterly) cycle of form submission, which has proved successful in GBW. A quarterly cycle allows the form to be of manageable size, reminds participants about the survey, and provides important opportunity for feedback through the newsletter.

7.5 Should other taxa be included?

We suggest that, if a Garden Mammal Watch is planned, serious consideration be given to the inclusion of reptiles and amphibians. Particularly in view of the serious declines of amphibians, even data not precisely identified to species (e.g. frog, toad) may be valuable. The add-on costs would be trivial.

7.6 Organisation of Garden Mammal Watch

There are two main options: as a stand-alone project perhaps run by The Mammal Society (or similar voluntary body) or as a bolt-on to Garden BirdWatch (which is run by BTO, but would need input from mammal experts if mammals were to be bolted on to it). We are hesitant to mention the second, as this possibility has not been assessed either by BTO or by the GBW sponsor; furthermore, it may appear to be a bid for BTO to reserve a high-profile element of mammal monitoring itself. However, we estimate it to be a much cheaper option. Our current estimates of approximate costs of (£000s per annum) of a GMW are as follows (they would need to be refined if this scheme was thought worth discussing further).

Number of participants	Stand-alone	Part of GBW
1000	85	20
2000	87	21
5000	90	25
10000	95	30

Note that there would be some additional cost for the second option, in the form of a few weeks per year for the input of time from mammal experts in interpreting the results and helping with feedback; it would probably be inappropriate for BTO to take on those tasks.

In addition, under either option, 2-4 man-weeks should be allowed for setting up the scheme and a further 4 man-weeks p.a. for any but the most routine analysis of the data (e.g. long-term trends, regional analyses, mapping, feeding records through into NBN; routine analysis would be confined to gross reporting rates in each quarter). These additional weeks would be at post-doctoral graduate scientist level.

The circulation of identification materials should be considered; the cost of this is not included above.

We do not believe that such a scheme could be funded through participants' subscriptions, since the mammals have fewer devotees than birds and generally provide less entertainment to garden-watchers than do birds (because there are fewer species - five per garden on average in the Garden Mammal Survey - and they are less commonly seen). We believe that most GBW participants would not pay more if asked to do so simply in order to include mammals on this project.