Management and Control of Populations of Foxes, Deer, Hares, and Mink in England and Wales, and the Impact of Hunting with Dogs

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Executive Summary

1. Why seek to control populations of foxes, deer, hares, and mink in England and Wales?

- A number of interest groups seek to control populations of foxes, deer, hares and mink for various, and often for several, reasons, summarised in Chapter 2. These reasons should be considered in the context of:
  - ♦ An often ambivalent attitude to the species and its control.
  - ♦ The general lack of a simple relationship between damage and abundance.
  - ♦ Differences between perceived and actual damage sustained.
- Foxes are widely controlled because they are perceived to kill livestock (lambs, poultry and piglets), game (including hares) and other ground-nesting birds.
  - ♦ Fox predation on livestock is usually low level, but widespread and sometimes locally significant. Evidence is strong that fox predation has a significant impact on wild game populations, but less so for other ground-nesting birds.
  - ♦ Fox culling by farmers is done, not in reaction to a current problem, but as a preventative measure, and out of fear of what might happen if the population increased.
- Deer are routinely and widely controlled, in part because they cause damage by feeding on crops and young plantations, but also for amenity reasons, such as sport shooting and venison.
  - ♦ Most studies to date of deer damage to agriculture in England and Wales suggest that this is rarely of economic significance at regional or national level. Where significant damage does occur, it tends to be localised and highly variable even between fields within a farm.
  - ♦ Deer cause widespread and significant damage to forestry, but estimating the economic impact of damage is difficult, not least because trees (and cereals) may recover or even benefit from browsing
- Hares are locally controlled because they are perceived to feed on crops and young trees, but there are few data with which to assess the damage caused. Following widespread declines, brown hares are subject to a Species Action Plan to increase their numbers, while mountain hares receive special protection under international law.
- Mink are patchily controlled because of their predation on wildlife (particularly the water vole and nesting sea-birds), livestock (primarily poultry), and game. While there is ample evidence of their impact on wildlife, there are few data on their impacts elsewhere.

2. What methods are available to control populations of foxes, deer, hares, and mink in England and Wales?

- At present, methods to control populations of foxes, deer, hares and mink in the UK all involve culling: shooting, hunting with dogs, trapping, and snaring. Chapter 3 summarises the methods used for each species; only a few available methods are not used to any significant degree (e.g. snares for mink, live-capture traps for foxes, hunting with dogs for fallow and roe deer).

- ♦ For foxes, dogs, snares, rifles, and shotguns are combined in various ways to create a range of culling methods suited to different situations. The prevalence of these different methods varies substantially between regions in response to local conditions, land-use and traditions.
- ♦ For deer, hunting with dogs is confined to parts of Somerset and Devon; otherwise shooting with a rifle is the predominant method of population control.
- ♦ For hares, population control is usually achieved through organised driven shoots; hunting and coursing make no claim to act as control methods, although illegal, unplanned coursing may locally suppress hare numbers.
- ♦ For mink, trapping (with either killing or live-capture traps) is the predominant method; hunting with hounds is also widespread.

- Methods available to control populations of foxes, mink, deer and hares are restricted by a number of statutes designed to satisfy the interests of individuals (game rights, crop, livestock and game protection), while also addressing general issues such as environmental and human safety, humaneness and conservation (Appendix 2).

- Most operators use a combination of lethal techniques to control populations and non-lethal techniques to control damage. Physical exclusion by wire netting is widely used to protect vulnerable crops, livestock or reared game against deer, foxes and mink. More sophisticated approaches have either proved unpromising (conditioned taste aversion) or are still far from realisation (fertility control).

- Assessing the extent to which different methods are used is made difficult by the disparate nature of available data. In particular, it can be difficult to clearly distinguish between control methods, as they might combine the use of several techniques, with the emphasis changing from place to place, day to day and person to person.
  - ♦ For example, a single Welsh hunt might operate as a mounted hunt in lowland areas, a foot pack in open uplands, and as a gun pack in plantations, and in some or all of these situations might also use terriers, rifles or shotguns.
  - ♦ Welsh packs, with their flexible *modus operandi* illustrate the difficulties in distinguishing clearly between methods that use dogs to kill, chase, locate or flush foxes (or indeed, other quarry).

3. What do simulation models suggest about the effectiveness of methods to control populations of foxes, deer, hares, and mink?

- In Chapter 4 we use three modelling approaches to investigate the potential effectiveness of culling populations of fox, red deer, hare and mink. These are: population-level matrix modelling (all four species); individual-based modelling (for foxes and mink only); and Population Viability Analysis (PVA; for foxes only).

- Population level matrix modelling simulated the impact of culling at a national level for all four target species. Individual-based modelling simulated the more detailed impacts of control strategies in simulated regions for which data were available to provide starting parameters. PVA examined long-term trends in regional fox population dynamics under different scenarios.

- In population level models, hares were extremely resilient to culling mortality because of high productivity. For foxes and deer, a simulated cull causing moderate mortality in addition to existing mortality caused a long-term decline in model populations.

- With individual-based modelling, simulated hunting with dogs (at levels of effort typical of the English Midlands for foxes) was ineffective at significantly reducing population sizes of either foxes or mink. For foxes, the simulated ‘culling at the earth’ and ‘shooting’ scenarios had significant impacts; in mink populations, both the ‘trapping’ and ‘trapping and hunting’ had significant effects on population size.

- An important output of sensitivity analysis of the individual-based modelling was that predictions for foxes were very sensitive to input parameters representing dispersal, which is currently poorly understood.

- PVA explorations showed that fox populations were resilient to changes in breeding and mortality rates, especially given adjacent populations with migration. Under the scenarios we explored, population isolation had more effect on population viability than moderate levels of culling (such as would result from foxhunting). The results however, did suggest that higher levels of culling could be used to hold fox populations at lower densities at which they were still viable.

- Much more research is required on the basic mortality and dispersal biology of each of the species before we could use modelling in any tactical management of these species. The results of the modelling are therefore ‘general’.

4. How effective are methods to control populations of foxes, deer, hares and mink in England and Wales?

- In Chapter 5, we distinguish two important aspects to the performance of wildlife management practices: effectiveness and efficiency. ‘*Effectiveness*’ expresses performance in terms of success in achieving the aims of management (e.g. reducing population size, reducing damage). ‘*Efficiency*’ expresses performance as the success achieved for a given cost (e.g. in time, in effort, or financial).
    - ♦ An accurate measure of population size is an essential component of any measure of the effectiveness of population control (i.e. have the measures brought about the desired change in population size), or its efficiency (i.e. the cost of killing an animal known to be present). However, population size is notoriously difficult to estimate.
    - ♦ Landholder’s perceptions of effectiveness and cost-efficiency may not be accurate. For example, farmers overestimate foxhunting bags by as much as 10-fold.
    - ♦ Commonly used measures of effectiveness (e.g. numbers of animals culled) and efficiency (e.g. financial outlay required to kill one animal) can be very misleading because they do not take into account the density of the quarry. Nonetheless, these measures are components of any estimate of effectiveness and efficiency.

- With the possible exception of red deer in the West Country, the data are not sufficient to calculate total numbers culled, or the proportion of the cull taken using each method; however, we do have relatively good data for organised methods of culling involving dogs (i.e. the various hunt Associations, and the National Coursing Club).
    - ♦ Registered packs of foxhounds and upland foot and gun packs probably take a cull in the region of 21,500-25,000.
    - ♦ Over the last five seasons, an average of 144 red deer were culled annually by the three Master of Deerhounds Association-registered staghunts, roughly 11-15% of the total cull required to prevent further population increases within the Staghunting countries.
    - ♦ On average, registered packs and in National Coursing Club competitions together kill less than 2000 hares.
    - ♦ We estimate that 400-1400 mink are killed by registered minkhound packs.

- If reducing numbers with the intention of protecting a game, agricultural or fisheries interest is a prominent aim, all the strands of our data suggest that hunting with dogs is generally less effective than the alternative methods, for all the species considered.
    - ♦ Possible exceptions include the use of terriers in fox culling, and the use of hounds and terriers in combination with methods involving shooting, particularly as practiced in upland Wales.
- Nationally, shooting foxes, deer and hares is the method which probably contributes most to population control, although there is regional variation in this contribution.
    - ♦ Shooting also contributes most to the local red deer cull within the Staghunting countries; the annual Staghunting cull is insufficient to control numbers in the area on its own.
    - ♦ Trapping is potentially the most effective method used to control mink.
- There are discretionary aspects to all control methods (e.g. number of hunt meets, number of stalking days), which have the potential to increase or decrease their effectiveness.
- Cost-efficiency analyses suggested that fox control using dogs was cost-effective for sheep farmers in mid-Wales. In the east Midlands, cost–efficiency of hunting with dogs was a more complex issue, with net costs for arable farmers and net benefits for others.

5. How acceptable are methods to control populations of foxes, deer, hares, and mink?

- The acceptability of a control method will depend on the balance between a number of criteria, only some of which are readily measurable. An important criterion is humaneness, which is the focus of Chapter 6.
    - ♦ Humaneness is a property of actions that do not involve ‘unnecessary suffering’.
    - ♦ Suffering can be assessed only indirectly, by combining objective measures of behaviour and physiology in the animal’s response to stress, with subjective consideration of the suffering associated with these in humans. This important approach is in its infancy, hence interpreting these measures is not straightforward, and the data are in any case sparse.
- Perceptions of the humaneness of different control methods differ between interest groups. Sometimes the basis for these perceptions is not clear and not readily commensurate with what fragmentary evidence exists.
    - ♦ Nonetheless, it is clear that a majority of the British electorate does not consider hunting foxes with hounds humane. Detailed surveys of other forms of hunting with dogs are not available.
- In hunting with dogs, welfare issues are primarily associated with the length of the chase and the mode of death.
    - ♦ Except for deer, which we do not consider as they are the subject of a separate contract, there have been no studies of the physiological effects of hunting foxes, hares, or mink. Studies on deer are not considered here, as they are the subject of a separate contract.
    - ♦ There are data on physiological responses of foxes to various stressors (including being chased with dogs in an enclosure), but their interpretation and relevance to hunting and other forms of control are not obvious.
- Shooting is widely regarded as humane if accurately done, but there are few data on how frequently it is accurately done.

- ♦ A study of deer in south-west England suggests that wounding rates from stalking average about 2%.
- ♦ Shooting adult foxes during the breeding season has particular welfare implications for orphaned cubs which contrast with its relative effectiveness.

- Welfare issues raised by trapping include the period within the trap (e.g. the stress of being restrained, dangers of starvation, dehydration, hypo- or hypothermia) and humane dispatch of captives.
  - ♦ Physiological data suggest that captured foxes exhibit a stress response. We cannot know for sure the extent to which that involves suffering, but the response is greater for leg-hold traps than for box traps, probably due to physical damage to the restrained limb, and to the animal struggling against the trap.
  - ♦ Humane dispatch of live-trapped mink is problematical.
- Welfare issues associated with snaring are similar to those of trapping, with the additional potential for strangulation in unstopped snares. Generally, less than half the captures in snares set for foxes are of non-target species
- In summary, science is not sufficiently advanced to provide simple measures of the humaneness of different control methods; any immediate judgement can be based on only fragmentary (although interesting) evidence and common sense. While science can inform many of the wider criteria in assessing acceptability (such as the effectiveness of the method in achieving its aims), ultimately acceptability is a judgement for society.

6. What would be the impact on populations of foxes, deer, hares, and mink of a ban on hunting with dogs, and how would this affect different interest groups?

- In Chapter 7 we consider, in the context of population control, three quite different questions which arise in view of a possible ban on hunting:
  - ♦ Can the contribution currently made by hunting to the required cull be provided satisfactorily by other means?
  - ♦ Will a ban on hunting make landowners less willing to tolerate the quarry at current density levels, and hence (through increased control by other methods) cause significant declines in those parts of their range where they are currently hunted?
  - ♦ Would a reduction in overall numbers necessarily be detrimental to the conservation and welfare of the quarry?
- The impact of a ban on hunting foxes with hounds will be highly regionally variable, depending on fox densities and the utility of other methods. In view of this, the impact of a ban on hunting with dogs, in terms of damage caused by foxes, would be regionally complex.
  - ♦ A ban is most likely to have important consequences for game managers and livestock farmers, especially in upland areas with difficult or remote terrain. It is least likely to have important consequences for fox population control in lowland areas.
  - ♦ In mid-Wales, there was strong evidence that hunting with dogs accounted for 73% of a regional cull that effectively suppressed fox numbers. Here, and possibly elsewhere, the result of a ban on hunting with dogs (including both hounds and terriers) would be to allow a rapid increase in fox numbers, unless the same cull could be achieved using other methods.
  - ♦ A ban may have consequences for habitat conservation on farmland; in the 1970s non-hunting farmers removed more hedgerow than hunting farmers, but this was no longer true during the

1980s. However, there was evidence that hunting farmers managed other non-productive habitats with more regard to conservation than non-hunting farmers.

- Staghunting currently contributes only a very small proportion of the red deer cull, even within the Staghunting countries, and this could readily be absorbed by stalking. There may be a decline in the population because of lower tolerance to red deer in the absence of hunting. However, current levels of deer are very high, and are likely to be reduced regardless of whether there is a ban or not.
  - ♦ A possible decline in red deer in the Staghunting countries means that farmers and foresters are unlikely to suffer from increased damage because of a ban. Visibility, which is an important amenity provided by deer, is not simply related to abundance, and will not necessarily suffer because of a decline in numbers.
  - ♦ The redistribution of red deer within their ranges is another potentially important change which may arise from changing culling methods; this will lead to changes in browsing and grazing pressures, and visibility.
- We have relatively few data on hares with which to assess the impact of a ban on hunting. However, we note that they are not regarded as a serious pest, except where locally abundant, and that organised hunting with dogs (with packs of hounds or in coursing competitions) takes only a tiny fraction of the cull.
  - ♦ We surmise, therefore, that there will be little impact on farmers or foresters of a ban on hunting hares with dogs. A ban could have a potentially detrimental impact on hare conservation in some areas, where they are encouraged for hunting with hounds or for coursing.
- We have insufficient data on the numbers of mink killed by different methods, and of the extent of the damage they cause, to assess the impact of a ban on hunting. However, because of the small number of hunts, any impact will be highly localised. Conservationists widely believe that mink hunting has the potential to cause disturbance to wildlife, but again we have no data with which to assess this.

7. In which areas is there agreement or dispute, and what data are lacking?

- There are many areas in which data are lacking and/or subject to widely differing interpretations. These include:
  - ♦ Data on the population biology of all species in the wild is rudimentary. Hence the effect of any culling intensity cannot be predicted with confidence.
  - ♦ The relationships between mortality due to different culling methods, and between these mortalities and other non-culling mortality, are completely unknown. Consequently, it is not possible to predict confidently the consequences for population density of a ban on hunting with dogs.
  - ♦ Estimates of culls by unregulated and/or illegal methods. In particular, data are needed to estimate: numbers of foxes killed by terriers outside the context of the hunt; numbers of hares killed by *ad hoc* coursing; the extent to which illegal methods are used for control.
  - ♦ Reliable field data on damage caused by wild mammals is essential to accurately estimate their true impact on human interests, but is generally lacking. Land users' perceptions of damage are often inaccurate.
  - ♦ Measurements of humaneness, particularly through physiological methods.

Contents

1. Introduction

1.1. The debate

Whether or not hunting some mammals with dogs should continue to be allowed in Britain has been highly controversial, with staunch protagonists on both sides. The twin issues of how some British mammals should be managed, and whether hunting with dogs makes a useful contribution, have formed an important part of the debate.

Of course, hunting in the broadest sense has its roots in prehistory, but hunting with dogs in Britain is also an ancient occupation (Macdonald, 1984; Macdonald & Johnson, 1996; Strutt, 1883). For example, stag hunting was an obsession with many of the Norman Kings. Of William the Conqueror it is said that it was better to have been a stag than his subject, so rigidly did he enforce the harsh Forest laws, under which the penalty for killing a red deer stag was death (Whitehead, 1964). Fox hunting also has a venerable history and was seen as a useful service. Strutt (1883) reproduces an engraving dated early 14th century, of a fox being unearthed by digging for a waiting dog to catch, with a bystander blowing a horn. He also records, "*in the 43rd year of Edward III* [i.e. 1370], *Thomas Engaine held lands in Pitchley in the county of Northampton, by service of finding at his own cost certain dogs for the destruction of wolves, foxes, & c., in the counties of Northampton, Rutland, Oxford, Essex and Buckingham*".

During this long history, the balance of motivations, for example between pest control and recreation, has doubtless varied, as it may at any time from place to place or even occasion to occasion. So too, the effectiveness of this pursuit in achieving different goals may vary. To evaluate effectiveness necessitates clearly identifying the goals, and distinguishing the criteria on which performance can be judged. It may also be necessary to consider whether the goals are well founded. For example, evaluating the contribution of hunting to the goal of pest control is only relevant insofar as it is established that the fox is a pest, and that limiting its numbers reduces the nuisance it causes.

Issues of this sort are not uncommon – worldwide they are the grist of the inter-disciplinary science of wildlife management. Invariably, wildlife management and conservation is made challenging not only because people hold contrasting views on the desirability and legitimacy of different goals, but also because the yardsticks whereby effectiveness may be measured are often incommensurable. That is, even when measures have been accurately made, they are often calibrated in such different units (e.g. population size, money, aesthetics, cruelty) that they cannot systematically be compared to give a single 'right' answer. In short, science can greatly inform debate, but ultimately judgement will decide it. The case of hunting with dogs illustrates vividly these difficulties: it is perceived by different people to have different goals and diverse consequences, some of which are contradictory, many are technically difficult to evaluate, and most are measured in incommensurable units.

In this report, we address the issues of how populations of foxes, deer, hares, and mink are controlled, and the impact on them of hunting with dogs. In this introductory Chapter, we cover some of the important biological background, including the processes underlying changes in mammal populations. In Chapter 2 we ask why these species are controlled, and in Chapter 3 we ask what methods are used. The effectiveness of hunting with dogs in comparison with other methods forms a key part of our report (Chapters 4 and 5). In Chapter 6 we outline the acceptability of hunting with hounds and other methods, with particular emphasis on the associated welfare implications. Finally (Chapter 7), we assess the potential impact of a ban on hunting with dogs on populations of foxes, deer, hares, and mink, and on wider issues related to their management and control. Throughout, we comment on regional differences and the need for further work. Descriptions of each species are given in Appendix 1.

1.2. Background

This report was written under contract to the Committee of Inquiry into Hunting with Dogs. The Committee's terms of reference are:

" To inquire into:

- The practical aspects of different types of hunting with dogs and its impact on the rural economy, agriculture and pest control, the social and cultural life of the countryside, the management and conservation of wildlife, and animal welfare in particular areas of England and Wales
- The consequences for the issues of any ban on hunting with dogs; and
- How any ban might be implemented.

To report the findings to the Secretary of State for the Home Department."

Our report relates to two contracts:

Contract 5: MANAGEMENT OF THE POPULATION OF FOXES, DEER, HARES AND MINK, AND THE IMPACT OF HUNTING WITH DOGS.

Contract 6: METHODS OF CONTROLLING FOXES, DEER, HARES AND MINK.

1.2.1. Aims

The aims of this report are based on the research objectives outlined in Contracts 5 and 6, which we have formulated as a series of questions:

- Why seek to control populations of foxes, deer, hares, and mink in England and Wales? (Chapter 2)
- What methods are available to control populations of foxes, deer, hares, and mink in England and Wales? (Chapter 3)
- What can simulation modelling tell us about effectiveness of methods to control populations of foxes, deer, hares, and mink in England and Wales? (Chapter 4)
- How effective and efficient are methods to control populations of foxes, deer, hares, and mink in England and Wales? (Chapter 5)
- How acceptable are methods to control populations of foxes, deer, hares, and mink in England and Wales? (Chapter 6)
- What is the likely impact of a ban on hunting with dogs on populations of foxes, deer, hares, and mink, and on different interest groups? (Chapter 7)
- Which matters are generally agreed upon, which are disputed, what regional differences are there, and what further research is necessary?

In answering these questions, we have focussed on five interest groups: farmers, game managers, foresters, fisheries managers, and conservationists. Of the two species of hare, we have concentrated on the brown hare, and among the six deer species we have concentrated on red and roe, as only these two continue to be hunted with dogs in England and Wales.

1.3. How does management and control affect populations of wild mammals?

1.3.1. What do we mean by management and control of populations?

Man's treatment of wild mammals has become a controversial subject in Britain. Unfortunately, its many aspects easily become confused by careless use of English. In this report we adopt precise meanings for a few common English words.

The term '*population'* is a convenient shorthand for 'the animals living in a defined area'. A population can only rarely be considered in complete isolation, for example on islands. Usually, immigration and emigration from surrounding areas are important, especially in small areas, which have a high ration between their frontier (for immigration and emigration) and area (for births and deaths).

By '*management'*, we mean any deliberate interventions by man to manipulate the number, structure, distribution, and impact of an animal population. An immense variety of techniques and approaches are used to manage mammal populations (see reviews in Caughley & Sinclair, 1994; Bookhout, 1994). We use the term '*population control'* to mean that numbers are held within desired limits by management. Since animal populations have an intrinsic tendency to increase when resources allow, management for population control usually involves intervention to either increase mortality (e.g. culling) or to reduce productivity (e.g. contraception). Methods specifically aimed at '*damage control'*, such as fencing, chemical repellents or habitat modifications, may sometimes be more appropriate than population control.

An important distinction is between intended and achieved population control. Except in the short-term, killing individuals does not necessarily limit a population (because these individuals may have been part of a seasonal surplus, or because they may rapidly be replaced through births and immigration), or diminish any damage caused. A parallel distinction is important between perceived and proven pest status.

Various people and organisations have an interest in managing mammal populations. These *'interest groups'* often wish to achieve different, sometimes conflicting, aims, including:

- To limit to within acceptable levels damage to human interests, such as livestock or crops, or to a wild species of game or conservation interest.
- To harvest animals from a population, usually for sport or food.
- To increase a species' abundance and range, primarily for conservation, and conversely, to prevent the further spread of non-native species.
- To reduce the spread of diseases.
- Other reasons might include welfare, amenity, or ecotourism.

There is often little contact between interest groups, resulting in many separate management strategies within any region, some of which might be diametrically opposite in aim, method and outcome. For example, the brown hare is simultaneously managed as a pest (for eating young crops), a game animal, and a species of conservation concern.

1.3.2. What happens if populations are not managed?

At any one time the number of animals in a population depends on the history of two opposing processes: animals entering the population through births and immigration, and animals leaving the population through deaths and emigration. A great many factors simultaneously influence these processes, among them resources, competition, behaviour, predation, and weather.

Under favourable conditions, populations of animals have the capacity to grow by producing more young than are lost through mortality. This intrinsic growth rate can sometimes be very fast. However, even unmanaged populations are eventually limited by resources such as food, water, or den sites. A population that has increased to the maximum supportable by a limiting resource is said to be at '*carrying capacity*'. Populations can be held below carrying capacity by mortality unrelated to a limiting resource, such as disease.

Often, as populations grow and approach carrying capacity, their growth does not increase at a constant rate, but starts to slow down; processes not operating at low population density come into play to decrease births or increase losses, or both. This change in growth rate with density is known as 'density-dependence', and its effect is to regulate or stabilise numbers, as a thermostat regulates temperature. Other processes affecting populations are not related to density and have either a constant or a random effect. There is thus a distinction between processes that limit the population and those that regulate it, the latter being the density-dependent subset of the former.

A useful concept is the 'equilibrium level', where the gains and losses of the population are in balance, so that overall the population is neither growing nor declining. The population density at which this balance occurs depends on the prevailing combination of density-dependent and density-independent processes. In modern environments dominated by man, virtually every process acting on even unmanaged wildlife populations will be linked to human activities, and many or most of the possible equilibrium levels will be heavily influenced by man.

Furthermore, populations do not reach an equilibrium level and stick there. In reality, population density will fluctuate around an equilibrium level, as in any system under regulation (e.g. the temperature of a hot-water tank). In addition, if the factors influencing births, deaths, immigration, and emigration are continually changing (an inconstant or continually perturbed environment), the population may never reach an equilibrium, except as a transient state. Thus, stability becomes an important concept in population biology. Again, one can identify processes that tend to increase stability, and others that tend to cause chaotic fluctuations.

1.3.2.a. What is the evidence for density-dependent regulation of fox populations?

There are two main strands of evidence for density-dependence in foxes. First, fox populations that are dense relative to food resources are generally less productive than those that are less dense. Second, there are wide variations in the proportions of vixens that reproduce each year, and in their average litter sizes (Macdonald, 1980; Layne & MacKeon, 1956; Englund, 1970; Lindström, 1982, 1983, 1988; Chirkova, 1955). These two aspects of productivity appear to be related to crowding effects, with lowest productivity tending to occur where fox density is high or food supply poor (Harris & Lloyd, 1991; Lindström, 1992, 1998; Lindström *et al.*, 1989; von Schantz, 1981; Harris, 1977; Pils *et al.*, 1981).

Where food is not limited, crowding itself can suppress reproduction, with only the dominant female breeding. Behavioural mechanisms by which this occurs include harassment of subordinates, infanticide and cannibalism of subordinate vixens' cubs, and possibly the dog fox courting only the dominant females (Macdonald, 1977, 1980). A hormonal mechanism whereby stress leads to lowered productivity through foetal reabsorption has also been identified (Hartley *et al.*, 1994). Consistent with this mechanism, Heydon & Reynolds (2000b) found that in populations where productivity was low, performance was depressed consistently at all stages of pregnancy, from conception to birth.

Similar evidence of reduced productivity in relation to density and resources is found in a wide variety of other mammal species (e.g. arctic fox, Angerbjörn *et al.*, 1991; racoon dog, Helle & Kauhala, 1995; badger, Cresswell *et al.*, 1992; white-tailed deer, Swihart *et al.*, 1998). This is significant because the hormonal processes governing reproduction and reactions to stress are basically the same in all mammals.

1.3.2.b. What is the evidence for density-dependent regulation of deer populations?

For red, roe, and fallow deer, three types of density-dependent changes which influence population dynamics have been reported: an increase or decrease in age at maturity; variation in the proportion of adult females conceiving or calving each year; and variation in first-winter mortality of juveniles. Each of these clearly has the potential to reduce population growth as density rises or resources become more restricted (Albon *et al.*, 1983; Langbein, 1991; Langbein & Putman, 1992a; Gaillard *et al.*, 1992; Hewison, 1993, 1996; Putman *et al.*, 1996). However, none of these authors studied populations actually at equilibrium, and it seems that while density-dependent factors do reduce recruitment rates when conditions are poor, they rarely result in zero-growth or regulation to equilibrium. On the other hand, effects of density-independent factors (e.g. severe winters, drought) are usually regarded as destabilizing rather than regulatory, due to their stochastic (random) impact on mortality or recruitment. In the case of deer, density-dependent changes may generally function to compensate or to dampen wider, erratic fluctuations in population numbers resulting from density-independent variation (Putman *et al.,* 1996).

1.3.3. What can happen when populations are managed?

Equilibrium levels reached by unmanaged populations may not be at a density that is compatible with human interests (e.g. commercial stalking is incompatible with very low densities of deer, and restoration of water vole populations may be incompatible with high densities of American mink). In these cases, human intervention through management may be necessary to increase or decrease densities, or dampen fluctuations.

Management can increase populations in two ways: by enhancing gains (e.g. through increasing limiting resources or increasing immigration via translocation); or by reducing losses (e.g. by removing causes of mortality such as predators, or by vaccination against disease or parasites). These techniques are commonplace in game management and wildlife conservation (Caughley & Sinclair, 1994; Bookhout, 1994).

Where the aim is to reduce the population density of a wild species, the mechanism is to create a density-dependent force that is sufficient - in combination with all the other processes acting on the population - to reduce density to the level required and hold it there. In principle, the options are to increase mortality or decrease productivity, but the latter is not yet practical (sections 3.6.1.a.iv and 3.6.1.b.v).

In the UK context, the aim of population control is usually to limit a population, rather than to exterminate it. However, there are cases where extermination might be a goal, such as American mink from sea-bird islands. Exterminating a pest is generally difficult. First, very intensive culling is usually required to launch an abundant and productive population into decline, because reducing density relieves density-dependent constraints, thereby facilitating increased growth rates. Clearly, to achieve a sustained decline will require a level of culling that exceeds the maximum rate of growth the population can attain. Second, culling is naturally density-dependent, becoming more difficult as population density decreases. Thus, effort must increase as density declines.

Where a wild animal is harvested, for example, for venison, management may involve a combination of positive management and selective culling. Carefully planned harvesting ensures that animals are killed without reducing the numbers produced in future years. Counter-intuitively, the maximum sustainable yield is produced by populations below carrying capacity, due to density-dependent processes.

1.3.3.a. Can control exert a selective pressure on a population?

In principle, any selective mortality has the potential to influence the genetic makeup of a population if it affects a heritable trait. For example, if population control selectively increased the mortality of animals with a particular heritable appearance or behaviour, not only might there be a short-term reduction in the proportion of such animals in the population, but individuals with these traits might leave fewer offspring, thereby eventually changing the genetic profile of the population. If old animals are selectively culled, this will affect the composition of the population, but not its genetic makeup, as these individuals would have already bred and thereby passed on their genes.

In practice, control methods generally do not exert sufficient selective pressure to have an impact on the genetic makeup of quarry species, and the few exceptions require very heavy selection and continuous maintenance. For example, over the last 300 years black herds of fallow in Epping Forest have been maintained by stringently culling any other colour strains, but recently, increased immigration into the herds by animals from expanding populations in Essex is making this difficult (Langbein, 1996).

The most common example of man’s attempts to exert selective pressure through culling is the management of deer to maximise high ‘quality’ trophy heads. Selective culling of males with antlers which show ‘poor’ characteristics in terms of shape and size is used to promote particular characteristics of antler form. Such approaches to culling are so ingrained in tradition that in some European countries the removal of ‘poor’ quality stags or bucks is a requirement laid down in cull plans agreed by hunting authorities (Gill, 1990). The ability to exert some degree of selection for particular characteristics of antler shape is well proven (e.g.

Whitehead, 1993; Ueckermann & Hansen, 1983). However, overall antler size and complexity is probably more influenced by environment than genetics, not least in view of the considerable nutritional demands on male deer in the brief period of antler growth (Fennessay & Suttie, 1985).

Concern is often expressed over loss of genetic variation through inbreeding. For example, deer hunts sometimes choose to kill a stag that has been with a herd of hinds for several years, to prevent inbreeding (MDHA submission to the Inquiry). Suffice it to say here that these concerns are largely erroneous. While problems arising from inbreeding do occur in (already highly inbred) domestic animals, and in very small, enclosed, populations of wild mammals, there is very little evidence for inbreeding in free-living wild mammals, which have sophisticated social systems to avoid it, and even less to suggest that when it does occur it has harmful effects.

1.3.4. What features of the quarry species' population processes are important to their control?

1.3.4.a. Fox population control in terms of fox biology

The fundamental aims of fox management in the UK are to reduce populations or prevent their increase. For this to happen, losses of foxes through deaths and emigration must equal or exceed gains through births and immigration.

In a large geographical region such as a whole county, immigration and emigration will be minor relative to births and deaths. Rural fox populations produce 2-3 cubs per adult annually. Thus if a population of foxes before breeding was 100, it would increase unless 200-300 foxes died before breeding the following year. Culling could account for most or all such deaths, but accidental (e.g. road traffic) or natural causes (e.g. disease, starvation) are also important, and might be sufficient on their own.

At a more local scale (e.g. within the confines of a single farm or shooting estate), two further aspects of fox biology – territoriality and dispersal – become important. To reduce fox density in such a small area, culling must remove not only any resident territory-holding foxes and their offspring, but also any 'replacement' foxes that would normally have been excluded by the territory-holders. These will be either foxes encroaching from neighbouring territories or foxes dispersing from territories farther away. As a result, culls can be locally as high as 25 foxes per square kilometre, even though rural fox densities are typically only 0.5-4.0 per square kilometre in autumn, after cub production.

Because it draws from a pool of potential replacement foxes in the surrounding countryside, intensive local culling does create a 'sink' effect, but it is wrong to imagine that local culling creates a vacuum that sucks foxes in from far away. Foxes on distant territories cannot be aware of the vacant space, so the local culling effort increases their risk of dying only if they are already committed to dispersal behaviour and actually arrive in the culling area. In spring and summer, when no dispersal occurs, the impact of localised culling on fox numbers does not extend more than a few kilometres outside the culling area.

How does local culling fit into the regional context? The countryside can be pictured as a mosaic of 'sinks' and 'sources'. In sink areas culling has ensured that mortality exceeded the local fox productivity, while in source areas culling has been insufficient to prevent an increase in fox numbers. Irrespective of whether a local culling effort meets its local aims (e.g. lower predation on game birds), it is inescapably a component of fox mortality in the region as a whole. Indeed, because dispersal allows high 'bags' to be attained on quite small areas of land, localised fox culling may contribute substantially to the total cull of foxes in a larger region. If many local culling efforts take place within a region, the impact of these alone could amount to regional limitation of fox numbers.

1.3.4.b. Deer population control in terms of deer biology

Although not all the deer species are territorial, many of the above points for foxes also apply here. For roe, which do exhibit territoriality of both males and females, the vacuum left by culling a mature male is often filled quickly by one or two younger males taking over or splitting the vacant territory. Even for non-territorial species, such as red and fallow, source and sink effects may be significant in achieving population control through culling, as their seasonal or daily home ranges will often cover several different landholdings. This is particularly so in southwest England where average estate sizes are comparatively low (*c.*100ha) compared to the individual mean range sizes of red deer hinds (*c*.400ha) and stags (1000ha; Langbein, 1997). Deer culling undertaken only on some small estates but not on surrounding land, may thus often have rather more wide-reaching effects. However, maintenance of close control over the structure of local populations may be difficult within small estates, as for much of the year males of the larger species often live in separate ranges from females.

The potential natural increase of deer populations is somewhat lower than for the other species considered in this report: red and fallow deer rarely produce more than a single young per year, and while roe frequently produce twins, they average nearer 1.5 young per adult female. Most lowland red, fallow and roe populations can sustain an annual cull of 20-25 % of the autumn population (equivalent in number to *c.* 25-33% of the spring population). In the more extreme conditions of the Scottish Highlands, culling levels of only 10-20% may suffice to prevent increases (Ratcliffe, 1987; Ratcliffe & Mayle, 1992). Based on present nationwide estimates for spring deer numbers of >1,000,000 for all six of our deer species combined (Staines *et al.*, 1998; Harris *et al.*, 1995), this nevertheless suggests an annual cull requirement of around 250,00 deer merely to maintain numbers at current levels. This size of cull is not unusual by comparison to other European Countries: in Germany where deer culls are monitored in some detail, the annual cull of roe deer alone regularly exceeds 1 million.

Britain today lacks significant natural predators of deer, and long-term research indicates that without management deer populations approaching an equilibrium level sustain a heavy impact on their habitat and health, and on human interests (see section 1.3.2.b). In order to obtain a sustainable balance among the varied conservation and economic objectives of deer management in the UK, population control is inevitable. The actual numbers of deer that can be sustained without causing unacceptable impact on vegetation can, nevertheless, be manipulated by careful consideration of deer in long-term plans for the design and restructuring of forests.

1.3.5. Regulation of wildlife management and control in Europe

The regulation of wildlife (including game and pest) management varies considerably among European countries (Myrberget, 1990). Although historically England, Wales, and Scotland were very advanced in enacting statutory protection measures, today the organisation of wildlife management in the UK is much less institutionalised than in many other European countries. Current legislative restrictions on control methods are described in Appendix 2. Below we briefly review key components of European regulations of wildlife management:

- **Official authorities:** Responsibility for wildlife management rests variously with agriculture, environment, forestry, environment and education ministries in different countries, or even with local authorities. Firearms licensing and security are usually handled separately by police authorities.

- **Hunting rights:** In the majority of European countries including the UK, game-hunting rights are associated with land-ownership. In some - particularly Mediterranean - countries (Portugal, Italy, Greece, Turkey, and Switzerland), citizens can hunt in all areas (excepting national parks and reserves). In Norway, landowners are obliged to make at least some hunting licences available to members of the public.

- **Hunting fee:** In most countries, but not the UK, individual hunters (including those holding the hunting rights) must pay a fee to be allowed to hunt. This provides a fund of state-held money that covers administrative costs, damage compensation, management, and research. The hunter licence also provides something that can be revoked in the case of individuals who behave irresponsibly. Licences to take individual animals usually require a further fee.

- **Reviers:** Most countries dictate a minimum size of land-unit over which hunting must be co-ordinated. Hunters are obliged to take responsibility for a defined ‘revier’, or to belong to a management group that co-ordinates hunting over the revier. Licences allowing a certain quota of game are allocated per revier. This obligatory community organisation of hunting is conspicuously lacking in the UK, though land-ownership and informal groupings (e.g. deer management groups, fox hunts, fox destruction clubs, shooting syndicates) achieve a similar effect.

- **Game reserves** where hunting is forbidden exist in many countries with the intention of providing source populations of game.

- **Closed seasons:** Under EU law, the basic status of birds and mammals is protection, but a hunting season of defined duration is allowed for game species. Protection of game animals during the breeding season allows them to breed and helps ensure that hunting is sustainable. Year-round hunting/culling is permitted for certain common ‘pest’ species (variable between countries and with time) that are damaging to human interests.

- **Habitat management:** In some countries there is an obligation for hunting groups to carry out management work either to improve habitats for game or to minimise damage. Such work can range from providing duck nest boxes to erecting deer fences to prevent access to crops by game.

- **Conflicts of interest:** Potential conflicts of interests between sport, pest control, and conservation aspects of wildlife management are recognised in certain aspects of its regulation. Thus, compensation claims for damage caused by game species are made to the state, which pays out of the revenue generated by hunter licences. This can lead to shortfalls when the damage caused by a population out of control far exceeds the revenue generated by hunting (e.g. wild boars in Italy and France). Where hunting rights are disengaged from land-ownership as in Mediterranean countries, the motivation to carry out habitat management for wildlife (or predator control) may be wanting, and a conflict can arise between the short-term interests of hunting and longer-term conservation.

- **Training and examination:** Conspicuously, there is no obligatory training and examination procedure in the UK (also the case in Italy). Compulsory hunter training and examination applies in most other European countries, and can be lengthy (e.g. 4 years in Poland), although standards vary greatly between countries. Qualification involves examination of both theoretical and practical aspects. Proficiency with firearms at a firing range is usually also examined. In the UK, voluntary training courses are well established in the UK for most aspects of shooting and hunting, and some of the qualifications are widely respected. The success of at least some aspects of hunter training has been studied in the USA, but we are not aware of similar studies in any European country.

2. Why seek to control populations of foxes, deer, hares, and mink in England and Wales?

2.1. Introduction

In this chapter, we consider the reasons why different interest groups seek to exert control over populations of foxes, deer, hares, and mink in England and Wales. Specifically, we consider the reasons for control by five major interest groups that are potentially affected by one or more of these species: farmers, game managers, foresters, fisheries managers and conservationists. In Britain, the methods that are used to control populations (rather than damage) always involve culling, and are dealt with in Chapter 3.

In general, the major reason why people seek to control populations of foxes, deer, hares and mink in England and Wales is that they believe these animals cause damage, for example, to livestock, nesting birds, or crops. Other reasons include prevention of the spread of disease and maintenance of ‘healthy’ populations. The exploitation of animals for sport or animal products is a separate motivation from the desire to control populations to limit damage. Neither motivation will necessarily result in effective population control (section 5.1), although both have the potential to do so, as both involve culling. In exploitation, the desire may be to provide a high yield of sport or animal products. The population level at which this optimum yield is attained will usually be greater than the population level desirable purely for damage limitation (section 1.3.3). Conflicting management aims therefore arise for species such as some deer and hares, which are simultaneously considered pests, game species, quarry, and are of conservation concern.

2.1.1. How well do the reasons for control relate to population control strategies?

Caughley & Sinclair (1994) point out that the original reason for the existence of a control campaign is frequently forgotten, after which the action itself becomes the objective. They discuss the case history of deer control in New Zealand where the official reason given for control operations has changed several times between 1920 and 1992. None of these changes had any effect on the management action in place. The means became the end.

People who deliberately seek to control animal populations generally reason that they do so because they consider a particular species to be a pest because of the damage it causes. As we demonstrate below, however, population control is not always a simple response to recent damage, nor is it necessarily an effective form of damage control, as there is often no simple link between the abundance of a species and the damage it causes.

2.1.1.a. What is the link between farmers’ experiences of stock losses and their fox control?

Farmers’ responses to damage in terms of culling methods and policy aims are related to both farm size and region. Heydon & Reynolds (2000a; section 2.2.1.a.i) found that only on farms of less than 200ha was recent (last 12 months) experience of livestock losses associated with a higher likelihood of (independent) culling effort; this was not the case for any other farm category. Recent losses did not increase or decrease the involvement of communally organised methods like hunting with hounds.

The aim of eliminating losses of livestock, game, or wildlife prey was also more common on small (<200ha) farms, while on larger estates (>200ha) reducing losses to an acceptable level was more often the aim. Small farmers in the east Midlands were more tolerant of losses due to fox predation than were their counterparts in mid-Wales and west Norfolk. Importantly, only a quarter of farmers had purely local aims: three-quarters cited regional control of fox numbers as an aim of their culling regime.

Data from a 1981 survey (Macdonald & Johnson, 1996; section 2.2.1.a.ii), provide evidence of a link between control and significant damage sustained at some time in the past (according to the subjective assessment of the respondent). Of 183 farmers who stated they had suffered significant damage by foxes, 77% carried out control, while only 13% of 395 who had not suffered damage controlled foxes[1]. For comparison, 45% of farmers reporting deer damage in 1981carried out deer control, but only 3% of those not reporting damage did so (see also Table 2-7 in section 2.3.1.b)

Regional patterns of reported fox damage generally mirrored those of control (Table 2-1), except in Warwickshire, where 41% of farmers carried out control, but only 26% suffered damage. A farmer's tendency to carry out fox control might be linked to the amount of damage he perceives he would incur *in the absence of any control measures*, as well as to the cost of control (see also section 5.3). One measure of this perceived potential for damage might be whether the farmers believes there are 'too many' foxes.

Table 2-1 Regional patterns of reported fox damage and control (1981 survey). % Answering yes to: 'Do you attempt to control foxes?' and 'Do you believe you suffer significant damage from foxes?' WestC. = Devon and Cornwall.

	County:								
	Dorset	Leic.	Oxon.	Salop.	Suffolk	Sussex	Warw.	WestC.	Yorks.
% Control:	38.5	27.1	31.5	22.8	26.1	39.5	40.8	37.8	21.4
% Damage:	31.7	25.0	27.0	29.1	19.7	40.9	26.3	41.2	27.0
N	122	70	127	57	69	86	76	98	89

We investigated the link between farmers' tendencies to report that they controlled foxes, with their tendencies to cite different reasons (Table 2-2), and found evidence that perceived pest status affects a farmer's tendency to control foxes independent of reported damage. In addition to reported damage, the 'too many' reason was a significant predictor of whether a farmer controlled foxes[2]. No other reason had this property. So, regardless of whether a farmer reported damage due to foxes, they were more likely to cull them if they thought there were 'too many'. This potentially explains the regional variation observed in Table 2-1: for example, in Warwickshire, where the highest proportion of farmers said there were 'too many' foxes, almost twice as many farmers controlled foxes as reported suffering damage.

Table 2-2 Regional variation in reported reasons for controlling foxes (1981 questionnaire; WestC. = Devon and Cornwall).

Reason:	County:								
	Dorset	Leic.	Oxon.	Salop.	Suffolk	Sussex	Warw.	WestC	Yorks.
Disease	53.3	49.2	50.4	50.0	53.3	60.2	43.1	29.2	38.8
Stock	68.6	76.9	61.0	72.0	71.7	74.7	66.7	71.9	71.3
Too many	62.9	67.7	71.3	70.0	63.3	74.7	73.6	65.2	54.4
Game	53.3	38.5	41.3	64.0	71.7	36.1	55.6	21.6	47.5
N	119	64	124	55	68	85	75	95	88

In Wiltshire, in 1995, farmers who considered the fox to be a pest on their own farm were more likely to believe that foxes should be controlled everywhere than farmers who did not consider the fox a pest on their farm (Baker & Macdonald, 2000; section 2.2.1.a.ii). Although most farmers thought the fox 'too numerous', farmers who perceived a fox pest problem did not estimate a higher density of foxes on their land, and the belief that there were too many foxes was widespread, even among farmers who were not affected by losses.

It seems, therefore, that farmer's fox control efforts are largely driven by their fears of what would happen if they stopped culling. At present, we have insufficient data to predict whether farmers' losses would in fact

[1] $\div^2_{[1]} = 225.26$, $P < 0.0001$

[2] logistic regression, $\div^2_{[1]} = 13.2$, $P = 0.003$

increase if they stopped culling foxes, and it is likely that only an unfeasibly large and long-term field experiment would be able to provide the answer.

2.1.1.b. Is there a link between a species' abundance and the damage it causes?

Foxes, deer, hares, and mink cause damage primarily through their feeding behaviour. What, and how much an animal eats is subject to many influences including the quality and quantity of alternative foods, competition with other group members, competition with other species, and individual dietary requirements and preferences. These factors mean that change in a species' abundance will not necessarily translate into a *pro rata* change in damage.

In North America, some studies, but not all, have found that sheep predation was a function of coyote density (Knowlton, 1999). There is some evidence that sheep predation by coyotes is restricted to a particular sector of the population. Sacks *et al.* (1999) found that breeding coyotes whose territories contained sheep were the principal predators of sheep, and further that predation was reduced only when territorial breeders known to kill sheep were removed. Comparisons between the North American coyote literature and UK fox literature must be made with caution because coyotes are capable of killing adult sheep, and of hunting these co-operatively.

Perhaps because of the difficulty of estimating fox abundance and of holding all other variables constant, there has been no formal attempt to relate livestock losses to fox abundance. Rowley (1970) presented evidence that lamb killing by foxes may be habitual in certain individuals, giving rise to serious losses locally. There are no grounds for speculating how fox density and sheep density might influence the risk of such behaviour developing, except that reduction of average fox age might lead to reduced occurrence of any acquired behaviour patterns. Knowlton (1999) showed that besides reducing population density, intensive culling substantially reduced the age profile of coyote populations. A similar effect would be expected from fox culling, although Heydon & Reynolds (2000a) did not detect one in heavily culled and moderately culled regional fox populations, possibly because their samples sizes were too small.

Severe damage by deer to woodland and agricultural crops is widely recognized as being associated with high population densities, but recent evidence suggests that there is no simple linear relationship (Gill, 1992; Kay, 1993; Putman, 1994, 1996b; Reimoser & Gossow, 1996; Nahlik, 1995; Tilghman, 1989). Factors such as proximity of the 'target' crop to cover and availability of alternative forage may have a greater influence on damage than density (Putman & Kjellander, in press).

For most deer species, damage levels tend to remain low - and relatively constant - until population density passes a certain threshold (which is itself highly variable between areas). Beyond this threshold, impact suddenly and dramatically increases. The existence of just such a curvilinear relationship between damage and density has been shown for white-tailed deer damage to forest regeneration (Tilghman, 1989), although it is usually difficult to demonstrate clearly in the field because of variation in other site factors. The actual extent of damage sustained would seem to be determined by a complex interplay of density with other factors such as forage diversity and quality, landscape and habitat structure, and climate, as well as the particular type of crop affected, its distance from cover, size of planted area, and distance of the vulnerable crop from alternative preferred forages. In emphasising the significance of those latter factors, Reimoser & Gossow (1996) suggest that levels of deer damage to forestry or agricultural crops relate not simply to deer density *per se* but to the effective balance between food-independent 'attraction factors' for deer (e.g. woodland edge) and the natural food supply. Where habitat structure is very attractive to deer yet the natural food supply is sparse, more damage may be anticipated than where the 'attractiveness' of an area is low in relation to the forage availability.

2.1.1.c. How realistic are estimates of damage?

Whether or not a species really causes the damage it is accused of is central to the validity of any control programme. If the species is not, in fact, the cause of the damage, then a programme to control its population will be misplaced at best, and counterproductive at worst, as resources will be directed away from the real source of the problem. Accurate estimates of damage are vital to a key series of questions: 'What are the relative costs of damage and control?' (section 5.3); 'What is the potential cost of damage if control is discontinued in the long term?'; and crucially, 'What population level is required to maintain effective damage reduction?'. At present data are generally insufficient to answer these questions accurately, but we address some of the issues for foxes in sections 2.2.1.c.i and 2.2.1.f.ii, and for deer in section 2.3.2.b.i.

2.2. *Why control fox populations?*

Fox population control in rural Britain is attempted by a number of disparate interest groups. These include both communally organized groups (fox hunts and fox control societies or clubs), and individuals (professional pest controllers, gamekeepers, wildlife reserve wardens, farmers and landowners). For many of these people, control by culling is perceived as a means of reducing predation by foxes on domestic livestock and poultry, reared and wild game and on other wildlife. However, foxes are also culled for sport, and sometimes to control the spread of disease (Macdonald & Johnson, 1996; Reynolds, 1998). Culling foxes for their pelts, once important (Macdonald & Carr, 1981), effectively ceased during the 1980s (Harris *et al.*, 1995).

In the UK, anyone may kill or capture a fox by a legal method, but they must have the authority to be on the land to do this, otherwise they commit a trespass (an armed trespass if carrying a firearm). Only the landowner or tenant farmer (for sporting rights this depends on the tenancy agreement) is in the position to grant this basic authority. Hence, whatever the personal motivations of those who actually carry out fox control (e.g. gamekeepers, hunts, pest controllers), the motivation of landowners or tenant farmers to cause or allow fox control is paramount.

The conclusions that fox numbers in some regions are suppressed by deliberate culling (Heydon & Reynolds, 2000a), and in most regions have shown strong changes through time (Reynolds *et al.*, unpublished), severely complicate any appraisal of the reasons that motivate fox culling. They mean that any investigation of the impact of foxes on human interests is specific to the fox density prevailing in that region and at that time.

2.2.1. Why do farmers seek to control fox populations?

Here and elsewhere in this report, our use of the term ‘farmer’ encompasses landowners, owner-farmers, and tenant farmers, and implies livestock and arable farming. While we recognise that control for reasons of game conservation might be an important consideration for many among this group, for simplicity we deal with it as game management in a separate section (section 2.2.1.f.ii).

2.2.1.a. Data and approach

Our data in this section come from three main sources:

- The Game Conservancy Trust’s (GCT’s) ‘Three-region study’ (Heydon & Reynolds, 2000a,b; Heydon *et al.*, 2000).
- A series of questionnaire surveys of farmers’ attitudes to wildlife and fox control carried out by the Wildlife Conservation Research Unit (WildCRU) at Oxford University (Macdonald, 1984; Macdonald & Johnson, 1996, 2000; Baker & Macdonald, 2000).
- A Produce Studies Limited (PSL) survey in March 1995 of 831 British farmers by face-to-face interview (PSL submission to the Inquiry). This survey was commissioned by the Campaign for Hunting. Breakdowns by category are provided, but no statistical tests of significance are made, nor do the data presented allow these to be made. There is no record of the questions asked.

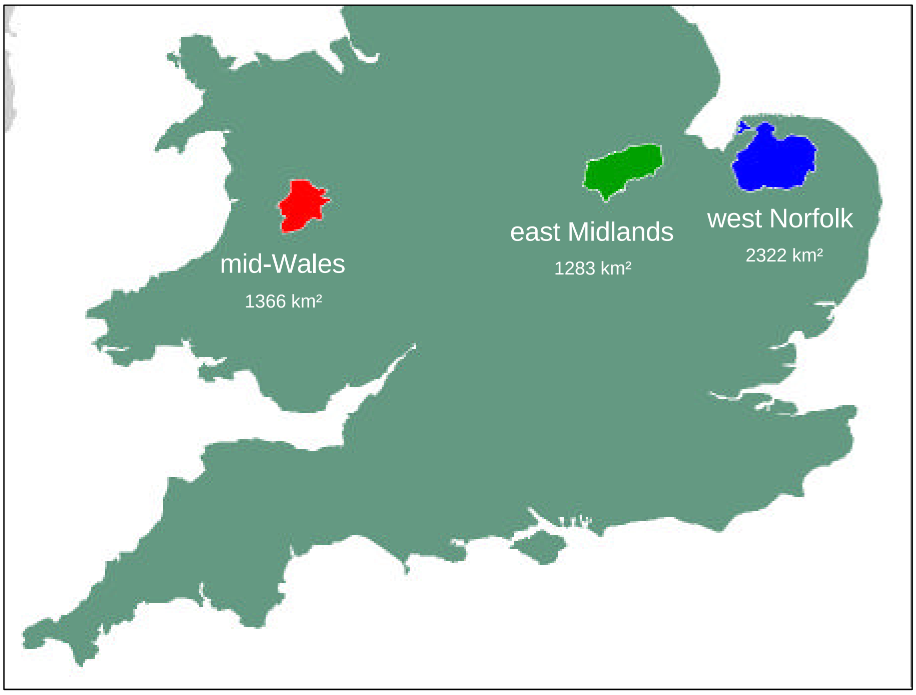
We detail the approach used to collect and analyse the GC and WildCRU data, because they are used widely throughout this report.

2.2.1.a.i. The Game Conservancy Trust’s ‘Three-region study’

During 1995 to 1998, The Game Conservancy Trust undertook their ‘Three-region study’ to determine the impact of culling - by all the interest groups involved - on fox numbers across large regions, the size of a whole county. Previous studies of fox culling had considered individual methods only, and on either a very local scale or on a national scale that ignored regional variations.

The three large regions - mid-Wales, the east Midlands, and west Norfolk (Figure 2-1) - were chosen to illustrate a range of landscapes, land-use and fox culling traditions, rather than to be representative of Britain as a whole. In the hills and valleys of mid-Wales, sheep farming is the primary motivation for fox culling. Fox density is low and most culling involves the use of hounds and terriers. The east Midlands is an area of mixed

Figure 2-1 The three large regions used in the Game Conservancy Trust's 'Three-regions' study.



agriculture and land-use. It has a mixed regime of fox culling, but hunting with hounds and mounted followers holds centre place. In west Norfolk, game conservation is the commonest motivation for fox culling, carried out by professional gamekeepers. The flat landscape and low fox density are well suited to culling with rifle and spotlamp.

The study used three principal sources of data: a questionnaire survey to all farmers, a field survey of fox density, and post mortems of dead foxes. The questionnaire aimed to determine numbers culled, reasons for culling, aims, and methods used. After posting a questionnaire to every farm property, the authors checked for bias by telephoning a random sample of the non-respondents. In all, data were obtained from an unbiased 51% of farm properties, giving excellent representation. (Opinion polls and other studies have typically covered less than 1% of farms in their survey areas.). Cull data were also obtained directly from communally organised culling groups, such as foxhunts, and gun-packs.

2.2.1.a.ii. The Wildlife Conservation Research Unit (WildCRU) surveys of farmers' attitudes to wildlife

The WildCRU holds data from farmer questionnaire surveys, carried out in 1981, 1992, 1995 and 1998. In this chapter, we use data from three of these (1981, 1992 and 1995).

The 1981 questionnaire comprised 130 questions soliciting information concerning diverse aspects of farming practices relevant to wildlife. Among these questions were a series asking about the damage that farmers attributed to foxes, and their attitudes to the humaneness (section 6.1.2.c) and effectiveness (section 5.2.1) of different control methods. These data are reanalysed here to address the specific issues raised in this report.

The 1981 questionnaire was dispatched to 2,288 farmers, and 859 (37.5%) responded in 'hunt Countries' in nine regions of England. The region referred to as 'West Country' comprised Devon and Cornwall. The regions were selected after examination of distribution maps in Coppock's (1976) Agricultural Atlas of England and Wales. Within England, each of the major agrarian regions was represented. The questionnaires were distributed by post, together with a pre-paid reply envelope, and a letter stressing our non-partisan position on countryside controversies. The questionnaire was designed after extensive consultation and pilot studies (along with much discussion) with farmers, and with the assistance of questionnaire experts. The questionnaire provided data for several projects, making it effectively impossible for farmers to anticipate how we would analyse their answers. A full account of these results is given in Macdonald & Johnson (1996) and Macdonald & Johnson (in press).

The second WildCRU survey of farmers used in this chapter was carried out in 1992. This was devised primarily to assess farmers' perceptions of the mole as a pest, and included questions concerning other common pests. With the aid of the National Farmers Union (NFU), 460 questionnaires were distributed to a representative sample of farmers in England, Scotland, and Wales. Of these, 157 (34%) were returned (Atkinson *et al.*, 1994).

The third WildCRU questionnaire used in this chapter (reported in full in Baker & Macdonald, 2000) was carried out in Wiltshire in the summer of 1995. In a postal questionnaire, 220 Wiltshire farmers were asked about foxes and foxhunting on their farms. These included all 120 tenant farmers of Wiltshire County Council (who farmed >5.0 hectares) and a random sample of 100 other farmers. Questions covered topics such as the perceived pest status of foxes on farms, the number of foxes killed on farms, and the methods used; 101 questionnaires (45.9%) were returned.

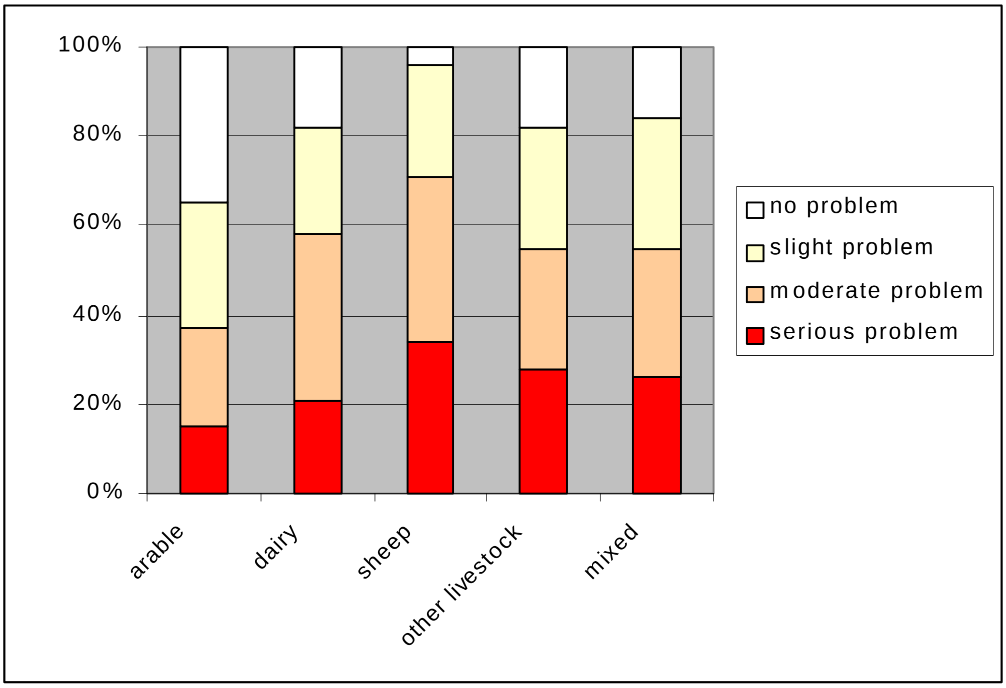
2.2.1.b. Do farmers consider the fox a pest?

Questionnaire surveys indicate that most farmers consider the fox to be a pest in general, although far fewer consider the fox a pest on their own land. In Wiltshire in 1995, the majority of farmers (64%) believed foxes should be controlled 'everywhere' (Baker & Macdonald, 2000), but only around a third (32%) included the fox on a list of animals that they considered a pest on their own farm. In the WildCRU's 1992 survey, 57% of 157 farmers said the fox was a 'pest', while in the PSL survey (PSL, 1995), 23% of farmers perceived the fox to be a serious problem, 56% a moderate or slight problem, and 21% no problem. In the 1981 WildCRU survey, 74% of a national sample of farmers thought foxes should be controlled in the country and 71% in towns.

There are clear differences between types of farming enterprise, leading to regional differences in the perception of the fox as a pest. In the PSL survey only 4% of sheep farmers perceived foxes as no problem, compared with 35% of arable farmers, and 18% of dairy and other livestock farmers (Figure 2-2). Similarly, the regional distribution of responses reflected land-use, with 16% of farmers in sheep-dominated Wales seeing foxes as presenting no problem, compared with 35% in the more arable East Anglia/Midlands region.

How do farmers rank the pest status of foxes compared with other species? Table 2-3 summarises the results of a number of different studies, including Packer & Birks' (1999) survey of gamekeeper's attitudes. In the WildCRU's 1981 survey of farmers, rabbits were most frequently cited as a causing damage, and foxes were

Figure 2-2 Perceptions of foxes as a problem by different categories of sheep farmers. Data from Produce Studies Limited (1995). *n* = 831.



listed fourth most frequently by 33% (Macdonald, 1984; Atkinson *et al.*, 1994). In the WildCRU's 1992 survey, fewer farmers considered the fox to be a pest than the rat, magpie, or mole (Atkinson *et al.*, 1994). In this survey 11% of farmers rated the fox their worst pest, 9% the second worst, and 10% third worst. In Wiltshire in 1995, farmers were asked to produce a list of the animals that they considered pests on their farm, in descending order of estimated financial damage caused. Around a third of farmers included the fox on their list, fewer than included rabbits (58%), or badgers (42%). Each species listed was assigned a score, according to the position it occupied in the farmer's list. Scores were averaged for each animal, and the pests then ranked according to the resultant scores attached to each. Overall, farmers ranked the fox as their third worst pest by this measure, after rabbits and badgers respectively, as did farmers of non-dairy stock. The 1998 Quantocks Deer Management and Conservation Group questionnaire (Langbein, 1998; section 2.3.1.a) also asked those farmers which considered deer to be a pest to rank them alongside other species, including foxes, from 1 (most damaging) to 6 (least damaging).

Table 2-3 Percentage of farmers considering various species a pest, and species rank as a pest. Data taken from: Macdonald, 1984 (1981 data); Atkinson *et al.*, 1994 (1992 data); Baker & Macdonald, 2000 (1995 data); Packer & Birks, 1999 (1996 data from gamekeepers); Langbein, 1998 (1998 data). A blank cell indicates that the species was not on the prompt list or was not mentioned by the farmers. Additional species not covered by two or more questionnaires were excluded.

	Survey:				
	1981, England: % reporting damage (prompted)	1992, Britain: % considering species a pest (prompted)	1995, Wiltshire: % considering species a pest (unprompted)	1996, Wales & Midlands: mean rank (gamekeepers)	1998, Quantocks: mean rank
Fox	30	57	27	1	4
Deer	13		11		1
Hare		7	2		
Mink			1	3	
Hedgehog	3			7	
Mole	41	64	11		
Rabbit	58		47		2
Rat	56	90	21	5	5
Squirrel			2		4
Badger		19	34	3	
Weasel/stoat	5			4	
Corvids	39	64	17	4	
N	795	157	101	66	60

Bearing in mind that farmers' perceptions may not accurately reflect reality (section 2.2.1.c.i), what damage do farmers ascribe to foxes? Lamb, poultry and piglet predation are the most frequently cited types of damage caused by foxes.

2.2.1.c. How much loss of lambs is attributed to foxes?

Wherever livestock farming takes place around the world, predation by canids (wolves, coyotes, foxes) is a perennial and controversial complaint, and in the case of foxes, one that has generally defied quantification (Lloyd, 1980; Macdonald, 1987; McDonald *et al.*, 1997; Saunders *et al.*, 1997). Predation on livestock is an extremely complex issue in which breeds, stock management, predator density and individual predator behaviour all appear to influence the outcome. There seems to be no doubt that foxes are capable of taking dead, moribund and healthy lambs, but how important this is to sheep farming enterprises remains unclear.

Studies around the world have estimated the proportion of viable lambs killed by foxes at 1-30% in different circumstances (Saunders *et al.* 1995). In the UK, Hewson (1984, 1990) and White *et al.* (in press) estimated that <2% of otherwise viable lambs in various Scottish sites were killed by foxes. MAFF (1996) attributed 5% of lamb losses to 'predator/misadventure', which includes predation by dogs, as well as various other causes. However, the logistical and scientific problems associated with studies of lamb losses (section 2.2.1.c.i) mean that for the most part we must rely on the farmer's judgement as to the extent of damage caused by foxes, even though this is likely to be an overestimate.

A total of 649 respondents to the WildCRU's 1981 survey kept sheep, in flocks varying in size from only two to several thousand. Of these, 54% said they had lost lambs to foxes at some time. There was regional variation[3]: the proportion of sheep farmers claiming a loss in the previous year was highest in the North and lowest in the Midlands & East (Table 2-4). When only flocks larger than 100 were considered, the proportion of farmers suffering losses in the past year was highest in the West Country (73%), and remained lowest in the Midlands & East (49%).

Sheep farmers in the 1981 survey were asked about the absolute numbers of lambs they thought they had lost to foxes in the three years before the study. On average, a farmer estimated that he had lost about two lambs annually (mean=1.74). This varied regionally: the mean in the Midlands & East was 0.75, compared with 4.45 in the West Country, where flocks were largest. As a proportion of the flock, reported losses were also highest in the West Country, with considerable regional differences (Table 2-4).

Table 2-4 Reported lamb losses due to foxes in England and Wales

	The GCT 3-Region Study			WildCRU 1981 Survey			
	Wales	Midlands	W. Norfolk	West Country	Midlands & East	North	South
% Flocks suffering fox predation	61	49	24	73.2	47.1	49.3	54.7
% Lambs born indoors	41	77	57	-	-	-	-
% Lambs killed by foxes:							
All lambs	0.6	0.4	0.0	1.9	0.4	0.8	0.5
Flocks where losses occur	1.0	1.3	1.1	4.7	2.8	1.8	1.6
Maximum losses	14.5	5.2	8.3	100[4]	13.3	4.9	10.0

In their 'Three-region study' Heydon & Reynolds (2000a) reported that 24-61% of sheep farmers, depending on region, had experienced lamb losses during the preceding 12 months that they attributed to foxes (Table 2-4). However, the losses reported amounted to only a small percentage of all lambs, in line with an earlier study in an upland area of western Scotland (Hewson, 1984). Maximum values for any single farmer were 5-15%, depending on region. The pattern of lamb losses among regions did not mirror fox abundance, but more likely reflected the vulnerability of lambs under the regionally diverse lambing practices (see also section 2.2.1.c.i).

In their review of available evidence, McDonald *et al.* (1997) concluded that losses of lambs to foxes were insignificant compared with losses due to other mortality. Improved husbandry would therefore give rise to greater productivity increases than would fox control. While this is probably true, it overlooks three points that are important for the sheep farmer. First, irrespective of other losses, one can nevertheless ask whether prevention of fox predation on lambs is in itself cost-effective (see section 5.3). Second, measured or perceived levels of loss may already be reduced because of fox culling. Thus MAFF "*does not consider foxes to be a significant factor in lamb mortality nationally, but it should be stressed that this is against a background of widespread fox control by farmers*" (MAFF, 1993). The only study that has aimed to measure lamb losses in the absence of fox control (Hewson, 1990; section 2.2.1.c.i), provided weak evidence that culling foxes did not influence lamb losses. Third, for many upland and marginal upland areas improved husbandry may not be an option, or may not be a cost-effective option.

2.2.1.c.i. How realistic are estimates of fox predation on lambs?

The WildCRU's 1981 survey, in which just over half of 649 sheep farmers claimed to have lost lambs, demonstrates the problems of relying on farmers' perceptions of losses. When asked what kind of evidence had implicated the fox in their reported lamb losses, a high proportion of farmers said that foxes had been seen in the area (83%). Just under a half (46%) said they had seen dead lambs at fox earths, and 39% said that at some stage in their lives they had seen a fox attacking a lamb. This figure also varied between the regions[5], and

[3] $\div^2_{[8]} = 37.8$, P <0.001

[4] A single farmers reported losing 8/8 lambs

[5] $\div^2_{[8]} = 18.5$, $P = 0.02$

tended to be highest in the southern counties: for example, 53% of Sussex farmers said they had seen such an attack while only 18% of Yorkshire sheep farmers said they had done so.

For predation on lambs, a key investigatory problem is the difficulty of monitoring events in a widely dispersed flock under low husbandry supervision. Lamb remains following predation or scavenging are usually obvious, but identification of the predator is not always easy or reliable. In assessing fox predation, it is crucial to distinguish lambs that were alive when taken, from those that were dead or likely to die, but the diagnostic field signs are difficult to interpret and easily missed. Indeed, the whole lamb may be missing, its removal by predators or scavengers passing unsuspected by shepherds, a fact not appreciated until recently when ultrasound scanning made it possible to forecast the number of lambs that will be born (Reeves, cited in Saunders *et al.,* 1995). These diagnostic difficulties affect the observations of both scientist and farmer.

The difficulty of correct diagnosis suggests that manipulative experiments, as have been carried out for deer (see below) would be a better approach, allowing comparison of flocks exposed to predation with others where predators are removed or excluded. In such an experiment, comparison of the number of lambs weaned between the two types of flock treatment would give a direct measure of the impact of predation on genuinely viable lambs. Unfortunately, foxes are rarely the only predator, and whether predators are removed by culling or exclusion, it is rarely possible to contrive an experimental treatment that does not affect more than one predator species. Distinguishing the role of individual predators then relies on field signs, just as in non-experimental studies. If, alternatively, only one predator species is removed or excluded, compensatory predation by the remaining species will cause the impact of the missing one to be underestimated. The problems with interpreting the effects of compensation are dealt with further in section 7.1.1.

Hewson (1990) attempted to address the influence of fox culling by estimating lamb losses in a 70km² area where foxes were not culled during the three-year study period. Although this study has never been submitted to a refereed scientific journal, it has been quoted as key evidence that lamb predation is not reduced by fox culling (e.g. McDonald *et al.*, 1997; LACS submission to the Inquiry, p. 26), hence it is important to consider its value as evidence.

There are three major flaws with the study. First, Hewson’s report refers to two estates, only one of which controlled foxes, but quantitative data on lamb losses and fox density were not presented from this site; the work was not, therefore, a controlled experiment in any accepted sense. Second, there was no measure of the fox population before, during, or after the study, on either estate. Four individuals were radio-tracked and two earths located, suggesting that the no-culling estate was big enough to hold 2-3 fox territories. Hence, at best, the study considered lamb predation by only nine foxes each year. Third, lambing on both estates was carried out on enclosed ground close to the farm, where supervision was intensive, and predation would be expected to be lower than on the open hill. Ewes were returned to the hill with their lambs already 3-5 days old, by which time twinned or orphaned lambs would have been fostered onto ewes that had lost their own lambs, avoiding much of the previously identified risk of predation (Saunders *et al.* 1995). Quantification of lamb losses after their return to the hill depended on the researcher regularly searching 70km² of difficult terrain for evidence, which meant there was no real possibility that losses could be accurately quantified or changes between years measured.

Other problems include doubts over the actual lack of culling on the intended ‘no culling’ estate; the method used to indicate fox numbers were limited by food supply (just one female was judged barren from external examination); and conflicting statements about the extent to which carrion was available and used. Overall, we consider the study to be scientifically weak, and not to allow the strong conclusions drawn by Hewson and by LACS.

Among three large regions of England and Wales, Heydon & Reynolds (2000a) reported a pattern of reported lamb losses that appeared to reflect the vulnerability of lambs under the regionally diverse lambing practices, rather than fox density. Thus, losses were most commonly reported in mid-Wales, where much of the lambing happens on unenclosed hill ground with minimal shepherding. In west Norfolk and the west Midlands, most lambing takes place either indoors, or out of doors under intensive supervision. Nevertheless, for all three regions the effect of having a gamekeeper was to halve reported lamb losses, suggesting that intensive local culling of foxes had a marked impact on perceived or actual lamb predation.

2.2.1.d. How much loss of poultry is attributed to foxes?

Among farmers with free-range poultry (excluding large commercial flocks) surveyed by Heydon & Reynolds (2000a) 49-78% reported losses in the preceding 12 months (depending on region – Table 2-5). For poultry, the regional incidence of losses (% of flocks affected) mirrored fox abundance, so that west Norfolk had the fewest occurrences, the east Midlands had the most, and mid-Wales was intermediate. It is noteworthy that, among the three regions, large-scale commercial free-range poultry units occurred only in Norfolk. Such operations may be feasible there only because of the low regional density of foxes, but each operator also put considerable independent effort into fox control. Again, presence of a gamekeeper significantly reduced reported losses.

Table 2-5 Reported poultry losses due to foxes (flocks <200 birds only)

	Wales	Midlands	W. Norfolk
% Flocks suffering fox predation	54	78	49
% Birds killed by foxes:			
All birds	18	25	0
Flocks where losses occur	50	50	15
Maximum losses	100	100	100

In 1995, more Wiltshire farmers reported that they had lost chickens (42%), than lambs (16%), game birds (11%), or other livestock, (16%; Baker & Macdonald, 2000). Chickens were generally kept on a non-commercial scale.

2.2.1.e. How much loss of piglets is attributed to foxes?

In recent years there has been a rise in the number of pigs raised on outdoor units (from 5% of the national breeding sow herd in 1987 to a predicted 40% in 2000), and there is a widespread belief among outdoor pig farmers that foxes take piglets, agitate sows and thus increase mortality from overlays, and transmit disease. There have been no scientific studies of the extent of these problems, and we must therefore rely on farmers' perceptions and individual experiences.

In April 2000 the National Pig Association requested information on this issue from 51 outdoor pig farmers. All of the six farmers who had replied by mid-May 2000 believed that foxes posed a potential problem, and reported losses at some sites but not others. Losses were variously described as 'persistent', 'occasionally severe', and 'small'. One farmer estimated losses exceeding 25% on occasions. A survey of outdoor sows by Cambac JMA Research in 1993/94 (privately contracted 'by various bodies') found that 20% of units reported a fox problem (summary from Dr H.J.Guise, pers.comm).

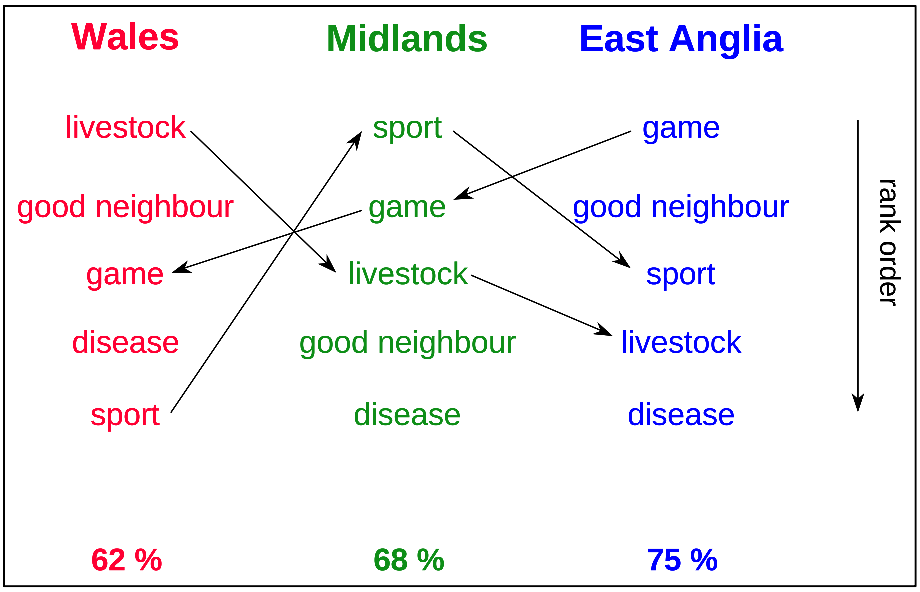
Fox-proof electric fencing can substantially reduce fox predation, but is a big financial investment, and occasional fox culling is still required as a back-up. Local Wiltshire farmers (R. and K. Shepherd, pers.comm.) handling 1700 litters of piglets annually in 1994, recorded an improvement of *c.*1 piglet per litter after erecting an electric netting fence (both live and dead piglets were counted). There was no such improvement in a nearby unfenced area. The difference of one piglet per litter amounted to 30% of the profit margin. Another farmer with 830 sows in 1995 reported piglet losses attributed to foxes of <1% of all piglets in late winter, rising to >5% (100 piglets/month) in late summer (Reynolds, unpublished data). Annual loss attributed to foxes totalled 390 piglets out of production of 21,000, amounting to £10,315-29,815 (depending on whether piglets were fattened on or off the farm).

2.2.1.f. What reasons do farmers give for controlling fox populations?

Heydon & Reynolds (2000a), asked farmers to indicate their reasons for fox culling on their land. Not surprisingly, these reflected variation in land-use between regions. Thus, 94% of farmers in mid-Wales cited protection of livestock, but only 28% did so in predominantly arable west Norfolk. Conversely, while only 29% in mid-Wales cited protection of game, game interests motivated 75% of west Norfolk farmers (a more detailed exploration of why foxes are controlled for game management is presented below). Most farmers (62-75%, depending on region) gave two or more reasons for culling foxes (Figure 2-3).

Local fox culling for the benefit of neighbours was widely cited by farmers in all regions (35-54%), but only 6.5% of this group gave 'good neighbour policy' as their sole reason for culling. Sport, too, was usually cited in

Figure 2-3 Reasons cited by farmers for killing foxes in Wales, Midlands, and East Anglia. The rank order of reasons like game, livestock, and sport reflected land-use in the three regions. Importantly, most culling was done for two or more reasons. The figures on the bottom line refer to the proportion of farmers citing more than one reason for fox culling.



combination with other reasons. In mid-Wales, not a single farmer cited sport alone. In the east Midlands 57% cited sport, but only 14% cited sport alone. No farmer claimed sale of pelts as a reason for culling.

In the 1981 WildCRU questionnaire, farmers who believed that foxes should be controlled were asked which of four options they thought justified the control. These were '*may spread disease*', '*kill domestic stock*', '*kill game birds*', and '*are too numerous*'. Overall, the domestic stock reason was most commonly opted for (70%), with 'too numerous' next most frequent with 67%. Disease was opted for by 47%, and 'game' by 46%. Only protection of game and disease showed significant regional variation[6] (Table 2-2). The variation in protection of game followed regional game-shooting patterns, while disease was cited markedly less frequently in the West Country compared with other regions.

When asked to select from the same five options, almost three-quarters (73%) of Wiltshire farmers said they were too numerous, and over half said because they kill domestic stock. Killing game birds was cited by 34% of farmers, and the spread of disease by 30%; 7% cited 'other' reasons. In comparison with the 1981 survey, relatively more Wiltshire farmers who believed foxes needed to be controlled, selected each of 'kill domestic stock', 'kill game birds', and 'spread disease', while relatively fewer believed foxes 'too numerous' (Macdonald, 1984).

2.2.1.f.i. Why do some farmers not cull?

In 1981, two-thirds (67%) of farmers surveyed reported carrying out no fox control (they were not asked about culling carried out by others on their land), but among those who reported fox damage, only 24% carried out no control. There was considerable regional variation in proportions of non-culling farmers[7], as would be expected from patterns of land-use. The lowest proportions were recorded in Yorkshire and Shropshire, and the highest in Sussex, Oxfordshire, and Dorset (Table 2-1). There was no evidence that a farmer's participation in hunting or approval of the active conservation of foxes had any effect on their tendency to control foxes[8].

In the Midlands and Norfolk in 1996, 12% of farmers did not cull foxes or allow fox culling (Heydon & Reynolds, 2000a). Among these, the commonest reason cited was lack of necessity, followed by a perceived

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[6] $\div^2_{[8]} = 23.7$, $P = 0.003$ and $\div^2_{[8]} = 58.1$, $P < 0.001$ respectively

[7] $\div^2_{[8]} = 15.7$, $P = 0.04$

[8] logistic regressions, all $P > 0.05$

benefit from the presence of foxes. Half of the non-culling farmers stated that they would consider culling in the future if the fox population increased. Only one fifth of non-culling farmers (i.e. 2.5% of all farmers) stated that they did not approve of fox culling.

2.2.1.f.ii. How realistic are farmer's worries about foxes as carriers of disease?

Of 92 farmers who responded to a questionnaire in Wiltshire in 1995, almost a third (30%) believed that foxes should be controlled because they spread disease (Baker & Macdonald, 2000; section 2.2.1.a.ii). These 19 farmers were asked to list the diseases about which they were concerned. Although they identified fourteen diseases, the majority of those suggested present no risk whatsoever with modern veterinary science (Table 2-6). Despite listing as potential fox-borne hazards various inappropriate pathogens, farmers did not cite *Bordetella bronchisceptica* (Kennel cough) or *Toxocara canis* as causes for concern, although foxes might transmit these. Other pathogens known to be carried by foxes include *Neospora caninum*, para-TB (responsible for Johne's disease) and parvo-virus. The risks of transmission to livestock and domestic animals are unknown. Although the sample was very small, it nonetheless illustrates that if poorly informed, farmers might overestimate some risks associated with foxes, while other possible threats go unrecognised.

Table 2-6 Farmers citing certain diseases and causes of infection, or death, as potentially spread by foxes (% of farmers), together with an indication of the likelihood of this happening.

Disease:	% Farmers worried (n=19)	Likely associated risk
Mange/mites	36.9	Unlikely - species specific
Tuberculosis	15.8	No evidence for this
Movement of carcases/afterbirth	15.8	Small
Abortion	10.5	Possible
Rabies, botulism	each 10.5	None, no rabies in Britain
Brucellosis	5.3	Small
Leptospirosis, tapeworms, Salmonella	each 5.3	Possible
Anthrax	5.3	Anthrax only possible on feet
Distemper, summer mastitis, foot and mouth	each 5.3	None

2.2.2. Why do game managers seek to control fox populations?

Predator control has been a feature of game management in both upland and lowland Britain since shooting estates first arose in the early 18th century (Tapper, 1992). Predator control to protect nesting game birds is a skilled and labour-intensive job reliant on the employment of a gamekeeper, and is regarded as essential to ensure the farmer has a viable shoot.(Tapper, 2000).

An important distinction that must be drawn is between the management of a wild game bird population to allow a harvest, and reliance - wholly or in part - on hand-reared game birds (see section 3.6.1.a.ii). Arguably, the former most closely conforms to the sustainable 'wise use' of a natural resource, and for many game managers is preferable if it can be attained (Tapper, 2000). However, the trend in recent decades has been towards the latter, partly because wild game birds on farmland have fared badly under modern intensive agricultural practices (e.g. Potts, 1980). This has implications for the timing of predator control (section 3.2.6).

The commercial aspects of shooting also need some clarification. Shooting is a saleable commodity. Although not all farmers sell or lease their shooting, many do. The provision of game shooting is a popular secondary land-use for many land properties; for some upland estates, grouse shooting is actually the primary land-use. This can and does lead to increased demands to intensify the production of game birds, and in recent years bodies like the British Association for Shooting and Conservation (BASC) and GCT have had to address the question of excess in commercially driven shoots. Tapper (2000) and co-authors argue a view on this that places limits based on environmental and ecological impacts of intensive shoots.

2.2.2.a. Do game managers consider the fox a pest?

In game management, man is directly in competition with the fox, since both are game predators. An important difference is that while man withholds predation during the game breeding season in order to benefit from population increase, predation by foxes is particularly intense during this same period because the prey are more vulnerable and because foxes are themselves breeding at this time. Several studies (reviewed by

Lindström, 1994; Reynolds & Tapper, 1995a) have indicated that adult foxes selectively provision their cubs with birds and mammals in the size range 0.3-3.5kg (e.g. rabbits, hares, game birds). Because of this breeding season predation, foxes can have a substantial impact on wild and reared game bird and hare population dynamics.

In a questionnaire survey of 66 Welsh and Midlands gamekeepers, the fox was ranked the most serious predator of game (Packer & Birks, 1999; Table 2-3). Similarly, in a 1994 survey of gamekeepers carried out by BASC, 96 % of 1624 keepers who responded said that foxes were present on their land and needed to be controlled. Control was considered necessary to '*ensure that damage to game, wildlife and livestock was reduced or kept at acceptable levels*' (BASC submission to the Inquiry).

A survey conducted in 1997 by Bristol University for the National Gamekeeper's Organisation (NGO submission to the Inquiry) asked gamekeepers to rate how serious a problem foxes posed to them. Of the 203 who replied, 63% stated that the fox was a major pest, and only 6% considered it a minor pest. None said it posed no problem to them.

2.2.2.b. What impact do foxes have on wild game bird populations?

The fox is a key predator in many ecosystems (Reynolds & Tapper, 1996), particularly the heavily altered man-made ecosystems of Western Europe. Usually, evidence of the importance of any single predator species is circumstantial: a study of a prey species – usually investigating poor productivity or population decline – finds high predation levels. Studies of this kind identifying foxes as a major predator exist for all British game birds. Accumulated evidence of this kind can be very persuasive that high predation is associated with population decline. Unfortunately, it remains ambiguous: the predation could be the cause of decline, or it could be merely symptomatic of some other cause.

Unambiguous evidence about the impact of foxes on wild game bird populations was specifically sought through research by the GCT. This evidence is of two kinds. First, an experimental study of wild grey prtridges on Salisbury Plain (Tapper *et al*., 1996), in which a suite of common predators, including foxes, were intensively culled on a 6km² 'removal' site for three years. A similar site nearby had no predator removal and acted as a 'comparison' area. After three years, 'predator removal' and 'comparison' treatments were switched between the two areas. Throughout the six years of the experiment, and for one year before and after, partridge numbers and productivity were monitored on both areas. The results were conclusive: under the predator removal regime, autumn partridge densities increased by 75% year-on-year, finishing 3.5 times greater at the end of three years, compared with the non-removal comparison regime. These improved autumn numbers also carried over to build up spring breeding stocks, which increased 25% annually, to finish 2.6 times greater after three years.

The Salisbury Plain experiment provided decisive evidence of the importance of predators for game, but did not indicate which predator species contributed most to the effect. After predator removal ceased, it was shown by radio-tagging partridges that foxes were by far the most important of the suite of predators removed, accounting for 81% of all annual losses and 91% of all breeding season predation losses (Reynolds *et al.*, 1992; Reynolds, unpubl.). By radio-tagging foxes to establish territory and group size (Reynolds, unpubl.), it was also shown that their annual food requirements (600-1000kg/km²) far exceeded what the partridge population could supply (spring density 1-3kg/km²). In fact, game birds as a whole formed less than 1% of fox diet in this area, although foxes killed 20% of breeding females (Reynolds, unpubl.), accounting for most of the experimental effect (there was a further 10% non-predation loss of breeding females). Thus, partridges were certainly not an important determinant of fox density, but foxes were very important for partridges. On Salisbury Plain, the food resources that allowed foxes to maintain such high numbers relative to partridges were rabbits and hares, which together made up 85% of fox diet.

The second type of evidence is not experimental, but is equally important. It is very rare for any field study to quantify predators, predator diet and prey numbers simultaneously. However, if foxes really are important to game, the number of game they eat must make a significant dent in the game population. Reynolds & Tapper (1995b) undertook this research in a mixed agriculture area in northeast Dorset with unremarkable populations of both game and foxes, and showed that the proportion of game birds taken by the resident foxes was substantial compared with the number of birds (24-100% of wild-breeding game birds), their productivity, and the shootable surplus.

For wild red grouse, a comparison between good and bad moors in Scotland and northern England (Hudson, 1992) demonstrated that louping ill was overwhelmingly the most important factor influencing grouse density and productivity; game keeping (incorporating both predator control and habitat management) and climate were also significant explanatory variables. Where predator control is sufficient to allow high grouse populations (and only there), threadworm parasites cause cyclical year-to-year fluctuations in grouse numbers. Low-density grouse moors appear to suffer severe predation pressure from raptors and foxes that kept numbers low. Of all these influential factors, fox density, habitat management, and threadworm burdens can currently be influenced by practical management measures.

McDonald *et al.* (1997), state that 'diseases are the main factor controlling grouse numbers', but overlook Hudson's (1992) point that on high density grouse moors the impact of predation has already been minimised by intensive predator culling.

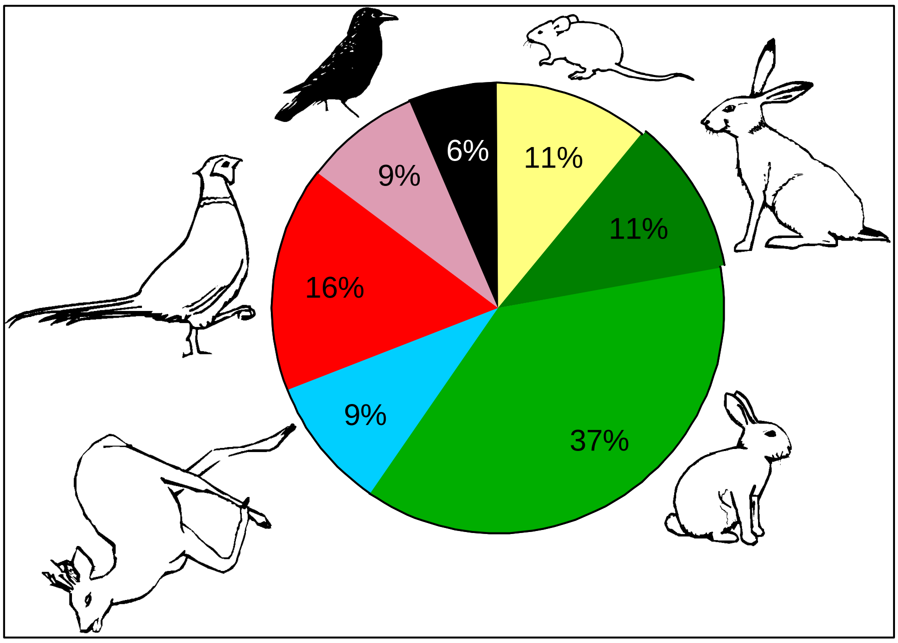
2.2.2.c. What damage to released pheasants is attributed to foxes?

Due to the success of rearing (section 3.6.1.a.ii), the pheasant has become an increasingly important game bird during the 20th century. At the turn of the century, it probably comprised 15% of all game birds shot in the UK, but by the 1980s had increased to >55%, roughly equivalent to 12 million birds per year (Tapper, 1992). It has been estimated that around 20 million birds are released annually. The wild pheasant population has probably declined during the same period, and now comprises only an estimated 10% of all pheasants shot (Tapper, 1999). If these estimates are reliable (they involve extrapolation from small samples), they imply that 40% of released birds (*c*. 8 million) die annually. An unknown proportion of these will be killed by foxes.

2.2.2.d. What impact do foxes have on wild brown hare populations?

In north-east Dorset, Reynolds & Tapper (1995b) found that in a population of hares (which was not subject to culling by man), foxes effectively wiped out the annual reproductive gains of the population. They did this by taking a biomass of 47-87 kg/km² annually from a hare population with a pre-breeding biomass of 42kg. (This is obviously only feasible because the hare population reproduces during the year.) This research supplemented many earlier studies of hares that had provided circumstantial evidence of the importance of foxes as hare predators (Jensen *et al.*, 1970; Nyholm, 1971; Spittler, 1974; Angerbjörn, 1977; Frylestam, 1980; Häkkinen & Jokinen, 1981; Lindlöf & Lemnell, 1981; Pegel, 1986; Hearn *et al.*, 1987; Dannell & Hörnfeldt, 1987; Angerbjörn, 1989; Small & Keith, 1992; Lindström, 1994), as well as manipulative experiments (Marcström *et*

Figure 2-4 Diet of foxes in north-east Dorset.



al., 1989; Tapper *et al*., 1993). The foxes in northeast Dorset had a more diverse diet than those on Salisbury Plain, and hares comprised just 11% by weight of their food intake (Figure 2-4). As with partridges on Salisbury Plain, the fox's influence on the hare population was far greater than the hare's importance to the fox.

In their submission to the Inquiry, IFAW argue that these studies are atypical, and that 'there is no evidence' that foxes are having an impact on prey species in most other situations. In fact, field studies of the type described above have not been conducted elsewhere: there is no evidence because it has not been sought. Furthermore, the grounds for regarding predator and prey densities at the Dorset study site as unremarkable are well supported (Reynolds & Tapper, 1995a,b).

2.2.3. Why do conservationists seek to control fox populations?

Foxes eat species of conservation concern throughout the world, particularly in countries where they are not native (e.g. Kinnear *et al*., 1988). In Britain, they are controlled by conservationists primarily because of their predation on some ground-nesting wild bird populations (Côté & Sutherland, 1995), particularly those that are already fragmented. Well-documented examples include red grouse *Lagopus lagopus* in the Stiperstones Reserve (Macdonald *et al*., 1999), and terns, *Sterna spp*, at various reserves e.g. Sands of Forvie (Patterson, 1977), North Denes reserve, Great Yarmouth (Paul Lewis, RSPB. pers. comm.).

Predation by foxes has become an increasing problem on coastal bird reserves in Norfolk, and many reserve-owning conservation bodies have carried out or commissioned fox culling to safeguard vulnerable bird populations (Reynolds, 1998a). In most cases, this has been a reluctant and controversial policy. Coastal reserves in north Norfolk may previously have been protected against foxes by a cordon of shooting estates and by the intensity of regional fox control. Foxes were apparently absent in west Norfolk earlier this century, probably the result of the very large workforce of gamekeepers (there were 1202 in Norfolk in 1911). Although today there is only one tenth that number of gamekeepers, the proportion of land with professional gamekeepers remains very high compared with the rest of Britain.

In many cases, foxes pose a threat to wild birds only for a short period, usually the nesting season (e.g. Birkhead & Nettleship, 1995). It is uncertain how significant nest losses are to bird species that are typically much longer-lived than game birds. Predator removal often has a large, positive effect on hatching success and post-breeding densities of the target bird species, but no impact on spring breeding numbers (reviewed by Côté & Sutherland, 1997). Since increasing breeding numbers is the usual conservation goal, Côté & Sutherland conclude that predator removal did not generally fulfil the conservationists' aims. They add that this could be attributable either to inherent characteristics of the birds' population dynamics, or to ineffective predator removal. To these comments, we add that reserve-based conservation measures that enhance productivity may be annulled by emigration and events occurring away from the breeding grounds.

The RSPB only consider fox control where it can be achieved legally and humanely, and then only if there is a risk of serious damage to conservation, agriculture or human health. Where fox control is necessary, the policy of the RSPB is to use trained staff to trap or shoot.

2.2.4. Why do foresters seek to control fox populations?

The Forestry Commission has for many years undertaken fox culling through its own wildlife rangers, and financially supported local fox destruction groups. In 1992, this 'good neighbour' policy was revised following a review of existing literature, changing the emphasis from extensive and systematic fox culling to providing a quick and effective response to lamb killing by foxes. The policy shift was applauded by conservationists, but criticised by sheep farmers and game managers (Chadwick *et al*., 1997).

A fox culling policy on the part of forestry bodies has no component of self interest except where shooting is leased out. As in arable farming, foxes may have a benefit to forestry interests as predators of pest mammals (see section 5.3.3). Although the impact of foxes as predators on populations of rabbits and voles is uncertain (Trout & Tittensor, 1989; Dyczkowski & Yalden, 1998), both are significant pests of young forest plantations. Foxes are also predators of roe deer fawns (Lindström, 1994), another wildlife species whose control in forestry enterprises costs large sums of money (see below).

2.3. Why control deer populations?

A wide range of people, with various motivations, have an interest in the control of deer populations and damage. Deer damage has long been a major concern to forestry (Gill, 1992a,b), and with recent expansions in deer ranges and abundances (Staines *et al.*, 1998; section 0), there is increasing concern about damage to agricultural crops and pastures (Scotland: Mitchell *et al.*, 1977; Callander & McKenzie, 1991; England and Wales: Putman & Moore, 1998; Packer *et al.*, 1998; Staines *et al.*, 1998). Furthermore, such concern now extends to natural tree regeneration and ground flora in semi-natural woodlands (Mitchell & Kirby, 1990; Cooke, 1994), with potential impacts on the diversity of birds, small mammals, and invertebrates (Stowe, 1987; Hill, 1985; Petley-Jones, 1995). Culling is also used to stop the spread of exotic deer species. On the other hand, however, deer are widely believed to be valued by the public for their aesthetic appeal (Exmoor NP submission to the Inquiry; Scottish Natural Heritage, 1994), form a valuable natural and renewable resource as venison, and generate stalking revenue.

Many interest groups perceive population control by culling as one of several means available to reduce damage levels (see also section 3.6.1.b). British deer species are also culled for sport (deer stalking and hunting with hounds; Whitehead, 1964; Hamilton, 1907), and stalking can generate significant revenues through venison, stalking fees and trophies. These various (competing) objectives often need to be balanced against one another within a single multipurpose landholding or area covered by a co-operative Deer Management Group, and the primary motivation for seeking to control deer will vary according to local land use.

2.3.1. Why do farmers seek to control deer populations?

2.3.1.a. Data and approach

While quantitative data on losses to farm crops because of deer damage remain limited (Putman, 1986; Putman & Kjellander, in press), a number of recent questionnaire-based surveys have helped to assess the perceived significance of such damage in England and Wales. We draw heavily on two of these.

2.3.1.a.i. ADAS questionnaire survey

The most comprehensive questionnaire survey thus far, conducted by ADAS during 1995, was distributed by post to 3322 landholders representing four main user groups (agriculture, forestry, nature conservation, recreation) to assess their attitudes and practices towards lowland deer (Doney & Packer, 1998; Packer *et al.*, 1998). Sampling extended to four geographic regions (Somerset and Gloucestershire; Essex and Suffolk; Northamptonshire; lowland Yorkshire), chosen to represent a range of agricultural uses and landscapes across the known distribution of lowland deer. The questionnaire asked for details of land area; crop or habitat types present; species of deer present; frequency and number of deer seen; nature and estimated cost of damage caused by deer; and methods and perceived effectiveness of damage prevention.

A total of 1546 (47%) valid responses were received, with high response rates across all sectors (ranging from 82% from conservationists to 36% from forestry holdings). The reliability of responses and perceived damage levels reported was assessed in 1997 by a ground-truthing study encompassing 25 agricultural holdings, focussing on damage to cereals and farm woodlands (Doney & Packer, 1998; Packer *et al.*, 1998).

2.3.1.a.ii. Quantocks Deer Management and Conservation Group questionnaire survey

The Quantocks Deer Management and Conservation Group circulated a similar questionnaire during March 1998 to all known landholders within or close to the boundary of the Quantocks AONB, one of the few remaining areas where deer are subject to hunting to hounds (Langbein, 1998). Its purpose was to obtain better information on the occurrence and current management of deer on local estates, including the landholder's own estimates of the financial losses associated with deer damage, and their views on how the deer should best be managed in future. The questionnaire was sent to 165 addresses of local farms or known landowners, eliciting 68 responses (43%); the actual percentage of landholders replying was, however, much higher, as many responses covered two or more holdings farmed by the same person. Together, the respondents accounted for 14,649ha, an area slightly larger than the AONB itself. Pasture and grass ley accounted for the largest proportion of the land (29%) followed by cultivated land (25%) and moorland (25%).

2.3.1.b. Do farmers consider deer a pest?

In the WildCRU's 1981 survey of farmers (section 2.2.1.a.ii), 13% of respondents reported that deer caused damage, though there was considerable regional variation (Table 2-7). Deer were most frequently reported causing damage in Dorset.

Table 2-7 Regional patterns of reported control of deer (1981 survey). % Answering yes to: 'do you attempt to control deer?' and do you believe you suffer significant damage from deer?' WestC=Devon and Cornwall.

	Dorset	Leic.	Oxon.	Salop	Suffolk	Sussex	Warw.	WestC.	Yorks.
% Control:	23.8	1.4	5.5	0.0	10.1	7.0	6.6	8.2	6.7
% Damage:	40.3	1.2	8.1	1.8	14.7	19.4	9.3	11.6	5.7
N	119	64	124	55	68	85	75	95	88

In the more recent ADAS survey across lowland England and Wales, deer were present on the holdings of 69% of 1192 agricultural respondents, and 38% believed that deer cause significant damage (Packer *et al.*, 1998). Most felt that agricultural damage from deer had increased (42%) or stayed the same (48%) between 1990-1995, but remains a limited and mostly localised problem (Doney & Packer, 1998.). Where deer were present, 18-51% of respondents reported damage, depending on region, and probably influenced by the species its and abundance. The majority of farmers agreed or strongly agreed that "*the damage caused by deer causes significant economic loss*" in Somerset (45% of 294 with deer present), and Essex and Suffolk (36% of 141), but disagreed or strongly disagreed in Northamptonshire (51% of 35) and North Yorkshire (49% of 100).

In the Quantocks AONB, where large number of red deer are present, 74% of landholders (most being farmers) considered that deer caused significant damage on their land, with most ranking deer as more damaging than rabbits, badgers, or foxes (Table 2-3). Just over half (52%) the farmers said deer were the most damaging of a list of six mammal species (Langbein, 1998a).

2.3.1.c. What damage to farming is attributed to deer and how significant is it?

Cereals were perceived to be affected by deer by almost half (44% of 822 replies with deer) of the farmers responding to the 1995 ADAS survey (Doney & Packer, 1998), followed by damage to trees (29%), grass (6%), root crops (3%), fruit (3%), vegetables (3%), and oilseed rape (3%). For farms growing mainly cereal crops, 17% of respondents claimed no annual cost of deer damage, and 85% perceived deer damage to be £500 or less per annum.

In the Quantocks AONB, 74% of 68 landowners, holding between them 89% of the 14,649ha covered by the survey, agreed that deer caused significant economic losses (Langbein, 1998a). The median annual losses due to deer damage was also estimated at around £500 per holding (mean holding size 92ha) if including the 30% of respondents suggesting zero or <£100 damage; or £800 if restricting analysis to those reporting at least >£100 losses due to deer. As in the case of results from the wider ADAS study above, most farmers believed that cereals were affected (54%); a high proportion of landowners also stated that they had suffered damage to pasture and sown leys (41%), hedges and banks (34%) and woodland (34%).

2.3.1.c.i. How realistic are estimates of the costs of deer damage to agriculture?

Although it is well established that deer cause damage to crops and forestry, without experience deer damage is often difficult to distinguish from that caused by other mammals such as rabbits and hares. Furthermore, even where landholders may have correctly ascribed damage to deer, their estimates of the costs incurred through such damage often bear no resemblance to actual losses.

This was demonstrated recently by an ADAS survey (Doney & Packer, 1998; Packer *et al.*, 1998; section 2.3.1.a), which found that while farmers were mostly (75-80%) accurate in reporting deer species and approximate abundance, they were generally incorrect about the economic value of damage to cereals. Farmers were as likely to underestimate the costs of damage as to exaggerate it. Actual losses due to grazing of winter wheat were assessed during follow-up ground truthing at up to 0.57 tonnes per hectare on farms which were visited regularly by roe or fallow deer, but at lower levels of grazing, a negligible economic loss, or an actual gain in yield, was recorded (Doney & Packer, 1998; Doney, 1998).

The costs of deer damage to crops is made more difficult to estimate because, over a period of months or years, plants can recover to some extent, or even benefit from grazing or browsing. For example, in Hampshire, roe deer cause substantial levels of apparent damage to cereal fields in spring, but by harvest this may often be negated through tillering and increased growth rates (Putman, 1986b).

A further complication to estimating the costs of deer damage is that deer are themselves a valuable resource. During the late 1980s, when the price of venison was low, the Red Deer Commission reported an increase in the numbers of complaints about deer on farmland (RDC, 1989 [in SNH, 1994]).

2.3.1.d. Which species of deer do farmers seek to control?

In the 1995 ADAS questionnaire survey, the perceived significance of damage levels recorded by farmers indicated very little differences between areas in which fallow, red, roe or muntjac deer were present (Packer *et al.,* 1998), although roe deer were the species most regularly associated with damage overall (partly reflecting their wider distribution across all sample areas).

Fallow, red and roe deer were the three species most frequently associated with damage to agriculture in an earlier review (Putman & Moore, 1998), based on the frequency of unsolicited requests for advice received by ADAS between 1985-1989. Different deer species tended to be associated with different types of damage. Most reports of damage to oilseed rape or to nursery crops, garden shrubs, and top fruit involved roe deer, which were relatively rarely implicated in reports of damage to grass or cereals. Reports of damage by red deer were largely in connection with pasture, silage crops, or field cereals, while 76% of all complaints concerning damage to field cereals cited fallow deer. By comparison, little damage was reported to cereals from sika deer, muntjac or Chinese water deer (Putman, 1995; Putman & Moore, 1999).

Damage by fallow (and red) is often localised to areas where large herds, sometimes of 70-200 animals, aggregate on favoured farmland feeding grounds (Langbein, 1996). Potential damage by red deer, partly by virtue of their larger individual size, tends to be viewed most seriously and can be locally significant (Callander & McKenzie, 1991; Langbein, 1998; Putman & Langbein, 1999).

2.3.2. Why do foresters seek to control deer populations?

The habitats preferred by all six of our deer species are associated with open forest or woodland edge, and at the national scale prevention or limitation of damage to young tree plantations is probably the single most important reason for which deer control tends to be undertaken in Britain. Aside from hindering the establishment of new commercial tree plantations, deer may also cause damage to amenity trees, natural tree regeneration from seed, coppice management, and standing timber (Gill, 1992a,b).

In many parts of Scotland, where already high but still rising deer densities have been experienced throughout much of this century, establishment of commercial tree plantations without either deer population control or fencing or both has long been regarded as non-viable (Staines & Welch, 1989). By comparison, deer distribution and abundance south of the border has been relatively restricted in the past, but the significant increases noted over the last 40 years, especially of roe, fallow and muntjac deer, have made this an equally important issue in England and Wales.

2.3.2.a. Do foresters consider deer a pest?

The recent ADAS questionnaire survey by Packer *et al.* (1998) indicated that 57% of foresters considered that deer cause significant damage, a greater proportion than any other user group (agriculture, conservation, and recreation).

The attitude among foresters that deer cause damage, and that their numbers therefore need to be rigorously reduced, is beginning to change in favour of more positive management. National and European policies for sustainable forest management are increasingly aimed at delivery of a wide range of benefits, including biodiversity, conservation and recreation, even from those forests still managed primarily for timber production (HMSO, 1994). Persistently high numbers of deer can be detrimental to these goals, but there is increasing consensus that in many types of woodland retention of some grazing by deer or domestic stock is preferable to their total exclusion (Kirby, 1993; Hester & Miller, 1995; Putman, 1986, 1996; Kuiters *et al.*, 1996). Deer play an important role in creating a diverse structure which benefits other species (Ratcliffe, 1998). Current Forestry Authority and Deer

Initiative advice therefore recommends that: "*Management should aim to maintain healthy deer populations in balance with their environment, rather than to eliminate deer from an area altogether*" (Forestry Practice Advice Note 2, 1995)

2.3.2.b. What damage to forestry is attributable to deer, and how significant is it?

Potential damage by deer to trees may take various forms. *Browsing* (biting off buds, foliage or shoots) and *bark stripping* (peeling of bark to eat) tend to be most common in winter when other food sources are scarce, and during shoot elongation in spring. *Fraying* (using antlers to abrade and partially remove the bark from stems and branches) may occur at different times of the year depending on the deer species involved, as male deer mark their territories and clean their newly grown antlers of velvet by rubbing them on young trees (Gill, 1992a; Langbein, 1993; Forestry Authority, 1995). Browsing and bark stripping in particular can result in serious losses of young trees, whether naturally regenerated, or in plantations (Mitchell *et al.*, 1977; Staines & Ratcliffe, 1987). The susceptibility of trees to damage is very variable between tree species, age, deer species, and frequency and type of damage.

Numerous published studies (Welch *et al.*, 1992; Van Hees, 1996; Putman *et al.*, 1989; Ammer, 1996; Langbein, 1997; Cooke, 1994) have experimentally quantified the effects of deer browsing on tree survival and growth form, natural regeneration, relative abundances of canopy tree species, and ground flora. In commercial sitka spruce plantations in Scotland, Welch *et al.* (1992) concluded that the overall effect of red deer browsing was equivalent to a check of about one year in the time taken to reach a height of 80cm. In semi-natural woodland, Putman *et al.* (1989) found that an area protected from fallow deer browsing had a density of 6440 saplings/ha after 14 years, in contrast to only 20/ha in an adjacent heavily browsed area. Deer (especially fallow) can significantly hamper establishment of Farm Woodlands (incentive schemes aimed at converting agricultural land to woodland). Of 74 (mainly broadleaved) farm woodland plantations in east Suffolk, 21% suffered substantial damage from fallow deer, with over 20% of the leader shoots damaged in one year; cherry and rowan were the most frequently browsed tree species (Key *et al.*, 1998).

Deer damage (by browsing and bark-stripping) can amount to as much as 40% of the crop of coniferous plantations in the Scottish uplands (Maxwell, 1967), and average losses close to 50% of the leader shoots have been recorded even in Sitka spruce crops (Staines & Welch, 1989). Allison (1990) estimates that deer damage to forestry in Galloway, Scotland, costs £2 million per year. Aside from actual costs of damage or failure of plantings, further costs are incurred through preventative measures aimed at reducing damage, such as fencing, or employment of stalkers to reduce deer numbers. Gill (in SNH, 1994) has estimated the total cost of red and roe deer damage and control in Forestry Commission plantations in Britain to be in the region of £5 million (net of revenues from venison and stalking – *c.* £1.26m), equivalent to *c.* 7% of the total revenue generated by Forestry Commission timber sales in 1989/90.

Comprehensive reviews of the damage caused by deer to forestry, and its likely economic significance in England and Wales, are provided by Gill (1992a,b), and Putman & Moore (1998).

2.3.2.b.i. How realistic are estimates of the costs of deer damage to forestry?

Damage to forestry by deer is not regularly surveyed in Britain, there are surprisingly few published data with which to estimate the true financial costs of deer damage, and how that relates to the expense of deer population control and protective fencing. The variability in the few available estimates serves to underline the difficulties in accurately assessing such costs (Gill, 1992b; see below). In Europe, estimates of the annual cost per hectare of woodland range from <£1 for browsing by moose in Sweden (Jantz, 1982) to £85 for red and roe deer browsing in Germany (Spiedel, 1980).

As with cereals, one of the difficulties of assessing the long-term costs of damage to trees is that unless they have been killed outright, they will often recover completely. Sitka spruce affected by terminal bud damage often respond by 'flagging', where one of the lateral branches migrates round to take over apical dominance, sometimes leading to a net increase in height compared to undamaged individuals (Staines & Welch, 1984).

A further complication is that different tree species may have different susceptibilities to, and recovery rates from, damage. Extensive exclosure studies in the Netherlands (Van Hees *et al.*, 1996) showed that browsing by red and roe deer had clear effects on mortality and growth rates of silver birch and oak, but less significant effects on beech. Interestingly, however, they note that there may also be an indirect 'benefit' of deer browsing in that grazing and browsing reduce the competition suffered from non-crop species.

2.3.2.c. Which species of deer do foresters seek to control?

Where present, all the deer species (with the possible exception of Chinese water deer which do not yet occur in significant numbers) tend to be implicated in damage to forestry (Gill, 1992a,b). In their ADAS questionnaire survey, Packer *et al.* (1998) found that the species of deer present (red, fallow, roe, muntjac) on each forestry holding had no effect on whether respondents perceived that damage to be of economic significance.

Fallow, sika, and red deer, as preferential grazers, tend to include rather lower proportions of woody browse in their diet than do the smaller roe and muntjac deer. Whereas this might be expected to lead to major differences in levels of tree damage, such differences are often largely negated by the greater total intake rates by the larger three species, as well as their greater tendency to form herds. The three larger deer species also pose a greater potential threat to forestry (and costs in protection) in view of the greater height to which they will browse, and hence the longer period of vulnerability until growing trees are out of reach to the deer. For establishment of Farm Woodlands, fallow and roe tend to be the most frequent species associated with damage, not least due to their wide distribution in England (and increasingly also Wales), and the tendency of fallow to build up high numbers even in agricultural areas offering only small patches of woodland for cover.

2.3.3. Why do conservationists seek to control deer populations?

Grazing and browsing by wild herbivores have always played a role in determining the structure and dynamics of natural ecological systems. It is considered ‘damage’ when the consequences are extreme and/or conflict with human interests or management objectives, but grazing and browsing by deer within conservation communities also has many positive, facilitative effects, such as preservation of ancient wood pasture and associated wildlife species (e.g. Bakker *et al.,* 1983, 1984; Putman, 1986a, 1996b). Persistently high numbers of deer can be detrimental to tree regeneration or lead to losses of ground flora sensitive to grazing, but there is increasing consensus among conservationists that some grazing is beneficial (Kirby, 1993; Hester & Miller, 1995; Putman, 1986, 1996; Kuiters *et al.*, 1996).

2.3.3.a. Do conservationists consider deer to be a pest, and why?

Deer were considered to cause significant damage by 34% of conservationists replying to an ADAS survey questionnaire (Packer *et al.,* 1998; see below); this is comparable with the farmers’ response to the same question.

In a recent survey of National Nature Reserves conducted on behalf of English Nature (Putman, 1996b), questionnaires were sent to managers of 162 sites designated as National Nature Reserves throughout England. Not all sites had deer present; of the 112 site managers recording deer visiting or resident within their reserve, 45% recorded a measurable impact at some level (browsing damage to coppice, lack of regeneration, impact on ground flora). Sites that reported damage from deer were without exception woodland reserves - managers of ‘open sites’ (grasslands, meadows, heath land or fenland sites) generally regarded the presence of deer as neutral or positively advantageous in suppressing encroachment by scrub. Within woodlands, the vast majority of complaints concerned browsing damage to coppice regrowth.

Overall, only 18% of managers of reserves with deer present considered that damage sustained was sufficient to cause difficulty in meeting management objectives for the site; all of these considered that current management measures (culling, fencing of vulnerable areas) were adequate at present to reduce damage to tolerable levels. However, Putman himself points out that such figures should be interpreted with some caution, in view of differences between sites, not only in levels of damage and numbers of deer, but also in methods of damage prevention already in place. As with foxes (section 2.1.1.a), management history thus confounds the responses, and the proportion of managers reporting lack of conflict in meeting management objectives should not simply be equated to those who would find no conflict in the absence of any management (Putman, 1996, 1998).

2.3.3.b. What damage to conservation is attributed to deer?

Grasslands and lowland heaths rely on the maintenance of grazing to maintain their characteristic structure and diversity, and such communities are more likely to be at risk from reduction of grazing than from increasing deer populations. In certain situations, however, there may be a case for controlling the level of grazing, for example, where impact has risen to such a level that it conflicts with other management objectives determined for a particular site. High densities of red deer are well known to have significant impacts on the growth and regeneration of heather in many parts of the Scottish uplands (e.g. Staines *et al.*, 1995; Clarke *et al.*, 1995; Stewart & Hester, 1998). However, in most parts of Exmoor, the contribution to heather off-take by red deer is of fairly minor importance, and much lower than off-take from sheep grazing (Langbein, 1997).

Woodland communities are most prone to damage from over-grazing by deer and other ungulates, with particular concerns about damage to natural tree regeneration and ground flora in semi-natural woodlands, and damage from deer to regeneration of coppice coupes (Mitchell & Kirby, 1990; Putman, 1994a, 1996b; Cooke, 1994).

Within woodlands, heavy grazing pressure may have a number of distinct effects on regeneration of woodland trees, on structure and composition of field and shrub layers, and on species of the woodland floor. Where losses of mature trees through browsing damage or simply old age are complemented by virtual lack of regeneration due to depletion of the seed source or heavy browsing pressure on new seedlings, browsing mammals start to exert a substantial impact on the entire woodland structure (e.g. Peterken & Tubbs, 1965). Even light but selective browsing, taking a preponderance of preferred species such as ash or oak by comparison to less preferred birch, alder or sycamore, can lead to quite pronounced shifts in species composition of canopy trees (Gill, 1992; Putman, 1996b; Van Hees *et al.,* 1996).

Within the Exmoor National Park, Langbein (1997) recorded significantly higher mortality and reduced growth of seedlings of oak, rowan, and beech in a number of old oak coppice woods grazed by red deer and sheep, compared to fenced plots in the same stands. Very few saplings reached heights significantly above the shrub layer in most old coppice stands studied, except those where sheep were excluded and red deer numbers were comparatively low (<5 per 100ha).

Browsing may also lead to elimination of the shrub layer, and the reduced field layer resulting from heavy grazing can reduce the abundance and diversity of invertebrates and small mammals (e.g. Hill, 1985; Putman, 1986a); this may in turn reduce diversity of raptors or mammalian predators (e.g. Tubbs, 1974, 1982; Putman, 1986; Putman *et al.*, 1989; Petty & Avery, 1990). Other species, however, may derive positive advantage from such heavy grazing. Wood warblers, pied flycatchers, and redstarts all depend on the park-like conditions of traditional wood-pastures (Stowe, 1987; Mitchell & Kirby, 1990).

Finally, heavy grazing pressure can result in dramatic changes in the composition and relative abundance of species of the woodland floor, which may be of serious consequence if that flora itself contains rare or threatened species. Recent declines in oxslip populations in many conservation woodlands of East Anglia (e.g. Hayley Wood, Cambridgeshire, Hales Wood, Essex) have been blamed - rightly or wrongly - on the coincidental rapid increases in range and number of fallow deer throughout that region (Rackham, 1975; Tabor, 1993, 1999). Cooke (1994a,b) has reported declines in bluebell and dog's mercury in Monk's Wood NNR and other Cambridgeshire woodlands, associated with heavy grazing pressure from muntjac. Deer browsing more generally results in increases in some species: for example, bracken, grasses, mosses, foxglove, and ragwort; and decreases in others: for example, bramble, honeysuckle, ivy, wild rose, and holly (Gill, 1997).

2.3.3.c. Which species of deer do conservationists seek to control?

All deer species pose some threat to conservation of habitats and other wildlife if allowed to reach very high densities. In Scotland, overgrazing by red deer is of particular concern in relation to conservation and restoration of native Scots pine woods (Holloway, 1967; Beaumont *et al.*, 1994; Palmer *et al*., 1998). In England and Wales, fallow, and in some regions red deer, tend to be the main species implicated in preventing natural tree regeneration, due to their tendency to build up very high densities locally (Putman, 1996; Langbein, 1997). Muntjac are implicated especially in damage to ground flora (e.g. Cooke, 1994).

In the questionnaire survey of National Nature Reserves undertaken on behalf of English Nature (Putman, 1996b), there was no statistically significant association between the type of deer species present and responses regarding ability to meet conservation objectives for the site. However, of the 25% of responses that suggested deer damage is of some concern in meeting management objectives, fallow and muntjac were the most frequently reported problem species (Putman, 1996).

2.3.4. What is the importance of deer population control for public amenity and as source of revenue for landholders?

Aside from the various needs to control of deer to prevent the damage outlined above, the presence of deer must be also be recognised as an aesthetic benefit (their very presence giving pleasure to people), and an exploitable resource, generating income for the landholder through recreational stalking, venison, and hunting.

2.3.4.a. How important are deer for public amenity?

Deer, as the largest terrestrial wild mammals remaining in Britain today, are highly regarded among the public, with people taking pleasure from seeing them in the wild or even merely knowing that they continue to thrive in our increasingly industrialised landscapes. In England, this is true in particular for many of the large traditional ‘deer forests’ such as Epping Forest, the New Forest, and Exmoor. Continued provision of deer viewing opportunities is an important aspect of published management plans for each of these three areas (Langbein, 1996; Putman & Langbein, 1999; Forestry Commission, 2000; Exmoor NP submission to the Inquiry).

The popularity of the red deer within Exmoor National Park (one of the remaining areas where hunting deer to hounds occurs) was highlighted by the results of a survey undertaken by the Park Authority (Park Life, July 1999). This showed that local residents placed red deer above any of the many other ‘special features’ that they valued most about Exmoor. The preservation of reasonable numbers of red deer on Exmoor and the Quantocks is thus of some importance to local tourism; however, for this, the total size of the deer population is likely to be of lesser importance than ensuring that good numbers remain near to those areas most frequented by the public.

2.3.4.b. How important are wild deer as a source of income?

As discussed above, each of the ‘quarry’ species considered in this report tends to be controlled partly (often primarily) because they are perceived to cause significant (if highly variable) damage to the interests of farmers, foresters, gamekeepers or conservation. The quarry species itself may also be of some economic value to the landholder, for example as saleable meat, pelts and for commercial shooting. Unlike the three other quarry species (hares, foxes, mink), the direct revenue potential from deer is relatively high.

Direct income from deer may be derived from a ‘harvest’ of venison, stalking fees or lets, and trophy fees. In view of the considerable size of some deer species, even the annual income through venison sales (*c.* £1.50-2.50/kg carcass weight) arising from any ‘necessary’ culling may contribute significantly towards covering any direct costs of deer control. Although variable, total revenues from venison in Scotland may exceed £3-6 million (SNH, 1994); no overall data are available for England and Wales, but here incomes are also likely to exceed £1 million per year.

Over and above the carcass value of culled deer, some landowners are also able to sell stalking opportunities on their land. This may be based on fees per outing (often accompanied by a professional deer ranger) or numbers of deer shot, usually with an additional premium charged for any stag or buck, depending on the quality of the (antler) trophy. Charges for accompanied stalking start at around £75 per day but vary widely between areas and between deer species, with a still wider range of additional charges for shooting a stag carrying ‘medal’ quality antler trophies (usually based on criteria laid down by the Conseil International de la Chasses). Alternatively, some landowners may simply let the deer stalking rights to a third party for an annual rent.

The sustainable exploitation of deer as a renewable natural resource can thus be an important motivation for some landholders to maintain deer populations on their land at particular levels. The extent to which each of the differing sources of income from deer can be exploited varies widely between sites, depending on the primary landuse objective of a given estate, the size of the holding or co-operative deer management area, and the habitats and deer species present. For a given estate the optimum size and structure of the deer population will also differ according to whether the main focus of exploitation is on venison sales, stalking lets, or trophies. Optimising return from venison sales usually requires maintenance of a fairly high population density and a high female to male sex ratio (*c.* >3:1), with the emphasis on ‘harvesting’ the surplus of young males (de Nahlik, 1992). However, maintenance of very high densities and heavily female biased sex-ratios is often inappropriate for management of wild deer populations, due to increasing impact on other landuses, and reductions in health and ‘trophy’ quality if resources become limiting. It may also become increasingly difficult to achieve adequate annual culls to keep population numbers under control.

Commercial exploitation of wild deer, especially through deer stalking, is widespread throughout Europe (Gill, 1990). While roe and fallow deer stalking is widely available throughout much of England, in Scotland the emphasis has long been on red deer stalking in the Highlands. There, it has been a major factor with regard to land management and the development of the large red deer populations. The average number of stags shot per annum has commonly been used as the main yard stick in assessing the capital value of the large Scottish shooting estates or 'deer forests' (e.g. Callander & McKenzie, 1991).

2.4. Why control hare populations?

As with deer, issues surrounding the control of brown and mountain hares are complicated by the fact that they are simultaneously considered a pest (by farmers, foresters and, for mountain hares only, gamekeepers), are a game species culled for their meat and for sport, and are of conservation concern.

Once considered abundant, the brown hare has declined substantially since the early 1960s to an estimated British pre-breeding population of 817,500 in the 1990s (Hutchings & Harris, 1996). This pattern of decline has also been seen in much of Europe during the latter half of the twentieth century. In Britain, the brown hare is subject to a Species Action Plan intended to "*maintain and expand existing populations, doubling spring numbers in Britain by 2010*" (Anon, 1995). However, the species is by no means rare (section 10.3.2): in Britain, brown hares are in fact the most numerous of all the species considered in this report.

In England, the mountain hare is found only in the Peak District and is extinct in Wales. Of the ten species covered in this report, the mountain hare is the only one subject to special protection under international law (Appendix III of the Berne Convention 1979; section 11.4). Control of mountain hares is nonetheless permitted.

2.4.1. Why seek to control populations of brown hares?

Hares eat crops such as oilseed rape and turnip, and particularly eat grasses and cereals. In addition, hares can eat high value market garden crops, and will often browse and kill newly planted young trees and shrubs. Some of this damage can be of economic significance to individual growers.

However, since the second world war, although commonly still regarded as a minor pest (Mellanby, 1981) the general agricultural damage hares inflict has never been serious enough to warrant the MAFF funded research efforts that went into the control of species like the brown rat, rabbit or woodpigeon. In livestock districts today, hares are not numerous and are rarely considered a pest. In arable areas, high numbers of hares on winter corn are considered damaging by most cereal farmers, and regular winter culls by shooting are undertaken where this occurs. Although the economic loss to cereals has not been calculated in Britain, hares spend much of the late winter feeding on winter cereals (Tapper & Barnes, 1986).

In some areas, such as the South Downs, landowners may also seek to reduce the abundance of hares on their land to deter illegal coursing. We have no data on the extent to which this occurs, but certainly it is a common perception, and the AMHB (submission to the Inquiry) state that the police support a policy of hare culling to remove the threat of poachers.

2.4.2. Why seek to control populations of mountain hares?

In some areas mountain hares may cause localised damage to forestry and moorland vegetation. They may also compete with livestock and grouse. Mountain hares are sometimes shot as a quarry. We are not aware of any evidence that the English population causes damage or is controlled, except in the context of game-shooting.

2.5. Why control mink populations?

American mink were the first carnivore to be introduced to Britain since the domestic cat, over a thousand years ago (Yalden, 1999). Mink are popularly regarded as voracious predators that roam the countryside devouring all wildlife, stock or pets in their path (Dunstone, 1993). The reality is, of course, much more prosaic, but as a recently introduced predator, the American mink brings with it some unique problems.

2.5.1. Why do farmers seek to control mink populations?

Mink are controlled by farmers because of their perceived impact on native wildlife, fish, game birds and livestock (particularly poultry, but sometimes lambs), and have a reputation for the surplus killing of confined animals (Dunstone, 1993).

In 1996, researchers at WildCRU distributed a questionnaire regarding mink to 40 farms along the River Thames in west Oxfordshire. Of the 32 farmers who replied, 29 (91%) considered the mink to be a pest (Strachan, unpublished data). The most frequently cited impact of mink was on waterbirds (37.5%), followed by water voles (28%), and fish (22%). Birds and general wildlife were cited by 19% of farmers, and only 12.5% cited damage to poultry.

Harrison & Symes (1989) examined the different types of damage attributed to mink from 195 reports to MAFF in South-west England by farmers during the periods 1961-70 and 1985-86. The killing of poultry and domestic or ornamental waterfowl accounted for 60% of the incidents reported. Although losses are undoubtedly important to individual landowners, where they are taken, poultry form only a small component of the mink’s diet: in one study of 685 scats, chickens and game birds formed less than 1% of the diet (Birks, 1986).

On the Scottish islands of Harris and Lewis, mink are believed to have made the keeping of outdoor poultry almost impossible. Before mink colonised these islands, 90% of the 4,000 registered crofts are estimated to have kept poultry; nowadays fewer than 10% continue to do so. Based on an average flock size of 10 birds, the net annual cost of this constraint to the crofting economy on the two islands is estimated to be £586,000 (M.C. Swan, unpubl. analysis).

2.5.2. Why do game managers seek to control mink populations?

A 1996 survey of gamekeepers in Wales and the West Midlands found that the 66 respondents ranked the mink as the third most serious predator of game, after foxes and feral cats (Packer & Birks, 1999; Table 2-3).

Mink are perceived to have an impact on game birds, ducks, fish, and native wildlife such as water voles. As with poultry rearing, penned pheasants and partridges have been killed after mink have gained access to the rearing enclosures with surplus killing frequently cited. Wild stocks of game birds and recently released game birds are also believed to be at risk, reportedly predated as adult birds or as chicks or eggs during the breeding season. However, we are not aware of any data quantifying the extent to which mink predate on gamebirds.

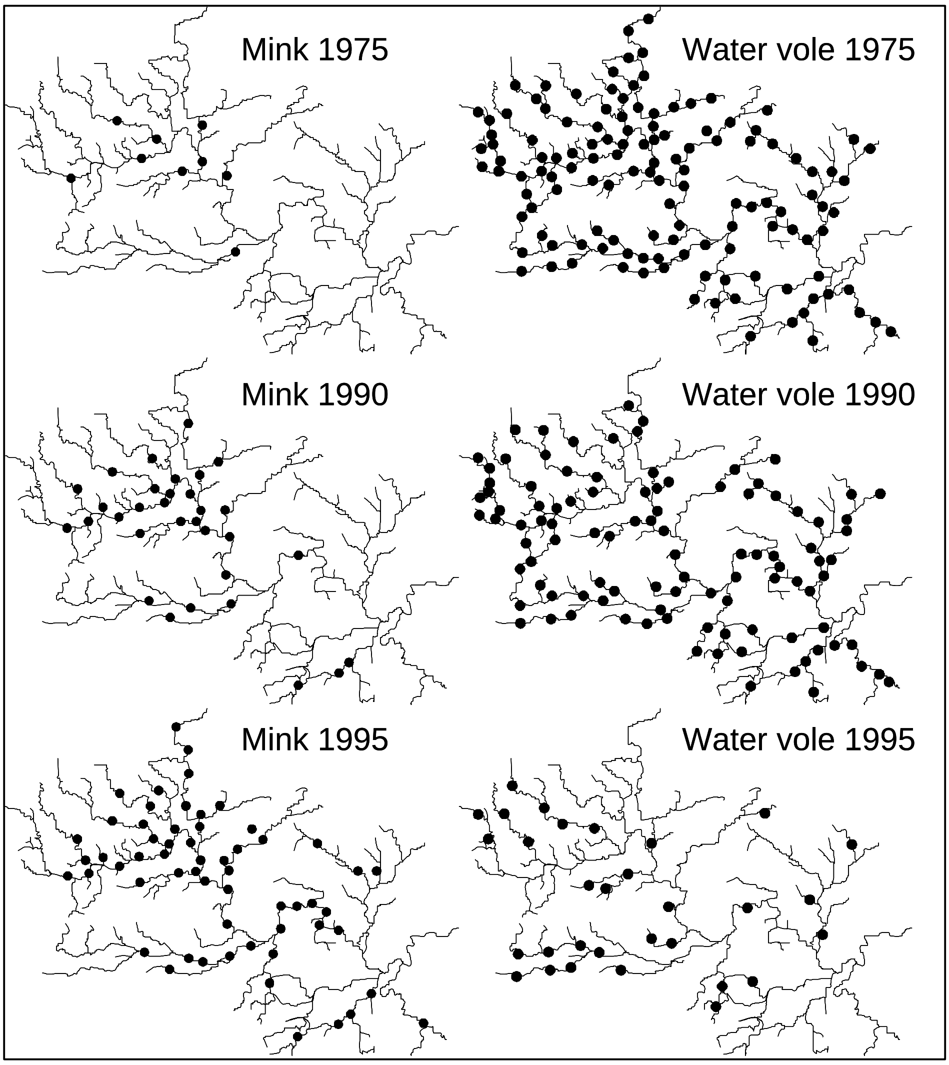
2.5.3. Why do conservationists seek to control mink populations?

Like farmers, conservationists believe that there is good evidence that mink have an impact on water voles, waterbirds (especially ducks, grebes, coots and moorhens), vulnerable ground-nesting birds such as the corncrake, lapwing or redshank, sea bird colonies (especially terns, gulls, eiders, black guillimot, shearwaters and puffins), and native white-clawed crayfish. There is an extensive and growing scientific literature regarding the impact of mink on wildlife (see reviews by Birks, 1990; Dunstone, 1993; Macdonald & Strachan, 1999; Craik, 1997, 1998). The UK Species Action Plan for the water vole encourages the control of mink as a conservation tool to protect key populations of the vole throughout its range.

2.5.3.a. What is the impact of mink on the water vole?

One of the strongest conservation arguments for controlling mink in Britain is to protect vulnerable water vole populations. The water vole has declined by an estimated 88% of its total population between 1989/90 and 1998 (Strachan *et al.*, 2000). Following pioneering work by Woodroffe *et al.* (1990) on the Yorkshire Moors, there is increasingly powerful evidence that predation by mink, in association with habitat degradation and fragmentation, is a causal factor in the vole’s decline (Macdonald & Strachan, 1999; Strachan *et al.*, 1998; Lambin *et al.*, 1998; Woodroffe *et al.*, 1990).

Figure 2-5 The changing distribution of mink and water voles in the Thames catchment.



A series of studies (Figure 2-5; Macdonald & Strachan, 1999) in the Thames catchment area indicate that the vole decline there has continued for over twenty years, since the mink first arrived in the 1970s. However, as recently as 1990, mink were uncommon in the catchment and water voles were still found in three quarters of the sites they had occupied earlier this century. A 1995 survey revealed a rapid increase in mink numbers and a catastrophic decline in the number of sites at which water voles persisted. No site was found on the Thames at which the two species occurred together. In 1991, Halliwell & Macdonald (1996) live-trapped American mink along 20km stretches of four rivers in the Thames catchment and found a negative correlation between the numbers of mink caught and the numbers of water vole signs.

Studies in the Thames, therefore, lead to the conclusion that, at least when mink numbers increase, their impact on water voles can be very rapid, leading to widespread losses within only five years. Indeed, evidence from both the Windrush and the Soar suggest that within one breeding season, resident mink may greatly reduce numbers of water voles, and within only two years eradicate them locally. Female mink, which hunt close to their nursery dens, may have a particularly rapid impact on local water vole colonies (Strachan *et al.,* 1998).

In a study of mink diet on 11 rivers in Derbyshire, Leicestershire, Staffordshire and Nottinghamshire over 1993-94, Strachan & Jefferies (1996) showed that water vole was the single most important species in the diet of colonizing mink. Indeed, in the May-June sample they comprised up to 32% of the volume of undigested prey remains. Similarly, analysis of 863 scats collected in 1993/94 along the River Soar demonstrated that water voles can be an important component of the diet of mink colonizing a river, but thereafter their importance declines as they are depleted (Strachan *et al.,* 1998).

2.5.3.b. What is the impact of mink on birds?

Mink have a serious impact on the breeding success of colonial ground nesting sea birds. They are believed to have caused terns colonies along the Scottish west coast between Mallaig and West Loch Tarbert to halve in the 11 years 1987-1998 (Craik, 1998).Craik (1993, 1995, 1997) describes surplus killing by single mink arriving at tern nesting islands before chicks have been fledged.

Less clear is the effect of mink predation on riparian bird species. Early studies led to contradictory conclusions (Lever, 1978; Linn & Chanin, 1978). As mink colonized different parts of Britain, considerable concern was expressed about the possible disappearance of moorhens (Smith, 1988). More recently, Halliwell & Macdonald (1996) found no significant correlations between mink and moorhen censuses from eight 10km sections of four lowland British rivers. In the Upper Thames, taking habitat into account, coot but not moorhen abundance was negatively related to the presence of mink, and fewer coot chicks were raised by pairs living in mink-occupied zones (Macdonald & Strachan, 1999; Ferreras & Macdonald, 1999). These results accord with faecal analyses which suggest that the mink’s impact on moorhens during the breeding season was less than on the coots. The reason for this difference may be that coots nest on the water surface among rushes and reeds, while moorhens nest in shrubs and trees.

In the Upper Thames, coots and moorhens together constituted 10% of the biomass ingested by mink, based on the analysis of 115 scats. Other water birds constituted 2% of ingested biomass. Higher proportions of waterfowl in mink diet have been related to high waterfowl densities and to the scarcity of other prey such as fish and crayfish (Chanin & Linn, 1980; Eberhardt & Sargeant, 1977).

2.5.3.c. What is the impact of mink on other species?

American mink arrived in Britain at a time when two other members of the guild of semi-aquatic predators were seriously reduced (Birks, 1989; Jefferies, 1989). These two, the amphibious otter and the more terrestrial polecat, are currently increasing and spreading into areas now occupied by American mink. The implications of this recolonization are likely to affect directly population processes of all three mustelids, and thereby indirectly to affect their prey.

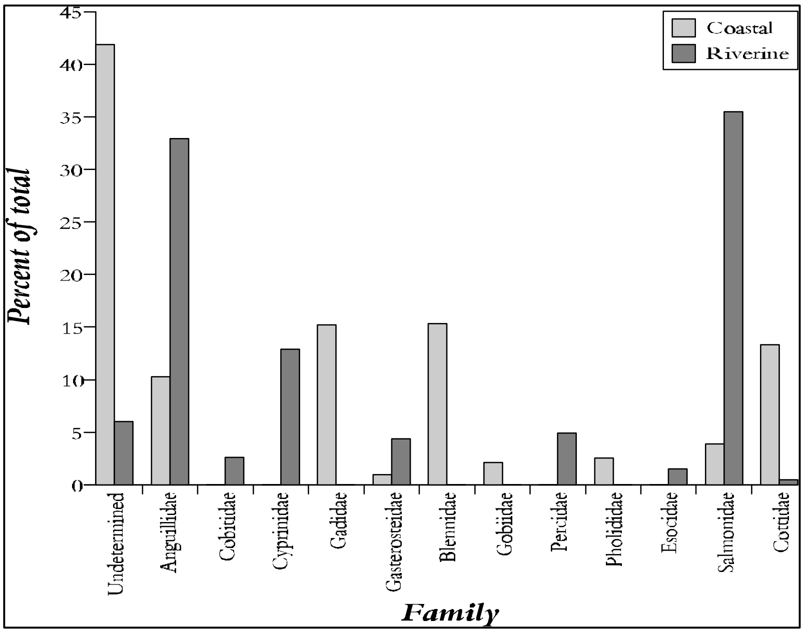
2.5.4. Why do fisheries managers seek to control mink populations?

Mink are perceived to have an impact on salmonid fisheries, trout farms, salmon farms, and commercial carp ponds (especially exotic koi carp). Concern is frequently expressed over the effect of mink on angling interests and particularly salmonid stocks (e.g. Lever, 1985).

In order to obtain a dispassionate opinion on the impact of mink on fish stocks, Birks (1990) carried out a questionnaire survey of the fisheries and conservation staff of the then newly formed National Rivers Authority. All ten NRA Regions responded, and nine of them stated that mink were not currently a threat to fish stocks (except perhaps in the artificial situation created by fish farms). Yorkshire Region felt that mink were a problem throughout the river catchments, and Southern Region expressed concern about the possible impact of mink on sea trout during periods of low flow, when this species lies in small pools in rivers and streams.

In its native range, fish are the second most important prey in the mink’s diet, after mammals. The Cyprinidae is the most diverse family of fish inhabiting fresh waters in North America, and are most frequently consumed by mink. In Europe, too, cyprinids are one of the mink’s main prey in fresh water ecosystems, particularly roach and chub. Salmonids, eels, stickleback and bullheads have also been reported as important fish prey for mink (Akande, 1972; Chanin & Linn, 1980; Gerell, 1967a; Skirnisson, 1980; Wise *et al.*, 1981). The representation of various fish families in the diet of British mink (as gleaned from a synthesis of 22 papers on mink diet; Macdonald & Strachan, 1998) is summarized in Figure 2-6. Mink generally take small fish: 97% of salmonids taken were < 25cm (Cuthbert, 1979).

Figure 2-6 Proportion of fish families in the diet of mink in Britain.



2.6. Conclusions

- Populations of foxes, deer, hares and mink are controlled by a number of interest groups for various, and often for several, reasons, summarised in Table 2-8. These reasons should be considered in the context of:
    - An often ambivalent attitude to the species and its control.
    - The complicated relationship between damage and abundance.
    - Inaccurate damage assessment.

Table 2-8 Why do particular interest groups seek to control populations of fox, hares, deer and mink?

		WHY DO THESE INTEREST GROUPS:				
		Farmers	Game Managers	Foresters	Fisheries Managers	Conservat-ionists
SEEK TO CONTROL THESE SPECIES?	Fox	Predate livestock	Predate game	Reactive good neighbour policy	NO CONTROL	Damage to wild birds
	Hares	Damage to crops (brown hare)	Compete with grouse (mtn. hare)	Damage to young trees	NO CONTROL	NO CONTROL
	Deer	Damage to crops	Management for stalking income	Damage to trees	NO CONTROL	Damage to plants and habitats
	Mink	Damage to poultry	Damage to game	NO CONTROL	Damage to fish	Predate water vole and wild birds

- Foxes are widely controlled because they are perceived to kill livestock (lambs, poultry and piglets), game (including hares) and other ground-nesting birds.
  - ♦ Of these different reasons, the best studied is wild gamebirds. Evidence is strong that fox predation has a significant impact on wild game populations, but less so for other ground-nesting birds.
  - ♦ Fox predation on livestock is usually low level, but widespread and sometimes locally significant.
  - ♦ Fox culling by farmers is done, not in reaction to a current problem, but as a preventative measure, and out of fear of what might happen if the population increased.
- Deer are routinely and widely controlled, in part because they cause damage by feeding on crops and young plantations, but also for amenity reasons, such as sport shooting and venison.
  - ♦ Most studies to date of deer damage to agriculture in England and Wales suggest that this is rarely of economic significance at regional or national level. Where significant damage does occur, it tends to be localised and highly variable even between fields within a farm.
  - ♦ Deer cause widespread and significant damage to forestry, but estimating the economic impact of damage is difficult, not least because trees (and cereals) may recover or even benefit from browsing
- Hares are locally controlled because they are perceived to feed on crops and young trees, but there are few data with which to assess the damage caused. Following widespread declines, brown hares are subject to a Species Action Plan to increase their numbers, while mountain hares receive special protection under international law.
- Mink are patchily controlled because of their predation on wildlife (particularly the water vole and nesting sea-birds), livestock (primarily poultry), and game. While there is ample evidence of their impact on wildlife, there are few data on their impacts elsewhere.
- The perceived significance to different interest groups of foxes, deer, hares and mink as pests are summarised in Table 2-9. The rankings are based on our subjective assessment of the literature and data

Table 2-9 How do particular interest groups rank the pest status of fox, hares, deer and mink? (1=highly significant pest; 5= not a pest)

		HOW DO THESE INTEREST GROUPS:					
		Livestock Farmers	Arable Farmers	Game Managers	Foresters	Fisheries Managers	Conservat-ionists
RANK THE PEST STATUS OF THESE SPECIES?	Fox	2	4	1	4	5	1-5
	Hares	5	4	5/4	4	5	5
	Deer	3	2	5	1	5	4
	Mink	3	5	3	5	3	1

3. What methods are used to control populations of foxes, deer, hares, and mink in England and Wales?

3.1. Introduction

Although a wide range of methods is potentially suited to controlling populations of foxes, deer, hares and mink, those used in the UK are restricted by legislation (see Appendix 2). Effectively, there are four legal methods in operation in 2000: shooting, hunting with dogs, trapping, and snaring. These are all lethal methods – that is, they aim to control the population by killing animals. In recent decades there has been an increasing drive to develop and implement non-lethal methods to achieve population control (Baker & Macdonald, 1999).

While most attempts to control populations involve deliberate culling, not all culling is done with the intention of controlling populations. Other aims include the harvest of natural products like meat and fur, and sport. Furthermore, different interest groups may have different aims for the same culling effort. Thus, culling may be instigated by farmers to control a population and the damage it causes; but culling may actually be carried out by a group of people who suffer no losses due to damage, but who find sufficient interest in the culling itself to spend their own time and money carrying it out.

Yet another layer of complexity is added by the fact that any one interest group will use a range of methods, but the emphasis varies between groups, even within the same area. There is also a degree of fluidity between methods, making strict definitions unrealistic and analysis difficult. For example, a single Welsh hunt might operate as a mounted hunt in lowland areas, a foot pack in open uplands, and as a gun pack in plantations, and in some or all of these situations might also use terriers, rifles or shotguns. Welsh packs, with their flexible *modus operandi* illustrate the difficulties in distinguishing clearly between methods that use dogs to kill, chase, locate or flush foxes (or indeed, other quarry).

In this chapter, our focus is population control through deliberate culling by man, for any reason. We include, therefore, sports that make no claim to control, such as hare coursing, as well as sports that do make a claim to control, such as mink and foxhunting. Our goal here is to examine which methods are used to cull each species, which interest groups use these methods, when and where. The numbers of animals killed by each method, and whether it effectively achieves management aims, is explored in Chapter 5.

There is an important distinction to be made between methods used to control a population, and methods used to control the *impact* of a population. In Chapter 2 we covered some of the reasons why people seek to control populations of foxes, deer, hares and mink. One major reason common to all these species is that they can cause damage to human interests, for example by eating crops or killing livestock. In these situations, it must be remembered that population control is not the only effective method, nor necessarily the most effective method, for reducing the impact of damage. Some of the alternatives to culling are explored at the end of this chapter (section 3.6).

3.2. What methods are used to control fox populations?

The methods used successfully to take adult foxes fall into two categories. Nighttime methods require either a lamp or image intensifier to make the fox visible to the operator, or else traps and snares that work in his absence. Daytime methods require the fox to be flushed out of cover. Additionally, the foxes' need for an underground earth to shelter cubs while they are very young creates a vulnerable period during which it is easier to locate both cubs and adults. Because adults have to provide solid food to their cubs, their activity may also extend further into twilight hours during spring and summer, creating more opportunities for shooting without spotlight or image intensifier.

Most fox culling is done without conscious selectivity, as it is difficult to distinguish males from females, or young from old, at any distance. From April to August cubs can be distinguished from adults, but this too becomes increasingly difficult as autumn approaches. Apart from obvious generalisations (e.g. a preponderance of females among adults shot at cubbing earths; a preponderance of young foxes among those caught in cage traps or shot after attracting with a squeak), few data are available to support the commonly held belief that different culling methods address different sectors of the fox population. Systematic exploration of this problem is difficult, because biologists too must use the same capture methods to study the population.

3.2.1. Data and approach

For fox culling in the context of game management, we used four sets of data held by The Game Conservancy Trust (GCT): the National Game-Bag Census; the Gamekeeper Fox Culling Methods Survey; the Fox Monitoring Scheme; and the Joint BASC/GCT Snares Trial. Our data on hunting with dogs come primarily from questionnaire surveys held by the Wildlife Conservation Research Unit (WildCRU), and from a questionnaire survey to all hunts organised by the Masters of Fox-Hounds Association (MFHA) and the Campaign for Hunting and analysed by the GCT. The most recent data available come from a January 2000 survey by Produce Studies Ltd (submission to the Inquiry). These data sets, and their limitations in terms of reliability and representativeness, are detailed below. Further data on the use of various methods were obtained by Reynolds & Heydon (2000a) from landowners and farmers in three large regions of Britain (mid-Wales, east Midlands, west Norfolk; see section 2.2.1.a.i), and by Baker & Macdonald (2000) in Wiltshire (see section 2.2.1.a.ii). Additional data were used are detailed in the text.

3.2.1.a. National Game-Bag Census (NGC), 1961-ongoing

This is a historical data base detailing ‘bag’ records for game species and predators for an annual sample of *c.* 500 shooting estates in all parts of the UK (Tapper, 1992). In general, predators (including foxes) were recorded only since 1961, although for a few estates records are available from much further back. Participation is voluntary, hence the sample of estates is self-selecting, and its geographical distribution varies with time as individual estates enter the scheme or drop out. No measure of culling effort is recorded. As a result of these peculiarities, great care is required to interpret the data correctly.

3.2.1.b. Gamekeeper Fox Culling Methods Survey, 1992-93

This survey was intended to fill a gap in our knowledge of the extent to which gamekeepers use different methods to achieve the ‘bags’ indicated by the NGC. The aim was to recruit a sample of *c.* 100 gamekeepers from around the UK to keep a daily record for 12 months of the effort and success of each culling method used. Recruitment effort was crudely stratified by region, in that regional advisors of Game Conservancy Ltd were asked to identify likely participants and estates that were either typical of the region or, conversely, unusual and of particular interest. Based on these recommendations, 83 estates were approached, and keepers on 47 of these agreed to take part. Head-keepers also involved a further 58 beat-keepers. After 12 months of recording, 65 record books were ultimately returned of which four were incorrectly filled out and were therefore unusable. The final sample therefore consisted of 61 gamekeepers from 36 shooting estates. Each participant was interviewed by telephone following return of the completed record books.

3.2.1.c. Fox Monitoring Scheme, 1994-ongoing

Because the above study suggested a way of monitoring relative fox numbers regionally and over a long time period, it was continued on a simplified basis in successive years. An annual sample of 40-60 self-selecting gamekeepers and amateur or professional fox controllers has contributed records for 6 years. As with the NGC, individual participants have come and gone. With such a small sample this means that the geographical origin of records has shifted with time, requiring very careful interpretation.

3.2.1.d. Joint BASC/GCT snares trial

This study was designed as an experimental comparison between a new type of neck snare and existing snaring practice. Sixty-four gamekeepers were recruited to record their snaring effort and success in a *pro forma* daily diary. Participants were not selected as a random sample, but rather as a sample of gamekeepers who already used snares to an appreciable extent in their work. Each snare location was described in detail, and the period for which the snare was set was recorded. At each daily inspection, any captures (foxes or non-target species) were recorded, also whether captives were alive or dead. The overall conclusion of this experiment was that the new variety of snare under trial performed no differently from those normally used by the participants. Consequently, data from these snare types may be combined and used as a single source of information on snaring.

3.2.1.e. WildCRU Questionnaire Surveys of MFHA Masters

The WildCRU hold data from a number of questionnaire surveys of farmers and MFHA hunt Masters. Data for farmers is detailed in section 2.2.1.a.ii.

In 1980, the Masters of all 206 MFHA hunts were circulated with a questionnaire soliciting information on the size of their hunts in terms of numbers and type of participants (Macondald & Johnson, 1996). They were also asked to supply the numbers of foxes killed and numbers of days hunting for the previous decade. A total of 80 Masters provided responses, and 60 yielded usable data on tallies. In the three years after this, Masters were asked to complete daily diaries recording the number of foxes moved, hunted and killed on each days hunting. Between 10 and 25 Masters provided data for each year. In 1994, these data were updated by circulating a further questionnaire, again via the MFHA, to hunt Masters. Of 60 circulated, 39 were returned (Macondald & Johnson, 1996). Foot-pack data were also obtained in 1994 (Macondald & Johnson, 1996), when the Welsh Farmers Fox Control Association supplied records for 18 of their (then) 28 member packs in 1993/1994 and 1994/1995 (at that time incomplete). They also recorded the number of lambs reported lost and the number of farms where losses were said to occur.

In 1995, Masters were interviewed from each of the thirteen recognised packs of foxhounds operating in Wiltshire (Baker & Macdonald, 2000). Face-to-face interviews lasting around 2 hours were conducted with Masters from each of the eight packs responsible for most of the hunting in Wiltshire, and the remainder were interviewed by telephone. Interviews provided information on hunt activity, and Masters from 7 of the 8 most active packs provided kill data from their diaries.

3.2.1.f. MFHA/Campaign for Hunting

The MFHA/CH data derive from questionnaires completed by 162 hunts, covering the 1990/91 to 1995/96 seasons. Data include the number of meets under early- and main-season rules, the number of foxes caught above and below ground, estimates of the percentage of foxes found that are killed, and the percentage of foxes found (moved) that are run to ground. For most hunts, the area of the hunt country, and patches of ground that were not hunted (‘no try’ areas) and where permission had been sought but denied (‘no go’ areas) were recorded. The percentage of foxes moved that were killed above ground, and the percentage of foxes run to ground that are dug out and killed by each hunt can be calculated from these data. Analysis was by The Game Conservancy Trust (Reynolds, unpublished).

3.2.1.g. Produce Studies Limited

In January 2000, Produce Studies Limited (PSL) sent a standard pre-coded self completion questionnaire to a Master or Chairman of all the 302 registered hunts for all species via their Masters’ Association (PSL submission to the Inquiry). Almost all (94%) responded. 178 of the replies related to foxhunting. The questionnaire covered five regions: South West including Avon, Gloucestershire, Wiltshire and Dorset; South, from Hampshire to Kent and Oxon to North London; Midlands & East Anglia to Norfolk, Lincolnshire, Nottinghamshire, Derby and Cheshire; North to the Scottish Border; Wales.

3.2.1.h. Are the data reliable and representative?

In none of the data sets could data be verified, and each may be prone to reporting and recording inaccuracies similar to those detected by Heydon & Reynolds (2000a). Inevitably, certain parameters are easier to measure than others; for example, the WildCRU survey (Macdonald & Johnson, 1996) asked Masters to note the numbers of foxes killed (distinguishing kills above and below ground), hunted (but not necessarily killed) and moved (but not necessarily hunted) – these measures are progressively more difficult to make accurately.

In the Culling Methods Survey and the Fox Monitoring Scheme (FMS), recruitment of volunteers aimed to minimise the risk of falsification. First, it was made clear that the task of recording data would be quite tedious; hence only those who were enthusiastic to help with research would take part. Second, absolute confidentiality was assured for each gamekeeper; data would not be revealed to his employer, his neighbours, the local hunt, etc., removing any temptation to falsify records to ‘keep up appearances’. Third, because of the involved nature of the *pro-forma* diaries, falsification would have been difficult to achieve convincingly. Fourth, because data were recorded daily, there was no risk of memory lapses as is the case with questionnaire studies referring to the previous twelve months. Finally, in the case of the FMS, a proportion of participants were willing to save body parts of foxes (lower jaw, uterus) for later analysis, verifying at least the number of foxes killed by these people.

None of the datasets could be gathered in a way that adequately took account of known regional variation in fox abundance, terrain and other environmental circumstances that might influence the effectiveness of different culling methods. Because of the ways in which participants were recruited, none of these surveys is representative of farmers, shooting estates or gamekeepers in the UK as a whole. In particular, despite obvious regional variation in circumstances, samples were not stratified by region. Only the NGC has a sample large enough to be broken down in this way. The WildCRU MFHA surveys were restricted to registered hunts, which may not be representative of other types of hunt, although this is partly remedied by the data from the Welsh Hill Farmers packs. The Fox Monitoring Scheme set out to involve people who would were interested enough to contribute data over a long time, and is biased towards operators who primarily use lamping to cull foxes.

3.2.2. Hunting with dogs

Foxes are primarily hunted with two very different types of dogs: large, fast hounds with enhanced trailing abilities that can pursue and catch a fox above ground; and very small terriers to locate and corner the fox that has gone to ground. Foxhunting mostly combines the use of the two breeds of dog to provide a unique daytime method for culling a nocturnal animal. However, some groups, particularly in hill country, use terriers without hounds; gun-packs use hounds without terriers to flush foxes from cover; and MFHA-registered hunts now do not dig out foxes (using terriers) unless specifically requested by the landowner. A few hunts – mostly fell packs – do no digging out on hunt days. Foxes are also hunted with lurchers and other ‘long-dogs’ on an *ad hoc* basis.

3.2.2.a. Mounted hunting with hounds

Mounted hunting with hounds is the most widespread form of fox hunting in England, and is most closely associated with the public perception of ‘foxhunting’. There are currently 184 foxhound packs registered with the MFHA, with 14,720 meets annually; five harrier packs also hunt the fox on horseback (MFHA submission to the Inquiry).

The structure and mode of action of each hunt is reasonably constant. The MFHA submission to the Inquiry gives a full account of this. In essence the huntsmen, their hounds and followers gather at the meet, and move off to a location where the huntsman has decided to searc for ('draw') a fox. The hounds then fan out, searching for fox scent. If a hound finds a fox, the hound barks ('speaks') and if disturbed but not instantly caught, the fox flees, and the chase commences.

Each registered hunt has an exclusive ‘country’ allotted by the MFHA, within which it negotiates permission to hunt from individual landowners. Historically, the purpose of the country was to avoid border disputes between neighbouring hunts – it implies nothing about rights of access.

3.2.2.a.i. Where, when and by whom are foxes controlled using mounted hunting with hounds?

Between them, hunt countries occupy a total of 145,000-164,000 km^2 (Macdonald & Johnson, 1996; PSL submission to the Inquiry) and in the south of England they are effectively contiguous. About 23% of England and Wales is not included within any registered hunt country (MFHA/CH data), although there may be hunting by unregistered packs there. Furthermore, within each hunt country there are ‘no-try’ areas where no attempt is made to hunt or to seek permission, usually because the land is unsuitable for hunting. On average, such no-try areas amount to 21% of the land included in hunt countries, and permission to hunt is sought but denied on a further 1-2% of allotted hunt countries (MFHA/CH data; PSL submission to the Inquiry). Excluding all of these un-hunted areas, the proportion of England and Wales actually hunted over is 61%.

Table 3-1 presents a regional breakdown of the structure of an average hunt country in 1981, in terms of their mean area, and the mean number of farmers and gamekeepers within them.

Table 3-1 Regional breakdown of the average structure of a hunt country in 1981.

	Wales and West	Midlands and East	South	North	Overall
Areas (km^2)	634.4	860.7	839.4	589.2	732
% of land owned by hunt participants	32.6	16.3	18.3	24.5	21.5
Number of farmers	488.5	642.9	442.1	321.5	488.9
Number of hunting farmers	54.4	65.3	34.3	30.0	46.3
Number of farmers banning hunt	4.7	6.7	5.1	2.6	4.9
Number discouraging hunt	0.4	1.7	2.5	1.7	1.7
Number of shooting estates	2.6	16.0	14.2	7.2	11.2
Number of Gamekeepers	4.5	24.3	19.0	12.2	16.9

The hunting season is usually September to March and can be split into two. In September, ‘cub hunting’ (now frequently referred to as ‘early season’ or ‘autumn’ hunting) takes place. This gives experience to young hounds and is said to promote dispersal (MFHA submission to the Inquiry). Between November and March, main season hunting occurs. Hunts meet 2-4 times a week, depending on the size of their country. The WildCRU data from 1981-1993 show that during an average season there are 71 hunting days. About two-thirds of these consist of main season hunting (44 days, on average), and a third consist of cub hunting (26 days, on average). There is wide variation within these averages, however, with full seasons lasting 33-125 days.

Clearly, mounted foxhunting affects about two-thirds of England and Wales, but how widely is it regarded as a population control method by farmers? One poll (PSL, 1995) reported that hunting occurred on 35% of Welsh farms and 26% of farms in the Midlands and East Anglia combined. Another found that the number of farmers allowing hunting was 63%, 61% and 56% for television network areas covering Wales, the Midlands and East Anglia respectively (NOP, 1996). These values compare with Heydon & Reynold’s (2000a) estimates of 41% of farms in Wales, 82% in the Midlands and 50% in East Anglia. In 1993/94 and 1994/95 respectively, 41% and 48% of farmers in Wiltshire reported that hunting had occurred on their land (Baker & Macdonald, 2000).

However, a farmer allowing hunting on his land does not necessarily see it as part of a strategy for fox control (Baker & Macdonald, 2000): only 31% of farmers in Wiltshire encouraged the hunt. Hunting was not allowed by 6% of Wiltshire farmers, leaving a further 63% who ‘tolerated’ or ‘discouraged’ it. The high proportion of tenant farmers, and the retention of sporting rights (Parkes & Thornley, 1994) by the Council may create this complex situation in Wiltshire. In 1995, the sporting rights on 88 (73%) of the 120 Council farms had been retained by Wiltshire County Farms Estate, and foxhunting was automatically permitted regardless of the farmer’s wishes. It is possible therefore that some Council tenants tolerated hunting on their land only because of their landlord’s policy. For the same reasons, however, tenant farmers without sporting rights would have had no motivation to control fox numbers for the purposes of game management. In Wiltshire, farmers reported that the hunt had visited their farm for an average of 1.8 days in each of the two hunting seasons. There was no statistical relationship between the number of kills made on a farm, and the farmer’s attitude to hunting[9].

In a questionnaire survey of National Gamekeepers Organisation members in 1997 just under half of 203 respondents (employed on shooting estates) cited hunting (48%) as one of the methods used on their ground to cull foxes (NGO submission to the Inquiry).

[9] Fishers Exact, P=1

3.2.2.b. Foot packs

In some (predominantly upland) areas, foxhunts operate as foot packs, with an October-March season. These function in a very different way to the mounted hunts, without horses, and making more use of shot-guns and terriers. There is considerable overlap between foot packs and gun packs (see below); in Wales, a foot pack might operate very much like a gun pack when in plantations (Federation of Welsh Packs, submission to the Inquiry). In general, foot packs use larger numbers of hounds than gun-packs. Some hunts also operate as both foot packs and mounted hunts. Some packs are also part of Fox Destruction Societies, which claim payment (currently £25 per day) for fox control from Forest Enterprise Wales (formerly the Forestry Commission).

In 1993/94, each Welsh foot-pack covered an average of 57.4km^2 (a considerably smaller area than a mounted hunt), and in total covered 2678km^2.

In Cumbria, there are 6 foot packs registered with the Central Committee of Fell Packs, and another three are affiliated (CCFP submission to the Inquiry). These make only limited use of shooting.

3.2.2.c. Digging with terriers

Digging out with terriers is widely practised by mounted and foot hunts and other communal fox control groups, as well as by small groups, or individuals such as gamekeepers. As with other methods involving dogs, terrier work has an enthusiastic following for its own intrinsic interest. When used by the hunt, terriers are usually entered into earths where hounds have marked a fox to ground. In other circumstances terriers are entered into holes where fieldsigns and local knowledge suggest an active earth, where tracks show that a fox has entered, or where there is evidence of a cubbing earth (section 3.2.3.c). Terriers can be entered speculatively into any earth, pile of straw bales, very thick cover, etc. to locate and either bolt, corner, or kill the fox. Field-craft skills are critical, as it is illegal to enter a terrier or dig in any place in regular use by badgers. This may be a very limiting condition in hill areas where rock piles are commonly used as shelter or cubbing earths, as neither species leaves much surface evidence in these situations.

Where foxes are dug out, a radio-transmitter collar on the terrier aids economical and rapid digging. Once exposed by digging, the fox must be dispatched humanely, for which a .22 rim-fire pistol or rifle firing a free round is recommended (Harris, 1985). In some cases the fox may be killed underground by the terrier – this is particularly likely with fox cubs. Foxes are usually prevented from bolting by lightly blocking tunnel entrances (e.g. with a spade), and those that attempt to bolt can be dispatched there. Sometimes nets are used to entangle a bolting fox.

The National Working Terrier Federation is an umbrella body for the major working terrier clubs, and has a code of conduct for pest control (NWTF submission to the Inquiry).

3.2.2.c.i. Where, when and by whom are foxes controlled using terriers?

The use of terriers lends itself best to communally organised fox culling efforts covering large areas of ground, particularly mounted hunting with hounds, or foot packs, which have been covered above. Foxes are not dug out by members of the MFHA unless this has been requested by the farmer on whose land the fox has run to ground. Known earths in the area to be hunted may be lightly blocked prior to the meet. Where this is practised, it will obviously influence the proportion of the cull taken by digging. There is also considerable variation in the extent to which hunts practice digging out with terriers: the PSL survey found that hunted foxes were most likely to be dug in Wales (where 53% of 2674 foxes were killed in this way) and least likely to be dug in the Midlands & East Anglia (25% of 2,519).

Although terriers are certainly used in the context of individual fox culling efforts (e.g. by professional gamekeepers), within a typical lowland beat of about 8km^2 (National Game-Bag Census, 1997 data), with fox breeding group densities of 0.1 to 0.4/km^2 and ongoing culling by other methods, each terrier would be entered to only a few (1-3) earths annually and would not build up much experience.

Nevertheless, 46% of 214 NGO gamekeepers use terriers (NGO submission to the Inquiry; ‘Terriers’ and ‘hunting’ were separate options under the question “How are foxes controlled on your beat?”). Among BASC gamekeeper members (BASC, 1994), 57% used terriers to cull foxes.

In Wiltshire, farmers did not report the use of terriers to control foxes outside the context of hunting with hounds (Baker & Macdonald, 2000).

3.2.2.d. Lurchers

The use of a spotlamp with a running dog (large lurcher or greyhound) rather than a rifle is popular in some areas of the UK. This tends to be practised as an unauthorised (and therefore illegal) sport rather than as legitimate fox control, and as a result no data are available.

3.2.3. Shooting

Foxes are shot in two main ways: at night with a spotlight and rifle ('lamping'); during the day by groups or individuals. They are also shot at the cubbing earth. Gun packs and shooting at earths may combine shooting with the use of dogs to find, bolt, or flush out foxes. Almost all gamekeepers use shooting to cull foxes (NGO submission to the Inquiry; BASC, 1994), mainly by lamping or by driving foxes to guns. Shooting is probably also widely used by other groups, such as farmers, on an *ad hoc* basis.

3.2.3.a. Spotlamp and rifle ('lamping')

In 'lamping' foxes are shot at night with a rifle (usually high powered centre-fire .22, .22/.250, or .243 calibre) with telescopic sight, in conjunction with a powerful spotlamp, usually from a 4WD vehicle. Lamping requires good vehicule access, an absence of cover, and terrain that allows safe shots. Red light reflected from the fox's retina can be detected from over a kilometre when there is no mist, although the fox must be close enough for its body shape to be distinguishable before a shot can be fired (Anon, 1998). Squeaking sounds often bring foxes running towards the lamp; this trick is most successful with young, naï ve foxes, allowing rapid culling by this method during autumn.

A spotlamp and rifle can also be used on foot or from a stationary high-seat, but away from cubbing earths this is extremely inefficient, because the likelihood of a fox passing in range within a reasonable space of time is very low. Most foxes shot from high seats are either at a cubbing earth, or are shot opportunistically during deer culling operations.

3.2.3.b. Gun-packs and standing guns with shotguns

These methods involve the use of a small pack of hounds, or a team of human beaters, to flush foxes out of cover towards a line of standing guns. This approach is most often used in dense woodland, especially commercial softwood plantations. The choice of hounds or human beaters varies regionally depending on availability, but hounds are clearly better in very dense cover and can also trail and catch wounded animals. Some foxes may be caught and killed by the hounds before they reach the guns.

For safety, and because the opportunity to shoot an emerging fox is usually brief, shotguns are almost invariably used. Gun-packs are communally organised, and are especially active in late winter/early spring.

In Wales, there are currently 30 gun-packs; while these operate largely on foot, one or two mounts may be used to keep up with the hounds (The Welsh Farmers Fox Control Society, submission to the Inquiry).

3.2.3.c. Culling at the cubbing earth

The cubbing earth provides a focal point within the territory where adults as well as cubs may be culled. Foxes culled at the cubbing earth must be either shot (with a rifle or shotgun), dug out or caught in nets (after sending down a terrier to locate and bolt or corner the fox), or trapped using cage traps set into the tunnel entrance (effective only for cubs older than 8 weeks; usually ineffective for adults).

Earths used for cubbing are difficult to recognise early in the spring, but become more obvious as evidence of occupation accumulates around them. Among gamekeepers, the aim will be to destroy resident breeding females as early as possible in the season and about 24% of breeding earths are located before cubs are active above ground. Correspondingly, 25% of vixens killed at the earth are killed before cubs can be culled or even counted, unless by the use of a terrier.

3.2.4. Snaring

Only neck snares are allowed under UK legislation. These are set on any route-way likely to be used by a fox, and will be successful only if they remain undetected. This is reasonably easy to achieve (Reynolds, 1998), making the snare a powerful tool against wary adults. The recommended way to dispatch a fox captured in a snare is to shoot it with a shotgun from close range (10-15 m) (Reynolds, 1997). At this range, choice of shot size is not important, and death is instantaneous.

A high proportion (81-86%) of gamekeepers use snares (NGO submission to the Inquiry; Reynolds unpubl.; BASC, 1994). Snares are the only culling method available where prolonged use cannot result in a fraction of the population being untrappable through selection against the unwary. One indication of this is that catch-per-effort for snares peaks in mid-winter when dispersal is at its height, rather than summer or autumn when the highest proportion of the population is naï ve. A further indication is that in ecological research on foxes, where animals captured in snares are tagged and released, recaptures and multiple recaptures are not uncommon (J.C.Reynolds, D.W. Macdonald, pers obs.)

Snares are unpopular in sheep-farming country during the lambing season due to the risk of lambs being caught. In upland regions, snares are most often set around a buried carcass bait, with a surrounding fence that keeps sheep out but allows free passage by foxes – this arrangement is known as a ‘midden’.

3.2.5. Trapping

In rural areas foxes are generally difficult to catch in live-capture traps (Harris, 1985; Macdonald, 1987; Reynolds, pers. obs.). Among professional gamekeepers, live-capture traps account for just 1% of all foxes taken. .

3.2.6. What strategies are adopted to attempt to control fox populations?

Consciously or not, farmers and landowners adopt various strategies to achieve their fox control aims. These strategies determine the combination of culling methods used, the timing of culling, and the amount of effort put into each method (and indeed, whether to cull at all). Because different methods are best suited to particular seasons, these aspects are closely inter-linked.

The choice of strategy will vary at a local level from one estate to another according to the needs and preferences of individual farmers. Independent culling efforts are most likely to involve night shooting, trapping, and snaring, often carried out by professional gamekeepers. In addition, there may be a regional strategy for fox control, involving the use of communally organised hunting with hounds and terriers, particularly in upland areas. These require the consent of several or many farmers, and are often supported by subscriptions from farmers, sometimes with formalised state backing. In Scotland, for example, there is limited financial support from the state for fox control groups; similar support was abandoned in England and Wales in 1979, but is continued in Wales by Forest Enterprise.

Other aspects of culling strategies will also change through time. For example, gamekeepers may have shifted away from snares in favour of night shooting with a rifle and spotlight (Reynolds & Tapper, 1994). As there are no historical data on the use of different methods, these are perceptions only, but it seems certain that during the last 30 years or so there has been a rise in popularity of lamping for fox culling. This has been helped by development of lighter and more powerful spotlamps, but also represents a move from the older tradition of managing wild gamebirds - requiring intensive predator control during spring and summer - towards an increasing reliance on hand-reared gamebirds (which are in rearing fields and therefore protected from foxes during spring and summer). Because lamping becomes less effective and snaring more effective as vegetative cover grows, the newer tradition favours lamping. The change of emphasis to autumn/winter lamping inevitably means that greater numbers of foxes will be culled to achieve the same aims.

3.2.6.a. How do local strategies translate into regional effects?

Although a regional impact on fox numbers is an aim for the majority of culling efforts, few people are in a position to organise a fox culling strategy over large geographical areas. Communally organised hunts and fox destruction clubs are better placed to do this than anyone else, but they do not have exclusive command over fox culling. Heydon & Reynolds (2000a) found that although organised groups operated on 88% of farms on which culling took place in Wales, the Midlands and Norfolk, 33% to 91% of these farms (depending on region and farm size) carried out additional fox culling independently. In some regions, therefore, the net impact on fox numbers is more the incidental result of local actions than the outcome of regional planning.

In their ‘Three-regions study’, Heydon & Reynolds (2000a) found fox control strategies used by farmers varied with a distinctly regional character. Communal methods such as hunting with hounds, gun-packs, and digging with terriers were practised on almost every farm in mid-Wales,

where fewer than 10% of land properties had a professional gamekeeper. Spring/summer culling was uniquely important here, and was reflected in high fox mortality during these two seasons. Gun-packs (involving hounds to drive foxes out of cover to standing guns) were used only in Wales. In Norfolk, the bulk of culling was carried out independently by professional gamekeepers on large estates, hence shooting with a rifle and spotlamp, and snaring, were the methods most commonly used.

3.2.6.b. How prevalent is fox control?

The prevalence of fox culling (which occurred on 88% of farms across mid-Wales, the Midlands and Norfolk) indicated by Heydon & Reynolds (2000a; see section 2.2.1.a.i for details) is much higher than that previously suggested. A 1981 survey reported that nationally only 33% of farmers replied “Yes” to the question “*Do you attempt to control foxes?*” (Macdonald, 1984; Macdonald & Johnson, 1996; Table 2-1). The difference between these studies may be explained by the form of question asked. Heydon & Reynolds asked, “*Is any fox culling carried out on your land?*” When they asked, “*Are foxes culled by anyone other than a hunt or gun-pack*”, 31% to 67% (depending on region) indicated that they took undertook such culling independently. In a more recent poll (Produce Studies Limited, 1995), 80% of farmers in Wales and 47% in an area covering the Midlands and East Anglia stated that fox control in some form occurred on their land. These values, however, included non-lethal control via electric fencing and assumed that where farmers did not consider foxes a problem there was no attempt to control them. The fact that some forms of control (particularly hunting) may occur on a farmer’s land without his encouragement (Baker & Macdonald, 2000) creates further difficulties in interpreting levels of active control.

3.3. What methods are used to control deer populations?

In contrast to the other species covered by this report (fox, mink, hare), only two groups of lethal methods (shooting and hunting with hounds) are in general use for controlling deer populations in England and Wales.

3.3.1. Hunting with hounds

The history of hunting deer with hounds stretches back at least as far as the Norman Conquest, when, Royal ‘Forests’ were created to provide and restrict sport and venison for the King. Such Forests, including amongst others The Forest of Exmoor, the New Forest, and Epping Forest, were widespread in England and Wales in Medieval times (Rackham, 1986), and some still exist today, stocked mainly with fallow deer. Although deer hunting itself is so ancient, the present-day style of hunting by riding to a pack of hounds is comparatively modern (Burton, 1969). On Exmoor, mounted hunting of red deer (staghunting) was revived around the middle of the 18th century, and, with some interruptions, it has now had a 200-year history. Staghunting on the Quantocks dates back to 1917, when red deer were reintroduced there, and the Quantock Staghound pack was reinstated with government support as a source of food supply and to revive this sport.

Today, organized hunting of red deer with hounds is practised in only a small area of southwest England (West Somerset and parts of North Devon), and only three hunts are registered with the Master of Deerhounds Association (MDHA). Some small, unregistered hunts, also mainly in the West Country, hunt roe deer with hounds. Until recently, fallow deer were hunted with hounds in the New Forest, but this ceased in 1997 following the Forestry Commission’s ban on deer hunting. While hunting with dogs is currently limited to red and roe deer in the West Country, there are no legal reasons why other species such as muntjac and sika could not be hunted in the same way.

Hunting red deer is referred to as 'staghunting', but both mature males (stags) and mature females (hinds) are hunted. The processes of stag and hind hunting vary slightly (MDHA submission to the Inquiry). Before a hunt, stags are selected by the 'harbourer' (an experienced local deer expert), who watches their movements at dawn. The chosen stag is separated from the herd by a group of experienced hounds ('tufters') before the remainder of the pack are introduced to the scent. Hinds are not harboured because individuals are almost indistinguishable. Instead, if hounds encounter a herd of hinds and calves, they will chivvy them until the herd fragments, and a single hind (with or without calf at heel) breaks away to be hunted alone. Once the stag or hind has been brought to bay it is killed by a shot at close quarters with a modified shotgun or humane killer.

3.3.1.a. Where, when and by whom are red deer controlled using hunting with hounds?

At present, three registered staghunts operate in Devon and Somerset (see also **Error! Reference source not found.**). According to a recent survey (PSL submission to the Inquiry), the three staghunt countries cover roughly 3,900km^2, of which 3% is not hunted over for safety or access reasons.

'Autumn stags' (usually 5-6 years or older) are hunted from the middle of August to the end of October. Thereafter hunting concentrates on hinds until the end of February, from which time 'Spring stags' (mainly 2-4 years old) are hunted until the end of April.

The Devon & Somerset Stag Hounds (DSSH) normally hunt three days a week (Tuesdays, Thursdays, and Saturdays) throughout the season, although occasional 'bye-days' may also be added on other days of the week. The other two hunts rarely hunt more than twice a week. As a result of the recent bans on deer hunting imposed by the National Trust and the Forestry Commission, which together hold very significant areas of land within the Quantock Hills, the Quantock Stag Hounds now commonly only hunt one day a week within their own area, with additional meets by invitation within the DSSH country.

Among landowners who responded to a questionnaire circulated by the Quantock Deer Management And Conservation Group (Langbein, 1998a; section 2.3.1.a), 37.5% used or permitted only hunting with dogs as a control method, and a further 27% used both hunting with dogs and shooting. Thus, 64% of farmers in this area use hunting with hounds to control red deer.

The support for hunting among farmers may be higher still: during 1998 over 90% of local farmers (100) with at least ten or more acres of land within the Quantock Staghunting country (including tenant farmers not necessarily always able to grant access for hunting) agreed to sign a letter in support of hunting by the Quantock Staghounds.

3.3.1.b. Hunting roe deer with hounds

While red and fallow deer have consistently been regarded as noble quarry of the *Forest* or *Chase* down the centuries, the status and hunting of roe deer has had a more chequered history (Whitehead, 1964). Following their demotion in Ancient Forest Laws in 1338 from 'beasts of the Forest' (preventing them and other animals such as red deer, fallow, and wild boar from being hunted other than by order of the King or his appointees), to 'beasts of the warren', roe deer gradually became exterminated throughout much of England and Wales by the seventeenth century. However, helped by some re-introductions and sustained re-afforestation schemes, roe have gradually re-colonised most English and many Welsh counties over the last 200 years. For example, in 1800 Lord Dorchester reintroduced the species in Dorset, and within fifteen years the first pack of roebuck hounds was formed to hunt them. Several packs of roebuck hounds hunted in Dorset during the 19th century (including the Blackmore Vale Hunt and Charborough Hunt) and seem to have continued in Dorset until the First World War (Whitehead, 1964). With their gradual spread further west, by 1900 roe also started to be hunted again occasionally in Somerset (e.g. by the Seavington Hounds near Chard).

Since then the practice of roe deer hunting has continued intermittently. Few published details are available, as most roebuck hunts are not registered, nor recognised officially by the Master of Deerhounds Association. However, it is known that two buckhound packs (Cheldon Buckhounds and Exe Valley Buckhounds) currently hunt roe deer in parts of the Devon and Somerset hunt countries used by the three packs of staghounds. The hounds used to hunt roe deer are usually either bassett/harrier crosses or beagles. Buckhounds commonly pursue their quarry followed only by a small core of mounted followers (huntsman, Master, whipper-in), with the remainder of followers on foot. The procedures for hunting roe are similar to those described for hind hunting above, with an average hunt resulting in a kill normally lasting around 1 to 1.5 hours (Exe Valley

Buckhounds, pers. comm.). At the end of the chase roe deer tend to lie down, rather than being brought to bay, and are then dispatched with a modified shotgun or humane killer.

Hunting of male roe deer usually takes place between the end of August and end of October, and also from April to early May, whereas females are hunted from October to February. Each of the two packs usually hunts once a week during the season, with a total of around 35 meets per pack. The main motivation for roe deer hunting is to provide sport with a total estimate of only 30-40 roe deer killed by hunting per annum by both the above packs combined.

3.3.2. Shooting

Shooting by stalking with a rifle or large bore shotgun is the most common method used to cull deer in England and Wales, as well as in Scotland and Northern Ireland (BASC and BDS submissions to the Inquiry). Shooting culls of deer, particularly as part of organised Deer Management Groups (DMGs – groups of adjoining landholders co-ordinating their deer management) are the method of deer control recommended by government (MAFF submission to the Inquiry).

3.3.2.a. Data and approach

Because there is, as yet, no single organisation in England and Wales which collates all culling data for deer, there are surprisingly few data available on the exact numbers of deer shot. The British Association for Shooting and Conservation's (BASC) 1996 survey on stalking "Deer, Deer Stalking and the Future" provides the most comprehensive account yet of deer stalking in this country, although results accrue necessarily from a self-selected sample of deer stalkers.

BASC sent the first questionnaire for the survey to a randomly selected 10,000 of their *c.* 120,000 members; 67% responded. This identified 13.4% of members who were active deer stalkers, and a further 22% interested but not active. A second questionnaire, to active stalkers only, produced 408 replies, providing a small but widely distributed sample (BASC, unpublished).

3.3.2.b. Shooting by rifle (deer stalking)

Shooting deer by rifle, generally known as deer stalking, is by far the most common and widespread of the lethal methods of legal deer population control (of any of the species) undertaken in England and Wales, as indeed it is in Scotland (Callender & McKenzie, 1991), and most other European countries (Deutscher Jagdschutz Verband, 1997). Rifle culls tend to be taken either from a high-seat (an elevated platform usually paced against a tree), or by stalking carefully up to the deer on foot at ground level to a safe shooting position. Many deerstalkers use a trained dog to 'point' to deer during stalking, and to locate fatally shot or injured deer.

Deer stalking may be undertaken for various complementary reasons, including population control to assist with crop protection, to obtain venison for own consumption or profit, or to provide income and sport through letting the deer stalking rights or offering accompanied stalking to shooting clients.

In the 1996 BASC deer stalking survey (see above), 24% of the total nationwide cull reported related to red deer, 49% to roe deer, 17% to fallow deer, 7% to muntjac, 3% to sika and <1% to Chinese water deer.

3.3.2.b.i. Where, when and by whom are red deer controlled using stalking?

Red deer are stalked with rifles throughout their range (see **Error! Reference source not found.**), particularly in Scotland, but also in England and, to a lesser extent, in Wales. The open season runs from August to April for stags, and November to February for hinds (see also Table 11-1).

Stalking is generally the main method of controlling red deer numbers across most private and public landholding types (forestry, moorland, farmland), including by Forestry Commission (Forest Enterprise), and on MOD land. In the Quantocks, shooting was used to control red deer by 50% of farmers who responded to a questionnaire circulated by the Quantock Deer Management And Conservation Group (Langbein, 1998a; section 2.3.1.a).

From their 1996 survey, BASC used a very conservative estimate of 9% (not 13.4%) to calculate that an estimated 10,000 of their members were active deer stalkers. Of these, 87.6% (8700) were 'recreational' stalkers and 12.4 % (1300) were 'professional' deer stalkers (rangers/ghillies etc). However, the professional stalkers accounted for 40% of the total deer cull. Extrapolated to the nationwide estimate of 180,000 deer shot by their members alone, these percentages suggest that around 72,000 deer are shot by professional rangers, and *c.* 108,000 by recreational stalkers.

3.3.2.c. Shooting by shotgun (for damage control)

Prior to the Deer Act of 1963, shooting with shotguns was a common method of deer control. In the West Country, any perceived surplus in deer numbers not taken by hunting was generally accounted for by means of organised shotgun drives (HMSO, 1951; Bonham-Carter, 1991). However, the Deer Act 1963, and most recently, the Deer Act 1991, prohibited the use of any smooth bore gun (shotgun) to kill deer for the purpose of general deer management, although they can be used for damage prevention or to kill an injured animal (Appendix 2). No separate data are available for the proportion of the cull currently taken in this manner.

3.3.3. What strategies are adopted to attempt to control deer?

Traditional approaches to deer control involve attempting to reduce population sizes. While this is often an appropriate response, it is associated with various problems. In particular, many deer populations range widely, over several estates; this means neighbouring landowners must coordinate their management efforts to achieve effective control. However, even effective control of deer numbers will not necessarily deliver an equivalent reduction in impact, since there is no simple relationship between damage and deer density (section 2.3.2.b.i).

In practice, effective control will generally only be achieved via integrated management involving both direct management of the deer population and independent control of its impact (Ratcliffe, 1998; Chapman & Harris, 1998; Putman & Langbein, 1999, 2000). Non-lethal methods (section 3.6) form a useful tool for management of deer and control of damage in an integrated approach. Strategies to manage local distributions and to reduce the significance of damage include permanent and temporary wire fencing, tree guards and shelters, chemical repellents and habitat manipulation.

In a recent large-scale questionnaire survey conducted by ADAS in lowland England, the majority of respondents with farming or forestry landholdings perceived organised culling via a Deer Management Group (DMG) as being the most effective way of preventing deer damage (Packer *et al*., 1998). Encouragement and formation of DMGs is the main aim of the Deer Initiative, a Forestry Commission-led partnership of a wide range of countryside and welfare organizations.

3.4. What methods are used to control hare populations?

Hares are only a minor pest, and are usually culled for sport and sold to game dealers (Harris & McLaren, 1998). Where they do need to be controlled, organised hare shoots are generally used (Stoate & Tapper, 1993)

3.4.1. Hunting with dogs

3.4.1.a. Hunting with hounds

Hunting hares with beagles appears to have been well developed by Tudor times and was popular in the 18th century (Stuttard, 1981). Today, hunting hares with hounds takes place on foot with packs of beagles or bassets, or on horseback with harriers. In England and Wales 102 packs of hounds (of which 72 are beagles, 10 bassets and 20 harriers) hunt hares and are registered with the Association of Masters of Harriers and Beagles (AMHB) or the Masters of Basset Hounds Association (MBHA) (AMHB and MBHA joint submission to the Inquiry).

The AMBH (submission to the Inquiry) estimate that hunts are normally restricted to 1-2 square miles and last 30-90 minutes. Typically, the hunt consists of a number of short chases and checks, as the hare evades and then is flushed by the hounds. The huntsmen draw a nearby field with a pack of beagles until a hare is put up or its scent is found. This is then pursued until the line is lost and the hare escapes, or the animal is caught and killed by the hounds.

Only one beagle pack (the Holme Valley Beagles) take mountain hares in England.

3.4.1.a.i. Where, when and by whom are brown hares hunted with hounds?

As with foxhunting, each hare hunt has its own country, registered with the AMHB, and also used by the MBHA. These average 1,300km^2 (PSL submission to the Inquiry), and cover an estimated 90% of rural England and Wales. In 1989 and 1990, the Game Conservancy Trust asked one hunt Master to map where his hounds ran on each of 14 days of hunting. The average of the 14 days was roughly three square kilometres (300ha).

Most beagle and harrier packs hunt twice a week (AMHB and MBHA joint submission to the Inquiry). The season extends from after harvest begins in late August or early September and extends until the end of March. AMBH rules determine that there is no hare hunting after March, though there is no statutory requirement on this.

3.4.1.b. Hare Coursing

Hare coursing as a sport was well established in Roman times in Gaul (Stuttard, 1981). Today, two types of legal hare coursing operate: competitive (‘two-handed’) coursing, and ‘single-handed’ coursing. Coursing competitions, usually with greyhounds, but also with other breeds including deerhounds or salukis, are organised events carried out by coursing clubs, which take place on land with permission of the land-owner.

Under rules laid down by the National Coursing Club (Stable & Stuttard, 1971; NCC submission to the Inquiry), coursing takes place on open ground to allow the hare to escape. Competition coursing is not considered a control method, and the aim is not to kill the hare (NCC submission to the Inquiry), although the hare will be killed if it is caught by the dogs. Hares are driven onto the running ground by beaters (‘driven coursing’), or are put up by participants and their dogs (‘walked-up coursing’). Dogs are released once the hare is sufficiently far away (about 80 yards). Competitions are usually knock-out competitions between pairs of dogs; the winning dog is chosen by a judge on horseback who awards points on the basis of how well they perform in relation to each other and the hare. Coursing competitions using deerhounds follow most NCC rules with additional Deerhound Coursing Club rules (DCC submission to the Inquiry). Competition coursing with lurchers follows NCC rules “as far as is applicable” (Association of Lurcher Clubs [ALC], submission to the Inquiry).

‘Single-handed’ coursing is usually carried out with a single lurcher, and the object is to catch and kill the hare. This form of coursing is not formally recognised as a field sport by the Countryside Alliance (ALC submission to the Inquiry), and does not abide by NCC rules. As with any other form of hunting, this type of coursing is illegal only when carried out without the landowner’s permission.

3.4.1.b.i. Where, when and by whom are hares coursed?

There are 224 greyhound coursing clubs affiliated with the NCC. The ALC estimates the total number of lurcher owners to be 112,422. Of 328 lurcher owners who responded to a questionnaire survey by the ALC, 52% coursed. There are no data on extent to which farmers and other landowners consider coursing to be a useful form of pest control, but we note that *“In many cases the courser pays* [in cash or in kind] *for the right to course on the farm”* (ALC submission to the Inquiry).

On average, between the 1990/91 and 1998/99 seasons, there were 1934 competition courses held over 93 days annually (data taken from the NCC submission to the Inquiry). The number of coursing events run on an estate ranges up to 112 (Altcar) and these generally take place on arable farmland with a gamekeeper.

National Coursing Club rules do not allow coursing between March 11th and September 14th; the ALC regulations also stipulate a closed season.

3.4.2. Shooting

Hare shooting is the most significant form of hare culling in the countryside and the method most frequently adopted by farmers in arable areas as a means of pest control (Tapper, 1987).

Hare shoots (using shotguns) are normally undertaken over the whole farm area, and smaller farms sometimes combine to take in a wider area of ground. The day is organised as a series of drives, and unlike game bird shoots, there is no separation between beaters and guns. Most participants (usually 20-40 people) carry guns and will be involved in either standing or walking lines during the course of the day. Separate areas of the farm are surrounded and the drives move inwards making a tighter area. Hares either are shot as they flush running forward, or are taken as they break out through one of the lines of guns.

February is the most common month for organised hare shoots, being after the end of the game bird shooting season, and at a time when most hares are feeding on winter cereals (Tapper & Barnes, 1986). Hares shot on hare shoots are treated as game, and provided the shoot is undertaken before March 1st it is usually sold to a game dealer who markets the hares in Britain or in Europe.

An unknown number of hares are also shot with rifles by or at the instigation of farmers, to reduce numbers and avoid the problems associated with illegal hare coursing. Although there are no data on this, contact with landowners and farmers suggests that this is quite a common strategy (Pye-Smith, 1998; S. Tapper, pers.comm.). It is extremely easy for hare numbers to be substantially reduced by steady attrition over a period of weeks or months, using a .22 rifle.

In 1997 an estimated 200,000-300,000 brown hares and 40,000-100,000 mountain hares were shot in the UK (Cobham Resource Consultants, 1997). The number of mountain hares known to be shot in England and Wales has been typically no more than 50 individuals annually during the 1990s (National Game-Bag Census, The Game Conservancy Trust, unpubl.), all on a handful of shooting estates in the north of England.

3.4.3. Other methods

The smaller mountain hare is a quarry for falconers using the larger raptors such as the golden eagle. Hares have also traditionally been snared and netted. All of these methods remain legal today.

3.5. What methods are used to control mink populations?

Mink are controlled using three methods: hunting with hounds on foot, trapping (either break-neck or live-trapping followed by shooting), and shooting.

During the 1960s, the Ministry of Agriculture, Fisheries and Food trapped over 5,000 mink in England and Wales, with a similar effort in Scotland by the Department of Agriculture and Forestry. By the mid-1970s this effort was emerging as futile and was abandoned (Birks, 1986). Subsequently, highly intensive mink control has been implemented patchily, where fishing interests prevail. Otherwise, attempted mink control has been haphazard or none-existent for the last two decades.

3.5.1. Hunting with hounds

Since the otter was legally protected from hunting in Britain, at least six otterhound packs began hunting mink instead. Several new packs hunting mink have also been formed, and there are currently 20 packs registered with the Master of Minkhounds Association (MMHA submission to the Inquiry). There are also some unregistered packs.

The hunt operate on foot, and drag rivers and streams across several farms where permission has been granted or in response to a request from fish farmers and others (MMHA submission to the Inquiry). Dens are located by dogs as they search the riverbank, and attempt is made to bolt their quarry. The dogs may catch the animal as it is flushed into the open or it may be treed and then shot or dislodged to fall to the hounds. As with foxhunts, minkhunts use terriers to kill mink which have gone to ground.

In January 2000, a Produce Studies Ltd survey (PSL submission to the Inquiry; section 3.2.1.g) estimated that mink countries covered 76,000km^2, 61% (46,000km^2) of which was actually hunted over. The hunting period runs from April to September when water level is low and the mink are raising young.

In 1996, Strachan (unpublished data) sent a letter with simple questions concerning mink to 40 landowners along 24km of the River Thames in west Oxfordshire. Of 32 respondents, 6 (19%) said they would use the mink hounds to control mink.

3.5.2. Shooting

Over the period 1995-98, Strachan *et al.* (unpublished) carried out interviews with farmers, gamekeepers and lock-keepers in the Thames Valley regarding their mink control practices. Shooting by farmers and gamekeepers was *ad-hoc* and largely incidental to other control activity (e.g. while shooting rabbits, foxes or pigeons). The majority of mink were killed during the winter months or in the autumn when juveniles were dispersing.

3.5.3. Trapping

Mink are either caught and killed in spring (break-neck) traps, or are caught alive in cage traps, and then killed. Spring traps must be approved to catch mink in the Spring Trap Approval Order 1995 (i.e. Fenn Mk 6, Springer No 6, Kania 2000, BMI Magnum 116). They risk killing non-target species (in a riparian habitat these are chiefly weasels, stoats, polecats, water voles, rats, and moorhens, but also possibly young otters).

Cage traps are best made of 14-gauge weldmesh, and should be wrapped in hay and baited (e.g. with sardine or day old chick carcasses) to provide food and shelter, and to increase their attraction to the mink. Traps must be checked daily, ideally first thing in the morning. Live-caught mink must be killed, as it is illegal to release them back into the wild. For humane killing, a 0.22 rimfire rifle, or 9mm and 0.410 shot-pistol or shotgun is recommended, but in practice it is not always easy to shoot a small, moving, target within a cage, and some practitioners simply drown the animals; this is particularly cruel (section 6.4.2.a).

Trapping is the main method of mink control in Britain, and is widely recommended (Advisory Service of the Game Conservancy, 1994; Strachan, 1998; MAFF, 1998). As a specific conservation tool at water vole sites, mink control by cage trapping is recommended in the months of January, February, March, and April (around the time of the rut but before birth). This will allow for the efficient removal of all resident female mink (Macdonald & Strachan, 1999).

Gamekeepers, water bailiffs, lock keepers, angling clubs, farmers, nature reserve managers, and private landowners may all trap mink, but there are few data on the extent to which it is used. In a 1996 survey of farmers along 24km of the River Thames, just over half (56.3% of 32 replies) said they took action against mink. Of these 18 farmers, 6 (19% of all those surveyed) used trapping (Strachan, unpublished data). In 1995-1996, 21 of the 47 Lock keepers on the River Thames upstream of Teddington, trapped for mink (Strachan, unpublished data). Trapping effort was very variable, from none (even where mink were known to be present) to up to 10 traps on one Lock island. Trapping was generally in response to the presence of mink.

3.6. What are the alternatives to culling?

In recent years, ethical and conservation concerns over culling have led to increasing interest in non-lethal methods of population and damage control. Non-lethal methods of control fall into three categories:

- Methods that seek to reduce populations, such as fertility control (Tuyttens & Macdonald, 1998a,b).
- Methods that alter behaviour, such as conditioned taste aversion, repellents or diversionary feeding (e.g. Durdin, 1992; Garrott, 1995; Moore, 1998; Pepper, 1978; Putman, 1998).
- Methods that alter the environment to reduce encounters with human interests, such as fencing or stock management (e.g.Haddon & Knight, 1983; Balharry & Macdonald, 2000; Minsky, 1980; McKillop & Sibly, 1988; Pepper *et al*., 1985, 1992).

Environmental alteration techniques are by far the most widely and reliably used; behavioural alteration and fertility control methods are still largely experimental, although they have been used effectively in some practical applications. The nature of the species, the damage it causes, and management aims, define the type of control needed. An important aspect of behavioural and environmental alteration approaches is to isolate the aim of damage reduction from any other aims, such as sport or meat.

Non-lethal methods of damage and population control could prove more effective than lethal control, because non-lethal methods aim to minimize perturbation, for example by retaining the predator in its original territory (Tuyttens & Macdonald, 2000). This avoids the density dependent population responses and immigration which can result from culling (section 1.3.2.a), whilst allowing the animal to continue with whatever effect it has on limiting other prey numbers or excluding conspecifics.

3.6.1.a. Non-lethal methods of fox control

The most widely used non-lethal method of fox control is exclusion using physical barriers. Hand-rearing of game-birds is a management technique which protects them from foxes at a vulnerable stage, but also creates other problems. Two further non-lethal approaches have been widely discussed: manipulating foxes' food preferences (conditioned taste aversion or CTA) and fertility control.

3.6.1.a.i. Physical barriers

Physical barriers such as wire netting are valuable for preventing loss of poultry, game-birds or livestock held in small areas, but are not practicable to protect them on any wider scale. Electric fencing has been used with partial success to protect wild ground-nesting birds on nature reserves, but usually needs to be backed up with lethal methods. It is most useful for colonial nesting species whose nests are concentrated in small areas (e.g. terns), or to a very few individual nests (e.g. stone curlews), and for a limited time (Minsky, 1980; Haddon & Knight, 1983; Patterson, 1977).

3.6.1.a.ii. Hand-rearing of game birds

Hand-rearing of gamebirds is itself a way of avoiding predation (and other causes of mortality) at a vulnerable life-history stage. However, it does create other problems, including:

- Concentrations of birds on the rearing field and in release pens. Both must be securely fenced against incursions by foxes, but any breach can be catastrophic (Reynolds & Tapper, 1995).
- Increased vulnerability to fox predation following release as poults (Krauss *et al*., 1987; Brittas *et al*., 1992; Putaala *et al*., 1997). Reared birds that survive do become 'street-wise' about predators, and after their first adult year in the wild are functionally indistinguishable from wild-reared birds (Hill & Robertson, 1988).
- Unexplained non-genetic vulnerability to fox predation in released birds during nesting in their first year after release, compared with wild-reared birds or older released birds (Hill & Robertson, 1988; Leif, 1994).
- High populations of birds made possible by hand-rearing and releasing are a valuable food resource for foxes (Reynolds & Tapper, 1996). In the absence of effective fox control these may attract substantial predation, and allow higher fox density, higher cub production and improved cub survival.

3.6.1.a.iii.Conditioned Taste Aversion (CTA)

Conditioned taste aversions are a well-researched component of vertebrate behaviour through which an animal's innate food preferences are modified as a result of experience (Reynolds, 1999). Characteristically, a single bad experience of a food that is mildly damaging (e.g. food poisoned by harmful bacteria or laced with an emetic) leads to a robust and lasting aversion towards any food that is similar in taste. In captive or laboratory animals, the power of CTA to cause lasting avoidance of certain food types, even in the absence of alternatives, is very impressive.

Since the early 1970s wildlife biologists have speculated that CTA might be exploited to curtail unwanted predation. In the UK, it has been envisaged that individual foxes might be made averse to game birds - for example - by feeding them dead game birds that have been laced with a mildly toxic chemical (Reynolds, 1999; Cowan *et al*., 2000). Once 'educated' to avoid the chosen prey, a fox would become a valued resource, because it would keep out 'uneducated' foxes through territorial behaviour. A single, loaded, bait would be sufficient to 'teach' an individual that the referent food type (e.g. game birds) was bad to eat. Thus a short baiting programme at the appropriate time of year might be all that was required to achieve management aims.

CTA has recently been the subject of recent intensive field research by both The Game Conservancy Trust and Central Science Laboratory (MAFF). The WildCRU at Oxford University have explored a related topic - the creation of food-aversions using a bitter-tasting substance (Baker & Macdonald, 1999). It is now clear that deployment difficulties and non-target hazards make CTA non-viable for fox management in the UK (although the WildCRU and CSL are continuing related research on repellents). The major problems identified are:

- Difficulty in delivering aversive baits to all individual foxes present (some individuals simply never take bait);
- The lack of any effective aversive chemical that is acceptable on safety and welfare grounds for field application;
- The involvement of non-target species (particularly badgers) as potential consumers of aversive baits;
- Sub-division of baits such that accurate dosing becomes impossible;
- Accurate dosing given the range of body sizes addressed;
- Incompatibility with lethal control methods and other high mortality, which removes ‘educated’ individuals.

3.6.1.a.iv. Fertility Control

The concept of controlling wildlife populations by reducing productivity rather than by increasing mortality has been around since the 1970s. For canids, chemosterilisation has been attempted in the USA using a number of chemicals (Balser, 1964; Asa, 1997). Hormonal control has also been trialled, both in captivity and in the field (Linhardt & Enders, 1964; Linhart *et al.*, 1968; Oleyar & McGinnes, 1974; Allen, 1982). However, because reproductive biology is so similar in all mammals, both kinds of fertility control carry risks for non-target species, including humans. The safest approach to fertility control for wild animals is by exploiting the body’s immune system (immuno-contraception) to create antibodies to sperm-coat or ovum cell wall proteins (Tyndale-Biscoe, 1994).

Immuno-contraception for foxes has been the subject of intensive research over several years by Australian and French government scientists. In Australia, the chief concern is to drastically reduce or eliminate the (introduced) red fox. In Europe, the interest is primarily to ensure that it remains possible to control rabies outbreaks through oral vaccination, despite increasing densities of foxes (Artois, 1997). Compared with Australia, a much more modest impact on fox numbers is sought to ensure efficacy of rabies vaccination.

Immuno-contraception was expected to be long-lasting, cheap, safe, reversible, more species-specific and more humane than traditional methods for reducing fox populations (Kirkpatrick & Turner, 1991). Initial hopes that contraceptives could be administered using modified viruses as vectors have largely been displaced by the less ambitious route of oral administration through baits. Despite enormous expenditure in America, France and Australia, many other practical problems remain to be resolved before a workable methodology is available. One view of this is that the probability of success and the environmental risks linked to species-specificity mean the money required for development would be better used in traditional methods of population control (Artois, 1997; Artois, pers.comm.). Optimism remains high within research teams (Boue, pers.comm.), though the time-scale to development of a vaccine for foxes in captivity is at best 5 years. One can speculate that development and testing for environmental application would take at least 10 years from now (2000).

3.6.1.b. Non-lethal methods of deer control

Non-lethal measures are more widely used against deer than any other species covered in this report. Non-lethal measures to limit the impact of deer on agriculture, forestry or conservation areas include a variety of fencing, individual tree guards or shelters, chemical repellents, species choice and habitat manipulation aimed at altering the behaviour and feeding patterns of the deer. In addition, fertility control is being developed to control (mainly enclosed) populations.

3.6.1.b.i. Physical barriers

Traditional wire mesh fencing is probably the most widely used form of protection against deer damage. In the planting year 1991/92 the Forestry Commission alone spent over £6 million on fencing, primarily to protect against deer damage (Gill, 1997). The Forestry Authority (Pepper, 1992) recommends the minimum fence heights between 1.5m (for roe and muntjac) and 2m (for red deer).

Full height (1.8-2.4m) high tensile fencing is also used extensively along many motorways in Britain and elsewhere, as it is also the most reliable method of reducing numbers of road-traffic accidents related to deer (e.g. Ward, 1982; SGS Environment, 1997). Electric fencing, can provide temporary protection to areas vulnerable only for a short period of time, but is unreliable and often ineffective (Pepper *et al.*, 1992), and requires maintenance.

Where the protection of new plantings of small numbers of trees is required, individual trees may be protected with guards or shelters (Pepper *et al*., 1995). A wide variety of such shelters are now commercially available. Plastic guards can provide protection from roe deer, but welded wire mesh guards are most effective for both broadleaves and conifers, particularly against larger species. Tree growth shelters (opaque plastic tubes), are widely used to establish broadleaved trees and shrubs and have the added benefit of providing a suitable microclimate for growing trees. As with fencing, the height and specification of the tree guards or shelter is important. The cost of individual tree protection tends to be high (currently *c.* £1.50-2.50 per tree).

3.6.1.b.ii. Species choice

Where new trees are to be planted the choice of species will depend on the particular amenity, sporting, or commercial objectives for the forest, and will be limited by site factors (soil type, climate, exposure) as well as by the anticipated future economic needs for timber. Some scope may nevertheless remain for the forester to chose those species least susceptible to wildlife damage.

As a general guide, willows, aspen, and silver fir are highly preferred by deer for browsing, and Norway spruce, lodgepole pine and ash are particularly susceptible to bark stripping. Sitka spruce, Scots pine and Corsican pine are less vulnerable (see reviews by Gill, 1992a,b). Damage even to low ranking species may nevertheless be severe where more highly preferred species are not available, and hence provision of some alternative browse alongside the main crop may be useful (e.g. Langbein, 1993). Damage to coppice woodlands across England shows clear differences in vulnerability to damage between coppice species, depending on the deer species present (Putman, 1994). Birch regrowth was browsed heavily by roe and red deer, but left virtually untouched by fallow, the latter causing more damage among chestnut and ash. Alder and maple seem least palatable to deer in general, while willow is universally preferred, again suggesting some scope by focussing attempts at coppicing at the least vulnerable species given local conditions.

3.6.1.b.iii.Chemical repellents

Chemical repellents may be applied to reduce deer damage in two main ways: as barrier repellents, leaving an ‘olfactory fence’ which animals will not willingly cross; or as feeding repellents, applied to individual vulnerable plants which, by scent or taste, repel or inhibit feeding. Only one barrier repellent (Renardine) is currently approved for use under the Pesticides Registration scheme, but it appears to be ineffective against deer. The Forestry Authority has tested the efficacy of over 65 chemicals or proprietary compounds sold for application to individual plants (trees), but results have generally been disappointing (Pepper, 1978; Pepper pers. comm*).* While various products still under development may have some potential to provide protection against deer damage, a wide variety of other folk-lore treatments (e.g. lion dung or human hair) proved to have no repellent properties. Only one, Aaprotect® has so far been found to give consistently good results, but even this offers protection for conifers for one winter only. A fundamental limitation with most repellents is that new foliage remains unprotected unless the repellent is re-applied after every growing season (Gill, 1997).

3.6.1.b.iv. Behavioural and habitat manipulations

The impact of deer may be manipulated by changing their patterns of habitat use or foraging behaviour, so that they cause less damage in vulnerable areas. Damage sustained by any given agricultural area or woodland block will be a function both of the amount of usage that area sustains (in terms of ‘deer-hours’ spent within it) and the proportion of that time which is spent actually feeding on vulnerable crops.

Both the attractiveness of an area for deer, and the probability that they will feed upon vulnerable species, can be manipulated by appropriate changes in management. Valuable crops may be protected from damage by intercropping them with more palatable ‘sacrifice crops’ (Petley-Jones, 1995; Putman, 1998), or animals may be drawn away from sensitive areas by increasing attractiveness of alternative sites through increasing forage quality or cover (Langbein, 1997). Conversely, habitat manipulations may also be employed to *reduce* the attractiveness of vulnerable areas for deer (e.g. removal of nearby cover). For example, changes in coupe size may affect the amount of usage of regenerating coppice or plantations, with large open areas often being less attractive to deer (Kay, 1993; Putman, 1994).

While such measures are very likely to have the potential to alleviate damage, especially if used in combination with other forms of damage control, their actual effectiveness has not been studied in detail (Putman, 1998).

3.6.1.b.v. Non-lethal population control

In the United States, fertility control trials have been undertaken on enclosed or island populations of various ungulates, including deer (white-tailed deer: Kirkpartick *et al.*, 1997, Turner *et al.*, 1992; Garrot, 1995; fallow deer: Fraker, pers comm), with some degree of success. However, numerous potential behavioural and physiological side-effects have yet to be fully investigated, and it is unlikely at this stage that such methods would ever become practical and cost-effective for general use on free-ranging deer populations. As with foxes, a major problem is simply the large numbers of animals which must be treated to achieve any significant reduction in recruitment rates (Garrot, 1996; Putman, 1997; Moore, 1998).

Although strictly speaking not a non-lethal method, a ‘natural’ method of population control might in theory be achieved through reintroduction of large predators of deer. Re-introduction of wolves and lynx to Britain is likely to be problematic (Macdonald *et al.*, 2000), particularly in view of the vast population of free-ranging sheep likely to suffer predation (Gill, 1997). However, the gradual re-establishment and spread of wolves in Spain, France, Norway and Sweden illustrates that wolves can co-exist with modern agriculture in parts of Europe (Boitani, 1998), although not without problems.

3.7. Conclusions

- At present, methods to control populations of foxes, deer, hares and mink in the UK all involve culling: shooting, hunting with dogs, trapping, and snaring. Table 3-2 summarises the methods used for each species; only a few available methods are not used to any significant degree (e.g. snares for mink, live-capture traps for foxes; hunting with dogs for fallow and roe deer).
    - For foxes, dogs, snares, rifles, and shotguns are combined in various ways to create a range of culling methods suited to different situations. The prevalence of these different methods varies substantially between regions in response to local conditions, land-use and traditions.
    - For deer, hunting with dogs is confined to parts of Somerset and Devon; otherwise shooting with a rifle is the predominant method of population control.
    - For hares, population control is usually achieved through organised driven shoots; hunting and coursing make no claim to act as control methods, although illegal, unplanned coursing may locally suppress hare numbers.
    - For mink, trapping (with either killing or live-capture traps) is the predominant method; hunting with hounds is also widespread.
- Methods available to control populations of foxes, mink, deer and hares are restricted by a number of statutes designed to satisfy the interests of individuals (game rights, crop, livestock and game protection), while also addressing general issues such as environmental and human safety, humaneness and conservation (Appendix 2).
- Most operators use a combination of lethal techniques to control populations and non-lethal techniques (Table 3-2) to control damage. Physical exclusion by wire netting is widely used to protect vulnerable crops, livestock or reared game against deer, foxes and mink. More sophisticated approaches have either proved unpromising (conditioned taste aversion) or are still far from realisation (fertility control).
- Assessing the extent to which different methods are used is made difficult by the disparate nature of available data. In particular, it can be difficult to clearly distinguish between control methods, as they

might combine the use of several techniques, with the emphasis changing from place to place, day to day and person to person.

- For example, a single Welsh hunt might operate as a mounted hunt in lowland areas, a foot pack in open uplands, and as a gun pack in plantations, and in some or all of these situations might also use terriers, rifles or shotguns.
- Welsh packs, with their flexible *modus operandi* illustrate the difficulties in distinguishing clearly between methods that use dogs to kill, chase, locate or flush foxes (or indeed, other quarry).

Table 3-2 Legal methods to control populations and damage. ✓ indicates the method is both lawful and commonly used in the UK. Some of the listed methods are lawful for species other than those indicated, but are not commonly used in the UK.

			Fox	Red deer	Fallow deer	Roe deer	Brown hare	Mtn hare	Mink
LETHAL	**Methods involving dogs:**	Mounted packs	✓	✓		✓	✓		
		Foot packs	✓				✓	✓	✓
		Terriers	✓						✓
		Coursing	✓				✓		
	Methods involving shooting:	Rifle by day	✓	✓	✓	✓	✓	✓	✓
		Gun packs/beaters	✓				✓		
		Rifle and spotlamp	✓						
		Shotgun	✓	✓	✓	✓	✓	✓	✓
	Snaring		✓						
	Trapping:	Live-trapping and shooting	✓						✓
		Kill-trapping							✓
NON-LETHAL	Barrier methods		✓	✓	✓	✓			✓
	Management		✓	✓	✓	✓			
	Repellents			✓	✓	✓			

4. What can simulation models tell us about the effectiveness of population control methods?

4.1. Why use simulation models to estimate effectiveness?

There are many factors that influence the distribution and abundance of animals in landscapes; these are complex and often poorly understood. The abundance of a species within a particular area is dependent upon an interaction of four basic processes: reproduction, mortality, emigration and immigration. These processes are, in themselves, determined by other landscape-dependent factors such as the availability of food and den sites, and the incidence of anthropogenic factors such as culling. The abundance of an individual species can therefore be considered as the resultant of a complex interacting system of different life history processes.

Discovering genuine trends in population sizes of animal species is, therefore, a difficult task. There are so many confounding factors occurring simultaneously, that, compounded with the difficulty of obtaining an accurate census of numbers, it may be impossible to distinguish actual effects from the morass of 'noise'. In addition, mammal populations tend to act on a time-span of years or even decades, and so population trends caused by the cessation or escalation of control measures may not become evident for some time.

Clearly, in order to assess the impacts of culling on the abundance of culled species it is necessary to assess the relative impacts of all of these different processes as well as the impact of culling on population size. The complexity of these systems means that they are ideal subjects for investigation using modelling approaches. The purpose of modelling is to simplify complex systems so that the processes determining their functioning can be understood. Population dynamics modelling has a very long history. Lotka (1925) and Volterra (1926) developed the first mathematical models to investigate how species competed. These were the first of a series of analytical models that have been developed over the last 70 years. These models were based on numerical analyses of differential equations. These models did not pretend to analyse how populations of animals might behave in real landscapes, and as such their use was purely strategic in so far as they illustrated general ecological principles rather than addressed tactical issues. Strategic models of this type are particularly useful for evaluating the impacts of individual life history parameters and culling strategies on broad populations and have considerable value didactically, because they can be used to assess the broad impact of changes in life history parameters, such as cull rate, on population size.

4.2. What types of models have we used to estimate effectiveness of population control methods?

In this Chapter we use both strategic and tactical modelling approaches to investigate the impacts of variations in life history parameters and culling on the likely dynamics of the controlled species. We first develop a series of matrix population models to investigate the impacts of culling at the large population scale for foxes, hare, red deer and mink. We use this approach in a didactic sense to investigate firstly, under what conditions large scale population change could be influenced by culling and secondly to provide a comparison with other simple population analyses based on net gains or losses of individuals form populations. We then go on to develop individual-based models for the fox and mink to investigate the impacts of different culling strategies on the populations of these two species at the scale at which population control is practised. We focus on the fox and the mink since these species have very specialised space use patterns and behaviours. Finally, we use population viability analysis to examine long term trends in fox population dynamics under different scenarios.

4.2.1. Modelling at the level of the population

Matrix population models are a relatively recent technique used in population ecology. They were independently developed by Bernardelli (1941), Lewis (1942) and Leslie (1945, 1948), but were not widely adopted by ecologists until the 1970s. The work by Leslie (1945) is by far the most influential in the field of ecology. By expressing the basic age-specific projection equations of a population in matrix form, complex calculations can be reduced to matrix algebra. The eventual rate of increase of a population, and the stable age distribution, can be derived from such matrices. Jenson (1995, 1996) described a simple and easily-employed matrix model for density-dependent population growth that requires no new functions, matrices or parameters. It is directly analogous to the models produced by Lotka and Volterra, except that it takes into account population age structure, and so is useful for looking at systems where culling is applied differentially to different stages of the life-cycle. Used strategically, matrix population models can be used to assess the sensitivity of an animal population to culling, and the likely result of escalating or ceasing any management strategies employed.

The purpose of the population modelling approach in this inquiry is didactic. The approach is not intended to simulate actual population changes, but instead to examine the sensitivity of an animal population to culling. This is a complex issue, as culling might be differentially employed on different age classes of the animals involved, and this may have a delayed effect on population dynamics.

For the matrix population models we attempt to model a population sufficiently large that immigration and emigration are made irrelevant (because there is no ‘outside’ for these migrants to come to or from). In our individual-based and population viability analysis, however, we specifically look at the possible influence of migration on population persistence (see below). We reconstruct the population dynamics of the species in question using established values for life histories and then impose different levels of culling mortality on these populations, to see what effects these have on long term trends in the population.

For this study, matrix population models have been constructed to model populations on a regional or national scale. The use of matrix algebra overcomes some of the theoretical objections to differential population models since the matrix operates over discrete periods of time. The age structure of the population is taken into consideration, and so differential mortality may be applied to each age class of the animal modelled.

By modelling at the population level, particularly with matrix algebra, we lose the fine-scale interactions found in individual-based models (see below). However, these effects become less important when looking at a regional or national population. Matrix models are also ‘data-hungry’, in that they require age-specific life history data (yearly production and probability of survival for every age of the animal’s life), and this often requires extrapolation of the available data.

4.2.2. Modelling at the level of the individual

Whilst simple matrix models can be used to assess the impact of culling at the overall population scale, they takes no account of how culling is undertaken in practice. In order to assess how culling measures such as hunting and shooting might impact on real populations it is necessary to consider how the control is practised, and more particularly, where it is practised.

Populations of culled species do not exist as homogeneous individuals spread at random through landscapes: they have very specialised interactions with the landscapes in which they are found. Furthermore the behaviour patterns they adopt in landscapes will have knock-on effects on how culling is practised. The majority of species utilise a permanent or semi-permanent den or nest for breeding. In such a situation animals forage for food away from the nest in a more or less defined area that makes up a home range. A home range may be more or less exclusively occupied by a single animal, as in the mink, or by a small social group with dominant adults, as in the fox. Humans involved in managing these species are well aware of the implications of these behavioural and space use patterns for population control and have adapted their own culling behaviour to ensure that their control utilises these patterns to their own advantage. The complex space use-animal behaviour interaction and human culling behaviours require a different modelling approach for fox and mink. We focus on the fox and the mink since these species have very specialised space use patterns and behaviours

Spatially explicit population dynamics models predict the distribution of animals in the landscape on the basis of interactions between the landscape structure and individual behavioural processes such as home-range behaviour, territoriality and dispersal, and the life-history processes of births and deaths. In these models space provides reference points on which the populations processes occur and the distribution of organisms amongst these points emerges as the model is run. They are inevitably much more complex than models based on statistical methods because they attempt to simulate individual processes, but they offer a potential route for investigating species with complex social interactions and highly dynamic distributions such as the mink and fox.

Spatially-explicit models can be broadly classified as either population based or individual based depending on the level at which the life history processes are modelled. In individual-based models the processes of mortality and reproduction are estimated at the level of the individual and the overall effects on the population are derived from summation of the life-histories of each individual. These approaches are also termed i-space configuration models (Caswell & John, 1992). In population-based models mortality and reproduction are not modelled at the level of the individual; rather they are applied to groups of animals such as individual cohorts or age classes that constitute components of the total population. In these models the various life history data used may be drawn from a statistical distribution that represents that observed in the population.

These modelling approaches require a detailed knowledge of the effects of all biotic and abiotic factors on the dynamics of fecundity, mortality and migration within the individual population. In addition, some understanding of the basic behaviour of the species is needed in order to be able to link the life history processes to the space where they occur and this information is often poor (Lima & Zollner, 1996). Whilst the utility of using these approaches has been debated (Bart, 1995; Wennergren *et al.*, 1995) these models have been used to investigate the distributions of many other species of conservation and pest significance with considerable success (Akcakaya *et al.*, 1995; Liu *et al.*, 1995; Rushton *et al.*, 1997; Rushton *et al.*, 1999; Rushton *et al.*, 2000). Spatially explicit models are inevitably more tactical than the matrix models described above, in the sense that they attempt to simulate processes of individual behaviours such as the social group behaviour of the species and in this case, the human behaviour associated with different types of culling.

We develop these approaches in landscape where the fox and the mink are culled or are considered pests. For fox we utilise the three study areas subjected to hunting and intensively studied by Heydon *et al*. (2000) and Heydon & Reynolds (2000a, b). For mink we utilise the river catchment of the Thames, which has been extensively studied by Macdonald and associates (Macdonald & Strachan, 1999).

4.2.3. Population Viability Analysis

Population Viability Analysis models (PVA) are a form of modelling widely used to explore the risk posed to a population by various threats. In the context of conservation, PVA models are often used to identify factors which limit-the abundance of a population. In terms of pest control, comparable analyses may serve instead to reveal the likelihood that a given source of mortality will limit a population’s numbers or diminish its persistence. Our population viability analyses use the commercially produced program Vortex (version 8.03) (Lacey, 1993). We used Vortex because it is a widely used system, amenable to sensitivity analyses, and for consistency with the earlier explorations of fox population dynamics by Macdonald & Johnson (1996).

4.2.4. Summary of models used

The models used for each species are summarised in Table 4-1. In population based models, each step in the calculations is based on average estimates for a given subset of the population and often fixed estimates of mortality and fecundity are used. The model results tend to be easier to interpret, but they are less realistic. These models are best used to explore broad patterns in population dynamics and to understand the possible interactions between key population parameters. These models make unrealistic predictions for very small populations because the mortality and fecundity estimates are based on fractional estimates of individual numbers.

In individual-based models, the program keeps track of each individual. The life history processes of fecundity, survival and migration are determined by age and sex classes and by predefined probability distributions. These models generally require greater computational power and the results may be more complicated to interpret, but they have the advantage that they are more realistic. We often use these models to understand the inherent variation in population processes. These models produced variable data and statistical analyses are often required to interpret the results.

Spatially explicit models examine population dynamics within a spatial framework and are useful when the distribution of habitat features or resources (food, den sites etc.) have an important influence on the distribution of the animal. This is particularly important where a species has specific habitat requirements and preferred habitats are not unevenly distributed in the landscape. Spatially explicit models can be designed to look at a range of questions and can be tailored to explore issues related specific species and human animal interactions.

The population viability analysis can be used to examine the influence of different population parameters and environmental variability on population size and population persistence. A recent version allows for simple metapopulation structure (Lacey, 1993). They are widely used and so the results of these studies can be readily compared with other studies in the literature. Their generality has its drawbacks, however, as these models are less flexible. For example, in Vortex, only one mortality rate can be used for all age classes above age at first breeding.

Table 4-1 Modelling approach applied to each species.

Modelling approach	Species			
	Fox	Deer	Hare	Mink
Matrix population models (15 year)	X	X	X	X
Individual based models (15 year)	X (2 dimensional)			X (1 dimensional)
Vortex (PVA) (100 year)	**X** (metapopulation, single population)			

4.3. What do population-based models predict about the effectiveness of different population control methods?

4.3.1. General methodology for each species

To model at the population level, Jenson's (1995) method was employed for foxes, red deer, brown hares and mink. This involves taking the matrix population models developed by Leslie (1945), and making them density-dependent. This latter method requires knowledge of the stable age structure of the population under investigation, and its carrying capacity.

The general methodology for each species was as follows:

First, national estimates for carrying capacity were calculated. These assumed the population could grow unrestricted by any factors other than intrinsic mortality. Population densities for different habitats were obtained (from estimates provided by Macdonald *et al*., 1998), and multiplied by the national coverage of each habitat type, derived from the Institute of Terrestrial Ecology's Land Cover Map (LCM) held in the Countryside Information System.

Predictions of actual population sizes and distributions from satellite-derived land cover data such as these has been attempted, but found to be unreliable (Cardillo *et al*., 1999). The concept of 'carrying capacity' was therefore used to represent the maximum possible land area that could be occupied by the mammal species in the UK, assuming that distribution is both determined and limited solely by habitat.

Initial population sizes in the model were assumed to be half of this calculated carrying capacity.

These estimates are almost certainly too high, as not all habitats included in each land cover category will necessarily be suitable habitat for the mammals studied. Mammal distributions also show considerable geographic variation which may be independent of land cover. Caution must therefore be exercised in interpreting these results.

Age-specific life history data for fecundity and survival were obtained from the available literature. These data were then used to simulate population sizes using population models, written in the programming language C.

One of the simplest models of population growth is the discrete time logistic equation (May, 1974):

$$N_{t+1} = N_t + drN_t ,$$

where N_t, N_{t+1} are the population sizes at time *t* and *t+1* respectively, *d* is a density dependent function involving carrying capacity and *r* is the intrinsic rate of increase. Matrix models of the form used by Jenson (1995) are direct analogies of this generalised equation, but the numbers of individuals of each age class are tracked, instead of just the total population size. The most important aspect of a matrix model is the element known as the *Leslie Matrix*, which appears as follows:

$$M = \begin{vmatrix} F_0 & F_1 & F_2 & \ldots & F_x \\ P_0 & 0 & 0 & \ldots & 0 \\ 0 & P_1 & 0 & \ldots & 0 \\ \hdashline 0 & 0 & 0 & P_{x-1} & 0 \end{vmatrix} ,$$

where *F* is the fecundity of age group (the number of young produced per individual of each age during the time period *t*) and *P* is the probability of survival from one age group to the next. The Leslie matrix (or a version of it) takes the place of the intrinsic rate of increase in the above logistic equation.

The matrix model was used to derive population trajectories over time for the four species modelled in this fashion – the red fox, the brown hare, the red deer and the American mink. Given sufficient time, the proportion of the total population in each age class will remain constant between generations – this is the stable age distribution, and is a property of the life history variables provided. These population trajectories represented the national trends of the four species over fifteen years.

In all modelling exercises, it is important to test the sensitivity of the model to the input parameters. To do this, upper and lower bounds for the life history variables were estimated, and sets of input data were generated at random between these bounds using a method called Latin Hypercube Sampling (Vose, 1996). Age-specific fecundity and survival were varied in this way, along with additional mortality caused by the culling methods employed. The model was run for 15 years for each of 1000 parameter sets. The population sizes that result from this analysis can be compared with the input parameters using partial correlation to discover which life history or control parameters have significant effects on the total population size. Binary logistic regressions of the same data set will reveal which life history or control parameters have significant effects on population increase (or presence/absence for less fecund species). This step reveals which culling parameters have a statistically significant effect on population size. We can then use that information to predict what level of control of these species would be required to prevent population increase, or cause population decrease.

The statistically significant culling factors elucidated in the previous modelling stage were used to investigate the sensitivity of fox, hare, red deer and mink populations to culling. Each significant parameter (here called JUVENILE CULL, SUB-ADULT CULL and/or ADULT CULL) was given a value between 0 and 0.9 (i.e. by culling between 0% and 90% of the age class). All combinations of culling mortalities were used to discover the response of the long term population trajectory, and thus levels of culling that would be needed to halt population increase or cause a population decrease.

The elements of the matrix population models are derived from life history characteristics of the species under investigation. Age specific fecundity and survival must be either available or estimated to construct these models.

4.3.1.a. Assumptions made

Models are simplifications of reality, and therefore assumptions must be made in the formulation of any modelling system. As long as the effects that these assumptions make on the output of the model are taken into account when interpreting the results, these assumptions should not reduce the usefulness of modelling. In the context of population modelling, the purpose is not to mimic reality, but to investigate the sensitivity of animal populations to anthropogenic change, and thus these assumptions can be accommodated.

Some general assumptions that apply to matrix population models are:

Any culling mortality (JUVENILE CULL, SUB-ADULT CULL and/or ADULT CULL) is applied to the pre-breeding population. Natural mortality is assumed to occur before culling mortality – the former is density-dependent, the latter is density independent. No pregnant animals are assumed to be culled.

The same density-dependent function was applied to both birth and death, as it is in the logistic equation. Although it would be more realistic to separate the effects of density dependence on these two processes, this would increase the difficulty of parameter estimation. It is simpler, for this investigatory approach, to estimate density-dependence using parameters that can be estimated – the carrying capacity (as described above) and the stable age structure (which is a property of the life history variables used).

4.3.2. How effective are methods to control fox populations?

4.3.2.a. Approach

In the matrix population modelling approach, anthropogenic control was simulated by culling a fixed proportion of the pre-breeding population. This culling was density independent, and different levels of culling could be applied to different age groups, namely the juveniles (defined as individuals yet to reach breeding age), sub-adults (individuals in the first year of breeding) and adults (individuals in a second or later breeding season).

The aim of this modelling approach is to discover what levels of culling are required to result in a long term population decline in Great Britain.

4.3.2.b. Data used

Red foxes are the most catholic species covered in this inquiry with regards to habitat preference, being found in nearly all land cover categories. Densities were estimated to be 1.5 foxes/km^2 for urban and suburban areas, 0.025 foxes/km^2 for bare land, bogs, salt marshes and other unfavourable habitats, and 2.5 foxes/km^2 for all other terrestrial habitats. The final estimation of the maximum number of foxes that Great Britain could sustain, given no anthropogenic restrictions, is approximately 434,400. See Table 4-28 for more details.

Estimates of annual fecundity and survival of foxes were derived from the available scientific literature. For the construction of matrix models, age-specific information for these two parameters is required. In many cases, these can be estimated from the available information.

Fecundity is at a peak in vixens of 2-4 years old (Harris & Smith, 1987), at which time the mean litter size for a vixen is 4.5 cubs (Harris, 1979). This potential fecundity of vixens is reduced by 20% due to late-term reproductive failure (Corbet & Harris, 1991). These data lead to the fecundity schedule in Table 4-2.

Table 4-2 Fox fertility schedule

Age	Potential litter size	Actual litter size	Young/fox
0-1	0.0	0.000	0.000
1-2	4.5	3.593	1.797
2-3	4.5	3.593	1.797
3-4	4.5	3.593	1.797
4-5	4.0	3.194	1.597
5-6	4.0	3.194	1.597
6-7	3.0	2.396	1.198
7-8	3.0	2.396	1.198
8-9	2.0	1.597	0.799
9-10	2.0	1.597	0.799

Survival data can be estimated from the age structure of a population. Culling occurs in most fox populations studied, and therefore survivorship data is likely to underestimate actual fox numbers in the absence of a cull.

The mean age structure over several populations of foxes was determined from Corbet & Harris (1991, p365). This can then be used to calculate age-specific survival – the probability of surviving to the next age class (Table 4-3).

Table 4-3 Fox age-specific survival

Age	% Population	% Surviving to next age
0-1	56.286	0.414
1-2	23.286	0.442
2-3	10.286	0.583
3-4	6.000	0.357
4-5	2.143	0.933
5-6	2.000	0.470
6-7	0.940	0.830
7-8	0.780	0.962
8-9	0.750	0.667
9-10	0.500	

4.3.2.c. Assumptions made

In addition to the general assumptions of the matrix models, the following initial assumptions are made in the fox models:

To derive age-specific fecundities it was necessary to extrapolate the mean fecundity of vixens in their prime to older animals. However, it is known that fecundity decreases in vixens over the age of four, and the mean yearly fecundity for all vixens of all ages was 3.14, the same as the mean yearly fecundity of the three regions given in Corbet & Harris (1991)

Density dependence is known to limit fox fecundity (see section 1.3.2.a). In these matrix models, as explained above, the effects on mortality and fecundity of density-dependence are combined into a single function.

The survivorship data used in the matrix modelling were taken from fox populations subjected to control. Therefore, levels of natural mortality are estimated to be higher than in reality, so the subsequent models underestimate fox populations and thus estimates of culling levels required to control fox populations are a little high (that is, a lower culling mortality will produce the same change). However, the general principles remain the same because the survival data used was taken from a number of fox populations with different levels of population control.

4.3.2.d. Results

Table 4-4 Fox age structure

Stage	Age	Proportion
juveniles	0-1	0.563
sub-adults	1-2	0.233
adults	2-10	0.204

The intrinsic rate of increase of the modelled fox population can be calculated from the Leslie matrix by eigen analysis. This results in a value of 1.13, reflecting the relatively high reproductive rate of fox populations. Given no form of population regulation (either natural or anthropogenic), this rate of increase indicates that the fox population will increase by 12.2% every year. This potential for population growth is almost never realised, because the habitat restricts the maximum size of a population and density-dependent processes mean that population growth rates decrease as the carrying capacity is reached. The stable age structure of the modelled population is shown in Table 4-4., and the results of the partial correlation in Table 4-5.

Table 4-5 Partial correlation results.*** indicates p<0.001; ** indicates p<0.01; * indicates p<0.05; ns indicates p>0.05

	Variable	F value (d.f.=1, 993)	Significance level
Life history factors	Fecundity	1134.54	***
	Juvenile survival	1644.12	***
	Sub-adult survival	216.25	***
	Adult survival	85.91	***
Control factors	Juvenile culling	2.50	ns
	Sub-adult culling	8.01	**
	Adult culling	7.99	**

The sensitivity analysis of the matrix model showed that long term population size of the model was significantly affected by applying ADULT CULL and SUB-ADULT CULL on a yearly basis, but JUVENILE CULL did not significantly affect population trends. This is interesting, because it suggests that modelled fox populations are able to compensate for high juvenile mortality, presumably by lowered mortality of the other age classes and an increased fecundity.

The intrinsic rate of increase indicates the ease with which the population recovers from perturbation. Because this rate of increase is achieved only at low densities, it is important to examine long term population trends under different culling regimes. When animals are removed from a population, compensation may occur, so that natural mortality or emigration is lower in the following year, allowing the population to grow faster than it did before the cull (see section 1.3.2). By using the population models we can discover the sensitivity of the fox population to man-made perturbation by imposing a yearly cull of different age groups. By examining all levels of cull mortality, we can investigate the responses of a population to this control

The sensitivity analysis reported above indicated that fox population sizes are significantly affected by SUB-ADULT CULL and ADULT CULL. These two parameters were therefore varied in tandem, resulting in an estimation of the overall predicted effect on final population size after fifteen years of simulation.

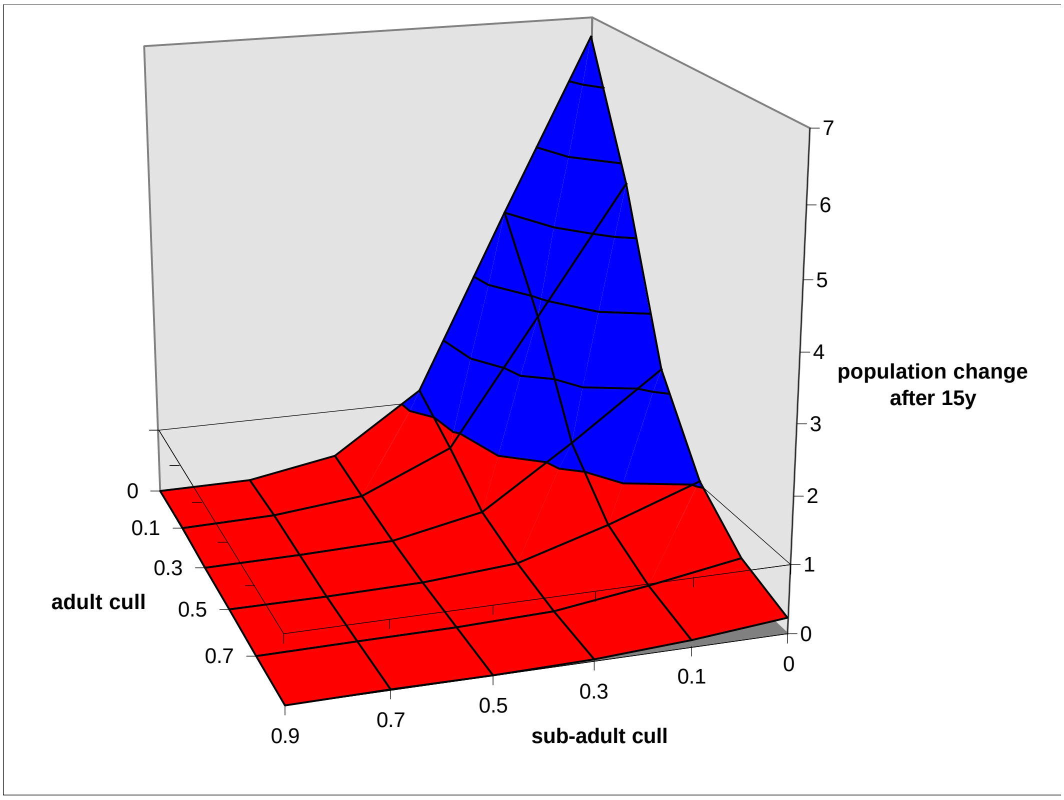
For the comparison of sub-adult fox cull and adult fox cull, the results can be seen in Figure 4-1, and summarised in Table 4-6

Table 4-6 Comparison of sub-adult and adult fox cull. ++ = moderate population increase (initial population doubles in 15 years); + = small population increase (between 1.2 and 2 times initial population in 15 years); 0 = no significant population change; - = small population decrease (between 0.8 and 0.5 of initial population in 15 years); - - = moderate population decrease (between 0.5 and 0.1 of initial population in 15 years); - - - = large population decrease (less than 0.1 of initial population in 15 years); X = population extinction

		Adult cull					
		0	0.1	0.3	0.5	0.7	0.9
Sub-adult cull	0	++	++	++	0	- -	- -
	0.1	++	++	+	-	- -	- -
	0.3	+	-	- -	- -	- - -	- - -
	0.5	- -	- -	- - -	- - -	- - -	- - -
	0.7	- - -	- - -	- - -	- - -	X	X
	0.9	X	X	X	X	X	X

The model suggests that a control of more than one age class will produce the most effective control of fox populations in the long term.

Figure 4-1 Illustrates the predicted effect of a fox cull. The red area indicated parameter values that result in either no increase or a decrease in total fox population size after fifteen years, the blue area indicates the parameter values resulting in net increase after 15 years.



Without ADULT CULL, over 30% of sub-adults must be removed from the population each year to see a small decrease in total fox numbers after 15 years. At this level of SUB-ADULT CULL, if over 10% of the adult foxes are also removed each year, then there will be a greater than ten-fold reduction of fox numbers after fifteen years.

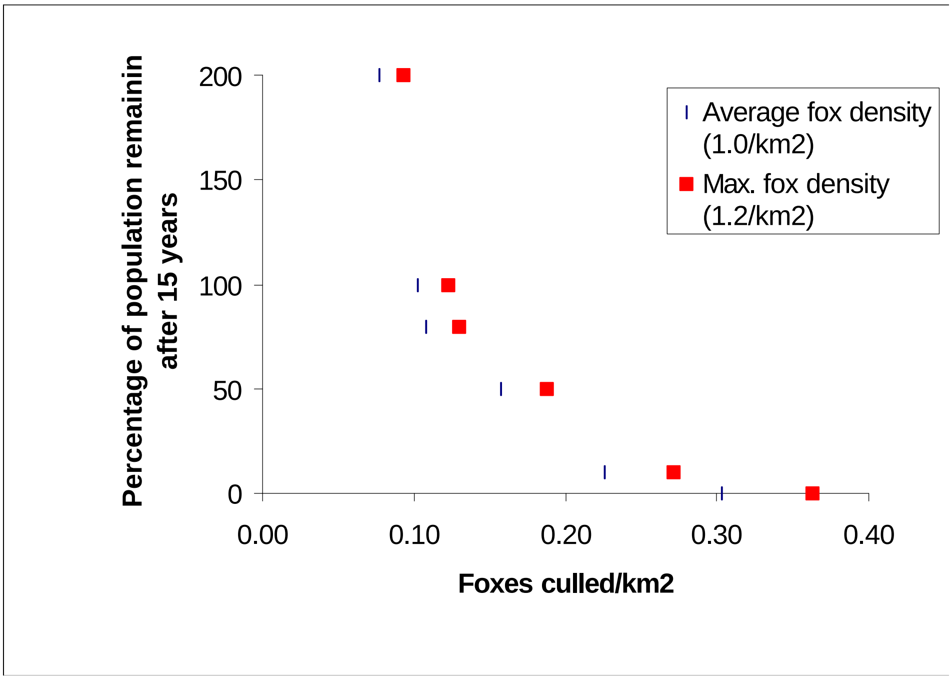
A high value for SUB-ADULT CULL (greater than 70% per year) will result in extinction of fox populations within 15 years, whatever the level of ADULT CULL.

The values given in Figure 4-1 and Table 4-6 represent the response of the modelled population only in terms of proportion of individuals in each age class affected by the levels of each CULL parameter. To compare this output to actual culling intensity it is necessary to convert the proportion affected by these parameters into densities affected.

To calculate the density on the ground of individuals culled, the stable age class distributions resulting from the matrix model were converted into fox density per age class by multiplying the proportion of the age class by the density per km^2 of the whole fox population at carrying capacity. Data for real fox densities were available from Heydon *et al.* (2000). These values are an average fox density of $1/km^2$ and a maximum of $1.2/km^2$.

This calculation allowed the production of population response curves for different levels of CULL parameters. These plots are shown in Figure 4-2.

Figure 4-2 The percentage of the fox population remaining after 15 years, assuming an average carrying capacity of 1.0/km^2 (diamond symbols) and a maximum carrying capacity of 1.5/km^2 (red symbols) against different densities of adults and sub-adults culled/km^2. Estimates of overall minimum cull density were estimated as 0.71, 0.37 and 0.41/km^2 for Wales, West Midlands and East Norfolk. These densities were adjusted to exclude juveniles culled (proportion juveniles = 0.563) assuming even selection of the age classes, giving 0.40, 0.21 and 0.23 respectively. This corresponds to limiting the population by 50-10%, causing population extinction over 15 years.



4.3.3. How effective are methods to control hare populations?

4.3.3.a. Approach

In the matrix population modelling approach, anthropogenic control was simulated by culling a fixed proportion of the pre-breeding population. This culling was density independent, and different levels of culling could be applied to different age groups, namely the juveniles (defined as individuals yet to reach breeding age), sub-adults (individuals in the first year of breeding) and adults (individuals who have reached peak reproductive performance).

The aim of this modelling approach is to discover what levels of culling are required to result in a long term population decline in Great Britain

4.3.3.b. Data used

Brown hare are found primarily in open grassland and farmland; and occur at a wide range of densities, even in similar habitats, and year-to-year variation is considerable. On arable land, average densities can be 14.4 hares/km^2, for pastoral and managed grasslands 9.8 hares/km^2, and for less favourable grasslands, densities average 7.2 hares/km^2. The final estimation of the maximum number of hares that Great Britain could sustain, given no anthropogenic restrictions, is approximately 1 million.

Brown hares have 1–4 litters per year (average 3) and 1–4 leverets per litter (max 10) (Macdonald & Barrett, 1993). With a mean of 2.5 leverets per litter, and 3 litters per year, the mean fecundity for a female in its reproductive prime (3-4 years old, according to Frylestam, 1980) is 7.5 young per year. Fecundity is known to be lower in hares younger and older than this age group (Frylestam, 1980), so we assumed a slow decrease in yearly reproductive output up to the age of 7 (the average lifespan of a wild hare, Corbet & Harris, 1991). Hares mature at about 8 months old, so the first age group (0-1 year) can produce approximately 4/12 of the young that a hare in her prime can achieve. These data lead to the fecundity schedule in Table 4-7:

Table 4-7 Hare fertility schedule

Age	Litter size	Young/hare
0-1	1.88	0.94
1-2	5.00	2.50
2-3	7.00	3.50
3-4	7.50	3.75
4-5	7.00	3.50
5-6	6.00	3.00
6-7	4.00	2.00

Survival data (Table 4-8) can be estimated from the age structure of a population. Culling occurs in most hare populations studied, and therefore survivorship data is likely to underestimate actual hare numbers in the absence of a cull.

The mean age structure over several populations of hares was determined from Tapper (1991); age categories are less than 1 year, 1–2 years, 2–3 years, 3 years or older). This can then be used to calculate age-specific survival – the probability of surviving to the next age class. We have extrapolated the age structure of the population to account for hares above the age of 3. This extrapolation is based on the fact that although first winter mortality is thought to be higher, mortality is otherwise constant with age (Broekhuizen, 1979). The number of hares in the 3+ category remains the same as that given in Tapper (1991), approximately 22.

Table 4-8 Hare survival data. * = extrapolated

Age	Number of hares	% Surviving to next age
0-1	2040.00	0.374
1-2	763.00	0.270
2-3	206.00	0.099
3-4	20.40*	0.100
4-5	2.04*	0.100
5-6	0.20*	0.100
6-7	0.02*	-

4.3.3.c. Assumptions made

In addition to the general assumptions of the matrix models, the following initial assumptions are made in the hare models:

Extrapolation of prime fecundity was required to obtain age specific production for older hares. With no data available on this, we assumed a slow decline from an average of 7.5 leverets per year for a hare in her prime, to 4 leverets per year for a seven year old individual. Because of this assumption, the model is more likely to underestimate hare productivity than overestimate.

The matrix models used here cannot accurately take into account the multivoltine pattern of breeding in hare populations (i.e. the fact that they have multiple litters in a year). However, the lowest age class is given a value for fecundity to account for the fact that individuals can achieve breeding success in the same year that they were born, at an age of 8 months. In real populations, fecundity of female hares changes throughout the year, and this is not accounted for either. Optimistic values for hare productivity have therefore been used throughout this modelling section. The effect of the failure of the matrix model to allow for multivoltine breeding is only likely to be important in small populations, as density dependent effects on fecundity will come into play in populations close to the carrying capacity.

The initial age structure of hare populations was derived from bag counts from spring shoots on estates in England. The data are therefore assuming that the numbers of hares shot of each age is representative of the age-structure of the pre-breeding hare population.

Survival data was estimated from a culled population. The matrix model is likely to output hare populations that are under-estimates in the absence of a cull

Extrapolation of the survival data was needed for hares over 3 years old. However, it is known that adult hare mortality stays constant with age (Broekhuizen, 1979), and we were able to use this information to estimate the number of hares from their fourth year onwards.

4.3.3.d. Results

The intrinsic rate of increase of the modelled hare population can be calculated from the Leslie matrix by eigen analysis. This results in a value of 1.65, reflecting the high reproductive rate of hare populations – given no form of population regulation (either natural or anthropogenic), this rate of increase indicates that the hare population will double every year. This potential for population growth is almost never realised, because the habitat restricts the maximum size of a population and population growth rate decreases as the carrying capacity is approached. The stable age structure of the modelled population is given in Table 4-9.

Table 4-9 Hare stable age structure

Stage	Age	Proportion
Juveniles	0-1	0.673
Sub-adults	1-2	0.252
Adults	2-7	0.075

The results of the partial correlation are as follows (Table 4-10):

Table 4-10 Partial correlation results for hare. *** indicates p<0.001; ** indicates p<0.01; * indicates p<0.05; ns indicates p>0.05

	Variable	F value (d.f.=1, 993)	Significance level
Life history factors	Fecundity	103.20	***
	Juvenile survival	235.43	***
	Sub-adult survival	23.74	***
	Adult survival	5.49	*
Control factors	Juvenile culling	245.68	***
	Sub-adult culling	39.38	***
	Adult culling	15.27	***

The sensitivity analysis of the matrix model showed that long term population size of the model was significantly affected by applying JUVENILE CULL, SUB-ADULT CULL and ADULT CULL on a yearly basis. Hare are so productive that huge populations can result from unperturbed populations. Removal of breeding individuals can substantially affect the long term population trends, as can removal of leverets. However, even with large levels of CULL parameters, the resultant populations in the long term are still huge (just not as huge as in situations without a cull), so it is necessary to investigate further to find the true effects of culling on the predicted population trends.

The intrinsic rate of increase indicates the facility of the population to recover from perturbation. Because this rate of increase is only achieved at low densities, it is important to examine long term population trends under different culling regimes. When animals are removed from a population, compensation may occur, so that natural mortality or emigration is lower in the following year, allowing the population to grow faster than it did before the cull. By using the population models we can discover the sensitivity of the hare population to man-made perturbation by imposing a yearly cull of different age groups. By examining all levels of cull mortality, we can investigate the responses of a population to this control

The sensitivity analysis reported above indicated that hare population sizes are significantly affected by culling of all age classes, juveniles, sub-adults and adults. These three parameters were therefore varied in tandem, resulting in an estimation of the overall predicted effect on final population size after fifteen years of simulation.

For the comparison of sub-adult hare control and adult hare control, the results can be seen in Figure 4-3 and summarised in Table 4-11. With only SUB-ADULT CULL and ADULT CULL parameters, the model output indicates that hare populations are impossible to control unless virtually every individual is removed. This is because of the high reproductive potential of each individual in the population.

Table 4-11Comparison of sub-adult and adult hare cull. ++ = moderate population increase (initial population doubles in 15 years); + = small population increase (between 1.2 and 2 times initial population in 15 years); 0 = no significant population change; - = small population decrease (between 0.8 and 0.5 of initial population in 15 years); - - = moderate population decrease (between 0.5 and 0.1 of initial population in 15 years); - - - = large population decrease (less than 0.1 of initial population in 15 years); X = population extinction

		Adult cull					
		0	0.1	0.3	0.5	0.7	0.9
Sub-adult cull	0	++	++	++	++	++	++
	0.1	++	++	++	++	++	++
	0.3	++	++	++	++	++	++
	0.5	++	++	++	++	++	++
	0.7	++	++	++	++	++	++
	0.9	++	++	++	++	++	++

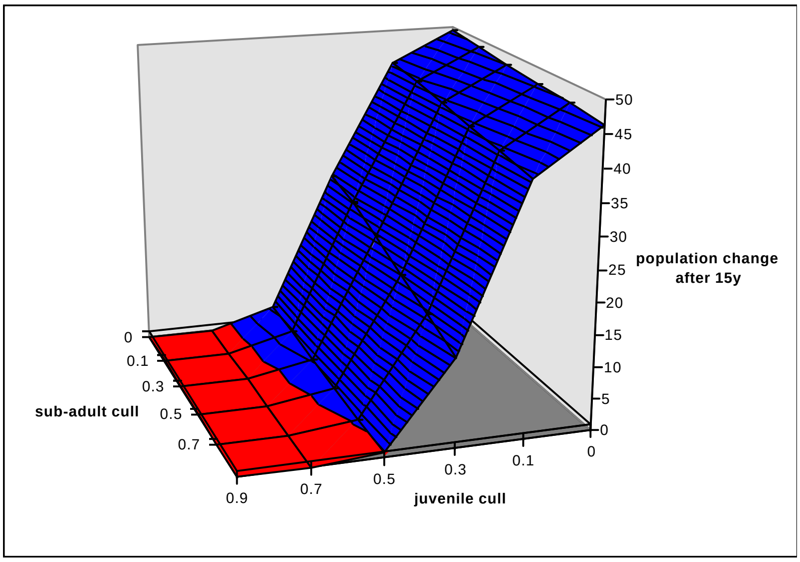
For the comparison of juvenile hare control and sub-adult hare control, the results can be seen in Figure 4-3, and summarised in Table 4-12. The model output indicates that control of hare populations is possible through application of both JUVENILE CULL and SUB-ADULT CULL. The effect would be to remove individuals before they have a chance to breed, and thus population control is achieved. Control levels need to be quite high to get this effect, with more than 50% JUVENILE CULL each year without sub-adult control, down to more than 10% JUVENILE CULL with more than 70% SUB-ADULT CULL.

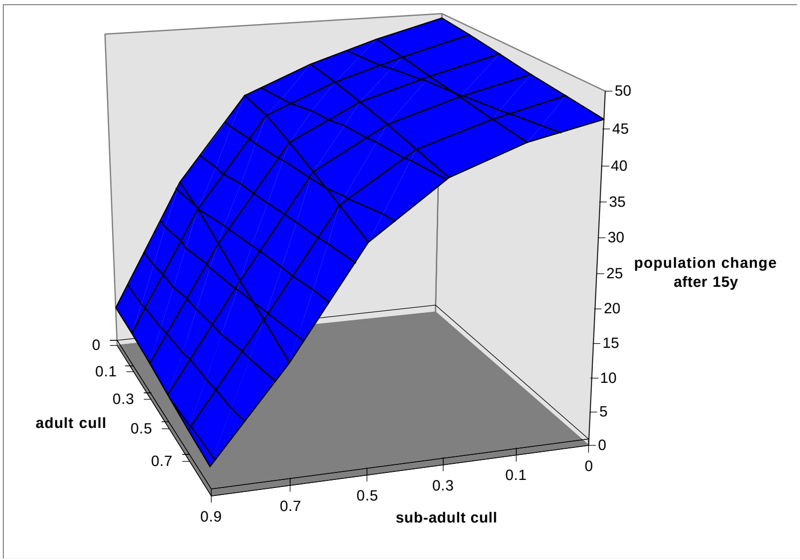
Table 4-12 Comparison of juvenile and sub-adult hare cull. ++ = moderate population increase (initial population doubles in 15 years); + = small population increase (between 1.2 and 2 times initial population in 15 years); 0 = no significant population change; - = small population decrease (between 0.8 and 0.5 of initial population in 15 years); - - = moderate population decrease (between 0.5 and 0.1 of initial population in 15 years); - - - = large population decrease (less than 0.1 of initial population in 15 years); X = population extinction

		Sub-adult cull					
		0	0.1	0.3	0.5	0.7	0.9
Juvenile cull	0	++	++	++	++	++	++
	0.1	++	++	++	++	++	+
	0.3	++	++	++	++	0	- - -
	0.5	++	++	-	- -	- - -	- - -
	0.7	- - -	- - -	- - -	- - -	X	X
	0.9	X	X	X	X	X	X

For the comparison of juvenile hare control and adult hare control, the results can be seen in Figure 4-3, and summarised in Table 4-13.

Figure 4-3 Illustrates the predicted effect of a hare cull. The red area indicated parameter values that result in either no increase or a decrease in total hare population size after fifteen years, the blue area indicates the parameter values resulting in net increase after 15 years. Each graph shows the effects of varying two of the three culling parameters





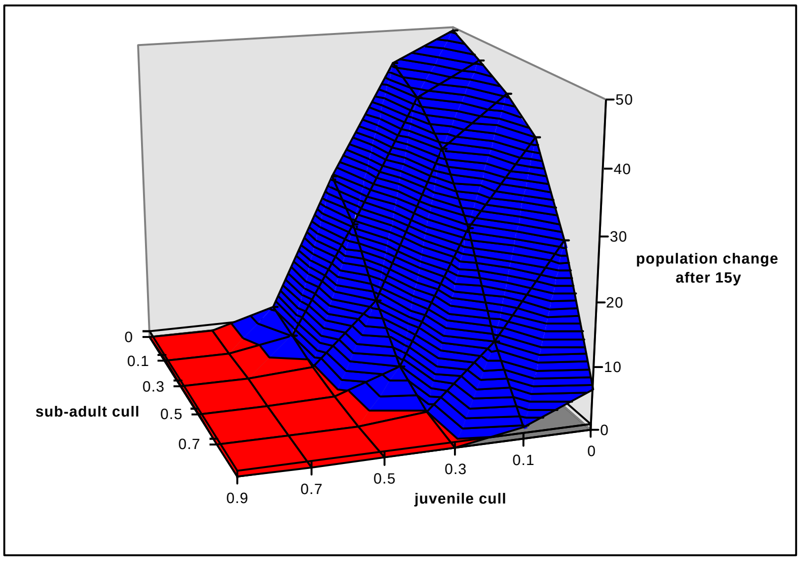


Table 4-13 Comparison of juvenile and adult hare cull. ++ = moderate population increase (initial population doubles in 15 years); + = small population increase (between 1.2 and 2 times initial population in 15 years); 0 = no significant population change; - = small population decrease (between 0.8 and 0.5 of initial population in 15 years); - - = moderate population decrease (between 0.5 and 0.1 of initial population in 15 years); - - - = large population decrease (less than 0.1 of initial population in 15 years); X = population extinction

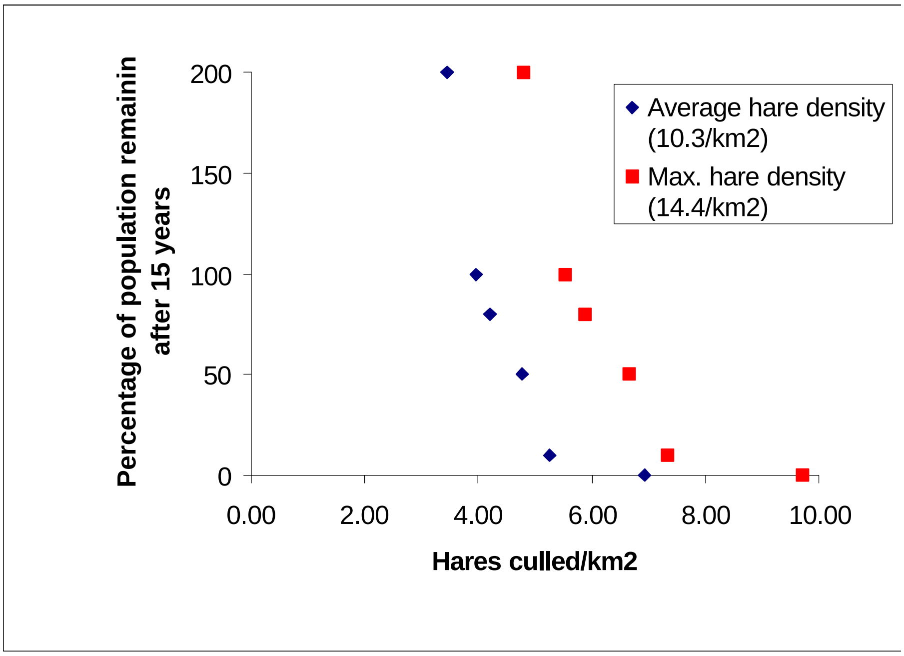
		Adult cull					
		0	0.1	0.3	0.5	0.7	0.9
Juvenile cull	0	++	++	++	++	++	++
	0.1	++	++	++	++	++	++
	0.3	++	++	++	++	++	++
	0.5	++	++	++	+	0	-
	0.7	- - -	- - -	- - -	- - -	- - -	- - -
	0.9	X	X	X	X	X	X

The predicted effects of juvenile control combined with adult control is similar to those given above, but ADULT CULL must be more severe than SUB-ADULT CULL to give the same effect.

The values given in the above tables and figures only represent the response of the modelled population in terms of proportion of individuals in each age class affected by the levels of each CULL parameter. To compare this output to actual culling rates it is necessary to convert the proportion affected by these parameters into densities affected.

To calculate the density of individuals culled, the stable age class distributions resulting from the matrix model were converted into hare density per age class by multiplying the proportion of the age class by the density per km^2 of the whole hare population at carrying capacity. Hare population densities were taken from Table 4-28 – average hare density is 10.3 / km^2 and a maximum hare density is 14.4 / km^2.

Figure 4-4 The percentage of the hare population remaining after 15 years, assuming an average carrying capacity of 10.3/km^2 (blue symbols) and a maximum carrying capacity of 14.4/km^2 (red symbols) against different densities of adults, sub-adults and juveniles culled/km^2**.**



This calculation allowed the production of population response curves for different levels of the CULL parameters. These plots are shown in Figure 4-4.

4.3.4. How effective are methods to control deer populations?

4.3.4.a. Approach

In the matrix population modelling approach, anthropogenic control was simulated by culling a fixed proportion of the pre-breeding population. This culling was density independent, and different levels of culling could be applied to different age groups, namely the juveniles (defined as individuals yet to reach breeding age), sub-adults (individuals in the first year of breeding) and adults (individuals who have reached peak reproductive performance).

The aim of this modelling approach is to discover what levels of culling are required to result in a long term population decline in Great Britain.

4.3.4.b. Data used

Red deer can utilise a range of habitats, including woodland, grassland, moor and scrub. Density is highest in open woodland habitats (22.5 deer/km^2) and lower in grassland and moor habitats (15.5 deer/km^2). These estimates are not achieved by English populations of red deer, which approach nearer 1 deer/km^2 even in open woodland habitats. However, the purpose of this exercise is to estimate the maximum density that the species could achieve, and so the higher values for Scottish populations were used. The final estimation of the maximum number of red deer that Great Britain could sustain, given no anthropogenic restrictions, is approximately 1.4 million individuals.

Scottish populations of red deer in open upland habitats have been extensively studied, and age specific life history data have been published (Lowe, 1969) and used in matrix models (Usher, 1973). However, the Scottish populations have a low fecundity in comparison with data collected from the populations in the south west of England, who dwell in a more productive environment (Ratcliffe, 1987). Data from the south west indicate 73% of yearling hinds and 91.5% of older hinds are pregnant (Langbein, 1997; data from the two regions have been averaged). We have assumed that fecundity is not constant with age, but reaches a peak by the 7th year, then declines slowly thereafter. This is in accordance with the fecundity schedule of Scottish upland populations (Lowe, 1969).

These assumptions produce the fecundity schedule in Table 4-14 (values for Scottish upland deer have been included for reference). In the absence of data which is more representative of English populations of red deer, the age-specific survival – the probability of surviving to the next age class – of the Scottish populations was used (Lowe, 1969) (Table 4-15).

4.3.4.c. Assumptions made

In addition to the general assumptions of the matrix models, the following initial assumptions arise from the deer models:

The models are designed to model populations in Great Britain, however, it is the English populations of deer that are of specific interest in this Inquiry. As the fecundity of English deer is greater than that of Scottish deer, we have chosen to use the English data, as it will produce a ‘best case scenario’. However, if these results were to be extrapolated to Scottish populations, caution must be used in coming to the same conclusions as in the following section.

Extrapolation was necessary to derive age-specific fecundities for deer over 2 years old. This information was available for Scottish deer, but these populations are less productive than English populations, and so could not be used. However, the pattern of age specific fecundity could be derived from this life history data – in Lowe’s (1969) data, fecundity steadily increases to a maximum at 6-7 years of age, and declines thereafter. The same pattern was used in the matrix models, ensuring that the mean productivity of hinds older than 2 remained the same in Langbein (1997).

In the absence of age-specific survival data for the English populations of red deer, details from the life table of Scottish deer have been used.

Table 4-14 Red deer fertility schedule

Age	Young/hind (SW England, Langbein, 1997)	Young/deer (SW England, Langbein, 1997)	Young/deer (Scottish uplands, Lowe, 1969)
0-1	0.00	0.000	0.000
1-2	0.73	0.365	0.000
2-3	0.85	0.425	0.311
3-4	0.90	0.450	0.278
4-5	0.92	0.460	0.302
5-6	0.92	0.460	0.400
6-7	0.98	0.490	0.476
7-8	0.95	0.475	0.358
8-9	0.92	0.460	0.447
9-10	0.92	0.460	0.289
10-11	0.92	0.460	0.283
11-12	0.92	0.460	0.285
12-13	0.92	0.460	0.283
13-14	0.92	0.460	0.282
14-15	0.92	0.460	0.285
15-16	0.92	0.460	0.284

Table 4-15 Red deer survival schedule

Age	% Surviving to next age
0-1	0.863
1-2	0.903
2-3	0.892
3-4	0.879
4-5	0.863
5-6	0.841
6-7	0.810
7-8	0.498
8-9	0.328
9-10	0.859
10-11	0.835
11-12	0.802
12-13	0.753
13-14	0.671
14-15	0.508
15-16	-

4.3.4.d. Results

The intrinsic rate of increase of the modelled red deer population can be calculated from the Leslie matrix by eigen analysis. This results in a value of 1.16, reflecting the relatively high reproductive rate of deer populations – given no form of population regulation (either natural or anthropogenic), this rate of increase indicates that the deer population will increase by 14.8% every year. This potential for population growth is almost never realised, because the habitat restricts the maximum size of a population and population growth rates decrease as the carrying capacity is reached. However, this intrinsic rate of increase indicates the facility of the population to recover from perturbation. The stable age structure of the modelled population is given in Table 4-16, and the results of the partial correlation in Table 4-17.

Table 4-16 Deer stable age structure

Stage	Age	Proportion
Juveniles	0-1	0.372
Sub-adults	1-3	0.37
Adults	3-16	0.258

Table 4-17 Partial correlation results. *** indicates p<0.001; ** indicates p<0.01; * indicates p<0.05; ns indicates p>0.05

	Variable	F value (d.f.=1, 993)	Significance level
Life history factors	Fecundity	266.75	***
	Juvenile survival	550.27	***
	Sub-adult survival	413.60	***
	Adult survival	6.23	*
Control factors	Juvenile culling	7.44	*
	Sub-adult culling	16.155	***
	Adult culling	5.09	*

All of the CULL parameters had a significant effect on the long term population trends of this matrix model. The annual population growth of this matrix model is comparable with that of the fox, however, this is due to different reasons. Red deer produce many less young than foxes, but they live longer and so the population growth rate is similar. Because fewer young are produced in deer populations, the matrix model is more sensitive to removal of yearling animals, each of which has a long and productive life ahead of it.

The intrinsic rate of increase indicates the facility of the population to recover from perturbation. Because this rate of increase is only achieved at low densities, it is important to examine long term population trends under different culling regimes. When animals are removed from a population, compensation may occur, so that natural mortality or emigration is lower in the following year, allowing the population to grow faster than it did before the cull. By using the population models we can discover the sensitivity of the red deer population to man-made perturbation by imposing a yearly cull of different age groups. By examining all levels of cull mortality, we can investigate the responses of a population to this control

The sensitivity analysis reported above indicated that red deer population sizes are significantly affected by culling of all age classes, juveniles, sub-adults and adults. These three parameters were therefore varied in tandem, resulting in an estimation of the overall predicted effect on final population size after fifteen years of simulation.

For the comparison of sub-adult deer control and adult deer control, the results can be seen in Figure 4-5, and summarised in Table 4-18. This simulation suggests that the model is quite sensitive to the SUB-ADULT CULL parameter. Fifteen years of removing sub-adults at levels of 10% or greater will result in population decline. The populations are more resistant to ADULT CULL – without removal of sub-adults, ADULT CULL values greater than 10% will halt population decline, but greater than 50% ADULT CULL is predicted to be required to see a significant sustained reduction in numbers.

For the comparison of juvenile deer control and sub-adult deer control, the results can be seen in Figure 4-5, and summarised in Table 4-19. The matrix model combining JUVENILE CULL and SUB-ADULT CULL predicts that any level of removal greater than 10% of either of these age classes will result in modelled population decline. Significant culling of the fawns and yearlings will result in population extinction within 15 years – a level of JUVENILE CULL and SUB-ADULT CULL of more than 30% will achieve this, although if more juveniles are culled, less sub-adults need to be removed to achieve the same result, and vice versa.

For the comparison of juvenile deer control and adult deer control, the results can be seen in Figure 4-5 and summarised in Table 4-20. This simulation compares well with the SUB-ADULT CULL / ADULT CULL scenario described above. It therefore indicates that the modelled population will decline if fawns are culled at an intensity of greater than 10%, but can withstand up to 50% ADULT CULL given no JUVENILE CULL.

Table 4-18 Comparison of sub-adult and adult deer cull. ++ = moderate population increase (initial population doubles in 15 years); + = small population increase (between 1.2 and 2 times initial population in 15 years); 0 = no significant population change; - = small population decrease (between 0.8 and 0.5 of initial population in 15 years); - - = moderate population decrease (between 0.5 and 0.1 of initial population in 15 years); - - - = large population decrease (less than 0.1 of initial population in 15 years); X = population extinction

		Adult cull					
		0	0.1	0.3	0.5	0.7	0.9
Sub-adult cull	0	+	0	0	0	-	-
	0.1	-	-	-	- -	- -	- -
	0.3	- -	- -	- -	- - -	- - -	- - -
	0.5	- - -	- - -	- - -	- - -	- - -	- - -
	0.7	- - -	- - -	- - -	- - -	- - -	- - -
	0.9	X	X	X	X	X	X

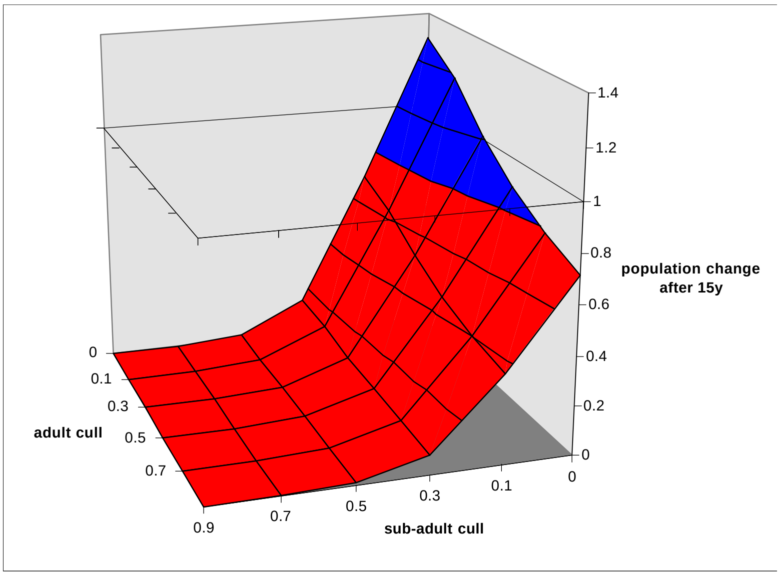
Table 4-19 Comparison of juvenile and sub-adult deer cull. ++ = moderate population increase (initial population doubles in 15 years); + = small population increase (between 1.2 and 2 times initial population in 15 years); 0 = no significant population change; - = small population decrease (between 0.8 and 0.5 of initial population in 15 years); - - = moderate population decrease (between 0.5 and 0.1 of initial population in 15 years); - - - = large population decrease (less than 0.1 of initial population in 15 years); X = population extinction

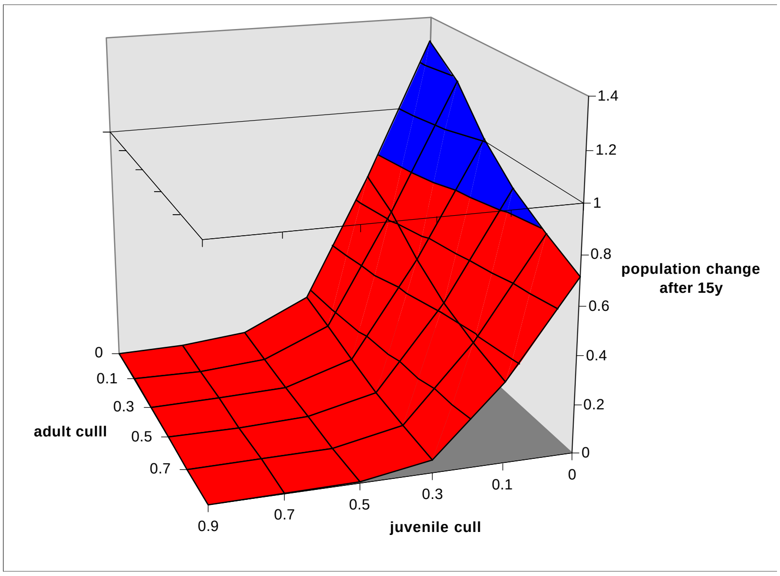
		Sub-adult cull					
		0	0.1	0.3	0.5	0.7	0.9
Juvenile cull	0	+	-	- -	- - -	- - -	X
	0.1	-	- -	- - -	- - -	X	X
	0.3	- -	- - -	- - -	X	X	X
	0.5	- - -	- - -	X	X	X	X
	0.7	- - -	- - -	X	X	X	X
	0.9	- - -	X	X	X	X	X

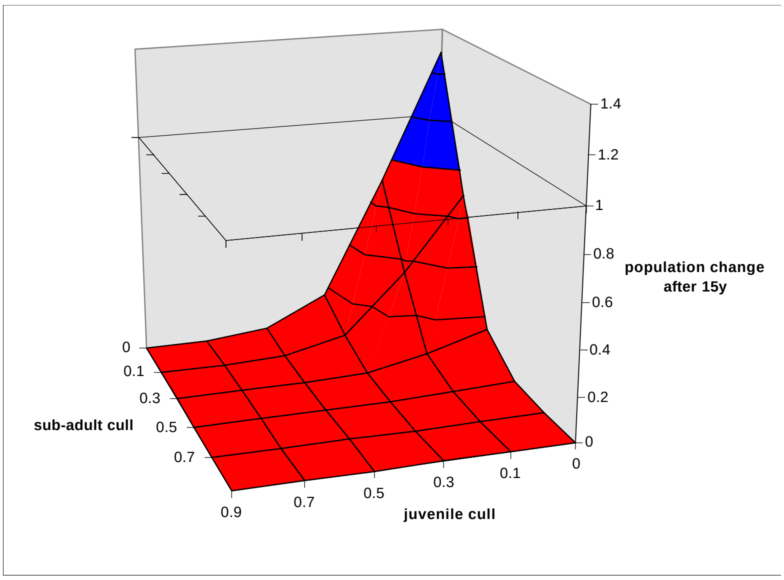
Table 4-20 Comparison of sub-adult and adult fox cull. ++ = moderate population increase (initial population doubles in 15 years); + = small population increase (between 1.2 and 2 times initial population in 15 years); 0 = no significant population change; - = small population decrease (between 0.8 and 0.5 of initial population in 15 years); - - = moderate population decrease (between 0.5 and 0.1 of initial population in 15 years); - - - = large population decrease (less than 0.1 of initial population in 15 years); X = population extinction

		Adult cull					
		0	0.1	0.3	0.5	0.7	0.9
Juvenile cull	0	+	0	0	0	-	-
	0.1	-	-	-	- -	- -	- -
	0.3	- -	- -	- -	- - -	- - -	- - -
	0.5	- - -	- - -	- - -	- - -	- - -	- - -
	0.7	- - -	- - -	X	X	X	X
	0.9	- - -	X	X	X	X	X

Figure 4-5 Illustrates the predicted effect of a red deer cull. The red area indicated parameter values that result in either no increase or a decrease in total deer population size after fifteen years, the blue area indicates the parameter values resulting in net increase after 15 years. Each graph shows the effects of varying two of the three culling parameters.



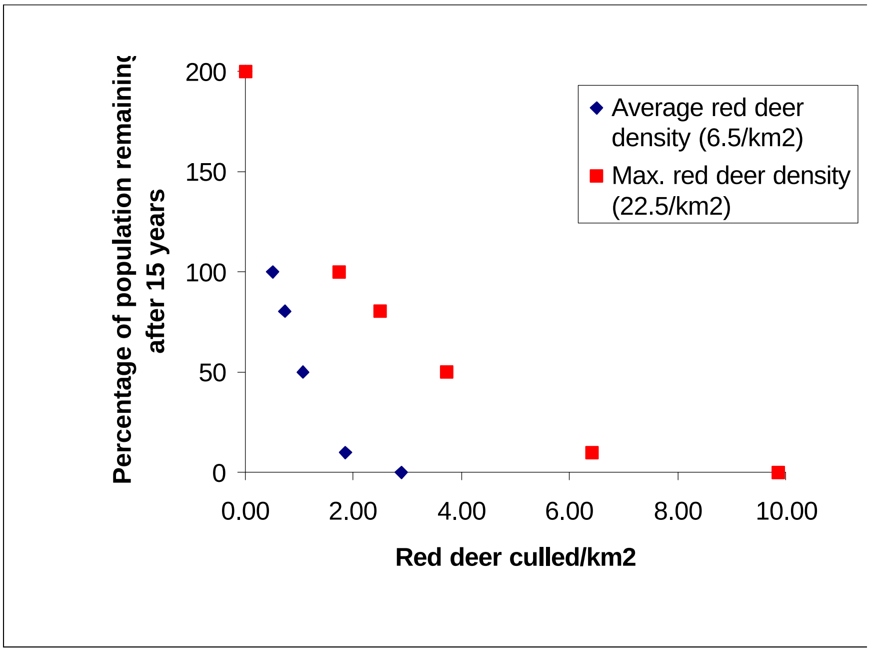




The values given in the above tables and figures only represent the response of the modelled population in terms of proportion of individuals in each age class affected by the levels of each CULL parameter. To compare this output to actual culling rates it is necessary to convert the proportion affected by these parameters into densities affected.

To calculate the density of individuals culled, the stable age class distributions resulting from the matrix model were converted into red deer density per age class by multiplying the proportion of the age class by the density per km^2 of the whole deer population at carrying capacity. Average deer population density were not taken from Table 4-28 as these respresent an average for the whole of Great Britain, and do not represent red deer populations in England. Instead, average deer population densities were taken from Langbein *et al*. (1998). The average red deer population was therefore set at 6.5 / km^2. Maximum densities of 22.5 deer/km^2 were taken from national data presented in Table 4-28. This calculation allowed the production of population response curves for different levels of the CULL parameters. These plots are shown in Figure 4-6.

Figure 4-6 The percentage of the deer population remaining after 15 years, assuming an average carrying capacity of 6.5/km^2 (blue symbols) and a maximum carrying capacity of 22.5/km^2 (red symbols) against different densities of adults, sub-adults and juveniles culled/km^2.



4.3.5. How effective are methods to control mink populations?

4.3.5.a. Approach

In the matrix population modelling approach, anthropogenic control was simulated by culling a fixed proportion of the pre-breeding population. This culling was density independent, and different levels of culling could be applied to different age groups, namely the juveniles (defined as individuals yet to reach breeding age), sub-adults (individuals in the first year of breeding) and adults (individuals who have reached peak reproductive performance).

The aim of this modelling approach is to discover what levels of culling are required to result in a long term population decline in Great Britain

4.3.5.b. Data used

Mink were introduced into Britain in the 1930s, and have been increasing on a national scale ever since. Mink are riparian, and found in association with both still and running water, as well as estuaries and rocky coastlines. Mink density in these regions are dependent largely on the availability of rabbits, their primary food source, but average density is about 0.35 mink/km of river (Macdonald *et al*., 1998). The final estimation of the maximum number of mink that Great Britain could sustain, given no anthropogenic restrictions, is approximately 378,600. See Table 4-28 for more details.

The details on mink life histories are sparse. Much of the available data is derived from studies on captive mink, and naturally these do not reflect the mortality factors experienced by wild populations. Mink have one litter per year, four to six young per litter (Corbet & Harris, 1991), although larger litters are possible (up to 17 in captivity). The percentage of barren females is smaller and the litter size tends to increase in older females (Dunstone, 1993). These data result in the fecundity schedule in Table 4-21.

Table 4-21 Mink fertility schedule

Age	Litter size	Young/mink
0-1	0.00	0.00
1-2	4.00	2.00
2-3	4.50	2.25
3-4	5.00	2.50
4-5	5.50	2.75

An age structure from Hatler (1976, by way of Dustone, 1993) indicates longevity is about 5 years in the wild, although 10 years has been recorded in captive animals (Macdonald, Mace & Rushton, 1998). This age structure allowed the calculation of age-specific survival – the probability of surviving to the next age class (Table 4-22).

Table 4-22 Mink survival data.

Age	Number of mink	% Surviving to next age
0-1	43	0.674
1-2	29	0.552
2-3	16	0.625
3-4	10	0.200
4-5	2	

4.3.5.c. Assumptions made

In addition to the general assumptions of the matrix models, the following initial assumptions arise from the hare models:

As before, it was necessary to estimate age-specific fecundity from the data provided in the literature. Mink productivity was assumed to increase with age, as reported by Dunstone (1993).

The survival data for the matrix models were taken from a study of mink populations on Vancouver Island. It is not known whether these data are representative of British mink populations, or whether that population was subject to hunting. It is likely that the data used overestimates mink mortality, and therefore populations should grow faster than indicated in the following results.

4.3.5.d. Results

The intrinsic rate of increase of the modelled mink population can be calculated from the Leslie matrix by eigen analysis. This results in a value of 1.49, reflecting the high reproductive rate of mink populations – given no form of population regulation (either natural or anthropogenic), this rate of increase indicates that the mink population will increase by 40% every year. This potential for population growth is almost never realised, because the habitat restricts the maximum size of a population and population growth rates decrease as the carrying capacity is reached. However, this intrinsic rate of increase indicates the facility of the population to recover from perturbation. The stable age structure of the modelled population is given in Table 4-23. The results of the partial correlation are given in Table 4-24

Table 4-23 Mink stable age structure

Stage	Age	Proportion
Juveniles	0-1	0.588
Sub-adults	1-2	0.266
Adults	2-5	0.146

.

Table 4-24 Partial correlation results for mink. *** indicates p<0.001; ** indicates p<0.01; * indicates p<0.05; ns indicates p>0.05

	Variable	F value (d.f.=1, 993)	Significance level
life history factors	Fecundity	90.57	***
	Juvenile survival	40.08	***
	Sub-adult survival	3.99	ns
	Adult survival	0.87	ns
Control factors	Juvenile culling	0.15	ns
	Sub-adult culling	38.26	***
	Adult culling	68.99	***

The sensitivity analysis of the matrix model showed that long term population size of the model was significantly affected by applying all CULL parameters on a yearly basis. Mink are so productive that huge populations can result from unperturbed populations. Removal of breeding individuals can substantially affect the long term population trends, as can removal of kits. However, even with large levels of culling, the resultant populations in the long term are still huge (just not as huge as in situations without a cull), so it is necessary to investigate further to find the true effects of culling on the predicted population trends.

The intrinsic rate of increase indicates the facility of the population to recover from perturbation. Because this rate of increase is only achieved at low densities, it is important to examine long term population trends under different culling regimes. When animals are removed from a population, compensation may occur, so that natural mortality or emigration is lower in the following year, allowing the population to grow faster than it did before the cull. By using the population models we can discover the sensitivity of the mink population to man-made perturbation by imposing a yearly cull of different age groups. By examining all levels of cull mortality, we can investigate the responses of a population to this control

The sensitivity analysis reported above indicated that mink population sizes are significantly affected by culling of all age classes, juveniles, sub-adults and adults. These three parameters were therefore varied in tandem, resulting in an estimation of the overall predicted effect on final population size after fifteen years of simulation.

For the comparison of sub-adult mink control and adult mink control, the results can be seen in Figure 4-7, and summarised in Table 4-25. The matrix model suggests that without SUB-ADULT CULL at a level of over 30% per year, no level of ADULT CULL will prevent the modelled mink population from increasing. If control measures can be employed that kill more than 50% of the sub-adults, populations are predicted to decline regardless of ADULT CULL.

Table 4-25 Comparison of sub-adult and adult mink cull. ++ = moderate population increase (initial population doubles in 15 years); + = small population increase (between 1.2 and 2 times initial population in 15 years); 0 = no significant population change; - = small population decrease (between 0.8 and 0.5 of initial population in 15 years); - - = moderate population decrease (between 0.5 and 0.1 of initial population in 15 years); - - - = large population decrease (less than 0.1 of initial population in 15 years); X = population extinction

		Adult cull					
		0	0.1	0.3	0.5	0.7	0.9
Sub-adult cull	0	++	++	++	++	++	++
	0.1	++	++	++	++	++	++
	0.3	++	++	++	++	++	+
	0.5	++	++	++	0	-	- -
	0.7	-	- -	- -	- - -	- - -	- - -
	0.9	- - -	- - -	- - -	- - -	X	X

For the comparison of juvenile mink control and sub-adult mink control, the results can be seen in Figure 4-7, and summarised in Table 4-26. This model prediction suggests that control of modelled mink populations can be achieved effectively through a combination of both JUVENILE CULL and SUB-ADULT CULL parameters. This will reduce recruitment into the reproductive age class of the mink, and prevent positive population trends. A combination of more than 30% JUVENILE CULL and 10% SUB-ADULT CULL will produce a significant decrease in mink numbers after 15 years. A higher culling intensity of either class results in a reduction of the cull needed in the other – if greater than 50% of either juveniles or sub-adults is achieved, the model predicts that no cull will be required of the other age class

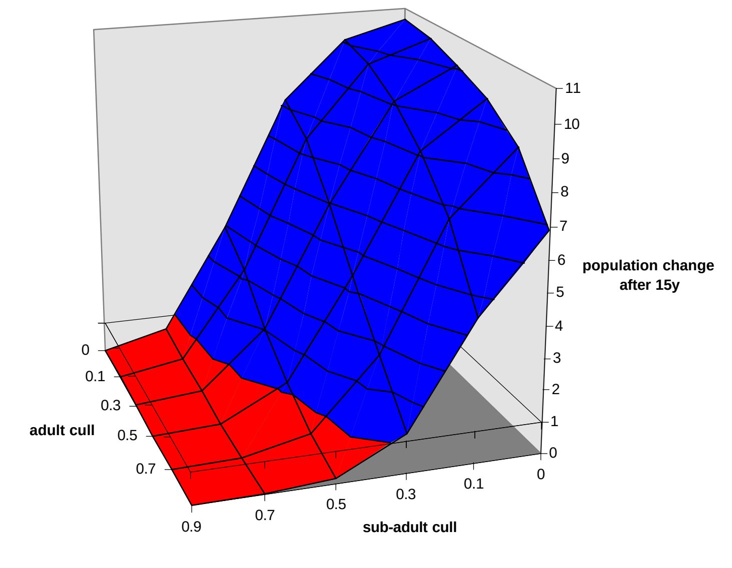
Table 4-26 Comparison of juvenile and sub-adult mink cull. ++ = moderate population increase (initial population doubles in 15 years); + = small population increase (between 1.2 and 2 times initial population in 15 years); 0 = no significant population change; - = small population decrease (between 0.8 and 0.5 of initial population in 15 years); - - = moderate population decrease (between 0.5 and 0.1 of initial population in 15 years); - - - = large population decrease (less than 0.1 of initial population in 15 years); X = population extinction

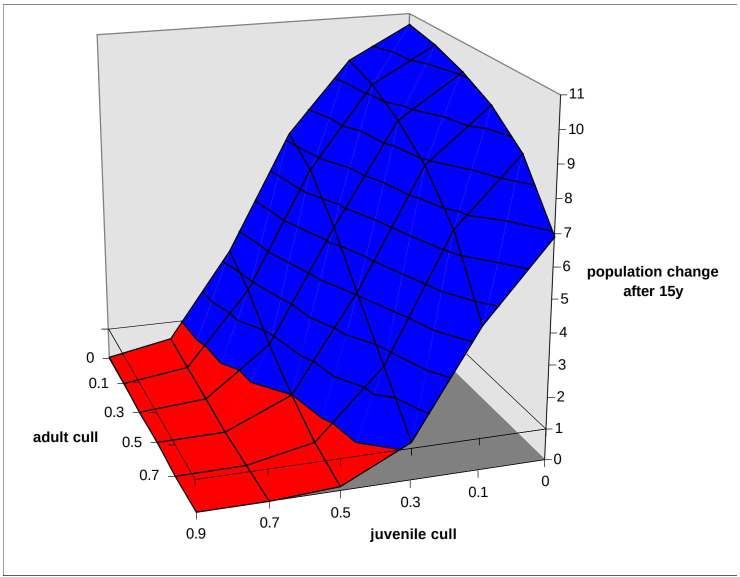
		Sub-adult cull					
		0	0.1	0.3	0.5	0.7	0.9
Juvenile cull	0	++	++	++	++	-	- - -
	0.1	++	++	++	++	- -	- - -
	0.3	++	++	++	0	- - -	- - -
	0.5	++	++	-	- -	- - -	X
	0.7	- -	- -	- - -	- - -	- - -	X
	0.9	- - -	- - -	- - -	X	X	X

For the comparison of juvenile mink control and adult mink control, the results can be seen in Figure 4-7 and summarised in Table 4-27.This simulation compares well with the SUB-ADULT CULL / ADULT CULL scenario. It therefore suggests that the matrix model can support a substantial level of ADULT CULL, as long as JUVENILE CULL and SUB-ADULT CULL is less than 50%.

The values given in these tables and figures only represent the response of the modelled population in terms of proportion of individuals in each age class affected by the levels of each CULL parameter. To compare this output to actual culling rates it is necessary to convert the proportion affected by these parameters into densities affected.

Figure 4-7 Illustrates the predicted effect of a mink cull. The red area indicated parameter values that result in either no increase or a decrease in total mink population size after fifteen years, the blue area indicates the parameter values resulting in net increase after 15 years. Each graph shows the effects of varying two of the three culling parameters.





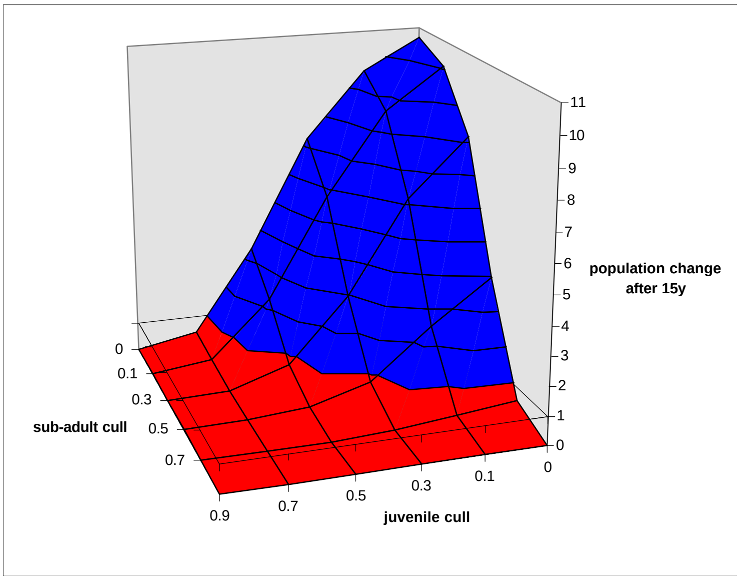
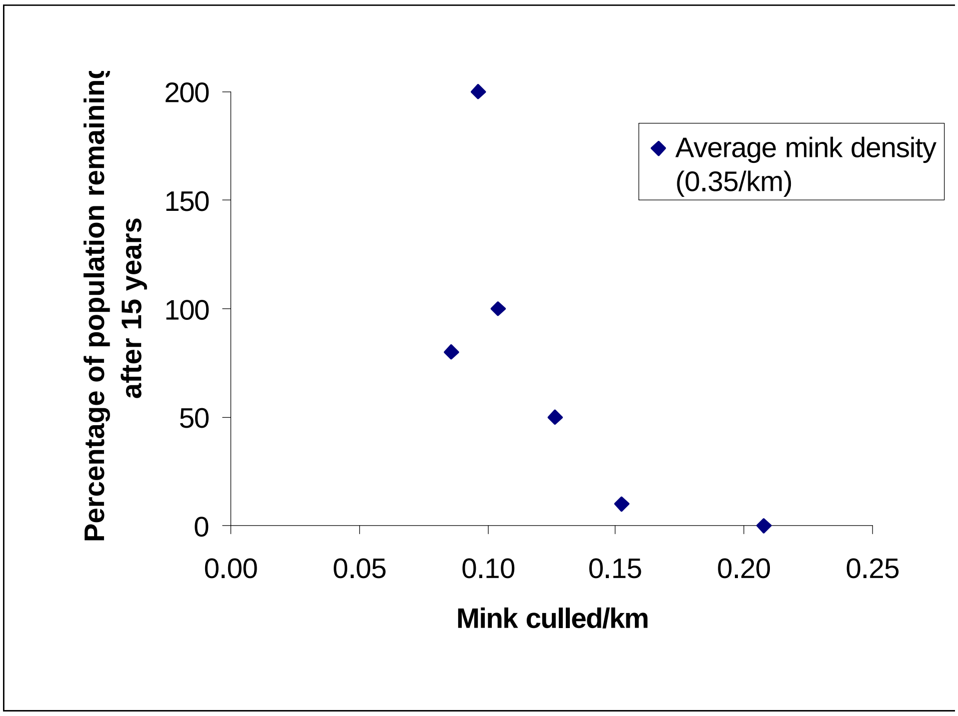


Table 4-27 Comparison of juvenile and adult mink cull. ++ = moderate population increase (initial population doubles in 15 years); + = small population increase (between 1.2 and 2 times initial population in 15 years); 0 = no significant population change; - = small population decrease (between 0.8 and 0.5 of initial population in 15 years); - - = moderate population decrease (between 0.5 and 0.1 of initial population in 15 years); - - - = large population decrease (less than 0.1 of initial population in 15 years); X = population extinction

		Adult cull					
		0	0.1	0.3	0.5	0.7	0.9
Sub-adult cull	0	++	++	++	++	++	++
	0.1	++	++	++	++	++	++
	0.3	++	++	++	++	++	0
	0.5	++	++	+	0	- -	- -
	0.7	- -	- -	- -	- - -	- - -	- - -
	0.9	- - -	- - -	- - -	- - -	X	X

To calculate the density of individuals culled, the stable age class distributions resulting from the matrix model were converted into mink density per age class by multiplying the proportion of the age class by the density per km of river of the whole mink population at carrying capacity. Mink population density were taken from Table 4-28 average mink density is 0.35/km of river. This calculation allowed the production of population response curves for different levels of the CULL parameters. These plots are shown in Figure 4-8.

Figure 4-8 The percentage of the mink population remaining after 15 years, assuming a carrying capacity of 0.35/km river against different densities of adults, sub-adults and juveniles culled/km.



4.3.6. Discussion of matrix modelling results

4.3.6.a. Red Fox

Foxes have relatively high reproductive rates (each pair produce on average 4 cubs per year, resulting an overall 12.2% annual increase at low densities), and thus age structured population models such as this are quite resistant to low additional mortality pressures through control.

Control of fox populations can be achieved through moderate levels of culling of both sub-adults and adults, according to the predictions of this model. If 30% of sub adults and 10% of adults are culled each year, then there will be a greater than ten-fold reduction of fox numbers after fifteen years. If fewer sub-adults are culled, more adults will need to be hunted to maintain the same level of control, and vice versa. Removal of cubs is predicted to have no significant effect on long-term population size unless it is complete (100% of cubs removed).

These data can be translated into estimations of numbers of foxes culled per kilometre squared. To produce a decrease in modelled fox populations after 15 years, more than 0.1 foxes per km^2 had to be removed, assuming that the number of sub-adults and adults removed was proportional to the numbers of sub-adults and adults present. To half the population density over 15 years, more than 0.16 foxes per km^2 had to be removed per year.

The result of the matrix modelling for foxes is dependent upon the initial assumptions. This output is likely to be overestimated, as the data used in formulation of the models was derived from a population under some level of control.

4.3.6.b. Brown hare

Hare have the highest reproductive potential of any of the species in this study (they are the only multi-voltine species, and have the largest litter size; resulting in an overall 50.1% annual increase at low densities). It is because hares are so fecund that controlling their populations is predicted to be difficult unless juvenile mortality is imposed. It is likely that hare populations are probably maintained close to the carrying capacity of the habitat that they live in (confirmed by estimating national carrying capacities), and subject to density-dependent mortality and / or emigration. Removing adults or sub-adults through culling will just release the pressures of density-dependence. Increasing the mortality of the juveniles on the other hand, will remove the larger part of the population, and thus its capacity for rapid growth. This could be achieved, for example, through fertility control – although adult and / or sub-adult control will also be needed unless levels of greater than 50% juvenile mortality (or 50% reduction in fertility) can be achieved. Killing adults or sub-adults, which are the current control measures taken, is predicted to have little effect on the long term population trends – modelled populations continue to increase even if 90% of the individuals of these classes are removed from the population.

These data can be translated into estimations of numbers of hares culled per kilometre squared. To produce a decrease in modelled hare populations after 15 years, more than 3.95 hares per km^2 must be removed, assuming that the number of juveniles, sub-adults and adults removed is proportional to the numbers of juveniles, sub-adults and adults present. To half the population density after 15 years, more than 4.76 hares per km^2 must be removed per year.

The results of the matrix modelling for hares is dependent upon the initial assumptions. The data for survival were derived from a population count, which in turn was based upon the numbers of hares shot. The most likely effect of this assumption is that the juvenile class (leverets) will be under-represented in the bag counts, as they are less likely to be seen or killed by hare hunters. This will naturally affect the output of the model, as survival from the juvenile stage to the sub-adult stage has been estimated to be higher than in reality. The overall effect would be to weaken the effect that juvenile culling would have on hare populations.

The matrix models are updated on a yearly basis, and so do not fully account for multiple litters of hares in a single year. The problem arises in that the individuals born in a year are assumed to have a very low value for fecundity compared to females that are older than a year. Therefore it is assumed that individuals that are 8 months old might produce a single litter. If the model was updated on a smaller interval (monthly, or seasonal, for example), a more accurate breeding pattern could be simulated. Such a model would incur a greater level of error than currently present, as any errors in parameter estimation would be compounded every time the population size was calculated. An additional problem with such a method is that the results would not be directly comparable with the other three matrix models.

The overall effect of a yearly calculation of reproductive output is to provide an optimistic estimate of hare fecundity. As we have assumed that hare density on a regional level is operating close to carrying capacity (ref from Jonathan), density dependent effects on hare fecundity will far outweigh any errors in parameter estimation due to an annually-updated population.

4.3.6.c. Red Deer

Red deer in West Country and most of England are more fecund compared with the Scottish Highlands. Analysis of the matrix model reveals an overall 14.8% annual increase at low densities when using the fecundity data for English deer populations, which compares with a 4.9% annual increase of the Scottish red deer populations.

This intrinsic rate of increase can be misleading, because it is an effect of the long reproductive life of red deer hinds. Fawns are produced one at a time – twins are very rare (less than 1 in 1500 births, Macdonald, Mace & Rushton, 1998). Thus recruitment to red deer populations is low, and while they can withstand adult culling up to 50% without causing a population decline (although anything over 10% is predicted to halt the growth of red deer populations in England), removal of fawns or yearlings in a regular and sustained fashion at levels of 10-30% of those age classes will result in long term population decline. This prediction is typical of ungulates, who have a low recruitment, relative to other mammal groups (such as the Carnivora or Lagomorpha).

These data can be translated into estimations of numbers of deer culled per kilometre squared. To produce a decrease in modelled deer populations after 15 years, more than 0.5 deer per km^2 must be removed, assuming that the number of juveniles, sub-adults and adults removed is proportional to the numbers of juveniles, sub-adults and adults present. To half the population density after 15 years, more than 1.07 deer per km^2 must be removed per year.

The results of the matrix modelling for red deer is dependent upon the initial assumptions. The survival data used in the formation of the matrix models originated from Scottish populations undergoing control in the form of shooting. Therefore, natural survival has been estimated to be lower than in reality, which will result in more deer in the modelled populations than dealt with in the analyses. This would tend to make the output of the culling simulation less sensitive to impose mortality in the form of a cull than natural deer populations are in reality.

4.3.6.d. American Mink

Mink are more productive than foxes (eigen analysis of the matrix results in an overall 39.9% annual increase at low densities), and they continue to reproduce at a high level throughout their lives. Age structured populations models such as this therefore predict that the population will be quite resilient to additional mortality pressures through control.

The population model predicts no population decrease if just adults are culled (up to 90% effectiveness), because of the high fecundity of mink – it doesn't take many mink to escape the cull to repopulate. This is a feature of highly fecund species, that, like the mink, have a relatively high natural juvenile mortality. It is likely that mink populations are probably maintained close to the local carrying capacity of the habitat that they live in, and subject to density-dependent mortality and / or emigration. Removing adults or sub-adults through culling will just release the pressures of density-dependence. Increasing the mortality of the juveniles on the other hand, will remove the larger part of the population, and thus its capacity for rapid growth. The most effective control strategy predicted by the model is by imposing juvenile control – perhaps in the form of fertility control – as well as sub-adult mortality. This effectively limits recruitment to the most fertile life stage, the adult. If adult control is applied in conjunction with juvenile control (a more feasible option), reducing recruitment by half will reduce mink populations in the long term with only a low intensity of adult control.

These data can be translated into estimations of numbers of mink culled per kilometre of riverine habitat. To produce a decrease in modelled mink populations after 15 years, more than 0.1 mink per km must be removed, assuming that the number of juveniles, sub-adults and adults removed is proportional to the numbers of juveniles, sub-adults and adults present. To half the population density after 15 years, more than 0.13 mink per km must be removed per year.

Table 4-28 Estimations of the overall carrying capacity for Great Britain for the four mammal species covered by this inquiry.Coverage of habitat types are taken from the Countryside Information System; estimates of habitat-specific density were derived from Macdonald, Mace & Rushton (1998)

Land Class category	total ha	Red Fox		Brown Hare		Red Deer		American Mink	
		density/ha	estimated number	density/ha	estimated number	density/ha	estimated number	density/ha	estimated number
urban	260300	0.015	3905	0	0	0	0	0	0
suburban	1317000	0.015	19755	0	0	0	0	0	0
tilled land	1531000	0.025	38275	0.144	220464	0	0	0	0
managed grassland	6567000	0.025	164175	0.098	643566	0	0	0	0
rough grass	430700	0.025	10768	0.098	42209	0.15	64605	0	0
bracken	360300	0.025	9008	0	0	0.15	54045	0	0
heath grass	2020000	0.025	50500	0.072	145440	0.15	303000	0	0
open shrub heath	2787000	0.025	69675	0	0	0.15	418050	0	0
dense shrub heath	722000	0.025	18050	0	0	0.15	108300	0	0
bog	430900	0.00025	108	0	0	0	0	0	0
deciduous woodland	1233000	0.025	30825	0	0	0.225	277425	0	0
coniferous woodland	772200	0.025	19305	0	0	0.225	173745	0	0
inland bare	256600	0.00025	64	0	0	0	0	0	0
saltmarsh	38940	0.00025	10	0	0	0	0	0	0
coastal bare	142100	0.00025	36	0	0	0	0	0.35	49735
inland water	171400	0	0	0	0	0	0	0.35	59990
sea/estuary	768300	0	0	0	0	0	0	0.35	268905
Max population size			434,457		1,051,679		1,399,170		378,630

The results of the matrix modelling for mink is dependent upon the initial assumptions. The survival data used, like in all of these matrix models, is perhaps not representative of an English population of mink that is not subjected to any anthropogenic mortality; however, the model assumes that this is the case. Thus the modelled mink life histories are likely to under-predicted mink population sizes, and thus the population is shown to be more sensitive to imposed culling than the model should be.

4.4. What do individual-based models predict about the effectiveness of different fox and mink population control methods?

4.4.1. How effective are methods to control fox populations?

4.4.1.a. Approach

The model was a spatially explicit individual-based population dynamics model that simulated the dynamics of fox social groups in terms of the survival and reproduction of individuals and their dispersal movements within the landscape. The spatial unit modelled was the fox social group, which was assumed to occupy an exclusive territory centred on one or more breeding earths, and to consist of a male, one breeding female, and a variable number of non-breeding females. The model was stage-structured (Caswell, 1989), in so far as discrete life-history stages (adults, subordinate adults and juveniles) were recognised, but the mortality, fecundity and dispersal rates appropriate to each life-history stage were applied at the level of the individual within each social group.

4.4.1.a.i. Model structure

Two-dimensional digital ‘landscapes’ were created, across which earths were distributed randomly at densities determined by input parameters. The grid co-ordinates of each earth were used as the spatial reference point by which to model the dynamics of individuals and social groups. At the start of each model run, every earth and associated home-range was assigned a maximum number of adults and subordinate adults. Each social group had separate dynamics, which interacted only through the process of dispersal. For each social group, changes in membership took place through: gains from breeding and immigration of subordinate adults from other social groups; and losses due to non-culling mortality, emigration, and culling.

It was not feasible to represent with detailed realism the many different methods of fox culling described in Chapter 6. Instead, the approach was to model a number of processes believed to encapsulate essential characteristics of the main types of culling method. These were applied to the population in a pre-defined sequence to mimic their seasonal nature. To distinguish these from the reality described elsewhere in this report, modelled processes are labelled in capitals as HUNTING, EARTH CULLING, SHOOTING, and CONTRACEPTION. These are described in operational sequence, beginning in spring:

- CONTRACEPTION
  This was assumed to occur in spring. Social groups in which contraceptive treatment took place did not produce offspring. The efficacy of this technique (proportion of female foxes treated) was varied as a model input. The effect of contraception was assumed to vanish by the end of 12 months, so that if the animal was still alive it was normally reproductive unless treated again. [

- (REPRODUCTION)
  This process was imposed just after reproduction. As there is evidence that animals other than males are Reproduction was assumed to occur once a year, with only one litter per social group, produced by the dominant female. The number of cubs in each litter was estimated by drawing deviates from a Poisson distribution with the mean number varied as a model input (following Akçakaya *et al.*, 1995; Rushton *et al.,* 2000).

- (NON-CULLING MORTALITY)
  The model included mortality processes other than culling mortality. Three such mortality rates were

used: adult mortality, subordinate adult mortality, and juvenile mortality. Mortality in adult foxes was assumed to occur after breeding. In juveniles and subordinate adults mortality occurred as they were recruited into the subordinate adult and adult populations respectively. Mortality was modelled at the level of the individual for all types. All mortality rates were varied as model inputs.

- CULLING AT THE EARTH
  This process was imposed just after reproduction. As there is evidence that animals other than males are preferentially culled by this practice, so mortality was imposed on the other age/sex classes. The likelihood of a social group being culled successfully by this method was derived stochastically varied as an input. Where an adult female was culled it was assumed that her cubs would also die.

- SHOOTING
  This process occurred in autumn prior to dispersal, and was imposed as a mortality factor on subordinate adults and adults. The chance of shooting success (foxes shot per excursion) and the number of shooting excursions made were varied as an input. Shooting was assumed to be a locally applied culling method, affecting only a proportion of fox social groups each year. However, it was not feasible in this relatively simple model to fix this culling effort to address social groups at the same earths in each year.

- HUNTING
  This process was imposed on the population during autumn and winter. In reality, hunts focus on a small section of their total country on each hunt day, and thus address only a small number of fox social groups. We assumed that meets are evenly distributed so that a typical hunt covers its entire hunt country once per season. In the model, this roving process was simulated by estimating the number of social groups that would be visited in one days hunting and using this to estimate the likelihood that an individual social group would be visited. This mean that a small number of potential fox home ranges were 'visited' on each hunt day. Within the social groups 'visited', the mortality among individuals due to hunting was also assessed stochastically, based on a predefined chance of hunting success. All age and sex classes were assumed to be equally at risk from hunting. The number of hunt days per season and the chance of success (foxes killed per hunt day) were varied as inputs.

- (DISPERSAL)
  In the model, dispersal between social groups occurred once a year in autumn. All subordinate adults that could not find space in their natal social groups dispersed. Dispersing foxes moved to social groups that were close enough and not already full. Dispersing foxes moved on if the social group was already fully occupied, and died if they did not join a social group within range of their maximum dispersal distance. In this way, the population could never grow above the carrying capacity defined by the starting population. The maximum distance animals could disperse was varied as a model input.

The full sequence of model processes including births and dispersal was reiterated for 15 cycles, each cycle representing 1 year. The 15 year period was chosen arbitrarily to investigate the likely medium term impacts of culling on fox population, a compromise between ensuring that the period investigated included several generations of foxes and avoiding the unrealism associated with running the model with the same sets of input parameters over extended time periods.

4.4.1.a.ii. Parameterisation

Three 'landscapes', referred to as WALES, MIDLANDS, and NORFOLK, were created. Input values for earth density and initial fox density (size and number of social groups) were attached to each landscape based on estimates by Heydon *et al.* (2000) for three large regions of England and Wales (mid-Wales, east Midlands, west Norfolk). Thus each modelled landscape was seeded with a fox population reflecting known real-world scenarios. Unlike their real-world counterparts, starting densities were assumed also to be the carrying capacity for each landscape. Culling parameters were varied through an identical range for all landscapes (see Sensitivity analysis below). In this respect, too, models in the three landscapes differed from their real-world counterparts, where the relative use and success of individual culling methods varied with a regional flavour (Heydon & Reynolds, 2000a).

Each culling method in the model had parameters representing effort (e.g. number of days HUNTING) and success (foxes killed per day's HUNTING). The likely range of values each parameter might take was estimated from such data as were available. The lack of suitable data to define these ranges with much accuracy was a problem, discussed further below.

Dispersal was difficult to parameterise. Foxes are considered good dispersers, and dispersal distances up to 394 km have been recorded. In field studies, measured dispersal distances (e.g. between 3 and 43km or 2 and 37km for males and females respectively are usually measured by recovery of tagged animals. Inevitably, many or most recoveries are of dead animals, with the result that estimates of dispersal distances are greatly influenced by mortality risk for dispersing animals. The probability of successful dispersal (ending in settlement in vacant territorial space) must also depend on territory size and population density. Finally, the probability that an individual of any sex/age class will disperse is related to density and other variables at the place of origin (Woolard & Harris, 1980). The maximum linear distance a fox could disperse was varied as an input parameter selected from the range of distances observed in the field.

Non-culling mortality was also difficult to parameterise. Fox life-history patterns are typified by high juvenile and subordinate adult mortality and lower adult mortality (Macdonald *et al.*, 1998; Harris, 1987; Heydon & Reynolds, 2000b). Non-culling mortality was based on the estimates derived from a study of mortality in Somerset (Reynolds *et al.*). In this, a 14% non-culling mortality was recorded. Ninety-five percent confidence limits on this estimated proportion were used to provide upper and lower bounds as inputs for the model. The range used (5-32% mortality per year) was used for each age classes, with a different value from this range drawn for each age class in each run of the model.

4.4.1.a.iii.Sensitivity analysis

The sensitivity of the model to life history and culling parameters was investigated by analysing how population size and population persistence varied in response to variations in model inputs, for each landscape. Latin Hypercube Sampling (LHS) following the methods of Vose (1996) was used to select input parameters for the model from the known or estimated ranges of the different variables in the model. LHS uses stratified sampling without replacement to select suites of input parameters from known distributions of those variables. In practical terms, the probability distribution for each variable is split into ‘n’ intervals of equal probability, where ‘n’ is the number of sets/suites of input variables selected. The creation of input variables for a model run proceeds as follows. A random number is used to select an interval and a further random number to determine the position within the interval and hence the value of the input variable for inclusion in the set. Once selected, an interval cannot be used again. In effect, each interval may be used once, but once only, in combination with values of the other variables selected in the same way. The aim was to provide a range of input values for each variable that could potentially occur under field conditions. In other words the model would be run a sufficiently large number of times to encompass <u>all</u> of the potential range of conditions that occur naturally rather than simply worst and best case scenarios (*sensu* Bart, 1995). Eleven parameters were varied in this way:

- Maximum linear dispersal distance
- Mean productivity of females (cubs per litter).
- Adult non-culling mortality (percentage per year).
- Subordinate adult non-culling mortality (percentage per year).
- Juvenile non-culling mortality (percentage per year).
- HUNTING success (percent chance of successful kill).
- Number of HUNTING days per year.
- CULLING AT THE EARTH success (percentage of earths where culling was practised).
- Social groups addressed for shooting (percentage of groups).
- SHOOTING success (foxes shot per excursion).
- Efficiency of CONTRACEPTION (percentage of female foxes successfully treated per year).

In most cases insufficient data were available to identify the distribution function for each parameter. A uniform distribution was assumed for each variable with upper and lower limits derived from the literature. The range of each parameter from which samples were taken is given in Table 4-29. The use of uniform distributions for all of the input variables leads to the selection of values for parameters that are near the extremes of their distributions more frequently than would be expected in reality. However, it also ensures that all potential values (within the known range of observed behaviours for each variable) are sampled. 200 sets of the input parameters were selected and the model was then run for 15 years for each set. The starter population for each run consisted of populations of animals present at carrying capacity in all social groups in the landscape. For each model run, the total number of foxes controlled, whether or not the population had gone extinct, and the total population of foxes present in the landscape were output at the end of the 15 years. These data were then correlated with the input variables and partial correlation coefficients were calculated to assess the impact of the individual life history parameters on the dynamics of the fox population in the landscape as simulated by the model.

Table 4-29 Values of life history and culling parameters and their sources used in the individual-based models

Variable		Min	Max	Source
Maximum dispersal distance (km)		1.00	38.00	Macdonald (1984)
Mean fecundity (cubs per breeding female)		4.00	6.00	Macdonald (1984)
Juvenile non-culling mortality(%)		5.0	32.0	Heydon & Reynolds (2000b)
Adult non-culling mortality (%)		5.00	32.0	Heydon & Reynolds (2000b)
Subordinate adult non-culling mortality (%)		5.0	32.0	Heydon & Reynolds (2000b)
HUNTING PARAMETERS	HUNTING mortality percent chance of a successful kill	80.0	100.0	Johnson & Macdonald (1996)
	Number of HUNTING sessions	32	125	The Game Conservancy Trust, unpubl.[10]
CULLING AT EARTH PARAMETERS	CULLING AT THE EARTH % of total earths visited	1.0	30.0	The Game Conservancy Trust, unpubl. [1]
SHOOTING PARAMETERS	% Of social groups subject to SHOOTING	10.0	80.0	The Game Conservancy Trust, unpubl. [1]
	Number of SHOOTING excursions	5	81	The Game Conservancy Trust, unpubl. [1]
	Number of foxes killed per SHOOTING excursion	0.22	2.00	The Game Conservancy Trust, unpubl. [1]
CONTRACEPTION PARAMETERS	CONTRACEPTION success (% of females rendered infertile)	10.0	80.0	No data

4.4.1.b. Results

Assessing the impacts of culling regimes on the populations of foxes in the three landscapes

Six culling scenarios were assessed in the model. These were:

i) HUNTING

ii) CULLING AT THE EARTH

iii) SHOOTING

iv) HUNTING + CULLING AT THE EARTH

[10] based on daily records of fox culling effort and success from 60 professional gamekeepers, 1992-93

v) CONTRACEPTION

vi) HUNTING + CULLING AT THE EARTH + SHOOTING

The effects of these culling regimes and the underlying fox life history parameters on model outputs were assessed using partial correlation coefficients following Vose (1996) and Rushton *et al* (2000). The partial correlation coefficient measures the extent to which each input variable contributes to the model outputs. Three model outputs were considered:

- The number of fox social groups in the landscape at year 15
- The total population of foxes left in the landscape at year 15
- The total number of foxes culled during year 15

F values assessing the significance of each partial correlation coefficient under each modelling scenario were calculated. The F values are used to assess the extent and significance with which the different model inputs contribute to the model output.

F values for the input variables for each of each of the three model outputs for each of the three landscape scenarios are given in Tables 1 to 3 respectively. Entries highlighted indicate that the individual variable concerned had a significant impact on the relevant model outputs at P< 0.05

4.4.1.b.i. The NORFOLK landscape (see Table 4-30)

In this landscape, the maximum distance over which foxes could disperse was always a significant partial correlate with the total number of social groups, and the total population of foxes in the landscape in all culling scenarios except CULLING AT THE EARTH and when all controls were applied in combination. In the case of the HUNTING, CULLING AT THE EARTH and CONTRACEPTION scenarios, dispersal was the most significant partial correlate. This result suggests that the dispersal ability of foxes to other earths in the landscape is a major factor determining the numbers of animals and social groups present. Considering the individual culling scenarios in turn, hunting by itself was not a significant determinant of total population size or the number of social groups present in the landscape. For the CULLING AT THE EARTH scenario the percentage of earths visited was a significant predictor of fox numbers and social groups. Similarly, the percentage of social groups where SHOOTING was practised was a significant predictor of both total population and numbers of social groups. When HUNTING and CULLING AT THE EARTH were used in combination then the number of hunting days became a significant predictor of population size and the number of social groups. When all three forms of control were used in combination, the percentage of earths where CULLING AT THE EARTH took place, the percentage of social groups at which SHOOTING occurred, the number of SHOOTING excursions and the kills per SHOOTING excursion were major predictors of both total population size and the number of social groups. The level of CONTRACEPTION was not a significant predictor of total population size or number of social groups.

Turning to the number of foxes killed, the level of control applied was, not unexpectedly, a major predictor of the number of animals killed.

Considering the individual life history parameters, only mean fecundity and young mortality were significant predictors of total population size and number of social groups and only under two scenarios, that involving shooting at the earth and that where all culling methods were used in combination. This result indicates that when reproduction is low and young mortality is high the impact of shooting mortality at the earth would be greatest.

4.4.1.b.ii. The MIDLANDS landscape (see Table 4-31)

In this landscape, the results were very similar to those in the Norfolk landscape except in so far as: a) the dispersal ability variable, whilst significant, was much less so than in the first landscape; b) the life history variables of mean fecundity, and adult and juvenile mortality appeared to be more important in determining total population size.

4.4.1.b.iii.The WALES landscape (see Table 4-32)

In this landscape, the results were similar to those in the Norfolk landscape.

Table 4-30 Norfolk landscape. F values detailing the significance of partial correlation coefficients relating three model outputs (predicted number of fox social groups, predicted total population of foxes and predicted number of foxes killed) to different input parameters in each culling scenario in the Norfolk landscape. In all cases F values greater than 5.15 indicate that the variable has a significant impact on the output at P<0.05 (identified in bold).

		Hunting	Killing at earth	Shooting at earth	Hunting and killing at earth	Fertility Control	All control mechanisms
Number of social groups in landscape	Maximum dispersal distance	20.24	32.28	1.23	14.84	15.41	2.72
	Mean fecundity	0	0.2	6.63	3.12	0.03	9.84
	Young mortality	0.03	0	6.01	0.93	0.12	9.66
	Adult mortality	0.02	0.33	0.96	2.33	0	3.95
	sub adult mortality	0.34	0.02	0.04	0.59	1.14	2.29
	Hunting mortality	1.09			0.49		1.05
	Killing at earth %		11.07		32.88		119.9
	Earths shot %			602.5		0.36	417.3
	Number of hunting days	0.03			5.59		2.77
	Number of shooting days			10.46			12.74
	Kills per day shooting			3.86			22.66
Total population size in landscape	Maximum dispersal distance	31.88	33.33	1.16	14.77	10.28	2.51
	Mean fecundity	0.24	0.4	5.9	2.64	1.87	9.86
	Young mortality	0.01	0.1	5.92	0.65	0.36	9.13
	Adult mortality	1.21	0.02	1.09	2.28	2.35	3.04
	sub adult mortality	0.03	0.15	0.02	0.66	0.42	2.15
	Hunting mortality %	0.32			0.35		1.11
	Killing at earth %		8.68		30.1		120.0
	Earths shot %			580.8		0.56	446.0
	Number of hunting days	0.11			5.59		3.11
	Number of shooting days			9.36			13.14
	Kills per day shooting			3.5			22.47
Number of foxes killed	Maximum dispersal distance	0.18	0.07	0.98	4.51	n/a	0.29
	Mean fecundity	0.34	55.51	0.75	42.65	n/a	18.01
	Young mortality	0.24	0.05	0	0.2	n/a	3.00
	Adult mortality	0	0.76	2.99	1.13	n/a	0.50
	sub adult mortality	5.51	3.25	0.43	0.09	n/a	0.98
	Hunting mortality %	0.2			0.17	n/a	0.35
	Killing at earth %		2586.1		959.9	n/a	0.80
	Earths shot %			2.89		n/a	210.9
	Number of hunting days	3109			38.31	n/a	0.52
	Number of shooting days			28.1	0.15	n/a	16.01
	Kill per day shooting			25.88	0.04	n/a	24.44

Table 4-31 East-Midland landscape. F values detailing the significance of partial correlation coefficients relating three model outputs (predicted number of fox social groups, predicted total population of foxes and predicted number of foxes killed) to different input parameters in each culling scenario in the East-midland landscape. In all cases F values greater than 5.15 indicate that the variable has a significant impact on the output at P<0.05 (identified in bold).

		Hunting	Killing at earth	Shooting at earth	Hunting and killing at earth	Fertility Control	All control mechanisms

Number of social groups in landscape	Maximum dispersal distance	8.47	9.07	7.35	12.57	5.87	0.08
	Mean fecundity	0.22	0.92	10.35	0.66	0.88	12.23
	Young mortality	0.06	0.66	1.21	0.49	0.9	9.91
	Adult mortality	0.14	0.64	0	0.49	0.75	1.11
	sub adult mortality	0.89	0.38	0.8	0.19	0.29	3.38
	Hunting mortality	1.75			1.1		1.73
	Killing at earth %		4.49		5.36		111.98
	Earths shot %			692.09		0.83	580.06
	Number of hunting days	0.04			0.02		0
	Number of shooting days			13.99			11.23
	Kills per day shooting			4.18			21.14
Total population size in landscape	Maximum dispersal distance	17.54	9.69	0.79	12.88	10.46	0.08
	Mean fecundity	16.98	1.54	6.35	1.14	14.74	11.44
	Young mortality	14.09	1.09	9.61	0.92	13.06	9.1
	Adult mortality	16.21	0.07	1.13	0.08	21.31	0.72
	sub adult mortality	4.5	0.17	0	0	5.78	2.89
	Hunting mortality %	0.04			1.08		1.75
	Killing at earth %		1.83		2.43		110.18
	Earths shot %			683.2		0.45	598.42
	Number of hunting days	0.27			0		0
	Number of shooting days			13.56			11.63
	Kills per day shooting			3.58			20.47
Number of foxes killed	Maximum dispersal distance	0.52	1.25	1.1	0.02	n/a	0.03
	Mean fecundity	0.26	434.2	0.84	326.3	n/a	16.25
	Young mortality	0.6	0.04	0	2.33	n/a	1.59
	Adult mortality	2.15	1.33	2.94	0.3	n/a	0.4
	sub adult mortality	4.16	0.34	0.44	0.21	n/a	0.82
	Hunting mortality %	2.64	0.01		1.24	n/a	1.42
	Killing at earth %		13555		11824.4	n/a	5.98
	Earths shot %			2.88		n/a	211.75
	Number of hunting days	1444.7			5.75	n/a	0.94
	Number of shooting days			28.87	0.37	n/a	12.38
	Kill per day shooting			26.46	1.6	n/a	16.06

Table 4-32 Welsh landscape. F values detailing the significance of partial correlation coefficients relating three model outputs (predicted number of fox social groups, predicted total population of foxes and predicted number of foxes killed) to different input parameters in each culling scenario in the Welsh landscape. In all cases F values greater than 5.15 indicate that the variable has a significant impact on the output at P<0.05 (identified in bold).

		Hunting	Killing at earth	Shooting at earth	Hunting and killing at earth	Fertility Control	All control mechanisms
Nu	Maximum dispersal distance	16.3	21.03	0	20.45	13.21	0.59
	Mean fecundity	0	0.46	8.35	0.16	0	14.28
	Young mortality	0	0.21	8.54	0.4	0.46	8.79
	Adult mortality	0	0.19	1.36	0.45	0.06	2.35
	sub adult mortality	0.57	0	0.03	0.01	0	2.95
	Hunting mortality	1.27			0.53		2.03
	Killing at earth %		8.43		10.91		118.1
	Earths shot %			669.7		0.19	534.4

	Number of hunting days	0.07			0.05		0.58
	Number of shooting days			14.46			10.68
	Kills per day shooting			2.61			23.21
Total population size in landscape	Maximum dispersal distance	12.85	20.14	0.01	20.56	13.08	0.46
	Mean fecundity	0.52	0.67	7.37	0.26	2.36	13.47
	Young mortality	1.42	0.36	8.36	0.51	2.08	8.61
	Adult mortality	6.53	0.04	1.46	0.23	3.29	1.66
	sub adult mortality	1.38	0.13	0.02	0	0.15	2.77
	Hunting mortality %	0.4			0.5		2.15
	Killing at earth %		5.35		8.59		117.12
	Earths shot %			657.3		1.81	561.9
	Number of hunting days	0			0.07		0.65
	Number of shooting days			13.98			10.76
	Kills per day shooting			2.26			22.83
Number of foxes killed	Maximum dispersal distance	0.18	0.8	0.95	0.2	n/a	0.11
	Mean fecundity	1.39	180.9	0.81	151.9	n/a	20.33
	Young mortality	0.45	0.16	0	4.05	n/a	2.45
	Adult mortality	4.94	2.52	2.83	0.6	n/a	0.17
	sub adult mortality	0.72	0.07	0.52	0.82	n/a	0.53
	Hunting mortality %	0.2			0.49	n/a	2.71
	Killing at earth %		5937		4642.1	n/a	1.59
	Earths shot %			2.88		n/a	209
	Number of hunting days	2289.4			34.83	n/a	0.64
	Number of shooting days			28.89	0	n/a	11.21
	Kill per day shooting			26.74	0.01	n/a	16.9

4.4.1.c. Discussion

The partial correlation coefficients (Table 4-30, Table 4-31, and Table 4-32) indicate that the dispersal ability of the foxes has a major impact on the extent to which population control measures can influence the abundance of fox populations, particularly at low fox densities. This over-arching influence of dispersal is not unexpected. The modelled SHOOTING and CULLING AT THE EARTH processes affected only a proportion of social groups each year, imitating the spatially fixed nature of local culling efforts. The tendency for foxes to even out their own density by dispersal would determine their ‘availability’ to local culling efforts: culling success is dependent on the opportunity to cull. By contrast, the modelled HUNTING process was assumed to distribute its effort randomly around the landscape, rather than concentrating effort in relation to fox density. Real-world hunt practice probably lies somewhere between this ‘blind’ roving behaviour and a rigid system of evenly distributed meet points: meets do tend to be concentrated where foxes are numerous, while venues for cub-hunting meets, by-days and lambing calls can be arranged at short notice to suit local need.

The results also suggest that CULLING AT THE EARTH and SHOOTING had more impact on fox numbers than did HUNTING in all three landscapes, and we must ask what characteristics of the modelled processes cause this difference? Is it the intrinsic nature of each form of culling (represented by the timing and mode of operation of each process within the model), or is it the range of culling intensities, as determined by starting parameters? In the first instance there are good reasons why HUNTING would not be expected to have as large an impact as SHOOTING because of the way in which the processes were modelled. The number of social groups affected by HUNTING was dependent on the number of days spent hunting. So when this was low, the chance that an individual social group was drawn during the course of a year was also low, over and above any inherent success rate in killing the animal (which was also varied as an input). In the SHOOTING scenario, in contrast, the minimum number of shooting excursion made was 5, with a minimum number of foxes killed of 0.22, meaning that at the very least one fox was killed in any social group where SHOOTING was undertaken in any year. In the majority of model runs more than one animal would be killed and for any runs where the number of excursions was greater than 25 all animals in the social group would be killed. This means that the chances of an individual fox being killed from any group by HUNTING were correspondingly lower than that for an individual subjected to SHOOTING. Is this model of HUNTING realistic ? The best way to assess this is to compare the kill rate per hunting day in the model with that actually observed in real hunts. Analysis of 208 registered hunts by Johnson and Macdonald (**) indicates that the mean number of total foxes killed was 75.2 per year (SE 7.7) over a mean of 71.0 hunting days (SE 3.5) or a kill rate of 1.05 foxes per session. In the model, equivalent kill rates for the three landscape scenarios were:

- NORFOLK 79.9 (SE 1.9) killed for 77.9 sessions per year (range 32-125); i.e. a kill rate of 1.02 foxes per session;
- the MIDLANDS 85.7 killed (SE 2.0) over 77.9 days (range 32-125) i.e. a kill rate of 1.10 foxes per session;
- WALES 80.4 killed (SE 1.9) over 77.9 days (range 32-125) i.e. a kill rate of 1.03 foxes per session;

These results suggest that the kill rate caused by HUNTING was similar in the model to that observed for all registered hunts taken as a whole. Similar analyses for the SHOOTING scenario are difficult to undertake because the data are not available.

Whilst the model appears realistic, the results do not match with the empirical understanding obtained by Heydon & Reynolds (2000b) in real-world regions in mid-Wales, east Midlands and west Norfolk. In both mid-Wales and west Norfolk, the total cull taken by all methods was deemed effective in controlling the fox population, but whereas in Norfolk 73% was taken using rifles and snares, in mid-Wales 73% of the cull was taken using dogs. There are two important differences between the model and reality here. First, the modelled HUNTING process best describes the more formal practices of hunting with hounds (whether on foot or with horses) typical of the east-Midlands landscape, whereas the characteristics of gun-packs and terrier work as practised in mid-Wales are not represented at all. Thus, terrier work in mid-Wales shares seasonal characteristics with both SHOOTING, and CULLING AT THE EARTH, but has the roving characteristics of the HUNTING process. It is likely that this latter form of culling is a more effective method of fox control and should be investigated further. Second, the same range of input parameters were used in all three modelled landscapes, with no attempt to mimic observed regional values. Thus, the highest levels of shooting intensity – empirically, found in west Norfolk – were also considered feasible in the WALES landscape. In reality, this is probably not the case. Norfolk has a uniquely high density of professional gamekeepers, financially supported by large estates, arable agriculture and the general suitability of this region for game management. Even leaving aside the unsuitability of the mid-Wales terrain for shooting with a rifle, it is unlikely that the sheep farming economy could ever support professional fox culling at this intensity.

Thus it seems likely that disparity between the model and reality reflects approximations both in mimicking real-world culling processes and in parameterisation. These approximations in turn reflect the imperfect state of our knowledge in the year 2000. However, it is also justified to view the models as reflecting the impacts of different culling methods - practised with the intensities observed in field studies - on fox populations at realistic densities, assuming no constraints on the suitability of different methods for different landscapes. Under this view, it is clear that SHOOTING (as practised in source studies) can be practised with sufficient intensity to have a very significant impact on fox numbers, whereas HUNTING (as practised in the East-Midland source study) does not. In drawing this conclusion it is important to remember that the source data for HUNTING was formalised hunting as practised by MFHA-registered hunts, all of which exhibit moderation ethics in one way or another (e.g. by observing a closed season); also that the source data for SHOOTING included both mid-Wales and west Norfolk which probably represent extreme values.

Besides suggesting the relative impact of the different modelled culling processes, the model also outputs the scale of increase or decline of each population. The mean total number of social groups and the mean total population of foxes output by the model in each of the three study landscapes at the end of 15 years of each culling scenario are given in Table 4-33. This table shows the means for all of the range of parameter values used in the models rather than any particular suite of life history and control parameters that might pertain in the field in any given year. Thus these data can be used only to illustrate the relative impacts of the different control strategies in each landscape over all possible scenarios rather than quantify the impacts of culling *per se*.

Table 4-33 The mean predicted numbers of fox social groups and total population sizes under five culling strategies. Note total number of earths in each landscape was 200 for Norfolk, 1400 for the East-Midlands and 500 for Welsh study areas..

		Hunting	Killing at earth	Shooting at earth	Hunting and killing at earth	Fertility Control	All control mechanisms
Predicted number of social groups in landscape	Norfolk	199.8	198.08	144.2	193	199.93	69.3
	East Midlands	1399.9	1396.6	1024.5	1395.8	1400	597.3
	Wales	499.2	497.2	363.5	495.8	499.9	197.3
Predicted total population size in landscape	Norfolk	799.6	795.4	588.6	776.4	800.06	287.1
	East Midlands	5601.6	5601	4174	5598.2	5602	2458.8
	Wales	1999.3	1995.2	1483.3	1989.1	1999.4	817.1

The most obvious feature of these results is that HUNTING and CULLING AT THE EARTH by themselves had little impact on the total population of foxes or the number of social groups in any of the three landscapes. In combination these two culling processes reduced the final number of social groups by around 3.5% in 15 years in the Norfolk landscape and under 1% in 15 years in the other two landscapes. In contrast, shooting decreased the number of social groups by 27% in 15 years in all landscapes and all controls used together by 65% in Norfolk, 57% in the East-Midlands and 60% in Wales. The impacts on the total population size were similar to those for the social groups. CONTRACEPTION, in common with hunting, did not have any effects on population size or the number of social groups. Research in Australia (Marks *et al.*, 1996) indicates that fertility control in the form of abortifacients, can reduce fecundity in fox populations. It is clear from this study that the extent to which abortifacients could control fox populations will depend on the extent to which dispersal of animals from non-treated areas will occur.

To evaluate the extent to which culling could be used to control fox populations the likelihood of culling reducing fox numbers below the carrying capacity of each landscape was investigated using logistic regression. In these analyses, the final population after year 15 of each model run was compared to the initial starting population in each landscapes. If the final population was less than the starting population at the beginning of the 15 year run, then this was scored as 1; if the population was lower then it was scored as 0. The categorical 1/0 variables produced formed a response variate in logistic regression, with the individual culling variables used as predictors. (This analysis was undertaken for the shooting scenario and not hunting with dogs as the latter was not shown to be a significant predictor of fox numbers.)

Plots of the probability that the fox population will decline as a result of the imposed SHOOTING regimes are shown for each study landscape in **Figure 4-9**. The most obvious feature is that for all three landscapes the probability that populations would be reduced by culling increased with the number of excursions into the landscape and the proportion of social groups visited in the landscape. The fox population response was greatest to variations in the proportion of earths visited rather than the number of days spent shooting. This latter variable was more important in the MIDLANDS and WALES landscapes.

It should be noted that the accuracy with which population trends were predicted depends heavily both on model processes and on input parameters. Thus errors in parameterisation of non-culling mortality could seriously affect the predictions of the model.

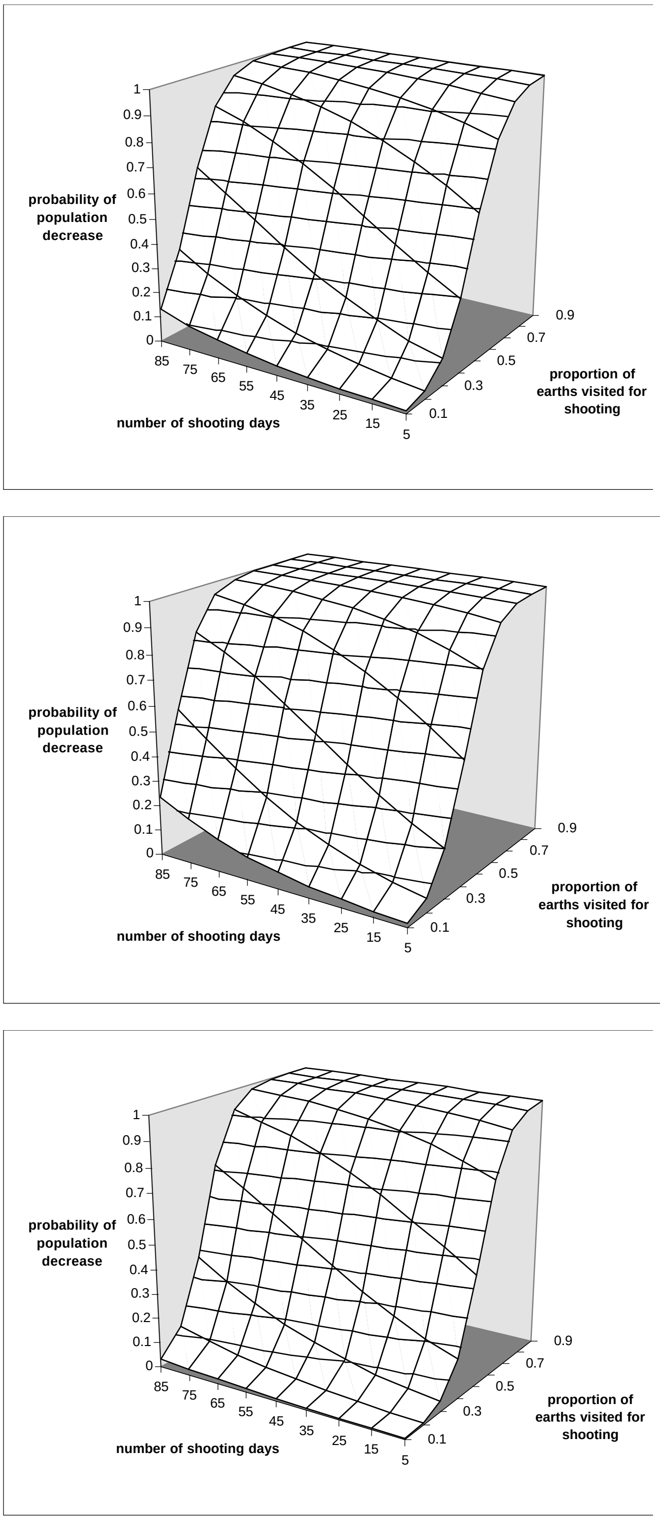
4.4.1.d. Desirable developments of the individual-based model

It must be remembered that all models are simplifications of reality. They attempt to summarise key aspects of the processes modelled so that the modeller can investigate the system under study. They should be used firstly to illuminate understanding and illustrate gaps in our knowledge before they can be used tactically to test or predict. In the present context, the modelling has essentially a didactic role. The introduction of spatial realism into the individual-based model focuses attention on the operational differences between hunting-with-hounds vis-à-vis localised culling efforts. It must be stressed, however, that this model is a first step only, and that much greater spatial realism is desirable before the models could be used tactically and impact of any management changes can be made with confidence. In particular, we would wish to investigate local variations in culling activity and then fix local-culling efforts spatially for the duration of each 15 year model run. In the model reported here, a proportion of social groups were targeted each year, but the spatial layout of these varied annually.

As already discussed, a very serious gap in current knowledge is the magnitude of cause-specific mortality risks, for both culling and non-culling mortalities. Furthermore, there is very little known of the dispersal process in foxes (in common with most other mammal species). The only way to resolve these issues would be a large-scale satellite tagging exercise of the kind already described in section 5.1.1.b. Once these data were known, individual-based spatial modelling would allow us to understand how these different mortalities interacted through compensation effects and dispersal. Indeed, one of the least realistic aspects of the model described here - the strict sequence of mortalities – would be avoided if these data were known, because risks could be applied to each individual on a much shorter time frame (e.g. day by day).

Currently the success of culling effort in the model is predefined by input parameters. In reality, success of culling – and hence the number of foxes killed and the impact of culling – depend on the opportunity to cull, i.e. on the availability of foxes where culling is attempted. This requires inclusion of some feedback between fox population density and hunting effort which would also require further information on the hunting process. Greater realism in these two respects would allow more confident prediction of the result of different culling regimes, including one in which hunting with dogs was banned. A relatively small amendment of the model would then also allow output in terms of financial costs to different interest groups; however, this step is scarcely worthwhile until the course of the model can be directed with greater realism by improving input parameters. A further requirement is to derive model outputs from the perspectives of different interest groups.

Figure 4-9 Predicted probability of population decrease in the Welsh, East Midlands and Norfolk landscapes under a range of culling scenarios. These were the proportion of earths visited for shooting and the number of shooting days each year at each earth.



4.4.2. Modelling the spatial dynamics of mink populations in relation to hunting with hounds and trapping

4.4.2.a. Approach

The model was a spatially explicit individual-based population dynamics model which simulated the dynamics of mink in terms of the life histories of individuals and their dispersal movements within the landscape. The spatial unit modelled was the mink home range, which was assumed to be a section of river between 1 and 4km long centred on one or more breeding dens. The model was written in the C programming language. The model was stage-structured (Caswell, 1989), in so far as discrete stages were recognised in each home range, but the life history processes of mortality, fecundity and dispersal were modelled at the level of the individual within the different age classes in each group.

4.4.2.b. Model structure

Mink live individually and occupy home ranges that vary in size with the quality of the riparian habitat. Home ranges have limited overlap between members of the same sex but male ranges may overlap those of more than one female. Two-dimensional digital ‘landscapes’ were created, across which mink were distributed randomly in home ranges at densities determined by the length of river available. The grid co-ordinates of a home range were used as the spatial reference point by which to model the dynamics of individuals. At the start of each model run, every home-range was assigned an adult. The life histories of each animal (reproduction, mortality and dispersal) in each range were simulated individually.

4.4.2.c. Parameterisation

A digital landscape was created for a 50 by 42 km study area in the catchment of the River Thames region defined by National Grid References (232000 190000 450000 40000). A river corridor map was created within the GRASS GIS system (Westervelt *et al.*, 1990). This consisted of all land within 100m of all watercourses in the study area. This landscape contained 750 km of river water course. Given mink home ranges may vary from between 1 and 4 km (Dunstone, 1993) this meant that the Thames study area was capable of supporting approximately 180 mink. Den sites corresponding to 180 home ranges were randomly distributed throughout the river corridor landscape. The grid coordinates of each den were used as the spatial reference point with which to model the dynamics of the individual resident mink. Each mink was assigned a sex assuming a sex ratio of 1:1.

Mink have a breeding season which can extend from March to May. They are potentially highly prolific and are capable of producing between 1 and 10 young per litter with an average of 4 (Corbet & Harris, 1991). Reproduction was therefore assumed to occur once a year, with only one litter produced by each female. The number of young produced in each litter were estimated by drawing deviates from a Poisson distribution with a mean varied as a model input (following Akcakaya *et al.*, 1995; Rushton *et al.*, 2000). Females were only allowed to breed if there were males present nearby in the landscape at a suitable distance. The distance over which male mink would travel in order to mate was varied as a model input. Mortality in mink is highly variable but is typified by high juvenile mortality (Macdonald *et al.*, 1997) and a lower adult mortality. Three types of natural mortality were used in the model in each social group. These were; adult mortality which was assumed to occur after breeding, juvenile mortality which occurred as they were recruited into the adult populations and dispersal mortality. Mortality was modelled at the level of the individual for all types. The probability of death for each individual was determined by sampling from a uniform distribution in the range 0 to 1, with mortality occurring if the deviate was in the range of the mortality for the relevant factor. All mortality rates were varied as model inputs.

There is very little information available on dispersal in mink. They are generally considered good dispersers, since animals have been recorded travelling distances 10km from the site of birth (Dunstone, 1993) and one animal was found 40 km from their birth place, (Gerell, 1970). Most dispersal of young takes place in autumn. In the model dispersal between social groups occurred once a year. All animals that could not take over their maternal home range (arising from adult mortality) dispersed. Dispersing animals interrogated the landscape, and stopped dispersal if they arrived at a site which was not already occupied. Animals moved on if the range was already fully occupied and died if they did not find an unoccupied site prior to reaching their maximum dispersal distance. The maximum distance animals could disperse was varied as a model input.

4.4.2.d. Mink population control

There are many different methods of mink control, ranging from *ad hoc* shooting by individual land owners through to organised and systematic hunting with packs of dogs. In order to investigate the impacts of different culling and control strategies on the population dynamics of mink the following culling mechanisms were modelled:

4.4.2.d.i. HUNTING

This was assumed to occur in autumn and overwinter. Hunting activity varies extensively over the country. hunts were assumed to hunt over sections of the river system in rotation with each visit focussing on a section of the total country each visit. It was assumed that hunts visited all of the habitat once during the course of a year. This in effect meant that for each hunt day only a small number of potential mink home ranges were visited. The chance that each social group was hunted was assessed stochastically, based on the number of hunt days, and the number of potential home ranges that could be visited in the course of a hunt day. Successful hunting of individuals was also varied stochastically. The number of hunt days and the chance of a successful kill were varied as input.

4.4.2.d.ii. TRAPPING

This was assumed to occur in winter during dispersal and prior to breeding, and was imposed as a mortality factor on subordinate adults and adults

4.4.2.e. Sensitivity analysis

The sensitivity of the population dynamics model to all of the life history and culling parameters was investigated by analysing how total population size and the persistence of populations of mink in each landscape varied in relation to variations in the model inputs. A Latin Hypercube Sampling (LHS) strategy following the methods of Vose (1996) was used to select input parameters for the model from the known or estimated ranges of the different variables in the model. Latin hypercube sampling uses stratified sampling without replacement to select suites of input parameters from known distributions of those variables. In practical terms, the probability distribution for each variable is split into ‘*n*’ intervals of equal probability, where ‘*n*’ is the number of sets/suites of input variables selected. The creation of input variables for a model run proceeds as follows. A random number is used to select an interval and a further random number to determine the position and hence the value of the input variable for inclusion in the set. Once selected, an interval cannot be used again. In effect each interval may be used once, but once only, in combination with values of the other variables selected in the same way. The aim was to provide a range of input values for each variable that could potentially occur under field conditions. In other words the model would be run a sufficiently large number of times to encompass all of the potential range of conditions that occur naturally rather than simply worst and best case scenarios (*sensu* Bart, 1995). Seven parameters were considered:

- Maximum dispersal distance, the distance between ranges that mink could disperse;
- The mean distance a male would go in pursuit of a female;
- Mean fecundity of females (cubs per litter);
- Adult baseline mortality (percent per year);
- Juvenile baseline mortality (percent per year);
- Hunting mortality (percent chance of successful kill)
- Trapping mortality (percent chance of success per year)

In all cases there were insufficient data available to identify the distribution function for each parameter. Furthermore, there were no data available to assess the extent to which each of the life history parameters were correlated with the others. A uniform distribution was assumed for each variable with upper and lower limits derived from the literature. The range of each parameter from which samples were taken is given in Table 4-34. The use of uniform distributions for all of the input variables is likely to lead to the selection of values for parameters that are near the extremes of their distributions more frequently than would be expected in reality. However, this ensures that all potential values (within the known range of observed behaviours for each variable) are sampled.

Table 4-34 Values of life history and culling parameters and their sources used in the individual-based models

	Min.	Max.	Source
Maximum move km	10.00	20.00	
Mean fecundity	2.00	6.00	Dunstone (1993) and Corbet & Harris (1991)
Max. mate searching distance (km)	1.0	10.0	no data
Young mortality	0.40	0.80	no data
Adult mortality	0.20	0.60	derived from Dunstone (1993)
Mortality when hunted	0.60	0.95	no data
Mortality for trapping/shooting	0.20	0.95	no data
Number of hunting sessions	5.00	20.00	no data

Two hundred sets of the input parameters were selected and the model was then run for 15 years for each set. The starter population for each run consisted of populations of animals present at all home ranges in the landscape. For each model run, the total number of mink controlled and the total population of mink present in the landscape were output at the end of the 15 years. These data were then correlated with the input variables and partial correlation coefficients were calculated to assess the impact of the individual life history parameters and CULLING on the dynamics of the simulated mink population in the landscape.

4.4.2.f. Results

Three culling scenarios were assessed in the model. These were:

- HUNTING
- TRAPPING
- HUNTING and TRAPPING

The effects of these culling regimes and the underlying mink life history parameters on model outputs were assessed using partial correlation coefficients following Vose (1996) and Rushton *et al.* (2000). The partial correlation coefficient measures the extent to which each input variable contributes to the model outputs. Two model outputs were considered these were, the predicted total population of mink left in the landscape at year 15 and the total number culled during year 15. F values assessing the significance of each partial correlation coefficient under each modelling scenario were calculated. The F values are used to assess the extent and significance with which the different model inputs contribute to the model output. F values for the input variables for each of each of the model outputs for each of the three landscape scenarios are given in Table 4-35. Entries highlighted indicate that the individual variable concerned had a significant impact on the relevant model outputs at $P< 0.05$. Considering the predicted total population of mink in the landscape at the end of the fifteen year run first. In contrast with the analyses of fox populations the predicted total population of mink under all culling scenarios appeared to be dependent on the background reproduction and mortality parameters applied in the model. This will have arisen as a result of two factors. Firstly there is considerable inherent variation in these life history parameters, mink do indeed have a greater potential range in fecundity than do foxes. Secondly, it may have arisen as an artefact of the ranges in the inputs used. Mink have been studied less extensively than foxes and the range of mortalities used as inputs, whilst realistic may reflect our poor knowledge.

The predicted impacts of culling on the mink population were different. The predicted population size was not dependent on HUNTING, either in terms of the mink ranges visited or the number of hunting days. The predicted population was dependent on the proportion of mink successfully culled in the TRAPPING scenario. This variable was also significant in the combined HUNTING and TRAPPING scenario.

Table 4-35 F values detailing the significance of partial correlation coefficients relating two model outputs (predicted total population of mink and predicted number of mink killed) to different input parameters in each culling scenario in the Thames river landscape. In all cases F values greater than 5.15 indicate that the variable has a significant impact on the output at P<0.05 (indicated in bold).

		Hunting	Trapping/ shooting	combination
Number of ranges in landscape	Maximum dispersal distance	0	0.83	0
	Mean fecundity	64.15	145.09	139.5
	Young mortality	5.91	9.81	0.74
	Adult mortality	41.57	124.44	156.7
	sub adult mortality	40.39	13.71	14.6
	Hunting mortality	0.13		0
	Trapping/shooting mortality		61.25	56.43
	Hunting days	0.07	0.29	0.61
Number of mink killed	Maximum dispersal distance	0.04	2.04	1.45
	Mean fecundity	1.79	103.24	112.1
	Max. mate finding distance	3.17	6.08	0.03
	Juvenile mortality	0.08	70.69	81.32
	Adult mortality	1.64	37.82	36.07
	Hunting mortality %	26.03		0.77
	Trapping/shooting mortality %		3.04	1.39
	Number of hunting days	178.38	0.11	1.41

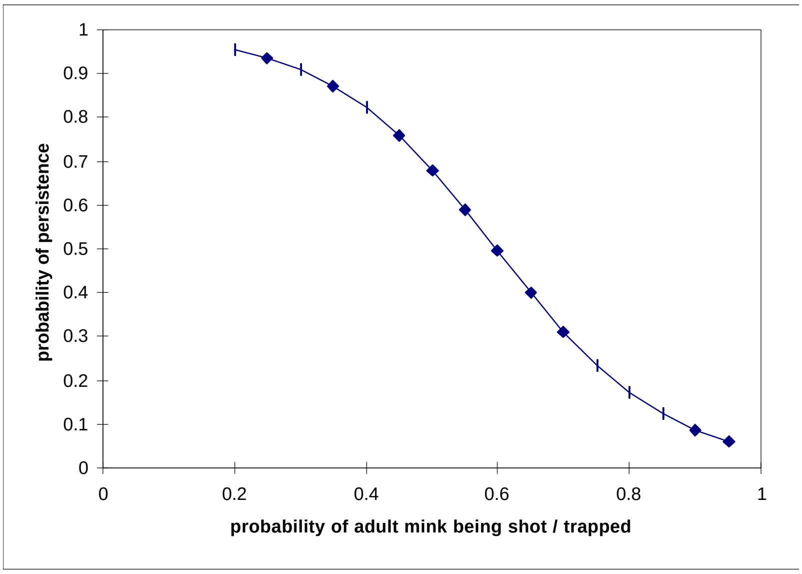
Considering the number of animals killed by each culling method. HUNTING was the only determinant of the number of animals culled in the HUNTING alone scenario, not surprisingly. In the TRAPPING, however, the number killed also appeared to be dependent on the other life history parameters of fecundity, the distance that males could travel to find a mate and adult and juvenile mortalities. Similar results were obtained for the combined control scenario with both HUNTING and TRAPPING. These results suggest that the impact of control would be dependent on how successful the mink population was in the landscape, in other words where reproduction was high and mortality low the number of animals killed would be greater because there were more there to kill.

In order to investigate the extent to which TRAPPING could be used to eradicate mink from the landscape of the Thames a logistic regression relating extinction (0) /persistence (10 to the control effort was undertaken. A predicted probability of rendering the mink population extinct in relation to the level of control applied is shown in Figure 4-10. The results suggest that in order to eradicate mink from this landscape control would have to be applied at an efficiency in excess of 95%.

4.4.2.g. Discussion

The partial correlation coefficients indicate that HUNTING was not an important factor determining the population size in the landscape. TRAPPING on the other hand was the third most important predictor after juvenile mortality and mean fecundity. As with the fox model discussed above great care has to be taken in interpreting these results. The model presented here has all of the same short-comings and oversimplifications as the fox model, with the added problem that the underlying life history processes of natural mortality for the mink are even less well documented than those of the fox. In comparison with the fox the spatial dynamics of the mink has been less studied, and the effects of different culling regimes on mink numbers little, if at all. The results suggest that HUNTING would have little impact on mink populations, but to what extent are they realistic? Birks (1981) analysed hunt records for the Devon and Cornwall Minkhounds that hunt over a large area comprising 12 river systems. He noted that, over 5 years (1976-1981) 84 mink (16 per year) were killed by HUNTING in 156 days (mean of 31 per year) that is a kill rate of 0.54 animals per hunting day. The mean number killed in the model by HUNTING was 10.8 per year so the number killed was similar, but the kill rate was much greater, at 0.96 animals per day but with comparatively fewer hunting days (12 days). It is difficult to assess the impacts of HUNTING in terms without knowing the original population size from which the animals

Figure 4-10 Probability of extinction of mink in relation to the efficiency of control imposed in the form of shooting / trapping.



have been culled. In the case of the Devon and Cornwall Minkhounds this would be impossible because of the large size of the hunt range. However, Birks (1981) analysed the small scale impact of <u>one</u> HUNTING episode on <u>one</u> population of known size on <u>one</u> river. He concluded that one in six mink were culled in this one episode. Whilst this is undoubtedly a small sample, this is much greater than the mean kill per head of population for all of the modelled scenarios presented here (mean population not HUNTED 176, mean population HUNTED 10.8). To achieve a similar proportional kill by HUNTING in the model would involve a threefold increase in hunting activity. Even given a kill rate of 16% it is unlikely that this level of kill would lead to population control since it is likely that vacated ranges are rapidly filled by non-resident animals from elsewhere (Birks, 1981). Indeed, Birks (1981) provides evidence that of all the animals culled in his Teign study area from 1975 to 1981 only 3.5% were culled by HUNTING. We conclude that HUNTING will have limited impact on mink populations.

TRAPPING in common with shooting in the fox models was predicted to be a much more effective form of population control, in that the total population of mink was related to the intensity of TRAPPING activity. As with the shooting control method used in the fox model, we have to consider how realistic the model was in terms of simulating real TRAPPING. Whilst the roving nature of HUNTING seemed to match broadly the activity as practised, it is likely that the TRAPPING scenario was over-simplistic. Mink are not easy to trap, they show considerable seasonal variation in their susceptibility to trapping and males are more easy to trap than females with young. Thus it is likely that the model was over-optimistic in its representation of trapping since none of these behavioural or seasonal effects were included. Given this constraint, it seems likely that TRAPPING has some potential for controlling mink populations even though it is very unlikely that control could ever be implemented at a level high enough to render mink extinct on any river system.

The main conclusion to be drawn from this work is that there is insufficient knowledge of the spatial dynamics of mink with which to assess the likely effectiveness of mink control measures.

4.4.3. Conclusions from individual-based models

The most important conclusion that can be drawn from the individual-based modelling is that there currently insufficient knowledge with which to evaluate the impact of culling on fox or mink populations in anything other than very general terms. It is clear that these modelling approaches could be used tactically in management if data on the population processes occurring in fox and mink populations were available, as they are for other species such as squirrels where these approaches already have tactical utility. Given these constraints we conclude that:

Of the processes modelled, shooting is the single most effective fox control method largely because the range of observed shooting activity can be very large and its impact correspondingly so.

Hunting with hounds in its conventional form has least effect on fox or mink populations compared to the other culling measures modelled.

Fertility control is likely to be ineffective as a method of controlling foxes

Further research on fox and mink mortality and dispersal and their variation in relation to culling activity are required in order to assess the impacts of a changes in fox culling policy .

4.5. What does Population Viability Analysis tell us about the effectiveness of hunting to control fox populations?

4.5.1. Approach

The behaviour and ecology of foxes varies substantially between habitats. For the purposes of illustration, we use data for lowland farmland in England. Some ecological data were available from our own field studies in Oxfordshire (Macdonald *et al*. submitted), but in the absence of sufficient demographic parameters from Oxfordshire we used values from Pech *et al*. (1997). Data on the interactions between foxes and farmers are few (e.g. Baker & Macdonald, 2000; Macdonald & Johnson, in press), but Macdonald & Johnson (1996) attempted to quantify the relationship between fox populations and foxhunting. They found it convenient to base their analysis on a typical fox population contained within a ‘hunt country’, a territory hunted over by a pack of foxhounds and averaging about $500km^2$. Our simulations also use $500km^2$ as the spatial unit.

Our PVA was used to examine population viability in relation to four main factors: fecundity, mortality, meta-population structure, and foxhunting. The results of our simulations are presented as the means of 1000 repetitions, each repetition was run for a simulated duration of 100 years. We used Vortex because it is a widely used system, amenable to sensitivity analyses, and for consistency with the earlier explorations by Macdonald & Johnson (1996).

4.5.2. Sensitivity Analysis

Vortex was used to test the influence of varying litter size, mortality rates, and metapopulation structure on fox population dynamics. A computational constraint is that Vortex allows only two rates of mortality to apply within a population, one for individuals prior to reproduction (juveniles) and another for reproductively adult animals. Initially, Pre-breeding mortality was set at 65.4% following Pech *et al*. (1997) (Table 2). The mortality of breeding foxes was based on the weighted mean of mortality estimates for age classes of 1 to 5 years (average 39.8%). The deviation in mortality rates was set at 33% and fecundity at 20% in accordance with Mills *et al*. (1996). These values ensured that our simulations encompassed extreme conditions that might increase the vulnerability of the simulated fox populations to extinction. We included this extreme variation to ensure that the impact of human-induced mortality factors was not underestimated. A range of mortality rates were then modelled, with adult and juvenile mortality rates being varied in tandem. Mean mortality was varied between 48.1 to 75.8% (juveniles) and 9.8 to 57.9% (adults) which represents the range (based on ±50% of mean survivorship). Litter sizes were varied between 2 and 6. All combinations of mortality rates and litter sizes were simulated. We assumed density dependence in that the percentage of breeding females was calculated as: % breeding = ((90.00*[1-((N/K)^16.00)]) + (40.00*[(N/K)^16.00]))*(N/(0.00+N)). We assumed environmental variation in the percentage of females breeding had an standard deviation of 2.0. We assumed no inbreeding depression.

Three sets of Vortex simulations were then produced, and each explored the impact of variables such as litter size and adult and juvenile mortality rates and the additional affect of foxhunting. The first set of results were based on a single isolated population. The second and third sets involved four adjacent populations (each the size of the single population) with migration between populations. Dispersal between hunt countries might be crucial to the response of fox populations to reduced numbers following high mortality and the four population scenarios are designed to reflect this.

Estimates of migration rates between populations were based on a number of assumptions. Our calculations assume that home ranges and hunt countries have the simplified geometry of a square and that individuals emigrating from one hunt country migrate to one of the remaining three populations. Migration rates were based on an average dispersal distance of 12.65km (based on the finding that foxes typically disperse 8.0 home ranges each with a cross section distance of 1.58km, Macdonald & Bacon, 1982; Macdonald *et al*.,submitted). These figures are well within the range reported in general reviews of fox ecology (e.g. Lloyd, 1980; Macdonald, 1984) and, importantly, are large with respect to the diameter of typical hunt countries which are our unit of analysis. This means that we should expect large fluxes of migration in and out of hunt countries. If we assume that foxes disperse in random directions, and are evenly distributed, we can calculate that 82% of the fox population will, on average, be within half of the typical dispersal range **-** we use half of the dispersal distance because foxes have an equal chance of moving in either direction, i.e. into either hunt country. Of the 82% of foxes within this perimeter zone, we assume that half (41%) has the potential to emigrate to neighboring hunt countries. This assumption is based on the fact that up to half of the fox population may be made up of transient (mostly juvenile) foxes during the annual population peak (Macdonald & Johnson, 1996). We assume migration did not begin from a population until the fox population reached half of the carrying capacity. The results are given both in terms of probability of persistence and average population size.

4.5.3. Results

The probability of persistence over 100yrs and mean population sizes (estimated from 1000 runs of the model) are presented in **Error! Reference source not found.** for a range of conditions. The area encompassed by the white square represents typical conditions for foxes, with average survivorship for juveniles and adults of 35% and 60% respectively, and litters sizes between 4 and 5. Given these conditions, single, isolated populations were rather unstable, with estimated levels of persistence ranging between 50-90%. Adding the effect of moderate hunting did not noticeably influence this pattern (**Error! Reference source not found.**). However, with four adjacent populations and levels of migration of 41% between one population and the others, the persistence remained high, between 90-100% (**Error! Reference source not found.**). This remains the case when the one of the four populations is exposed to average levels of foxhunting. **Error! Reference source not found.** combines these results to facilitate comparison of the effects of hunting and meta-population structure. Under conditions where litter sizes are low, all populations are susceptible to extinction **Error! Reference source not found.**. As litter size increases, the populations are more likely to exceed half of their carrying capacity and migration between populations occurs leading to increased levels of population persistence. Under these conditions the added effect of mortality due to hunting is minimal. Considering the average results from all of the simulations combined, fox population down to about 400 individuals remained very stable **Error! Reference source not found.**. In figure 4 the overall average affect of hunting and meta-population structure are illustrated. One can see that hunting rarely has much impact on average population sizes or persistence. Average sizes between hunted and non-hunted populations typically differed by less than the number of foxes hunted (estimated at 50/year/hunt country). The mean difference between these populations was approximately 11 to 31 for the single and four populations scenarios respectively. Overall, single populations show lower levels of persistence and smaller population sizes than one population with three adjacent populations. PVA analysis of hunting with hounds:

Our population viability analyses used techniques designed to assess the robustness of populations to the threat of extinction due to human interference, and determine which factors pose the greatest risk to their persistence. The results are expressed in terms of the average number of years of persistence or until extinction (based on a large number of simulations, to accommodate the wide variation in outcomes likely to arise due to stochastic factors). Such analyses are often undertaken to assess the fragility of threatened populations. However, we used the same methodology to model the impact of foxhunting, and other methods of population control, on the stability of fox populations. One relevance of this question is that any such reduction in fox numbers might correspondingly diminish the impact of foxes on rabbits. Whilst our simulations suggest that isolated fox populations are more extinction-prone than those linked to other sub-populations through dispersal within a metapopulation. Thus, small, island populations of foxes may be vulnerable. Nonetheless, our Vortex simulations revealed that simulated fox populations were notably resilient. Furthermore, when the simulation permitted migration between sub-populations, foxes were extremely resilient to extinction irrespective of highly variable conditions. This is exactly as expected for a notoriously opportunistic, r-selected species. Furthermore, it is also expected that the mortality likely to arise through foxhunting has only marginal impact on extinction risk except in cases where the population is already on the verge of extinction (as illustrated by some of the isolated population scenarios). The importance of metapopulation structure in stabilising the fox population and limiting risk is noteworthy.

Figure 4-11 The results from a population viability analysis (Vortex), showing the probability of persistence under variation in adult and juvenile survivorship (changed in tandem) and litter sizes, for four scenarios: single population without foxhunting a) population persistence, b) mean population size: single population exposed to moderate levels of foxhunting, c) population persistence, d) mean population size: one of four adjacent populations with migration between them with no foxhunting present e) **population persistence, f) mean population size: one of four adjacent populations but with one exposed to foxhunting, g) population persistence, h) mean population size. See text for details.**

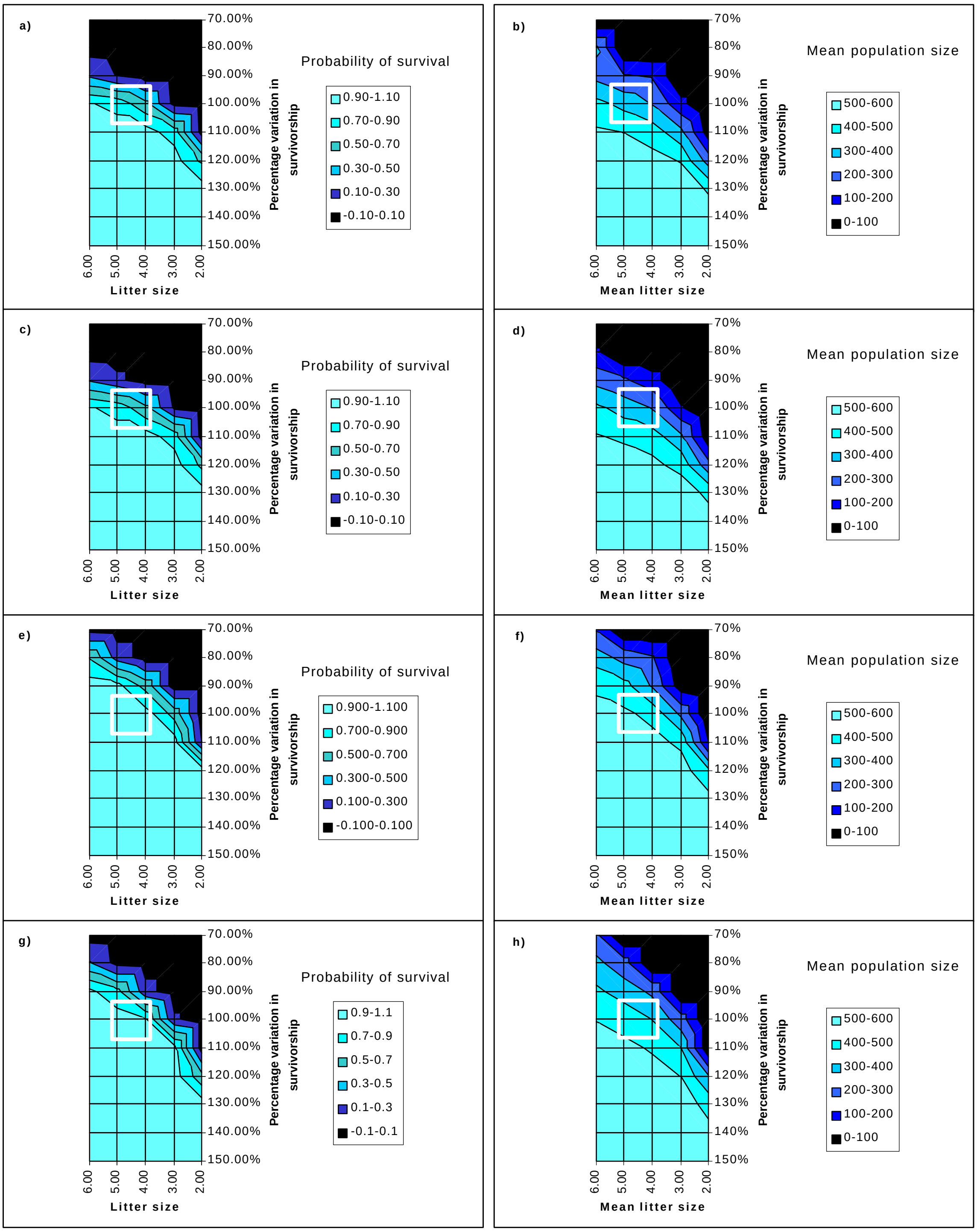


Figure 4-12 The probability of persistence against adult mortality (juvenile mortality was varied in tandem) for three scenarios: single population with moderate hunting; one of four adjacent populations; and one of four adjacent populations with one population sustained by moderate hunting (see text for details). Litter size:a) mean=2; b) mean=4; c) mean=6.

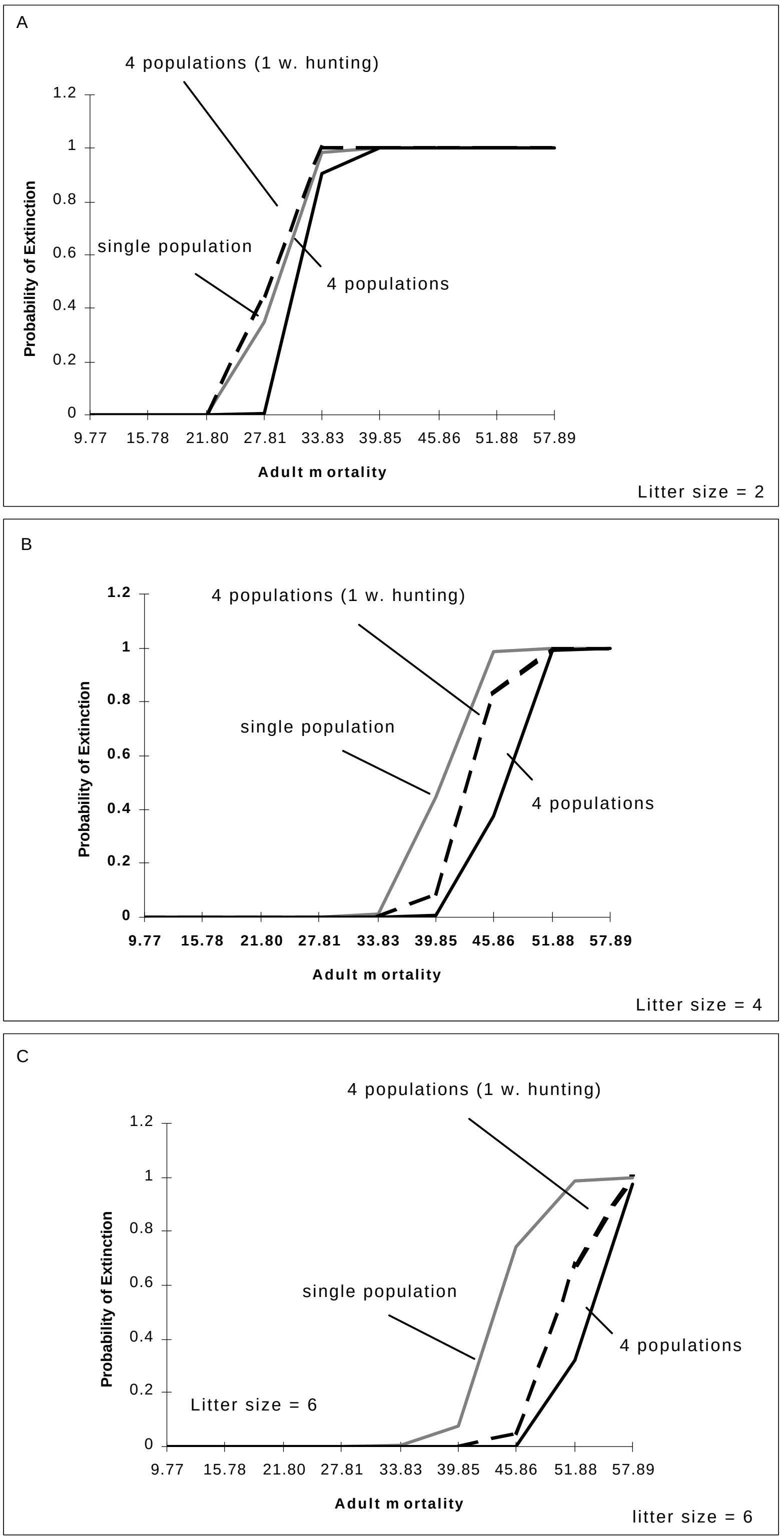
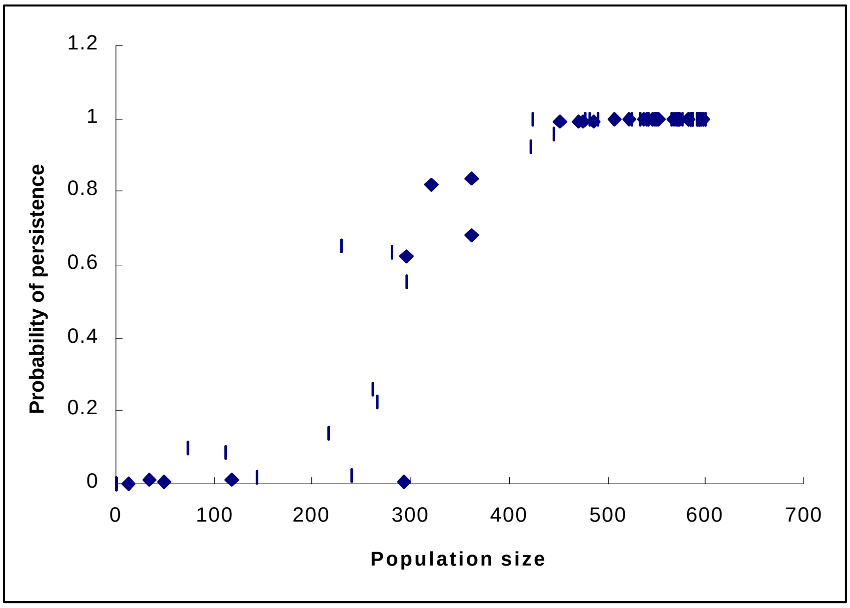


Figure 4-13 The mean probability of extinction against mean population size predicted by the population viability analysis. All simulations from Figure 1 are included.



4.6. Conclusions

- Three modelling approaches were used to investigate the potential effectiveness of culling populations of fox, red deer, hare and mink.

- The first approach, MATRIX POPULATION MODELLING was used to discover under what conditions large-scale populations could be affected by a cull.

    - ♦ Fox populations were found to be sensitive to a moderate level of simulated culling; at a level of about 0.16 foxes (both adult and sub-adult) per km^2 modelled populations reduced in the long term.

    - ♦ Hare populations were seen to be more productive, and juvenile mortality was revealed to be the key feature in the model. As long as yearly production of hares was halved every year, modelled hare populations reduced in the long term.

    - ♦ Modelled deer populations were most sensitive to the removal of juveniles and young adults – if 10-30% of these classes were removed each year, the long-term modelled population declined.

    - ♦ Finally, modelled mink populations needed reduction of recruitment to substantially affect population size – about 1.3 mink per km of river, with 60% of the mink removed being juveniles or sub-adults.

- In the second approach, INDIVIDUAL BASED MODELLING, the simulated effects of hunting with dogs was found to be ineffective at significantly influencing the population sizes of either foxes or mink.

    - ♦ For foxes, the ‘culling at the earth’ and ‘shooting’ scenarios had significant impacts

    - ♦ In mink populations, both the ‘trapping’ and ‘trapping and hunting’ were significant.

    - ♦ The most important conclusion of the individual based modelling is that there is insufficient data to investigate the impacts of culling on anything but a very general level.

- The third modelling approach, POPULATION VIALBILITY ANALYSIS, was used to examine long term trends in fox population dynamics only. This was done to explore the affects of varying adult and juvenile mortality and litter size on population persistence and size.
    - Fox populations were found on the whole to be very resilient given typical breeding and mortality rates – especially given adjacent populations with dispersal between them.
    - Given the scenarios used in this analysis, population isolation had a more pronounced affect on population viability than moderate levels of culling due to hunting with hounds. The results however, did suggest that higher levels of culling could be used to hold fox populations at lower densities at which they were still viable.

5. How effective are methods to control foxes, deer, hares, and mink in England and Wales?

5.1. *What do we mean by* ‘effective’*?*

There are two important aspects to the performance of management practices: effectiveness and efficiency. In everyday speech, these terms are often used loosely and interchangeably. However, here we draw the distinction that efficiency is ‘*doing the thing right*’, while effectiveness is ‘*doing the right thing*’.

Translated into management terms, ‘e*fficiency*’ expresses performance of a management technique (e.g. in terms of animals killed) relative to cost (e.g. in time, in effort, or money). Thus, an efficient trap would catch more animals per unit time than would an inefficient one in the same circumstances. ‘*Effectiveness*’ expresses performance in terms of the aims of management (usually the farmer’s aims). Thus, if the aim is to achieve population control, then culling can be deemed effective if it results in the population density required. If the aim is to control damage levels, however, successful population control by culling may or may not be an effective approach, depending precisely how damage is related to population density. Returning to efficiency, one method may be more efficient than another if it achieves the same aim in a shorter time, or more cheaply.

Arising from considerations of both efficiency and effectiveness are the concepts of ‘*strategy*’ and ‘*targeting*’. Consciously or not, the landowner or his representative adopts some strategy to manage wildlife, embracing the choice of methods, and the amount and timing of effort (see section 3.2.6 for a discussion of fox control strategies and 3.3.3 for deer control strategies). Accurately targeted management strategies are likely to be more effective and cost-efficient than broadly defined strategies. For instance, if lamb killing is a characteristic of individual foxes, then a strategy of selective removal of those individuals **-** if feasible **-** might be more appropriate to reducing lamb losses than a strategy to reduce fox numbers overall. Similarly, reducing losses of free-range poultry by wire fencing may be more cost-efficient than regional fox population control.

In this chapter, we assess and compare these performance measures as far as available data allow. Perceptions of effectiveness and efficiency are important because they help determine which methods are actually used; where we have pertinent data, we explore this question. Then we attempt to assess the actual effectiveness of different methods, and the cost-efficiency of three fox control scenarios. Because of the provisos listed below, these assessments are necessarily rough. Where data are simply unavailable, or where the real-world situation is so complex that the human brain can barely grasp its implications, we turn to the computer models developed in Chapter 4 for their didactic benefits. Computer modelling is an especially valuable activity in these circumstances because it helps us to understand how processes are likely to interact in the real world.

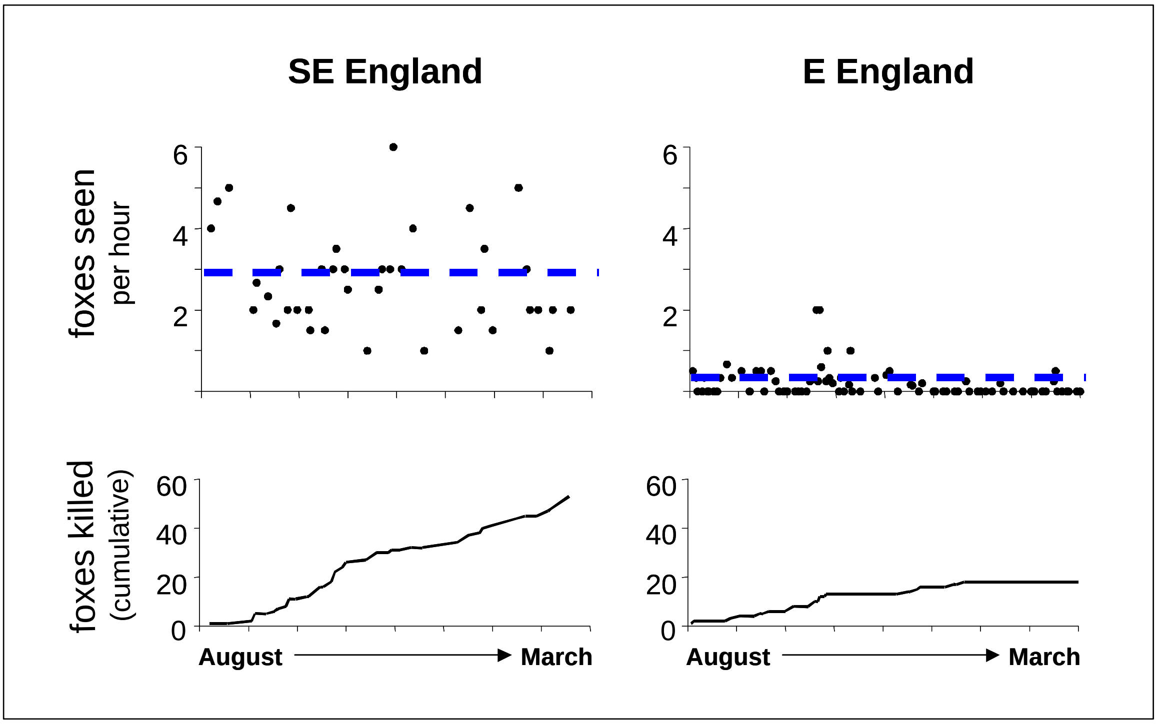
Before we turn to an assessment of the performance of different control methods, we explore some of the difficulties involved in making these comparisons, using fox culling as an illustration. . However, the general principles apply to deer, hares, and mink as well.

5.1.1. Comparing the efficiency of different methods: fox culling as an illustration

5.1.1.a. Problems of comparison

Comparison of different culling methods can be very misleading. Part of the problem is how to measure the efficiency of a culling method. The comparison often made is of foxes killed in a given time-span. Unfortunately, this ‘*catch-per-unit-effort*’ is generally not a useful measure of a method’s efficiency because it does not take into account the opportunity to cull. For instance, a high catch-per-unit-effort is possible only where there are many foxes to cull. Fox density varies three-fold within a year, and at least six-fold between regions of Britain. This will clearly distort comparisons of catch-per-unit-effort between seasons and regions even if a single method is used (Figure 5-1). It will particularly distort comparisons of methods that differ in their seasonal or regional use. For example, snares are less suited than lamping to the winter months when there is little vegetative cover, but in spring and early summer when cover is higher (and therefore favours snares) the fox population is at its minimum and capture rates by any methods will inevitably be lower. Nevertheless, in spring and summer snares can contribute substantially towards a focused and effective culling strategy, not least because high vegetative cover and short nights diminish the value of lamping at this time of year.

Figure 5-1 Judging the success of localised culling. This figure contrasts a gamekeeper in southeast England operating over 8.1 km² (left-hand side), with one in west Norfolk operating over 6.1 km² (right-hand side). The upper graphs show a dot for each occasion the keeper went out with a spot-lamp and rifle (once or twice a week in both cases); the lower graph indicates the accumulating 'bag' of foxes (killed through this and other methods), from August to March. The keeper in southeast England saw on average about 3 foxes per hour (the variation between successive occasions is characteristic and is due to chance) and gradually accumulated a 'bag' of 60 foxes, but there was no indication that this cull had made any difference to the large number of foxes present. Despite flatter terrain, the keeper in Norfolk typically saw foxes at a rate of less than 1 per hour. As a result, despite similar effort, his 'bag' only amounted to 20 foxes, but by the latter half of the period (January-March) foxes were essentially absent. These examples probably represent extreme situations for the UK, but they illustrate how 'bags are a poor indicator of either efficiency or effectiveness.



Another part of the comparison problem is that the time taken to kill a fox may not be the best measure of efficiency. Neck snares, for instance, have a low daily capture rate, yet they function largely in the absence of the operator. Similarly, if the numbers of followers was taken into account, then foxhunts would have an apparently low capture rate per man-hour of effort, but in fact most of the followers are irrelevant to the success of the hunt. Financial efficiency may be as important a consideration as efficiency of time or effort. Hunt follower's subscriptions essentially pay for the hunt in terms of the upkeep of the hounds and hunt employees, so the financial cost to the farmer may be zero or very small (see 5.3).

Perhaps the most serious problem with comparing efficiencies is that a given culling method is rarely used exclusively. Usually each fox is at risk of being killed by several culling methods. A fox killed by one method is no longer available to be killed by any other method, and as a result there is interdependence between the culls obtained by each method. All else being equal, if one method were banned, the culls taken by other methods would inevitably increase, without any increase in effort, simply because more foxes would be available to be killed. Biologists know this as '*compensation*'. In effect, different methods compete for foxes. Because of this interdependence, comparisons between methods may be confounded by the mix of methods used and the relative emphasis applied to each.

5.1.1.b. What would it take to resolve these problems?

The only reliable measure of efficiency is the cost (in financial terms, or in time or effort) required to catch a fox that is known to be present. Efficiency could then be quoted as cost, time, or effort per capture per fox-day (a fox-day being one fox present for 24 hours). To measure this would be a testing assignment for any wildlife biologist, but the point is not merely pedantic: any other measure is misleading and constitutes an unsafe basis for political decision-making.

To measure efficiency comparably for different culling methods, without incurring the problem of compensatory mortality, it would be necessary to monitor the fate of tagged foxes in a landscape where they are at risk from all those methods, as well as from non-culling mortality. Although not previously possible, such a study would be feasible nowadays by tagging a large sample of foxes with satellite transmitters, and monitoring their fate day by day. Simultaneously, it would be necessary to evaluate the cost of each culling method incurred by different interest groups within the landscape. The study would then give a measure of the risk of death from each method (method-specific mortality per fox-day per unit cost). Large sample sizes would be necessary, making the study expensive in terms of capture and tagging costs. Ancillary work would also be necessary to measure mortality in young foxes too small to carry satellite tags.

5.1.2. Comparing the effectiveness of different control strategies: fox culling as an illustration

5.1.2.a. Problems of assessing effectiveness

Assessing the effectiveness of culling strategies to achieve particular aims is no easier than assessing efficiency. The major problem, shared by all those who practise fox culling, is the difficulty in accurately assessing the number of foxes present, and the significance of the damage they cause (section 2.2.1.c.i). The best indication most operators have of the size of the fox population is the number of encounters with foxes, either in the course of everyday life, or through damage caused, or in the course of culling efforts. As already shown, levels of damage may not relate well to abundance, and the number of foxes culled is a poor indicator of effectiveness (Figure 5-1). Even for professional wildlife biologists using standardised monitoring techniques, estimating the number of foxes present has proved a very difficult task.

Even when fox density can be estimated, we encounter further complexities. Perhaps the most difficult of these is the question of geographical scale. Certain culling methods (like hunting with hounds, gunpacks, or terrier groups) are organised communally and characteristically implemented on a wide geographical scale. Others (like shooting or snaring) are characteristically implemented by individuals operating intensively within a limited area of ground. The two approaches are not independent, because both contribute to the overall mortality for the region as a whole, and because they draw on the same population of foxes via dispersal.

The aim of a game manager may be to reduce the density of foxes on his estate as close to zero as possible. The aim of a sheep farming community may be to bring the fox population of an entire region at what is perceived as an acceptable level. In terms of these aims, a local culling effort could well be locally effective even where regionally culling does not effectively control numbers. Conversely, even in a region where culling does effectively control numbers, a poor operator could be ineffective in achieving his local aim.

5.1.2.a.i. Effectiveness at a local scale

Tapper *et al.* (1993,1996) used a relative sighting index to compare fox presence in a 6 km² experimental predator removal area against a similar area without predator removal. A reduction of foxes in the removal area indicated that fox removal during spring and summer, mainly by shooting with a rifle, was locally effective in the context of this experiment. Reynolds *et al.* (1993) pieced together the likely impact of the same cull (from the age structure of the culled foxes and their known territory size) and concluded that it annulled the fox population on the 6 km² removal area, but would have had no impact 8 km away at the comparison area. In other words, because it was restricted to spring and summer months during which dispersal did not occur, it achieved a strictly local impact. Recolonisation of the predator removal area by dispersing foxes took place each autumn and winter, hence the experimental design was able to include switching of removal and non-removal treatments between the two areas after three years. The effective, localised control of fox numbers allowed the experiment to demonstrate a substantial difference in wild grey partridge survival and productivity between predator removal and non-removal areas (Tapper *et al.* 1996), despite their proximity.

5.1.2.a.ii. Effectiveness on a larger (regional) scale

The National Game-Bag Census (Tapper, 1992) shows significant variation between the regions of Britain in the numbers of foxes culled on shooting estates. Fox sighting data from gamekeepers (Reynolds, 1995) confirmed that this reflected regional differences in fox abundance. Furthermore, although much fox culling effort was locally organised and had local aims, some culling effort - such as hunting with hounds - was organised over much larger geographical areas. Regional variation in fox management practices were also apparent. These observations suggested that to fully appreciate the effectiveness of culling would require study on a regional scale. In regions with low fox abundance one would expect it to be easier to attain both local and regional control over fox numbers.

Heydon & Reynolds (2000b) assessed the likely impact of all culling efforts on fox population density in three large regions (1283-2322km^2) of England and Wales, having separately estimated fox density and the cull taken by different interest groups (Heydon & Reynolds, 2000a; Heydon *et al.*, 2000). In two regions (mid Wales and west Norfolk) the population was low and there was no evidence of reproductive suppression (Heydon & Reynolds, 2000b). The authors concluded that the population was well below carrying capacity (and therefore able to reproduce at a high rate; section 1.3.2), probably because of the intensive cull indicated by farmers. In the third region, low productivity suggested that the population was closer to the maximum sustainable by resources; here the cull indicated by farmers and others was substantial, but lower than in the other two regions.

It was not possible in this study to assess the separate effectiveness of individual methods or combinations of methods to control the fox populations. Importantly, the two regional culls which effectively suppressed populations were attained by quite different strategies: in mid-Wales 73% of the cull was taken using dogs, while in west Norfolk 64% of the cull was taken by rifle and spotlamp. In the east Midlands area 53% of the cull was taken by rifle and spotlamp, but here the overall cull had a much lower impact.

5.1.2.a.iii.Effectiveness on a national scale

A natural question for politicians, concerned with a national electorate, is to ask the effectiveness of culling to achieve population control at a national level. For biologists, such questions have less importance and are awkward to answer for a number of reasons.

In all scientific research, there is a trade-off between generality and precision: thus the wider the scale on which we wish to discuss population control, the less reliable our answers will be. Furthermore, study of the variation to be found at a smaller scale is more informative than attempts to answer a question at a single gross scale. Thus the comparisons made above between mid-Wales, the east Midlands and west Norfolk - which differ substantially in fox density, land-use and fox control traditions - tell us more about the effectiveness of culling than can any attempt to estimate the fox population and cull of Britain and to deduce effectiveness from them. Without such comparisons, the problem of how to estimate non-culling mortality becomes crippling to the exercise. Indeed, the precision with which one can determine even culling mortality declines rapidly as one moves from the single farm to large regions to the whole of Britain, as will be apparent later in this chapter.

For the land-holder, too, the questions of whether he is in an upland rather than lowland area, and whether in Cumbria or Surrey, are more immediately important than whether he is in England, Wales or Scotland. Depending on the land-uses he practices, the number of foxes on his land, and the number of foxes likely to affect this by dispersal from the surrounding region, may be important to him; the number of foxes in the whole of Britain is unlikely to be. Equally, the national importance of a particular culling method will concern him less than whether it is effective under the conditions found on his land.

In the rest of this chapter, we attempt to derive national figures because these are popular. Usually, however, these throw little light on the effectiveness of different culling methods. Given the data currently available, effectiveness can only be glimpsed by cautious consideration of a few rather specific studies, which indicate a regionally complex answer to the question.

5.1.2.b. What would it take to resolve these problems?

Resolving the question of effectiveness for different combinations of methods (strategies) would certainly be a major undertaking, comparable with the current MAFF 'Krebs Experiment' on badgers (Krebs, 1997). The correct approach is an experimental study in which comparable areas are subjected to different culling treatments. In each treatment one or other method of culling is suspended. A blocked design with replication of each set of treatments would be required. Evaluation of population trends and numbers killed would indicate the effectiveness of each culling method to achieve population control on the scale studied.

The scale required for such an experiment almost certainly makes it impracticable. Heydon & Reynolds (2000a,b) and Heydon *et al.* (2000) demonstrate that fox numbers, culling practices and population demography all vary markedly between regions the size of hunt countries. Furthermore, measurement of fox density with acceptable accuracy is not possible in regions much smaller than this. Although the suspension of hunting with dogs by registered packs might be feasible for regions of this size, the suspension of shooting or snaring on many separate land properties would be much harder to dictate.

5.2. How effective are methods to control fox populations?

Few comparative assessments of the effectiveness of different fox control methods are to be found in the literature. While hunting for sport or pelts obviously has some effect on fox density, it is widely agreed that, throughout the fox's range, the population impact of these activities is small (Phillips *et al.,* 1972; Hewson & Kolb, 1973; Storm *et al.,* 1976; Harris, 1977; Macdonald, 1980; Hewson, 1986; Voigt, 1987; Wandeler, 1988). This generalisation may arise either because of the strategy of sustained use (e.g. involving a closed season) rather than population control, or because of the methods used. The former possibility means that studies in countries where fox culling is practised primarily for sport or fur may be misleading. In Australian conditions, where the fox is universally regarded as an alien pest, hunting using traps, shooting, or dogs have all been ruled out as effective means of population control by the Australian Nature Conservation Agency (Saunders *et al.,* 1995); instead, poisoning is the preferred method, with 1080 the most common poison.

5.2.1. What is the perceived effectiveness among farmers of methods to control fox populations?

Two WildCRU questionnaire surveys in 1981 and 1995 (section 2.2.1.a.ii) asked farmers whether they believed various control methods were an effective means of controlling foxes in terms of ameliorating a specific pest problem, such as stock loss (rather than simply removing individuals). Table 5-1 presents their responses to this question.

Table 5-1 Percentage of farmers replying 'yes' when asked whether they believed a method was effective in controlling foxes. (WestC = Devon and Cornwall).

	Dorset	Leic.	Oxon	Shrop.	Suffolk	Sussex	Warw.	WestC.	Yorks	Overall (1981)	Wilts. (1995)
Gassing	64.6	55.6	61.7	71.7	61.3	67.0	65.8	50.5	57.6	61.0	38.9
Snaring	40.9	25.9	39.0	25.0	38.7	42.6	48.8	52.3	30.4	39.1	7.0
Hunting	49.6	45.7	46.8	30.0	33.3	29.7	41.5	53.3	51.1	43.7	54.9
Poison	48.0	40.7	39.0	30.0	46.7	43.6	39.0	34.6	45.7	41.2	22.2
Shooting	64.6	66.7	67.4	65.0	77.3	76.6	67.1	67.3	69.6	68.8	62.5
Terriers	33.0	29.6	38.3	21.7	30.7	34.0	32.9	45.8	32.6	34.2	19.4

The responses indicate that, overall, shooting was most widely considered an effective control method. The great majority of farmers responded that shooting was effective (69% in 1981), and there was no evidence for any regional variation. In 1981, gassing (now illegal) was also considered effective by a majority of farmers (61%), though by 1995 that figure was only 39% in Wiltshire. Hunting with hounds was considered effective by less than half (44%) of farmers overall in 1981, and by 55% in Wiltshire in 1995.

There was a five-fold difference between Wiltshire farmers' estimates of the density of foxes killed by the hunt (averaging 0.46 foxes per km^2) and figures provided by the hunt Masters (which revealed that in fact just 0.09 foxes were killed per km^2). It is almost certain that farmers' estimates of the number of foxes killed by the hunt on their land includes some double counting, possibly resulting from uncertainty regarding the fate of a fox which, at some stage of the chase, crossed their land. In contrast, figures provided by the Masters of Foxhounds were taken from their professional diaries, but do include some dead ground between farms within their territories. The same discrepancy was discovered independently by Heydon & Reynolds (2000a), who found that farmers over-estimated the hunt bag by 7 to 12 times.

We conducted new analyses of data collected in 1981, which suggested that farmers in sheep-rearing counties were more likely to report that hunting was effective. The rate at which hunting was reported to be effective was also linked to game shooting; significantly more farmers with game-shooting on their farm reported the method efficient (49% compared with 38%[11]). The effectiveness of shooting was also more frequently reported by game-shooting farmers (78% compared with 65%[12]).

5.2.2. What does modelling suggest about effective fox population control?

The population-level model (section 4.3.2) suggested that control of fox populations could be achieved through moderate levels of culling of sub-adults and adults. Because culling in the model was expressed in terms of *additive* culling mortality, it is difficult to translate these levels into real terms. If approximately 25% of sub-adults and adults were culled each year over and above other mortality, the model suggests a decline amounting to a ten-fold reduction in fox numbers after fifteen years. In reality, even more animals would need to be killed than this percentage implies because an unknown component of culling is actually wasted effort. This wasted effort would occur because it is impossible for the operator to identify those animals that will die of other causes. The model predicted that, unless it is very heavy, culling of cubs has no significant effect on long-term population size. These figures are likely to be overestimates as the intrinsic rate of increase was estimated for a population already experiencing culling and mortality.

In the individual-based model (section 4.4.1), hunting with hounds at levels of effort typical of MFHA-registered Hunts has little impact on fox abundance at a regional scale. Given the realistic hunting intensity modelled (*c*. 1-2 per hunt day; Macdonald & Johnson, 1996; Reynolds, unpublished) (section 5.2.3), there are insufficient hunting days in a season to have much impact on overall population size (more intense levels of hunting were not modelled). Shooting, also modelled at realistic intensities, had a much greater capacity to reduce fox populations regionally. The model indicated that fertility control is unlikely to be of value in controlling fox populations.

In our population viability analysis (section 4.5) we looked at the effects on population persistence and size – again at a regional scale - of varying adult and juvenile mortality (in tandem), fecundity (number of cubs per litter), and migration between populations. In addition, we added moderate levels of hunting representative of typical rates killed by hunting with hounds in the west Midlands (0.1/km^2 annually: Macdonald & Johnson, 1996). Under these levels of culling, hunting with hounds had a trivial effect on population persistence and size. Isolated populations of 600 individuals were more susceptible to increased mortality or reduced fecundity. In fact, population isolation had a greater effect on population persistence than did culling at this intensity..

Taken together, the most obvious feature of the results from all three model types is that - by itself - hunting with hounds at levels typically observed in MFHA-registered hunts can have little impact on the abundance of

[11] $\div^2_{[1]} = 11.8$, $P < 0.01$

[12] $\div^2_{[1]} = 5.1$, $P < 0.05$

foxes at a national or regional. Shooting was predicted to be the best control method by itself. Individual-based models suggest shooting to be effective in decreasing populations regionally provided it takes place over a high proportion of the region.

Fox for fox, the greatest impact on population growth is achieved by culling during early February to late March, following the period of highest 'natural' (non-culling) mortality associated with dispersal, and when the population is largely settled into territories and the highest proportion of vixens are pregnant. This also immediately precedes the period when fox predation is most significant for livestock, wild game (but not reared game), and conservation interests. However, this window of opportunity is generally insufficient to achieve the level of culling necessary to control fox numbers.

5.2.3. How many foxes are killed by hunting with hounds?

Because cull success of MFHA-registered hunts is relatively well documented, there is general agreement about the number of foxes killed by this formal style of hunting with hounds, most of which takes place with mounted huntsman and field. **Error! Reference source not found.** shows the mean density of foxes killed by the MFHA hunts for all seasons between 1959/1960 and 1992/1993 (simple arithmetic means across hunts; Johnson & Macdonald, 1996; section 3.2.1.e). During this period the mean intensity of culling by these hunts was about 0.1 foxes/km^2, though there was evidence for an upward trend in recent years. Extrapolation from these data to the entire area within MFHA hunt countries implies that *c.* 14,500 foxes were killed annually between 1960-1993 by MFHA-registered Hunts in Britain. Similarly, MFHA hunt countries the cull intensity implies that *c.* 14,500 foxes were killed annually between 1960-1993 by MFHA-registered hunts in Britain. MFHA/CforH data (section 3.2.1.f) for 127 hunts for which the hunted area was known suggest an overall mean of 0.17 foxes/km² (0.15 foxes/km² weighted for area; 95%-ile range 0.04- 0.41), for England and Wales during 1990-1996 (Table 5-2). These data suggest an annual cull of 13,800-15,400 foxes, depending on the year (Reynolds, unpubl.). Data from the diaries of seven hunt Masters in Wiltshire (section 3.2.1.e) suggested that they had killed 0.11 foxes/km^2 in 1995, and an average of 0.09/km^2 (both weighted by the area hunted over) during the previous five seasons.

Table 5-2 Regional breakdown of mean cull densities, 1990-1994 (MFHA/CforH data).

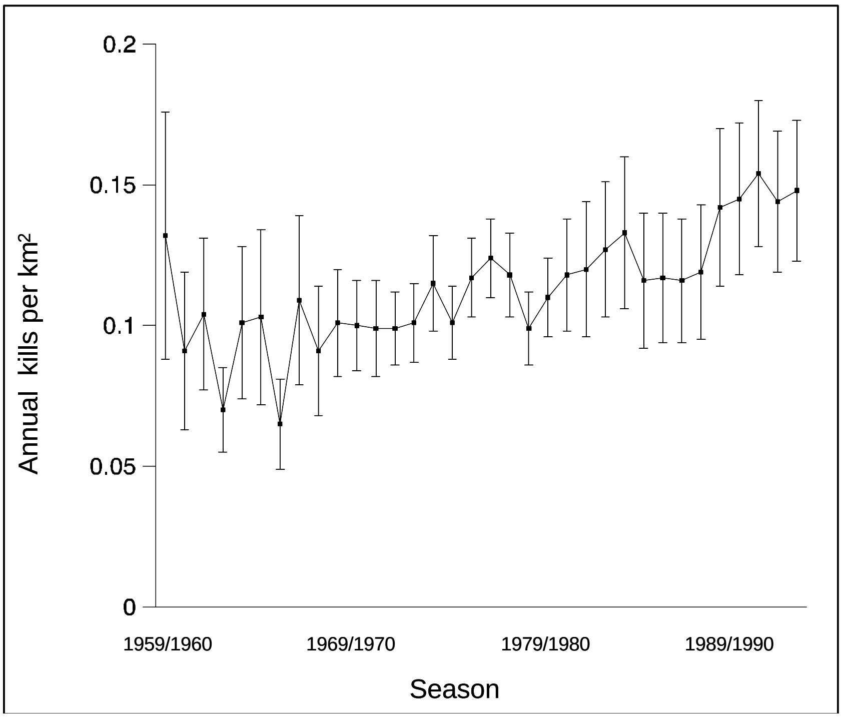
Characteristic:	Wales and West	Midlands and East	South	North	Overall
Cull Density, foxes/km^2 (unweighted)	0.23	0.15	0.15	0.11	0.17
Cull Density, foxes/km^2 (weighted by area)	0.20	0.13	0.15	0.09	0.15
N	49	31	21	23	127

In the 1998/99 season 178 foxhound packs recorded 13,987 foxes killed in England and Wales (PSL data, section 3.2.1.g); if these figures are extrapolated to cover all 184 MFHA-registered hunts, a total of 14,458 foxes may have been killed. Clearly, in different years there will be different numbers of hunting days and different numbers of foxes killed, and there may be an underlying temporal trend. However, we consider 14,500-15,000 per annum to be a good estimate of the number of foxes killed in Britain by mounted hunts and footpacks registered with the MFHA.

Both fox density and hunt culling success vary between regions of Britain. Analysis of MFHA/CforH data show that fewer foxes are killed per unit area by northern hunts (Table 5-2)[13]. The frequency of finds (foxes moved per km² per meet) also followed a similar pattern, with higher numbers found in the south and west, lower numbers in the north and east (Reynolds, unpubl.). Reynolds & Heydon (2000a) estimated culling intensities of 0.09, 0.13, and 0.02 foxes/km² taken by mounted and foot packs (but excluding gunpacks) in mid Wales, east Midlands and west Norfolk respectively, further indicating clear regional variation in the impact of these methods.

[13] $F_{3,18} = 3.3$, $P = 0.043$

Figure 5-2 The national trend in foxhunt kill density, 1959-1993.

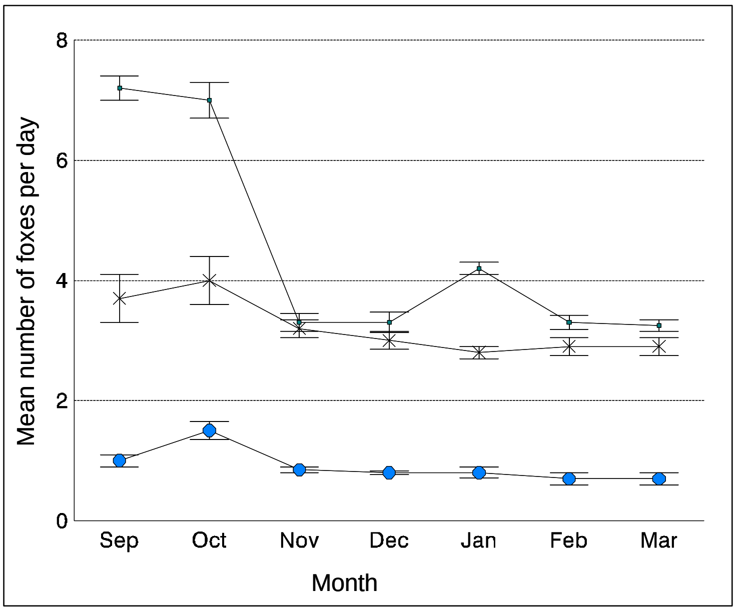


Of the 13,987 foxes recorded killed in 1998/99 (PSL data), the majority was taken in the southwest (32%), and fewest were taken in the south (11%). Similar proportions were taken in the Midlands & East Anglia (18%), the North (20%), and Wales (19%), but as these figures are not weighted by area they tell us nothing about the density of culling.

There are also seasonal differences. Over the entire period for which WildCRU have MFHA data for the autumn ‘cubhunting’ part of the season, the national proportion of total kills that occurred in this phase did not fall below 43% (in 1989/89) or exceed 51% (1991/92). There was no evidence for any significant regional variation in this proportion. In Wiltshire, according to Masters’ data, a mean of 43% of the foxes hunted throughout the entire season were killed during cubhunting, even though cubhunting days made up only 34% of the season. MFHA/CforH data from 131 hunts during 1994-1996 suggests that 39% of meets and 45% of the annual cull taken by hunts occur during the cubhunting season (Reynolds, unpubl.). Pye-Smith (1997) stated that approximately half of all hunting kills occurred during cubhunting. In the 1998/99 season 41% of kills occurred during ‘cubhunting’ (PSL data).

Examining the month by month pattern for a single hunt for which we have the most detailed accounts (Bicester and Warden Hill), reveals that the number of times which the hunt encountered foxes, and its success in terms of numbers of hunted foxes that were ultimately killed, differed markedly between ‘cubhunting’ and main-season hunting (Figure 5-3). More foxes were moved during a day’s hunting, and more of the hunted foxes were ultimately caught, during the September and October ‘cubhunting’ season.

Figure 5-3 Seasonal activity of the Bicester Hunt. Mean numbers moved (upper), hunted (middle) and killed (lower). 1964-1982.



5.2.3.a. How many foxes are killed with terriers in the course of hunting with hounds?

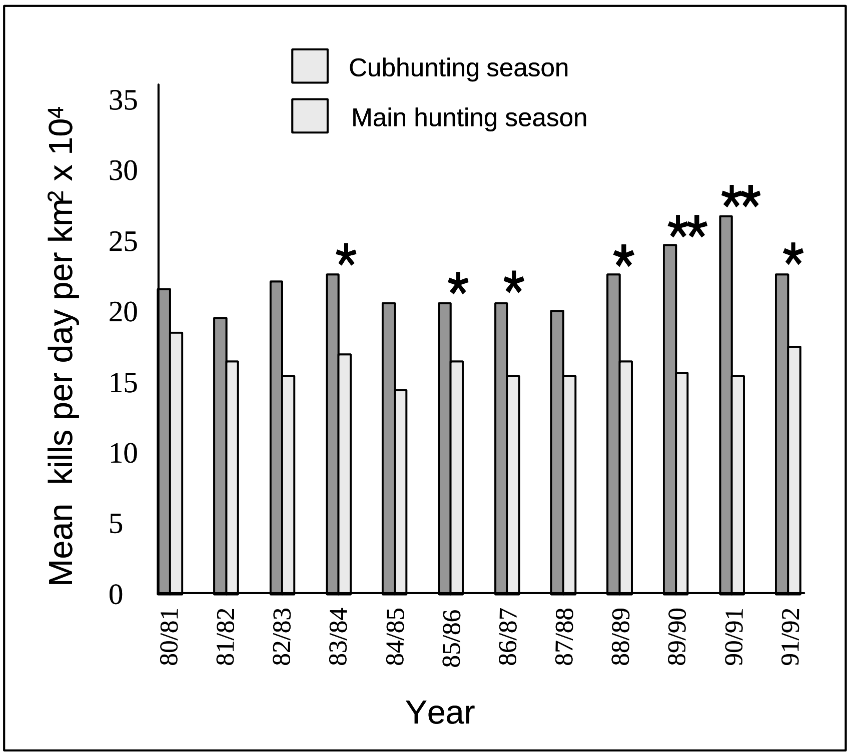
Overall, 36% of 13,987 foxes killed in the 1998/99 season were killed by digging out with the aid of terriers, and this figure was similar both cub (33%) and main (39%) hunting seasons (PSL submission to the Inquiry). For fox hunts registered with the MFHA from 1990/91-1995/96, 37% of foxes killed were taken by digging out (average for 145 hunts over 4 seasons; range for individual hunts 0-86%; MFHA/CforH data, section 3.2.1.f). Macdonald & Johnson (1996) arrived at a similar figure using data from hunt diaries: of 2062 kills for which data were available, 32.5% were made under ground. There was strong evidence for variation between hunts in this proportion. Some hunts made the majority of their kills after the fox had gone to ground: the maximum observed was 74.8% (80 out of 107). On the basis of data for numbers of foxes killed by hunting with hounds, 30-40% of which are taken by terriers, this would suggest that about 5,000-6,000 foxes are killed by terriers during hunting.

5.2.3.b. What factors influence numbers of foxes killed by hunting with hounds?

Several discretionary aspects of present-day foxhunting influence the number of foxes killed. Obviously, the amount of land any pack attempts to hunt, the number of meets per season, the distribution of meets in relation to fox abundance, and the length of the hunting season all determine culling intensity. The decision whether to dig out foxes that have gone to ground, and the proportion of the season run under early season (cubhunting) rules (with a limited field and early morning meets), and location are also extremely important. The effect of varying the hunting effort and digging out (using terriers) are explored further in sections below.

There is evidence that foxes are easier to catch during cubhunting phase of the season. For a standard effort (kills per day per $km^2 \times 10^4$), more foxes were caught during cubhunting than during the main season (Figure 5-4). On average, *c.* 15% more effort was required to catch a fox during main season hunting.

Figure 5-4 Comparison of the rate of foxhunt kills per unit effort in cubhunting and hunting proper during the 1980s. Stars denote statistically significant differences.



In Wiltshire, the hunt made fewer visits to council farms, which were most likely to perceive a fox problem and had higher reported densities of foxes, than to non-council farms (Baker & Macdonald, 2000). The hunt was more likely to visit larger, arable, upland farms, than the smaller, lowland farms better suited to stock rearing. An interesting suite of factors may explain these results. Council farms in Wiltshire tend to be smaller, low-lying, and located on heavy land (Baines *et al.,* 1995). Interviews with Masters revealed the following. The chances of the hunt choosing to visit an area on a smaller farm are less than on a larger one, they may be less suitably located for access (smaller or lowland farms might not offer the same opportunities to avoid roads, urban areas etc.), or for the hunt to be successful. Heavier land causes problems for the hunt, and the Master needs to avoid damage to the ground, which is most critical in spring. Masters can influence the direction of a hunt by arranging meets offering opportunities to hunt in different areas, therefore giving greater flexibility. If it is likely to be wet, they can meet on free-draining soils/uncultivated land. Masters also have to take into account the expected size of the Field and followers, and parking of horseboxes.

For many hunts, current choices on these discretional aspects are such that the impact of hunting could be increased if desired. Of course, we are aware that the participants consider existing practices to be largely traditional. However, none of them is ‘fixed’ other than by convention, and their manipulation is theoretically feasible. Our modelling work has to some degree explored the potential effect of increasing hunting effort in terms of numbers of days per season (section 4.4.1). Here we present an empirical investigation of the same issue. First, we consider the likely effect of increasing the number of meets, and then the effect of increasing the proportion of foxes run to ground that are dug out.

5.2.3.b.i. What is the potential effect of increasing the number of meets in a season?

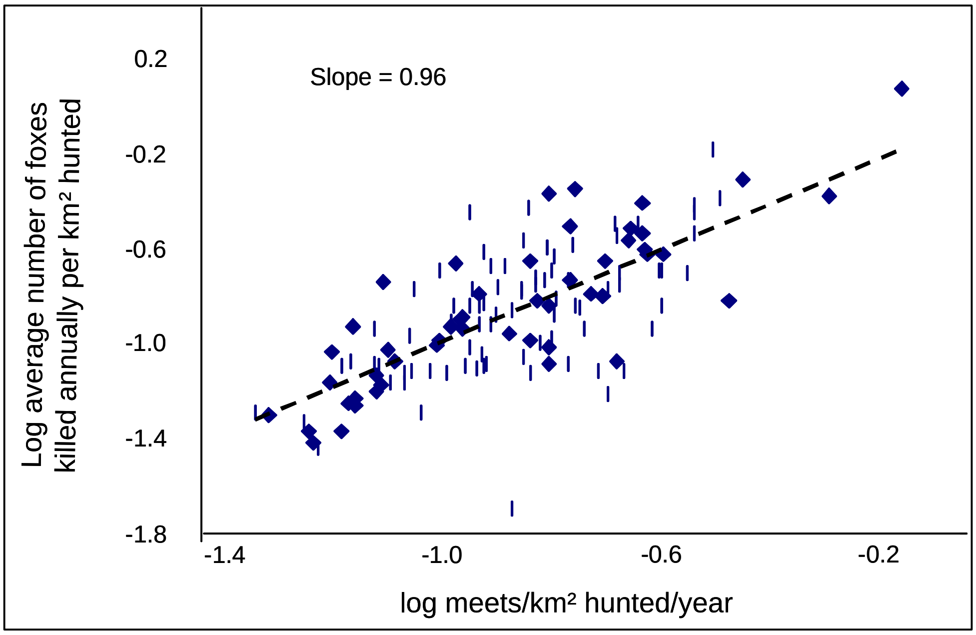
We ask what the relationship is between the effort of the hunt and its bag. As a measure of effort, we used the number of meets per unit area for each hunt (i.e. effort density) and expressed the bag per unit area as well. We allocated each hunt to one of four regions: North, South, Midlands & East, and Wales & West, allowing for differences in fox density at this scale. We then fitted statistical (GLM) models predicting cull density as a function of both region and effort (both bag and effort density were logged).

There was no evidence that the relationship between bag and effort density differed between regions[14]. There was, however, strong evidence that bag density differed between regions and in relation to effort[15]. Table 5-3 describes regional variation in current cull densities (unadjusted for variation in effort). Figure 5-5 illustrates the relationship between bags and effort.

Table 5-3 Current average hunt kill densities by region (1990-1994)

		Mean	SE	Min	Max
Midlands & East (N=31)	Bag km^{-2}	0.15	0.02	0.05	0.42
	Meets km^{-2}	0.14	0.02	0.05	0.56
North (N=23)	Bag km^{-2}	0.11	0.01	0.02	0.27
	Meets km^{-2}	0.15	0.01	0.06	0.36
South (N=21)	Bag km^{-2}	0.15	0.02	0.04	0.31
	Meets km^{-2}	0.15	0.01	0.07	0.28
Wales & West (N=49)	Bag km^{-2}	0.23	0.03	0.04	1.18
	Meets km^{-2}	0.20	0.02	0.05	0.76

Figure 5-5 The relationship between cull density and meets for MFHA hunts 1990-1994 (log scale)**.**



In the current context, the magnitude of the slope of the relationship between bag and effort is of more interest than the observation that it differs significantly from zero. The observed slope, common to all regions was 0.98 (SE = 0.07), which plainly does not deviate significantly from a value of 1.0[16]. A 95% confidence interval for the value of the slope is approximately 0.84-1.15. The practical significance of a value of 1.0 here is that it describes a situation where fox bags increase exactly in proportion with effort, that is twice the effort implies

[14] non-significant interaction term: $F_{3,116} = 0.62$, $P = 0.606$

[15] $F_{3,116} = 15.2$, $P = 0.0001$ and $F_{3,116} = 149.0$, $P = 0.0001$ respectively

[16] $t_{116} = 0.11$, P > 0.90

exactly twice as many foxes are caught. Further, there was no evidence that the relationship deviated from a straight line as it might if there was a 'levelling off' in the rate of increase in bag at high efforts[17].

A simplistic extrapolation of the predictions of this unsophisticated model implies that if hunts were to extend their season, increase the number of meets, or if there were more packs, each with a smaller hunt country, the cull achieved would increase proportionately (assuming variability between packs remained at current levels).

5.2.3.b.ii. What is the potential effect of increasing kills of foxes run to ground?

One potentially important attribute of the hunt that could be adjusted without extending either the duration or intensity of the hunting season is the policy of each hunt with respect to a hunted fox that has gone to ground. We know that hunts vary considerably in the proportion of kills made underground. In fact, this aspect of the hunt is not solely under the control of the hunt itself, and such factors as, for example, the number of terrier men available to it. Under MFHA rules, when a fox is run to ground it may now only be dug out at the request of the farmer, landowner or shooting tenant (MFHA submission to the Inquiry). What if all such interest groups were to request this activity?

The extent to which the effectiveness of an individual hunt could be increased by digging out depends on two factors (at least). First, the proportion of foxes found that are run to ground, and second the proportion of these that are subsequently killed. (We observed that these were interrelated; where a high proportion of foxes were run to ground, the proportion of these that were subsequently killed tended to be lower[18]. One explanation might be that there is an upper limit to the number of foxes that can be dug out during a day's hunting).

We estimated the impact on the effectiveness of the hunt (in terms of its total seasonal kill) of increasing the proportion of foxes dug out. We used MFHA/CforH data for hunting bags between 1990 and 1994 (section 3.2.1.f), split into those killed above and below ground. These data are available for a large sample of hunts (150), and include estimates by hunt Masters of the proportion of foxes 'found' that were subsequently killed, or run to ground.

The proportion of foxes run to ground averaged 36% across hunts, though this varied widely, from 2-95%. Doubtless, this largely reflected terrain differences. On average, 28% of foxes run to ground were subsequently killed was 28% (the median was 23%), but again there was enormous variation among hunts in this figure. Nine hunts reported that no kills were made in this way: no foxes were dug out. There was no apparent geographic pattern among these hunts; they included hunts in Devon, the Midlands, Yorkshire, Wales, and Northumberland, probably reflecting a choice rather than terrain constraints. Eight hunts reported that more than 80% of foxes that went to ground were subsequently killed.

If we assume that the first of these variables is not under the control of the hunt while the second is, we can estimate potential increase in efficiency by hypothesising that all foxes run to ground are killed, and calculating the effect of this on the density of foxes killed. Table 5-4 presents the result of this exercise, including the mean percentage increase in density of foxes killed. hunt bags did not increase in proportion with area hunted[19].

On average, the density of foxes killed would more than double through this one discretionary aspect of hunting (although the mean is distorted by a small number of high values - the median increase is x 1.8). The highest of these values are likely to be unrealisable due to the practical upper limit on the number of foxes that can be dug out during a day's hunting.

Given the variability between hunts, however, generalisation is of limited meaning. hunts where a low proportion is run to ground and a high proportion of these are killed, have little scope to increase cull intensity by this means (six hunts would increase efficiency by less than 10%). The two hunts recording increases of x 10 or greater by this means both record large numbers found, and a high proportion of these going to ground, very few of which are then killed.

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[17] quadratic term, $F_{1,118} = 0.27$, $P = 0.606$

[18] $r = -0.300$, $P = 0.0003$

[19] A regression equation predicting log(mean bag-size) from log(area) has a slope of significantly less than 1.0, which is predicted from simple proportionality; $t_{123} = 9.8$, P < 0.001)

Table 5-4 The predicted impact on hunting efficiency (national pattern) of increasing the mortality of foxes chased to ground to 100%.

	Mean	Median	Min	Max
Current kill density (foxes/km^{-2})	0.17	0.19	0.03	7.91
Potential kill density (foxes/km^{-2})	0.42	0.34	0.04	18.07
Magnitude of improvement :	x 2.50	x 1.8	x 1.02[20]	x 27.9[20]

It may be more instructive to examine the scenario in example hunts corresponding to the three areas of the country for which the most detailed data concerning fox management and demography are known (Heydon & Reynolds, 2000a,b; Heydon *et al*., 2000). The prediction for the David Davies hunt in mid Wales is an increase of 16% (0.20 to 0.23km^{-2}), that for the Llangeinor 52% (0.14 to 0.21 km^{-2}), and for Llangibby 29% (0.17 to 0.22km^{-2}). Within the English Midlands, there appears to be no greater consistency than within the entire sample. The prediction for the Bicester Hunt, for example, is 31% (0.20 to 0.26km^{-2}). That for the Vale of Aylesbury is 525% (0.1-0.5 km^{-2}). In the Vale of the White Horse by contrast, the increase would be a mere 1.8% (0.30 to 0.31km^{-2}).

5.2.4. How many foxes are killed by upland foot and gun packs?

Many upland areas are not suited to mounted hunting with hounds, and here hound packs are operated flexibly as mounted packs, foot packs, and gun packs (section 3.2.2.b). The same groups of people may also operate as terrier groups and provide a lambing call service. Hence, bag records provided by such groups include foxes taken by several methods. Although most of these involve dogs, it is not possible to distinguish between culls where hounds were used to flush the foxes from cover and the fox was subsequently killed by guns, and culls where the fox was killed by the hounds.

Data from the Welsh Farmers Fox Control Society (section 3.2.1.e), for example, encompass all these forms of culling. In the 1993/94 season 18 WFFCS packs killed 2376 foxes (an average of 132 per pack). Taking into account the area covered by each pack, Macdonald & Johnson (1996) calculated average culling intensities of 1.16 and 1.00 foxes/km^2 for the 1993/1994 and the (incomplete) 1994/1995 seasons respectively, ten times the culling intensity of mounted hunts. In 1998/99 six members of the Federation of Welsh Packs averaged a cull of 200 (the range was 130-285; FWP submission to the Inquiry). Although these packs may represent the most effective ones, extrapolation to the full 48 members would suggest a total cull of up to 9600 (or nearer 7000 if using a figure of 132 foxes culled per pack).

During ‘lambing calls’, Welsh packs kill twice as many cubs as adults. In 1998/99, eight packs from west and mid Wales (FWP submission to the Inquiry) between them killed 80 adults and 162 cubs in 129 lambing calls (equivalent to just under 2 foxes per call).

In the 1998/99 season, the six Cumbrian Fell packs which are full members of the CCFP (and make relatively little use of shooting) caught 431 foxes.

5.2.4.a. How many foxes are killed with terriers by upland packs?

A high proportion of Welsh packs kill foxes using terriers. In 1998/99, the range for six packs was 30-90%, and the average was 62% (FWP submission to the Inquiry). (The Cumbrian Fell Packs use terriers, but we have no data on the proportion of foxes dug out). However, among predominantly gun packs, the Welsh Farmers Fox Control Association estimate that terriers account for les than 15% of the cull with the remainder taken by guns above ground (WFFCA submission to the Inquiry).

[20] This represents the minimum and maximum improvement for any of the 150 hunts for which data were available.

5.2.5. Use of terriers in three contrasting regions of England and Wales

Heydon & Reynolds (2000a,b) estimated that in mid-Wales, a fox population of 1.74 foxes/km² (post-production) was subject to a culling pressure (by all methods) of 0.71-1.91 foxes/km² that was effective in suppressing fox density. Of these, 37% (i.e. 0.26-0.71 foxes/km²) were taken by digging with terriers. In the east Midlands and west Norfolk, the equivalent percentage was only 5% and 9%, respectively (these estimates are based on Heydon and Reynolds 2000a, Table 3, with further unpublished data on the percentage of foxes taken below ground by Hunts).

The derivation of these data creates some potential for error here, and requires caution. The percentage of the non-hunt cull taken by each method was estimated from farmers' questionnaire replies. Comparison with hunt records showed that these replies overestimated the number of foxes culled by Hunts. It is not known whether farmers' replies also overestimated culls by other methods to the same extent. In addition, we have assumed that the foxes stated by farmers to have been taken using terriers, rather than by the hunt, were additional to the foxes taken by terriers under the auspices of the hunt. This seems likely as these figures were always much smaller than hunt figures for foxes killed underground.

Accepting this interpretation, lower effort rather than practical difficulties probably explain the lower cull taken using terriers in the east Midlands and west Norfolk compared to Wales (Heydon & Reynolds, 2000a). In the east Midlands, the proportion of foxes killed underground by the hunt was low (18% compared with 77% in mid Wales), while the non-hunt cull per km² using terriers was much lower than in mid Wales, despite the much higher fox density. In other words, there is evidence of a restrained culling policy by all interest groups in this region. By contrast, in west Norfolk the low figure for culling with terriers probably reflects the very high intensity of autumn/winter culling by rifle, resulting in a low hunt cull (of which 56% is taken using terriers) and a spring breeding population less than half that in mid Wales.

5.2.6. How many foxes are killed by other methods?

Our data are insufficient to calculate with any meaningful accuracy the numbers of foxes killed by each method. Unlike hunting, no organised bodies keep reliable records of numbers of foxes killed by shooting or other non-hunt methods. Furthermore, we know (Haydon & Reynolds, 2000a) that there is considerable regional variation not only in fox density, but also in the density of culling and the combinations of methods used. Yet more complications arise from the fact that our data for numbers of foxes killed by farmers, gamekeepers, and the hunt all overlap to varying degrees, and with regional differences.

Pye-Smith (1997) gives estimates for the numbers of foxes killed by terriers, lurchers, shooting, and snaring, but we do not believe the data are adequate to support these figures.

5.2.6.a. How many foxes are killed on farmland in England and Wales?

In their 1981 survey of farmers (section 2.2.1.a.ii), Macdonald & Johnson (1996) asked "*How many foxes were killed on your land in the last twelve months*?". The response suggested that on average 2.3 foxes/km^2 were killed. Extrapolating to the area of England and Wales (340,640km^2), of which approximately 70% is farmed, and without any account of regional variation, produces a figure of approximately 477,000 foxes killed on farms in 1980.

Clearly, this figure is extremely rough and 20 years old. However, it does at least give us a handle on the extent of culling that occurs on farms. Reassuringly, in 1995 data from Wiltshire farmers and hunt Masters indicated that 2.26 foxes per km^2 were killed.

5.2.6.a.i. How many foxes are killed by gamekeepers?

Pye-Smith (1995; also adopted in Tapper, 1999) suggests that gamekeepers kill around 70,000-80,000 foxes a year in the UK. This is probably an over-estimate. For England & Wales, the National Game-Bag census records 7692 foxes killed by 359 gamekeepers in 1992, and 6840 by 330 gamekeepers in 1998. Extrapolating to the estimated total of 1790 gamekeepers in England and Wales (Tapper, 1992), these figures suggest only around 38,000 foxes in 1992 and 37,000 foxes in 1998 were killed by gamekeepers. A stratified analysis, in which Britain was divided into the ten roughly equal-sized regions (as in Tapper, 1992), suggests that an average of 39,000 foxes were killed annually between 1990 and 1998 on shooting estates, including those with no gamekeeper.

It should be noted that not even the number of gamekeepers in England and Wales is known with any precision. Tapper (1992) estimated 2,500 gamekeepers in the whole of Britain on the basis of census data, with 95% confidence limits of 590 to 2989. BASC (submission to the Inquiry) estimate there to be 5,500 gamekeepers in Britain. Both BASC and The Game Conservancy Trust have 3,000 to 3,500 'gamekeeper members', but these range from full-time professionals to amateurs with less available time. The National Gamekeepers' Organisation has about 2000 professional gamekeeper members, including some retired. The extent of overlap between these three memberships is unknown. A de-duplicated tally is likely to be around 3,800 people who define themselves as gamekeepers, of whom 22% are not actually practising for various reasons (J. Ewald, pers.comm.). In the present context, this self-defined group is of less importance than the number of people who actively cull foxes. This is also unknown.

5.2.6.b. What proportion of the cull is taken using each method?

The GCT's 'Three-region' study (Heydon & Reynolds, 2000a) indicated regional differences in the cull taken by shooting. In mid-Wales, east Midlands and west Norfolk (respectively), the figures were: rifle 21%, 53% and 64%; shotgun 25%, 9% and 4%. This makes the proportion of the cull taken by shooting 46%, 62% and 68%, respectively in the three regions. (Note that the 25% taken by shotgun in Wales also involved the use of hounds.)

Heydon & Reynolds (2000a) obtained estimates of the number of foxes killed using terriers outside the context of hunting in mid-Wales, east Midlands and west Norfolk. These were (respectively) 0.06, 0.19 and 0.24 foxes/km² respectively (these figures include those killed in this way by gamekeepers).

5.2.6.b.i. What proportion of the gamekeeper cull is taken using each method?

In the 1994 BASC survey (BASC submission to the Inquiry) gamekeepers estimated that an average 57% of culled foxes were shot, and 9% taken using terriers.

Roughly 38% of 1621 foxes killed by 61 professional gamekeepers in 12 months were taken using a spotlamp and rifle ('lamping'), though this proportion varied considerably by season (50% during autumn/winter, 20% in spring/summer) and between individuals (Gamekeeper Fox Culling Methods Survey [GFCMS] data, section 3.2.1.b). GFCMS data indicated that cubs and adults taken at earths constituted about 25% of the annual cull for gamekeepers, though this figure varied according to the type of shoot (wild or released birds) and overall fox control strategy; the proportions of these taken using rifle, shotgun and terriers are unknown. Together, lamping and culling at the earth took about 63% of the cull, a figure not very different from BASC's 57%.

In the 1992-3 GCT study (section 3.2.1.b), about 25% of 1621 foxes killed by 61 professional gamekeepers were taken using snares, though this proportion varied greatly between individuals (0 to 85%), probably reflecting both local conditions and personal choice. The BASC survey estimated that about 30% of the cull was taken with snares.

5.2.6.b.ii. What factors influence numbers of foxes killed by shooting?

As with hunting, numerous discretionary and non-discretionary factors influence numbers of foxes killed. Among those over which the operator has an influence are the type of shooting (lamping, culling at a cubbing earth, gun packs and standing guns), the number of shooting excursions, and their length. Some of the effects of these have already been explored. Our modelling exercise, which included the likely spatial dynamics of shooting, suggested that the percentage of social groups visited, and the kills per excursion were significant determinants of effectiveness in all three landscapes modelled.

In autumn 45% of foxes seen with a lamp are killed. This falls to 28% at its minimum in March. Although average autumn/winter values are 0.2 to 0.6 foxes per hour (dependent on region) the return dwindles towards zero as the pool of replacement foxes dries up.

A farmer employing a professional gamekeeper in the interests of game management may or may not operate a moderation ethic. A gamekeeper costs around £30,000 a year, and this expenditure must be considered worthwhile in terms of the shooting generated. On wild bird shoots especially, fox culling effort will be particularly intense during April-July, when breeding earths are easily located and gamebirds are nesting. If reducing predation on nesting gamebirds were the sole aim, this should be a sufficient strategy. However, predation on reared game, and contribution towards regional control are often additional aims, so most gamekeepers cull when they have the opportunity, accepting that culling efficacy is lower at other times because dispersal causes foxes to be replaced. Among gamekeepers in general, roughly half the foxes culled are taken during April-July; 80% of these are cubs.

In Wiltshire in 1995, a greater density of foxes was shot on ‘pest’ than on ‘non-pest’ farms, while in contrast, there was no difference in the density killed by the hunt. This suggested that hunting occurred at a fairly inflexible level, regardless of pest control considerations. As with hunting, some shooting of foxes may take place for ‘recreation’ (e.g. on arable land), but it appears that in contrast to hunting, shooting efforts can be radically increased when necessary, for pest control purposes.

5.2.7. How does the fox cull taken using different methods relate to population size and to the overall cull?

5.2.7.a. What is the national cull, how is it taken, and how does it relate to population size?

Some appreciation of the likely impact of each culling method as a determinant of fox density in England and Wales can be obtained by comparing the cull with the total number of foxes likely to die each year. Harris *et al.* (1995) estimated that the pre-breeding fox population of England and Wales outside of urban areas was 186,900 and that this population would produce around 330,000 cubs annually. This implies that 330,000 foxes must die annually if the rural fox population of England and Wales is not to increase. These estimates were derived by extrapolation on the basis of land-class from a very small number of local studies, and were given a low reliability rating by the authors. Heydon *et al.* (2000) have shown that such land-class predictions can be incorrect: the fox density observed in west Norfolk was one-fifth of that predicted by the approach used in Harris *et al.* (1995).

However, accepting the estimates of Harris *et al.* (1995) as the only basis on which to proceed with this line of argument, the 14,500-15,000 foxes killed by MFHA registered hunts (as derived above), represents about 4% of the total annual mortality. On this basis, it seems unlikely that hunting with hounds could have much impact as a determinant of the number of foxes in Britain as a whole. Similarly, Pye-Smith (1997) (also cited in Baker & Harris, 1997), reported that foxhunts are responsible for no more than 4% of total fox mortality in Britain.

The hazards of this kind of argument are illustrated by also considering the estimated cull by gamekeepers as a proportion of total fox mortality. Taking the estimate of 39,000 foxes killed by 1790 gamekeepers in England and Wales (this report), the proportion works out at 12% of total fox mortality in England and Wales. (If we use the figure of 80,000 proposed by Pye-Smith (1997) for 2500 gamekeepers in Britain as a whole, it comes to 19% of total fox mortality). Again, these seem quite small proportions, yet we know for certain (through manipulative experiment) that culling by gamekeepers can be extremely effective locally (see section 5.1.2.a.i above). Further, the cull of foxes sustained on some shooting estates is as high as 25 foxes/km², whereas rural fox densities are typically only 0.5-4.0 foxes/km²: on this basis, one would expect the gamekeeper cull to be significant on quite a wide geographical scale. For gamekeepers, and all other culling methods too, it is clear that effectiveness will vary with geographical scale. This reminds us that the aim, against which we must judge effectiveness, is to control fox numbers over a specified area: a shooting estate, or a hunt country. No group of people has control of fox numbers on a national scale as their aim.

One important factor to be considered here is the proportion of the countryside that is actually addressed by each method. For instance, roughly 65% of England and Wales is actually hunted over by hound packs registered with the MFHA. BASC (submission to the Inquiry) estimate that 6-11% of agricultural holdings are gamekeepered. These proportions are also regionally variable, as illustrated by Heydon & Reynolds (2000a).

Furthermore, it is not necessarily true that the mortality caused by a culling method must be substantial if that method is effective. It will be apparent from the theoretical discussions in section 1.3 that if a population’s ability to increase is slow (either because it is unproductive or because other mortality is extremely high) it may be controlled by quite a small additive culling mortality. When we consider one culling method at a time, ‘other’ mortality includes other culling methods. We do not know the extent of natural or non-culling mortality, nor do we know to what extent culling is additive to such mortality; we also know very little about the interplay between different culling methods (section 5.1).

How do we resolve these difficulties? One approach is to consider what contribution each method makes to the total cull of foxes, and to determine separately whether the cull as a whole is effective in controlling population size. This was the approach taken in The GCT's 'Three-region study' (Heydon & Reynolds 2000a,b; Heydon *et al*. 2000).

5.2.7.b. What are regional culls, how are they taken, and how do they relate to populations?

In The GCT's 'Three-region study' (Heydon & Reynolds 2000a,b; Heydon *et al.* 2000), the impact of culling as a whole was inferred by comparison between three contrasting regions, avoiding the need to know about other mortality. Among the three regions, a high cull was associated with low fox density and evidence that the population was well below capacity (mid-Wales and west Norfolk); a low cull was associated with a high fox density and evidence that it was closer to the capacity of the environment (east Midlands).

Within this overall picture, considerable regional variation in the importance of different methods was apparent. Methods involving dogs were estimated to account for 73% and 11% of the regionally effective culls in mid-Wales and west Norfolk respectively, and 18% of the less effective cull in the east Midlands. In the region where these methods were most important (mid-Wales) gunpacks accounted for 0.50 foxes/km^2 while hunts using hounds and terriers alone killed 0.09/km^2. In the other two areas, where shooting accounted for the bulk of the overall cull, hunt culling success reflected fox abundance: 0.13 foxes/km^2 in the Midlands compared to 0.02 in west Norfolk. In another study, records of the number of foxes killed by the hunt indicated that foxhunting accounted for approximately 5% of foxes killed on farms (per farm: 0.11 hunted, 2.26 killed in total; Baker & Macdonald, 2000). The low density west Norfolk fox population was probably suppressed well below carrying capacity. In this region, game shooting was an important secondary land-use, and culling by gamekeepers accounted for the majority of the fox cull (Heydon & Reynolds, 2000a); 64% was taken using a rifle and spotlamp. This suggests that, where the costs of carrying it out are justified by the protection of game or stock achieved, shooting is effective to achieve fox control not only locally but also on a regional scale.

In west central France, Brun *et al.* (1999) recorded a total culling pressure of 1.4 foxes/km^2 across two regions of 192 and 143 km^2. Half of this cull (i.e. 0.7 foxes/km^2) was taken by digging with terriers. The fox population in these regions was unknown. This high culling intensity by digging with terriers gives a very different perception of its effectiveness from that expressed by Saunders *et al.* (1995), who state that in Australia '*the use of small terriers as den dogs... is more of a sport rather than a control tool*' (although these authors give no evidence either way).

A second important approach to side-step our lack of knowledge about fox mortalities other than culling is to use computer modelling didactically, accepting the lack of data with which to parameterise such models, but searching for basic truths about the nature of each culling process which will determine its effectiveness. This is the approach we have taken in Chapter 4 (see section 5.2.2 above).

5.3. How cost-efficient are methods to control foxes?

We were unable to obtain estimates of the costs of many of the methods of control for the different species. However, the cases for which we do have data are useful to illustrate the issues involved in evaluating cost-efficiency of pest control methods. In this section we present three case-studies of the cost-efficiency of hunting with dogs to control foxes. The first concerns an upland hunt in Wales for which accounts were available, the second a typical mounted hunt in the English Midlands, and finally we consider how foxes might be of benefit to arable farmers.

Costs of a control method per head are relatively easily estimated. However, such costs are likely to be an over estimate of the true cost if other benefits are obtained from the method in addition to pest control. These additional benefits may be difficult to quantify, but may be very important to farmers (e.g. the collection of fallen stock by hunts). The apparent costs of sport hunting will of course be higher than other methods of control, but when evaluating such costs, it is important to identify who pays for the activity. Sporting activities may be inefficient on a cost/head basis, but if they are self supporting, then they are a cost-efficient method of control for the farmer, though not necessarily a highly effective one. Indeed, one possible disadvantage to sporting hunting is that, by definition, it is not designed solely as an effective method of controlling pest populations. Sporting groups may even encourage higher densities of their quarry (e.g. stag hunters) if this improves the hunt and they may discourage other more effective forms of control.

In terms of the efficient management of resources, it makes sense to compare the costs of control methods with the potential damage caused by the species being controlled. In one of our case studies (Welsh farmers), the cost of foxhunting was less than the estimated damage caused by the loss of lambs due to fox predation. However, the estimates of some types of damage are not substantiated and may be too high (sections 2.3.1.c.i, 2.3.1.c.i, and 2.3.2.b.i); indeed, some species may even provide benefits. We present a case study that shows the potential of foxes to benefit cereal farmers by feeding on rabbits that damage crops. However, more research is needed to quantify the impact of species like foxes in different settings; our calculations are merely illustrative of the complexities of seeking to estimate the consequences of particular policies in the absence of adequate data.

5.3.1. Cost-efficiency of hunting foxes with dogs for the sheep farming community in mid-Wales

In the 1366km² region of mid-Wales studied by Heydon & Reynolds (2000a,b), at least 968 foxes, and probably around 2602 foxes were culled in the survey year; 73% of these were killed by methods involving dogs (hunts, terrier groups, fox destruction clubs or gun-packs). To calculate the cost-efficiency of these methods for the sheep farmer, we assumed that the entire cull was taken in these ways; this avoids the unknown cost per fox for other methods, and the unknown dependability for the sheep farmer of other fox control efforts, most of which are inspired by game management.

5.3.1.a. What are the costs of culling in mid-Wales?

The Afonwy Hunt, which operated partially within this region, had an expenditure of £12,411 for the year 1998/99, in which they accounted for 259 foxes, implying a cost of £47.90 per fox (FWP submission to the Inquiry). If the entire cull for mid-Wales had been taken by methods involving dogs, the likely operational cost would thus have been between £46,382 and £124,675. At most, 80% of the cost of culling by these methods was born by the sheep farmer (6% was paid as a good neighbour policy by forestry bodies; FWP submission to the Inquiry), so the maximum likely cost to the sheep farming community in this study region was actually £99,258. Because this figure included costs for only one part-time working participant plus hounds and terriers, other time and labour costs are by implication found within the community.

Of 830 farmers in this region, 94% had sheep, with a mean flock size of 558 ewes, implying a total flock of 435,352 ewes. Mean lambing rate was 1.24 lambs per ewe, which was low compared with the east Midlands (1.73) and west Norfolk (1.5) regions; 1.35 is typical for ‘upland’, and 1.1 for hill flocks (Nix, 1996). Only 40.5% of lambs were born indoors.

Total reported pre-weaning losses among all 522,422 lambs (indoor and outdoor) were 6%. Those attributed to foxes were 0.6%, or 3,134 lambs. These are the losses that occurred <u>despite</u> the fox culling regime prevailing at the time. (If fox losses were confined to outdoor lambs, this loss amounts to 1% of outdoor lambs.) At a typical market price in 1996 of £31.50 per lamb sold (Nix, 1996), the total regional loss of income would have been £98,738, a figure comparable with the cost of fox culling. (Subsequently, the market value of lambs has fallen drastically).

To set these costs of fox culling into perspective, the annual medical (worming, dipping) and veterinary costs associated with each ewe and her lambs were £2.80 (Nix, 1996). For instance, it costs in the region of 60p to drench a ewe with Dectomax, and this would be repeated twice annually. The regional fox cull, expressed per ewe, costs a maximum of £124,675 for 435,352 ewes, equivalent to 28p per ewe, or one-tenth the medical and veterinary costs.

5.3.1.b. What are the potential costs of not culling in mid-Wales?

The fox population found in mid-Wales by Heydon & Reynolds (2000a,b) was below capacity, probably because of the heavy culling pressure. To calculate the lamb losses prevented by the culling regime, it is first necessary to postulate what might happen to fox density if culling ceased. We could not predict what fox density might become if culling ceased, but instead consider what might happen if it doubled. (This is quite plausible: density was only 40% of that in the east Midlands; productivity was high, suggesting that resources would allow an increased density; 968-2602 foxes were culled annually and if culling ceased, the population would double in one year if just 560 of these foxes survived to breed.) We assumed that twice the density of foxes causes twice the level of losses (i.e. 1.2% of lambs or 6,268 lambs). The income lost through these lamb deaths would be £197,442, considerably more than the money saved through not culling foxes (highest estimate £124,675 - see above). In other words, although lamb losses represent a small percentage of gross income, fox culling apparently was a cost-effective solution (given the assumptions made here) because it prevented much worse losses from occurring. These figures are, of course, sensitive to variation in lamb prices.

Further complications follow. Let us suppose that after fox density has increased, culling is now seen to have been cost-effective after all, and is reinstated. (This actually happened during the Second World War in a nearby region of mid-Wales hunted by the Plas Machynlleth Foxhounds - The Federation of Welsh Packs submission to the Inquiry, Appendix K). If the fox population is double what it was previously - as we posited - its annual production of cubs (2-3 per adult) will be 4-6 times greater. For the population to show zero growth would require annually the deaths of 4-6 times as many foxes as previously. We cannot predict to what extent other causes of mortality would also increase with fox density; hence we cannot assume that the cull required to ensure zero growth would also be 4-6 times as great as previously. However, it seems plausible to suppose that twice as many foxes might have to be killed annually to begin to reduce the population.

On top of the lost income of £197,442, this would cost £249,350 (unless it proved much easier at the higher fox density - but this too seems unlikely as the basic costs of running a pack are invariant, and furthermore 45 to 90% of foxes taken in this region by packs are dug out). In this scenario, culling is clearly no longer cost-effective on an annual basis, but instead has become a burden for the sheep farming community. Nevertheless, if the community risks not culling, fox density and lamb losses might increase still further. It is likely that in a sheep farming community culling by man acts as a ‘super predator’, and may be expected to show a functional response to fox density over a period of years, so that as fox numbers increase, human tolerance declines, and hunting effort is increased until numbers fall again.

Costs are involved for the public purse, too. The Forestry Commission paid £2 per fox killed by the Afonwy Hunt (FWP submission to the Inquiry, p.9.). In this region 49% of sheep farming income in 1996 came from farming subsidies (*c.* £26.45/ewe, compared with 45% from sale of lambs). Our no-culling scenario might therefore save £1936-5204 of public money in forestry grants, but would not affect the cost of subsidy, £96,747. (A full exploration of public funding issues for such a heavily subsidised industry with low profit margins would obviously include the cost of unemployment and the environmental impact if sheep farming became untenable, but is beyond the scope of this report – here we seek merely to illustrate some of the interacting factors).

5.3.2. Cost-efficiency of hunting with hounds in the Midlands

As a first attempt at considering the cost-effectiveness of a typical Midlands mounted hunt, we adopt the premise that the hunt exists solely for the purpose of culling foxes. As for the Afonwy Hunt case-study, there are two parts to this exercise. First, the cost of rendering each fox dead by this method. Second, and more problematically, estimating the economic impact that these foxes might have had on farming enterprises had they not been removed from the population.

5.3.2.a. How much does it cost to hunt?

For the first part of this problem, what are the elements of the hunt and how much do they cost? The principal sources of expense are plainly the horses, the hounds, and their associated costs. How much does the annual upkeep of a pack of hounds cost? We do not have detailed accounts for any individual hunt available, so the figures used here are speculative, and explicitly intended to be no more than a ‘back-of-an-envelope’ exploration.

We estimated costs for a hunt with two full-time employees, costing £20-£25,000 annually including whatever accommodation is maintained by the hunt for their staff (but not the cost of property upkeep, nor kennels [feed, and veterinary costs]). For comparison, the MFHA (submission to the Inquiry) quote a range of 1-5 such staff excluding part-time staff, and a recent questionnaire survey of hunt Masters suggests an average of 2.6 (PSL submission to the Inquiry).

From hunt Masters' responses, Macdonald & Johnson (1996) reported a median field of mounted followers of 50. However, a proportion of these may be considered superfluous to the hunt's task of fox culling (they might be considered as a 'moving grandstand'). The MFHA cite a range of 4-18 'hunt horses'. We used a figure of 10 horses as our baseline. Livery for a hunting hack costs between £90-£140 per week (International Association of Masters of Bloodhounds [IAMBH] submission to the Inquiry), which we used to approximate the annual cost per horse.

Additional costs include veterinary attention, equipment for the riders, transport to the meets, tack repair and farriery. There is also the initial cost of buying a horse (which given an initial purchase price of £5,000 and the prospect of 10 seasons hunting for an 8-year-old suggest *c.* £500 per year). For these, we used the IAMB figure of 'not less than' £6,000 per year. Informal inquiries among a sample of equestrian hobbyists suggest this to be realistic. In running a hunting stable rather than commercial livery, the hunt may save a proportion of this, and we incorporated a notional 20%, giving £48,000 per year for a stable of 10 horses. This totalled an annual cost of £68,000. The total figure for revenue expenditure for all hunts of *c.* £14,000,000 cited by Produce Studies Limited (submission to the Inquiry) implies a mean per hunt of *c.* £50,000, which was described as 'remarkably consistent'.

A typical Midlands hunt kills in the order of 100 foxes in the course of a season (the mean is 71), so we estimate a cost to the hunt of *c.* £680 per fox.

5.3.2.b. What is the potential cost of not hunting foxes in the Midlands?

How can we set a hunting cost of £680 per fox against the likely cost, at the level of the hunt country, of not carrying out culling by this method? The question is not a simple one. We know that other methods of culling are widespread in this area. The survival prospects of the foxes surviving hunting are unknown. We assumed that, without hunting, the foxes would remain in the population and on average, be equally likely to cause offence to farmers as unhunted foxes. How might this be translated into a financial impact on farmers?

The proportion of farmers in the Midland who report damage due to foxes was *c.*25% (Table 2, Heydon & Reynolds, 2000b). Atkinson *et al.*(1994) found that nationally 29% of farmers incurred damage from foxes, averaging *c.* £200 (at 1991 rates) where damage occured. An NOP Market Research Poll in the early 1970s (cited by McDonald *et al.*, 1997) reported that 64% of farmers suffered no damage to foxes and fewer than 2% suffered losses of more than £100 annually.

One possibility is that farmers who report no fox damage are not susceptible to fox damage, and that those who do report it would suffer increased damage in proportion to the (temporarily) increased density of foxes resulting from cessation of hunting. If there was a simple linear relationship between fox density and damage, this would imply an approximate maximum 10% increase in damage (the hunt cull numbers amount to 10% of the pre-breeding fox population). There is a possibility that removing hunting mortality would lead to a sustained upward trend in fox numbers, but given the estimated magnitude of this source of mortality in comparison to total mortality, we consider this to be unlikely. In the Midlands, fox numbers are probably closer to the carrying capacity of the environment than elsewhere (there is evidence for reproductive suppression [section 1.3.2.a]) As a result, any increase is likely soon to be halted by density-dependent effects. If a temporary increase in fox numbers of 10% is realistic, this translates (according to the damage estimates by the farmers) to an increase in cost to the farmer of the order of £20.

In a typical Midlands hunt country there are *c.* 600 farmers (Macdonald & Johnson, 1996; see section 3.2.2.a.i). There is considerable variation between surveys in the proportion of farmers sustaining damage by foxes. If 25% are susceptible to fox damage, the extra total cost to farmers in the entire hunt Country would be between £3000 and £7500 (=600 x 25% x £20). This calculation, based on many assumptions and unverified estimates of fox damage, serves to reveal that even taking the largest available estimate of fox damage, the entire hunt-scale impact would be of the order of £10,000 against an expenditure of at least five times greater.

5.3.2.c. The farmers' perspective

The foregoing discussion, as we have said, rests on the assumption that hunting exists solely to control foxes. If it is accepted that mounted hunting in the form we consider here is financially supported by participants as recreation, then plainly it makes no sense to include its costs as any part of a farming enterprise (irrespective of whether hunting has any impact on fox numbers).

Other potential inputs to the equation need to be considered for all farmers, but most of these are outside our remit. Perhaps the most substantial is the free removal of fallen stock that would otherwise incur disposal costs. The hunt also maintain hedges and woodlands for a proportion of farmers (MFHA submission to the Inquiry). Some hunts (e.g. the Cottesmore) supply hedge-laying materials (stakes and binders) to those who do their own hedge-laying, although we have no information on how widespread this maintenance is within hunt countries. No non-hunting farmer pays anything to the hunt. So for a non-hunting farmer, there may be a net benefit, arising both from removal of foxes and from woodland/hedgerow maintenance. While from this perspective the hunt may be cost-effective, it may not be efficient. On the cost side, the hunt occasionally causes damage to the farm, though the hunt will usually compensate for any damage reported to them. For a cereal farmer, removing foxes potentially carries a cost via the resulting survival of rabbits and their consumption of the crop (we derive an estimate for this cost below).

5.3.3. Can foxes reduce rabbit damage?

A farmer with no livestock or gamebirds has little reason to believe that foxes are damaging to him personally. Indeed, a few arable farmers believe that the fox is at least potentially a benefit to them because it eats the rabbits that in turn graze their crops. Are these farmers over-optimistic, or could foxes really translate into an economic benefit? Here we assess that likelihood.

5.3.3.a. Our approach and methods

We used a simple spreadsheet calculation of fox population and fox predation to estimate numbers of rabbits killed by foxes at four-week intervals from April 1st. This in turn allowed us to estimate the value of the crop damage averted per fox. We projected these estimated benefits over three years to account for the corresponding reduction in future rabbit population growth.

Within a notional area of 500km^2, our model categorised fox and rabbit populations as adults (foxes >1 year old; rabbits >20weeks) and juveniles (juvenile foxes entered the rabbit-eating population in July). The initial fox population size was set at 500 adults and 200 juveniles. Annual survivorship was assumed constant throughout the year (0.6 and 0.3 for adults and juveniles respectively).

An average of four offspring from each of 200 pairs of foxes was recruited into the rabbit-catching population in the fourth time period at the end of July. We calculated the number of rabbits killed to provide 3.0 kg per week assuming adults and juveniles weigh 1.0 and 0.45 kg respectively and that 77% of all rabbits killed were juveniles (J. Calzada, pers. comm). The total number of adult and juvenile rabbits killed annually was then converted into rabbit-years (based on ½ life expectancies of 11.2-12.8 and 15.2-15.6 weeks for juveniles and adults respectively at moderate and good survivorships; Smith & Trout, 1996). We assumed predation occurred, on average, in the middle of a rabbit's life span.

Next, we calculated the damage averted per head of wheat and barley (£6.5 and £4 respectively: Economist, May 23rd 1998, p. 119). The overall costs averted per ha were calculated assuming that agricultural land comprised 12% wheat and 8% barley (MAFF, 1997). For both the annual and longer term impact estimates, we used two extreme scenarios: a high estimate assuming that the rabbits were distributed on crop lands only (we used wheat and barley as examples); and a low estimate assuming that rabbits were evenly distributed throughout the hunt country. On this basis, these two cereals might be planted on *c.* 10,000ha in an area of 500km^2. Our analysis does not take into account whether acreage of cereals or crop type might affect the rabbit population density.

Projections over three years assume the rabbits killed would increase by a factor of 1.24-1.61 per year (based on average and good survivorship, Smith & Trout, 1996). These calculations were repeated a) in the absence of fox control; b) under foxhunting (50 foxes were killed across the year, 43% during cub hunting); and c) shooting a total of 500 individuals with 45% of mortality in April, and the rest spread evenly throughout the year. These estimates do not include the long term benefits of fox predation in terms of future fecundity of females.

For the second set of calculations, we wanted to account for the knock-on effect of predation on future population growth. To do this we calculated the numbers of rabbits with and without foxes present over a longer time scale for 1 to 3 additional years. In Table 5-5 the benefits of fox predation are expressed as the expected costs due to rabbit damage based on the average number of rabbits years for the first year and fourth year.

Table 5-5The predicted value in pounds (£) sterling of savings made due to diminished damage to crops resulting from fox predation on the rabbits. The results explore scenarios of average and good rabbit population growth (lambda), duration in years, crop type (wheat and barley), with and without fox control.

		CROPS ONLY				EVEN DISTRIBUTION			
		No Control		Fox control		No Control		Fox control	
Growth rate	Year	Wheat	Barley	Wheat	Barley	Wheat	Barley	Wheat	Barley
1.24	1	19.96	12.29	12.77	7.86	3.98	2.45	2.54	1.57
1.24	4	38.07	23.42	24.35	14.98	7.58	4.6	4.85	2.99
1.61	1	22.46	13.82	14.37	8.84	4.47	2.75	2.86	1.76
1.61	4	93.73	57.68	59.96	36.9	18.67	11.49	11.95	7.35

5.3.3.b. Results

Assuming rabbits were distributed only on cereals, our model estimates that the annual impact of fox predation saves the farmer damage worth approximately £20-22.5/ha for wheat, and £12.3-13.8/ha of barely. At the other extreme, if rabbits are evenly distributed throughout the entire landscape, the predicted value of fox predation is £4-4.5 and £2.5-2.8/ha, for wheat and barley respectively. The value of fox predation, however, is considerably greater when projected over the longer term (Table 5-5). Assuming rabbits are distributed only on cereals, after four years fox predation is estimated to save £38.1-93.7/ha for wheat, and £23.4-57.7/ha for barely. At the other extreme, if rabbits are evenly distributed throughout the entire landscape, the predicted long term value of fox predation is £7.5-18.7/ha for wheat and £4.6-11.5/ha for barley under average and good conditions respectively.

5.3.3.c. Can foxes benefit farmers through rabbit control?

Our results clearly show a potential value of foxes as predators limiting the expansion of rabbits. Rabbits have been identified as the most damaging agricultural pest in the UK (McKillop *et al.*, 1996). Previous work suggests that fox predation may limit rabbit populations, but that their ability to do so depends on the population density of the rabbits (Macdonald *et al.*, in prep). Trout & Tittensor (1989) came to similar conclusions from a survey of rabbit densities in Britain in relation to levels of predatory pressure. In areas with intensive predator control, rabbit populations densities were on average twice as high, and their distribution was 1.5 times more wide spread as areas where people exerted no control on predators.

In contrast, the effect of fox predation was marginal in areas with the highest rabbit population densities. Nonetheless, as these authors note, rabbits experience unstable population cycles with population crashes attributable to food shortage, disease and weather conditions. After such crashes a population is inevitably at low density, and therefore susceptible to limitation by foxes. Therefore, even a rabbit population that is currently so numerous as to be unaffected by fox predation may, in the future, find itself reduced to a level at which foxes do limit its recovery. In this sense, if people succeed in reducing rabbit numbers, foxes may ‘work with them’ to retard the rabbits’ recovery.

We present simple calculations to estimate the immediate annual benefits due to fox predation on croplands as well as the potential benefits in limiting rabbit growth over the longer term. Similar calculations could be made for pasture, where rabbit grazing competes with livestock. Our models indicate that foxes may contribute substantially to reducing the costs to farmers of crop losses. Clearly, in deciding whether to treat foxes as pests or assets, farmers must offset these benefits of fox predation against losses, and both merit careful quantification. We considered whether foxes lost to foxhunting would significantly diminish their asset value as rabbit control agents. Because foxhunting appears to have only marginal impact on fox numbers it has little impact on rabbit numbers. Nonetheless, there is at least a plausible case that such a cost-benefit analysis would reveal that killing foxes was a counter-productive activity for farmers concerned about mammalian herbivores causing crop-damage. We estimate that overall, given half of their diet is made up of rabbits, each fox may save between £26-£145 worth of rabbit damage per year and that this value would have a knock-on affect of reducing future damage due to rabbit population growth of between £49 - £608 per fox per year.

Our previous results suggest (Macdonald et al. in prep), in line with the best available field data, that above a certain rabbit density (which may be approx. 7ha^{-1}), foxes continue to eat them, but cannot regulate them. Foxes are therefore most valuable if farmers find some other means to reduce rabbits to a density at which foxes can then regulate them further. What, then, are the financial consequences of killing foxes. In the case of foxhunting, they are probably minimal, because for lowland British farms the impact of foxhunting on fox numbers is probably negligible: we calculate that without control the mean number of foxes alive each month in 500km^2 (an average hunt country or territory) is 900 which falls only to 884 following simulated hunting of 50 foxes (a cull of 0.1km-^{2}). Under this level of foxhunting, the total of 136k rabbits that foxes kill would be reduced by only 2k or so. However, suppose the farmers were to shoot foxes at a rate of about 1km^{-2}pa? All else being equal, the vulpine tally of rabbits would be reduced to just 82k. Depending on which scenario of rabbit survivorship and dispersion we select from Table 2, the result is that, in the first year, wheat worth between £1.63-9.25 ha^{-1} would be consumed by rabbits that would otherwise have been eaten by foxes (given a more extreme scenario, where fox predation runs over three additional years, then, during a fourth year the savings due to fox predation would diminish rabbit damage by up to £34 ha^{-1} in the fourth year (the farmer not only saves on the rabbits that the foxes eat, but also on those that would have been alive had the fox not eaten their ancestors in the first three years). The farmer, about to kill a young fox just recruited into the rabbit-eating population, might pause to consider whether doing so is worth the £26-608 of saved rabbit grazing that he thereby may forfeit (given 1300 foxes per population kill 136k of rabbits every year). The value of the fox is calculated as the damage the rabbits killed by the fox during its foraging life would have done had it not killed them.

5.4. How effective are methods to control deer populations and damage?

There are only two legal forms of killing deer for the purposes of control in England and Wales: shooting and hunting with dogs (see section 3.3). Shooting is widespread, but hunting deer to hounds is now restricted almost entirely to one small part England lying within West Somerset and North Devon (**Error! Reference source not found.**).

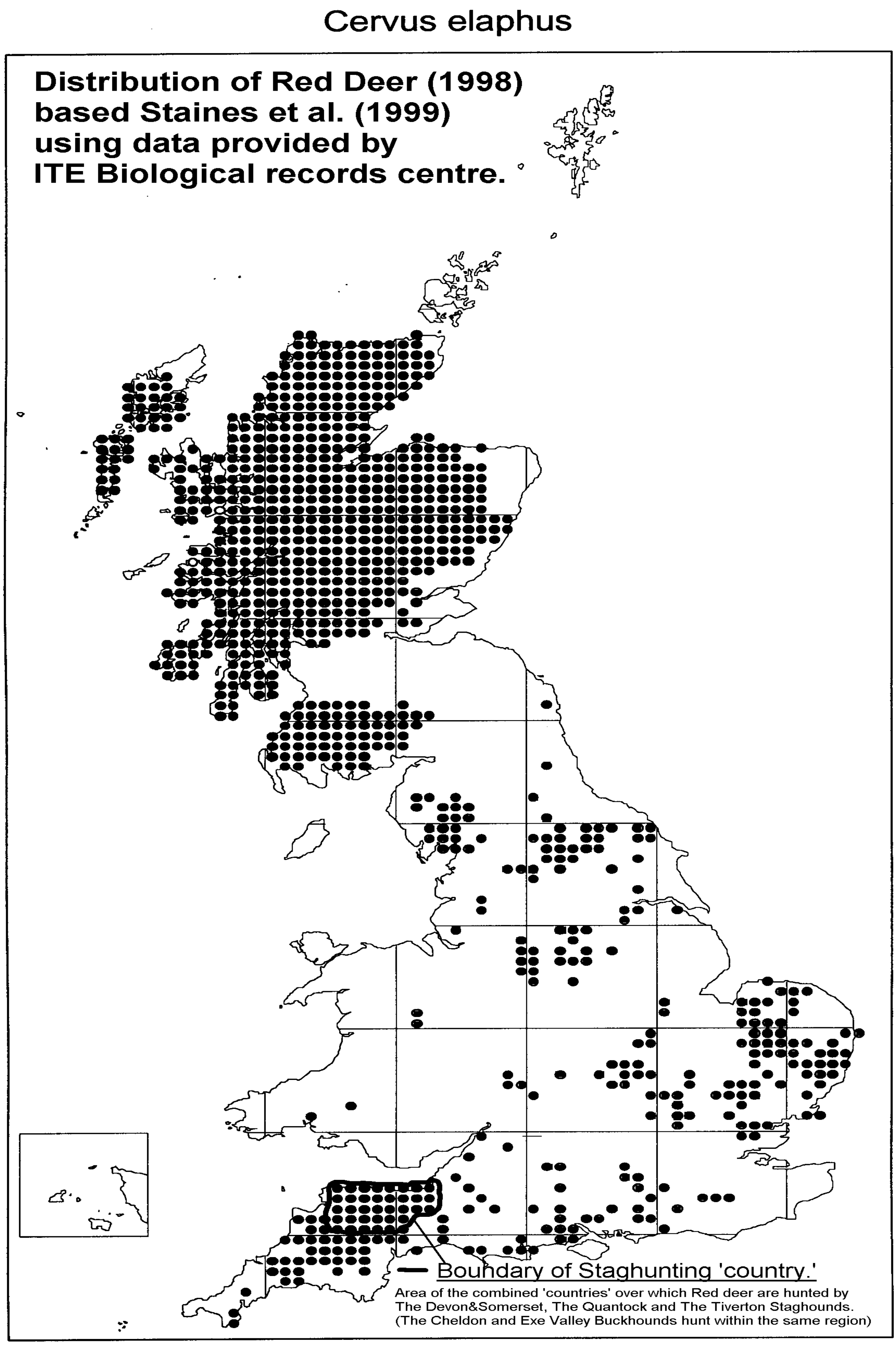
5.4.1. What is the perceived effectiveness of methods to control deer populations?

In a 1998 questionnaire (Langbein, 1998b; section 2.3.1.a), landholders in the Quantocks AONB, embracing most of the Quantock Staghunt 'country', were asked whether they believed their deer management practices (which included non-lethal measures) were effective in containing or reducing red deer damage. Of the 20 landholders who used only hunting, 80% believed it was effective; similarly, 85% of the 13 who used only shooting believed it was effective. However, only half (53%) of the 15 who combined hunting and shooting believed they achieved effective management. Doubtless, bias results if a landholder uses a method which he favours for reasons which are other than purely practical, but equally, judgments of 'effectiveness' depend on management aims and expectations.

Some proponents of staghunting claim that it helps to disperse the deer population, or keep herds on the move preventing them from settling in large numbers on vulnerable crops (e.g. MDHA submission to the Inquiry). Others also note that the deer usually return to their original areas within 36 hours (e.g. Exmoor Parish Council submission to the Inquiry; and see section 0). However, dispersal of deer and prevention of congregation of large local herds was at least *perceived* as one of the benefits of hunting by several landholders replying to the Quantocks questionnaire.

In the sections below, we assess the effectiveness of hunting to hounds and shooting in terms of population control. However, we would note that none of the three staghunts, or the MDHA, claim that hunting on its own is (or ever has been) able to exert control over deer numbers. They do argue, however, that it makes a significant contribution to the overall cull, and that it assists with other management objectives (such as increasing farmer tolerance of high deer numbers and damage levels, and dispersing large herds of deer in areas vulnerable to damage).

Figure 5-6 Location of the Staghunting countries in relation to red deer distribution.



In reality, because of the data we used, higher culling levels than those predicted by the model would be needed to maintain English red deer populations at or below carrying capacity. The problems of translating additive mortality into cull levels, and of wasted culling effort, described for foxes in section 5.2.2, also apply to the deer model.

5.4.3. How many red deer are killed by hunting with hounds?

The combined total number of red deer hunted and killed by the three packs of Staghounds in the West Country is well documented, and since 1982/83 has averaged 130 annually. Despite significant increases in population estimates over the last 30 years, the annual hunt take has remained fairly stable over that period (Langbein & Putman, 1992, 1996), with perhaps a slight increase over the last five seasons (mean for 1994/5-1998/9 = 144; range = 110-164) by comparison to the previous decade (mean for 1982/3-1991/2 = 130).

From 1982/83 to 1991/92, an additional 48 deer per annum, on average, were casualties. These are dealt with during hunt days or when the hunt is called out to kill an injured deer (e.g. car accidents, shooting, fence injuries). The total 'accounted for' by the three packs was, on average, 178 per annum, of which *c.* 25% were injuries.

The overall sex ratio of deer killed by hunting is approximately 1:1 (Table 5-6), but according to census counts, the population sex ratio is biased approximately 2:1 towards females (Langbein, 1997). Juveniles (<1 year) are not normally hunted, with hinds over 1 year old and stags over 2 years old tending to be selected.

Table 5-6 Annual average and range of numbers of red deer taken for by the three packs of West Country Staghounds during the 10 years 1982/3 to 1991/2. 'Extras' are animals of either sex dealt with by the hunts, such as e.g. fence or road casualties. Data obtained from Langbein & Putman, 1996. Averages rounded to the nearest whole number. 1985/86 data were missing for the Quantock Staghounds, and data from this season are excluded from means calculated for all hunts combined.

		Stags	Hinds	Total hunt take	Extras
Devon & Somerset Staghounds	Annual mean	33	36	69	26
	Annual range	38-26	65-20	52-101	37-10
Tiverton Staghounds	Annual mean	17	14	31	4
	Annual range	11-23	9-22	24-41	2-5
Quantock Staghounds	Annual mean	18	12	30	22
	Annual range	12-22	5-26	19-45	14-56
All Staghounds	Annual mean	68	62	130	52
	Annual range	56-78	38-95	106-170	34-95

5.4.3.a. What factors influence numbers of red deer killed by hunting with hounds?

Clearly, the most important factor influencing numbers of deer killed by hunting is its current distribution, restricted to a small part of the West Country (Figure 5-6). Within those West Country hunt countries, the effectiveness of deer control by staghunting is limited by the number of staghound packs, the number of meets organised by each, and the success rate in terms of kills achieved per meet.

The number of meets organised during the full 42-week season (August to April inclusive) is usually up to three per week by the Devon & Somerset Staghounds, and two per week each by the Quantock Staghounds and the Tiverton Staghounds. This gives a current maximum of 294 meets per year. More than half of all deer roused and hunted escape without being brought to bay (Harris *et al.*, 1999), and kills are achieved on about half of all hunting days (Bateson, 1997), depending on season, weather, scenting conditions, and location. Under current conditions, the maximum number of deer killed through staghunting (excluding the casualty service) is therefore unlikely to exceed much above 150 per annum.

Any potential regulating effect of the hunt is further limited by the proportion of females taken. The hind-hunting season, in line with the statutory close seasons for England and Wales, is limited to the 18-week period from November to February, providing a maximum of 126 hind hunting days by the three packs annually. As a rather higher proportion of meets are abandoned due to bad weather during the hind season than during either of the two periods of stag hunting, this maximum is rarely achieved. Even assuming an optimistic success rate of 2 kills per 3 outings the potential annual take of hinds under current arrangements is therefore limited to around 84 per year.

5.4.3.b. Does hunting with hounds help disperse red deer?

Deer on Exmoor and the Quantocks are relatively easy to view, not least because they often congregate in very predictable feeding or resting areas, with groups in excess of 40 hinds or 20 stags commonplace (Langbein & Putman, 1992b). These aggregations are most likely an anti-predator response to the open terrain, and disturbance such as hunting might further encourage them (Putman, 1988). If this hypothesis were true, deer dispersal would be influenced by the frequency with which particular areas are disturbed (by hunting or any other means), and by the amount of nearby cover to which the animals can retreat. Thus, regular disturbance of open farmland might help to prevent deer from settling on that farm (at least during the day), whereas similar levels of disturbance on landholdings near to woodland or scrub might have no effect.

In fact, several strands of evidence, though largely fragmentary, suggest that hunting has no lasting impact on red deer dispersal. During radio-tracking studies on Exmoor, several hinds were never recorded outside of their small home ranges of 300-700ha throughout two or more years, despite numerous hunt meets within those ranges (Langbein, 1997). During regular deer census work between October 1991 and May 1992 within the Holnicote Estate (Exmoor), several large (>30) herds of deer were observed on almost every visit, despite over 20 hunt meets on the estate during that season (Langbein & Putman, 1992). A later study (Bateson, 1987) in which hunted and non-target deer were observed during hunting also concluded that it does little to disperse deer, and other local observers indicate that deer commonly return to their original areas within 36 hours after the hunt (e.g. Exmoor Parish Council submission to the Inquiry).

Although the effectiveness of hunting in dispersing red deer is unproven, and at best is likely to be variable, many local farmers welcome the fact that they are able to request that the hunt meets on their land to attempt to disperse deer when a problem is perceived.

5.4.4. How many deer are killed by shooting?

5.4.4.a. How many deer are killed by shooting nationally

There are few precise estimates of numbers of deer shot, as there is at present no national system for monitoring deer culls in England and Wales (Macdonald & Johnson, 1998).

Table 5-7 The estimated number of deer (rounded to nearest 250) of each species shot in the UK by deer stalking members of BASC (BASC, unpublished). Figures are for all UK stalkers assume that estimates based only on BASC members are 72% of true national totals. SW England is Devon, Cornwall and Somerset, inclusive.

	Red	Roe	Fallow	Muntjac	Sika	Water deer	Total
All UK (all stalkers)	55,500	112,500	40,250	15,250	7500	ins. Data	231000
All of UK (BASC only)	40,000	81,0000	29,000	11,000	5500	Ins. Data	166,500
Of which: England & Wales	3750	47,000	27,500	11,000	1,300	ins. Data	90,500
Of which: SW England	2435	6379	1482	ins. Data	ins. Data	ins. Data	7676

For the 1995/96 culling season, a nationwide randomised questionnaire survey indicated that BASC members alone shot over 166,500 deer (of all six deer species combined) across the United Kingdom (BASC unpublished survey data, section 3.3.2.a; Table 5-7). By extrapolation to all stalkers in the country including those not subscribing to BASC (estimated at 28%; based on comparison of BASC survey estimates with minimum reported red deer culls in Scotland), the likely true total number of deer shot that year was closer to 231,000.Of these, an estimated 55,500 red deer were culled by all stalkers nationwide. More recently, the total reported cull of red deer in Scotland alone has risen from 49,000 reported to Deer Commission for Scotland in 1995/6 to over 72,000 in 1998/9, without any apparent decline in numbers (DCS, 2000).

5.4.4.b. How many deer are killed by shooting within the West Country staghunting countries

Deer stalking with rifles is the predominant method of culling red and other species of deer, both within and beyond the areas of West Somerset and Devon where hunting with hounds continues. We estimate that the total number of red deer culled annually by rifle within the area hunted by the three staghunts lies somewhere in excess of 1000-1100 head.

A figure in excess of 1000 red deer culled annually by shooting within the hunted area was first put forward in an unpublished 1988 report by West Somerset, Taunton Deane and Sedgemoor Environmental Health Consultative Panel, who examined records maintained by 20 game dealers operating in that region. Inspection of just the top three or four game dealers operating in the Exmoor area suggested that they dealt with more than 700 red deer that year; allowing for some transfer between dealers might slightly lower this number.

Tentative estimates based on results of the 1996 BASC survey (Table 5-7) for numbers of red deer culled by BASC members within the Somerset-Devon-Cornwall region (2435), also suggest that well over 1000 red deer might be taken by shooting within the staghunting country.

Following significant increases in local red deer numbers between 1960–1990, the deer population seems to have remained fairly stable or increased slowly in most parts of the hunt country during the 1990s (Langbein and Putman, 1992, Langbein, 1997), and can thus apparently sustain this cull.

5.4.5. What is the relative contribution of hunting and shooting to local, regional, and national deer population control?

Because hunting has such a limited geographic distribution, its contribution to culling might be more significant at a local and regional scale than at a national scale.

5.4.5.a. What is the contribution of hunting and shooting to red deer population control within the West Country staghunt countries?

The main sources of recent data on red deer population sizes in the staghunt countries are spring visual censuses, organised annually since 1991 by the Quantock Deer Management and Conservation Group, and since 1994 also by the Exmoor and District Deer Management Society (see Langbein, 1997). Additional census information for the Tiverton country for the last three years is also available from the Tiverton Stag Hunt (TSH, pers. comm). These counts are undertaken over one or two specific days each February or March, using large teams of local volunteers with an interest in deer. Usually 30-50 people are involved on the Quantocks and 150-250 on Exmoor. In general, each participant is allocated to a separate area of land, (which despite the numerous people involved still extends to several hundred hectares per person), and asked to observe deer for 1½ to 2 hours after sunrise, either from good vantage points overlooking the area or by means of a slow survey walk through wooded areas. Records are made of the size and location of each group of deer seen, broken down as far as possible into numbers of stags, hinds, and calves. Recent count data are summarised in Table 5-8.

Table 5-8 Spring visual censuses of red deer

	1997	1998	1999	2000
Exmoor & District DMS	2449	2567	2399	2752
Quantocks DM&CG	478	524	802	745
Tiverton Staghounds	?	675	740	800
Total	?	3766	3941	4297

These counts establish minimum numbers, and help to indicate long-term trends. Despite variability caused by weather conditions, they suggest a total spring population size of at least 4000 red deer within the region covered by the three adjoining packs of staghounds in the West Country. However, visual census methods are notorious for underestimating population size (Ratcliffe & Staines, 1987; Mayle *et al.,* 1999), and we estimate under-counting of around 25 % (see Langbein, 1997) or worse (Langbein & Putman, 1992).

In the absence of accurate records for numbers of deer killed by shooting (which under current legislation do not have to be declared to anyone by the landowners or stalkers), it is not possible to directly assess the contribution of shooting to deer population control in the West Country staghunting countries. However, we surmise that the remainder of the 85-90% of the required cull not taken through hunting to hounds is largely taken by shooting (including a small but unknown proportion taken illegally by poachers).

With their cull of *c.* 130 per annum, the hunt therefore take only about 3% of the pre-breeding population of those red deer subject to hunting with dogs. To evaluate how significant this contribution is to controlling red deer populations we assumed a spring population of between 3750 (the lowest total census figure returned over the last three years) and 5000 (assuming only 75% of actual numbers are seen at census) red deer for the three hunt countries combined. Based on a breakdown of the census returns by sex and age class, and estimated average calving rates of hinds determined from red deer culled locally (85% for adult hinds >2yrs old, and 65% for hinds conceiving as yearlings; Langbein, 1997), we predict that the *harvestable* autumn population lies between 4700-6250 head in the hunted region as a whole. From computer simulations (Langbein & Putman, 1992; Langbein, 1997; see also sections 4.3.4 and 5.4.2), we predicted a cull requirement over and above natural losses of approximately 20% of the harvestable autumn population in order to maintain the population near current levels. On this basis, we calculate that a total cull of 950-1250 would be needed to maintain current red deer populations in the Staghunting countries (assuming the cull is split by sex in similar proportion to the prevailing population sex ratio).

The average number of deer killed annually by the three packs of Staghounds in recent years (144) thus contributes only 11-15% of the total cull required to prevent further population increases.

5.4.5.a.i. What size of population could the three staghunts potentially control?

The maximum size of the population it would be possible to control through hunting alone depends not so much on the overall numbers accounted for by hunting, as on the total number of females it is possible to kill during the hind hunting season. As discussed above (section 5.4.3) the total number of females currently accounted for during the hind hunting season by the three West Country deer hunts is around 84. From this, we calculate the maximum population size that staghunting alone would be able to control, making the following assumptions:

- A minimum of 20% of the population (including at least 20% of all females present) must be culled in addition to natural mortality factors, to prevent population increase.
- Females (including female calves) outnumber males by a ratio of at least 2:1 (in line with results of recent visual censuses by Exmoor and Devon Deer Management Society, and the Quantocks Deer Management and Control Group; see Langbein, 1997).
- A hind is killed on 67% of (*c.* 126) scheduled hind hunting days.

On this basis, we calculate that the maximum autumn population size controllable by hunting alone, would lie in the region of 625 head post-breeding (falling to *c.* 500 by spring).

Although the population sex ratio of West Country deer populations is already skewed significantly towards females, hunting tends to kill approximately the same total number (but a higher proportion) of males and females each year (see Table 5-6), thus contributing further to that disparity between the sexes. In the event of the population sex ratio being re-adjusted close to parity (achievable through consistent culling in direct relation to prevailing sex ratio) the maximum population controllable by the three hunts would estimated to rise somewhat to around 840 post-breeding (falling to *c.* 660 by spring).

5.4.5.b. What is the contribution of hunting and shooting to control of red deer in South-West England?

Although the best known and most readily viewed herds of red deer occur within Exmoor and the Quantock Hills, their distribution in the West Country extends well beyond the areas covered by the three packs of staghounds (**Error! Reference source not found.**), stretching from West Somerset, throughout most of west and north Devon, and south into Cornwall.

The entire population of red deer in South-west England has been tentatively estimated at around 10,000 based on preliminary research by Langbein & Putman (1992) and reported in Harris *et al.* (1995). This is corroborated by data from BASC's nationwide questionnaire survey of deer stalkers (see Table 5-7), which estimates that in excess of 2400 red deer were culled by shooting during 1995/96 within Somerset, Devon and Cornwall (as well as 6300 roe, and 1500 fallow). Nevertheless, this higher estimate remains disputed (Yandle, pers. comm.; White, pers.comm.), with the view held by some that total pre-breeding numbers in South-west England as a whole are unlikely to lie any higher than 5000 or 6000 head. The red deer cull taken by hunting thus amounts to somewhere between 1.3-2.7 % of pre-breeding population in South-west England (equivalent to 1-2 % of post-breeding numbers).

5.4.5.c. What is the contribution of hunting and shooting to national deer population control?

The number of red deer hunted and killed by the three staghound packs (*c.* 130–144 per annum) represents around 1% of the total estimated pre-breeding population of 12,500 for England and Wales (Harris *et al.*, 1995), and is clearly dwarfed entirely by the *c.* 350,000 non-hunted red deer in Scotland.

Shooting is the most common and widely used method to control population numbers of all six of the deer species present in Britain, as well as in most other countries throughout Europe and North America. The total annual culls of red and roe deer taken by shooting during 1995/6 within just six countries [Germany, Austria, Norway, Sweden, Holland, Switzerland] in Western Europe were 110,000 and 1,750,000 respectively (Deutscher Jadgschutz Verband, 1997).

We calculate that, as a percentage of the <u>pre-breeding</u> population, shooting kills approximately 14-20% of red deer, 29-40% of fallow, and 16-22.5% of roe. We arrive at these figures using Harris *et al.*'s (1995) estimates for pre-breeding population sizes, and estimates of numbers killed by shooting as detailed above (section 5.4.4.a).

For most wild or feral populations of deer in Britain, the annual cull necessary to contain population increase, as a percentage of the <u>post-breeding</u> population, is widely accepted to fall within 5-25% of red and fallow deer, and 10-25% for roe deer, depending on local conditions and fecundity of the populations (Ratcliffe, 1987; Ratcliffe & Mayle, 1992). Re-expressed as a percentage of the spring population this translates to around 6% for rather low performance populations, and nearer 25-35% for (the majority) of higher performance populations. Our estimated levels of culling undertaken by shooting thus seem to be in the order of those considered likely to achieve control over numbers, provided population sizes themselves are not greatly underestimated. In practice, whether shooting culls are able to halt population increases is highly dependent on the age structure and sex ratio of culls taken at a local level.

5.4.6. How effective and efficient are non-lethal methods to control deer damage?

Traditional pest control involves killing, but current trends look towards non-lethal management. These are particularly important in the context of deer damage control, and can be both effective in preventing damage and cost- and time-efficient.

5.4.6.a. How effective and efficient are physical barriers?

Traditional wire mesh fencing, when constructed to the right specifications and well-maintained, provides the most effective protection from deer in all contexts (agricultural, woodland/forestry, conservational; Pepper, 1992; Poore, 1995; Robinson, 1995; Ratcliffe, 1994, 1998). Fencing to fully deer-proof specifications is expensive (£4-6/m), although cheaper, temporary fencing designs may limit, but not completely prevent, damage (e.g. Poore, 1995; Robinson, 1995).

Electric fencing is prone to breakdown, shorting out and interruption of supply; current equipment tends not to produce a sufficient shock to deter red, fallow, sika or roe deer (Pepper *et al.*, 1992), although Petley-Jones (1995) reports success with double-fence lines against roe.

Plastic growth shelters or welded net guards of the appropriate height for the deer species concerned provide good protection, but are expensive. Current (2000) prices are £1.50-2.50 per tree.

The relative cost-efficiency of fencing and tree protection depends primarily on the total area requiring protection (the perimeter requiring fencing), and planting density (the number of trees requiring individual protection). Individual tree protection is generally cheaper than fencing on areas less than 2-5ha (where fencing costs are calculated on the basis of construction to Forestry Authority recommended specifications; Ratcliffe & Pepper 1987; Hibberd, 1988). Mayle (1994) suggests that costs of whole site fencing fall below those of individual tree protection for fallow deer (which are particularly prone to damaging tree shelters with their antlers) for areas greater than 0.7-1ha. Clearly the break-point of relative cost will be different when costs of guards are set against costs of cheaper styles of fencing (Poore, 1995; Robinson, 1995).

5.4.6.b. How effective are other non-lethal methods?

Selection of tree species that are less vulnerable to browsing provides only limited protection, as the extent of damage depends on the availability of more preferred alternative foods. Most chemical feeding repellents are ineffective against deer damage; just one (Aprotect) has been found to give consistently good results, but offers protection for conifers for one winter only. Only one olfactory barrier repellent (Renardine) is currently approved for use under the Pesticides Registration scheme, but appears to be ineffective against deer.

5.5. How effective are methods to control brown hare populations?

5.5.1. What does modelling suggest about effective brown hare population control?

Our population-based model (section 4.3.3) predicted that, on a national or regional scale, brown hare populations at low densities had the potential to increase by 50% annually, and that controlling their numbers would be difficult unless juveniles (<1 year old) were removed. Hares have the highest reproductive potential of any of the species in this study; they are the only multi-voltine species, and have the largest litter size.

Most hare populations in the UK are probably close to carrying capacity, and subject to density-dependent mortality or emigration. Removing adults or sub-adults through culling will release the pressures of density-dependence. Removing juveniles, on the other hand, will remove the larger part of the population, and thus its capacity for rapid growth. Theoretically, juveniles could be removed not only through direct culling, but also indirectly, by killing pregnant or lactating females, or through fertility control. In spring, for example, an adult cull of 10%, half of which are females (each with three leverets that die when their mother is shot), could represent the removal of 11% of juveniles, assuming that 45% of the remaining females each produce three leverets.

Our model predicted that additional adult and/or sub-adult control was needed if less than 50% of juveniles were removed. In our model, removing adults and/or sub-adults, but not juveniles, had little effect on the long term population trends: modelled populations continued to increase even if 90% of the adults and sub-adults were removed from the population.

The model's predictions apply only to populations on a regional or national level; local populations may not have the large fecundities we assumed for hare populations as a whole. The problems of translating additive mortality into cull levels, and of wasted culling effort, described for foxes in section 5.2.2, also apply to the hare model.

5.5.2. How many brown hares are killed using hounds?

In their joint submission to the Inquiry, the Association of Masters of Harriers and Beagles and the Masters of Basset Hounds Association report that, on average, 5% of hares seen during a day's hunting are caught. In the 1998/99 season, beagle and harrier packs recorded sighting approximately 30,000 hares during hunting of which 1650 were caught. This figures amounts to one-fifth of one percent of the British pre-breeding population of 817,000, or less than 1% of the English and Welsh pre-breeding population of 759,500 (as estimated by Harris *et al.*, 1995).

5.5.3. How many brown hares are killed by coursing?

For the 1990/91 and 1998/99 seasons, an average of 13% of the hares chased in National Coursing Club (NCC) competitions were killed (taken from figures from coursing inspectors' reports given in the NCC submission to the Inquiry). Over the nine seasons, a total of 17,406 hares were coursed in NCC competitions, of which 2269 were killed, equivalent to 252 per year. However, this figure does not take into account coursing taking place outside NCC competitions, for which we know of no data.

In 1988, 1989, 1990 and 1999, 21, 18, 14 and 14 hares respectively were reported killed during the Waterloo Cup (coursing's premier event) at the 1127ha Altcar Estate (GCT submission to the Inquiry). On the 759ha Chippenham Estate near Newmarket, 3 and 4 hares were killed during a coursing event in 1989 and 1990. These figures are equivalent to 0.4-2/100ha

5.5.4. How many brown hares are killed by shooting?

Shooting makes by far the largest contribution to the annual cull. At least 200,000-300,000 brown hares are shot each year (National Game-Bag Census data; section 3.2.1.a), equivalent to 26-39.5% of the English and Welsh pre-breeding population. The estimated culls of hares on hare shoots are typically very high and much more than would be considered appropriate for a purely game species (Stoate & Tapper, 1993).

The sustainability of this culling has been questioned (Hutchings & Harris, 1996). In particular, Hutchings & Harris suggest that large numbers of hares immigrate onto these estates only to be shot in a subsequent year. Work by the GCT suggests there is little evidence that this is the case.

Table 5-9 Hare populations before and after shooting. Hare numbers are shown as an estimated density (N/km^2) before shooting, and as estimated populations over the whole of the estate before and after shooting. These estimates are expressed as upper and lower limits and based on 95% confidence intervals around the calculated mean. The population change is the calculated difference between the two counts and the number recorded killed is the size of the bag reported by the gamekeeper.

Locality	Year	Density (N/km^2)	Population			N^o killed	% killed
			Before	After	Change		
Whitchurch	1989	28.7	179-223	65-88	-125	117	58
Andover	1989	13.6	47-80	40-55	-16	34	53
Andover	1991	26.1	100-144	75-99	-35	34	28
Basingstoke	1988	23.3	282-336	140-172	-153	215	69
Micheldever	1991	55.5	112-178	81-113	-48	58	40

Stoate & Tapper (1993) estimated hare population sizes before and after shooting on four arable farmland estates ranging in size from 262-1165ha. These estates were all managed as game shoots, and gamekeepers were employed for rearing game, predator control, and to deter poaching. Each estate was visited before and after each hare shoot, and hare counts were made using a spotlight (Barnes & Tapper, 1985). The number of hares killed at the end of the shoot was recorded by the game keeper, and this recorded bag was taken to be the number of hares killed (although it may have been an underestimate). Results of the counts are summarised in Table 5-9. The average density of the hare population on these four farms was 29/100ha. The size of the cull ranged from 40% to nearly 70% of the population, and averaged nearly 50%. In every case, shooting reduced the estimated population size, but even these reduced populations were higher than those on average farmland

(Hutchings & Harris, 1996). This is probably because most areas where hares are shot are places that are highly suited to hare breeding (Hutchings & Harris, 1996) and where other mortality (e.g. predation) is kept low (Stoate & Tapper, 1993).

5.6. *How effective are methods to control mink populations?*

5.6.1. What does modelling suggest about effective mink population control?

Mink are more productive than foxes (modelling suggests an overall 40% annual increase at low densities), and they continue to reproduce at a high rate throughout their lives. Indeed, our population-level model (section 4.3.5) predicted no population decrease when just adults were culled (up to 90% effectiveness), because of their high fecundity. As with hares, removing adults or sub-adults through culling simply releases the pressures of density-dependence. Increasing the mortality of the juveniles on the other hand, removes the larger part of the population, and thus its capacity for rapid growth. The population-based model predicted that the most effective control strategy was to impose juvenile and sub-adult mortality together. This limits recruitment to the most fertile life stage, the adult. If adult control is applied in conjunction with juvenile control (a more feasible option), reducing recruitment by half will reduce mink populations in the long term with only a low intensity of adult control. The problems of translating additive mortality into cull levels, and of wasted culling effort, described for foxes in section 5.2.2, also apply to the mink model.

Our individual-based model (section 4.4.2) suggested that 'HUNTING' would have little impact on mink populations because the number of animals that can be culled by this activity is low relative to the reproductive potential of the species and the ability of other animals to colonise vacant areas. The model predicted that 'TRAPPING' was a much more effective form of population control, in that the total population of mink was related to the intensity of 'TRAPPING' activity. However, the level of control that would need to be implemented may be in excess of that which could be achieved in practice. Indeed, it is unlikely that control could be implemented at a level high enough to exterminate mink on most river systems.

5.6.2. How many mink are killed by hunting with hounds?

Our request for data from the Masters of Minkhounds Association has not been successful. Birks (1986) estimated that most hunts killed 40-50 mink in a season and that the annual nationwide total for the 20 mink packs was therefore 700-800. However, annual culls can range more widely, probably around 20-70 (R. Strachan, *pers. comm.*); this extrapolates to 400-1400 per annum, equivalent to 0.7-4% of the most recent population estimate of 35,000-60,000 mink in Britain (see section 10.4.2).

5.6.3. How many mink are killed by trapping?

We have only snippets of data on numbers of mink killed by trapping. The National Game-Bag Census (section 3.2.1.a) records the capture of *c.* 1,000 mink annually on *c*. 500 shooting estates. At a study site in Devon, a riverside landowner trapped and shot 34 mink on his 500m stretch of the river during 27 months (Birks, 1989) and killed 119 mink along this stretch over 5 years; transient mink appeared to bear the brunt of the campaign. In 1995, the average number of mink killed by 21 lock keepers was 12 (although there was wide variation within this) and the total bag for the River Thames locks was an estimated 200 (R. Strachan, unpublished data).

These data are insufficient to estimate the total number or the proportion of the population taken by trapping.

5.6.4. What influences the effectiveness of methods to control mink?

For maximum effectiveness, any mink killing should be carefully planned, tackling questions such as where and when effort should be prioritised, how large an area should be targeted, and according to what pattern of revisitation.

For example, mink might most fruitfully be controlled where water voles still maintain enclaves, and in areas of importance to conservation such as local, County Trust and National Nature Reserves, and National Parks. There are also some special cases: there are strong arguments for targeting the eradication of mink from the Western Isles and, if they were to arrive in mink-free islands (such as the Isle of Wight, Anglesey, the Isle of Sheppey and North Uist) that would be a cause for immediate action. On the other hand, the pattern in which to focus mink control in mainland Britain (both regionally and locally) is by no means obvious, considering the severe limitation of resources.

The timing of mink control is particularly important, first because it affects the removal of juveniles (via breeding females), second because at the end of the summer there are a large number of transients that are part of a ‘doomed surplus’ and therefore involve a waste of culling effort; and third, because the chances of recolonisation vary seasonally. Removing adult females before they settle and breed over the three months of the rutting period has the potential effect of preventing residency of breeding mink at a site for that year, as any female mink arriving later are unlikely to be mated and will remain so until the next rutting season (January-March) the following year. A nursing mink with a dependant litter has been demonstrated to have an impact on local vulnerable prey species (Macdonald & Strachan, 1999). Macdonald & Strachan (1999) suggest control is best achieved by targeting breeding females between January and April.

Thus, the effectiveness of hunting with dogs may be different early in the season (when females have dependent young, so that their removal will cause the loss of the whole litter) compared to later in the season. From mid-summer onwards, those mink killed are likely to be dominated not by adult territory-holders, but by pre-dispersal juveniles that are part of the ‘doomed surplus’. Removal of lots of young mink then may give the impression of control, but this is probably compensated by juvenile dispersal when hunting stops around September. Furthermore, the timing of the mink hunting season is such that it coincides with a period when mink activity along a waterway is actually reduced (many more adult mink can be encountered during the rutting period January-March; R. Strachan *pers comm.*).

An additional factor influencing effectiveness of trapping is trap density; where trapping is intensive and involves neighbouring landowners, it can remove the majority of mink locally. This has been shown along a number of chalk stream game fisheries (Macdonald & Strachan, 1999).

The relative catching efficiency of cage traps and spring traps is unknown. Many professional gamekeepers believe spring traps to be more efficient, as they are less labour intensive, but this belief is not founded in any systematic study.

5.6.4.a. How could we measure the effectiveness of mink trapping?

Accurately measuring the proportion of a population that is captured by trapping is very difficult. However, by live-trapping, individually marking and then releasing animals over a period of several days it is possible to compare the likelihood of recapturing different classes of animal (males, females, adults, juveniles) and to measure seasonal variations in this. A measure of trappability (likelihood of recapture) overcomes the problem of measuring effectiveness in areas with very different abundances of mink.

Researchers in the WildCRU have carried out such an exercise along 24km of the River Thames in Oxfordshire, which was occupied by an estimated seven mink each month (N. Yamaguchi, pers. comm.). Using *c*. 20-30 traps at any one time, 52 different individuals were captured on 192 occasions over 5299 trap-nights from February 1995to August 1997. On average *c.* 1.64 new mink were trapped per month. The ease with which mink were caught varied seasonally, and with sex. For example, in May, males were highly trappable (each individual was caught, on average, 2.3 times) while no female was recaptured. Conversely, in January, females were slightly more trappable (each was caught twice, on average) than males (captured 1.7 times, on average). From this experience, the researcher suspects that the use of 20 traps for 3 days per month would have been sufficient to keep the 24 km study area free of mink. While these data are certainly not sufficiently extensive to be of practical relevance, they nonetheless show an approach by which one could measure the effectiveness of trapping in different circumstances.

5.6.4.b. How cost-efficient is mink trapping?

No data are available on the cost-efficiency of mink trapping for population control. However, WildCRU researchers working on the River Thames in Oxfordshire set live-traps at 200m intervals, maintaining a set of

25 traps along 5km of a river for 4 days (N. Yamaguchi, pers. comm.). They checked each trap early in the morning, a task taking *c.* 2-6 hours by boat, including travel time. Baiting these traps consumed up to *c.* 600g of sprats daily. Moving the line of 25 traps from one section of river to the next took another day. Thus, 5km of river could be covered in 5 days. If we base our estimate on an annual cost of trapping, then in one year (approximately 200 working days), an operator could cover 200km of river containing (at densities of 0.25-0.30/km; N. Yamaguchi, pers. comm.) 50-60 mink. At a salary of £40 per day, £0.63 per day for bait, and £1/day petrol (depending on the boat), total annual salary and expendable costs would come to £10,508. If we assume a further annual cost of £1000 for a car, £400 for a boat, £400 for field clothes and miscellaneous, and £625 for traps (25 traps at £25 each), this comes to £2425 for equipment. The total annual costs then comes to £12,933 pounds. Based on experience, at these trap densities we expect to capture 50% of the population within our 100 trap-night period. If 50% of 50-60 mink are caught, the cost of trapping would be approximately £431-517 per mink.

Of course, this scenario is unrealistic in the context of predator control for several reasons. Many of the costs (field clothes, vehicle expenses, salary) would be spread over a number of tasks, and some would be one-off costs (such as equipment). Trapping would be targeted to the most effective times of year (January to April), when a higher trap success rate might be expected, and the target area would likely be much smaller. A more realistic cost might be estimated as follows. If an operator restricts his activities to three months from January to March, he can cover 60km, which would contain approximately 18 mink. If trapping success was higher at this time of year, say 80%, he might catch 14 animals. If he spent no more than half a day on mink control, his salary, bait and petrol would come to £1290. Assuming equipment and other costs were covered from elsewhere, this represents a cost of £89 per mink killed.

In arriving at these estimates, our intention is not to arrive at accurate figures, but merely to illustrate some of the issues that need to be considered when estimating cost-efficiency of trapping mink.

5.7. Conclusions

- We distinguish two important aspects to the performance of wildlife management practices: effectiveness and efficiency. '*Effectiveness*' expresses performance in terms of success in achieving the aims of management (e.g. reducing population size, reducing damage). '*Efficiency*' expresses performance as the success achieved for a given cost (e.g. in time, in effort, or financial).
    - ♦ An accurate measure of population size is an essential component of any measure of the effectiveness of population control (i.e. have the measures brought about the desired change in population size), or its efficiency (i.e. the cost of killing an animal known to be present). However, population size is notoriously difficult to estimate.
    - ♦ Landholder's perceptions of effectiveness and cost-efficiency may not be accurate. For example, farmers overestimate foxhunting bags by as much as 10-fold.
    - ♦ Commonly used measures of effectiveness (e.g. numbers of animals culled) and efficiency (e.g. financial outlay required to kill one animal) can be very misleading because they do not take into account the density of the quarry. Nonetheless, these measures are components of any estimate of effectiveness and efficiency.
- With the possible exception of red deer in the West Country, the data are not sufficient to calculate total numbers culled, or the proportion of the cull taken using each method; however, we do have relatively good data for organised methods of culling involving dogs (i.e. the various hunt Associations, and the National Coursing Club).
    - ♦ Registered packs of foxhounds and upland foot and gun packs probably take a cull in the region of 21,500-25,000.
    - ♦ Over the last five seasons, an average of 144 red deer were culled annually by the three Master of Deerhounds Association-registered staghunts, roughly 11-15% of the total cull required to prevent further population increases within the Staghunting countries.

- ♦ On average, registered packs and in National Coursing Club competitions together kill less than 2000 hares.
- ♦ We estimate that 400-1400 mink are killed by registered minkhound packs.

- If reducing numbers with the intention of protecting a game, agricultural or fisheries interest is a prominent aim, all the strands of our data suggest that hunting with dogs is generally less effective than the alternative methods, for all the species considered.
  - ♦ Possible exceptions include the use of terriers in fox culling, and the use of hounds and terriers in combination with methods involving shooting, particularly as practiced in upland Wales.
- Nationally, shooting foxes, deer and hares is the method which probably contributes most to population control, although there is regional variation in this contribution.
  - ♦ Shooting also contributes most to the local red deer cull within the Staghunting countries; the annual Staghunting cull is insufficient to control numbers in the area on its own.
  - ♦ Trapping is potentially the most effective method used to control mink.
- There are discretionary aspects to all control methods (e.g. number of hunt meets, number of stalking days), which have the potential to increase or decrease their effectiveness.
- Cost-efficiency analyses suggested that fox control using dogs was cost-effective for sheep farmers in mid-Wales. In the east Midlands, cost–efficiency of hunting with dogs was a more complex issue, with net costs for arable farmers and net benefits for others.

6. How acceptable are current legal methods to control populations of foxes, deer, hares, and mink?

6.1. What do we mean by acceptability?

The acceptability to society of methods of controlling mammal populations by culling is clearly an important issue. Indeed, it is concern by some sectors of society over the acceptability of hunting with dogs that has prompted this Inquiry. The difficulty with judging acceptability is that different groups will use different criteria. Among the criteria likely to influence acceptability of a control method are, in random order:

1. Its humaneness.
2. Its safety to people and to non-target species.
3. Public opinion.
4. The aims of control by that method.
5. How widely the method is used, and what the alternatives are.
6. The number of animals killed using the method and the impact on their population.
7. The effectiveness of the method in achieving the aims.
8. The quality of the information available to those making the judgment.

These criteria are themselves frequently intangible and difficult to measure, particularly humaneness. Even apparently straightforward criteria, such as the number of people using a method or the number of animals killed, may be difficult to quantify so that different investigators arrive at different results.

Much of our report is relevant to gauging the acceptability of hunting with dogs and other methods. Chapter 2 is pertinent to criteria 4 in our list above, while Chapter 3 deals with criteria 5. In Chapters 4, 5 and 6 we deal with criteria 6 and 7, while 8 is assessed throughout.

In this Chapter, we explore some of the factors that need to be taken into account when judging the humaneness and safety of the different control methods. For methods detailed in Chapter 3 we document aspects that might reasonably be expected to have a bearing on humanness, such as the length of time an animal is held in a cage trap, the speed with which it is killed, or the wounding rate. We also discuss studies relevant to assessing the physiological and behavioural effects of different control or restraint techniques. As background, we discuss ways in which we might gauge humaneness, and then report on the attitudes of the general public and of those who carry out the control.

6.1.1. How can we assess humaneness?

In order to assess the small amount of quantitative data relevant to the humaneness of different control methods we must first define our terms, and then understand the problems associated with the measurements made.

6.1.1.a. Definitions: what is humaneness?

Animals maintain an internal environment that is conducive to their proper functioning. This is done by homeostatic controls: a collection of feedback mechanisms that result in a dynamic but relatively stable internal state that can be taken as 'normal' for the animal. Such systems operate within an optimal and a tolerable range (Maas, 2000). Many external events and processes can perturb this normal state, and are known as *'stressors'*. Examples include temperature, crowding, confinement, exposure to unfamiliar conspecifics, and physical injury or disease. The impact of a stressor in disturbing the homeostatic equilibrium is correctly known as '*stress'*, while the processes of responding to stress in individuals subject to stressors are collectively the '*stress response'* (Broom & Johnson, 1993). These definitions have been widely taken up in the biological and medical literature; it is unfortunate that in everyday usage, stress has been carelessly adopted to include the stressor, its impact, and its eventual consequences if not satisfactorily dealt with. In this report we adhere to the strict definition.

Usually stress is quickly reversed, by means of the *'general adaptation syndrome'*, which involves three stages: *'arousal'*, '*adaptation'*, and (if the stress continues) '*exhaustion'* (Dawkins, 1980). The process of arousal is common to all kinds of stressor, and involves stimulation of the sympathetic nervous system, and the production of ACTH and cortisol. Adaptation involves different mechanisms appropriate to each kind of stressor, e.g. erection of hair, comfort behaviour, avoidance. If adaptation is unsuccessful, exhaustion will be the inevitable consequence. This emphasizes the important point that dealing with stress is expensive (in terms of physical effort, the chemicals manufactured in the body, and lost time for other activities) and may involve processes (such as running away) that are not conducive to normal life functions. Indeed, prolonged or repeated stress-response can itself lead to pathological conditions.

Nevertheless, in most circumstances the stress response is normal, beneficial and adaptive, and can usually be replaced with the word '*coping*' (defined as having control of mental and bodily stability - Broom & Johnson, 1993). An animal's '*welfare*' depends on the success of its attempts to cope with its environment (Broom & Johnson, 1993; Maas, 2000). A great number of inputs affect this ability, including phylogenetic and individual genetic background, and individual long and short term history. Welfare is not a directly quantifiable property, though certain behavioural and physiological measures are used as welfare 'indicators'. In accepted usage, there is a *welfare continuum* between 'very good' and 'very bad' in which good welfare is not categorically distinguishable from poor welfare (Broom & Johnson, 1993). However, by definition, prolonged stress, caused by a stressor that will not go away and cannot be avoided, translates into poor welfare and may involve suffering. *Suffering* is defined to occur when unpleasant subjective feelings (such as pain, fear, or boredom) are acute or continue for a long time (Dawkins, 1990; see below).

Finally, *'humaneness*' is defined as the quality of any action of humans towards animals (like culling) that causes no '*unnecessary suffering'*. Suffering occurs when unpleasant subjective feelings (such as pain, fear, or boredom) are acute or continue for a long time (Dawkins, 1990; see below). An accepted definition of 'unnecessary suffering' is illusive (e.g. Fox, 1986), but is nevertheless the legislative benchmark by which cruelty to animals is defined (Barnard *v.* Evans, 1925), and is a measure what society will accept and tolerate in its relationship with animals. If a person causes an animal to suffer unnecessarily, they may have committed an offence under the Protection of Animals Act 1911 or the Wild Mammals Protection Act 1996. Hunting is currently lawful because it is exempted in animal protection legislation.

6.1.1.b. How can we measure humaneness?

As defined above, there are two components that must be measured and balanced to decide whether an action is humane or not: the extent of the suffering involved, and how necessary that suffering is.

Humaneness is ultimately decided by society, usually taking the extent of suffering into account. For example, we sanction the use of strychnine for poisoning moles, despite widespread agreement that it causes a prolonged, painful death. Information pertinent to the necessity of the various control methods, in relation to their underlying strategies and aims, is presented throughout this report. Here we consider how science can help to assess the suffering component.

Whereas welfare is defined in terms of processes that can be measured, suffering is not even indirectly measurable. It can only be inferred from observational data, by supposing that it accompanies behavioural and physiological states in animals that are normally associated with suffering in humans. This important approach is in its infancy, hence interpreting objective measures of behaviour and physiology is not straightforward (Bateson & Bradshaw, 1997; Bateson, 1997; Harris *et al.*, 1999; various submissions to the Inquiry). Many animal welfarists argue that as an ethical principle the moral costs of wrongly assessing suffering should not be considered symmetrical: assuming an animal is suffering when it is not, is more desirable that the reverse

(Kennedy, 1992). In other words, the animal must be given the benefit of doubt. (For comparison, the relative undesirability of opposing and mutually exclusive errors must also be balanced in scientific hypothesis testing; however, this is usually decided in the context of each hypothesis, rather than by an ethical maxim as here.)

6.1.1.b.i. What are the difficulties associated with measuring suffering?

There are a number of key areas of contention in the use of behavioural and physiological correlates of stress to measure suffering.

First, is there a relationship at all? To what extent are animals capable of suffering? Summarising the enormous difficulties presented to scientists by this issue, Kennedy (1992) concluded that while "..*there is no direct evidence that animals suffer, the indirect evidence is so strong that the proposition can be accepted"*. Dawkins (1990) argues that an understanding of an animal's mental state is achievable through analogy with processes occurring in an animal and our own feelings in similar circumstances. Bateson (1991) endorsed this view and emphasised the importance of "*comparable mechanisms and comparable behaviour*". This indirect approach is currently accepted to be the only means of assessing suffering.

Second, suffering is widely regarded as a continuum between mild and severe (just as welfare is a continuum between good andy bad). We have little basis for locating measured correlates precisely on this continuum, although we may be able to locate them relatively (i.e. more or less severe). This relative location may nonetheless be used. For example, one can compare the physiological state of animals exposed to one method of attempted population control with the state of animals whose condition attracts a consensus view that they are suffering. This approach was taken by Bradshaw & Bateson (1999).

Third, a major interpretational difficulty is that of deciding whether deviations from physiological normality illustrate the capacity of the animal to cope with stressors or whether they represent an unnecessary increase in suffering. The ability to respond readily and rapidly to threatening stimuli may, after all, be one distinction between wild and domestic mammals. This possible distinction – namely that some domestication typically involves selection to make animals less 'jumpy' and more manageable (Hemmer, 1989) – would complicate the use of domestic species as models for studying stress in their wild counterparts.

6.1.1.b.ii. How useful are measures of suffering?

Is it better that one animal suffers considerably for half an hour, or that 25 animals suffer moderately for several weeks? Is it better to hold a population at a low level by culling or to let it find a higher level, given that the number of deaths occurring each year will be greater in the latter scenario? These questions are examples of the many major ethical issues that touch the hunting debate, and in the absence of a single metric, it is not obvious to us how to make these judgments.

The lack of a single currency for suffering makes assessing the relative humaneness and acceptability of some scenarios difficult. For example, if it is assumed that deer being killed by canids suffer, and that humans have a moral duty to take responsibility for their actions, one view might judge it absurd to discuss the acceptability of wolves killing deer, but valid to discuss the acceptability of domestic dogs to be encouraged to do so. Alternatively, others might ask whether it makes sense to apply the standards of laboratory or abattoir to wild animals, given the physical difficulties of achieving them, and the patent unpleasantness of any form of death in the wild. Potential answers, taking into account avoidability, human intervention, and even 'naturalness', involve fine distinctions that are seldom straightforward.

While these debates are central to judgements on hunting with dogs, and while some may be informed by science, they fall outside our remit in this report, which is restricted entirely to quantifiable aspects. Here, therefore, we will restrict our review to reporting numerical data.

6.1.1.c. Lessons from Biodiversity

Problems faced in evaluating the acceptability of hunting with dogs or any other control method in terms of humaneness are loosely equivalent to other perplexing evaluations facing conservationists, of which the one that has received most scientific attention has been the value of non-tradeable (incommensurable) facets of biodiversity. While biodiversity has enormous direct and passive use values, in this context the relevance is that also has intrinsic worth, irrespective of utilitarian values, and this intrinsic worth often has an aesthetic, cultural, spiritual or ethical aspect. What value is society to put on these non-market attributes when faced with the cost-benefit analysis of alternative policies? This question has stimulated a literature on indirect valuation techniques (described in Macdonald, in press).

These techniques employ statistical and survey methods to calibrate models of consumer choice. For example, expressed preference techniques include contingent ranking (people choose between baskets of options, thereby establishing their relative preferences for particular ecosystem services relative to goods that have an easily-measured market price) and contingent valuation (people state a monetary value that they would be willing to pay for an improvement in service, or would be willing to accept for a degradation in service). Alternatively, indirect preference techniques include aversive expenditure (how much people pay to reduce the magnitude of an impact), travel cost (the value of the service provided is assumed to be at least the cost in time and money spent in travelling to reach it) or hedonic pricing (which uses statistical methods to estimate the relationship between the market price and the attributes of the service). In the latter case, the impact of access to areas of outstanding natural beauty on the value of a house can be estimated through a statistical analysis of the variation in house prices with proximity to parks. Where valuation measurement techniques are not appropriate, either because of poor public understanding of the issues or because the preferences cannot be gathered, a fallback position is the so-called ‘Delphi technique’ namely to rely upon the consensus of a jury of experts. This, however, is increasingly unlikely to satisfy stakeholders where the environmental impacts are complex and are weighted against social and economic development benefits. It also rests on the demonstrable impartiality of the experts.

The nature of these valuations is clearly subjective, and thus Macdonald (in press) judges them a perilously flimsy basis for valuing biodiversity. Nonetheless, they provide a means of ranking peoples’ priorities in a way that might be adapted for the study of acceptability in the context of animal welfare. This approach to welfare issues would be systematic and at least somewhat quantitative, but we know of no attempt at this in Britain.

6.1.2. What are attitudes to controlling populations of foxes, hares, deer and mink?

Public opinion (whether well-informed or not) is an important component in the acceptability equation, and a driving force behind changing legislation. In the UK, most available data on public acceptability are collected through opinion polls. Few opinion polls conducted for political purposes apply significance testing to their results, quantify the effects of different approaches to their target audiences, or attempt to assess how well-informed respondents are. In short, they are often scientifically wanting. One should note that surveys restricted to well-informed or experienced people may also give a misleading image of the views of society if these are a small sub-set of the population. In the US, wildlife scientists have established a longer tradition of studying public attitudes towards wildlife issues (e.g. Messmer *et al.*, 1999; Cockrell, 1999; Manfredo *et al.*, 1999). Farmers’, game managers’, foresters’, fisheries managers’ and conservationists’ attitudes to controlling the different species have largely been covered in Chapter 2. Farmer’s attitudes to the humaneness of different methods is dealt with below.

Whether or not their judgements are consistent or rational, public opinion differs towards different species (and differs between people for any one species). Clearly, the public is not entirely hostile to foxes. In a questionnaire survey of 836 households in Oxford (24% response rate), 20% reported putting scraps on compost heaps, 68% put out scraps for birds, and at least 100 deliberately provisioned foxes (equivalent to four households feeding each fox in the city, even were the non-respondents to supply no food to the foxes; Macdonald & Newdick, 1982). Deer too are sometimes highly regarded by the British public, both where they are hunted with dogs (Exmoor Life Survey; Exmoor NP submission to the Inquiry), and where they are not (e.g. Scottish Natural Heritage, 1994). To judge by press coverage, feral American mink are almost unanimously unpopular, but there is considerable concern over the welfare of the same species in fur farms. Since peoples’ actions are linked to their opinions, it is worthwhile to quantify their views, and to ask whether they are well founded. In particular, does the general public perceive a need to cull the species in question?

6.1.2.a. Does the general public perceive a need to cull?

While we know of few systematic studies of public perceptions regarding the need to attempt control of British mammals, this so-called ‘human-dimension’ of conservation biology is growing in North America (e.g. Messmer *et al.*, 1999; Cockrell, 1999; Manfredo *et al.*, 1999). In Britain, there are data to show that, unsurprisingly, even farmers' perceptions of the fox as a pest depend on their particular circumstances (section 2.1.1.a).

We also have limited (and 20 year old) information on some of the factors that might affect the way people feel about culling foxes and some other wildlife. In 1978 Macdonald & Newdick (1982) delivered a questionnaire to 14,000 households in Oxford; 3468 (26%) were returned the following day. Of these, 201 were known to live within fox home-ranges, and were asked whether foxes needed to be controlled, where, and why. In Table 6-1 we compare these urban dwellers' responses with the national responses of 859 rural farmers in 1981 (see section 2.2.1.a.ii for survey details). Urban dwellers were much less likely to state that foxes needed to be controlled, either in towns or in the countryside, and were less likely to state than any of the listed motives were acceptable. These differences were statistically significant[21]. Urban dwellers were more likely to approve of the active conservation of foxes.

There was evidence that for our urban respondents, the acceptability of different motives and of killing foxes in principle was affected by their background. In a separate question, the respondents were asked whether they were brought up in an urban or rural environment. Country-bred respondents were significantly more likely to favour fox control in the countryside than were those brought up in the city (53 % compared with 46%)[22]. Similarly, they were more likely to approve of foxes being controlled to protect game and stock.

Table 6-1 Comparison of urban and farming sources for responses concerning fox control.

		Urban	Farmer
Where do foxes need to be controlled?			
	In the county	47.7	73.9
	In towns	61.9	70.7
Why do foxes need to be controlled?			
	To control disease	56.6	45.7
	To protect stock	48.7	67.6
	To protect game	14.4	44.5
	Foxes too numerous	21.1	65.1
Do you approve of fox control for these reasons?			
	To improve shooting	6.7	42.0
	For pelts	3.3	16.8
	For sport with hounds	11.8	68.4
Do you approve of active conservation of foxes?		46.0	19.3

While there is considerable variation in the perceived need to cull foxes, the need to control mink and deer using culling is very widely accepted. In the case of deer, all the Deer Initiative Partners (Forestry Commission, Forest Enterprise, British Deer Society, British Association for Shooting and Conservation, Royal Society for the Prevention of Cruelty to Animals, Country Landowners' Association, Department of the Environment, Transport and the Regions, English Nature, Game Conservancy Trust, Highway Agency, Ministry of Agriculture, Fisheries and Food, National Farmers Union, National Trust, The Wildlife Trusts, Woodland Trust, Timber Growers Association, NW England Association of Deer Management Groups) accept the need for culling in achieving sustainable management of deer. The need for culling by rifle is also fully accepted by a wide range of animal welfare organisations. The mission statement of The Deer Initiative (DI) Partners is: "*To ensure the delivery of a sustainable, healthy, well managed wild deer population in England*", and its objectives include promoting a responsible approach towards deer stalking within a framework of sustainable management, and encouraging sound deer management training.

In the case of American mink there is at least implied acceptance amongst the statutory agencies of a need to control mink to promote the conservation of their prey (Macdonald & Strachan, 1999). The need for anything other than occasional localised hare control is not widely accepted by conservation or welfare organisations.

[21] All $\div^2_{[1]}$>23, $P < 0.0001$

[22] $\div^2_{[1]} = 10.9$, $P < 0.001$

6.1.2.b. What are public attitudes to hunting with dogs?

A MORI poll on foxhunting, conducted by phone in July 1999 for *The Mail on Sunday* interviewed 801 adults across Great Britain by telephone; 69% lived in a town or suburb. Data were weighted to the known profile of Great Britain. When asked "*To what extent do you support or oppose a ban on hunting with dogs in Britain?*", 52% strongly supported a ban and a further 11% tended to support a ban. Only 14% strongly opposed a ban. Most people (69%) disagreed (47% strongly) with the statement that foxhunting is a necessary means of preserving the balance of wildlife in our countryside. There was a difference between urban and rural responses: of urban respondents, 20% either agreed or strongly agreed that foxhunting was necessary, while 39% of core rural respondents did so[23].

An October 1997 MORI telephone poll of 3010 people, weighted to the national profile, also found little public support for foxhunting. Of those polled, 73% supported Michael Foster's bill to ban hunting with dogs and only 8% were in favour of hunting with hounds. Almost three-quarters (74%) of all those questioned, and 64% of rural people, disagreed with the statement: "*hunting with dogs is necessary to control animal numbers, such as foxes*". Similarly, 60% of rural, and 74% of the total, disagreed with the statement: "*Hunting with dogs is an important part of the British way of life*". The poll found that hunting was not considered an important issue for countryside areas and communities.

In numerous other polls (see IFAW submission to the Inquiry) most respondents express anti-hunting sentiments. Britain's most respected animal welfare organisation, the RSPCA, is opposed to any hunting of animals with dogs or other animals.

6.1.2.c. How humane do farmers believe different fox control methods to be?

Our data for foxes in this section come from two questionnaire surveys, one covering 859 farmers from 10 regions in England in 1981, and the other covering 72 farmers in Wiltshire in 1995 (section 2.2.1.a.ii). Farmers were asked to say whether they thought certain methods of fox control were 'humane' (undefined). In both surveys, and all regions, shooting was consistently considered the most humane method of fox control (69% overall in 1981, 58% in 1995; Table 6-2); in 1995 49% considered it effective as well as humane. In 1981 a high proportion of farmers believed both hunting (55% overall) and gassing (49%) to be humane, but in Wiltshire in 1995 only 29% believed gassing was humane, although over half still thought hunting humane. Gassing has not been legal since 1987, and we speculate that its passing as a control method may have altered farmers' perceptions. (Several mechanisms are possible here. First-hand experience may formerly have convinced farmers that gassing was humane. Farmers today, who lack recent first-hand experience, may inevitably have unfavourable perceptions of methods that have already become illegal. Alternatively, the possibility of gassing being made illegal may formerly have caused farmers who saw it as an effective tool to defend it more strongly.)

We were also able to investigate if farmers' judgements on the humaneness of different methods, and the justification of different motives, was influenced by the damage they had sustained that they attributed to foxes, and the field sports they participated in. In Wiltshire, the proportion of farmers who considered each method to be humane did not vary significantly between those who had, and had not, designated the fox a pest on their farm (Baker & Macdonald, 2000). However, more farmers reporting actual stock loss to foxes in the previous year, said hunting was humane, than those who did not. This contrasts with findings in 1981 (Macdonald & Johnson, 1996), which suggested that farmers were more likely to think shooting, snaring, poisoning or the use of terriers humane if they had suffered losses to foxes, but their opinions of hunting and gassing were not affected. The differences between these studies could reflect regional variation, changes since 1980, or the smaller sample size in the Wiltshire study.

[23] $\div_{3} = 81.3$, $P < 0.001$; we gratefully acknowledge John Leeman of MORI for making the raw data available to us for this analysis

Table 6-2 Percentage of farmers replying 'yes' when asked whether they believed a method was humane in controlling foxes. (WestC. = Devon and Cornwall).

	Dorset	Leic.	Oxon	Shrop.	Suffolk	Sussex	Warw.	WestC.	Yorks	Overall (1981)	Wilts. (1995)
Gassing	50.4	50.6	48.2	58.3	53.3	55.3	54.8	35.5	43.8	49.2	29.2
Snaring	13.4	7.4	9.2	10.0	17.3	14.9	11.0	21.5	13.0	13.2	1.4
Hunting	56.7	55.6	60.3	33.3	53.3	51.1	61.0	52.3	59.8	54.8	52.1
Poison	11.8	9.9	12.8	3.3	8.0	2.1	4.9	8.4	7.6	8.3	8.3
Shooting	67.7	65.4	61.7	70.0	72.0	77.7	72.0	65.4	71.7	68.8	58.3
Terriers	18.9	13.6	22.0	20.0	29.3	22.3	24.4	35.5	20.7	23.0	9.7

Using 1981 data, farmers who had sustained damage by foxes were more likely to say that killing foxes to improve pheasant shooting or for fur were acceptable motives (Table 6-3). Damage had no effect on the likelihood of approving the active conservation of foxes. Hunting farmers were less likely to say that shooting and gassing were humane, and more likely to state that digging with terriers was humane. They were also more likely to approve of the active conservation of foxes.

Table 6-3 The effect of fox damage and hunting participation on the perceived humaneness of different control methods. % of respondents approving of the motive, or stating that the control method is humane.

		Fox damage?		Farmer hunts?		Farmer shoots?	
		No	Yes	No	Yes	No	Yes
Motive	Protect pheasants	35.3	56.5	43.8	39.5	22.9	62.8
	For fur	12.8	25.1	18.9	12.2	12.7	22.1
Humaneness	Shooting	70.7	77.3	80.5	59.1	76.4	77.4
	Gassing	48.8	62.0	59.9	38.7	56.6	64.1
	Snaring	9.1	24.6	14.3	13.4	11.4	18.3
	Hunting	59.0	59.3	41.4	91.2	44.0	44.0
	Poisoning	17.4	32.6	24.2	36.5	24.2	28.9
	Terriers	21.2	34.8	19.1	36.5	21.3	36.1

For the UK as a whole, 75% of farmers (including those who did not consider foxes a problem on their farms) said they would instruct their MPs to vote for 'no change' in the legislation governing foxhunting (Produce Studies Ltd, 1995; *n* = 831); 11% said they would instruct their MP to vote for a ban, while 14% held no strong view. Regionally, those in favour of no change varied between 86% (south-west England) and 56% (Scotland). Those in favour of a ban varied between 6% (south-west England) and 26% (Scotland).

6.2. How humane is hunting with dogs?

6.2.1. How humane is hunting foxes with dogs?

Three elements of foxhunting have welfare implications: the chase, killing by the hounds, and digging out with terriers.

6.2.1.a. How humane is the chase?

Foxes hunted above ground by hounds in England and Wales can be pursued for over an hour (Phelps *et al.*, 1997), although the average hunt lasts about half an hour (Macdonald & Johnson, 1996; Campaign for Hunting, submission to the Inquiry). Three-quarters of foxes 'found' evade capture (n=149 hunts, MFHA data, 1990-96).

In trying to assess the humaneness of pursuing foxes for such time periods, various aspects must be considered. One is whether they suffer fear and physical suffering whilst the hunt is on, and whether this is necessary. Another consideration is whether the foxes experience physiological stress sufficient to cause lasting harm and prolonged suffering, if they are not killed by the hunt. The potential for scientific research in this arena to contribute significantly to management and welfare has been richly illustrated by Bateson & Bradshaw's (1997) work on deer and the interest it has spawned. However, such studies are few and their interpretation not straightforward.

Below, we discuss a series of studies recording physiological changes in foxes that have been chased, trapped, or handled. The humaneness of trapping is discussed further in section 6.4.1, but the physiological data are included here because a comparison with chased foxes is pertinent to assessing the relative humaneness of hunting with hounds and leg-hold trappng (which is banned in Britain).

6.2.1.a.i. What physiological changes have been recorded in chased foxes?

Although it was preliminary work conducted under circumstances not directly equivalent to those being discussed in Britain, the publications from Minnesota by Kreeger *et al.* (1989, 1990) and White *et al.* (1991) are especially relevant to assessing the extent and nature of some of the physiological changes experienced by a fox being chased by hounds. Kreeger *et al.* (1989) used radio-telemetry to monitor heart rate and body temperature in 24 captive-raised red foxes held in a 4.1ha enclosure. Transmitters were placed abdominally, attached with sutures close to the xiphoid cartilage. Six behavioural states were assessed: sleeping, resting awake, hunting (foraging), feeding, running (alone or with a mate), and being chased by a dog for approximately five minutes. Table 6-4 shows their measurements for heart rate and body temperature (derived from Figs 1 and 3 in Kreeger *et al.*, 1989; note the number of foxes chased with dogs was only six, a precariously small sample for statistical analysis. Furthermore, chasing with dogs was the only activity which was anything like experimentally manipulated; it was impossible to control the amounts of time animals spent in other activities, or how much effort was put into them.).

Table 6-4 Heart rates (beats per minute) and body temperatures (^{o}C) in red foxes

Behavioural state:	Median heart rate	Median body temperature
Sleep	85	39.8
Awake	125	39.9
Hunt	163	40.2
Feed	210	40.6
Run	325	40.9
Chased	385	41.3

All differences in heart rate and body temperature for behaviours in this table were statistically significantly different from each other (P=0.0001, but test not explicitly described), apart from the difference between body temperature during voluntary running and while being chased by dogs. There was variation about these means

– for example, individual heart rate measurements for running exceeded the lowest chasing values, and individual body temperature measurements for running exceeded the highest values for chasing. Kreeger *et al.* note that the temperatures quoted were standardised to overcome calibration difficulties and allow comparison between behaviours, but did not accurately reflect either core body temperature or absolute temperature values.

6.2.1.a.ii. What physiological changes have been recorded in trapped foxes?

White *et al*. (1991) took heart rate, body temperature, endocrine, biochemical, haematological and pathological measurements of red foxes caught in box traps, and compared these with Kreeger *et al.*'s measurements for foxes caught in steel foothold (gin) traps, and with Kreeger *et al.*'s (1989) data for unrestrained foxes.

Heart rate and body temperature (measured by transmitter implants) in foxes caught by either foothold (310bpm) or box traps (290bpm) increased rapidly immediately after capture, but fell again to near pre-trapping levels after 80 minutes. Average heart rate for foxes 2-8 hours after foothold-trapping was 193bpm, significantly higher than when foxes were hunting (169.2bpm), but less than when running (318.5bpm). Similarly, the mean body temperature for foothold-trapped foxes 2-8 hours after trapping (40.9 ^{o}C), was significantly higher than for hunting (foraging) foxes (40.3 ^{o}C), but not significantly different from that of voluntarily running foxes (41.0 ^{o}C). However, maximum heart rate and body temperature, occurring during the first few minutes of capture, were 310bpm (in a foothold trap) and 40.7ºC (in a box trap) respectively.

Endocrine, biochemical, haematological and pathological values measured in captured foxes reflected a mixture of psychogenic stress response, physical injury, and exertion, and were less pronounced in foxes caught in box-traps than in leghold-traps. Kreeger *et al.* (1990) concluded that "*red foxes caught in foothold traps developed 'classical' stress responses characterized by increased heart rate, increased HPA (hypothalamic-pituitary-adrenocorticol) hormones, elevation of serum chemicals, and neutrophilia*." Alkaline phosphatase, lactate dehydrogenase and aspartate aminotransferase were significantly elevated, especially in foxes caught in leg-hold traps; leukocyte counts were higher and there was significant neutrophilia and lymphopenia. These findings were interpreted by the authors as resulting from prolonged physical activity during the 8 hour restraint period and from the stress response itself (elevated cortisol and activation of the sympathetic nervous system). Creatine kinase levels (usually elevated in the case of skeletal or myocardial damage, and during prolonged exercise) were similar to those of untrapped foxes, but elevated aspartate aminotransferase levels were thought to reflect increased muscle cell leakage due to physical exertion.

6.2.1.a.iii.What physiological changes have been recorded in handled foxes?

Moe & Bakken (1997) provide observations on the physiology of silver foxes that were physically restrained by human handling for one hour, a situation that would generally be accepted as stressful. (Silver foxes are a strain of red fox bred for their fur colour and we cannot assume their responses are always comparable with those of wild red foxes. Behavioural variation in foxes is known to be genetically linked with coat colour such that '*red foxes have more fear than silvers*' (Keller, 1975). Furthermore, any captive breeding exercise is likely to involve conscious or unconscious selection.). Rectal temperatures were monitored, and blood samples were taken at five minute intervals for measurement of plasma cortisol, plasma glucose and leucocyte number. Rectal temperatures varied according to familiarity with the technician doing the measurement, but could not be measured before the onset of stress. Deep body temperatures (measured with transmitter implants) in six undisturbed foxes ranged between 38.0-38.4ºC with some diurnal variation.

Handling caused plasma glucose, cortisol and rectal temperatures to increase (rectal temperature by *c.* 0.7ºC), and leucocyte number to decrease. Foxes began to pant after 15-20 minutes. Deep body temperature for a single individual exposed to the presence of a human showed an immediate rise of c 1.0ºC, which returned to normal within 30 minutes when the human withdrew. The magnitude of these temperature changes is not large by comparison with known within-individual variation for mammals. However, Moe & Bakken (1997) also report a maximum rectal temperature of 42.0ºC in foxes after capture and handling; the relationship between rectal and deep temperatures is not-known. Other physiological changes were interpreted as indicating activation of the hypothalamic-pituitary-adrenal (HPA) and sympatho-adrenal medullary (SAM) systems, both potentially indicating a stress response. As a universal component of the stress response, elevated plasma cortisol is considered particularly relevant as a welfare indicator (Broom, submission to the Inquiry).

Bakken *et al.* (1999) made further observations on how different stressors affected deep body temperatures of silver foxes. Acoustic stimuli caused the least temperature elevations, while the greatest were associated with handling and the presence of unfamiliar foxes. Although physical activity increased both heart rate and body temperature, the authors concluded that a state of stress-induced hyperthermia was recognisable that was unrelated to physical activity and that this "*may be an expression of an emotional response, such as fear or anxiety*".

6.2.1.a.iv. How should these data be interpreted?

These data together take on a tantalising significance because they are all that exist in the context of stress measurements for foxes. For example, the observation that heart rate and body temperature were both higher in chased foxes than those held in leg-hold traps (section 6.4.1) is noteworthy insofar as the latter were made illegal in the UK on the grounds of cruelty under the Animal Protection Act 1911. One might thus ask what can be inferred about the relative stress and the relative distress induced by the two methods. One view might have it that the observed higher heart rate and body temperature in chased foxes could have been correlated with parallel higher rates in other aspects of stress-related physiology. A different view might be that heart rate is likely to be near maximal under any circumstances when a confined animal tries to evade capture by a predator, and thus may be unshackled from some other biochemical measures of stress. So far as we know, neither the maximum heart rate of foxes, nor its corollaries as an index of stress, have been studied, making these possibilities amongst the many important questions raised, but not answered, by these studies.

A further illustration of our dilemma, as ecologists (physiologists may know the answers), is whether to be more impressed by the observations that, at 385bpm, the heart rate of foxes being actively chased was 18% higher than that of foxes running voluntarily, or by the observation that the heart rate of those running voluntarily was only 18% less than those being chased.

Similarly, how significant is the median body temperature of 41.3ºC measured in chased foxes? While it seems eminently plausible that very high body temperatures are indicative of stress (as defined above), precisely how to calibrate this is another important topic that is beyond our expertise. As background, body temperature in mammals is typically dynamic over a range of 2-3 ºC when awake. Absolute temperatures of 45ºC and above can cause protein breakdown, although so-called 'heat shock' proteins act protectively to stabilise other cell proteins at temperatures below this. (T. Helliwell, pers.comm.). Heat stroke in humans is associated with body temperatures of 40.6ºC and above, but is complicated by the inseparable effect of dehydration. 'Hyperthemia-induced mortality' in captured wild animals is associated with core temperatures of 41ºC and above (Spraker, 1982), though it involves a complex of processes as well as the extra factor of restraint following capture. On the other hand, core body temperatures and tissue temperatures of 41ºC and above are observed in mammals and humans apparently without damage occurring. For instance, abdominal temperatures of 40.5ºC have been measured using radio-telemetry in exercising free-living red squirrels (J.C.Reynolds, unpubl.) and body temperatures of 41ºC have been recorded in human athletes (Oxford Handbook of Medicine; R. Harris, pers.comm.). Hodgson *et al*. (1993 - cited in Harris & Bateson, 2000, Report on Contract 7) recorded *muscle* temperatures of 43.9±0.3ºC (mean±s.e.) in horses after 10 minutes of treadmill exercise.

With regard to the specific question of the humaneness of hunting, the obvious question is whether the chasing situation studied by Kreeger *et al*. (1989) is a good model for hunting with hounds in Britain? The answer is probably not: the foxes studied were chased by dogs for five minutes in a large (4ha) enclosure, whereas hunted foxes can be pursued for over an hour, though more usually for about half an hour. Unlike Kreeger's captive foxes, wild foxes can and do evade capture. Even if chasing in an enclosure were a good model, we do not currently have a full enough picture of the normal range of any of the physiological parameters, either within or between individuals, to interpret measurements so as properly to inform judgements on welfare. This seems likely to be an area where further research may well advance our ability to make such judgements

While we believe that physiological techniques have great potential to contribute to assessing animal welfare, we are also mindful that absence of data should not blind us to the obvious: if foxes are capable of suffering at all, then the circumstances when they are most likely to do so is when they seek to escape imminent death. Physiological data on stress may be unable to reveal whether non-human animals experience suffering, but if they do suffer, and if suffering is a corollary of stress, then the moments immediately prior to capture by dogs must surely be amongst the most severe.

6.2.1.b. How humane is killing by the hounds?

On average, approximately 64% of fox kills are made by the hounds (section 5.2.3.b.ii), notwithstanding substantial differences between hunts in the proportion of their tally killed by hounds above ground or terriers underground.

One argument advanced regarding the humaneness of a death delivered by the hounds, is that it is ‘natural’. As defined above, humaneness is a property of human actions which avoid unnecessary suffering; in this case the human action is to use a pack of specially bred and trained dogs to kill a fox. Whether or not the mechanism by which the fox is killed is similar to that used by wild canids is irrelevant to humaneness because the kill occurs because of the actions of humans.

In contrast, it is relevant to ask how much suffering is involved for the fox killed by hounds, and proxy measures of this might include time to death and cause of death (dismemberment versus crushed skull or spinal column). There is evidence that both methods of despatch occur (by biting at the body: Edwards, 1999; Hansard, 1997; RSPCA and LACS submissions to the Inquiry; biting at the back or neck: Cunningham, 1999; Thomas & Allen, submission to the Inquiry), but no data on their relative frequencies.

6.2.1.c. How humane is the use of terriers?

In 30-40% of cases where a hunted fox is killed, death involves the use of a terrier either to kill the fox underground or to locate it or flush it out so it can be shot or killed by hounds (section 5.2.3.b.ii). There is considerable regional variation and in some regions the majority of kills involve terriers. Where terriers have marked the fox, digging down to it can take anything from ten minutes to three hours (Phelps *et al.*, 1997). In some cases, fox and terrier fight underground, which can result in serious injuries to the fox before it is killed (and also to the dog). We know of no data on the frequency with which this occurs or on the severity of the injury.

If a rifle is used to kill adult foxes at the cubbing earth, terriers may be the only way to ensure that dependant cubs are killed.

The National Working Terrier Federation have a code of conduct that identifies best practice for humane and efficient use of terriers for pest control.

6.2.2. How humane is hunting deer with dogs?

This is the subject of a separate contract to the Committee of Inquiry.

6.2.3. How humane is hunting hares with dogs?

Hares are hunted with dogs in three main ways (section 3.4): on foot with packs of hounds; in coursing competitions with pairs of dogs; and on an *ad hoc* basis with single dogs or small groups of dogs. We know of no data relevant to the humaneness of coursing outside organised competitions.

6.2.3.a. How humane is hunting with hounds?

One of the objectives of the Association of Masters of Harriers and Beagles is to encourage the maintenance of ‘a healthy and balanced population of hares’; control is not an objective. The AMHB imposes a closed season from 1st April to September.

According the joint AMHB and MBHA submission to the Inquiry, a hare hunt is usually restricted to about 1-2 square miles, and lasts for an hour to an hour and a half. In 1998/99 beagle and harrier packs recorded sighting 30,000 hares of which 1650 were killed.

6.2.3.b. How humane is competition coursing?

There have been three independent inquiries into hare coursing, the most recent being the House of Lords Select Committee, 1976, but none has resulted in legislation against it. In 1971, at the instigation of the British Field Sports Society, Stable & Stuttard examined coursing. They made nine recommendations, which were accepted by the National Coursing Club (NCC), and have reduced the proportion of coursed hares that are killed from around 20% to around 13% (NCC submission to the Inquiry).

Four aspects of competitive coursing have welfare implications: driving the hares into the field, the chase, the kill, and translocation of animals. Wild hares are driven by a team of beaters from adjacent farmland into the coursing field. Hares are sometimes translocated from one part of the country to another to provide sport. We know of no data from which the stress of these activities can be evaluated, but transportation of domestic animals is widely acknowledged to be one of the most stressful husbandry operations, for which extensive legislation exists (to avoid distress).

6.2.3.b.i. How humane is the chase?

Coursing dogs are not released to chase the hare until it is at least 80m away, and the 'slipper' must consider that the hare is 'in a fit condition to be coursed'. An average greyhound course lasts 35-40 seconds (NCC submission to the Inquiry) and covers an estimated third of a mile. The Deerhound Coursing Club (submission to the Inquiry) state that a course that lasts two minutes would be considered exceptionally long. We know of no independent, systematically collected data pertaining to the mean length of chases or the state of animals after the chase.

NCC rules stipulate that the hare must not be enclosed, nothing must hinder its escape, and it must have 'sufficient knowledge' of the ground. On some coursing grounds special refuges called 'soughs' are provided to help the hare to escape. The dogs chase by sight and stop when the hare is out of sight (NCC submission to the Inquiry).

6.2.3.b.ii. How humane is the kill?

For the 1990/91 and 1998/99 seasons, an average of 13% of the hares chased were killed (taken from figures from coursing inspectors' reports provided by the NCC submission to the Inquiry). Hares are killed both by the dogs and by four 'pickers-up', who, under NCC rules, are positioned two on each side of the course and have a duty to ensure that hares are killed quickly and humanely. There is no statutory close season for the hare, but the NCC imposes one between March 11th and September 14th in any year.

The only study of the injuries sustained by hares killed during coursing was undertaken by the Universities Federation for Animal Welfare (UFAW), on behalf of the Joint Advisory Committee on Hare Coursing. UFAW staff attended 17 coursing meetings during the 1977-1979 seasons. In total, 53 hares (16 mountain hares and 37 brown hares) were obtained from these meetings, and detailed post-mortem examinations were carried out at the Department of Animal Husbandry and Hygiene, Royal Veterinary College. In addition, a very small number of brown hares were shot with shotguns. The nature of the injuries inflicted are summarised in Table 6-5.

Of the 53 hares killed, 43 had neck injuries, 18 (42%) of which were inflicted by the handler (as evidenced from a clean break and no teeth marks). No clean breaks were believed to have been caused by dogs (where tooth marks were evident). The UFAW team's assessment was that all chest injuries would have been quickly fatal (in six cases these included a punctured heart); 10 animals without neck injuries had chest injuries. Abdominal injuries included six punctured livers, but generally involved a ruptured gut. In the UFAW team's opinion, hindleg and back injuries could have been extremely painful until chest or neck injuries were inflicted.

Of the five hares shot, two had their necks broken by the shooter, and one sustained neck injuries; one had to be shot twice. All five had chest injuries, and four had hindleg injuries. While the small sample size precludes any valid comparison with coursed hares, it does indicate that shotguns will also cause multiple injuries.

Table 6-5 Percentage of injuries sustained to different parts of the body of hares killed by coursing and by shotgun.

	N	% Injuries sustained to:				
		Neck	Chest	Abdomen	Hind leg	Back
Coursing	53	81	77	79	74	58
Shotgun	5	60	100	40	80	20

6.2.4. How humane is hunting mink with dogs?

According to popular descriptions (Hunting with Country Illustrated, October 1997, The Countryman's Weekly, 24 December 1999, Countryweek Hunting, October 1996) mink hunts take an average of 1 hour 46 minutes and range from 20 minutes to five hours. Mink are either killed when they are caught by the hounds, or are shot or dislodged when treed, or are dug out and shot or given to the hounds. Generally, about a third to a half of the mink found are killed. Hunting takes place during the summer when females are likely to have dependent young.

6.3. How humane is shooting?

The issues involved in judging the humaneness of shooting are: the speed at which an animal dies after being shot; the likelihood of wounding; the chances of shooting a female with young, thereby leaving them to starve to death. As discussed earlier, shooting is widely held to be a humane method of control in skilled hands. Unlike the use of hounds and terriers, where the chase and kill are largely beyond human control, shooting is susceptible to at least some human influence on standards through training, codes of conduct and target practice. In Britain, training courses and target practice are voluntary for practitioners.

Some animals are undoubtedly wounded and left to die, but, with the exception of work by Bradshaw and Bateson (2000) for deer, there are no quantitative scientific data about the likely extent of wounding in Britain. Sainsbury *et al.* (1995) reviewed instances where animal welfare is compromised, and concluded that cases of prolonged pain and distress could be caused by bullet and gunshot wounds. They cite studies in America (where there is little regulation of shooting compared with Britain) that indicate that between 8-42% of deer were wounded rather than killed instantly.

As Broom (submission to the Inquiry) concludes, the welfare implications of shooting require further study.

6.3.1. How humane is shooting foxes?

Foxes are shot using either rifles or shotguns (section 3.2.3).

6.3.1.a. How humane are rifles?

Rifles appropriate to cull foxes are high powered centre-fire .22/.250, or .243 calibre. The single projectile means that rifles are not appropriate for shooting moving targets, except in very practised hands. The large explosive charge and relatively small (50 grain) bullet give a trajectory that is flat enough to ensure that range judgement is a minor consideration up to a distance of 250-300m. In night shooting there is rarely time to use a rangefinder for distance estimation. Although vertical drop of the bullet is not a practical issue, small bullets of this kind are relatively susceptible to strong winds. (Low velocity centre-fire .22 rifles are not suitable for foxes, except for humane dispatch at very close range.)

The target accuracy of shooting foxes with a rifle in field conditions is unknown. Although a heart or head shot is the ideal, each offers a target only about 4-5cms across. At a distance of 100m, the main body of a fox (assumed 40cm excluding head and tail) will subtend an angle of 0.23 degrees. A movement of 3mm by the rifle operator will sweep the entire length of the fox. (For comparison, the British Deer Society's Level 1 deer stalker's certificate requires candidates to place shots within a 10cm circle at 100m, using a rest for the rifle.) Under field conditions, the operator will usually have a rest, but is often standing on the back of a 4WD vehicle that rocks slightly in wind, the fox may be stationary for only 10 seconds or so, and shots may be taken at distances up to 250m.

Given the obvious potential for missing vital target organs, one must consider the consequences of the bullet striking other parts of the fox's body. Because soft-nosed friable bullets are the norm, extensive wounding normally occurs wherever the bullet strikes on the fox's body. Experienced operators state that most foxes are so badly wounded by a high power .22 round - wherever it strikes - that death is effectively instant and on the spot. The only admitted exceptions to this are if the bullet strikes the lower jaw or a leg – wild foxes can and do survive complete amputation of one leg (J.C.Reynolds, pers.obs.). Implicit in such opinions is the assumption that operators can reliably judge (by sound or by the fox's behaviour) when the bullet has hit the fox.

There are no hard data to suggest what proportion of foxes hit by bullets might escape with lethal or other wounds. The analysis by Bradshaw & Bateson (2000) suggesting a wounding loss of 2% in deer stalking (see section 6.3.2) is not really relevant here: deer present a considerably larger target, are shot in daylight, and typically at closer ranges than foxes. The deerstalker has a vested interest in placing his shot carefully to ensure that the animal is recovered and that the meat is edible. The fox shooter must usually operate by artificial light at night, is aiming at a target about half the size of a roe-deer and is motivated solely to cull the fox. One might therefore expect a higher wounding loss in fox shooting than in deer shooting, although risky shots are against the operators' interests because they can lead to 'lamp-shy' individuals.

The GCT's Fox Monitoring Scheme (section 3.2.1.c) requires that contributors record the number and sex of culled foxes on a daily basis. For 90% of foxes recorded as 'killed' using rifle and spotlamp at night, the sex is recorded. This suggests that the remaining 10% were hit (bullet heard to strike, or fox seen to drop), but were not examined for some reason. One interpretation could be that up to 10% of foxes believed to be hit actually escaped wounded. This could overestimate wounding losses because there are other possible reasons for not recording the sex (e.g. laziness of the operator; fox dead but not found because it drops in thick cover or stubble; fox the other side of a large river, etc.). On the other hand, it may underestimate wounding losses if operators do not always perceive that the fox has been hit.

Baker & Harris (1997) suggest that if wounding with guns were a significantly frequent occurrence, foxes with shot wounds would be more frequently treated in wildlife hospitals, but this argument is unsafe. Even if data on different types of injuries were collated, we have no basis for knowing what proportion of foxes wounded by shooting (or other means) are likely to be taken to a wildlife hospital, and therefore no way of extrapolating from numbers in hospitals to numbers wounded in the field. However, we note that foxes taken to wildlife hospitals are likely to be biased towards urban and traffic-related situations where members of the public would encounter the afflicted fox.

In summary, although shooting foxes with a rifle is widely perceived to be a clinically accurate and thus humane method of culling, data to verify this view are few.

6.3.1.b. How humane are shotguns?

Shotguns are more useful than rifles against a moving target, but are effective only at close range. As range increases, the risk of non-fatal wounding also increases. (The number of shot per unit area of the 'pattern' decreases with increasing distance. A cartridge can hold more small shot, giving a denser pattern, but their momentum falls below a critical threshold at a lesser distance than with larger shot. Larger shot hold their momentum better, but because there are fewer of them the 'pattern' will have bigger gaps. Either way, wounding can result.) For this reason one British code of practice recommends a maximum range of 20m (BASC, 1996), though an Australian handbook suggests up to 35m (Saunders *et al.*, 1995).

Generally speaking, opportunities to shoot foxes at such close range occur only at cubbing earths, or when foxes are driven towards the guns. Drives of this kind are necessarily daytime exercises. If hounds are used, wounded foxes may be trailed and caught by them. Occasionally, foxes may be caught by hounds before they pass the line of guns.

6.3.1.c. How humane is shooting at the cubbing earth?

Culling in spring and summer is especially effective because, fox for fox, it has the greatest impact on population growth at this time, and because fox predation on lambs, wild game and other wildlife is concentrated during this period. However, a spring/summer culling policy carries an increased welfare cost associated with the need to locate and destroy orphaned cubs.

The following example, using data collected by the GCT Fox Monitoring Scheme (detailed in section 3.2.1.c) illustrates the likely scale of this cost. Of 707 vixens killed in 1996 by a sample of 60 gamekeepers, 179 were killed during the period 10th March to 21st June in which births occur, and roughly half of these would have given birth by the time they were killed. Thirty-nine were killed at the earth, and attempts made to destroy the cubs; 88% of cubs seen at the earth were killed. The average litter size counted was 3.74. A further 31 litters were destroyed, but no vixen associated with them was killed at the earth. From this we can calculate that 162 cubs are likely to have died through lack of maternal care due to the cull of 179 vixens.

This calculation takes no account of the existence of sub-dominant vixens who may nurse and provision the litters of dominant vixens, though it is unlikely that more than one-third of vixens killed were non-breeding helpers in this way. Arguably, the loss of sub-dominant vixens - and of adult males, who also provision cubs - would also reduce cub survival.

Thus, although logical and effective for a number of aims, the strategy of culling during spring and summer has a welfare cost not shared by culling in other seasons (though this is no worse for foxes than for any other species). It is noteworthy that a closed season for lactating female badgers is deemed necessary even under the extreme imperative to cull badgers for attempted control of bovine tuberculosis in cattle (and for the sake of a nationally important MAFF experiment).

6.3.2. How humane is shooting deer?

Deer are the one species group for which we have data with which to assess the humaneness of shooting: Bateson & Bradshaw (2000) asked how often stalkers (using rifles) wound red deer, and Bradshaw & Bateson (2000) compare the welfare implications of culling deer by hunting with hounds and by stalking. These papers are based on essentially the same data relating to wounding by stalkers (summarised in Table 6-6), and both sought to estimate wounding rates in four ways:

- Eight stalkers were asked to recall how many deer they shot and wounded the previous (1995/96) season. Of 372 deer killed, 3.5% escaped wounded.

- A further seven stalkers (all experienced marksmen) were able to provide records kept at the time of culling for the 1995/96 season; none of the 171 deer killed escaped wounded.

- Six carcass dealers recorded the position of bullet and other wounds for 40 deer, and on the basis of these diagrams seven stalkers estimated how far each deer ran, how long it survived, and whether it would have required a second shot. Two of the carcasses examined had old shotgun wounds, but none had rifle wounds.

- Records for injured or diseased animals which hunters and stalkers are called out to kill in South-West England were examined. Of records for 23 deer in 1995/96, three had rifle wounds and one had a shotgun wound. Devon and Somerset Staghound hunt records for a further 88 deer, including those found dead (but excluding those killed by poachers), suggested that about 20% had been wounded by stalkers. On this basis, and assuming there are 2500 deer on Exmoor, an estimated 4.5% of shot red deer escaped wounded

Table 6-6 Estimated wounding rates for red deer using records from various sources (data from Bateson & Bradshaw, 2000). Weighted means were calculated from data obtained from stalkers memories and records, and carcass analysis (Bradshaw & Bateson, 2000).

	Stalkers' memories	Culling records	Carcass analysis	Weighted mean	Injured deer
Mean % surviving >2 mins	7.5	4.4	14.6	7	
Mean % killed with >1 shot	12	9.8	10	11	
Mean % escaped wounded	3.5	0	0	2	4.5
Number of deer	372	171	40		111

Welfare of stalked deer could be compromised in ways other than wounding, for example by disrupting the normal behaviour of the deer, or by leaving orphaned calves. (However, the statutory close season for red deer [1st March to 31st October] lasts until calves are generally considered 'weaned', even though lactation may continue into the winter; the extent to which calves orphaned after October are disadvantaged is unknown, but is an issue which applies equally to shooting and hunting).

A variety of organizations, such as the British Deer Society, BASC and the St Hubert Club, have for many years been running deer management training courses and stalker certification schemes. Although training in deer stalking and management is not currently compulsory in this country, the Deer Initiative and the

BDS/BASC Code of Practice both state that 'culling must be carried out safely, legally and humanely'. To that end a new progressive system of voluntary qualifications based on the standards laid down in the government's National Standards has been introduced through a new awarding body, Deer Management Qualifications Ltd. These qualifications – The Deer Stalking Certificate Levels 1 and 2 - enable deerstalkers to demonstrate that they have the necessary theoretical knowledge and practical skill. The qualifications are increasingly recognized by landowners and managers and are endorsed by the Deer Initiative.

6.3.3. How safe is shooting?

6.3.3.a.i. How safe is fox culling with a rifle?

A code of practice has been produced by BASC, instructional videos are commercially available, and practical range training and instruction is obtainable through BASC and at local rifle clubs. No training is required to cull foxes with a rifle in Britain. However, almost anyone with authority to cull foxes can obtain a firearms certificate, buy a rifle, and begin. There is no officially recognised qualification for fox shooting and no mandatory test of proficiency. (On the other hand, shooters usually learn the ropes in the company of other, more experienced sportsmen, just as young foxhunters learn in the company of the experienced members of the Hunt.)

Safety is primarily a matter for the operator, though police firearms officers in some constabularies will view the ground from a safety angle before issuing a new firearm certificate. Because bullets are dangerous to humans and livestock over very large distances (several miles), terrain limits the number of locations where shots may safely be taken. A solid back-drop behind the target is essential. Counter-intuitively, this is easier to ensure in flat open country from the back of a 4WD vehicle, than in hill country. Rifle shooting in hill ground where the distance between valley sides is >250 m, and where visibility on the hillside is limited by convex or rocky terrain, is severely constrained. Similarly the presence of livestock, small field size, many gates, poor vehicular access, dense cover, rough terrain and steep slopes can all make fox culling with a rifle unworkable.

Quarry identification is a serious safety concern for night shooting with a rifle and spotlamp. The fox is discovered in the first place by the reflection of light off the back of its eyes. Cats and dogs, deer, sheep, badgers, and owls are all potentially confusable by their eye-shine. Both cats and dogs may be accompanied by owners. The BASC Lamping (Night Shooting) Code of Practice (BASC, 1996), and other training material, emphasise that target identification at night should be made by body shape, not by eye-shine.

Intuitively, one would expect a trade-off between safety and efficiency, such that efficiency is sacrificed in order to attain a high standard of safety.

6.3.3.a.ii. How safe is deer stalking?

The potentially increased risk to the public through shooting if hunting deer with dogs ceases is often quoted as a negative impact of a ban. Clearly, high regard must be given to minimising risks associated with firearms, but this must be seen in the context of present shooting activity. Should a ban be introduced, the additional numbers of deer likely to be culled annually by shooting (<200; see section 5.4.3) are minimal compared to even just those 1000 deer already culled by shooting within staghunt countries. In the event that red deer numbers would decline and be tolerated only at lower density levels following any future ban on hunting with dogs (as argued by e.g. MDHA, 2000), any annual culls taken by shooting would clearly also decline in the longer term.

It is also sometimes suggested (e.g. Exmoor NP submission to the Inquiry) that deer stalking could not be done safely on Exmoor due to the large numbers of tourists and general public in the area. However, Exmoor is far from unique is this respect. In The New Forest, an even more popular tourist area (with an estimated 7-20 million visitors per year: Ecotec, 1992; Portsmouth University Survey, 1996), annual deer culls by Forestry Commission rangers have exceeded 750 throughout the last decade (Putman & Langbein, 1999); no accidents involving public have occurred during those culls.

Similarly, the British Deer Society (BDS submission to the Inquiry) are not aware of any serious accidents and state: "*With the exception of the deer packs in the Exmoor area, control of deer numbers throughout Britain is achieved by stalkers, professional and recreational, using firearms. They may operate as individuals or as* *members of a Deer Management Group. The safety record of stalkers is good. We are not aware of any accidents involving stalkers and firearms in the countryside*."

As outlined above, a variety of organizations run deer management training courses and stalker certification schemes.

6.4. How humane is trapping?

The issues around which the humaneness of live-trapping revolves include the length of time the animal is in the trap and, related to that, the extent to which it suffers hypothermia, hunger or thirst; the extent to which it is stressed simply by being held captive; and the potential for injuries to occur if it tries to escape, for example, by attempting to dig out. With kill traps, the most important issue for humaneness is the speed and manner of death. Another important consideration of both types of trapping is the extent to which non-target species might be caught. This is particularly a problem with mink traps, as there are a number of other similar-sized species that could be caught.

6.4.1. How humane is trapping foxes?

In England and Wales, only live-traps may be used to catch foxes, which must then be killed by shooting or with a humane killer. In our experience, it is generally difficult in Britain, even in urban settings, to lure foxes into box traps (this has not been our experience elsewhere in the world, and apparently reflects local selective pressures due to man). In addition to the frequency with which live traps are checked, their construction is also clearly a welfare issue, (foxes are likely to snap their canine teeth on mesh, and injure themselves on sharp surfaces). In their study of a different species, American swift foxes (*Vulpes velox*), Moehrenschlager *et al.* (in press) found that trapping and handling protocols could minimise injuries to live-trapped animals.

Data on the physiological changes that occur when foxes are trapped are presented in section 6.2.1.a.ii and discussed in section 6.2.1.a.iv. On the basis of these changes, White *et al.* (1991) concluded that marked stress responses occurred in both foothold and box traps. The principal difference between capture in box traps and in foothold traps was physical trauma to the restrained limb in foothold traps, and the physical exertion of struggling against the foothold trap, compared with less violent physical activity in the box trap. Foxes caught in foothold traps pulled against the trap for 13-18% of the time they were held, equivalent to *c*. 15-86 minutes (Kreeger *et al.*, 1990). Unpadded foothold traps caused limb injuries, giving rise to higher values of those serum measures that reflect tissue damage, than for foxes caught in padded foothold traps.

Leg-hold trapping of foxes is illegal in the UK, although an abundant literature on injuries associated with it exists elsewhere (e.g. Kreeger *et al.*, 1990). Interestingly, there appears to be a cultural difference in perceptions of humaneness between Britain and much of continental Europe and North America: in Britain leg-hold traps were banned long ago on grounds of cruelty, whereas snares are still acceptable; elsewhere, snares were generally banned long ago on grounds of cruelty, whereas leg-hold traps generally remain in use. In Sweden, neither are allowed, but leg-hold snares are allowed.

6.4.2. How humane is trapping mink?

Mink can be caught in live or kill-traps set in natural or artificial tunnels. We are not aware of any data with which to gauge the humaneness of kill-trapping mink.

6.4.2.a. How humane is live-trapping mink?

Mink can readily be caught in live-traps. A critical issue regarding acceptability is the method by which they are killed (Macdonald & Strachan, 1999). Drowning is illegal but may be widespread. This is widely regarded as inhumane (particularly in view of the mink's diving adaptations; e.g. Ladders *et al.*, 1999). It is general practice to despatch trapped mink with a high powered air rifle – while no data exist, the difficulty of shooting cleanly a fast moving mink within the confines of a small box trap is not to be underestimated, and may even destroy the trap.

The other major welfare consideration concerns the frequency with which the traps are checked, the placement of traps and provision of bedding and food, and a wire gauge which minimises broken teeth. In our experience, mink caught in carefully supervised live-traps are very rarely injured (D.W. Macdonald, R.Strachan, pers. obs.) and do not emit their noisy stress-calls until handled. However, welfare issues will arise if traps are not checked often enough.

6.4.2.b. How species-specific is trapping?

Done properly, limited data from Oxfordshire suggest that live-traps set for mink can be highly species-specific. In a total of 5299 trap nights (numbers of traps x number of nights set), in which there were 192 captures of mink, there were an additional 24 captures of non-target species (Yamaguchi, unpublished data). Nine of these were mammals (one hedgehog, three rats and five rabbits), and the remainder were birds, including four coots and five moorhens. Non-target captures thus represented 11% of all captures, but less than half a percent of all trap nights. All non-target species were alive and uninjured and were quickly released back into the wild at the site of capture.

6.5. How humane is snaring foxes?

Foxes are the only species covered by this report which are regularly snared (section 3.7). Issues surrounding the humaneness of snaring are similar to those around trapping: to what extent are non-target species caught; how long does the animal remain in the snare and under what circumstances; what injuries can occur?

UK legislation (Wildlife & Countryside Act 1981; section 11.1) prohibits the use of ‘self-locking’ snares and the setting of snares in circumstances where they are likely to kill or injure protected species. It also requires that snares are inspected at least once a day. Debate at the time of this Bill suggests that the intention was to prevent the deaths (by strangulation) of non-target species. However, evidence held by MAFF at the time demonstrated no difference in this respect between self-locking and other snares (H.G.Lloyd, pers.comm.). Current expertise holds that the use of a ‘stop’ to prevent closure of the snare beyond a minimum diameter is far more relevant to strangulation and its prevention, but a stop is not a statutory requirement. This statutory concern with the deaths of animals in snares by strangulation implies that the intended use of snares in Britain is to catch and hold animals (i.e. a ‘restraining’ device, *sensu* ISO standard rather than a ‘killing’ device).

In Australia, snare trapping is officially classified as inhumane (Saunders *et al.*, 1995). Judgement on humaneness is, of course, only part of judgement on acceptability – in the case of Australia the suite of variously rare native species that might be snared accidentally will affect judgement on the overall acceptability of this method (in England and Wales, the risk of accidentally snaring badgers and wildcats - protected under Schedule 6 of the Wildlife and Countryside Act - is the major consideration in this context). The RSPCA is opposed to the use of all snares and has urged successive governments to ban them.

Some organizations have produced codes of practice regarding the snaring of foxes (BASC, 1994; Game Conservancy, 1998). Adherence to such guidelines could reduce some of the problems but such codes are voluntary with no legal status and failure to follow them is not necessarily an offence.

6.5.1. Non-target captures in snares

Snares are selective for target animals through their manner of placement (see below), but in common with live-capture traps have the potential to involve non-target animals. Records of snare captures investigated by RSPCA inspectors have been cited in three publications (Harris, 1985; Baker & Harris, 1997; RSPCA submission to the Inquiry) with steadily increasing sample sizes (n = 222, 246 and 360, respectively). We assume these to represent successive analyses of the same accumulating data, and have taken RSPCA (2000) as the most recent. The number of snare operators involved in these records is unknown. *A priori*, one would expect snares to be investigated by RSPCA inspectors when captured animals are reported by members of the public, and thus expect these data to reflect a bias towards town/village fringes, domestic animals and larger wildlife species, and towards snares that have been misused or neglected. Comparison between studies of the proportion of non-targets amongst captures suggests that this is indeed the case (Table 6-7). The combined

GC/BASC gamekeeper snaring data have a significantly different species composition to the RSPCA figures[24]. The MAFF studies cited by Lloyd (1980) and by Chadwick *et al.* (1997) are not otherwise published, but appear to refer to snaring carried out to catch foxes for tagging; the number of snare operators involved is again unknown, but is probably small. Collection of data from the GCT and the joint BASC/GCT studies is described in section 3.2.1.d.

Table 6-7 Non-target captures in snares: a comparison of studies. GCT, 1992-3, unpublished; BASC/GCT, 1994-5, unpublished; MAFF, 1968, cited in Chadwick *et al.*, 1997; RSPCA submission to the Inquiry.

	Professional gamekeepers:		MAFF studies:		RSPCA inspectors:
	GCT 1992-3	BASC/GCT 1994-5.	Lloyd 1980	MAFF 1968	RSPCA 2000
% Of captures:					
Foxes	43	55	79	54	31
Cats	2	1	0	0	42
Dogs	1	3	1	0	8
Badgers	5	6	21	0	14
Deer (roe/muntjac)	6	9	0	0	0
Rabbit	9	5	0	0	0
Hare (brown/mountain)	29	16	0	0	0
Unclassified/other	5	6	0	46	5
Sample size (n captures)	739	516	136	287	360
Sample size (n operators)	61	64	unknown	unknown	unknown

Several points need to be made about these data. First, it is clear that snares placed to catch foxes are genuinely selective for foxes: of the chief non-target species, badgers are nationally about as common as foxes, roe deer about twice as common, while hares are about 3½ times as common (Harris *et al.*, 1995). This crude comparison overlooks the fact that none of these four species is equally abundant throughout Britain, but the limited data do not allow an analysis suitably stratified by region. This selectivity towards foxes is achieved not by the design of the snare, which is capable of catching all the species listed, but by the field-craft involved in its placement. However, despite this selectivity, and even accepting GCT and BASC/GCT figures as more illustrative of the general use of snares by gamekeepers, there clearly is a non-target involvement.

Second, although hares and rabbits are unintentional captures when snaring for foxes, both are game species that may legally be taken using snares. In population terms, both species may benefit numerically from the removal of foxes (Trout & Tittensor, 1989; Macdonald *et al.*, in press; Reynolds & Tapper, 1995b; section 5.3.3).

Third, most of the non-target animals were alive and uninjured when the snare was inspected and would probably not have suffered lasting ill-effect after release. This is known for badgers because early biological studies employed snares to catch badgers for radio-tagging. Behaviour after release was not abnormal (Kruuk, 1978). In the joint BASC/GCT trial, non-target captures and deaths were highly variable between individual operators, and may be largely avoidable through appropriate training (see below).

For comparison, snares have been used to catch foxes for radio-tagging during approximately 3 months in each of seven years of field research by The Game Conservancy Trust (J.C.Reynolds, unpublished data.). Snares were set only in open locations free of potential entanglements, were always fitted with stops and were inspected twice each night and once during daylight hours. No snare was ever set where a trail led under an obstacle such as a wire fence. 28 different foxes were caught, some of them two or three times. Two further foxes died as a result of capture. Because the intention was to catch and tag every fox present on just 2 or 3 territories, the intensity of snaring effort was far greater than used by any gamekeeper; this might be expected to increase non-target capture risk. Non-target captures were not recorded during these studies, but certain definite statements can nevertheless be made. Roe deer and fallow deer were common in the area, but were

[24] $\div^{2}_{[6]} = 117.9$, *P*<0.001

never caught. Rabbits were extremely common but likewise were never caught. Badgers were certainly more common than foxes and were occasionally caught (*c.* 10 captures), but in every case were released without visible damage. Brown hares were caught about as frequently as foxes; about half of these were released without detectable injury, the others were injured in the hind legs by the snare wire, and were dispatched. Brown hares were, and have remained, moderately common on this site (*c.* 15/km², about 15 times as common as foxes).

Snares have been the preferred method used by wildlife biologists to capture foxes for radio-tagging in every UK study in a rural area (Lloyd, 1980; Macdonald, 1987; Hewson, 1990; Reynolds & Tapper, 1995). While few data have been published on trapping statistics, anecdotal study of the behaviour of radio-tagged animals before and after capture has led to the consensus that any impact of capture is short-lived (in this context the impact of capture is a separate topic to the impact of tagging).

6.5.1.a.i. Developments since 1993

It is important to note that the situation described above has now changed. Following the GCT and joint BASC/GCT studies quoted here, recognition of the hazards of snares for non-target wildlife has caused BASC and GCT to address these issues and to organise training initiatives. A code of practice was produced (BASC, 1993), and training in the use of snares has been thoroughly revised, particularly to improve field-craft. Stress is laid on how to assess the circumstances in which it is appropriate or inappropriate to use snares. The risk to non-targets and how to avoid it through choice of snare, choice of location, and inspection frequency are emphasised. Inspection of all snares early in the day, and more than once per day if possible, are recommended. The use of stops on snares is urged as mandatory to minimise non-target captures, and to avoid the death or injury of captured animals. Some manufacturers have been persuaded to fix these in place in the factory. Several manufacturers now also use improved swivels, which are believed to reduce the potential for injury to captured animals. Operators are taught to avoid traditional snare sites such as holes in hedges, gaps under gates, etc. where there is a high risk of potentially fatal entanglement for captured animals. Instead they are taught how to use snares in open situations where fox capture rate and target specificity are greater, the risk of entanglement avoided, and where snare height can be set to suit the target species instead of being dictated by the location.

The Fox Snaring Code of Practice (BASC, 1993) is now regarded as standard by the police (it is a component part of the National Foundation Course for Police Wildlife Liaison Officers), RSPCA, ADAS, FRCA, Forest Enterprise, statutory conservation agencies (EN, CCW, SNH) (included in training courses), North West Water, Partnership For Action Against Wildlife Crime (PAW), The Game Conservancy Trust and others.

BASC fox snaring courses are run for game managers, and employees of statutory agencies and non-governmental organisations.

6.6. How humane are other methods?

Most other culling methods for foxes, deer, hares and mink, such as poisoning, have been banned on welfare grounds. In this respect gassing is an exception, because it is still technically legal in the UK, but cannot be used at present (section 11.2). Fertility control methods are also discussed here because their importance as a control method is likely to increase in time.

6.6.1. How acceptable is gassing foxes?

Formerly, it was argued that gassing foxes with Cymag reduced cruelty insofar as it increased the likelihood of killing orphaned cubs once their parent(s) had been shot at an earth. Cymag is no longer licensed for use in the attempted population control of foxes in the UK. While we know of no study of the humaneness of this technique, we recall, from the era of its use, concern amongst practitioners of foxes suffocating in air pockets within gassed dens. It may also be noteworthy that the only systematic study of gassing that we know of – that of gassing badgers in attempted control of bovine tuberculosis – led to the immediate banning of the technique on grounds of its cruelty.

In Australia den fumigation is officially classified as inhumane (Saunders *et al.*, 1995).

6.6.2. How humane is fertility control?

Long-term population reduction through fertility control has not yet been achieved for any wild carnivore, but this method might nevertheless become more important in future (Tuyttens & Macdonald, 1998a,b; section 3.6.1.a.iv). Abortifacients are currently more promising than contraceptives, but Tuyttens & Macdonald (1998) reported that the former were less acceptable to the ethicists they consulted. R. Short (pers. comm.) reports effective pilot trials with the chemosterilant cabergoline to eradicate urban foxes in Australia (a continent to which foxes are introduced aliens and a major threat to endemic biodiversity, a situation which shifts the utilitarian balance of what is acceptable to a radically different point than would apply in Britain).

Trials of immuno-contraception have also been undertaken for various ungulates, including deer (white-tailed deer: Kirkpartick *et al.*, 1997; Turner *et al.*, 1992; Garrot, 1995; fallow deer: Fraker, pers comm) in the United States with some degree of success (section 3.6.1.b.v). Numerous potential behavioural and physiological side-effects, as well as practical difficulty in humane dosing of a sufficiently high proportion of animals in wild populations, have yet to be fully investigated, before the acceptabilty of these methods can be evaluated fully.

6.7. Conclusions

- The acceptability of a control method will depend on the balance between a number of criteria, only some of which are readily measurable. An important criterion is humaneness.
  - ♦ Humaneness is a property of actions that do not involve ‘unnecessary suffering’.
  - ♦ Suffering can be assessed only indirectly, by combining objective measures of behaviour and physiology in the animal’s response to stress, with subjective consideration of the suffering associated with these in humans. This important approach is in its infancy, hence interpreting these measures is not straightforward, and the data are in any case sparse.
- Perceptions of the humaneness of different control methods differ between interest groups. Sometimes the basis for these perceptions is not clear and not readily commensurate with what fragmentary evidence exists.
  - ♦ Nonetheless, it is clear that a majority of the British electorate does not consider hunting foxes with hounds humane. Detailed surveys of other forms of hunting with dogs are not available.
- In hunting with dogs, welfare issues are primarily associated with the length of the chase and the mode of death.
  - ♦ Except for deer, which we do not consider as they are the subject of a separate contract, there have been no studies of the physiological effects of hunting foxes, hares, or mink. Studies on deer are not considered here, as they are the subject of a separate contract.
  - ♦ There are data on physiological responses of foxes to various stressors (including being chased with dogs in an enclosure), but their interpretation and relevance to hunting and other forms of control are not obvious.
- Shooting is widely regarded as humane if accurately done, but there are few data on how frequently it is accurately done.
  - ♦ A study of deer in south-west England suggests that wounding rates from stalking average about 2%.
  - ♦ Shooting adult foxes during the breeding season has particular welfare implications for orphaned cubs which contrast with its relative effectiveness.
- Welfare issues raised by trapping include the period within the trap (e.g. the stress of being restrained, dangers of starvation, dehydration, hypo- or hypothermia) and humane dispatch of captives.

    - Physiological data suggest that captured foxes exhibit a stress response. We cannot know for sure the extent to which that involves suffering, but the response is greater for leg-hold traps than for box traps, probably due to physical damage to the restrained limb, and to the animal struggling against the trap.
    - Humane dispatch of live-trapped mink is problematical.

- Welfare issues associated with snaring are similar to those of trapping, with the additional potential for strangulation in unstopped snares. Generally, less than half the captures in snares set for foxes are of non-target species

- In summary, science is not sufficiently advanced to provide simple measures of the humaneness of different control methods; any immediate judgement can be based on only fragmentary (although interesting) evidence and common sense. While science can inform many of the wider criteria in assessing acceptability (such as the effectiveness of the method in achieving its aims), ultimately acceptability is a judgement for society.

7. What would be the impact on populations of foxes, deer, hares, and mink of a ban on hunting with dogs, and how would this affect different interest groups?

7.1. Introduction

In this chapter, we assess the likely changes in populations of foxes, deer, hares, and mink were hunting with dogs to be banned, and the potential impacts of these and other changes on six main interest groups: farming, game management, forestry, fishing, conservation, and public amenity. We concentrate largely on impacts relating to damage, habitat management, and conservation, although clearly the impacts of banning hunting with dogs range far beyond our remit of population control and management.

In the context of population control, three quite different questions arise in view of a possible ban on hunting:

- Can the contribution currently made by hunting to the required cull be provided satisfactorily by other means?
- Will a ban on hunting make landowners less willing to tolerate the quarry at current density levels, and hence (through increased control by other methods) cause significant declines in those parts of their range where they are currently hunted?
- Would a reduction in overall numbers necessarily be detrimental to the conservation and welfare of the quarry?

The impact of the population management and control associated with hunting depends on various entangled issues, many of which we have already addressed. Primary amongst these will be the extent to which our target species cause serious damage (Chapter 2) and the effectiveness of hunting with dogs as a control method (Chapters 4 and 5). Paradoxically, another important issue is the impact of the hunt on its quarry species – not as a control agent, but as an agent for conservation and tolerance. Only the mink is not, so far as we are aware, actively encouraged to provide sport for the hunt.

7.1.1. Compensatory culling

In assessing the first of our questions, (“*Can the contribution currently made by hunting to the required cull be provided satisfactorily by other means?”),* we need to have an understanding of the interrelationships between different culling methods. We have already made the point that in most cases foxes, deer, hares, and mink are culled for more than one reason (Chapter 2). This typifies the context in which hunting with dogs is embedded. At one level, some of the species under consideration here are problematical for agricultural interests or for small-scale enterprises like poultry keeping. At another level, they may impact on secondary land-uses, such as game management. Finally, some interest groups (who may be the same as those who farm or shoot) find some diversion, fascination, entertainment, sport – call it what you will – from the business of culling these animals with dogs. Populations of the species under consideration here are therefore culled by and for diverse groupings of people.

Different culling methods are almost certainly '*compensatory*' to some extent: i.e. in the absence of hunting with dogs there would be more animals alive at risk of culling by other methods (see also section 5.1.1.a). Without any increase in effort, the culls obtained by these methods would rise to some extent, appearing to compensate partially for the lost method. A similar effect would of course occur if rifles and snares were banned, creating more opportunities for foxes to be killed by hunting with dogs. There is also likely to be compensation between hunting mortality and non-culling mortality such as road traffic casualties and disease.

To complicate matters further, there is another and potentially more potent aspect to compensation. We have seen that a degree of compensation will occur without any change in the risks created by individual causes of mortality. But in practice, these risks may be inter-dependent. For instance, if one method of culling was prohibited, some interest groups might increase the effort put into other culling methods, either in an attempt to achieve the same end effect, or in pursuit of some new and alternative goal. An oft-suggested scenario is that if hunting with hounds were banned, there might be an increase in shooting and snaring to achieve the same effect of controlling fox numbers. Another scenario is that foxhunting enthusiasts, frustrated in their preferred sport, might turn to other field-sports, such as game bird shooting, creating more shooting estates on which fox culling is carried out intensively. Yet another, is that farmers' tolerance of damage caused by deer, foxes, or hares might lessen if the interest generated by their pursuit with hounds were removed; in these circumstances, it is argued, culling pressure might intensify, leading to lower densities than at present. We explore some of these arguments below.

7.2. What would be the impact of a ban on hunting foxes with dogs?

7.2.1. What is the likely impact on fox populations of a ban on hunting with dogs?

In the event of a ban, the key to likely changes in fox population is the extent to which natural mortality and culling by other methods compensate for foxes not killed with dogs. The extent to which compensation occurs will vary greatly across the country, depending on the size of the fox population, the extent to which it is currently suppressed by culling, and the contribution made by hunting with dogs.

At one extreme is a scenario in which mortality, and therefore the population, remains unchanged. At the other extreme is a scenario in which no compensatory mortality occurs. Thus, if hunting with dogs were banned *and nothing else changed*, anything between 0 and 812 extra foxes (the number culled using dogs) might survive the first winter of the ban in Heydon & Reynolds' (2000a,b) mid-Wales region. Given that the spring breeding population of the region was about 560 adult foxes, there is the potential for breeding numbers and cub production to increase up to 2.5 times (i.e. (560+812)/560) in the first year of a ban alone. By the same logic, fox numbers in the east Midlands and west Norfolk might increase by 10% increase in the first year of a ban.

Of course, these extreme scenarios are unlikely; reality is generally somewhere in between, but the difficulty lies in knowing where. Most causes of mortality intensify with increasing density (are density-dependent – see section 1.3.2.a). Thus, 'natural' causes of death like epidemic disease are facilitated by higher population density. Dispersal is also more common at higher densities, and leads to increased mortality. However, the very high productivity observed in mid-Wales and west Norfolk suggests that there is plenty of space for more foxes in those regions, and density-dependent 'natural' mortality factors may not increase much until numbers built up considerably. Thus unless checked by culling, numbers could build up rather fast.

A third scenario involves a decline in overall fox numbers because of the removal of forms of protection afforded them by the hunt. This protection is of two kinds. First, there is evidence that hunting with hounds in the east Midlands leads to greater tolerance by farmers, including those with shooting interests (Reynolds & Heydon, 2000a; GCT submission to the Inquiry). Second, hunts actively improve habitat for foxes through woodland maintenance (see below), and even by providing artificial earths, consisting of a pipe ending in a chamber (MFHA submission to the Inquiry). We have no data on the extent to which these actions affect fox populations.

7.2.1.a. Could the fox cull currently taken by hunting with dogs be accounted for by other methods?

We already know (section 2.2.1.f.i) that a further 5% of farmers would begin culling if fox numbers increased. We do not know to what extent farmers who already cull would intensify their efforts, nor to what extent those increased efforts could make up for the lost cull formerly taken with dogs. It is probable that there would be an increase in shooting and snaring effort. In mid-Wales, 95% of farms had some form of fox culling, and 69% relied exclusively on communally organised methods that involve dogs (Heydon & Reynolds, 2000a). Given that this is intensive sheep farming country where snaring is difficult and unpopular, and assuming that they do not already have one for other reasons, 573 farmers might therefore wish to acquire firearms certificates. For the east Midlands and west Norfolk, the equivalent figures are 347 and 186 farmers.

In the east Midlands and west Norfolk, where 35% and 50% of farms (respectively) have a professional gamekeeper, it seems likely that a 10% regional increase in fox numbers could be absorbed through an increased cull using rifle and snares. However, this assumes that the extra foxes would make themselves available to gamekeepers on shooting estates by dispersal. This is more likely to happen in the east Midlands where fox density is already high. In west Norfolk, a more plausible scenario is that fox density outside of shooting estates would increase somewhat. This might not affect game management interests unduly, but it would have consequences for livestock and poultry farming, and of course for wildlife outside of shooting estates.

What about the rest of England and Wales? Heydon & Reynold's three study regions were chosen for their variety of conditions, not for their representativeness. We cannot say how much of England and Wales is like any of them. For instance, does Wales typify upland regions, or are other upland regions like the Pennines or the Yorkshire Moors radically different? What is clear is that appreciating the regional variation in land-use, fox density and fox culling practices is crucial to a proper understanding of fox control.

7.2.2. What is the likely impact on different interest groups of a ban on hunting foxes with dogs?

The principal interest groups who might be concerned by potential increase in the fox population because of a ban on hunting with dogs are livestock farmers (especially upland sheep farmers) and game managers. Those living in areas where a high proportion of the cull is currently taken by methods involving dogs are most at risk of a potential population increase, particularly those in areas where other methods are not as effective (e.g. mid-Wales). A third interest, widely considered to benefit from the landscape created by hunting enthusiasts (e.g. MFHA submission to the Inquiry), is conservation.

7.2.2.a. What is the likely impact on farmers and game managers of a ban on hunting foxes with dogs?

The impact on farmers and game managers of a ban on hunting is likely to vary considerably from region to region. In the GCT's 'Three-region' study (Heydon & Reynolds 2000a,b; Heydon *et al.*, 2000), fox culling in total was shown to be the likely cause of population suppression in mid-Wales and west Norfolk, while the fox population of the east Midlands region was thought to be little affected by the culling levels pertaining there.

In mid-Wales, 73% of the cull was taken using methods involving hounds or terriers. The stated aim of this cull was primarily to prevent lamb losses. Lamb losses were low (reported to be <1% of lambs born; section 2.2.1.c), but we cannot say to what extent they would increase in the absence of culling. However, it seems probable that fox density would increase rapidly if hunting with dogs were suspended in mid-Wales.

Compared with the other two regions, the intensity of fox culling by the hunt was restrained in the east Midlands (for example, although 40% of foxes were run to ground during hunting, only 18% of the hunt cull was taken by digging out; see also section 5.2.3.b.ii). Total culling pressure was insufficient to achieve population control, and fox density was closer to the carrying capacity of the environment. Fewer farmers had livestock or game at risk of predation, and the most vulnerable group – those with farms smaller than 200ha – were more tolerant of losses than in the other regions. Where culling was practised independent of the hunt - as on shooting estates – its intensity (foxes killed per square kilometre) was lower than in either mid-Wales or west Norfolk, despite fox density being over three times greater than in mid-Wales and over four times greater than in Norfolk.

In west Norfolk, hunting with dogs accounted for one tenth of all foxes culled. Here the primary aim of fox culling was to reduce predation on game, and the overwhelmingly important methods were rifle (with spotlamp) and snares. It is likely that these methods could make up for the 10% culled using dogs, were the use of dogs banned, provided the foxes were available to be culled on shooting estates. Shooting estates make up about 50% of the landscape in this region, so it is possible that fox densities might increase outside of shooting estates, on other land currently hunted with hounds.

As noted earlier (section 3.2.2.c.i), terriers are used by roughly 50% of gamekeepers. Because of their unique role in culling at the earth, we must conclude that fox control in the context of game management would be impaired by a ban on hunting with dogs, and that this would apply especially to the management of wild gamebirds, and especially in upland areas. Thus grouse moor management would be especially affected. A complete ban on the use of terriers (including their use for locating and bolting a fox) would mean that focussed spring and summer fox control suitable to the protection of wild game-bird populations could be practised only using shooting and snares, resulting in a higher incidence of orphaned cubs and non-target captures.

The accepted passage of hunts across game shooting estates suggests that if there is any disturbance to gamebird coverts, this is widely tolerated. A code of practice exists to minimise conflicts of interest between hunts and shoots.

7.2.2.b. What is the likely impact on conservation of a ban on hunting foxes with dogs?

It is sometimes argued (e.g. Cobham Resource Consultants, 1997) that farmers allow their pastimes, most notably their interests in field sports, to influence their land management. For example, hedgerows can contribute to hunting by providing challenging jumps, or cover for the foxes. We were able to use the WildCRU's 1981 and 1998 questionnaire data (described in section 2.2.1.a.ii) to investigate the proposition that hunting and non-hunting farmers differed in their management of wildlife habitats.

The factors that influence the agricultural landscapes of Britain have changed markedly between these surveys. At the time of the first survey, the emphasis was on increased production, and non-productive habitats such as hedgerows and spinneys were extensively removed to increase field size and allow the use of large machinery. By the time of the 1998 survey, however, overproduction was a problem, and there had been many and varied changes in countryside policy. A patchwork of habitat protection and creation measures had evolved. In this section we seek to establish if there were any links between farmers' participation in hunting and their habitat management, and how this changed between the surveys. A full account of these results are given in Macdonald & Johnson (2000). Of course, we cannot predict from these observations what the effect of a ban on hunting would be. It is possible that any differences might persist in the presence of a ban. We do not know if farmers with a tradition of sensitive habitat management would change their policy as a result of the removal of hunting.

In our questionnaire surveys, farmers whose primary interest was in hunting reported removing less hedgerow in the past decade than those who shot or had no field sport interest, but this was statistically significant only in 1981. In an overall analysis, the variation between categories of field sport interest (hunting, shooting, both, neither) was significant for absolute length[25]: farmers who hunted, or said they were equally interested in hunting and shooting, removed least hedgerow in both surveys (Table 7-1). However, there was no such difference when only data from the 1998 survey were used; during the 1980s much less hedgerow in total was removed than during the 1970s. In our 1995 survey of Wiltshire farmers (section 2.2.1.a.ii), we could find no evidence that hunting farmers had removed any less hedgerow than non-hunting farmers[26], or differed in the order that they ranked hedgerows, trees, spinneys and shelter belts in terms of their value to wildlife. Farmers who said they had hunted had used the Environmentally Sensitive Area Scheme more frequently than non-hunting farmers [27]).

[25] $F_{3,897} = 5.9$, $P = 0.0089$

[26] Mann Witney test, Chisq approx = 0.26, $P = 0.60$

[27] 4/28 v 0/73Fishers Exact test, P < 0.001

In 1981, a smaller proportion of hunting than non-hunting farmers planned to remove more hedgerow: only 5.3% of hunting farmers, compared with 15.5% of farmers interested primarily in shooting, 8.2% of farmers interested in both, and 9.2% of farmers interested in neither[28]. In the1998 survey, where very few farmers in any category stated plans to remove hedgerow in the future, there was no significant difference with respect to field-sport involvement. There was a marked increase in the second survey compared with the first in the percentage of farmers who cited hunting as a reason for retaining hedgerow (Table 7-2)[29].

Table 7-1 Hedgerow removal and farmers field sport interest in 1981 and 1998. Means per farm for each category of farmer. Losses as reported by the farmer in the decade preceding the survey.

Field sport Involvement	Absolute loss m/farm (SE)		Loss m/ha/farm (SE)	
	1981	1998	1981	1998
Both	318.4(59.2)	16.4(9.7)	2.44(0.56)	0.10(0.05)
Hunting	277.1(59.2)	35.5(18.2)	2.52(0.48)	0.97(0.60)
Neither	525.1(90.6)	63.2(16.4)	5.77(1.00)	0.57(0.17)
Shooting	709.2(91.4)	77.3(15.8)	4.99(0.89)	0.58(0.16)

Table 7-2 Farmers citing participation in hunting with hounds as a reason for retaining hedgerows in 1981 and 1998.

County :	% Farmers (n = 1241)	
	1981	1998
Dorset	25.6	51.9
Leic.	24.7	55.6
Oxon.	22.9	53.6
Shropshire	6.8	32.7
Suffolk	13.2	18.2
Sussex	24.7	41.5
Warwicks.	10.3	66.7
West country	18.8	79.2
Overall	17.9	49.0 ***

There was also evidence that field sport involvement was linked to the management of other non-productive habitats. Farmers in the 1998 survey who said they were interested in neither hunting nor shooting encouraged shelter belts significantly less frequently than those with some level of participation in these pastimes[30]. Non-hunting farmers were less likely to cite encouragement as their principal strategy for ponds and shelter belts; 64% of farmers who said they had participated in hunting said their principal strategy with respect to shelter belts was encouragement, compared with 53.5% of non-hunting farmers [31]. Similarly, 63% of hunting farmers said they encouraged parkland trees compared with 47.5% of non-hunting farmers[32].

There was a tendency in both surveys for more of the farmers interested in either hunting or shooting to have sought advice on wildlife (The Farming and Wildlife Advisory Group were the most frequently cited source) than those interested in neither[33]. In 1981 only 6% of those with no field-sport interest in 1981 sought advice,

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[28] $\chi^2_{[3]} = 9.2$, $p = 0.02$

[29] $\chi^2_{[1]} = 127.0$, $P < 0.0001$.

[30] 3-way interaction log-linear analysis $\chi^2_{[1]} = 21.7$, $P < 0.001$

[31] $\chi^2_{[1]} = 3.94$, $P = 0.06$

[32] $\chi^2_{[3]} = 6.4$, $p = 0.01$

[33] $\chi^2_{[3]} = 9.8$, $p = 0.021$

while 22 % of those interested primarily in hunting, and 14 % of those interested primarily in shooting, did so. Inexplicably, the rate for those farmers who said they were interested in both hunting and shooting was also only 6%. In the later survey, fewer farmers interested in neither fieldsport answered ‘yes’ to this question.

The view that farmers who are interested in hunting are more concerned for the environment finds some support in our data; there was evidence that some unproductive habitats may have been better treated by hunting farmers. However, the evidence that hunting farmers particularly valued their hedgerows was weak - while hunting farmers reported removing significantly less hedgerow in the 1981 survey, in recent years reported removal rates were generally much lower, and unrelated to field sport involvement. Since 1997, hedgerows have, in any case, enjoyed limited legal protection (DOE, 1997).

There may be environmental effects at the level of the hunt Country that are not detectable at the level of individual farms. Hunts may be responsible for woodland management, and may buy woodland in order to ensure access (MFHA submission to the Inquiry). We observed that 45.5% of Masters responding to our survey said that the hunt maintained at least one spinney or thicket; 5% of Masters said they considered it part of a their job to influence land management practice within the hunt Country. This latter figure varied significantly between regions being much lower in the Wales and West (9%) than elsewhere (31-54%). This may reflect a difference in attitude, if foxes in the uplands are considered a pest, while in the lowland they are seen as more of a quarry.

In addition to a possible impact of a ban on hunting on habitat management, any increase (or decrease) in fox populations might be cause for concern among conservationists, particularly with regard to ground-nesting birds. As with farmers and game managers, the impact of any change will vary regionally. However, in terms of its use as a control method, foxhunting is generally not considered appropriate in protected areas. For example, the RSPB, while neutral on the hunting debate (Julianne Evans, pers. comm.), does not allow foxhunting on its land (Conservation Management, RSPB unpublished). While the direct catching of foxs using dogs is not allowed on land under RSPB control, dogs can be used to flush foxes from cover to be shot.

7.3. What would be the impact of a ban on hunting deer with dogs?

7.3.1. What is the likely impact on deer populations of a ban on hunting with dogs?

7.3.1.a. Could the deer cull currently taken by hunting with dogs be achieved by other means?

Currently, hunting with dogs accounts for only *c.* 130-150 red deer and 40 roe annually, representing less than 0.1% of the total estimated annual cull of wild deer (all six species) in Britain. However, as all red deer hunting, and most roe deer hunting, is restricted to Somerset and Devon, its contribution to deer culls should be considered primarily at that regional level. Here, those *c.* 130-150 red deer deliberately hunted and killed by the three West Country staghunts constitute at most one sixth (and possible less than one ninth) of the minimum cull requirement to prevent further population increases (section 5.4.5.a).

A ban on hunting with dogs would thus necessitate only a relatively small increase of the total cull already taken by shooting. Full time wildlife rangers employed by major forestry companies or deer stalking estates, will quite commonly cull between 70-100 or more deer each per ranger per annum (e.g. see Putman & Langbein, 1999), suggesting the cull currently taken by the hunts could readily be undertaken by the equivalent of just one or two rangers. In reality, it is much more likely that the additional cull would be absorbed through a small increase by each of the 50-100 or more licensed deer stalkers known to cull deer within West Somerset /north-west Devon (Langbein, 1997; Police wildlife liaison officers, pers. comm.).

Similarly, while we have no firm data on actual numbers of roe deer hunted and killed, this is believed to be unlikely to exceed 30-40 per annum (Master Exe Valley Buckhounds, pers comm), and is thus likely to have only minimal effects on the total roe population in these counties.

7.3.1.b. What will be the likely impact on deer populations of a ban on hunting with dogs?

It is widely believed that many landowners on Exmoor and the Quantock Hills, whether their primary interest is farming, forestry or nature conservation, tolerate rather higher deer densities on their land than they would choose to keep if hunting were no longer permitted (MDHA and Exmoor NP submissions to the Inquiry). However, whether and to what extent individual landowners are likely to attempt to reduce deer densities on their land in the event of a hunting ban is very difficult to predict with any accuracy.

Some limited data come from a recent questionnaire survey into views on deer management of local landholders in the Quantock AONB (Langbein, 1998; section 2.3.1.a.ii). When asked: "*If hunting with hounds were to be abolished, how do you expect this would affect your own deer management*", 35% of 65 landowners said they would probably tolerate fewer deer and increase their cull. The remaining 61% did not envisage increasing the overall cull on their land, although 38% thought they would need to increase the shooting cull to compensate for the reduced hunting cull. Collectively these 61% of respondents held 9,256 hectares of land, while those (23) landholders likely to tolerate rather fewer deer held a further 4,243 hectares (note: although the Quantock Hill AONB only covers 95km^2, the land area covered by responses received is larger as some estates straddle or fell just outside the designated AONB area). These results must be treated with some caution, not least because any change in deer culling policy on one holding may also influence deer behaviour, and distribution on neighbouring holdings.

Some limited insights may also be gained from recent census information on the National Trust's Holnicote estate, where deer hunting was banned in 1997. Holnicote is the largest contiguous estate owned by the National Trust on Exmoor, and extends over some 5000ha encompassing some of the prime red deer areas in the West Country. Although it makes up less than 8% of the land area of the Exmoor NP, around 17% of all red deer recorded during the annual visual deer census organised by the Exmoor & District Deer Management Society are usually seen within this estate alone. Some culling by rifle has regularly been undertaken on this estate. In the four seasons prior to the ban (1993-1997) numbers shot were 19, 33, 13, and 18. Since the ban was introduced in 1997, cull levels have been increased moderately (83, 35, 53) with the aim of maintaining a population of *c.* 380-480 red deer on the estate (National Trust, 1998). Results of the spring census counts in the four years prior to the hunting ban (387, 453, 435, 382) have remained virtually unchanged in comparison to spring counts in the three years since its introduction (486, 374, 481).

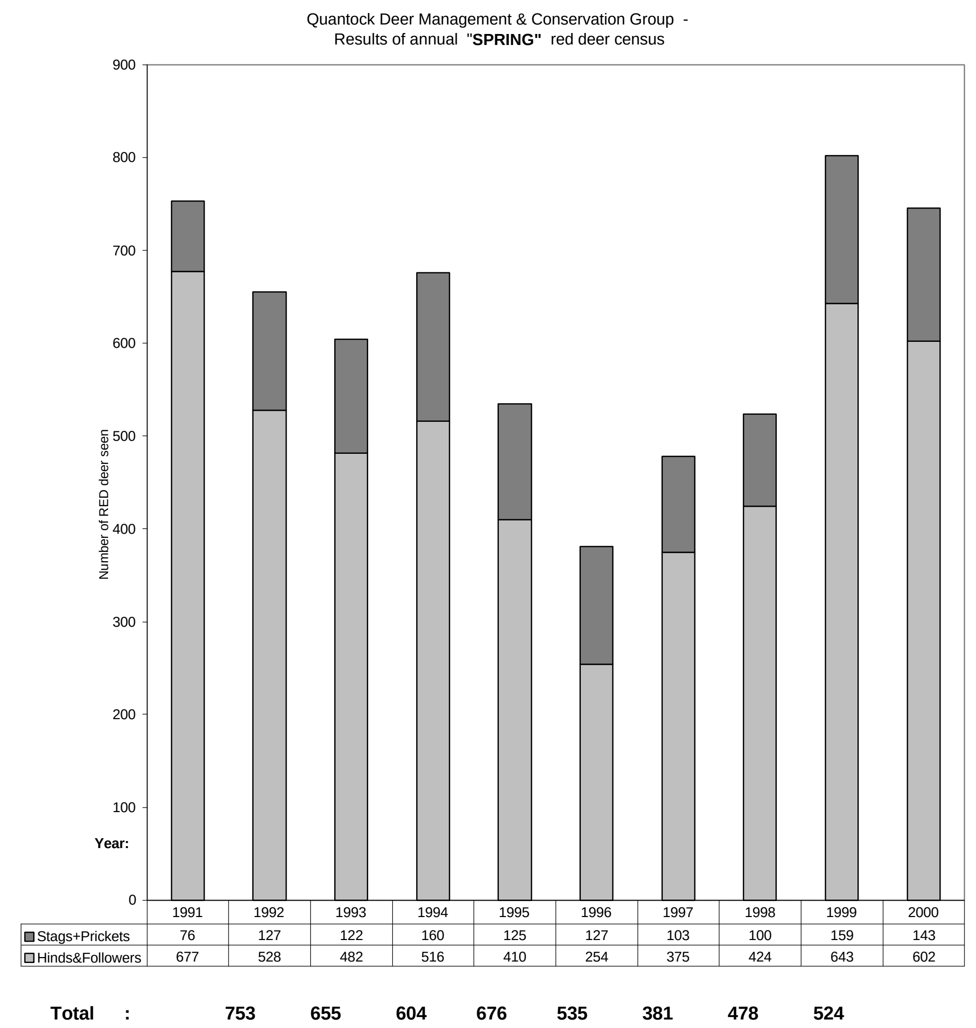
Extensive bans on hunting have also been implemented National Trust (in 1997) and the Forestry Commission (in 1998) in the Quantock Hills. Here, the ability of the QSH to hunt much of the central hill has been severely restricted, as they now have to avoid trespass of hounds into several moderate sized holdings distributed across the Hills. This in turn has led to a reduction in the number of deer hunts held annually within the Quantock Hills (with a significant proportion of QSH meets now transferred to Exmoor by inivitation of the DSSH). Although the bans elicited great anger and opposition from many Quantock landholders, leading at least some to significantly increase their deer culls in direct reaction, overall deer numbers in the Quantock Hills would seem to have remained fairly stable or have increased somewhat since imposition of those bans (see **Error! Reference source not found.**])

In the event of a ban on hunting, it is likely that at least some foresters would aim to manage deer numbers at lower than the current densities. However, the significance of deer damage tends to vary widely according to the growth stage of plantations, being most severe in the establishment and pre-thicket phase, but rather less important in mature, pre-felling stands (Mitchell *et al.,* 1977; Staines & Welch, 1989). Consequently, the extent to which foresters would be inclined to introduce tighter control over deer numbers would be likely to vary widely between differing forestry holdings across the region.

It is quite clear from the above, and also from discussions held with pro-hunting landholders and others neutral or opposed to hunting in other parts of the West Country (Savage, 1993; Langbein & Putman, 1992; Bateson, 1997) that reactions to a ban would vary widely between individual landholders. Some landholders passionately opposed to a ban may initially decide to attempt extermination of deer from their land, to demonstrate their anger, as happened to some extent following introduction of a deer hunting ban by the National Trust and Forestry Commission.

It is inconceivable that any such extreme action would occur on land owned by major landholding organisations (such as e.g. the Exmoor NP authority, National Trust, Wildlife Trusts, Crown Estates, Local Authorities, or Forestry Commission), who between them account for >25% of land within both the Quantock Hills AONB,

Figure 7-1 Quantock Deer Management Group – Records of annual spring red deer census



and within the Exmoor National Park. However, some landholders suffering high levels of deer damage, probably in particular those with commercial forestry interests (Packer *et al.*, 1998) or high value agricultural crops, have indicated that following a ban they would retain deer populations at somewhat lower density levels than at present. Others suffering comparatively little deer damage, and who view provision of deer viewing opportunities on their land as a positive asset in fulfilling their objectives for public amenity and conservation (e.g. Exmoor NP, National Trust), are likely to adjust their culling in such a way as to attempt to continue to maintain stable populations on their estates close to levels held prior to any ban (e.g. National Trust, 1998).

Considering the various compensatory factors, it seems probable, though by no means certain, that overall numbers of red deer maintained within the currently hunted area would decline to a certain extent. However, in the event of any drastic reductions being noted to occur in some areas, these would be increasingly likely to be compensated for by reduced culls and increased efforts at conservation of red deer by other landholders in the region, making a degree of re-distribution of the present herds across the region one of the most probable outcomes.

7.3.1.b.i. Are current densities of red deer in the Staghunting countries appropriate?

Red deer populations subject to staghunting in Southwest England (and many other deer populations throughout England, not subject to hunting) have increased significantly over the last thirty years (Langbein & Putman, 1992; Trout *et al.*, 1996; Staines *et al.*, 1998). The most widely accepted population estimates for red deer in Exmoor NP during the 1970s was around 500-800 deer (Lloyd, 1970, 1975), rising to 1500-1900 during the 1980s (Allen, 1990; Floyd, 1990, 1998). By the late 1990s, they reached at least 2300 (E&DDMS, in Langbein, 1997) and are therefore higher now than throughout most of the past century.

Whether current densities of red deer in Exmoor NP and the Quantocks are too high, too low or about right, is a question which cannot be answered on purely ecological grounds. The 'correct' figure depends entirely on management objectives - or where there are numerous, conflicting, pressures, on reaching a suitable compromise between the objectives of different interest groups. High densities enhance the chance for visitors to the region to see wild deer, and are thought to add greatly to the tourist appeal of the area (Exmoor Life, 1998; Exmoor NP submission to the Inquiry). A reasonably high deer density is also desirable for Staghunting and to maximise venison revenues. However, densities appropriate to these interests may cause unacceptably high levels of overgrazing of sensitive natural habitats (section 7.3.2.b), or damage to agriculture or forestry crops (section 2.3.3.c); such damage may become quite severe at population levels well below the biological carrying capacity of the environment (e.g. Ratcliffe, 1998).

The recent increases in deer density in the Quantocks are of some concern not merely from view of damage to forestry, agriculture, and conservation (Langbein, 1997, 1998), but also in view of rising numbers of road traffic accidents involving red deer in this area (Avon & Somerset Constabulary, pers. comm). A population of around 550-700 red deer at census has recently been agreed by most landholding and other members of the Quantock Deer Management Group as a suitable target figure towards which deer management on the Quantocks should be directed. Recent visual census information (QDM & CG, 1999, 2000) suggests minimum red deer numbers in spring of 750-800, and rather higher than this target. This helps to illustrate that some reduction over present numbers may well occur here in future as result of deliberate management actions, irrespective of whether or not a ban on hunting is introduced.

7.3.2. What is the likely impact on different interest groups of a ban on hunting deer with dogs?

7.3.2.a. What is the likely impact on farmers and foresters of a ban on hunting deer with dogs?

Farmers and foresters in the Staghunting countries, and within the West Country in general, are highly unlikely to suffer increased damage levels as a result of banning stag hunting. Indeed, the opposite may be true, as overall red deer abundance could decline.

Hunting with dogs is rarely used as a means of preventing damage to forestry, due to the difficulty in controlling and overseeing the hounds in thick cover, especially within thicket stage conifer plantations. However, some effects could arise in the long-term as a result of the current partial bans on deer hunting in the West Country, such as came into effect on National Trust land in 1997, and on all Forestry Commission land during 1998. The main area of commercial forestry affected by this ban is Great Wood, a large block of woodland covering *c.*650 ha at the centre of the Quantock Hills. This area is surrounded by land that is frequently hunted by the Staghounds, resulting in an increased tendency for deer to settle within the wooded area when hunting is in progress (J. Langbein, pers. obs.). This tendency for deer to use the woodland as a sanctuary from hounds may be expected to increase over time, further increasing the risk of damage to forestry unless greater deer control is exercised within the wood in future.

Although the actual effectiveness of hunting with hounds to limit crop damage through dispersing herds of deer is rather debatable (see section 5.4.3.b), inability to call on the Staghounds any longer to assist in this way would be regarded as a negative aspect of a ban by many farmers. There are potentially some negative aspects of hunting though damage incurred to pastures and crops, and disturbance of livestock, but these have never been quantified. Such damage is clearly of greatest relevance to the minority of farms in the hunt countries that do not permit hunting on their land, but do at times suffer from accidental trespass by hounds onto their land.

7.3.2.b. What is the likely impact on conservation and amenity of a ban on hunting with dogs?

Red deer are the largest terrestrial animals living wild in this country, and as such are highly valued by many people who delight simply in seeing them in the countryside. By far the largest and most readily viewed herds in England and Wales are based on Exmoor and the Quantock Hills. It is widely believed that red deer have been present continuously on Exmoor since soon after the last ice age (Whitehead, 1964; Allen, 1990; Lloyd, 1975). In their own submission to this Inquiry, Exmoor National Park Authority state: "*Any description of Exmoor must mention* *the importance of red deer and the National Park Authority's logo carries the head and antlers of the red deer stag. When Exmoor was designated a National Park in 1954, its red deer herds were undoubtedly a significant factor in the decision to designate.*". The conservation of a healthy deer population remains amongst the stated objectives of long-term management plans for both Exmoor and the Quantock Hills

Habitats of conservation value could potentially suffer increased damage because of a ban, if deer populations are redistributed within the Staghunting countries in response to changing culling pressures. If deer culling increased on cultivated land and forestry areas, this could lead to a shift in deer distribution, with a relatively greater build up of deer numbers in the semi-natural woodland and moorland areas where they are presently thought to cause little damage. Batcheler (1968) demonstrated that selective shooting of red deer in high quality forest habitats could cause them to shift their distribution into lower quality areas. Greater accumulation of herds in semi-natural habitats could pose some risk of increased damage to certain habitat types (e.g. coppice regeneration). By contrast, where some level of grazing is thought beneficial for conservation (e.g. heather moorland, pasture woodland) or reasonably high deer numbers are considered desirable for public amenity reasons, the potential for such shifts in distribution could be enhanced further through active manipulation of the habitat (to provide improved areas of grazing or shelter) or artificial feeding (see e.g. Stenin, 1970; Graham, 1995; Putman, 1998; Langbein, 1997).

The extent to which public amenity benefits associated with the presence of deer might be compromised by a ban on hunting depends not merely on whether such a ban would result in a reduction in overall deer numbers, but probably more so on changes in the distribution of the remaining population. If significantly increased shooting pressure does indeed occur on farmland and forestry, this would be expected to lead to some reduction in overall numbers in the short-term. However, provided the deer are not shot equally heavily in those areas regarded as prime sites to provide deer viewing opportunities for the public (i.e. particularly the open moorland and semi-natural woodland areas) increased culling on surrounding farmland may in fact serve to gradually concentrate deer herds more so in those public access areas. Thus even if a significant reduction in overall numbers should arise, this would not necessarily be predicted to have a major impact on the current prime sites for viewing deer (e.g. such a Dunkery Beacon/ Horner Woods on Exmoor, and the hilltop commons on the Quantocks), where number might remain affected to a lesser degree.

The number of followers during stag and hind hunting on Exmoor and the Quantocks commonly exceeds 100 to 300 vehicles (including high proportions of 4-wheel drive cars and motorbikes). Some areas of moorland on Exmoor are thought to have been damaged by vehicle followers leaving the road for closer views of the hunt (e.g. Exmoor NP submission to Inquiry). Such damage by vehicles is, however, considered less of a problem than the impact of more frequent use of commons and bridleways on Exmoor by trekking groups (Exmoor NP submission to the Inquiry). The large 'field' of horse followers associated with deer hunting causes significant damage to public footpaths and bridleways from time to time, and the National Park Authority has at times had to undertake repair works as a result.

7.4. What would be the impact of a ban on hunting hares with dogs?

Formalised hunting with dogs (with a pack of hounds or as part of National Coursing Club competitions) currently contributes only a tiny fraction of the cull, and in any case is rarely practiced for population control. A ban on these means of hunting with dogs would therefore be highly unlikely to impact on farmers or foresters in terms of increased populations and damage.

In addition, an unknown amount of informal or illegal coursing takes place in some areas, such as the South Downs, for which we have no data. Coursing carried out illegally at present will probably not be prevented by banning hunting with dogs, so would not directly cause a change in current levels of mortality. However, if landholders no longer have an incentive to conserve and protect hares for sport, they may be more inclined to remove hares from their land to prevent unwanted poachers.

A ban on hunting with dogs might have a negative impact on hare conservation, though this is a contentious issue (Hutchings & Harris, 1996). The Association of Masters of Harriers and Beagles (AMHB) requires its Masters to "*maintain a basic knowledge of the hare population throughout their registered country and to encourage that population at an appropriate level*". AMHB hunts keep records of numbers of hares seen on each day's hunting. These data are analysed and represent the only wide-scale source of data on hare numbers on a year-to-year basis from which to infer population trends. The AMHB argues that hunts therefore provide "*a unique and comprehensive source throughout England and Wales of practical knowledge and expertise about the hare in its natural habitat*" (AMHA & MBHA submission to the Inquiry).

The GCT's submission to the Inquiry suggests that the organised shooting of hares is generally absent from coursing estates, and that an interest in coursing encouraged farmers to tolerate higher levels of agricultural damage than would otherwise be the case. Lord Leverhulme's estate at Altcar, where coursing's Waterloo Cup has been run since 1836, provides an example of how hare numbers can be increased for coursing. Following the death of the last Lord Sefton in 1973, the previously high hare numbers on the Altcar estate fell so low that from 1978 to 1980 the Waterloo Cup was not run. In the meantime, the Waterloo Cup Committee, with the co-operation of the new owner, Lord Leverhulme, his tenants, and keepers, introduced fresh stock from other areas such as East Anglia. This action was accompanied by intensive predator control and poaching prevention. Farming changed from almost exclusively spring cereals in 1980 to a more diverse mixture of winter and spring cereals interspersed with vegetables, root crops, together with game crops. The stubbles left for the Altcar Club coursing in the autumn and the permanent grass meadows of the Waterloo Cup running grounds at the Withins and Lydiate contribute fodder and shelter. The density of hares on the estate increased by 39% between 1980 and 1988 (S.C.Tapper, unpubl.).

Hares are now sufficiently numerous at Altcar that there are plans to translocate some to other sites. Hutchings & Harris (1996) state that during the national hare survey a number of farmers and landowners in eastern England reported large numbers being caught and moved to coursing areas to ensure a good meet. However, there are no data that we know of to indicate how often this occurs, nor what the impacts are on the donor and recipient populations, or on the behaviour or survival of individual hares.

7.5. What would be the impact of a ban on hunting mink with dogs?

Our data for the numbers of mink killed by each method, and for the impact of mink on each interest group, are insufficient for us to meaningfully assess the impact on mink populations of a ban on hunting. However, we would note, first, that mink hunting probably takes less than 5% of the population, and second, as there are only 20 mink hunts in Britain any impact they have will be localised.

Hunting mink with dogs is widely regarded as having a detrimental effect on wildlife conservation because of the disturbance involved when the hounds and the whips investigate potential lying-up spots or dig out mink that have gone to ground. There is particular concern for the disturbance of otters and trampling of bird nests.

There are no quantitative data regarding this matter. However, Dorset Wildlife Trust sought the opinions of 15 leading academics and researchers who had worked extensively with otters. Of these 15, 12 replied, and 8 believed that mink hunting with hounds posed or probably/possibly posed a significant threat to breeding otters. Seven respondents believed that mink hunting reduced (6) or potentially/possibly (1) reduced the rate or potential for a recovery of the otter. Only two respondents believed that mink hunting did not pose a more significant risk of disturbance than other recreational activities. However, seven respondents had not actually come across examples of mink hunting causing otter disturbance or mortality, and otters have expanded their range including in areas where mink hunting with hounds takes place. In the light of the questionnaire results, it is Dorset Wildlife Trust policy to resist mink hunting on its reserves.

7.6. Conclusions

- In the context of population control, three quite different questions arise in view of a possible ban on hunting:
    - ♦ Can the contribution currently made by hunting to the required cull be provided satisfactorily by other means?

- ♦ Will a ban on hunting make landowners less willing to tolerate the quarry at current density levels, and hence (through increased control by other methods) cause significant declines in those parts of their range where they are currently hunted?
- ♦ Would a reduction in overall numbers necessarily be detrimental to the conservation and welfare of the quarry?

- The impact of a ban on hunting foxes with hounds will be highly regionally variable, depending on fox densities and the utility of other methods. In view of this, the impact of a ban on hunting with dogs, in terms of damage caused by foxes, would be regionally complex.
  - ♦ A ban is most likely to have important consequences for game managers and livestock farmers, especially in upland areas with difficult or remote terrain. It is least likely to have important consequences for fox population control in lowland areas.
  - ♦ In mid-Wales, there was strong evidence that hunting with dogs accounted for 73% of a regional cull that effectively suppressed fox numbers. Here, and possibly elsewhere, the result of a ban on hunting with dogs (including both hounds and terriers) would be to allow a rapid increase in fox numbers, unless the same cull could be achieved using other methods.
  - ♦ A ban may have consequences for habitat conservation on farmland; in the 1970s non-hunting farmers removed more hedgerow than hunting farmers, but this was no longer true during the 1980s. However, there was evidence that hunting farmers managed other non-productive habitats with more regard to conservation than non-hunting farmers.
- Staghunting currently contributes only a very small proportion of the red deer cull, even within the Staghunting countries, and this could readily be absorbed by stalking. There may be a decline in the population because of lower tolerance to red deer in the absence of hunting. However, current levels of deer are very high, and are likely to be reduced regardless of whether there is a ban or not.
  - ♦ A possible decline in red deer in the Staghunting countries means that farmers and foresters are unlikely to suffer from increased damage because of a ban. Visibility, which is an important amenity provided by deer, is not simply related to abundance, and will not necessarily suffer because of a decline in numbers.
  - ♦ The redistribution of red deer within their ranges is another potentially important change which may arise from changing culling methods; this will lead to changes in browsing and grazing pressures, and visibility.
- We have relatively few data on hares with which to assess the impact of a ban on hunting. However, we note that they are not regarded as a serious pest, except where locally abundant, and that organised hunting with dogs (with packs of hounds or in coursing competitions) takes only a tiny fraction of the cull.
  - ♦ We surmise, therefore, that there will be little impact on farmers or foresters of a ban on hunting hares with dogs. A ban could have a potentially detrimental impact on hare conservation in some areas, where they are encouraged for hunting with hounds or for coursing.
- We have insufficient data on the numbers of mink killed by different methods, and of the extent of the damage they cause, to assess the impact of a ban on hunting. However, because of the small number of hunts, any impact will be highly localised. Conservationists widely believe that mink hunting has the potential to cause disturbance to wildlife, but again we have no data with which to assess this.

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10. Appendix 1: Species descriptions

Among the four groups of species with which this report is concerned are two predators (the fox and the American mink), and eight herbivores (two hares and six deer). The two species of hare present in Britain are the brown hare and its smaller upland cousin, the mountain hare. Of these, the brown hare is the more important quarry species. There are six species of deer: red, roe, fallow, sika, Chinese water, and muntjac; red, roe and fallow deer are widely subjected to sport culling as, increasingly, are sika and muntjac. None of the species is internationally threatened. Indeed, red fox, American mink, mountain hare, and red deer are widespread in Eurasia and North America.

Of the ten species covered in this report, the mink, sika deer, muntjac, and Chinese water deer are alien species introduced in the 19th and 20th centuries, while fallow deer and brown hare result from much earlier introductions in the 11th century and in the Bronze or Iron Age respectively (Yalden, 1999). However, some British populations of all ten species have arisen through introduction, reintroduction, or translocation by man.

By comparison with birds, the abundance of mammals can sometimes be misleading because most are secretive and nocturnal. For example, nationwide estimates for brown hares in Britain at c. 800,000 are comparable with magpies, which number 640,000-1,180,000 adults in spring (Gibbons *et al.*, 1993), and are about three and half times as common as the fox (Harris *et al.,* 1995).

A summary of the ecology and natural history of foxes, hares, deer, and mink is given below; full accounts can be found in Macdonald & Barrett (1993) and Corbet & Harris (1991). Population estimates given below are for pre-breeding populations and are taken from Harris *et al.* (1995). Further estimates relating to population growth and mortality are given in section 4.3.

10.1. Red fox (Vulpes vulpes)

10.1.1. Where are foxes found in Britain?

Red foxes are found throughout the British mainland. They are present on the Isle of Wight as well as on Anglesey, and have been introduced to some Scottish islands. They were introduced to the Isle of Man in the early 1980s, but a 1999 survey (Reynolds & Short, in prep.) showed that they are now either absent or at undetectably low density (<1/km²). Foxes are highly adaptable, making use of both countryside and urban areas. Colonisation of urban areas and (particularly) suburban fringes has taken place since the 1940s. Favoured habitats include farmland, woodland, scrub, and residential suburbs, but foxes are also common on moorland, mountains, sand dunes, and in cities.

10.1.2. How many foxes are there in Britain, and are their numbers changing?

The pre-breeding British fox population totals an estimated 240,00 (195,000 in England, 22,000 in Wales), ranking them 21st in a league table of abundance amongst 64 British mammal species. Local densities are highly variable. In a typical square of mixed suburbs and farmland in Oxfordshire, overall adult spring density is *c.* 2/km^2. Numbers of foxes killed by gamekeepers, particularly in East Anglia, have been increasing since the 1960s, but it is not clear whether this represents an overall increase in fox abundance.

10.1.3. What do foxes eat and how do they behave?

Although they are Carnivores, foxes have an omnivorous diet including invertebrates and plant material. They feed opportunistically on prey ranging from earthworms and beetles, to berries and fruit, to small mammals and scavenge. They typically kill birds and mammals up to about 3.5kg.

Foxes live in pairs or groups of up to six, generally comprising one adult male and several females. Their ranges can cover as little as <40ha, or as much as >4000ha, depending on habitat. Their dens are called ‘earths’ and can be enlarged rabbit burrows, badger setts, rock crevices, drains, or under buildings. Foxes are mainly nocturnal and crepuscular, though they will be active by day if undisturbed.

10.1.4. What are the fox’s life history traits?

In the UK, usually only one or two females in a group breed (see section 1.3.2.a). Mating generally occurs between December and February, and females are receptive for only three days. Cubs are born from March to May in litters of 4-5, and are weaned after about six weeks. Both parents, and sometimes other females in the group, care for the young.

People (through road accidents, rodenticide use, and deliberate culling) are typically the major cause of fox mortality, which is especially high amongst dispersers. Foxes have few predators in Britain although golden eagles may kill both cubs and adults, and badgers and dogs may kill cubs. Foxes can live up to 9 years in the wild, though only an estimated one in ten thousand will do so. Roughly 75% of foxes die in their first year, and thereafter mortality is approximately 50% in each adult year.

10.2. Deer

In this section we briefly compare the six species of British deer. The most salient characteristics and distribution of each species are compared in Table 10-1. More detailed accounts are given below for red (section 10.2.5), and roe deer (section 10.2.6).

Adult males of five of the species carry antlers, which are shed and re-grown every year; Chinese water deer are the exception, and have upper canine teeth developed as tusks. Male deer are generally somewhat larger than females, but the degree of such difference between the sexes varies between species.

10.2.1. Where are deer found in Britain?

One or more species of deer are known to occur in every county of England, Wales, and Scotland. In comparison to the rest of the country, the distribution of deer in Wales and the northern part of the Midlands is still relatively patchy, and restricted largely to fallow, but roe and muntjac are now increasingly colonising these regions too.

While all deer resident in the wild in Britain tend to show a preference for wooded habitats, there are subtle differences in the actual type selected by different deer species, as well as by the same species in different regions of the country. Habitats favoured by the different species are selected not only on the basis of food resources, but also for shelter from harsh weather and concealment from disturbance.

The larger species thrive in large broadleaf or mixed forests, but sika appear to do equally well on more acidic terrain such as offered by conifer dominated forests with access to some rough grazing and deciduous browse. The smaller species tend to be more dependent on cover, so that roe and muntjac deer prefer woodlands with dense and varied undergrowth, while Chinese water deer favour marshy woodland and woodland margins bordering good quality grazing areas. However, all the species adapt readily to using less favoured habitats. For example, fallow, and increasingly also roe, have adapted well to living in mosaics of woodlands and fields, often offering only small wooded areas, but an abundance of suitable feed on the surrounding agricultural pastures and crops.

Table 10-1 Characteristics of free-living deer in Britain

	Species: Red	Sika	Fallow	Roe	Muntjac	Water deer
Adult wt Males: Females:	90-190kg 55-115kg	60-110kg 35-50kg	55-105kg 35-50kg	18-32kg	10-18kg	12-19kg
Max. ht at shoulder	120cm	85cm	100cm	75cm	50cm	60cm
Coat colour	Dark red to brown; pale buff rump patch	Brown to black, with white caudal disk	Very variable: light brown, black, white, with or without spotting	Reddish brown to brown-grey in winter; white rump patch	Chestnut to brown-grey in winter	Reddish brown
Visible tail	Yes	Yes	Yes	No	Yes	Yes
Tail colour	Brown	White	Variable, often black stripe	Anal tuft in females	Brown	Brown
Antlers	Round; single spikes; up to >20 points	Round; single spikes; up to 8-10 points	Flattened 'palmate' in old males, round in juveniles	Round; up to 6 points	Single short curved spikes	None
Main habitats used in UK	Open mature forest; remote open Scottish hills and English moorlands	Dense conifer, deciduous, or mixed woods or wood fringes and boggy scrub.	Open deciduous forest; also extensive use of mixed woodlands/farmland areas.	All woodland, preferably thickets; sometimes also moorland and hedgerow/ field systems	Dense, preferable diverse woodland; e.g. neglected coppice, scrub unthinned plantations	Reed beds, woodland with access to fields
Diet	Grass, crops heather, leaves and twigs, trees and shrubs.	Grass, herbs, shrubs nuts, berries; woody browse in winter	Grass, crops, nuts, berries; some browse, especially hardwood shoots	Tree and shrub leaves, buds and twigs; herbs, some grass, cereals and fruit	Mostly shrubs, especially bramble and raspberry; nuts, broadleaved browse	Grass, herbs crops, browse
Social behaviour	Groups, often sexually segregated outside rut	Groups, often sexually segregated outside rut	Groups, often sexually segregated outside rut	Males territorial Apr-Aug.; loose pair bonds	Solitary, loose pair bonds	Solitary; in pairs Dec-Apr
Rut	Late Sep/Oct	October	October	July/August	non-seasonal	Nov/Dec
Birth	June	June	June	May/June	any; 7 month intervals	Jun/Jul
Young p.a.	1, rarely 2	1, rarely 2	1, rarely 2	1-2, up to 4	1	2-3, up to 6
Max height Browsing: Fraying:	1.5 m 1.8 m	1.4 m 1.6 m	1.4 m 1.6 m	1.2 m 0.8 m	0.9 m 0.5	0.9 m

10.2.2. How many deer are there in Britain, and are their numbers changing?

Roe deer are the commonest deer in Britain (500,000 pre-breeding individuals), followed by red (360,000) and fallow deer (100,000). Of the three recently introduced species, the muntjac is the most abundant at 40,000; there are an estimated 11,500 sika deer, and at 650 the Chinese water deer is currently one of the least numerous British mammals (ranked 59th).

Populations of many species are increasing. Red and roe are discussed separately below. The number (but not range) of fallow deer is thought to be increasing, but the magnitude of the increase is unknown. Muntjac are currently rapidly increasing in numbers in England, possibly due to a recent lack of severe winters, and there is scope for continued expansion. Populations of sika deer in England are relatively small and appear to be increasing only slowly, although throughout Scotland they are increasing in density and in range. Chinese water deer populations are not increasing, and are likely to remain small.

10.2.3. What do deer eat and how do they behave?

The most common foods for each of the six species are summarised in Table 10-1. The three ‘small’ species (roe, muntjac, water deer) are particularly selective in their feeding style, browsing highly nutritious plants or plant parts such as buds and shoots. The larger species (red, sika, fallow) are intermediate feeders tending towards grazing (particularly fallow), but also taking advantage of ephemeral food sources such as tree mast and browse when available.

Deer tend to alternate between quite intensive feeding bouts in the open along rides, clearings, or fields adjoining the forest, following which they return to rest and ruminate in shelter and away from disturbance. Feeding may take place throughout the entire day at intervals of 2-3 hours, although in areas of high disturbance deer may become increasingly nocturnal, feeding in open areas only at dawn and dusk or during the night.

Red, sika and fallow tend to be herding animals and are commonly seen in groups of 2-8 in woodland, with much larger feeding groups sometimes forming on open ground. In these species, males and females form segregated groups for much of the year, with mixed sex groups being encountered mostly during the autumn rut and early winter.

Roe, muntjac, and Chinese water deer exhibit rather more solitary life-styles, although roe again may form groups when feeding in open fields, especially during winter. Adult roe bucks may defend exclusive territories against other males for several months of the year (April to August), although these can overlap with one or more female ranges, as well as with ranges used by non-territorial males. Muntjac and Chinese water deer are essentially solitary, but does may often be seen with their most recent offspring and/or in pairs with an adult male.

10.2.4. What are the deer species’ life history traits?

Five of the British species are seasonal breeders with a well-defined rutting season, and most females producing offspring between May and July. In this respect it is the muntjac that are the exception, as they may produce young at any time of the year at approximately 7-month intervals. Roe are also unusual because they exhibit delayed implantation (the egg does not implant in the womb until around five months after fertilisation in July or August).

Only two of the species (red and sika) are able to interbreed to produce fertile hybrid offspring; such hybrids are known in several areas in Scotland and the north of England. As sika deer are also extending their range in the South and South-West of England some further interbreeding with red deer in these areas in the near future seems inevitable.

10.2.5. Red deer (Cervus elaphus)

10.2.5.a. Where are red deer found in Britain?

The distribution of red deer is patchy in Britain, although locally they may be very abundant. The English population is particularly numerous in the south-west. Numbers in Wales are low.

Red deer are found in a great diversity of habitats, including woodland, grassland, moor and scrub, but are rarely found in large tracts of dense forest. In some areas, such as the Scottish Highlands, Exmoor and the Quantocks, they are found on upland moors above the tree line (although this may not reflect the natural tree line due to artificial moor maintenance). While in Scotland their ranges may include much open moorland year-round, the majority of red deer herd in the South-west of England tend to be closely associated with wooded valleys often surrounded by farmland.

10.2.5.b. How many red deer are there in Britain, and are their numbers changing?

The total pre-breeding population of red deer is estimated at 360000, making it the 24th commonest species. The majority of red deer are in Scotland, with <50 in Wales and 12500 in England.

Densities can vary from <1-40 individuals per km². Average densities in forestry plantations are in the region of 5-15 individuals per km², and in open ground are about 9 per km^2. High densities are associated with lower growth rates, higher over-winter mortality in calves and yearlings, reduced fertility and fecundity in hinds and retarded antler development in young stags.

There has been a steady increase in numbers of red deer in Scotland since the 1960s; this is thought to have stabilized now, with annual culls well over 50,000 (<70,000 in 1999). Red deer are increasing both in range and in numbers in south-west and north-west England, but are probably static elsewhere.

10.2.5.c. What do red deer eat and how do they behave?

Red deer are versatile feeders. Grass predominates, supplemented in woodland by browse (up to 180cm), shrubs and tree shoots, and in moorland by sedges, rushes and heather.

Red deer are social animals, although adult males and females are segregated for much of the year. Female herds typically consist of several matrilineal groups, while stag groups are less stable and may consist of unrelated males. Dominance contests occur amongst mature stags during the rut and breeding males defend a harem of females and their young. Group size depends on habitat and weather: herds tend to be larger on moorland (with groups >40 not uncommon on Exmoor and The Quantocks or the Scottish Highlands). Home ranges of hinds on Exmoor are 250-750ha (Langbein, 1997), but can be over 1000ha in forest with unplanted or mature compartments and open moorland in Scotland (Catt & Staines, 1987). Ranges of stags in both regions tend to be somewhat larger (1000-3000ha), and move seasonally between rutting and non-rutting areas often several kilometres apart.

10.2.5.d. What are the red deer's life history traits?

Mating occurs between September and October, births from late May to early June. The proportion of hinds conceiving as yearlings, which may vary from 0-90%, is closely related to their ability to reach certain threshold bodyweights, and reflects environmental conditions. Calves are weaned at 6-10 months and are cared for only by the female, which they begin to accompany after 7-10 days. Antler development and age at sexual maturity also depend on habitat quality, although males tend to reach their physical prime by about seven years old.

Annual adult mortality depends on habitat and increases rapidly after the ninth year of life. Mortality is highest in calves less than ten months old, the rate being related to birth weight, relative population density and climatic conditions.

10.2.6. Roe deer (*Capreolus capreolus*)

10.2.6.a. Where are roe deer found in Britain?

In Britain, roe deer are now the most widespread of the deer species. Although common in England in medieval times roe deer became extinct during the 16th or 17th century, except near the borders with Scotland. Only a few decades ago roe were still largely restricted to northern England, the Southeast and Norfolk, but they have expanded their distribution steadily, and are now present once more in most English and many Welsh counties.

They occur in a wide variety of habitats, most commonly patchworks of woodland and fields or in woodlands containing open ground.

10.2.6.b. How many roe deer are there in Britain, and are their numbers changing?

With an estimated pre-breeding population of 500,000, roe are the most abundant deer species in Britain, and the 20th most common terrestrial British mammal. There are very few roe in Wales (50), and most of the population is in Scotland; at around 150,000 head they are nevertheless also the most abundant deer species in England. Over-winter pre-breeding densities commonly vary from 5-30 per km^2, with up to 70 per km^2 recorded in some areas in southern England (Staines *et al.*, 1999; Gill, 1994).

10.2.6.c. What do roe deer eat and how do they behave?

Roe deer are selective and versatile feeders, feeding predominantly on herbs and shrubs, buds and shoots of trees and bushes, as well as grasses and fungi.

Roe deer may be active throughout 24 hours, feeding in or close to forest edges, and only venturing into more open areas from dusk to dawn. They rest in dense woodland stands (preferably conifer thickets) or dense shrub. In summer, roe deer are usually solitary or occur in small groups consisting of a doe and her kids, and sometimes a buck. Yearlings of both sexes may accumulate to form a non-territorial group. Yearling females also sometimes choose to stay with a territorial buck, and during the rut they form pair bonds for a few days. Sometimes, in large fields during the winter, roe deer may congregate in herds of up to 30 individuals with a linear rank amongst the males. Rank changes when antlers are cast (November-December), and is reasserted when cleaned of velvet (March). A male’s dominance diminishes with distance from his summer territory. Rank amongst females is less marked, and in late winter, even the previous year’s young males may be dominant to the highest-ranking female.

Home range sizes vary widely, but most European studies have determined average range sizes of 30-170ha, with smallest ranges in woodland and largest for farmland populations (Staines *et al.*, 1999). Within populations, the largest territories tend to be held by males, and are retained from year to year and defended from April to September.

10.2.6.d. What are the roe deer’s life history traits?

The roe deer is the only artiodactyl known to show delayed implantation. Mating occurs in July and August, implantation in late December and births the next May or June. A false rut occurs in October, the significance of which is unknown. Delayed implantation may allow parturition date to be modified by environmental conditions in spring: a late spring leads to late births, and vice versa. Up to three young may be born (more commonly one or two), and are usually weaned at 6-10 weeks. Young fawns are nursed only briefly, spending approximately 23 hours alone per day during the first few weeks of life.

Mortality is highest up to a few weeks after birth. There may be heavy predation on fawns by foxes and dogs, and many are killed road accidents and agricultural machinery, or die during their first winter due to starvation and respiratory infections, or during spring dispersal.

10.3. Hares

There are two species of hare in Britain: the brown hare, *Lepus europaeus*, and its smaller, less numerous cousin, the mountain hare, *Lepus timidus.*

10.3.1. Where are hares found in Britain?

Brown hares are widespread throughout lowland Britain and on some offshore islands. Their distribution is very patchy: they are scarce in some counties, such as Cornwall, and abundant in others, notably East Anglia, probably a response to the distribution of foxes. Mountain hares, in contrast, are essentially a Scottish species, occurring especially on the grouse moors of northeast Scotland. In England, mountain hares occur only in the Peak District and the Isle of Man; the Welsh population is probably now extinct.

Brown hares are found in flat countryside amongst open grassland and farmland up to 500m. They favour mixed arable and livestock farming systems that produce a patchwork of crops and grass at different stages of growth. Hedgerows and woodland are often used temporarily for food or shelter. In the Peak District mountain hares occupy higher altitudes than brown hares, and are found predominantly in heather moorland, although they can live in grassland and forestry elsewhere

10.3.2. How many hares are there in Britain, and are their numbers changing?

In a league table of 64 terrestrial British mammals the brown hare is the 17th most abundant out of a total of 64, while the mountain hare is 25th. There are an estimated 817,500 brown hares in Britain, of which 572,250 were in England and 58,000 in Wales. Mountain hares have a pre-breeding population of around 350,000, only 500 of which occur in England.

Densities of both species are very variable even within habitats, and from year to year at the same site; there is an approximately ten-year interval between peak numbers of mountain hares in Scotland. Brown hare densities range from <1-140 per km^2, and may be more than twice as high in arable areas as pastoral areas. Mountain hare densities are usually less than 10 per km^2.

Game bag records indicate a significant decline in brown hare numbers in Britain during the 1960s and 1970s, probably due, at least in part, to modern arable farming methods, particularly agricultural intensification. The brown hare is now included as a UK Biodiversity Action Plan species on the short list. However, brown hares are still locally common, and records from game bags and hunting records (from beagling) indicate that, nationally, numbers have been stable since 1983. There is some controversy over this data, with other surveys showing a decline (e.g. Hutchings & Harris, 1996).

10.3.3. What do hares eat and how do they behave?

Hares feed on grasses, herbs, and the bark of shrubs and trees such as willow and juniper. Short young heather forms a large part of the diet of mountain hares, while brown hares take more grass and herbs, including arable crops

Hares of both species are predominantly nocturnal, with some morning and evening activity in summer, and both are essentially solitary animals, though they come together in loose aggregations when feeding or breeding. Hares dig shallow depressions called ‘forms’, which they use as resting places and to rear their young. Mountain hares have discrete day and night ranges for resting and feeding respectively, which may be up to 2km apart with an altitude difference as great as 200m. Their total range can therefore be large: in northeast Scotland, average home range size is 44-130ha. Recorded ranges for brown hares are 16-78ha.

10.3.4. What are the hares' life history traits?

Breeding occurs from February through to August (mountain hare) or October (brown hare), and occasionally in winter. Litters of up to three leverets are tended by the female only, and are usually weaned at around 3 weeks old.

Populations of both species have a rapid turnover. Brown hares in their first winter may survive less well than older individuals, but otherwise survival is probably constant with age. Brown hares are regularly shot for game and as a pest. Mortality also results from numerous other causes including predation by foxes, agricultural machinery, road casualties, poor nutrition, and disease.

Among mountain hares, most natural adult mortality occurs between February and May while juvenile mortality peaks between August and October. Up to 75% of adults may die annually during population declines. Shooting by gamekeepers is a major source of mortality; poaching and lurchers are also a problem in the Peak District. Predation is generally not thought to be a major mortality factor, though wet, freezing weather is, particularly when blanketing snow reduces food availability or leverets become wet.

*10.4. American mink (*Mustela vison*)*

10.4.1. Where are mink found in Britain?

American mink were first confirmed breeding in the wild in 1956 in Devon (Linn & Stevenson, 1980), and are now widespread in the British Isles. There are relatively high densities of mink in south-west England, west Wales and west Scotland: they are not found in north-west Scotland and north-west Wales, but are spreading into East Anglia and east Yorkshire.

Mink are adapted to both terrestrial and aquatic habitats. Streams, rivers, marshes, ponds, and lakes with abundant waterside vegetation are favoured habitats, as are some coasts and estuaries, especially those with broad littoral zones, abundant cover, and rockpools. Mink can live near urban areas if there is sufficient cover and prey.

10.4.2. How many mink are there in Britain, and are their numbers changing?

The very fluid status of mink in Britain has been monitored during national surveys by the Vincent Wildlife Trust (VWT), designed to show trends in otter (surveyed 1977-78, 1984-86 and 1991-94) and water vole populations (1989-90 and 1996-1998) (Strachan & Jefferies, 1993, 1996; Strachan *et al.*, 2000, in prep). In general, mink are continuing to expand their range in eastern England (91% increase in sites in East Anglia over the period 1984-1994) and into northwest Scotland. Western Britain, however, has seen a considerable decline in the number of sites occupied by mink.

The otter surveys of the VWT detected a decline of mink in the South West region between 1985 and 1992. More recently, WildCRU has re-surveyed a random sample of the VWT sites in the South West region. This survey revealed that mink have suffered a further decline between 1992 and 1999. Mink are today found only at 30% of the surveyed sites (Bonesi pers.com.) compared to 70% in 1985 (Strachan & Jefferies, 1993). WildCRU has also detected a decline in the mink population inhabiting the Upper Thames catchment, which is likely to be a consequence of the recent otter re-introductions (Bonesi pers.com.).

Harris *et al.* (1995) estimated the mink population in 1990 at over 110,000, with fewest in Wales (9750) and the remainder split between England (46,750) and Scotland (52,250). These figures need revision. Surveys during the 1990s suggest that half to two-thirds of the population of mink has been lost in the last decade (Strachan pers. comm.), indicating a revised figure of 35,000-60,000 mink in Britain

It has long been recognized that local mink densities do not remain constant throughout the year, but follow a seasonal pattern in response to the reproductive cycle and behavioural changes. A stretch of river, which usually supports a single mink, may in some months provide a temporary home for more than five. Optimal habitat can average a density of 0.44 individuals per km of river.

10.4.3. What do mink eat and how do they behave?

Mink are carnivorous and opportunistic. In summer, rabbits are often the most important food. Hares, water-birds, rodents, crustaceans, and fish will be taken, as well as amphibians, snakes, and small invertebrates. In winter, there is a greater emphasis on fish. In coastal habitat, gulls are the most common avian prey, mostly taken as juveniles or carrion.

The mink is nocturnal or crepuscular, but may be active at any time, depending on the activity and availability of prey. On the coast, activity may be influenced more by the tidal cycle than day or night.

Mink climb well, and dens may be above ground in scrub or brush piles, or among tree roots, stones, in a hollow tree or in a rabbit burrow. An existing cavity is nearly always used. There are several dens within one home range, most being within 10m of water. Home ranges are linear where they occur along rivers and shores, or irregular in marshland, the size of which is inversely proportional to habitat productivity. In coastal areas, the average home range length is 1.3km. In riverine habitats home ranges span 1-6km. Mink are largely solitary, although the ranges of males and females may overlap considerably.

10.4.4. What are the mink’s life history traits?

Mating occurs between January-March; both sexes are promiscuous, and no pair bonds are formed. Between the end of April and mid-May most females give birth to 3-6 kits in a nest lined with fur, feathers and dry vegetation. The young are weaned at 5-6 weeks and are cared for by the female only. Most mortality is caused by man.

11. Appendix 2: What are the current legislative restrictions on control methods in England and Wales?

11.1. Statutory basis for control

The UK statutes and European directives governing wildlife management have been reviewed by Stroud *et al.* (1998).

- The Wildlife and Countryside Act 1981 is the major relevant piece of legislation, embodying provisions to address conservation, welfare and pest control interests. This Act prohibits the use of certain methods of killing or taking any mammal, regardless of their status as a pest, species of conservation concern, or their protection elsewhere.
    - Mammals listed in Schedule 5 are given full protection against any attempt to kill or take them, while certain methods (traps, snares, electrical stunners, poisons, stupefying substances, semi-automatic weapons, illuminators for night-shooting, mirrors, gas and smoke, live decoys, motor vehicles) are also forbidden against Schedule 6 mammals. None of the species under review in the present context is listed in these Schedules, but this legislation is nonetheless relevant as Scheduled species might be non-target victims.
    - Sections 11 prohibits the use of self-locking snares, bows or crossbows, explosives (other than ammunition for a firearm), and live animals as a decoy for killing.
    - Section 11 requires that snares are inspected at least once a day
- The Wild Mammals (Protection) Bill 1996 makes provisions to protect wild mammals from specific listed acts of wilful cruelty: “If, … any person mutilates, kicks, beats, nails or otherwise impales, stabs, burns, stones, crushes, drowns, drags or asphyxiates any wild mammal with intent to inflict unnecessary suffering he shall be guilty of an offence”.
- The Pests Act 1954 restricts the use of spring traps to models that have been approved by MAFF.
    - The Spring Traps Approval Order 1995 lists those that are approved. In the present context these models are relevant to mink only. All mechanical traps are also now addressed by the very specific performance criteria of the Agreement on International Humane Trapping Standards between the European Community and Canada and Russia. The result of this agreement is that all currently approved traps (including live-capture traps) will be re-tested by MAFF in the near future.
- The Protection of Animals Bill 1911 restricts the environmental use of poisons, with increasingly tight restrictions following in the Animal (Cruel Poisons) Regulations 1963, the Agriculture (Miscellaneous Provisions) Act 1972, and the Grey Squirrels (Warfarin) Order 1973. The net result of these restrictions is that the use of poisons against wild mammals is restricted to moles in underground tunnels, squirrels (in the interests of forestry only), and rats and mice in buildings.
    - Environmental protection aspects of pesticide use are governed by the Food and Environmental Protection Act 1985, which sets up the mechanism by which the Pesticides Safety Directorate must approve pesticides and similar substances for widespread use.

- ♦ The existing Food and Environmental Protection Act 1985 also affects future implementation of any non-lethal methods (e.g. repellents, aversive conditioning, contraception) in which chemical or biological preparations would be used in the environment; these preparations must also be approved by the Pesticides Safety Directorate.

- Both rifles and shotguns are covered by firearms legislation, requiring the holder and operator to possess a firearms certificate on which individual weapons are listed (Firearms Act 1968). Conditions to be met for the granting of a firearms certificate vary among police forces, but in general similar security arrangements must now be made for both types of firearm. For a rifle held for pest control, the holder must have written authority for at least one area of land. Many police firearms officers inspect the land with public safety in mind before granting the licence, although safety is ultimately the responsibility of the operator.

- The Agriculture Act 1947 provides a mechanism whereby MAFF can serve notice to control certain pest species (rabbits, hares, rats, mice and other rodents, deer, foxes, moles, and birds not listed in Schedule 1 of the Wildlife and Countryside Act 1981) to protect crops, pasture, foodstuffs, livestock, trees, banks, hedges or works on any land. Currently, a control notice exists only for rabbits.

Additional species-specific legislation also applies.

11.2. What additional legislation affects control of foxes?

Approval of Cymag® for gassing foxes in underground tunnels expired in 1987, and no application was made to renew it. This effectively prevented gassing as a technique for controlling foxes, as no other substance is currently licensed for this purpose. In principle gassing is not forbidden (it is permitted under the Agriculture Act 1947), but to be approved for use any product would need to satisfy pesticide safety standards on the efficacy, humaneness, human safety and non-target hazards. The market has apparently not justified these development costs.

11.3. What additional legislation affects control of deer?

Legislation regarding deer in Scotland differs from that in England, with the main legislation introduced by the Deer (Scotland) Act 1959 and its most recent revision, Deer (Scotland) Act 1996. Legislation concerning deer in England and Wales is as follows:

- The Deer Act 1991 (England and Wales) is the main piece of legislation covering deer in England and Wales.
  - ♦ Section 1 requires the consent of the landowner or occupier to kill or pursue a deer, or to remove a carcass; a person is not considered guilty if they believe that consent would have been granted.
  - ♦ Section 2 prohibits killing wild-living deer during a close season (Table 11-1).

Table 11-1 Closed seasons for deer.

		From:	To:
Red deer	Stag	1 May	31 July
	Hind	1 March	31 October
Fallow deer	Buck	1 May	31 July
	Doe	1 March	31 October
Roe deer	Buck	1 November	31 March
	Doe	1 March	31 October
Sika deer	Stag	1 May	31 July
	Hind	1 March	31 October

- ♦ Section 3 prohibits killing deer at night.
- ♦ Section 4 prohibits killing, or harming deer by various methods except by shooting with a rifle, and dogs, and prohibits shooting from a vehicle or using a vehicle to drive deer.

  Rifles must have a minimum calibre of 0.240 inches and minimum muzzle energy of 1,700 foot-pounds, and must be used in conjunction with hollow-nosed or soft-pointed ammunition.

  Smooth bore guns (shotguns) must be not less than 12 bore, with a barrel length not less than 609.6mm and a cartridge with shot at least 5.16mm in diameter. Shotguns can only be used for ‘mercy killing’ or where deer are causing damage.
- ♦ Other partial exceptions to Section 4 are granted for:

  Deer on enclosed land.

  Any action with the authority of the occupier of the land.

  Any act by the requirement of MAFF under the Agricultural Act (1947).

  Preventing suffering of an injured or diseased deer.

  Landowners, to allow them to kill deer which are causing damage, where this is deemed necessary to prevent further damage.
- ♦ In addition, the purchase and sale of venison must be by a licensed game dealer and records of such purchases must be kept.

- All species of deer are listed in Appendix III (protected animals) of the Berne Convention 1979. Exploitation of Appendix III species is subject to regulation “*in order to keep populations out of danger, taking into account the requirements of article 2*”.
  - ♦ Article 2 requires parties to take measures to maintain the population at, or adapt it to, levels which correspond with ecological scientific and cultural requirements, while taking into account economics and recreational requirements and the needs of subspecies, varieties or forms at risk locally
  - ♦ Article 8 bans the use of large-scale and non-selective killing methods for Appendix III species. It prohibits the use of indiscriminate means of capture killing and the use of means capable of causing local disappearance, or serious disturbance to, populations. Prohibited methods of hunting include snares, gassing and poisoned baits.

- Section 14 of the Wildlife and Countryside Act 1981prohibits release or escape into the wild of muntjac, sika deer, and sika/red deer hybrids.

11.4. What additional legislation affects control of hares?

Neither species of hare are given special protection under the Wildlife and Countryside Act 1981, but instead are protected by much older acts such as the Game Laws, the Ground Game Act and the Hare Preservation Act. Interpretation of this complex legislation is described in Parkes & Thornley (1994).

- The Ground Game Act 1880 gives tenant farmers the right to kill brown and mountain hares on their land at any time of the year to protect crops.

- The Hare Preservation Act 1892 forbids the sale of brown and mountain hares during their normal breeding season, March to July inclusive.

Mountain hares are given additional special protection under international legislation:

- Mountain hares are listed in Appendix III (protected animals) of the Berne Convention 1979 (see above).

- Article 14 of the EC Habitats and Species Directive (92/43/EC) requires that any taking or exploitation of mountain hares is compatible with the maintenance of a "*favourable conservation status*".

  - ♦ The Conservation (Natural Habitats, etc.) Regulations 1994 make provision for implementing the Habitats Directive in Great Britain. Section 41 restricts the methods for killing or taking mountain hares.

11.5. What additional legislation affects control of mink?

- Section 14 of the Wildlife and Countryside Act 1981 prohibits release or escape of American mink into the wild.