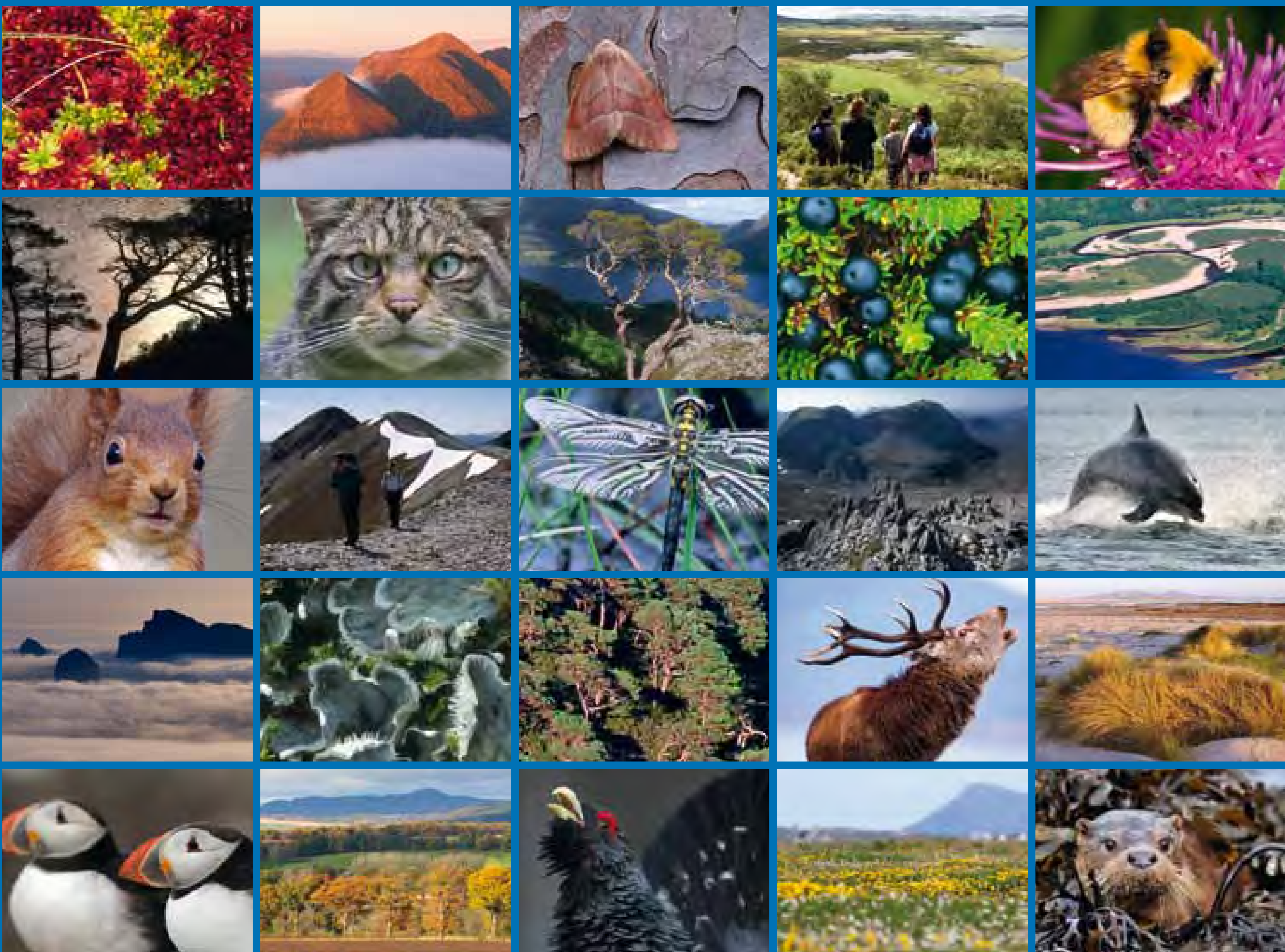
Scottish Natural Heritage
Commissioned Report No. 457

Analysis of cost of preventing establishment in Scotland of muntjac deer (*Muntiacus spp.*)



COMMISSIONED REPORT

Commissioned Report No. 457

Analysis of cost of preventing establishment in Scotland of muntjac deer (*Muntiacus spp.*)

For further information on this report please contact:

James Scott
Scottish Natural Heritage
Great Glen House
INVERNESS
IV3 8NW
Telephone: 01463-725 231

E-mail: James.Scott@snh.gov.uk

This report should be quoted as:

Ward, A. I. & Lees, K. (2011). Analysis of cost of preventing establishment in Scotland of muntjac deer (*Muntiacus spp*.). *Scottish Natural Heritage Commissioned Report No.457.*

COMMISSIONED REPORT

Summary

Analysis of cost of preventing establishment in Scotland of muntjac deer (*Muntiacus spp*.)

Commissioned Report No. 457

Contractor: Alastair Ward, The Food & Environment Research Agency, York.

Year of publication: 2011

Background

Chinese muntjac (*Muntiacus reevesi*) are well established in England and Wales, and their national range is expanding northwards. It may only be a matter of time before they spread or are released into Scotland. There are 12 species of *Muntiacus* worldwide, and some of these are held in captive populations in Great Britain. Escapes from collections could pose additional risks of establishment in Scotland. *M. reevesi* are invasive; they have the capacity for population growth and spread and substantial environmental harm.

The aim of this study was to estimate the costs of preventing muntjac from establishing in Scotland. The objectives were to review their spread and potential for spread in Great Britain; to produce a qualitative assessment of the risk of *Muntiacus* species becoming established in Scotland via different potential vectors; to provide recommendations for the development and deployment of a surveillance system for the detection and quantification of muntjac in Scotland; to propose a strategy for responding to muntjac incursions into Scotland and methods for controlling them; and to estimate the range of costs likely to be associated with preventing the establishment of muntjac in Scotland, and those associated with managing an established population in perpetuity.

Main findings

- Muntjac have been expanding their range throughout England and Wales at a compound annual rate of between 8.2% to 11.6% per year over the past 40 years. However, the rate of spread has varied between locations as a result of various environmental factors, including the presence of other deer species, bioclimatic factors, topography and land cover type.
- It seems only a matter of time before muntjac spread into Scotland, but they could also invade after escape from wildlife collections or deliberate introduction into the wild. Each route presents different challenges. Spread from England may require a regular programme of detection and control to stem the tide. Escapes from collections could relatively easily be resolved if all collections are registered, including details of species and numbers of each sex, and are obliged to report escapes on detection. Illegal deliberate release into the wild may be extremely challenging to prevent, but it could be discouraged via education and penalty. It would also represent the most challenging invasion to resolve since information on

location and numbers might not be known until the population has become well established.

- While *M. reevesi* is highly likely to enter Scotland in the future, no other species of the genus *Muntiacus* were found to pose such a risk. All registered collections of *Muntiacus spp.* in Scotland are of *M. reevesi* and in England, other species of muntjac in collections are rare and located in the south. However, we cannot discount the existence of unregistered *Muntiacus* spp. collections, and these pose an unquantifiable risk to Scotland.
- Faced with the threat of a muntjac invasion into Scotland the policy options seem to be to do nothing, to manage an established population in perpetuity or to ensure the country stays free of muntjac. The former two options would be non-reversible, the final could be reversed if required.
- Costs to the national economy of managing an established muntjac population in perpetuity are likely to range from £457,821 to £1,915,411 per year. Costs of eradicating an outbreak of muntjac are likely to range from £3,683 to £60,625 per outbreak for populations of up to 200 animals.
- Plans to eradicate muntjac outbreaks could be usefully informed by a surveillance system to detect their presence and quantify their abundance and distribution. We propose a staged process, with each step requiring enhanced investment, but yielding more robust results. However, such a system requires development and validation before it could be deployed.
- The culling of muntjac may require different approaches to those traditionally used for other deer species in Scotland. Stalkers that may be involved in muntjac eradication campaigns may benefit from training and experience in muntjac stalking. Other management techniques, such as trapping, should also be considered.

For further information on this project contact:
James Scott, Scottish Natural Heritage, Great Glen House, Inverness, IV3 8NW
Tel: 01463 725231
For further information on the SNH Research & Technical Support Programme contact:
DSU (Policy & Advice Directorate), Scottish Natural Heritage, Great Glen House, Inverness, IV3 8NW.
Tel: 01463 725000 or pads@snh.gov.uk

Table of Contents **Page**

List of Figures **Page**

List of Tables **Page**

Acknowledgements

We are grateful to the many people who provided data, advice and other information whilst compiling this report: Trevor Banham, Norma Chapman, Charles Critchley, Michael Cundy, Dominic Griffith, Charles Smith-Jones, Andy Page, Hugh Rose, James Scott, Peter Watson, Rosie Wilcock, and all the zoos and local authorities who responded to our requests for information. We are also grateful to an anonymous reviewer for helpful comments on an earlier draft of this report.

1 Introduction

The invasion by species of new areas can occur by natural and anthropogenic processes. In comparison with natural rates of spread, humans have greatly raised the rate at which species invade new areas, and when newly-arrived species become invasive they can cause substantial ecological and economic harm (Mooney *et al*. 2005). Most introduced species do not succeed in becoming invaders, in fact, only approximately 10% persist after each of the three main stages of invasion (transport, establishment and growth and spread) (Williamson 1996). However, by definition, those that do survive, persist and spread are invasive, and these typically become major drivers of ecosystem change (Elton 1958). One such species in the UK is Reeves' or Chinese muntjac (*Muntiacus reevesi*).

Muntjac are small cervids originally from south-east Asia. There are 12 species in their native range, only one of which (*M*. *reevesi*) occurs as an invasive non-native in Britain. The first examples were brought to the UK in 1838, and the first recorded release into the wild took place in 1901 when 11 animals were released into woods within and around Woburn Park in Bedfordshire (Chapman and Harris 1996). Further releases are known to have taken place in the 1930s in Northamptonshire and Warwickshire and between 1947 and 1952 in Kent, Northamptonshire, Oxfordshire and the Suffolk/Norfolk border. Following these releases in different parts of England, muntjac spread rapidly. Although most records came from near the main release sites, other records from areas such as Cheshire, the Lake District, Northumberland, Scotland and Wales suggested that human-assisted translocation had played a key role in their spread, and this has been confirmed using molecular analytical techniques (Chapman *et al*. 1994). Partly as a consequence of this, their rate of spread has been greater than any other native or non-native deer species in the UK (8.2% per annum; Ward 2005). Current information suggests that muntjac persist in the border counties of England, and possibly extend into Dumfries and Galloway, and there have been reports as far north as Inverness albeit in isolated pockets (Ward *et al*. 2008).

Concerns have been expressed about the effect of muntjac browsing on native vegetation, especially sensitive ground flora including bluebells, dog's mercury, primrose and orchids, as well as on hardier species such as bramble. Coppice regeneration is attacked and may be checked, and tender tree seedlings are browsed, sufficient for them to be checked. Damage appears to be most severe when muntjac populations are allowed to build up to high densities. The resulting denudation of the understory affects overall biodiversity in affected areas, including invertebrates and birds (Pollard & Cooke 1994; Dolman & Wäber 2008).

Muntjac mate at any time of year and have a gestation period of 210 days. The average interval between kids is only 233 days (Chapman & Harris 1996) and given that muntjac longevity is up to 16 years, this can lead to significant increases in populations. The current population of muntjac in England and Wales has been estimated at around 250,000 animals (Ward & Etherington 2010), with densities of up to 120 muntjac per km^2.

Reports from Scotland of free-roaming muntjac are rare and disparate. However, since there are reports of sightings around the border between Scotland and England, and since the northern extent of the core English population is now in North Yorkshire it would seem prudent to consider whether to prevent establishment of this invasive non-native species in Scotland and if so, how to achieve it. Here we present an assessment of current information pertaining to the requirements and costs of preventing the establishment of muntjac in Scotland.

The objectives of this study were:

- To review the spread and potential for spread of *Muntiacus* species released into the wild in Great Britain.
- To produce a qualitative assessment of the risk of *Muntiacus* species becoming established in Scotland via different potential vectors.
- To provide recommendations for the development and deployment of a surveillance system for the detection and quantification of muntjac in Scotland.
- To propose a strategy for responding to muntjac incursions into Scotland and methods for controlling them.
- To estimate the range of costs likely to be associated with preventing the establishment of muntjac in Scotland, and those associated with managing an established population in perpetuity.

2 Methods

2.1 Reviews

Reviews of several related topics underpin the empirical work undertaken and interpretation of the resultant data. Literature for review was identified from the authors' own collections, supplemented by searches of the online databases Web of Knowledge and Google Scholar. The key word used during searches was 'muntjac'. A brief review of the scientific literature on the spread of muntjac in Great Britain was presented in the Introduction. The potential for further spread is reviewed in the Discussion section. Methods for the detection and surveillance of muntjac are presented in the Discussion, we further review the applicability of a system for the surveillance of English wild boar populations to the potential situation of muntjac in Scotland.

2.2 Risk assessments

It was our intention to perform a risk assessment exercise for each species of muntjac found in collections in Scotland and the far north of England, following the template used by the Non-Native Species Secretariat for Great Britain. In order to identify which species of muntjac are kept in British collections the International Species Information System, (ISIS) online database (www.isis.org, accessed 19th January 2011) was interrogated for records of *Muntiacus* spp. However, as this is a members-only database it is unlikely to represent a full census of muntjac holdings. We also contacted the British and Irish Association of Zoos and Aquariums (BIAZA), who provided additional records, not stored online, dating from 2007. No other databases of licensed or unlicensed muntjac collections were found despite internet searches and discussions with knowledgeable stakeholders. Consequently, our results might represent an underestimate of the total number of muntjac collections in Scotland and northern England and the diversity of *Muntiacus spp*. held within them.

Due to the lack of a comprehensive centralised database of zoo stock lists, information on potential muntjac species and numbers was requested directly from local authorities in the north of England and Scotland. Local authorities are responsible for issuing zoo licenses necessary for public collections and all license applications are required to submit an accompanying stock list detailing all animals. All local authorities listed on www.direct.gov.uk for the North West England (39), North East England (12) and for mainland Scotland (28) were contacted. Information was requested by email directly from the relevant department, through an online enquiry form or via a Freedom of Information (FOI) request. Those that did not respond were later contacted by telephone.

In addition to this several enquiries to relevant individuals or large organisations were made in order to supplement council records or to identify other sources of information. Other organisations that were contacted via email were Blackpool Zoo, Auchingarrich Wildlife Centre, Heads of Ayr Farm Park, Highland Wildlife Park and the Scottish Deer Centre. Jedforest Deer and Forest Park was contacted by telephone after an initial email enquiry. In order to gain information on any *Muntiacus spp.* held in private collections the Veterinary Deer Society were contacted by email and Woburn Abbey were contacted by telephone.

2.3 Effort and costs of muntjac control and eradication

Data on the efficiency or cost of muntjac control in Scotland were not available since muntjac are considered absent from Scotland (Ward 2005). Consequently, we collected this information from a sample of deer managers in areas of England that hold muntjac. The cost of muntjac control is a function of fixed costs (*e.g*. equipment, such as rifles, vehicles *etc*) and variable costs (effort expended on culling duties, and consumables expended during these duties), and these are partially offset by venison sales. Since it is likely that no additional capital resources will be purchased to specifically facilitate muntjac control in Scotland, we assumed that there are no fixed costs. In order to estimate the likely costs of muntjac control in Scotland, we collected data from third parties and these were used to quantify the effort expended (in man.days) on muntjac control in England and the muntjac cull return per unit of effort.

Forestry Commission chief rangers and keepers (five, representing 24 rangers and keepers who shoot muntjac), private professional deer stalkers (two), and amateur stalkers (two groups, including Defence Deer Management, who provided information for 35 volunteer stalkers) were contacted with a request for information. Thus information was sought from 62 people who cull deer in areas where muntjac are present. Information sought was the area of the land over which they stalked deer, the relative density (zero, low, moderate or high) of each deer species present, the number of man.days expended stalking for each of the past three years (2007, 2008 and 2009 seasons), and cull returns for each species for each of these three years. Those contacted represented areas believed to hold very low to high densities of muntjac. From the data provided we estimated the proportion of the population that was culled per unit of effort, which required estimation of the total size of each muntjac population.

To estimate the total number of muntjac within the population pre-cull (N), we applied the following equation:

$N = C+(C(1+r_c))/1+r_b$

Where:
C = Total number of muntjac culled
r_c = Annual rate of cull increase (assumed to equate to the annual population growth rate)
r_b = Annual female birth rate (*i.e*. 0.785 does born per doe per year at an average of a 233 day interval between births).

The rate of cull increase was calculated as the mean annual change in cull numbers from 2007 to 2009. From the estimated total number of muntjac and the area of land we calculated the absolute density. The proportion of the muntjac population culled per man.day of stalking effort was calculated by dividing the total number of muntjac culled by the total number estimated in the population, the result being divided by the number of man.days expended on deer culling to produce an estimate of culling efficiency. This assumes a shoot on sight policy for muntjac, so that all stalking trips would preferentially be focussed on muntjac. This was indeed the policy of the Forestry Commission and Defence Deer Management.

The effort required to eradicate a population of muntjac was modelled using Monte Carlo simulations. We assumed that the population was closed (*i.e*. no immigration or emigration) and that eradication would be implemented sufficiently rapidly to negate impacts of births and non-cull deaths on the population size. Effort was simulated from 0 to 127 man.days in increments of 1 man.day. The starting population was set at 5, 10, 20, 50, 100 and 200 individuals during separate analyses. The number of muntjac culled per unit of effort at time t was calculated as the number in the population at time t-1, multiplied by the culling efficiency and the effort invested at time *t*. The cumulative cull return was calculated and read against effort invested to identify the effort at which the population became eradicated. Culling efficiency was simulated with a triangular distribution, with the lower 95th percentile of estimated culling efficiency representing the left hand limit of the distribution, the median estimated efficiency the most likely and the upper 95th percentile the right hand limit of the distribution. Simulations were run 1000 times for each of the starting populations, whereby the culling efficiency value used for each run was drawn at random from the triangular distribution. Outputs from these simulations were the median and upper and lower 95th percentile of the predicted cull return from the 1000 runs for each level of invested effort. These were then converted to the median and 95th percentile of effort required to eradicate the population. A population was considered eradicated when fewer than 0.5 animals remained.

In order to estimate the costs of each eradication scenario we assumed no setup costs (*e.g*. no vehicle or rifle purchases, no larder setup costs, no planning costs). Staff time was estimated at SNH C grade, which was £159 per day during the financial year 2010/11. Ammunition was estimated at £1.50 per muntjac killed to allow for some zeroing and practice costs, and some variation in calibres used. Travel costs were estimated at £0.80 per mile, with 20 miles travelled per staff member per day for local stalking and 240 miles per week for more distant stalking (*i.e*. if centralised staff are required to work away from home, this will require a 200 mile round trip for travel from home to site per week, 20 mile round trip per day from local accommodation to site). For distant stalking requiring time away from home, subsistence costs were estimated at £76 per person per day, which included £50 for bed and breakfast costs. Benefits were assumed to accrue from the sale of muntjac venison at £7 per carcass (in recent years muntjac venison was purchased by game dealers in England at £0.40 to £0.60 per kg; P. Watson pers. comm.). Since eradication of small muntjac populations (<200 animals) should be achievable within a single year, costs and benefits were not discounted.

To provide comparability, the same costs and benefits were applied both to eradication campaigns and perpetual muntjac control, assuming that they become established in Scotland. Hence, the approach we used provided an estimate of the costs of perpetual muntjac control relative to those for the eradication of smaller, discrete populations, but was not intended to produce a robust estimate of the absolute cost of deer culling throughout Scotland. The costs of perpetual muntjac control are more challenging to estimate reliably since they would in all likelihood become an additional species shot during routine culling duties, as opposed to a species for specific and intense targeting. Acevedo *et al*. (2010) estimated that the relative favourability across most of Scotland for muntjac was 0 to 0.2 that for roe deer. Therefore, we predicted the approximate maximum number of muntjac that might be culled annually if muntjac become fully established throughout Scotland as 0.05, 0.1 and 0.2 times the national roe deer cull from the previous 5 year statutory cull returns (Deer Commission for Scotland 2010) to represent the lowest, most likely and highest predicted cull returns. The mean effort required to cull a deer of any species was calculated by dividing the total number of deer culled by the effort expended in England, pooled across all deer managers for whom data were available. The result was multiplied by the total number of each species culled in Scotland (including the prediction for muntjac) to derive an estimate of the total effort expended on deer culling in Scotland. This assumed that the same strategy used in England would operate in Scotland; that muntjac would be preferentially targeted during culling activities. The predicted proportion of the cull that muntjac would constitute was then multiplied by this total effort and the cost of the effort as described above to produce an estimate of the likely annual cost of muntjac control if muntjac are allowed to become fully established in Scotland. This approach assumed that competition between other species and muntjac would limit the numbers of muntjac, but not those of other species, so may over-estimate likely muntjac cull returns and hence total costs of perpetual muntjac control. On the other hand, statutory returns are likely to under-represent the numbers of each species of deer culled in Scotland, so the number of muntjac likely to be culled, and hence its cost, may also be under-estimated relative to the costs of eradication. However, since the likely muntjac cull was estimated as a proportion of the total reported deer cull, the estimated costs of the muntjac cull are valid relative to the costs of culling the other deer species. Cost were estimated for a single year, and so were not discounted.

3 Results

3.1 Risk assessments

All 28 local authorities in Scotland responded to our request for information, and only Fife Council had issued zoo licenses to keep muntjac which comprised three *M. reevesi* at Fife Animal Park and three at the Scottish Deer Centre. The Scottish Deer Centre confirmed these as one male and two females. The Highland Wildlife Park confirmed that they hold no *Muntiacus spp*. Dundee City Council confirmed no *Muntiacus spp.* are currently held by collections however, they did report that they had previously been held at Camperdown Wildlife Centre. Auchingarrich Wildlife Centre in Perthshire, Heads of Ayr Farm Park in South Ayrshire, and Jedforest Deer and Forest Park also confirmed no muntjac within their collections.

The paper records from BIAZA dating from December 2007 detailed two collections in the North of England holding muntjac. Two males and two female *M. reevesi* were held at Blackpool Zoo and one male and one female *M. reevesi* were held at Lakeland Wildlife Oasis. Lakeland Wildlife Oasis confirmed that they no longer hold muntjac.

All 39 local authorities in North West England responded and only three confirmed collections of *Muntiacus spp*. Blackpool Borough Council reported that Blackpool Zoo hold two males and one female *M. reevesi* and Blackpool Zoo confirmed this. Barrow in Furness Borough Council have issued one zoo licence for one male and one female *M. reevesi* held at South Lakes Wild Animal Park and Bolton Borough Council hold one male and one female *Muntiacus spp.*

Of the 12 local authorities contacted in the North East of England one (Stockton-on-Tees) failed to reply and the remaining 11 confirmed that no zoo licenses were issued for *Muntiacus spp*. Ten other animal collections throughout England reported holding muntjac. All except two of these kept *M. reevesi*, one (in Berkshire) kept a single male *M. muntjak* and the other (in Hertfordshire) kept two female muntjac of unknown species.

Despite attempts to determine the extent of *Muntiacus* spp. held in private collections no official database for this information exists and enquiries to organisations such as the Veterinary Deer Society and Woburn Abbey provided no additional information.

Due to the infrequency of species other than *M. reevesi* in collections, that two of the three known collections hold only one sex and the other holds one male and one female, and that they are all a long distance from the border between Scotland and England, the risk of muntjac species other than *M. reevesi* establishing in Scotland via any means seems vanishingly small, unless there are viable breeding populations held, unregistered, in private collections.

3.2 Effort and costs of muntjac control and eradication

From the 62 deer managers for whom data was provided, useable data were available for 50. These were professional private stalkers (one), amateur stalkers (28) and professional rangers (21), who in 2009/10 culled 1,204 muntjac from 71,265 ha of land. Although some respondents provided data from areas believed to hold high densities of muntjac, our extrapolations of density produced no high estimates, which instead ranged from 0 to 15.2 km^{-2} (median: 3.7; 95^{th} percentile: 0-14.8 km^{-2}). Nevertheless, cull returns generally increased annually between 2007 and 2009 at a median rate of 5.1% per year (95th percentile: -18.0% to 30.8%). Extrapolated numbers of muntjac within populations prior to culling ranged from 3 to 414 (mean: 229 ± 142) and numbers culled had a mean of 227 ± 141 (standard deviation), representing 33.3% ± 28.7% of the total pre-cull population. A median of 1.09 man.days (95th percentile: 0.72 to 20.53 man.days) of effort were expended per muntjac culled, resulting in a median cull efficiency of 0.361% (95th percentile: 0 to 1.06%) of the population culled per man.day.

Culling efficiency was not significantly different between low (0 to 3.7 km^{-2}) and moderate density (5.3 to 14.1 km^{-2}) populations (Mann-Whitney *U* test: P = 0.151), and no relationship was evident between culling efficiency and muntjac density (Spearman's rank correlation: P = 0.385). Despite zero culling efficiency for some stalkers of very low density populations, efficiency was also highest for other low density populations. Consequently, culling efficiency data were pooled, which created the assumption that the range of culling efficiencies estimated here can be expected for muntjac at very low to moderate densities, which are likely to characterise those invading Scotland.

The median estimated effort required to eradicate a population did not differ substantially regardless of the starting size of the population. Median effort required varied from 31 man.days for a population of 5 muntjac to 50 man.days for 200 muntjac (Table 1). This was due to the low efficiency of muntjac culling typically observed in England and the long right hand tail of the eradication curve caused by the proportional nature of the population decay (Figure 1). However, this is biologically plausible and has been the nature of previous eradication attempts, with at least half of the total effort required to cull the last few remaining individuals (Gosling and Baker 1989).

Table 1: Estimated effort required to eradicate muntjac populations of fixed sizes.

	Effort required (man.days)		
Starting population size	Median	Lower 95th percentile	Upper 95th percentile
5	31	66	21
10	35	81	25
20	39	84	27
50	43	90	30
100	47	105	32
200	50	127	34

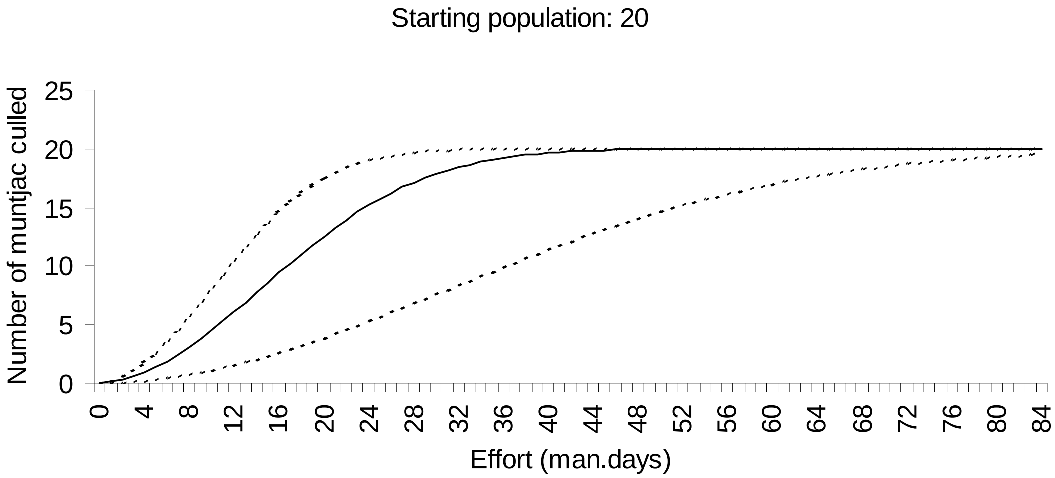


Figure 1: Predicted muntjac cull returns with increasing effort for a starting population of 20 animals. The solid curve represents the medium estimated return for effort, the lower line the lower 95th percentile and the upper line the upper 95th percentile. (© Crown copyright 2008. All rights reserved).

Assuming direct tradability between the number of staff and the number of days needed (*i.e*. assuming no staff subsistence costs and only modest travel costs), median estimated costs of muntjac eradication varied from £5,433 for a population of five animals to £9,050 for a population of 200 animals. However, there was considerable variability in estimated costs due to uncertainty in the amount of effort required to eradicate a population (Table 2). If staff were required to work away from home, the addition of subsistence costs and enhanced travel costs raised total estimated costs substantially (Table 3).

Table 2: Basic benefits and costs of muntjac eradication for fixed population sizes. Costs assume direct tradability between the number of staff and the number of days and no subsistence costs. Benefits from venison sales are given as the median with the range in parentheses.

		Costs (£)		
Starting population size	Venison sales (£)	Median	Lower 95th percentile	Upper 95th percentile
5	35 (30-40)	5433	11558	3683
10	70 (60-80)	6140	14190	4390
20	140 (120-160)	6855	14730	4755
50	350 (300-400)	7600	15825	5325
100	700 (600-800)	8375	18525	5750
200	1400 (1200-1600)	9050	22525	6250

Table 3: Costs of muntjac eradication for fixed population sizes. Costs assume direct tradability between the number of staff and the number of days, and include subsistence costs and enhanced travel costs. Benefits from venison sales (Table 2) can be subtracted from these costs to produce an estimate of net benefit.

	Costs (£)		
Starting population size	Median	Lower 95th percentile	Upper 95th percentile
5	14733	31358	9983
10	16640	38490	11890
20	18555	39930	12855
50	20500	42825	14325
100	22475	50025	15350
200	24050	60625	16450

The mean effort required per deer culled from responses in England was 1.63 ± 0.67 man.days. Statutory cull returns in Scotland from 2005/06 to 2009/10 are listed in Table 4, along with predicted numbers of muntjac if they were allowed to become fully established. Applying to these the cost structure for local culling, the predicted cost of all deer culling (excluding predictions for muntjac) in Scotland was estimated at a mean of £28,404,595 ± £1,444,400 per year from 2005/06 to 2009/10. The annual cost of muntjac control was predicted to vary from £457,821 (muntjac cull return 0.05 times roe deer cull returns, minus the standard deviation) to £1,915,411 (muntjac cull returns 0.2 times roe deer cull returns, plus standard deviation), with a most likely value of £916,057 ± 41,648 (muntjac cull return 0.1 times roe deer cull return).

Table 4 Deer cull returns and predicted cull returns for muntjac, had they been full established in Scotland. Muntjac 0.2 indicates the predicted return assuming 0.2 times as many muntjac returns as roe deer, similarly for Muntjac 0.1 and Muntjac 0.05.

Species	05/06	06/07	07/08	08/09	09/10	Mean	Standard deviation
Red	63611	62625	61354	58496	54203	60058	3795
Roe	33597	31787	32058	32626	29663	31946	1452
Sika	5110	5765	5167	5465	5074	5316	295
Fallow	1634	1398	1818	2036	1798	1737	237
Muntjac 0.2	6719	6357	6412	6525	5933	6389	290
Muntjac0.1	3360	3179	3206	3263	2966	3195	145
Muntjac 0.05	1680	1589	1603	1631	1483	1597	73

4 Discussion

4.1 The spread and potential for spread of *Muntiacus* spp. in Great Britain

The original species of muntjac released into the wild in England was *M. muntjak* (Indian muntjac) in 1838, but these were rapidly eradicated by the Duke of Bedford due to their aggressive nature, and were replaced by *M. reevesi* (Chapman & Harris 1996; Smith-Jones 2004). Several other introductions and translocations during the 20th century have supplemented the spread of their national range (Chapman *et al*. 1994).

By 1972, muntjac occupied approximately 1.4% of the land area of mainland Great Britain, all within England. From 1972 until 2002 they spread at a compound annual rate of 8.2% to cover 16.9% of Britain (Ward 2005). From 2003 to 2007 their rate of spread had increased to 11.6% and they covered 29.1% of mainland Britain, having spread from England into Wales and with a few reports around the border between Dumfries and Galloway and Cumbria (Ward *et al*. 2008; Figure 2). Range expansion has been in all directions away from their original release site in Bedfordshire, at a fairly constant wave speed (1km.year^{-1}; Chapman *et al*. 1994), although locally spread has been much more rapid (up to 2.4km.year^{-1}: Harding 1986).

The Wildlife and Countryside Act 1981 made release of muntjac into the wild illegal in Great Britain without a statutory licence (currently issued by Scottish Natural Heritage, the Countryside Council for Wales or Natural England, as appropriate). Evidence of unlicensed releases (Chapman *et al*. 1994) implies that efforts to prevent deliberate releases may currently be inadequate in isolation to prevent their establishment via this route. Indeed, despite being unconfirmed, reports have been made of muntjac presence in Fife, Angus and Invernessshire, which, if accurate, are extremely unlikely to have arisen through natural spread from England. However, the Muntjac Keeping (Scotland) Regulations 2011, which come into effect on 1st July 2011, require any person to be licensed by the Scottish Ministers if they keep and/or transport *Muntiacus* spp. within Scotland, and make it an offence under the Destructive Imported Animals Act 1932 to not comply with the Regulations or the terms of a licence. The Regulations include strict restrictions on the specifications of enclosures and transport cages. In addition, keepers of muntjac are required to notify Scottish Ministers of any muntjac escapes within 24 hours of becoming aware. These Regulations may substantially reduce the risk of unintentional escapes and the speed with which any escapes can be addressed. Whether they are capable of preventing deliberate, illegal releases should be demonstrated in time.

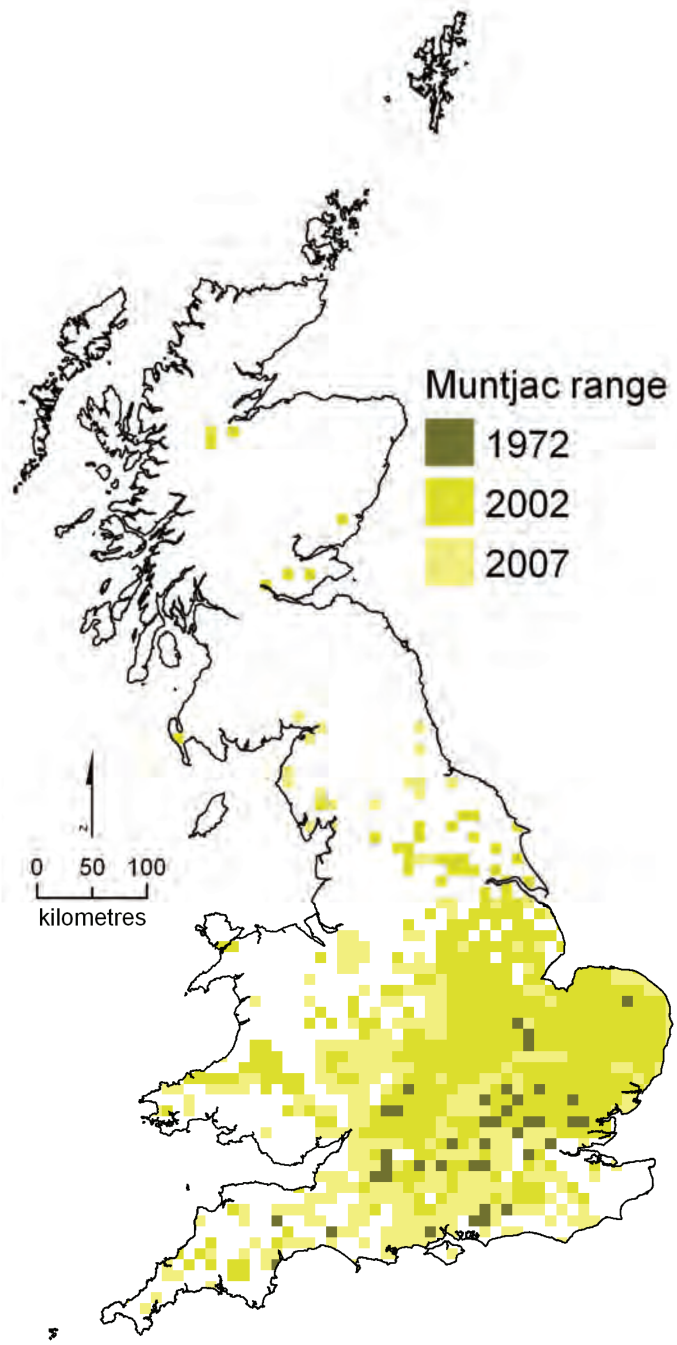


Figure 2: Distribution and range expansion of Reeves' muntjac from 1972 to 2007 (From Ward et al. (2008)).

Acevedo *et al*. (2010) modelled the presence and absence of all free-living deer in Britain and found the following variables to be significantly associated with muntjac presence: decreasing latitude, large variability in seasonal temperatures, decreasing distance to primary roads (which was collinear with the area of urban land), large areas of broadleaved and coniferous woodland, large areas of arable/horticultural land, decreasing area of mountain/upland, larger areas of standing water/canals. In a similar, but importantly different analysis, Ward *et al*. (2010a) modelled the rate of spread of all deer from their 1972 distribution to their 2002 distribution, and found the following variables significantly associated with low rates of spread: presence of fallow, roe and CWD, absence of sika, areas of forest (weak), littoral, mountain, pastoral land, sea, and low environmental diversity. Thus, it is likely that spread from England into Scotland might be more constrained (*i.e.* much slower) than it has been throughout England due to the northern direction, relative paucity of woodland, relative abundance of pasture and area of uplands along parts of the border, and the presence of other, competing deer species. Moreover, plans to prevent muntjac establishing in Scotland via spread from England or those to eradicate escapes or releases from focal points within Scotland should consider exploiting these landscape factors when planning management campaigns.

Unlike other British deer species, muntjac presence was associated with factors related to urbanization. Indeed, their persistence and increasing presence in many urban areas within their range (McCarthy & Rotherham 1994) suggests that there may be parts of Scotland additional to those identified by the models of Acevedo *et al*. (2010), and particularly within the heavily urbanized central belt, which may be available for muntjac to colonise and persist.

Despite having filtered out the unsubstantiated presence records of muntjac in central and northern Scotland, Acevedo *et al*. (2010) predicted that the parts of Invernessshire, Fife and Angus in which these observations were made were more favourable for muntjac than any other parts of Scotland (Figure 3). Nevertheless, parts of Grampian, East Lothian, Lanarkshire and the Borders were also estimated to be moderately favourable, so muntjac could be expected to persist well in these areas if they arrive there. Indeed, a comparison of the relative favourability of these locations for muntjac with roe deer and red deer suggested that muntjac could persist as well, and probably at similar densities, as roe and red deer currently do (Figure 4). In all other parts of Scotland muntjac were estimated to fare less well, with the environment favouring roe and red deer above muntjac (Acevedo *et al*. 2010).

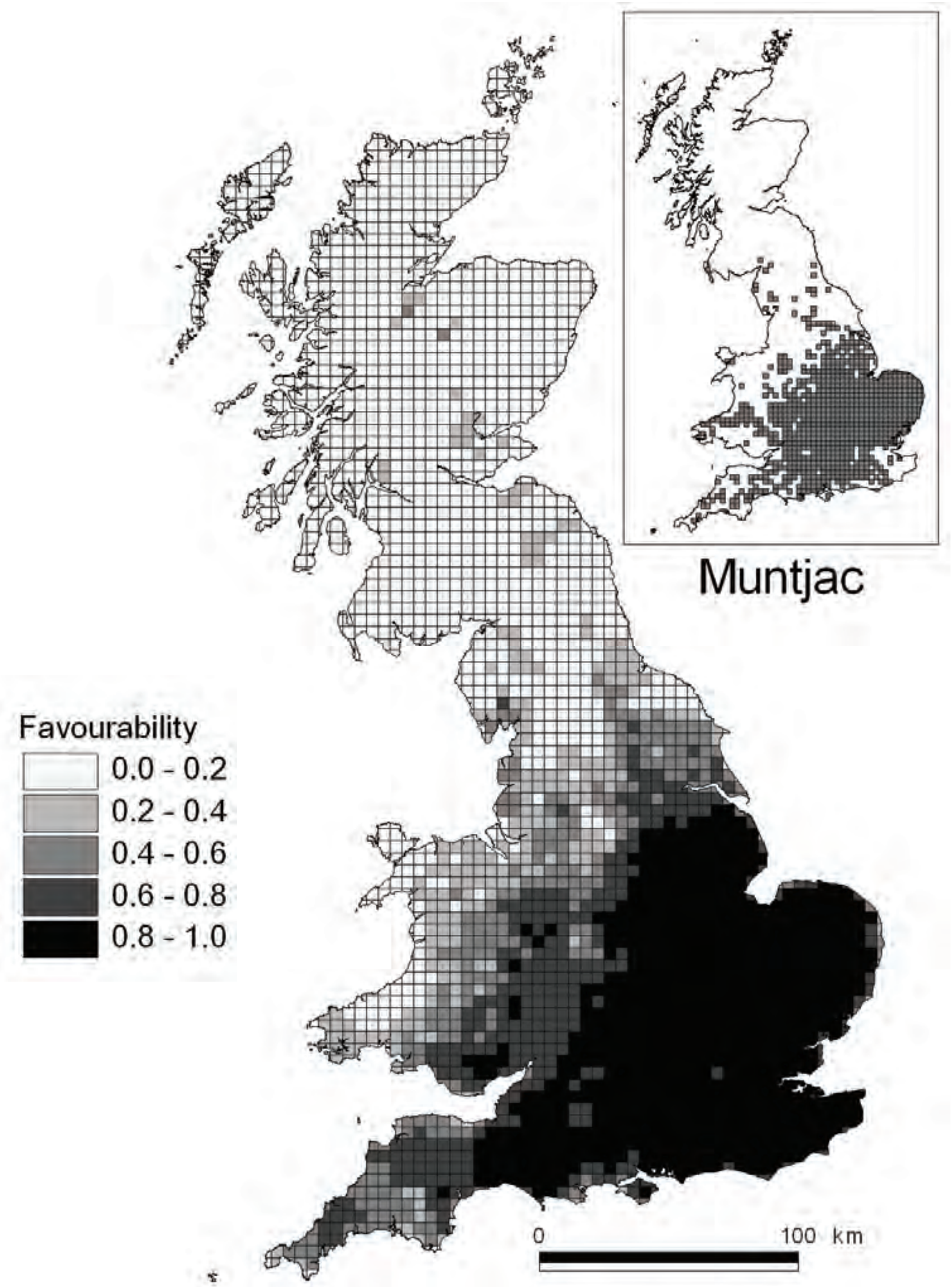


Figure 3: Species favourability in UTM 10 x 10 km squares (0 represents minimum favourability and 1 represents maximum favourability). Current distribution is depicted in the inset. (From Acevedo et al. (2010)

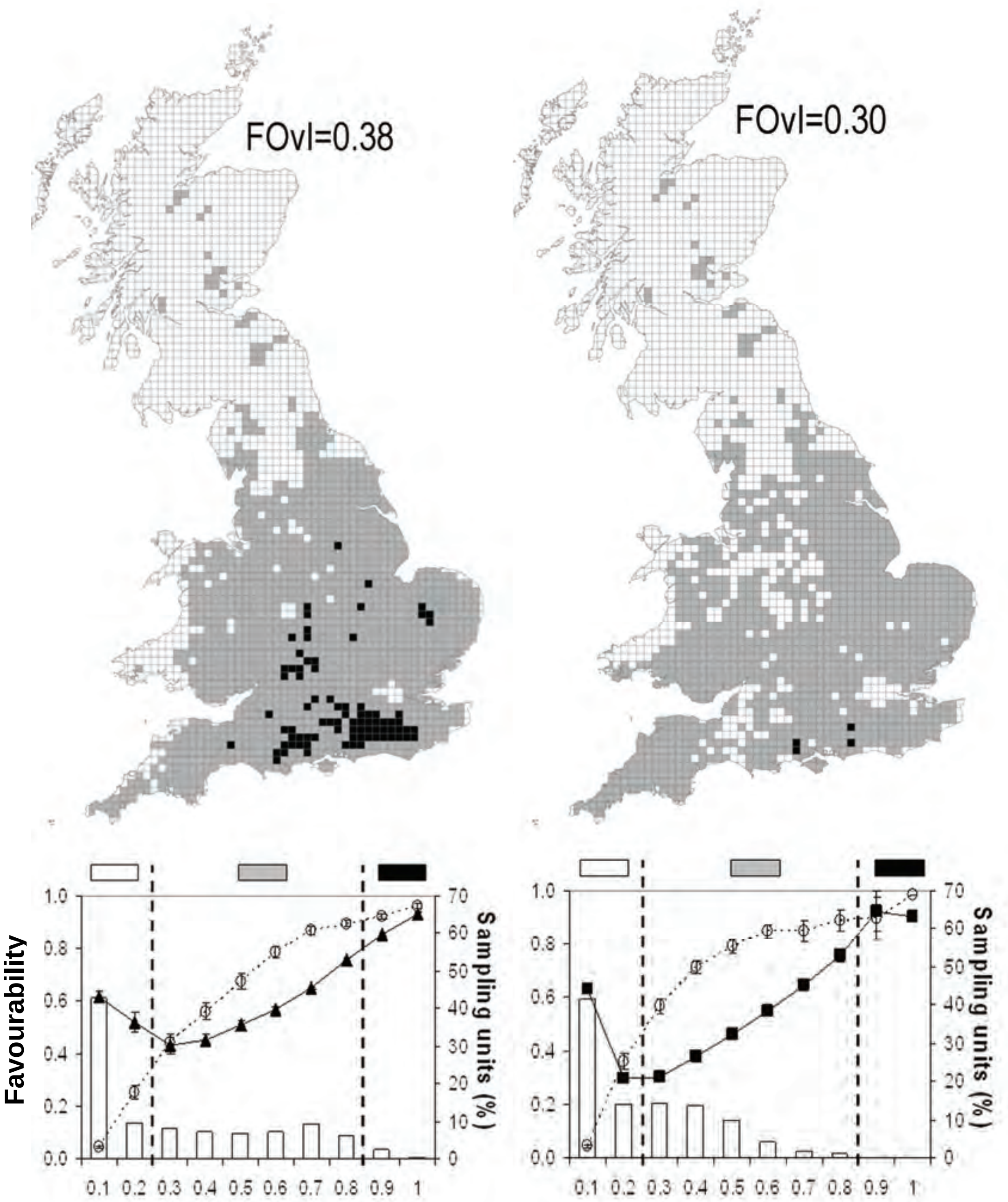


Figure 4: Biogeographical relationships between muntjac (circles and dotted line) and roe deer (left image: triangles and solid line) and red deer (right image: squares and solid line). Variations of mean favourability scores along the relative fuzzy overlap indices are displayed in the graphs. The indices were divided in natural intervals, and mean favourability values (95% confidence intervals) are shown. The number of sampling sites at each interval is also shown in columns. Fixed and unfixed intervals are defined (intermittent and dotted vertical lines, respectively) in the charts and are mapped. In areas with no shading, red and roe deer are likely to out-compete muntjac. In areas with grey shading all three species may coexist, competing on broadly equal terms. In areas with dark shading, muntjac are likely to out-compete red and roe deer. (From Acevedo et al. (2010) © Crown copyright 2010. All rights reserved).

4.2 Routes of entry into Scotland

Muntjac could invade Scotland via three routes: natural spread from England, escapes from private collections, and deliberate releases. Natural spread will most probably be extremely challenging to prevent since there is currently no national policy of range management for any wild cervid in England. While prevention of escapes from collections cannot be guaranteed, rapid reporting of escapes could result in a rapid response for the capture or destruction of escaped muntjac, to prevent them from establishing in the wild. Deliberate and unlawful release of muntjac, for whatever reasons, cannot be completely prevented, but could be minimised, for example via legislation and subsequent punishment, and/or education of those likely to have an interest in muntjac on the potential implications they could have for Scotland. However, it is these unlawful releases that are likely to be the most challenging to detect and monitor. In addition, if anecdote is correct, released animals might have formed established populations before they are first detected by passive surveillance.

4.2.1 Spread from England

Spread from England is likely to be slow and uncertain due to the relative unfavourability of much of the environment around the border (see above). Moreover, some locations are more likely than others to facilitate spread due to differences in favourability. In particular, an almost continuous, if rather tortuous path (at a resolution of 10km by 10km) of favourable environments runs from the core of the English population, through the northwest of England to the Solway Firth. In addition, patches of favourable environment exist around Kielder, Berwick-upon-Tweed and Dunbar, at this same resoultion. Consequently, it would seem prudent to focus muntjac surveillance efforts around these areas when the threat of spread is sufficient. However, the 10km by 10km resolution used by Acevedo *et al*. (2010) is coarse relative to the scale at which muntjac move. Therefore, it would be advisable to more closely map potential environmental corridors that might facilitate spread across the border so that surveillance and control efforts can be more precisely targeted. The resolution of such an analysis should be of the order of 100m by 100m or finer.

Since the rate of spread has been approximately 1km per year (Chapman *et al*. 1994), that muntjac are typically not detected by passive surveillance until they are well established (Smith-Jones 2004), and the British Deer Society's national deer distribution survey reports every 5 years, the point at which surveillance might be upgraded most appropriately is when national distribution data suggest that established muntjac populations are within 10km of the border. At this point regular (e.g. quarterly) review of reports of muntjac distribution from a variety of data collection initiatives (see below) and local estates and forestry organisations could form a useful and cost –effective mechanism for detecting approach to the border. When muntjac are considered likely to be at the border, cost –effective active surveillance for signs of muntjac could be deployed at the key locations mentioned above. Since considerable areas of southern Dumfries and Galloway and the Borders are under the control of the Forestry Commission, a collaborative approach to muntjac surveillance and management between SNH and the FC would seem appropriate and the involvement of other key local stakeholders should be encouraged.

4.2.2 Escapes from collections

A list of collections licensed to hold muntjac in Scotland is held by each local authority, and is known to SNH. Collections are now required to rapidly report escapes should they ever occur under the Muntjac Keeping (Scotland) Regulations 2011. This could facilitate a rapid, intensive response to remove any escaped muntjac from the wild (see Simberloff 2003a). With knowledge of the number of escapees, and little opportunity for them to disperse or expand their population, there seems little reason why the complete removal from the wild should not be possible. Indeed, Indian muntjac originally released into woodlands surrounding Woburn Abbey were rapidly eradicated by shooting at the turn of the 19th/20th century (Smith-Jones 2004). However, it is possible that un-registered collections of muntjac exist in Scotland, and escapes/releases from these are also likely to go unreported, with the same consequences as for unlawful releases (see below).

4.2.3 Unlawful releases

As discussed above, these are likely to constitute the most challenging outbreaks of muntjac to predict, detect, monitor and control of all the possible entry scenarios to Scotland. The reasons for this are that the location of release cannot be predicted and may remain unknown for some time, and the number of muntjac involved and their distribution will be uncertain (Lockwood et la. 2007), and may require estimation to inform eradication plans (Simberloff 2003a). At all but a few locations, it is highly uncertain how well muntjac could survive if released into the wild. It would seem prudent to assume that any escaped population comprising males and females could survive and breed.

4.3 The risk of *Muntiacus* spp. becoming established in Scotland

In order to become invasive a population must overcome a series of ecological and physical barriers (Williamson 1996). In addition, some of these can be stochastic in nature, such as periods of extreme cold or drought. With respect to the potential invasion of Scotland by muntjac, the ecological and stochastic barriers might prove beneficial for limiting or preventing this invasion. We have already discussed how bioclimatic variables are likely to limit the muntjac's ability to colonise Scotland, and periods of sudden extreme cold may provide further benefits. Indeed, during 1962-63 the severe winters experienced in Britain were associated with mass mortality amongst muntjac in eastern England (Chapman *et al*. 1994). Nevertheless, we have also identified that there may be some locations in Scotland in which muntjac could become established, and if they are transported they may spread from these loci. Moreover, while Acevedo *et al*. (2010) predicted highly favourable areas for muntjac, they were not able to identify any areas where muntjac were highly unlikely to persist.

If the experience of England is repeated in Scotland, and any escapes or releases into the wild are left unchecked, we can expect to see muntjac transported to new locations, establish self-sustaining populations in the wild, and after a period of slow growth, they are likely to grow and spread rapidly, at which point impacts on the environment are likely to become all too obvious. If allowed to achieve this status, with time they are also likely to become embedded into local biological and cultural systems (Lockwood *et al*. 2007), which could hamper efforts to control or eradicate them.

The proliferation in recent years of non-native species interest groups is consistent with a much greater awareness of the potential negative impacts caused by non-native species. In addition, it appears that all licensed wildlife collections in Scotland and English counties on and close to the border hold *M. reevesi*, which, in the event of any escapes, are therefore unlikely to present more of a threat to Scotland than are the free-living populations in England. The few records of *M. muntjak* (one in Berkshire) and muntjac of unknown species (two females in Hertfordshire, one male and one female in Lancashire) are consistent with muntjac species other than *M.*

reevesi posing negligible to no risk of establishment in Scotland. We therefore conclude that, at present, and assuming we have seen accurate and complete information, no species of muntjac other than *M. reevesi* pose any perceivable threat to Scotland. This is because collections reportedly containing other species of muntjac are very few, are far from Scotland and contain very few individuals. Consequently, there seems little requirement to conduct a formal risk assessment for other muntjac species, and no requirement to further assess the risks posed by muntjac, since such an exercise has already been published (https://secure.fera.defra.gov.uk/nonnativespecies/index.cfm?sectionid=51 accessed 20/01/11 via www.nonnativespecies.org). That document concluded, with a high degree of certainty that the risk of muntjac establishment was very high (they are already established in England and Wales), that continued spread was inevitable (although this might be slowed by co-ordinated, intensive culling), and that impacts to conservation woodland and via collisions with road traffic were likely to be high. It also concluded that some displacement of roe deer might be observed, but as discussed above, in Scotland this might only occur over a very limited range in isolated pockets. Perhaps the most crucial message arising from that risk assessment was that performance of a risk assessment 50 years ago would have been welcomed. Presumably this would have facilitated adequate action to prevent muntjac from becoming established in England. It is therefore commendable that such thought is being given in Scotland in advance of the potential arrival of muntjac.

4.4 Policy options

Once muntjac have been detected in Scotland, a decision will need to be made about how to respond. The response could be a) an attempt to eradicate, b) a decision to allow the management of an established population of muntjac in perpetuity, or c) to do nothing. Simberloff (2003a) recommended thorough analysis of the potential outcomes of competing policy options, because the decision to eradicate and the ensuing campaign's success will be complicated by both political and social dynamics. While the purpose of this study was not to perform one, it does provide information pertinent to such an analysis, and so it may be appropriate to consider issues in addition to the financial costs of control here.

Doing nothing will inevitably lead to the growth and spread of muntjac populations. Although there is no clear evidence on which to hypothesise the maximum distribution and carrying capacity of muntjac populations in Scotland, densities rose to over 100km^{-2} in the east of England when the population was left unmanaged (Cooke 2004). This resulted in a dramatic decline in the diversity and abundance of fauna and flora within the area, with an almost totally denuded ground flora (Cooke 1994). While it seems that the ecological conditions throughout much of Scotland might be insufficient to host these densities of muntjac, it is likely that they could nevertheless reach sufficient densities to negatively impact native flora with consequent repercussions for the rest of the ecosystem if left unchecked.

A decision to allow the management of an established muntjac population in perpetuity would undoubtedly result in enhanced net costs for landowners. The experience from much of England is that despite unrestricted culling, annual muntjac culls continue to grow. It is particularly noteworthy that in the east of England, two neighbouring land managers (the Forestry Commission and Defence Estates), both of which impose a shoot on sight policy for muntjac over vast tracts of land, continue to see their annual culls increase. These observations support the view of Smith-Jones (2004), who stated that it is very difficult to over-shoot a muntjac population. For most British deer species, the sale of venison at least partially offsets the costs of culling, but there is currently a very limited market for muntjac venison, which consequently attracts a low price (Smith-Jones 2004). In addition, in those parts of Scotland where muntjac might persist favourably (Figures 3 and 4), competition with red and roe deer might result in reducing culls of these species, with potential consequences for those stakeholders who gain revenue from stalking. An industry based on muntjac stalking might hold potential to at least partially replace income lost from the stalking of native deer species, but this and the total economic impact that muntjac might have on Scotland have yet to be estimated.

In addition to the issues outlined above, it is very important to be aware that the outcome of a 'do nothing' policy or a 'perpetual management' policy will, in all probability, be irreversible. Once established, abundant and widely distributed, eradication of invasive populations becomes exceptionally challenging, if not impossible, given the likely financial and logistical constraints (Lockwood *et al*. 2007). In contrast, if an eradication policy is implemented, there will always remain the option of retreating to one of the other policies once lessons have been learned.

The decision to eradicate will have to be made rapidly, or in advance of any muntjac invasion if it is to stand a chance of success before populations grow and spread. It is considered less important to invest in gaining a detailed ecological insight into the invasion than it is to rapidly engage in the eradication campaign (Simberloff 2003b). In order to facilitate this, restrictions to the progress of the campaign should be lifted. Nevertheless, it is considered important to develop rapid methods for the evaluation of the invading population to inform campaign strategy and effort (Simberloff 2003a), and for muntjac in Scotland, a process for such an evaluation is presented below. Finally, it is also important to consider the non-target effects of a potential eradication campaign (Simberloff 2003b), which are often characterised by 'brute force' (Lockwood *et al*. 2007). Non-target effects might be ecological, for example and in this case, shooting disturbance to sympatric fauna might promote behavioural changes. Equally, they may be social and political (Simberloff 2003a), for example if landowners remain non-compliant with the eradication policy. These latter issues are more likely to arise if muntjac are allowed to become embedded in the local cultural system (Lockwood *et al*. 2007).

4.5 Effort and costs of control and eradication

Contingent on the validity of the assumptions underpinning our estimates, the estimated costs of perpetual control were much greater (minimum: £457,821 per year) than estimated costs of eradication, despite the substantial difference between the estimated minimum (£3,683) and the maximum (£69,625) eradication costs. The estimated cost of eradication would become equivalent with the estimated cost of perpetual control when outbreaks of 200 muntjac occurred 6.6 to 27.5 times per year. Outbreaks of fewer animals would need to be more frequent for costs to be equivalent. However, the context for these comparisons is eradication of sporadic outbreaks with control of a fully established muntjac population throughout Scotland. During the years between invasion and complete establishment, total costs of perpetual muntjac control are likely to be lower than those quoted here. Moreover, while it is likely that pressure would be exerted on the Scottish Government to assist with or deliver eradication campaigns, the costs of perpetual management would likely be more diffuse, lying mainly with landowners. Nevertheless, it does seem that in the long-term and from the perspective of the Scottish economy, prevention of muntjac establishment would be more cost –effective than perpetual management of an established population. It is also worth noting that the tangible financial costs reported here do not include tangible costs of, for example, collisions with road vehicles, impacts to forestry and game shooting interests and non-tangible costs of impacts to the natural heritage. However, neither do they include the tangible benefits of an industry based on muntjac stalking (a gold medal trophy buck can attract a fee of £600, but there is little to no profit in harvesting muntjac for venison due to their small carcass size and low value; P. Watson pers. comm.) or non-tangible benefits of muntjac presence. For example, some of those who provided data for this study, all of whom were deer managers, reported admiration and fondness for muntjac, implying that their social welfare might be lower in the absence of this species. It is possible that such values may emerge if muntjac are allowed to establish in Scotland, but they are unlikely to if establishment is prevented.

The key assumptions used when modelling the effort and costs required to prevent muntjac from establishing in Scotland were that the cull returns data represented deer management programmes where muntjac were preferentially shot to other species, that they were shot on sight and that traditional stalking techniques were used to obtain the cull. All of these are true for the data used, but approaches may need to be different in Scotland. No muntjac eradication campaign has been undertaken in England (other than the Duke of Bedford's eradication of several *M. muntjak* during the 19th century), or anywhere else in the world, to our knowledge, so there is some uncertainty regarding the applicability of perpetual control data to eradication scenarios. The assumption of a fixed proportion cull has been used by other eradication modellers, and there is some evidence from the literature on feral pig eradication from islands that this pattern generally holds true (McCann and Garcelon 2008). Nevertheless, for the purposes of planning, we recommend that the estimated upper costs of eradication presented here are viewed as the minimum investment required to support a successful eradication programme.

4.6 A surveillance system for the detection and quantification of muntjac

A cost –effective system for the reliable detection and reporting of muntjac presence will be central to a strategy for the prevention of muntjac establishment in Scotland. However, when muntjac have been reliably detected, decisions on whether or how to deploy control efforts and how much effort is likely to be required will depend on muntjac distribution and abundance within the area.

Methods for estimating density and relative abundance of ungulates have been extensively reviewed (Staines & Ratcliffe 1987, Buckland *et al*. 2001 and 2004, Mayle & Staines 1998, Focardi *et al*. 2002, Hebeisen *et al*. 2007, Mayle et al. 2008). These include kilometric indexes of abundance (KIA) derived from activity signs, such as faecal pellet counts, distance sampling and driven censuses. Some of these methods, such as kilometric indexes of abundance, are used as indicators of population size and trends (i.e. increase, decrease or stable). Such methods typically yield estimates of unknown accuracy and precision, but can be very cost –effective due to the limited amount of survey work required and the simple nature of the data recorded. Consequently, these methods may most appropriately be used to monitor populations whose impacts are below some critical threshold. An involved, combined method for assessing muntjac abundance and impacts was devised in Monks Wood National Nature Reserve, a site with moderate to high densities of muntjac (Cooke 2006). How well this system works for very low density populations, which are likely to characterise invading populations, is uncertain. More expensive methods, such as distance sampling (Buckland *et al*. 2001) can provide accurate, precise estimates of population size and rates of population change, which can be used to determine the effectiveness of population control (Smart *et al*. 2004), which may in turn be suitable for detecting the approach of critical density thresholds (Putman *et al*. *in press*). Recent developments in distance sampling analytical procedures (Buckland *et al*. 2004) and technology (e.g. infra red or thermal imaging) have resulted in significant improvements so that population surveys can be undertaken quickly over fairly large areas of land (e.g. tens of km^2) and yet yield precise and accurate density estimates (Gill et al. 1997; Focardi *et al*. 2002, Smart *et al*. 2004).

Some techniques (such as pellet counts) have been used successfully for high-density muntjac populations (Hemami et al. 2005) but may not be appropriate for surveying muntjac populations at relatively low densities due to low detection rates, which may be compounded by low defecation rates (Chapman 2004). However, from a comparison of dung counting with direct observation, Cooke (2006) deduced that muntjac were unlikely to be detected by direct observation along an 8km transect unless they were present at densities in excess of $10km^2$. Consequently, it is advisable to avoid direct observation and dung counting methods for low density populations, which are likely to characterise an invading population. Indeed, methods relying on direct observation, which have been used traditionally to estimate the abundance of red deer on the open range (Daniels 2006), are likely to be less appropriate for muntjac due to their use of concealing cover and consequent lower detectability. Nevertheless, direct distance sampling using thermal imaging, which relies on this technology to increase the detection rate, has been successfully tested on relatively high density muntjac populations in England (Hemami *et al*. 2007), but has yet to be evaluated for low-density populations. The high level of precision and accuracy achievable using this technique recommends it as the gold standard for moderate to high density populations (Smart *et al*. 2004). Capture-mark-recapture, with recaptures based on camera traps, have yet to be tested for muntjac, but have produced promising results for other ungulates (Hebeisen et al. 2007, Focardi *et al*. 2002). Since muntjac are typically difficult to observe directly, largely due to their secretive nature and occupation of densely vegetated areas (Smith-Jones 2004; Hemami *et al*. 2005) camera traps offer considerable advantages for surveillance in environments that are otherwise difficult to survey. Rowcliffe *et al*. (2008) developed a method for quantifying ungulate abundance using trail cameras deployed at sampling points throughout an area. Following this method, the area covered by the camera defines the area sampled. The use of several cameras increases the proportion of the area that was sampled. The method proved reliable for muntjac at densities ranging from 6.1 to 21.7 km^{-2}, and has been proposed more generally for use on forest ungulates (Rovero and Marshall 2009). This approach is likely to be appropriate for cost –effectively estimating muntjac abundance in the wild for low-high densities, but requires validation, particularly in low-density settings.

The most appropriate method for a given situation depends on the relative abundance of local populations, local environmental conditions, the level of estimate accuracy, precision and statistical power required and the intended final use of the estimates. For example, if data are required to inform whether a population is increasing, stable or decreasing, a simple index of abundance, repeated over time, may present the most cost-effective option. Alternatively, if an accurate and precise density or abundance estimate is required, for example because the population is thought to be approaching a density at which action should be triggered, then a more robust method, such as direct distance sampling, may be more appropriate than an indexing method.

During Defra project WM0318, Fera developed a staged-approach to the surveillance of wild boar populations. Under this approach, a series of three steps were designed, each being more technical and resource-intensive, hence expensive and capable of producing more robust estimates than the previous one. For wild boar presence, the first step involved collation of reports on wild boar presence from third parties to produce a distribution map against which new reports could be compared. Sightings from previously unoccupied areas triggered a move to the second step, which involved confirmation of wild boar presence using baited camera traps sited in woodlands. Confirmation of wild boar presence triggered the final step, where the local distribution and the rate of range expansion were measured using trail cameras at bait points throughout a number (>20) of nearby woodlands. For wild boar density, the first step was a survey for the abundance of boar signs (track ways, footprints, wallows) along transects. The second step was quantification of relative abundance (low, moderate, high density) from counts of boar, using trail cameras at bait points, or camera traps deployed throughout an area in a grid design. If relative abundance was considered likely to be high enough (*i.e*. to yield at least 50 observations from a given survey effort) then distance sampling line transect surveys using thermal imaging were undertaken.

Following these staged approaches, wild boar were reported, confirmed, and their populations quantified as they reached range and/or abundance thresholds beyond which more robust, detailed information was required. This efficient approach means that resources can be allocated to surveillance and control efforts based on information of sufficient quality for the current circumstances.

We propose that a similar strategy is developed for muntjac. For confirmation of presence/absence, the first step should be to regularly review reports from the public to the various information gathering initiatives (see below). In addition, it would be beneficial to forge links with NGOs such as the SSPCA, RSPCA, local wildlife hospitals and also game keepers on country estates particularly in areas close to the border with England, and areas predicted to be favourable for muntjac (Figure 3) in order for them to provide reports of muntjac shot, involved in traffic collisions or otherwise captured and treated for injuries. SNH already has a suitable protocol for responding to reports of muntjac (Deer Commission for Scotland 2005) and this provides a useful framework for capturing information for the first step of muntjac detection.

At least four information repositories currently exist that capture data that could be used as a primary indication of the likelihood of the presence of muntjac in Scotland. These are:

The Biological Records Centre (www.brc.ac.uk), which collects presence records of all wildlife species in Great Britain through a network of voluntary recorders. The BRC provides the opportunity to download distribution information via the National Biodiversity Network Gateway (www.searchnbn.net).

The British Deer Society (www.bds.org.uk/deer_distribution.html), who survey their members every five years for observations on deer presence throughout GB to produce distribution data at a resolution of 10km squares. The BDS provide distribution data on request.

The Deer Collisions Project (www.deercollisions.co.uk) which collects reports of collisions between deer and motor vehicles on Britain's roads.

Recording Invasive Species Counts (www.nonnativespecies.org/recording) which collects reports on sightings of all non-native species throughout Britain, but focuses on 10 species, including *Muntiacus reevesi*. Distribution data can be visualised as customisable maps, which can help to illustrate range expansion.

On receipt of a report of a muntjac sighting, the second step should be to confirm their presence. SNH's system for verifying muntjac reports (SNH 2005) could be used to fulfil this step. Further verification could take the form of passive detection, such as a rapid sign survey, involving, for example, searching for muntjac slots in mud, consumption of ivy stems planted into the ground during February to April (Cooke 2006) or photographs from baited trail cameras. However, these approaches have yet to be evaluated for low-density muntjac populations or those sympatric with other deer species. Nevertheless, muntjac can be captured by trail cameras; during

Defra-project WM0318 muntjac were regularly photographed by trail cameras set to monitor wild boar (Figure 5). Detection of muntjac via this approach should then lead to the third step, where the local distribution of muntjac is estimated. From experience with low-density wild boar populations, the deployment of digital trail cameras at bait points in at least 20 woodlands can yield sufficient data to estimate the probability of boar presence in each of the woods, even if they were detected imperfectly (that is, they were not detected in some woods despite being present) (Mackenzie *et al*. 2003). While this method can be effective, the greatest challenge is preventing trail cameras from being vandalised or stolen.

Figure 5: Muntjac captured by a digital trail camera (Pennswoods DS07) at a maize bait point in woodland near Ross-on-Wye.

It is unlikely that invading populations of muntjac will be either abundant or at high density. For low-density populations the effort required to detect sufficient animals or their signs to yield precise abundance estimates can be prohibitive. Consequently, the most appropriate approach to the first step for the quantification of muntjac abundance is likely to be a rapid indexing method. Such a method should yield sufficient information to judge whether muntjac are present at low, moderate or high densities, but without necessarily producing any absolute estimate or measure of precision. A rapid indexing method could take the form of trackway surveys, whereby the number of deer paths are counted crossing a woodland boundary (Mayle *et al*. 2000). However, since it would be necessary to distinguish those used by muntjac from those used by other species to avoid high-biased estimates, each path would need to be searched to identify slot marks to species. Alternatively, depending on the number of trail cameras used per woodland during presence/absence surveys, the number of photographs taken of muntjac is likely to index their abundance (Rovero and Marshall 2009).

Once the relative abundance of muntjac has been estimated, an appropriate method for the second step can be identified. For low density populations there is unlikely to be a method that is capable to deliver accurate, precise estimates given limited resources. However, for moderate to high density populations, camera traps deployed in a grid pattern or line transect distance sampling may offer appropriate approaches. Again, it is advisable to develop and validate these approaches on muntjac populations of moderate to high density.

While density and abundance information can usefully inform control plans, it is likely to be challenging to survey low-density muntjac populations sufficiently to yield accurate and precise density estimates in a timely manner, and it may be more cost - effective to invest the required resources in attempting to eradicate them, if that was the chosen policy option. In the absence of an estimate of density and absolute abundance it will be challenging to estimate the amount of effort required to eradicate the population, but this could be estimated after culling has commenced. For fixed-effort culling, and assuming a straight-line relationship between the density of muntjac and cull returns, the cull returns will represent a fixed proportion of the population per unit effort. Plotting the cumulative cull numbers against cumulative effort results in a sigmoidal curve (Figure 1). The point at which the curve starts to plateau is typically the point at which approximately half the required effort has been expended, and hence it is possible to estimate the total effort that is likely to be required.

4.7 Culling muntjac

It is generally accepted by stakeholders that muntjac should be controlled using the same approaches as for other deer species in Britain; that is, using appropriate, legal firearm and ammunition combinations (Smith-Jones 2004). However, there are reports of muntjac being moved to waiting shotguns on some English estates (Smith-Jones 2004), and some respected deer managers perceive this practice as acceptable (anon. pers. comm.), although a deer welfare charity does not (www.bds.org.uk/deer_and_shotguns.html accessed 21st January 2011). Moreover, even when using rifles, there seems to be an increasing move towards collaborative woodland deer management whereby groups of stalkers cover an area of ground at the same time, either by passively waiting for passing muntjac, or by having other people gently move (as opposed to drive) muntjac towards them (Smith-Jones 2004). It is recommended that this form of management is undertaken for brief periods at infrequent intervals to avoid muntjac from habituating to it (Smith-Jones 2004). However, during an eradication campaign, an approach such as this might need to be prolonged until it is considered highly likely that the last muntjac has been killed. Indeed, leaving a single muntjac risks the population re-establishing if the animal is female (a 50:50 chance, all else being equal), and since there is a high likelihood of females being pregnant at any time of the year. It seems likely that a co-ordinated, intensive approach using several stalkers at the same time over a patch of land will increase the frequency of muntjac observations per stalker, and hence the frequency of shooting opportunities. In fact, muntjac behaviour might facilitate the success of this practice since they typically occupy small, overlapping ranges and seem to be constantly moving (Chapman *et al*. 1993).

Smith-Jones (2004) supported the views of several people who supplied data during this study; muntjac stalking can be quite different to the traditional stalking of other British deer species. While traditional woodland stalking can be effective during the spring, the use of high seats, particularly at the junction of woodland rides, at woodland edges and over bait points (baited with rotting apples, carrots, whole strawberry plants) is considered effective. However, muntjac are very different to other British deer in their appearance and the way they use their environment (preferring dense cover). Consequently, experienced deer stalkers lacking experience of muntjac have failed to shoot muntjac when they have been present on their ground in reasonable abundance, possibly because the stalkers' search image and stalking tactics are inappropriate for this species (Smith-Jones 2004). Consequently, it would seem prudent for stalkers likely to be involved in any eradication campaign to receive training and experience in muntjac stalking in advance of a potential invasion of Scotland by muntjac.

An approach rarely considered in Britain for deer management is trapping. Such practices are conducted using Clover traps, long nets, drop nets and rocket nets for deer management in the USA, but their use can illicit vocal responses from those interested in animal welfare as much as those interested in hunting (pers. obs.). Nevertheless, it would seem prudent to retain as many appropriate tools as possible when considering an eradication campaign, and trapping might present an option for this. Indeed, while targeting wild boar and badgers during scientific research, muntjac have been caught in traps of a variety of specifications (pers. obs. and see Figure 6 for muntjac in a corral trap set for wild boar). Moreover, in comparison to shooting, trapping can be far more cost –effective. Ward *et al*. (2010b) estimated that feral pig control was ten times more costly when conducted by stalking than by trapping. It is also worth noting that feral pig eradication campaigns on islands and in enclosed areas have usually used a combination of trapping and shooting to ensure complete eradication (Barrett *et al*. 1983; Lombardo and Faulkner 2000; McCann and Garcelon 2008). Putman (1995) quoted high mortality rates for trapped deer, but Chapman *et al*. (1987) provided guidance to reduce welfare impacts of trapping on muntjac. Such guidelines should be considered if trapping is to form any part of a muntjac eradication campaign.

Figure 6: Two muntjac feeding on maize bait in a corral trap set to capture wild boar near Ross-on-Wye. Image taken by a Reconyx RC60 digital trail camera.

5 Conclusions

Reports of muntjac in Scotland are occasionally received, and the English population is spreading northwards. It seems likely that they may spread into Scotland eventually, although the rate of spread may be much lower than that experienced in England. Escapes from collections and deliberate releases of translocated muntjac also risk establishing populations in Scotland. It seems likely that *M. reevesi* is the only species of muntjac that poses any real threat to Scotland. Most of Scotland does not appear to offer suitable habitat for thriving muntjac populations, but some locations do. It would seem prudent to establish routine surveillance at such locations close to the border with England as English muntjac populations approach the border.

There is a well-established culture of deer control in Scotland, and muntjac could become integrated with this. However, this risks perpetuating their presence as they become embedded in local custom. The costs of perpetual control of an established muntjac population are likely to be substantially higher in any one year than the eradication of a population of fewer than 200 animals, but the benefits, in terms of venison sales are likely to be much lower than for other deer species. This assessment of costs does not include the cost of impacts on the natural heritage or economic activities such as game shooting enterprises and forestry. Consequently, even on the basis of likely costs of control alone, prevention of establishment by eradication of invading populations seems to present the most cost –effective of the two options.

If a policy of eradication is to be pursued it will need to be under-pinned by quality information to guide the most appropriate course of action and the requisite investment. We have proposed a staged approach to surveillance to fulfil this requirement when information on the muntjac population is lacking. However, such a system will need to be properly developed and evaluated before its reliability can be judged. It should also be born in mind that when resources are limited, they may be better spent on eradication efforts than on surveillance.

Muntjac control would present novel challenges to Scottish stalkers, who may benefit from training and experience of muntjac surveillance and control in advance of any outbreak in Scotland. Moreover, we recommend retaining options to use the widest variety of tools possible to assist with muntjac control, and these are likely to include trapping. Eradication efforts should have a rapid inception and delivery in order to avoid pregnant females producing offspring and growing the population. A co-ordinated, intensive approach may therefore be more appropriate than traditional lone stalking.

Any eradication campaign is likely to stand a greater chance of success if muntjac are detected soon after their emergence into the Scottish countryside. We have presented the case for a system for their detection, and SNH have started to consider how to respond well in advance of any future outbreak. This approach provides Scotland with the best chance of remaining muntjac-free.

6 References

Acevedo, P., Ward, A. I., Real, R. & Smith, G. C. (2010). Assessing biogeographical relationships of ecologically related species using favourability functions: a case study on British deer. *Diversity and Distributions*, 16, 515-528.

Barrett, R. H., Goatcher, B. L., Gogan, P. J. & Fitzhugh E. L. (1988). Removing feral pigs from Annadel State Park. *Transactions of the Western Section of the Wildlife Society,* 24, 47–52.

Buckland, S. T., Anderson, D. R., Burnham, K. P., Laake, J. L., Borchers, D. L. & Thomas, L. (2001). Introduction to distance sampling: Estimating abundance of biological populations. Oxford University Press, UK.

Buckland, S. T., Anderson, D. R., Burnham, K. P., Laake, J. L., Borchers, D. L. & Thomas, L. (2004) *Advanced distance sampling: Estimating abundance of biological populations*. Oxford University Press, Oxford.

Chapman, N. G. (2004) Faecal pellets of Reeves' muntjac, Muntiacus reevesi: defecation rate, decomposition period, size and weight. *European Journal of Wildlife Research,* 50, 141-145.

Chapman N.G., Claydon K., Claydon M., Forde P. G., & Harris S. (1993) Sympatric populations of muntjac (*Muntiacus reevesi*) and roe deer (*Capreolus capreolus*) – a comparative analysis of their ranging behaviour, social-organization and activity. *Journal of Zoology*, 229, 623-640.

Chapman N.G., Claydon K., Claydon M.,& Harris S. (1987) Techniques for the safe and humane capture of free-living muntjac deer (*Muntiacus reevesi*). *British Veterinary Journal*, 143, 35-43.

Chapman, N. & Harris, S. (1996) *Muntjac*. The Mammal Society, London, and the British Deer Society, Fordingbridge.

Chapman, N., Harris, S. & Stanford, A. (1994) Reeves' muntjac *Muntiacus reevesi* in Britain: their history, spread, habitat selection, and the role of human intervention in accelerating their dispersal. *Mammal Review*, 24, 113–160.

Cooke, A. S., (2006). Monitoring muntjac deer Muntiacus reevesi and their impacts in Monks Wood National Nature Reserve. *English Nature Research Report No. 681.* English Nature, Peterborough.

Cooke , A. S., (2004). Muntjac and conservation woodlands. In *Managing Woodlands and their Mammals,* (eds C. Quine, R. Shore & R. Trout), pp. 65-69. Forestry Commission, Edinburgh.

Cooke A. S., (1994). Colonisation by Muntjac deer *Muntiacus reevesi* and their impact on vegetation. In Monks Wood national Nature Reserve, the experience of 40 years 1953-93, (eds M. S. Massey & R. C. Welsh), pp. 45-61. English Nature, Peterborough.

Daniels, M. J (2006).Estimating red deer *Cervus elaphus* populations: an analysis of variation and cost-effectiveness of counting methods. *Mammal Review,* 36, 235-247.

Deer Commission for Scotland (2005). *DCS staff guidance. Recording and investigating reports of free living muntjac in Scotland*. Deer Commission for Scotland, Inverness.

Deer Commission for Scotland (2010) Annual Report 09-10. Deer Commission for Scotland, Inverness.

Dolman, P.M. & Wäber, K. (2008). Ecosystem and competition impacts of introduced deer. *Wildlife Research*, 35, 202-214.

Elton, C.S. (1958). *The ecology of invasions by animals and plants*. Methuen and Co, London.

Focardi S., Isotti R., Raganella, Pelliccioni E. Iannuzzo D. (2002). The use of distance sampling and mark-resight to estimate the local density of wildlife populations. *Environmetrics*, 13, 177-186.

Gill, R. M. A., Thomas, M. L. & Stocker, D. (1997). The Use of Portable Thermal Imaging for Estimating Deer Population Density in Forest Habitats. *Journal of Applied Ecology,* 34, 1273-1286.

Gosling, L. M. & Baker, S. J. (1989). The eradications of muskrat and coypus from Britain. *Biological Journal of the Linaean Society* 38, 39-51.

Harding, S. P. (1986). Aspects of the ecology and social organisation of the muntjac deer (*Muntiacus reevesi*). *DPhil Thesis,* University of Oxford, Oxford, UK.

Hebeisen C., Fattebert J., Baubet E. & Fischer C. (2007). Estimating wild boar (*Sus scrofa*) abundance and density using capture–resights in Canton of Geneva, Switzerland. *European Journal of Wildlife Research*, 54, 391-40.

Hemami, R. H., Watkinson A. W., Gill, R. M. A. & Dolman, P.M. (2007) Estimating abundance of introduced Chinese muntjac *Muntiacus reevesi* and native roe deer *Capreolus capreolus* using portable thermal imaging equipment. *Mammal Rev*iew, 37, 246–254.

Lockwood, J., Hoopes, M. & Marchetti, M. (2007). *Invasion ecology*. Blackwell Publishing, Oxford, UK.

Lombardo, C. A., & Faulkner. K. R. (2000). Eradication of feral pigs, (*Sus scrofa*) from Santa Rosa Island, Channel Islands National Park, California. Pp 300–306 in D. R. Browne, K. L. Mitchell & H. W. Chaney (Eds). *Proceedings of 5th California islands symposium*. U.S. Department of the Interior, Mineral Management Services Pacific Outer Continental Shelf Region, study MSS99–0038, Washington, D.C., USA.

MacKenzie, D. I., Nichols, J. D., Hines, J. E., Knutson, M.G. & Franklin A.B. (2003). Estimating site occupancy, colonization, and local extinction when a species is detected imperfectly. *Ecology,* 84, 2200-2207.
Mayle B. A. Peace, A. J. & Gill. R. M. A. (2008. *How Many Deer? A Field Guide to Estimating Deer Population Size*. Forestry Commission Fieldbook 18. Forestry Commission, Edinburgh.

Mayle, B. A., Putman, R. J. & Wyllie, I. (2000). The use of trackway counts to establish an index of deer presence. *Mammal Review*, 30, 233-237.

Mayle, B. A. & Staines, B. W. (1998). An overview of methods used for estimating the size of deer populations in Great Britain. Pp 19-31 in *Population Ecology, Management and Welfare of Deer* (C. R. Goldspink, S. King and R. J. Putman, eds). Manchester Metropolitan University, UK.

McCann, B. E. & Garcelon, D. K. (2008) Eradication of feral pigs from Pinnacles National Monument. *Journal of Wildlife Management,* 72, 1287-1295.

McCarthy, A. J. & Rotherham, I. D. (1994). Deer in the Sheffield region including the Eastern Peak District. *The Naturalist*, 119, 103-110.

Mooney H. A., Mack R. N., McNeely J. A., Neville L. E., Schei P. J., & Waage J.K. (Eds) (2005) *Invasive alien species: A new synthesis*. Island Press, Covelo, California.

Pollard, E. & Cooke, A.S. (1994). Impact of muntjac deer *Muntiacus reevesi* on egg laying sites of the white admiral butterfly *Ladoga camilla* in a Cambridgeshire wood. *Biological Conservation*, 70, 189-191.

Putman, R. J. (1995). Ethical considerations and animal welfare in ecological field studies. *Biodiversity and Conservation,* 4, 903-915.

Putman, R. J., Langbein, J., Green, P. & Watson, P. (*in press*). Identifying threshold densities for wild deer in the UK above which negative impacts may occur. *Mammal Review*.

Rovero, F., & Marshall, A. R. (2009). Camera trapping photographic rate as an index of density in forest ungulates. *Journal of Applied Ecology,* 46, 1011-1017.

Rowcliffe, J. M., Field, J., Turvey, S. T. & Carbone, C. (2008). Estimating animal density using camera traps without the need for individual recognition. *Journal of Applied Ecology*, 45, 1228-1236.

Simberloff, D. (2003a). Eradication – preventing invasions at the outset. *Weed Science*, 51, 247-253.

Simberloff, D. (2003b). How much information on population biology is needed to manage introduced species? Conservation Biology, 17, 83-92.

Smart, J. C. R., Ward, A. I. & White, P. C. L. (2004) Monitoring woodland deer populations in the UK: an imprecise science. *Mammal Review*, 34, 99-114.

Smith-Jones, C. (2004). *Muntjac – managing an alien species*. Coch-y-Bonddu Books, Machynlleth, Powys.

Staines, B. W. & Ratcliffe, P. R. (1987). Estimating the abundance of red deer (*Cervus elaphus* L.) and roe deer (*Capreolus capreolus* L.) and their current status in Great Britain. *Symposia of the Zoological Society of London,* 58, 131-152.

Ward, A. I. (2005). Expanding ranges of wild and feral deer in Great Britain. *Mammal Review*, 35, 165-173.

Ward A. I. & Etherington T. R. (2010). Appendix Three: Mapping local and national abundance of wild deer in England and Wales. In *Predicting abundance and distribution of wild deer in England: supporting a vision for Natural England* (R. Putman and A. Ward). Natural England, Peterborough.

Ward, A., Etherington, T. & Ewald J. (2008). Five years of change. *Deer*, 14, 17-20.

Ward, A., Massei, G. & Cowan, D. (2010b). Feasibility study of the effectiveness of culling feral pigs during a CSF outbreak. Report to Defra, London.

Ward, A., Pietravalle, S., Acevedo, P. & Etherington, T. (2010a). Appendix 5. Predictors of range expansion of wild deer in Great Britain. In *Predicting abundance and distribution of wild deer in England: supporting a vision for Natural England* (R. Putman and A. Ward). Natural England, Peterborough.

Williamson, M. (1996). *Biological invasions*. Chapman & Hall, London

www.snh.gov.uk

ISBN: 978-1-85397-765-7

Policy and Advice Directorate, Great Glen House,
Leachkin Road, Inverness IV3 8NW
T: 01463 725000

You can download a copy of this publication from the SNH website.

recycle

Scottish Natural Heritage
Dualchas Nàdair na h-Alba
All of nature for all of Scotland
Nàdar air fad airson Alba air fad