

Control of an invasive species: the American mink in Great Britain

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American mink © A.L. Harrington

The American mink as an invasive species

American mink (*Neovison vison*) are semi-aquatic mustelids, native to North America. They are bigger than a stoat but smaller than an otter and have a luxuriant, thick, soft coat (Dunstone 1993). Demand for this highly-prized fur led to the development of mink fur farming and consequently American mink were farmed in many parts of Europe, Asia, and South America, as well as their native North America. The European mink (*Mustela lutreola*), an ecologically similar species (belonging to a different genus, Macdonald et al., Chapter 1, this volume) of similar size to the American mink, but with inferior fur, was never successfully farmed, either in Eurasia (where it is a native species) or elsewhere, although historically it was hunted for its pelt across much of its Eurasian range as the American mink is across its native range. In Europe, fur farming

dates back to the early 1900s but became increasingly popular from the 1960s onwards. Many countries in Europe, including, amongst others, Denmark, Finland, Norway, Sweden, Spain, France, Italy, and the Netherlands, (at the time of writing in 2017) still practise mink farming. As of 2015, there were approximately 5000 fur farms in Europe producing over 40 million mink pelts at a value of around 1.6 billion euros (www.fureurope.eu). Denmark is the largest producer of mink pelts in Europe, followed by Poland and the Netherlands (Fur Europe 2015, www.fureurope.eu). Fur farming is banned in Austria and the UK, and in a number of eastern European countries, and is being phased out (by 2024) in the Netherlands (www.furalliance.com). The last 13 mink farms in England and Wales (producing up to 100,000 mink skins each year) were closed down in the early 2000s prior to the Fur Farming (Prohibition) Act 2000 that came into effect in 2003. There were

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no fur farms operating in Scotland at the time (the last had been closed in 1993) but future fur farming was nevertheless prohibited in Scotland by the Fur Farming (Prohibition) (Scotland) Act 2002. Most countries with a history of fur farming now have feral populations of American mink, established from escaped or intentionally released mink, which are classed as invasive species (Macdonald and Harrington 2003). The European mink, in contrast, is now absent or rare across much of its former range (Maran et al., Chapter 17, this volume) and was never present in the UK (Macdonald et al., Chapter 1, this volume). Despite the presence of fur farms, American mink have not established in the Netherlands (Bonesi and Palazón 2007; Dekker and Hofmeester 2014). This might be because intensive trapping for coypus and muskrats incidentally controls feral American mink (Dekker and Hofmeester 2014; there are no European mink in the Netherlands). In this chapter, we refer exclusively to the American mink; interactions between the two mink species (where they now overlap) are discussed further in Maran et al. (Chapter 17, this volume).

In their introduced range, American mink (hereafter mink) inhabit riparian strips of both coastal and freshwater systems, preferring areas of sheltered rocky shore in coastal habitats (Bonesi et al. 2000) and areas of tree, scrub and grass cover in riverine areas (Previtali et al. 1998; Yamaguchi et al. 2003a; Melero et al. 2008; Macpherson and Bright 2010). Their biology in lowland England is summarized by Macdonald et al. (2015a), where mink tend to occupy linear home ranges and are rarely found further than 100 m from the water's edge (Dunstone 1993). They do not excavate their own den sites but instead occupy rabbit holes, crevices, exposed tree root systems, rock piles, or above ground vegetation, and utilize multiple sites within their territory (Dunstone 1993; Dunstone and Birks 1987). Mink are highly mobile (Gerrell 1970) and can swim and dive in both salt and freshwater (Dunstone 1993; Macdonald et al. 2015a). They are opportunistic, generalist predators, feeding on aquatic and terrestrial prey including small and medium-sized mammals, fish, crustaceans, birds, eggs, amphibians, and insects (Dunstone 1993). Predation by mink has led to the widespread demise of many native species within the mink's invaded range (Macdonald and Harrington 2003) and this is a major driver in promoting mink control. Studies in Britain, Finland, and Patagonia, for example, have demonstrated declines in native bird populations as a result of direct killing, desertion of breeding colonies, or breeding failures due to disturbance during incubation

(Craik 1997; Nordström et al. 2003; Peris et al. 2009). On the Outer Hebridean islands of Scotland colonized by mink, tern colonies are larger (a possible anti-predator response), and produce fewer chicks than on islands not colonized by mink (Clode and Macdonald 2002). On the mainland, on the west coast of Scotland, the number of breeding pairs of common terns (*Sterna hirundo*) decreased by 36% between 1987 and 1996, and for black-headed gulls (*Larus ridibundus*) and common gulls (*L. canus*) the number of breeding pairs decreased by 52% and 30% respectively between 1989 and 1996 (during a period when both the distribution and numbers of mink were believed to have increased, Craik 1995, 1997). The loss of approximately 88% of water vole populations (*Arvicola amphibius*) in Britain during the 1990s has also been largely attributed to predation by mink (Jefferies 2003).

Impact mitigation and legislative background

The impact of mink can be mitigated through removal and control, but only if performed on a suitably large scale so as to overcome compensation through dispersal from uncontrolled areas. A large-scale experimental removal of mink from an area in the Finnish Archipelago (>120 small islands covering an area of approximately 200 km²) over a 15-year period resulted in marked increases in the breeding densities of ringed plover (*Charadrius hiaticula*), arctic skua (*Stercorarius parasiticus*), arctic tern (*Sterna paradisaea*), and rock pipit (*Anthus petrosus*), and the re-establishment of breeding sites for razorbills (*Alca torda*) and black guillemots (*Cepphus grylle*) (Banks et al. 2008; Nordström et al. 2003). And, at a local scale, Moorhouse et al. (2009; see also Moorhouse et al. 2015a, b) demonstrated that, over the short term, water vole populations could be reintroduced successfully on short (20 km) stretches of river where mink were subject to continual monitoring and reactive trapping (the idea being that localized control could then be expanded over a larger area to allow long-term persistence of re-established water voles). Several European countries have attempted to control and/or eradicate mink, but the overall efficacy of control efforts has varied (Bonesi and Palazón 2007). Working mink farms in many countries pose a continual risk of reinvasion (Pertoldi et al. 2013). In the UK, there are no longer any fur farms so there is very limited risk of reinvasion, but no area subject to control is completely safe from recolonization.

The UK is one of many European signatories of the Convention on Biological Diversity (CBD), which

came into force in 1993. The CBD requires signatories to 'control or eradicate those alien species which threaten ecosystems, habitats, or species'. Despite taking over a decade to formulate a formal, co-ordinated approach to tackling the problem of invasive species, Britain was one of the first in Europe to develop a national policy framework with the launch of the Invasive Non-native Species Strategy for Great Britain in 2008 (www.nonnativespecies.org). Although the framework highlighted the 'prevention' and 'early detection' of invasive species as the highest priorities for action, it also specified that, for those species already established, priorities for control should be identified, and the feasibility of eradication considered. A key action of the framework was to 'develop and resource key GB level action programmes that are cost-effective and evidence-based'.

In this chapter, we provide an overview of a research project focussing on the control of American mink in the west and north of Scotland, UK. The aims of the research were primarily to obtain additional, and specific, information about the ecology of the American mink in Scotland, with a view to strengthening and expanding existing management plans, but also to explore methods by which such management plans could be implemented. We addressed the latter by considering the possibility of using volunteers to help with the manual work required, and specifically the possibility of recruiting ecotourism boat operators. At the end of the chapter, we summarize our case study, and discuss whether lessons learnt from Scotland may be applicable to mink control elsewhere in their invaded range, suggest relevant questions that currently remain unanswered, and outline very briefly some of the wider issues associated with the control of any invasive species. First, we briefly discuss the methods and strategies used to control American mink, giving a short summary of some of the earlier work that has taken place in Great Britain, and specifically in Scotland. In so doing, we highlight some of the issues associated with invasive species control in mainland, as opposed to island, systems.

Trapping methods and strategies

Island eradication

Eradication of invasive species from islands is becoming increasingly successful (e.g. feral goats from Santiago Island, Cruz et al. 2009, black rats from Anacapa Island, Howald et al. 2010) and is achievable and

sustainable due to the fact that islands are isolated, and their boundaries are defined and defensible, hence reducing the risk of recolonization.

The Outer Hebrides is an extensive (approx. 2800 km²) network of isolated islands in the North Atlantic Ocean, c.50 km west of mainland Scotland. They are home to a number of internationally important ground-nesting and migratory bird species, including waders, divers, and seabirds that have suffered significant disturbance and losses in the presence of mink (Scottish Natural Heritage 2012) that either escaped or were released from a mink farm on one of the islands. In 2007, as part of the Hebridean Mink Project, a team co-ordinated by Scottish Natural Heritage (www.snh.gov.uk) embarked on an eradication programme across the largest islands, Lewis and Harris (approx. 2200 km²), following on from previous trapping efforts on the southerly islands of North Uist, Benbecula, and South Harris (between 2001 and 2006, Roy et al. 2015). A dense network of over 8000 traps was deployed across Lewis and Harris by individually digging each trap into the ground. Traps were systematically operated by up to 14 full-time professional trappers for seven years. Traps were set in all available habitats, regardless of distance from roads, and encompassed small offshore islands, rocky coastline, bog moorland, machair (dune pastures), and agricultural land. Most traps had to be accessed on foot but the most remote locations were accessed by helicopter and offshore islands were visited by boat. Several trappers had dogs trained to locate the scent of mink, which was particularly useful in locating dens during the breeding season. Once a den was found traps were deployed around the den to catch the breeding female mink when she exited to bring food back for her kits. Between 2007 and 2014, more than 1500 mink were captured, and it was believed that only a few individuals remained to be captured (Lambin et al. 2014).

Mainland control

Successful eradication of invasive species in mainland systems is rare, generally because the scale of a project is too large and there is continued risk of reinvasion (successful examples include Canadian beavers in France, Genovesi 2005, and coypu in UK, Gosling and Baker 1989). Tackling a mink eradication project on the British mainland, where mink are almost ubiquitous, using similar strategies to those employed in the Outer Hebrides has been considered to be prohibitively expensive (Macdonald and Strachan 1999). Further, the certainty

of reinvasion from surrounding areas means that control to near zero abundance or reduction of the impact on native fauna to sustainable levels, rather than total eradication, is a more realistic option in mainland areas. However, since even these more 'realistic' projects can incur significant initial and ongoing cost, it is important to consider where mink control is likely to be feasible and yield benefits for native species populations.

Mink were first introduced to Britain in 1929 and were recorded in the wild almost immediately, but breeding in the wild was not documented until 1956 (Usher 1987). A control programme in 1962 failed to prevent the spread of mink (perhaps due to insufficient resourcing and scale considering the dispersal abilities of mink), as did various follow-up trapping events (Usher 1987). Hunting estates in Scotland have frequently logged mink in game bag records since 1964 but localized trapping effort did not result in an overall reduction in mink populations (Bryce et al. 2011). The traditional method of trapping mink (whether the intention is to kill them or not) using cage traps set on riverbanks is time consuming and requires considerable field skill as well as large numbers of traps (one every 300 m) that require checking every 24 hours (for animal welfare reasons, Yamaguchi et al. 2002). Fortunately, the development of the simple but ingenious idea of using a combination of tracking plates and traps on floating rafts, developed by the Game and Wildlife Conservation Trust (Reynolds et al. 2004; www.gwct.org.uk), has revolutionized large-scale mink trapping. This system provides an effective trapping platform (that requires little skill to set, and few rafts—one every 1 or 2 km in Britain), appears to be especially effective at catching females (that are often under-represented in traditional bank-side trapping programmes), and is particularly effective at low densities (Harrington et al. 2008b). More importantly, by enabling detection of mink prior to trapping, the raft system allows trapping to be targeted such that traps are only set on rafts on which mink presence is confirmed. Floating rafts consist of a tunnel (on a raft) protecting a clay pad that records footprints of any animal passing through the tunnel (Figure 16.1a, b). Mink are naturally curious and so invariably investigate rafts in rivers when travelling along the riverbank. Traps can then be set on the rafts (or under similarly designed tunnels on the riverbank, Figure 16.1c) in response to the presence of footprints. The method is considerably more efficient than traditional bankside traps because tracking plates can be checked once a week (rather than once a day) and the number of traps set is minimal (i.e.

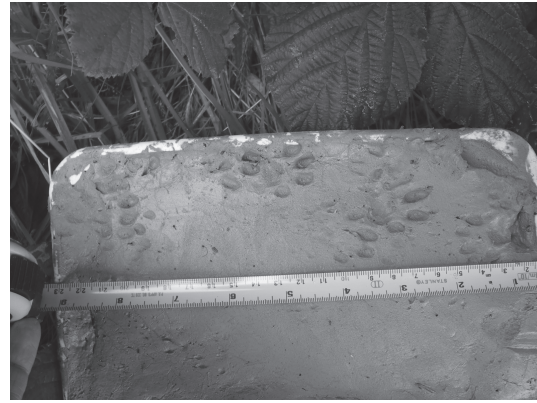
traps are only set on those rafts on which footprints were detected). The raft tunnels can potentially be constructed to accommodate any type of trap. However, the kill traps and poison baits that are used to control invasive mustelids in countries such as New Zealand are not appropriate on the British mainland where native mustelids are also present, and may be illegal in areas where polecats are known to be present (J. Birks, pers. comm.). Lethal spring traps are used in rafts in the UK on offshore islands where native mustelids are either absent or are larger than mink and can thus be safely excluded (for example, by installing bars that reduce the size of the tunnel entrance—'otter excluders', Short and Reynolds 2001). On the mainland, live cage traps on rafts provide a simple and reliable alternative that can be used safely in the presence of non-target species. Mink captured in cage traps (Figure 16.2) are humanely dispatched using a 0.22 mm calibre air pistol or air rifle, and a shot through the front of the skull (GWCT Mink Raft Guidelines, www.gwct.org.uk). Another tool which is proving useful, especially at low mink density, is an alarm system that alerts the trapper when the trap has been sprung (www.minkpolice.com). This saves the trapper from having to check the trap daily but allows for non-target animals to be released and target animals to be humanely despatched by the trapper as quickly as possible after being trapped.

Mink rafts have proven highly successful in aiding mink control throughout Britain, and have enabled what is probably the 'largest mainland invasive species eradication effort worldwide' (Bryce et al. 2011). Because checking rafts and setting traps on rafts does not require any specific skill or training (beyond that that can be provided in an afternoon), their use allows, and is ideal for, the employment of large numbers of volunteers to assist with labour, which can be especially important over large areas with low mink densities. The project was started by staff from the University of Aberdeen in response to major losses of water vole populations in the Cairngorms National Park in north-east Scotland. The initial focal area was thought to be relatively marginal for mink and because of this, acted as a low productivity refuge for an extensive network of water vole colonies linked by dispersal, with the consequence that mink-induced declines (seen elsewhere in Britain) had been delayed (Aars et al. 2001; Strachan and Jefferies 1993). Mink control began in the headwaters of river catchments and gradually progressed downstream, leaving a refuge for water voles in its wake (Bryce et al. 2011). The project expanded out of

(a)



(b)



(c)



Figure 16.1 Images illustrating the floating raft system for monitoring and trapping mink, showing a) a raft on a river in Scotland, b) mink footprints on the clay pad, and c) a live cage trap set within a tunnel on the riverbank. In Scotland, where rivers are fast flowing and subject to spight, bankside tunnels were sometimes used instead of floating rafts; these were used in the same way as floating rafts using a clay pad to detect the presence of mink before setting traps. © E.J. Fraser



Figure 16.2 An American mink in a live cage trap. © E.J. Fraser

the national park within a year and continued to deploy rafts throughout river catchments in northeast Scotland, recruiting volunteers to help monitor the rafts on a fortnightly basis. Mink monitoring and trapping was initially co-ordinated by scientists at the University of Aberdeen but primarily implemented by volunteers (gamekeepers, fisheries staff, wildlife conservation professionals, land managers, and local residents) who had convergent interests in the biodiversity of the local area (Bryce et al. 2011). Although local mink trapping by gamekeepers had occurred on an annual basis prior to the beginning of the project, control only became regionally effective once the volunteers were co-ordinated at a landscape level (Bryce et al. 2011). By combining novel community participation and adaptive

management approaches (continual spatial expansion with retention of monitoring behind the control front to detect immigrants) the project successfully eradicated breeding mink from an area of 10,570 km² in northeast Scotland between 2006 and 2009 (Bryce et al. 2011). Over the three years of the project, 376 mink were captured (47% of which were females) with the help of 186 volunteers. Since 2006, the project has expanded across north and east Scotland and is now managed by local non-governmental organizations with added input from scientists and civil servants. The project area is approximately 29,000 km² (an area the size of Wales, and ten times the size of the Outer Hebridean archipelago), and aims to remove breeding mink from across north and east Scotland. An area is declared 'mink-free' when no female mink have been captured for at least two consecutive quarterly trapping sessions, but monitoring has to continue indefinitely.

A number of different studies, using mink rafts, have shown that mink control can be effective (subject to the definition of 'effective', which we discuss later in this chapter) at the local (20 km) (Harrington et al. 2009a), river catchment (J. C. Reynolds et al. 2013), and regional (Bryce et al. 2011) scale, and thus demonstrate the potential for effective control on a much larger, possibly even national, scale through the systematic linking of adjacent control regions. This would be a step-by-step process and would intuitively benefit from the creation of new control regions that neighbour existing ones. One region that provides obvious progression from an existing control area is the west of Scotland. This area was until recently omitted from the large scale mink project covering north and east Scotland, yet is occupied by mink and suffers serious annual losses of native wildlife, particularly ground-nesting birds (e.g. see Craik 1997) due to mink predation. Expansion of the project to the west of Scotland, a predominantly coastal habitat (Figure 16.3), had been hampered by a perceived lack of knowledge of mink biology in this kind of landscape. There have been several attempts to control mink in mainland west Scotland with the aim of protecting seabirds (e.g. Craik 1997) but the success of these has been short lived, primarily due to their localized focus.

In the following section of this chapter, we focus on our case study in the west and north of Scotland. We consider first how mink have spread across Scotland, and, specifically, to what extent habitat suitability and food availability has influenced the current distribution of mink. We then consider how we might be



Figure 16.3 Coastal habitat in northwest Scotland. © E.J. Fraser

able to use knowledge of population structure across the landscape to target control efforts. Finally, (before moving on to the lessons we have learnt) we discuss the potential use of volunteers in this type of work.

The case study

The distribution of mink in Scotland

The spread of mink through Scotland has been tracked alongside the changing distribution and abundance of two native species of conservation concern, the water vole and European otter (*Lutra lutra*), that share the riparian habitat of mink (Green and Green 1987; Strachan and Jefferies 1993; Green and Green 1997; Jefferies 2003; Strachan and Jefferies 2006). Mink are now found throughout Scotland except, perhaps surprisingly, in the far north (Harrington et al. 2010). Is it just a matter of time before this area is colonized? Or, is northern Scotland simply not suitable for mink? We considered the latter unlikely given that mink have successfully colonized the harsh environments of Iceland and Norway (Bonesi and Palazón 2007) and indeed are native to the even harsher climes of Alaska and much of Canada (Reid et al. 2016). We posed the following questions: is it possible to prevent range expansion of mink into the far north of Scotland through increasing understanding of the distribution, movements, and colonization history of the current mink population? Could this increased understanding also help guide mink management across the rest of Scotland, particularly west Scotland?

To gain a better understanding of the historical and potential future spread of mink through Scotland, the

pattern and rate of range expansion was initially investigated and related to habitat suitability across the landscape. Using records of mink presence, from 1960 to 2012, collated from a variety of sources including national open access databases and riparian mammal surveys, the area of mink distribution over time was calculated using home range analysis techniques (Fraser et al. 2015b). Rates of expansion were calculated from the changes in occupied area and revealed that the pattern and rate of expansion were not constant over time. The spread of mink occurred in three phases: a fast increase in occupied area from 1960 to 1978, a slow, almost stalled advance, from 1979 to 1989, followed by a steady increase in occupied area from 1990 to 2012. The rate of expansion by area ranged from 101 km² to 2866 km² per year, with a mean of 1327 km² per year. Radial expansion rates ranged from 1 km to 23 km per year (mean 10 km per year) and linear expansion rates ranged from 8 km to 27 km per year (mean 14 km per year) (Fraser et al. 2015b). The range of mink in Scotland appears to still be spreading but expansion is primarily restricted to coastal areas of northwest Scotland. The initial spread of mink is likely to have been influenced by the distribution and density of fur farms, but feral populations did not establish in the vicinity of all farms (Fraser et al. 2015b). Presumably the surrounding habitat influenced whether feral populations of mink were able to establish.

To assess the extent to which habitat had influenced the spread and settlement of mink, a habitat suitability model for Scotland was created using the 'maxlike' method developed by Royle et al. (2012). Habitat suitability (assessed according to location of individual mink records from 1960 to 2012) was positively related to areas of improved grassland and indented coastline, and negatively associated with areas of rough grassland, bog, and high elevation. Expansion of mink range had been fastest through areas of highly suitable habitat but variation in the rates of range expansion could not be wholly attributed to the availability of suitable habitat. Good habitat, such as low lying, improved pastures, encouraged high rates of expansion, but seemingly unsuitable habitat, such as mountainous areas, bog, or rough grassland, did not always slow it. In northwest Scotland, the habitat suitability model indicated that preferred habitat was restricted to the coast.

A study of mink diet, based on stable isotope analyses of liver tissue, reinforced the hypothesis that suitable habitat in northwest Scotland was restricted to the coastline (Fraser 2013) and highlighted the importance of food availability. The proportion

of marine protein, such as crustaceans and fish, in the diet varied geographically and was higher in coastal areas where the availability of surrounding suitable terrestrial habitat was low. This was primarily in northern locations. Specifically, the area of suitable habitat within a 10 km radius of mink captured in coastal locations showed a strongly negative relationship with increasing latitude, suggesting that coastal mink in northern areas of west Scotland were more reliant on marine protein than coastal mink in southern areas of west Scotland. In southwestern areas, the habitat inland from the coast is generally low lying, with improved grassland and larger river catchments. This is in direct contrast to inland habitat in more northerly areas that is primarily rough grassland or is mountainous, and where rivers are narrow and fast flowing. The wider range of terrestrial (e.g. small mammals) and aquatic prey (e.g. fish) that is available in southern areas presumably accommodates a greater range of individual mink diets than in northwest Scotland, where the diet of coastal mink is limited due to the restricted distribution of suitable habitat. It is possible that the diversity of prey available in coastal habitats alone is sustaining mink populations and facilitating colonization of northwest Scotland (Fraser 2013).

Defining population structure to improve targeted mink control

Habitat suitability, linked to food availability, has clearly shaped and defined the current distribution of mink in Scotland, but, beyond the identification of the importance of coastal habitats, how can this information help us to control this invasive species? Two questions were particularly relevant:

- 1) Is there any structure within the mink population that could be used to help target control efforts more effectively?
- 2) With a view to preventing further northwards spread (into predominantly mink-free areas) is it possible to locate the origins of the mink that were recorded most recently in northwest Scotland?

Our aim was to identify any intrinsic patterns of population structure, and to assess how the landscape might affect gene flow within the wider population (Fraser et al. 2013). To resolve genetic structure, delimit sub-populations, and establish patterns of connectivity across areas, over 500 mink were genotyped at 12 microsatellite loci (Fraser et al. 2013).

Our results revealed that mink in Scotland belonged to two genetic clusters—one based primarily in west Scotland and the other in east Scotland. The clusters were determined by looking at differences in the frequency of alleles at microsatellite markers. These are non-coding regions of DNA that can be used as genetic markers but are not genes themselves and therefore do not contribute to local adaptation. Mink populations positioned in central areas demonstrated admixture between the clusters.

Genetic divergence (the accumulation of independent genetic changes in a population) and diversity between the eight mainland sub-populations indicated genetic structure between sub-populations of mink in Scotland. Landscape features were found to influence gene flow among them. Significant genetic divergence between mink in the far northwest (NW1, Figure 16.4) and northeast (NE1, Figure 16.4) of Scotland was attributed to the presence of a mountain range acting as a barrier to dispersal. Conversely,

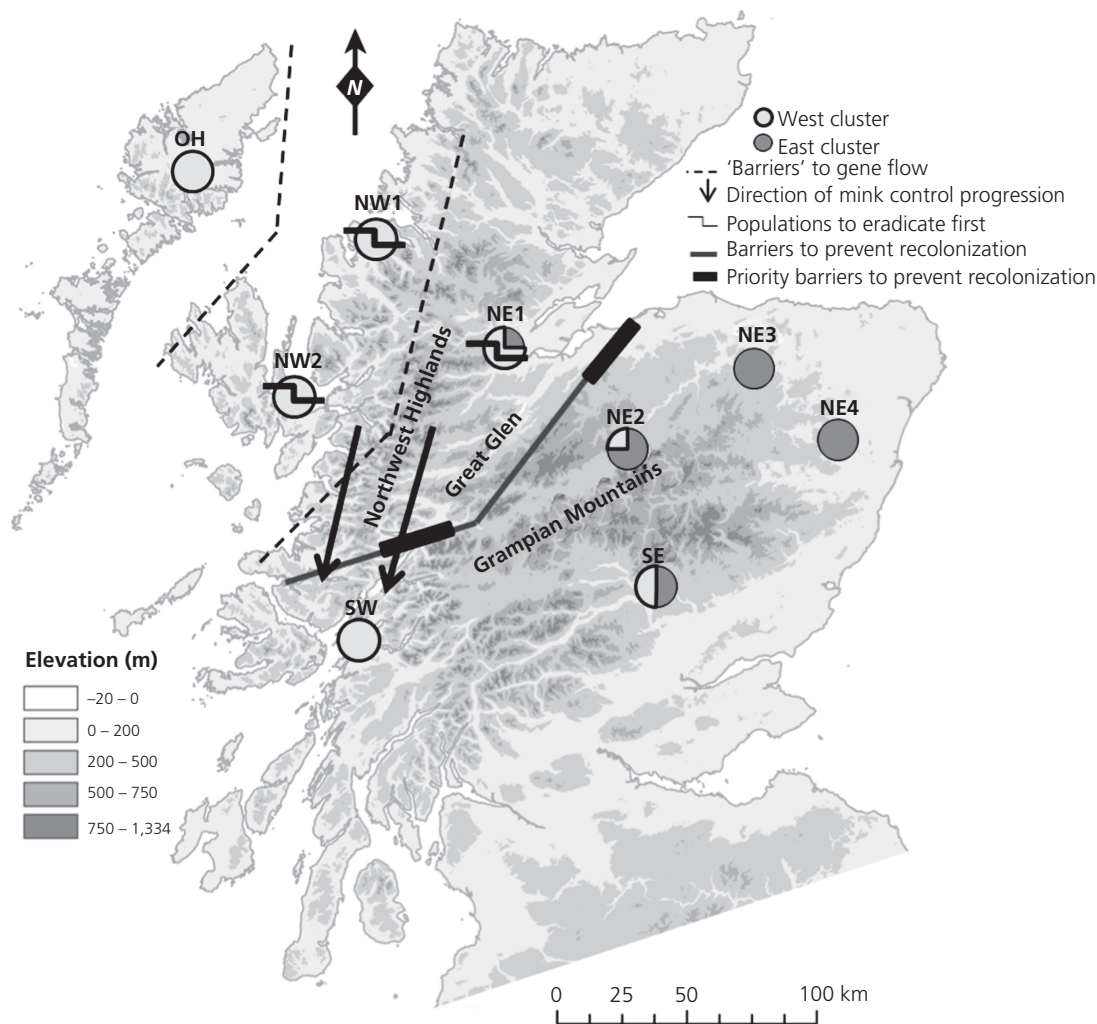


Figure 16.4 Schematic diagram of recommendations for mink control in Scotland to contract distribution to south of the Great Glen with illustration of the (approximate) genetic clusters each population was assigned to and barriers to gene flow as identified by BARRIER analysis. It is suggested that the NE1 population should be eradicated of breeding mink and recolonization of this area prevented by concentrating control efforts at key contact points. The NW populations should also be heavily depleted of breeding mink and pushed south. The natural barrier of the mountains should maintain isolation. The SW population should be incorporated into the management encompassing SE and remaining NE population areas.

the more moderate genetic divergence between mink populations in southwest (SW) and northeast (NE1) Scotland suggested that a mountain valley (the Great Glen) connecting the two areas is acting as a dispersal corridor.

Genetic population structure suggested that western areas of Scotland had been colonized independently from eastern areas, and that gene flow from eastern areas to northwestern areas is limited. It was concluded from allelic richness, heterozygosity, and pairwise genetic differentiation values that the southwest (SW) population of mink was the primary source of mink recently detected in northwest Scotland (NW1 and NW2). Combined with results from the habitat suitability model, this suggests that the coastline is likely to be the main dispersal route for mink moving in to northwest Scotland.

It has been suggested that mink eradication should concentrate on contact zones (Bifulchi et al. 2010; Zalewski et al. 2009) because admixture between genetic groups can increase adaptive potential which, in turn, can enhance invasion success (Kolbe et al. 2004; Lavergne and Molofsky 2007; Miller et al. 2005). Recommendations for targeted mink control in Scotland include breaking links between well-connected populations, particularly between NE1, SW, and NE2, to prevent further northwards invasion. The natural barrier of the mountains to the west of NE1 should maintain isolation from NW populations. Control of mink in northwest Scotland should be concentrated on managing west coast populations. By pushing NW populations south, eradicating the NE1 population and preventing recolonization of the area, it should be feasible to contract mink distribution and confine it to south of the Great Glen which acts as a dispersal corridor (see Figure 16.4). Although NE1 appears to be a demographic sink rather than a source, it would be precautionary to remove this population and suppress potential immigration. In the west, the recommendation was to heavily deplete the population in the north and push the invasion frontline south (Fraser et al. 2013). Enormous challenges stand in the way of this ideal, as removing all mink once they are established on a coastline encompassing numerous offshore islets inaccessible to volunteers requires resources that are not presently available. Thus controlling mink to low density so as to protect vulnerable nesting coastal birds and minimizing inland dispersal, in the full knowledge that some refugia will remain uncontrolled, is probably a more realistic target on the west coast of Scotland.

These conclusions and recommendations for future mink control demonstrate the usefulness of being able to compartmentalize a population and, subsequently, the task of control. This not only makes it easier to plan but also reduces the need to dissipate resources over a larger area than is necessary. Practical application of this type of approach is currently being demonstrated on South Georgia by those aiming to eradicate invasive brown rats (*Rattus norvegicus*). The discovery that some rat populations were isolated by glaciers and were unlikely to be recolonized once eradicated (Robertson and Gemmell 2004), enabled the eradication effort to be partitioned into geographical units as well as allowing it to be conducted over multiple seasons (www.sght.org). The three phases of laying poison bait are now complete and post-project monitoring to assess success is scheduled for 2017/2018 (www.sght.org). This staged approach is particularly useful in situations where budgets limit the spatial extent of suitably intensive control that can be applied simultaneously but is likely to be more effective than the localized attempts of well-intentioned individuals that achieve only a cropping of demographic surplus.

Identifying volunteers to help implement mink control

Now that we have some recommendations for strategic phasing of mink control in Scotland, what happens next? Planning a conservation or management project can be challenging but implementing it can be equally, if not more, testing. Volunteer involvement in conservation projects is being accepted increasingly as a cost-effective way to gather ecological information and implement conservation over large spatial scales, and given appropriate methods, training and supervision can be highly effective (Silvertown et al. 2013). Involving unpaid volunteers in collating information on the distribution (e.g. Delaney et al. 2008; Dickinson et al. 2010; Gallo and Waitt 2011; Buesching et al. 2014) and control of invasive species on a large scale is a feasible alternative to the costly use of professional staff (Bryce et al. 2011; Tidwell and Brunson 2008) where finances and support are otherwise limited. People willing to participate in invasive species management often have a 'stake' in the environment (Selge et al. 2011; Tidwell and Brunson 2008) making it possible to connect with the motivations of such stakeholders to promote voluntary participation in the management of invasive species.

Volunteers have proven to be an invaluable resource in the implementation of mink control in northeast Scotland (Bryce et al. 2011). Volunteers from a variety of sources and backgrounds were readily recruited to monitor rafts in locations local to their home or work. Initially, many of the volunteers were land managers on sporting estates that encompassed the project starting area and had convergent interests in protecting native wildlife (Bryce et al. 2011). Sourcing volunteers can be more difficult in a sparsely populated area, such as west Scotland. However, the rural economy in west Scotland is increasingly focused on tourism, including wildlife-watching (Blake et al. 2010; Bryden et al. 2010). Wildlife-watching tour operators are assumed to have an interest in conserving local wildlife and therefore could be a potential source of volunteers for mink control. Fraser et al. (2014) investigated whether it could be possible to harness assumed interest by ecotourism boat operators in low cost, coastal, localized, mink control in areas beyond the reach of current coordinated mink control projects. The study sought to assess the perceived impact of mink on coastal wildlife and opinions on the ensuing effect on businesses, and to determine the level of support required to mobilize this potential source of volunteers. The aim was to provide clear guidance on the external input required to mobilize volunteers who could work towards achieving local protection of key bird colonies in their coastal communities.

The perception of ecotourism boat operators in west Scotland towards mink was assessed using a questionnaire, structured with both closed-format and open-ended questions. Respondents were first asked to give opinions about their business priorities, followed by questions directed towards considering the benefits of mink control, their willingness to participate, and at what cost they would involve themselves. The questions were interspersed with informative text about mink and mink control, including the cost of equipment and an outline of the time commitment required when volunteering.

Responses were received from across west Scotland (Figure 16.5). The majority (64%) of respondents were concerned about the presence of mink in their area, agreed with control in principle, and were willing to become involved in a volunteer capacity. Respondents who would not volunteer but agreed with control (21%) indicated that they might reconsider if mink had a visible impact on their local wildlife. Those who disagreed (14%) with mink control would not consider volunteering under any circumstance. People who had

seen mink were more likely to volunteer to become involved in mink control than those that were only aware of their presence.

The overall results of the study suggested that people who are willing to participate in mink control would do so at their own cost and initiative if they were given written guidance on how to carry it out. Guidance required is likely to require minimal resources and could simply be a brochure or information pack detailing where to source monitoring equipment and traps, how to deploy equipment, and how to humanely and lawfully dispatch mink once caught. Respondents appeared to be more concerned about their local wildlife, particularly ground-nesting seabirds, than the direct impact on their business and this was clearly a motivation for volunteering in mink control. This small study suggested that the level of community involvement that could be developed on the west coast of Scotland has the potential to reduce the number of mink and thus provide local protection to seabird colonies.

Mink control elsewhere—what are the lessons and limitations of the Scottish experience?

There are clearly benefits to understanding how mink invade. The ability to predict the speed and direction of the minks' movements through the landscape, in particular, allows management to be targeted appropriately for maximum effect. Using adaptive management to improve ecological understanding, as well as capitalizing on local interest to get the work done, an area about the size of Wales (bigger than the US state of New Jersey) previously invaded by mink has now largely been freed from its influence. While efforts must continue to prevent further colonization and push the frontline of mink invasion south, the potency of the approach has been demonstrated. Most of northern Europe is now invaded, as was Scotland 10 years ago. However, while the area under the influence of mink is enormous and mink invasion cannot be wholly undone, the success in Scotland has demonstrated that, using a participatory approach grounded in ecology, it is possible to push back mink from areas of high biodiversity value.

The take-home message from our work in Scotland is that where colonization is ongoing, there is a degree of predictability that can be used if there is a strong commitment to push back this invasive species. We have shown that this is feasible in some areas, not

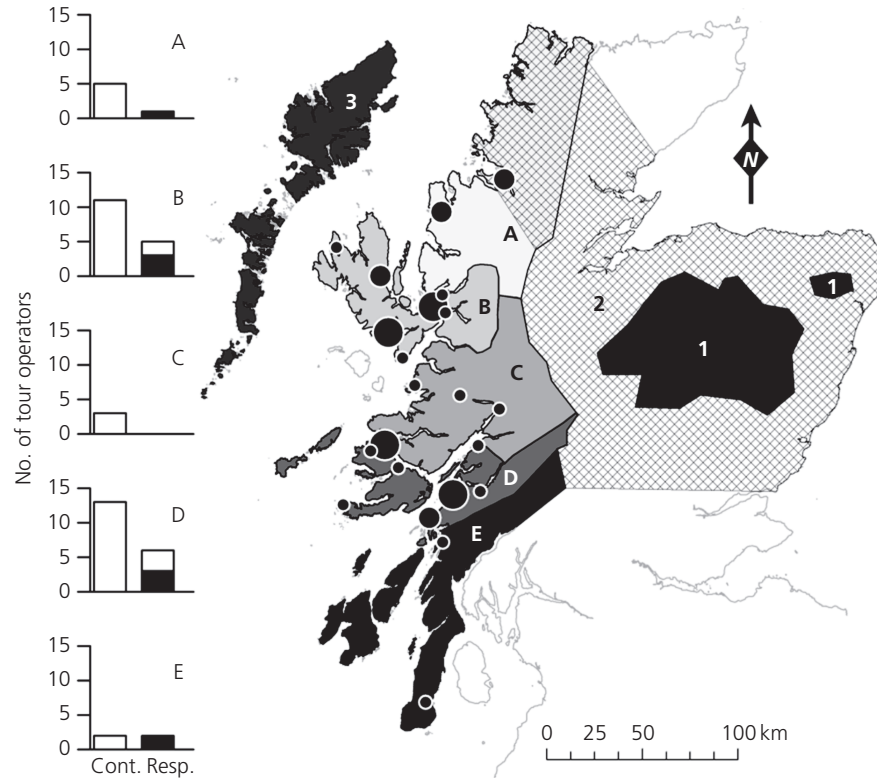


Figure 16.5 Map of Scotland showing all geographical locations mentioned in the text. Areas A–E show regional boundaries used to describe locations of west coast tour operators (A, North Highland; B, Skye and Lochalsh; C, Fort William and Lochaber; D, Oban and Mull; E, Mid Argyll and Kintyre). The gradation of shading from south to north (areas E to A) represents the pattern of colonization by mink on the west coast. Other mink control project areas are represented by numbers (1, original project area; 2, current project area; 3, the Outer Hebrides). The locations of all businesses contacted for the questionnaire are indicated by black dots, where the size represents number of businesses at each location (small = 1, medium = 2, large = 3). Bar graphs show the number of tour boat operators who were contacted (Cont.) and responded (Resp.), and, of those respondents, the number who were willing to volunteer (black section of bar) in each area (letters correspond as above).

least by relying on a motivated and supported volunteer work force, but it is also evident that it is difficult to reduce density durably without tackling a very large area or an area that by virtue of its topography is not swamped by recolonization.

There are other areas, in southern Europe, where invasion is ongoing, and mink spread is still limited, where preventing further spread and eliminating early foci of invasion may be both feasible and cost-effective. Here, the understanding of barriers can be really useful. However, fur farms still exist in all these countries, and none can be said to be escape-proof.

In Italy, where mink appear to be restricted to separate non-intermixing populations, eradication may still be possible (Iordan et al. 2012; Bartolommei et al. 2013). Application of a spatially explicit

population dynamics model (that simulates individual life histories and dispersal within a GIS-held habitat map, based on estimated population parameters, Bonesi et al. 2007) to the Italian landscape also highlighted the Italian island of Sardinia as a priority for action. On Sardinia, there is a mink farm, existing records of mink presence, a high density of waterways that presumably facilitates spread of mink populations, and a number of endemic species of freshwater amphibian that could be at risk from mink predation (e.g. the mountain newt, *Euproctus platycephalus*, Lecis and Norris 2004). The model predicted that without control mink would colonize 30% of the water bodies in Sardinia within 20 years (Iordan et al. 2012). Because Sardinia is an island, with reduced risk of recolonization, eradication here would also be a reasonable aim

(although reality, as elsewhere, will be dependent on resource availability, public and/or governmental willingness and interest to address the problem, and the closure of existing mink farms).

Because mink respond to ecological barriers and not political boundaries, mink eradication is probably not feasible in Portugal where they are currently increasing their range, are at a relatively early stage in their colonization (first recorded in the country in the late 1980s), but come from well-established populations near the border with Spain (Rodrigues et al. 2015). However, there are clear parallels with the situation in Scotland, insofar as range expansion of mink in Portugal appeared initially (in the first 20 years since its presumed introduction in 1985) to be very slow (2.75 km per year) in an area of presumably low-quality habitat (with fast-flowing, low-productivity rivers, and a high number of competitors). The possible role of the coast in facilitating the spread of mink south in Portugal has not been investigated, but a recent significant increase in range (at a rate of 22.5 km per year) was thought to relate to an increase in food availability due to the simultaneous increase in range of the American red swamp crayfish (*Procambarus clarkii*) in Portugal (Rodrigues et al. 2015; see also Melero et al. 2014). Rodrigues et al. (2015) recommend immediate mink control in the northwest where there is a significant influx across the border from Spain (although the requirement in Spain, and a number of other countries in mainland Europe, that all mink be killed by a vet make any control efforts logistically more difficult and certainly more costly, see Melero et al. 2014).

Ruiz-Olmo et al. (1997) found that the highest range expansion rates in Spain occurred in areas with few or no other mustelids leading to the conclusion that the spread of mink will be slowed by the presence of native competitors. However, even in the absence of competitors, these authors recorded a maximum spread of mink of only 10 km per year, yet rates of spread in both Scotland and Portugal, where there are otters, polecats, and one or more *Martes* spp., were twice that or more (in some areas, or at some times). Further, although earlier studies of interactions between mink and otters (which are much larger than mink) suggested that recovering otter populations in Britain might exclude mink, and thus facilitate a reduction in mink density (Bonesi et al. 2004, see also McDonald et al. 2007), there is no evidence of this actually happening. In Scotland, otters are widespread, yet mink have successfully colonized, as they have in Belarus where both species are now abundant on the same rivers (Sidorovich and

Macdonald 2001). None of this precludes there being a competitive interaction among mink, otter, and other sympatric small carnivores. Indeed, Harrington et al. (2009b) found a shift in activity patterns of American mink in an area of southern England when otters and polecats recolonized, and coexistence in Belarus is probably facilitated by the surrounding wetlands and possibilities for resource division (see discussion in Harrington and Macdonald 2015). Nevertheless, interaction does not imply exclusion and thus, although these intraguild interactions are of academic interest, the presence of the otter (or other sympatric competitors) has little management relevance.

Another case where lessons from Scotland could be applied is in Argentina, where mink are reaching isolated plateau lakes (where they predate on endangered endemic hooded grebes, *Podiceps gallardoi*, Roesler et al. 2012a). These lakes are not connected, even by seasonal streams, which means that mink are travelling long distances over largely unsuitable habitat. The vast and unpopulated area (5000 km²) (traversed by only one or two main roads) and apparently very low density mink population means that large-scale eradication is probably not feasible without substantial expenditure (and even then may be prohibited by difficult access to rivers and the harsh climate that is only suitable for trapping over a short period of the year). Initial trapping, over two years, at a number of the lakes has successfully reduced mink numbers and (at least temporarily) stopped predation (Fasola and Roesler 2016), but, given the endangered status of the grebes, a method to prevent mink reaching the lakes over a larger area is urgently needed. Presuming dispersal movement kernels follow an inverse power law function (cf. Byrne et al. 2014a for badgers), it is inevitable that a mink will eventually reach even the most remote places and the distances moved to reach the lakes are in accordance with mink moving long distances through unsuitable habitat in Scotland. There, the Cairngorm project was able to reduce colonization of poor quality mink habitat that is nevertheless valuable for native species by creating vacancies in good quality mink habitat, which mink colonize first, and from where they could be readily removed (Bryce et al. 2011). This use of 'attractive sinks' (Delibes et al. 2001) in management is somewhat different to the idea of a buffer area insofar as mink are specifically attracted to another area away from areas of conservation concern. Focussing effort on areas that are attractive to recolonists is now central to the management of very low residual mink numbers in the Scottish project

(Oliver et al. 2016). It is not yet clear whether such an approach might be suitable in Argentina; a heavily trapped buffer area around each lake (or a number of lakes) might be required, simply because a single mink can destroy an entire colony of this endangered bird (Roesler et al. 2012a, b).

Despite several decades of research by a number of different organizations across the UK and elsewhere in Europe, there remain a number of unanswered questions pertaining to efficient mink management and reduction of their impact on native fauna. Key issues include: How transferable is the Scottish volunteer approach to different countries or cultural environments? Where eradication is not achievable, what residual mink density is compatible with preserving biodiversity assets (bearing in mind that this may, for some species [e.g. hooded grebes], be zero; see also Maran et al., Chapter 17, this volume)? How can one optimize the trade-off between cost of control and biodiversity return? And, linked to these, what scale of mink control is required to minimize reinvasion (given other complicating factors such as density dependence in reproduction and dispersal, Sidorovich 1997, Melero et al. 2015)?

Although it may be feasible to remove mink at any scale given sufficient resources, a key conservation question is: at what cost can initial removal be sustained over time? The dispersal ability of mink, and hence scope to reinvade, dictates that mink control should only be attempted where it can be done on a large scale. Yet, large-scale projects should only be undertaken where there is long-term commitment and where there are geographical assets (as in northern Scotland) that can facilitate human management. Small-scale projects, while responding to the

legitimate desire by individuals to ‘do something’, need to balance the ethical costs in terms of numbers of mink killed (accounting for the likely continual influx of dispersers), against small-scale and temporary biodiversity gains, and are probably only appropriate as a short-term emergency action prior to a longer-term, larger-scale sustainable management plan. Killing invasive species simply because they are invasive, in ways that offer no prospect of limiting their impact, is as tawdry in the context of alien species as is inflicting suffering and squandering resources in any other context. With this in mind, we must be clear what we mean and what we expect by ‘effectiveness’. To compare future projects, it will be important to clarify to what density mink were reduced, over what area, for how long, and at what cost. Moving beyond the UK, the presence of mink farms in many countries in mainland Europe certainly makes justifying effort by society (as opposed to farm owners, i.e. ‘polluter pays’) questionable. Thus, whilst there are ecological and biological hurdles to overcome, there are perhaps even bigger societal and financial hurdles, and all this lies against the backcloth of an ethical imperative that killing invaders can be justified only if it is effective in delivering a conservation gain.

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