Invasive species removals and scale – contrasting island and mainland experience

P.A. Robertson[1], S. Roy[2], A.C. Mill[1], M. Shirley[1], T. Adriaens[3], A.I. Ward[4], V. Tatayah[5] and O. Booy[1,6]

[1]Centre for Wildlife Management, Newcastle University, Newcastle upon Tyne, NE1 4DD, UK. < peter.robertson@ncl.ac.uk> [2]International Union for the Conservation of Nature, Gland, Switzerland. [3]Research Inst. for Nature and Forest (INBO), Wildlife Management and Invasive Species, Havenlaan 88 bus 73, B-1000 Brussels, Belgium. [4]School of Biological, Biomedical and Environmental Sciences, The University of Hull, HU6 7RX, UK. [5]Mauritian Wildlife Foundation, Grannum Road, Vacoas 73418, Mauritius. [6]Animal and Plant Health Agency, Sand Hutton, York, YO41 1JW, UK.

Abstract Recent years have seen large increases in the number and size of successful invasive species eradications from islands. There is also a long history of large scale removals on larger land-masses. These programmes for mammals and terrestrial plants follow the same cost-area relationship although spanning 10 orders of magnitude in scale. Eradication can be readily defined in island situations but can be more complex on larger land-masses where uncertainties defining the extent of a population, multiple population centres on the same land-mass and ongoing risks of immigration are commonplace. The term 'complete removal' is proposed to describe removal from an area with ongoing effort to maintain the area as clear, as features in many larger scale mainland programmes. Examples of complete removal to a boundary, in patches and in habitat islands are discussed. While island eradications continue to grow in scale, new legislation such as the lists of Species of European Union Concern will also drive increasing management on larger land-masses. However, these lists include large numbers of species that are already widespread. Methods are needed to prioritise species to reflect both the risks posed and the feasibility of management, including the effects of scale on cost and effectiveness.

Keywords: control, eradication, invasive alien species, non-native species

INTRODUCTION

The removal or eradication of invasive alien species is increasingly used as a conservation tool. New legislation, for example the European Union's Invasive Alien Species Regulation, will also place increasing responsibilities on states to remove or eradicate high risk species. Both of these considerations are driving an increased number of management programmes at increasing scales and there is a need to understand how the costs and constraints change in relation to scale. A large number of published eradications have been based on islands, often at relatively small scales, while a small number of larger programmes have been based on mainland experience. There is a need to pull together these different sources of evidence, to support an assessment across a wider range of scales than can be achieved by considering islands or mainland eradications in isolation.

REMOVAL AT SCALE – ISLANDS AND MAINLAND EXPERIENCE

Recent years have seen a large increase in successful invasive species eradications from islands, as well as significant increases in the size of islands involved. The number of successful eradications continues to increase, and in 2012 the Database of Invasive Species Eradications (<http://diise.islandconservation.org>) recorded 1,182 whole-island introduced invasive animal species eradication projects either completed or underway on 762 individual islands. In terms of scale, recent years have seen a number of large island eradications. Cruz, et al. (2009) describe the eradication of goats from the 584 km^2 Santiago Island in Galapagos; Parkes, et al. (2014) predicted the effort required to remove cats from the 1,680 km^2 Stewart Island in New Zealand, while the current rat removal on South Georgia will cover 3,538 km^2 (Piertney, et al., 2016).

Although the point at which an island becomes a mainland is arbitrary, there is also a long history of invasive mammal removals from larger land masses in Northern Europe (Robertson, et al., 2017). These include muskrat (*Ondatra zibethicus*) eradications from the mainlands of Britain and Ireland in the 1930s; the eradication of the Himalayan porcupine *(Hystrix brachyura*) (1970s) and coypu (*Myocaster coypus*) (1980s) from the British mainland; a variety of American mink (*Neovison vison*) and grey squirrel (*Sciurus carolinensis*) removals from the larger British islands together with the removal of Pallas' squirrel (*Callosciurus erythraeus*) from Flanders on the European mainland (since 2000). Few of the programmes covered more than a fraction of the total land mass, so size was defined as the area over which species sightings occurred and trapping took place. The larger of these species programmes have covered areas of 3,411 km^2 (the two phases of the Hebridean mink programme), 5,219 km^2 (the five separate muskrat eradications) and 19,210 km^2 (coypu) (details and full references given in Robertson, et al. 2017). The ongoing ruddy duck (*Oxyura jamaicensis*) eradication from Europe (Robertson, et al., 2015) covers six states totalling 1,535,509 km^2.

Data on the costs of eradications are available for projects covering ten orders of magnitude of scale. Studies have described the costs of successful mammal eradications from islands (Martins, et al., 2006; Howald, et al., 2007) and larger land-masses (Robertson, et al., 2017), while Rejmánek & Pitcairn (2002) describe costed plant eradications in California. For mammal eradications, those on large land-masses covered significantly larger areas than those reported from islands while successful plant eradications were confined to smaller areas. Data from these different sources, appear to follow the same relationship (Fig. 1) whereby the cost per unit area is reduced by approximately 10% as the area involved doubles (Robertson, et al., 2017). As experience of eradications on larger islands grows, the overlap between island and mainland experiences is increasing (Cruz, et al., 2009; Parkes, et al., 2014; Piertney, et al., 2016).

It is worth recording that two small datasets describe programmes that fall outside this relationship. Rejmánek & Pitcairn (2002) also record three aquatic plant eradications which appeared more expensive than comparably sized terrestrial plant programmes, while the ruddy duck

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eradication (Robertson, et al., 2015) has been significantly less expensive compared to similarly scaled mammal programmes (Robertson, et al., 2017). More data are needed on the management of other taxa in different environments before firm conclusions can be drawn. These results are based upon currently available methods of eradication. As new technologies, such as gene-drives (Webber, et al., 2015), e-DNA self-resetting (Carter, et al., 2016) and self-reporting traps (Jones, et al., 2015) become available it is likely that these costs will decrease.

Eradication and complete removal

In their classic paper, Bomford & O'Brien (1995) make a clear distinction between eradication and on-going control, presenting these as alternative objectives for management. They also identify three key criteria for successful eradication; that the rate of removal exceeds the rate of increase at all densities; there is no immigration; and all reproductive animals are at risk.

These definitions and criteria have guided many successful eradications and are particularly applicable to islands where the population extent and risks of immigration can be readily assessed. However, at the scales found on larger land masses, these criteria may be more difficult to apply or achieve, for example where the boundaries of a population remain poorly defined, where multiple population centres may occur on the same land mass, or where immigration remains a risk. Despite this, large scale programmes frequently lead to the removal of species from large areas of land. Although not meeting Bomford & O'Brien's (1995) definition of eradication, these situations are also not well described as on-going control as no active management is required across the majority of the area. In these circumstances 'complete removal' may be a better definition of the objectives, sitting between Bomford & O'Brien's (1995) definitions of eradication and on-going control.

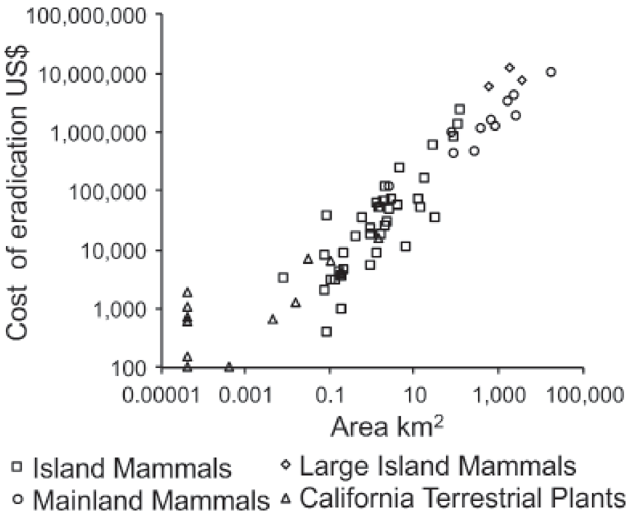

Fig. 1 The relationship between the area (km2) of a successful removal and the total cost (US$). The square symbols represent island mammal eradications reported by Martins, et al., (2006). The circles are for removals of mammals from larger land masses in Northern Europe (Robertson, et al., 2016). The three diamond symbols are recent examples or predictions of large-scale mammal eradications from islands: (Cruz, et al., 2009; Parkes, et al., 2014; Piertney, 2016). Plant eradications from California are triangles (Rejmánek & Pitcairn, 2002). Where the study recorded effort as man-years or man-days, total cost is estimated based on US$50k per man-year (Rejmánek & Pitcairn, 2002; Parkes, et al., 2014; Robertson, et al., 2016).

Eradication, the complete removal from an area, with no immediate prospect of recolonisation from neighbouring areas.

Complete Removal from an area but with ongoing effort to maintain the area as clear.

On-going Control within an area to reduce abundance, associated damage and the risk of spread.

Based on this definition, complete removal has been applied in a number of forms.

1 - Complete removal to a boundary

One objective of large scale programmes can include complete removal of a species up to a boundary across which the risk of reinvasion remains. Control along the boundary, or in a neighbouring buffer zone, can reduce the risk of reinvasion and help keep the main area clear. The nature of the boundary may vary, including fences (Saunders & Norton, 2001), landscape barriers such as water bodies or mountains (Schuchert, et al., 2014), or bottlenecks through which invading animals must move (Roy, et al., 2015). These boundaries can be permanent features of the management, requiring ongoing inputs (Saunders & Norton, 2001), or may be part of a phased programme to clear a larger area (Yamada & Sugimura, 2004; Bryce, et al., 2011; Robertson, et al., 2015; Russell, et al., 2015). If the aim is the removal of the species from a large area, but the funds or resources are insufficient for the simultaneous management of the entire population, then removal to a boundary is likely to feature.

The North American ruddy duck was introduced to the UK in the late 1940s, and its subsequent spread into Europe threatens the native white-headed duck (*Oxyura leucocephalus*) through hybridisation. The plan to eradicate the ruddy duck from Europe involves coordinated management across the continent. As the UK was the original source of this population and contained the majority of the birds, it was the focus of initial control (Robertson, et al., 2015). However, once the UK no longer contained breeding birds (currently it is thought only a few males remain), the English Channel became a boundary between a cleared area and the remaining continental populations. Control of the remaining European birds is ongoing, in the meantime the UK maintains surveillance and, if required, control along this boundary to maintain its cleared status.

In the UK, the native red squirrel (*Sciurus vulgaris*) is threatened by the ongoing spread of the invasive grey squirrel (*S. carolinensis*). This is mediated by the spread of a poxvirus by the asymptomatic greys which is typically fatal to the reds (Rushton, et al., 2000). The island of Anglesey on the north coast of Wales contained a small relict population of the native reds although greys were spreading onto the island. A control programme removed the greys (Schuchert, et al., 2014), allowing the reds to spread and recolonise the entire island. Anglesey is separated from mainland Wales by a narrow tidal channel, crossed by two bridges. There is evidence that grey squirrels can cross this boundary and the risk of recolonisation remains. To reduce this risk and maintain the island as grey squirrel-free, management has included a surveillance and rapid response programme to pick up incursions (Shuttleworth, et al., 2016), trapping to reduce the density of greys on the mainland side of the boundary, and a plan to extend the area of complete removal to clear greys from the North Wales coast up to a more distant boundary formed by a geographic bottleneck where the mountains meet the coast.

The American mink (*Neovison vison*) spread through the Western Isles of Scotland following its escape from fur farms in the 1950s. Its spread threatened internationally important populations of ground nesting birds as well as local economic activities such as salmon fishing. The decision was taken to aim for the eradication of this species from the archipelago but logistic and funding constraints, combined with the need to gain experience, led to a phased programme. In the first phase, mink were completely removed from the Uists, the southernmost islands of the chain (Roy, et al., 2015; Faulkner, et al., 2017). A buffer zone was maintained (South Harris) between this cleared area and the remaining mink population on the main island (Lewis) to the north. This buffer included a narrow, island strewn channel between the Uists and South Harris. Trapping on these 'stepping stone' islands together with South Harris itself provided an effective barrier to recolonisation. Once the Uists' work had provided confidence that eradication was feasible, a second phase extended mink control north to cover the remainder of the archipelago (Lambin, et al., 2014).

2 - Complete removal from patches

In some cases the primary objective of management may be the reduction of the impact of an invasive species with no prospect to eradicate. In many cases this constitutes ongoing control rather than complete removal (Bomford & O'Brien, 1995), although in some circumstances it can lead to complete removal. For this to occur, two criteria must be met, the species must be controlled at a rate sufficient to remove all of the resident animals in an area, and the scale of control should be such that the risk of recolonisation is so low in the centre of the controlled area that the central area is effectively maintained clear. The prospects of this occurring are scale-dependent, with the cleared area forming a larger proportion of the total as scale increases.

This approach has been used in New Zealand with the creation of 'mainland islands', areas maintained predator-free through the use of fencing combined with continuing control (Saunders & Norton 2001; Gillies, et al., 2003). The same results can be achieved without fencing, for example in Mauritius where the introduced small Asian mongoose (*Urva auropunctata*) (Patou, et al., 2009) is a major threat to the continued survival of a range of native bird species (Bunbury, et al., 2008). The mongoose is widely spread across the island, inhabiting a range of habitats, while the native birds are largely confined to remaining patches of good quality native forest. Control of the mongoose has been carried out in a number of these forest areas to create 'mongoose free' patches within the wider mongoose distribution. A network of box traps has been in place since 1989 and maintains a year-round effort to remove mongoose. As the size of the trapped area increases, the number of animals captured per unit area decreases (Fig. 2). Areas less than 5 km in extent continue to catch high numbers of mongoose per unit area, presumably because they face constant recolonisation pressure from neighbouring habitats. However, in larger areas, particularly those over 10 km^2 in area, mongoose catch per unit area drops dramatically. This is consistent with catching animals in a boundary area, with the proportion of the area maintained as mongoose-free increasing as the total area trapped increases. Achieving this requires ongoing effort, but complete removal provides many of the benefits of eradication, and has been a key element of efforts to conserve a suite of species endemic to the island. These include the Mauritius kestrel (*Falco punctatus*), the pink pigeon (*Nesoenas mayeri*), the echo parakeet (*Psittacula eques*) and a number of passerines such as the Mauritius black bulbul (*Hypsipetes olivaceus)*, and Mauritius fody (*Foudia rubra)*. Only through intensive trapping to maintain these predator-free patches, combined with a captive breeding and release programme, disease management and supplementary feeding, have these species managed to persist.

3 – Complete removal from habitat islands

Islands as blocks of land surrounded by water are widely recognised, but isolated blocks of habitat within a matrix of other land uses share many of the same characteristics. When invasive alien species are confined to discrete habitats within this matrix, they can be considered as inhabiting 'habitat islands'. In these cases, limited rates of species movement or colonisation between habitat islands may produce isolated populations, with particular opportunities for management within large land masses.

The monk parakeet (*Myiopsitta monachus*) has established a number of discrete populations in different European cities (Munoz & Real, 2006; Rodríguez-Pastor, et al., 2012). Although an attractive species widely kept as a pet, in the wild this species builds large communal nests on tall trees or man-made structures such as electricity infrastructure or radio masts. The large size and volume of nest material can lead to electrical short-outs and fire risks, with consequent economic costs (Avery, et al., 2002). The discrete nature of its current distribution, with isolated populations including London, Amsterdam and a variety of Spanish cities suggests that different populations have resulted from separate releases rather than natural spread from a single point of release. The management of this species reflects this, with some regions attempting the complete removal of isolated populations (Parrott, 2013).

The introduced Pallas' squirrel (*Callosciurus erythraeus*) also has a highly fragmented distribution within Europe, suggesting a number of separate introductions rather than spread from a single point of release. A rapid response in Flanders, Belgium removed a population whose distribution was constrained to a suburban setting in a small community surrounded by farmland (Adriaens, et al., 2015). In effect this species was present on a habitat island which aided its removal.

The current removal of rats from South Georgia (Piertney, et al., 2016) uses a similar approach. Glaciers on the island separate a number of discrete rat populations, which appear to be genetically isolated (Robertson & Gemmell, 2004). This allows the complete removal of discrete populations as steps to achieve the larger goal of island wide eradication.

These examples illustrate the potential for effective removal of isolated populations to be undertaken within larger land masses, using the principles applied to island

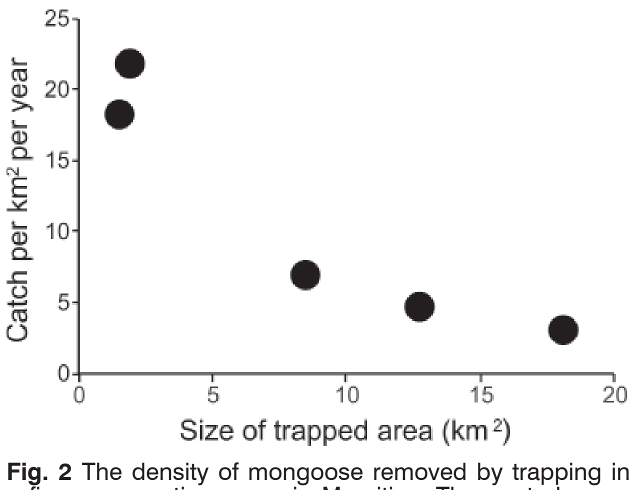

Fig. 2 The density of mongoose removed by trapping in five conservation areas in Mauritius. The control areas were surrounded by habitat containing mongoose populations.

eradications. However, as species establish and spread these discrete populations will become less pronounced. Identifying whether the distribution of a species represents a number of discrete clusters will have important implications for management, for example the decision to consider complete removal or on-going control. Spatial analysis of distributional data can be used to indicate the presence of discrete populations of a species. A range of spatial and spatio-temporal clustering algorithms (Velázquez, et al., 2016) can detect spatial point patterns and may be useful to differentiate clusters as they form.

EFFECTIVENESS AND SCALE

We used published accounts to assess the costs of removal at different scales. Doing so requires dealing with a number of biases. Firstly, it is commonly recognised that the published literature preferentially records success (Dwan, et al., 2008). For example, the successful coypu eradication in the UK is well documented (Gosling & Baker, 1987; Baker & Clarke, 1988; Gosling, et al., 1988; Gosling & Baker, 1989; Baker, 2006); the failed UK attempt to eradicate the American mink is barely recorded (Sheail, 2004) although it took place on a similar scale. Other failures are likely to have gone unrecorded. A publicly available database of island eradications is available (Keitt, et al., 2011), it would be useful to extend this to also include details of eradications on larger land masses. More importantly, the literature only records attempts, there is very little information on those situations where no action was taken, either through inaction or a judgement that it was not worthwhile. Inaction remains the most common response to invasive species. The successful island eradications are based on only a tiny proportion of the world islands, while the number of attempted eradications of alien species in Europe (Genovesi, 2005) is a similarly small proportion of the 20,000 species thought to have established.

If we are to make more objective decisions, we need to decide if, and when, management is appropriate in both island and mainland situations. Prioritisation methods have been applied to islands to identify those where management may be most beneficial (Harris, et al., 2012; Dawson, et al., 2015). Booy, et al. 2017 describe a method to assess the feasibility of eradication which incorporates the consideration of scale. If, as seems likely but has yet to be convincingly demonstrated, the prospects for successful eradication or complete removal decrease as a species spreads, then these methods offer a route to assess at what scale eradication or complete removal may no longer be a realistic outcome.

The application of methods to assess the feasibility of management is a critical need. The current EU invasive alien species regulations include the listing of species considered to be of 'Union Concern' and place reporting and management obligations on member states in which they occur. The selection of species for listing is largely based on established methods of risk assessment (Roy, et al., 2014), identifying species which pose a risk without similarly considering the feasibility of management. This focus on risk can result in the listing of species for which there are few realistic prospects for management. For example, of the 79 species currently listed or under consideration as Species of Union Concern, over half are already present in at least five member states. To date there are no successful examples of species eradication or complete removal in Europe when a species has already spread to this number of countries, although these may occur in future. Listing species based on risk assessment alone, without considering the scale and feasibility of management, risks committing resources into the on-going management of already widespread species, rather than the more productive routes of prevention and rapid response.

CONCLUSIONS

The experience of island eradications continues to grow, and to be applied at increasing scales. Alongside this, new legislation will drive increasing management on larger land masses. As island eradications grow in scale they will face many of the challenges experienced on larger land-masses, such as problems defining populations, multiple population centres on the same land mass, ongoing risks of immigration and the need for interim objectives. We suggest the term 'complete removal' to reflect the situation regularly encountered on larger land masses where a species may be removed from an area but with the need for an ongoing effort to maintain the area clear given the risk of reinvasion. The literature contains examples of successful eradications or complete removals in island and mainland situations covering 10 orders of magnitude. These island and mainland programmes appear to follow the same cost-area relationship. They also demonstrate an advantage of scale, with the costs per unit area of control reduced as the area of control increases. On larger land masses, such as the EU, care is needed to focus species listing on species where prevention, eradication or complete removal are realistic outcomes rather than committing member states to the on-going control of already widespread species. Methods of prioritisation which balance both risk and the feasibility of management, including the effects of scale on cost and effectiveness, are needed to guide future actions.

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