PRIORITISING THE MANAGEMENT OF INVASIVE NON-NATIVE SPECIES

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Abstract

Invasive non-native species (INNS) are a global threat to economies and biodiversity. With large numbers of species and limited resources, their management must be carefully prioritised; yet agreed methods to support prioritisation are lacking. Here, methods to support prioritisation based on species impacts, pathways of introduction and management feasibility were developed and tested. Results provide, for the first time, a comprehensive list of INNS in Great Britain (GB) based on the severity of their biodiversity impacts. This revealed that established vertebrates, aquatic species and non-European species caused greater impacts than other groups. These high impact groups increased as a proportion of all non-native species over time; yet overall the proportion of INNS in GB decreased. This was likely the result of lag in the detection of impact, suggesting that GB is suffering from invasion debt. Testing methods for ranking the importance of introduction pathways showed that methods incorporating impact, uncertainty and temporal trend performed better than methods based on counts of all species. Eradicating new and emerging species is one of the most effective management responses; however, practical methods to prioritise species based both on their risk and the feasibility of their eradication are lacking. A novel risk management method was developed and applied in GB and the EU to identify not only priority species for eradication and contingency planning, but also prevention and long term management. In this way, long lists of species were reduced to management focussed short lists that provided better cost-benefit than risk assessment alone. These pathway ranking and species prioritisation methods complement risk assessment and horizon scanning tools within a wider risk analysis framework for prioritisation. While applied here to identify management priorities in GB and the EU, they are flexible and could help prioritise INNS management at local, national and international scales.

Dedication

To my grandparents, Phyllis Elizabeth Christina Harrison (née Haydon) and Ivan Redvars Harrison, who inspired my interest in ecology and started me on this PhD.

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* (I do not promise)

Declaration

The analysis in Chapters 2 and 3 utilised data held by the Non-native Species Information Portal (NNSIP) project, based at the Centre for Ecology and Hydrology and funded by Defra. This database has been collated by a wide range of experts and has benefited from extensive volunteer input across Great Britain (Roy et al., 2014c, Roy et al., 2015c).

The expert elicitation methods used in Chapters 2, 4 and 5 utilised the knowledge and experience of a wide range of experts (n=93), all of whom are listed in the appendices to this thesis (Appendix A, E and F). Jess Ward helped to collect data and organise the experts involved in the EU risk management workshop (Chapter 5), which I co-facilitated with Piero Genovesi. Matt Grainger helped me to produce Figure 2.2.

Chapter 4 has been published as Booy, O., A. C. Mill, H. E. Roy, A. Hiley, N. Moore, P. Robertson, S. Baker, M. Brazier, M. Bue, R. Bullock, S. Campbell, D. Eyre, J. Foster, M. Hatton-Ellis, J. Long, C. Macadam, C. Morrison-Bell, J. Mumford, J. Newman, D. Parrott, R. Payne, T. Renals, E. Rodgers, M. Spencer, P. Stebbing, M. Sutton-Croft, K. J. Walker, A. Ward, S. Whittaker and G. Wyn (2017). "Risk management to prioritise the eradication of new and emerging invasive non-native species." *Biological Invasions* 19(8): 2401-2417. I devised, led and completed this work. All co-authors provided individual risk management scores and comments on the draft manuscript.

Chapter 5 is in preparation for being published.

With support from my supervisorial team, all other work was done by Olaf Booy.

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Chapter 1. Introduction

Non-native (or alien) species are those that have been moved by human agency, intentionally or unintentionally, to new locations outside of their native range (CBD, 1992); although definitions vary and have been the subject of much debate (Essl et al., 2018). The rate of non-native species introduction around the world has increased as a result of increasing movements of people, vehicles and goods (Levine and D'Antonio, 2003, Meyerson and Mooney, 2007, Westphal et al., 2008, Hulme, 2009, Bradley et al., 2012, van Kleunen et al., 2015) and there is no sign of saturation in the accumulation of species (Seebens et al., 2017). Indeed, continued expansion in global trade is anticipated to accelerate the rate of new invasions (Levine and D'Antonio, 2003, Hulme, 2009, Seebens et al., 2015, Seebens et al., 2018). This pattern is true of non-native species from a wide range of taxa that occur in different environments and at different scales (Dawson et al., 2017, Dyer et al., 2017, Seebens et al., 2017, Pagad et al., 2018). It is also apparent in Great Britain (GB), where the number of established non-native species has been increasing rapidly since the industrial revolution (Roy et al., 2014c). Indeed, between 1950 and 2017 non-native species established on average at a rate of 10.7 each year, compared to just 0.9 per year in the period 1600-1799 (NNSIP, 2017).

The majority of non-native species introduced to a new area either fail to establish or establish but have few negative impacts in their new environment (Williamson and Fitter, 1996, Lockwood et al., 2009, Blackburn et al., 2011). Indeed, some non-native species provide benefits, for example as crops, livestock and for ornament (McNeely, 1999, Richardson, 2001, Shackleton et al., 2019). However, a small proportion of those that establish cause negative impacts (Williamson and Fitter, 1996, Mooney and Cleland, 2001, Simberloff et al., 2013). These are termed invasive (Jeschke et al., 2014, Russell and Blackburn, 2017, Essl et al., 2018); however, Ricciardi et al. (2013) argue that all established non-native species cause at least some impact by definition and suggest that invasiveness should be considered a continuum from weak to strong impact, rather than a dichotomy between two states. Some have argued that species should not be distinguished based on their native or non-native origins but whether or not they cause impacts (Davis et al., 2011, Schlaepfer, 2018). However, others have emphasised the fundamental ecological differences between native and non-native species and the impacts that can occur as a result (Richardson and Ricciardi, 2013, Wilson et al., 2016, Pauchard et al., 2018). It has also been suggested that the impacts of invasive non-native species have been over-emphasised (Briggs, 2017); however, this too has been widely refuted (Richardson and Ricciardi, 2013, Russell and Blackburn, 2017).

The impacts of invasive non-native species are many and severe. Few, if any, regions worldwide have not been affected by them (Pain, 2007, Early et al., 2016, Duffy et al., 2017, Turbelin et al., 2017), with impacts affecting a diverse range of taxa in many different habitats (Vilà et al., 2011, Pyšek et al., 2012, Genovesi et al., 2015, Measey et al., 2016, Dueñas et al., 2018) caused by many different mechanisms (Mooney and Cleland, 2001, Nentwig et al., 2016). They are one of the main drivers of species extinction worldwide (Bellard et al., 2016), generating biotic homogenisation (McKinney and Lockwood, 1999) and causing a wide range of other impacts on native species, ecosystems and ecosystem services (e.g. Pejchar and Mooney, 2009, Vilà et al., 2010, Simberloff et al., 2013, Genovesi et al., 2015, Cameron et al., 2016, Gallardo et al., 2016, Dueñas et al., 2018). While impacts are most severe on small, isolated islands (Vitousek, 1988, Reaser et al., 2007, Courchamp et al., 2017, Dawson et al., 2017, Russell et al., 2017) and other isolated systems (Perrings et al., 2002, Moorhouse and Macdonald, 2015), they also occur on continents (D'antonio and Dudley, 1995, Vilà et al., 2011, Dueñas et al., 2018). In addition to biodiversity impacts, invasive non-native species are a massive drain on global resources (Early et al., 2016, Cassey et al., 2018b). For example, the cost of invasive non-native species to the US economy is estimated to be $120bn (USD) per year (Pimentel et al., 2005), while in the EU the equivalent figure is estimated to be at least €12.5bn (EURO) (Kettunen et al., 2009). They also cause human health impacts (Pyšek and Richardson, 2010, Hulme, 2014, Lazzaro et al., 2018, Mazza and Tricarico, 2018, Shackleton et al., 2019). For example, a single non-native species in Europe, *Ambrosia artemesifolia*, has become the leading cause of hay fever suffering in several European countries (Lazzaro et al., 2018).

In GB, invasive non-native species have caused a wide range of impacts (Boag and Yeates, 2001, Aldridge et al., 2004, Dehnen-Schmutz et al., 2007, Gallardo et al., 2015, Roy and Brown, 2015, Smith et al., 2018) although these have not been comprehensively compiled and evaluated (Manchester and Bullock, 2000). For example, the invasive *Rhododendron ponticum (*possibly *Rhododendron x superponticum,* Cullen, 2011) has been found to cause dramatic declines in lower plant and fungi diversity in highly ecologically important Atlantic oak-woods in Scotland (Long and Williams, 2007); while the America mink (*Neovison vison*) has caused important declines in water vole and ground nesting bird populations (MacDonald and Harrington, 2003). In addition to environmental impacts, these species cost the GB economy approximately £1.7bn per annum, although the actual figure is likely to be higher given indirect economic impacts (Williams et al., 2010). Health impacts also occur in GB, for example non-native deer species are responsible for numerous road traffic accidents that have caused injury and death (Langbein, 2007) and some species predicted to be likely future invaders may be serious health threats (e.g. Roy et al., 2009).

The extensive impacts caused by invasive non-native species and their rapid increase in numbers worldwide has resulted in urgent calls for them to be managed (Genovesi and Shine, 2004, Hulme, 2006, UNEP, 2011, Pino de Carpio et al., 2013, Bonnaveira, 2017, Carboneras et al., 2018, Roy et al., 2018b). However, management can be expensive, with costs for managing individual species frequently in the tens of thousands and often millions of pounds (Robertson et al., 2017). Examples from GB include considerable costs (GBP) to eradicate coypu (£6.14M) and muskrat (£3.13M), remove mink from the Uists (£6.17M) and black rats from the Shiant islands in the Hebrides (£1.12M), continue to eradicate ruddy duck (£5.79M) and develop a biocontrol agent for Japanese Knotweed (in excess of £1M) (cost sources: coypu, muskrat and mink corrected for inflation from Baker (1990), black rat (EC, 2019), ruddy duck (Iain Henderson 2019, *pers. comm*.), Japanese knotweed extrapolated from Williams et al. (2010) over 15 years). Even small scale eradications can be expensive, for example the removal of twelve Himalayan porcupine individuals from Devon cost ca. £174,000 (corrected for inflation from Baker, 1990). Failed eradications can be particularly expensive in terms of cost-benefit. For example, an attempt to eradicate the only known population of *Didemnum vexllum* (a highly invasive marine sea-squirt) from Wales cost in excess of £800,000 (GBP), but was unsuccessful with the population quickly re-establishing after management (Sambrook et al., 2014).

Despite the large resources needed for invasive non-native species management, funds are limited and often small in comparison to the scale of the threat. For example, despite the €12.5bn annual cost of invasive non-native species to the European Union economy, the European Commission's funding of invasive non-native management over a 15 year period totalled only ca. €132M (EURO) (an average of €8.8M per year), although this figure does not include the expenditure of individual Member States (Scalera, 2009). In GB, resources spent on invasive non-native species management were estimated to be only approximately

0.4% of the total biosecurity budget (i.e. including biosecurity funding for animal health, plant health, fish health, bee health and invasive non-native species) and approximately 2.3% of the total biodiversity expenditure (Moore, 2018). Given limited budgets and high costs, management efforts must be prioritised (CBD, 1992, Kumschick et al., 2015b, McGeoch et al., 2016, Scalera et al., 2016). This must be done carefully to ensure resource allocation is cost-effective, efficient and to avoid lost causes (Cassey et al., 2018b, Courtois et al., 2018).

Authors have emphasised the need to prioritise the management of non-native species, pathways and sites (McGeoch et al., 2016); however, most focus has been on the first two of these (UNEP, 2011, Scalera et al., 2016, Carboneras et al., 2018). Prioritising species and pathways is complex, not least because of the large numbers of both involved, the wide range of possible management options, considerable uncertainties and conflicting pressures faced by decision-makers (Woodford et al., 2016). Managers must decide, in the face of considerable uncertainty (Liu et al., 2011b, Moon et al., 2017), where to allocate resources between preventing the introduction of new species, eradicating emerging species and managing those that are already widespread (Wittenberg and Cock, 2001, McGeoch et al., 2016). The most cost-effective management is generally to prevent invasions occurring by managing introduction pathways and enhancing biosecurity (Mack et al., 2000, Leung et al., 2002, Keller et al., 2008, Epanchin-Niell and Liebhold, 2015) or, failing that, to rapidly detect and eradicate newly establishing species (Myers et al., 2000, Veitch and Clout, 2002, Simberloff, 2003a, Vander Zanden et al., 2010, Jones et al., 2016). However, deciding which introduction pathways and species to manage is not a trivial task. It requires a complex assessment of the risks, costs and benefits of invasion and management, as well as the trade-off between different management approaches (McGeoch et al., 2016, Epanchin-Niell, 2017). A further problem is that, while management focussed on prevention and eradication will usually be more cost-effective, decision-makers are often under considerable pressure to divert resources towards the management of widespread species (Woodford et al., 2016, Brancatelli and Zalba, 2018). This dilemma is caused by the impacts of widespread species being more apparent and immediate, while those of new or emerging species are less apparent, less certain and usually only emerge years after the initial invasion (Brancatelli and Zalba, 2018). Evidence-based methods are therefore needed to support the prioritisation of management that take into account not only the risk posed by species and pathways, but the feasibility of their management (D'hondt et al., 2015, Booy et al., 2017, Tollington et al., 2017, Vanderhoeven et al., 2017, Roy et al., 2018b). These need to be generalizable and practical, such that they can be applied to large numbers of species, pathways and contexts, as well as able to deal with limited data and uncertainty (Andersen et al., 2004, Hulme et al., 2009, Nentwig et al., 2010, Roy et al., 2018b, Vilà et al., 2018). Such methods should support the efficient use of limited resources and provide justification to help gain public, political and stakeholder support for management despite uncertainty (Shine, 2007, Cook and Fraser, 2008, Leung et al., 2012).

A range of methods have been developed to support the prioritisation of non-native species for management (Heikkilä, 2011, Scalera et al., 2016). Risk analysis is one of the main methods (Roy et al., 2018b) and has long been used to assess other biosecurity threats (i.e. plant and animal health pests and diseases, FAO, 1995, OiE, 2017). It usually comprises (at least) risk assessment, risk management and risk communication and is designed to help manage uncertainty (FAO, 1995, Lodge et al., 2016, OiE, 2017, Vanderhoeven et al., 2017). However, in relation to non-native species risk assessment has received considerably more attention than other components of risk analysis (Heikkilä, 2011, Booy et al., 2017). A large number of different invasive non-native species risk assessment methods have been developed (reviews in Essl et al., 2011b, Roy et al., 2018b). In some cases, separate tools have been developed to support the assessment of specific risk assessment components, for example climate matching tools to support assessments of establishment (e.g. Thuiller et al., 2005, Poutsma et al., 2008) and specific methods for impact assessment (e.g. Hawkins et al., 2015, Nentwig et al., 2016, Bacher et al., 2017). A special case of risk assessment is horizon scanning, which often uses a rapid (or shortened) form of risk assessment to identify species for their potential to become future invaders in a given region (Roy et al., 2014b). In contrast, while a number of methods have been developed that consider aspects of risk management for invasive non-native species (e.g. Vander Zanden et al., 2010, Darin et al., 2011, Drolet et al., 2014, Schmiedel et al., 2016, Courtois et al., 2018), this component of risk analysis has received considerably less attention (Heikkilä, 2011). As a result, practical methods to evaluate the feasibility of management are generally lacking (D'hondt et al., 2015), yet this information is essential for decision-makers and needed to support the prioritisation of management (Simberloff, 2003b, Dana et al., 2014, Kerr et al., 2016, Lodge et al., 2016, Epanchin-Niell, 2017).

In relation to prioritising introduction pathways, perhaps the most important recent advance has been the development of a consistent pathway classification scheme, which has been adopted internationally (CBD, 2014a). This provides a means of consistently collecting and comparing data on introduction pathways (Harrower et al., 2018a). However, methods to prioritise the management of pathways based on such a classification are still in the early stages of development (Essl et al., 2015, McGeoch et al., 2016). A number of approaches have been used, for example based on numbers of species introduced, vector analysis and pathway risk analysis (e.g. Carlton and Ruiz, 2005, Copp et al., 2010, Leung et al., 2014, Brancatelli and Zalba, 2018); however, work is required to develop methods that can be practically applied and clearly linked to management objectives (Essl et al., 2015).

While international commitments have been made to halt or slow the impacts of invasive non-native species (e.g. CBD, 1992), progress on implementing management has so far been slow (Early et al., 2016) and has had little apparent effect on the numbers of species establishing (Seebens et al., 2017). There is broad consensus about what needs to be done: prevent new incursions, detect and eradicate those that get through, and reduce the impacts of widespread species where eradication is not feasible (Lodge et al., 2016). However, practical methods to prioritise the management of specific species and pathways actions are lacking (Hulme et al., 2009). Such methods are urgently needed given the complexities and uncertainty involved and must consider not only the severity of the threat from invasive non-native species, but what can feasibly be done. The need for methods to support the prioritisation of management are recognised in GB (Defra, 2015), which already benefits from a comprehensive risk assessment scheme (Mumford et al., 2010), horizon scanning (Roy et al., 2014b) and extensive data on all established non-native species (Roy et al., 2014c). These resources provide an excellent, and potentially unique, foundation from which to develop and test further methods to support the prioritisation of management.

1.1 Thesis aim

The overall aim of this thesis is to develop invasive non-native species and pathway prioritisation methods that can be used to support management in different contexts and at different scales, building on existing tools where they are already available. The intention is that these methods will not only support the prioritisation of strategic management in GB, but contribute to international efforts to prioritise management in response to EU legislation (EU, 2014b) and global biodiversity targets (CBD, 1992).

1.2 Thesis outline

Given that the majority of non-native species cause little, if any, impact, an important starting point for the prioritisation of management was to be able to identify which species caused the most severe impacts. While there was already a comprehensive database detailing the established non-native species in GB (Roy et al., 2014c), a robust evaluation of the impacts of these species was lacking. Chapter 2 therefore explores a method for evaluating the environmental (biodiversity) impacts of all established non-native species in GB. The application of this method provides the basis for the ranking of introduction pathways (Chapter 3). It also provides an opportunity to explore patterns and trends in the types of species that cause impacts in GB over time.

Prevention is considered one of the most cost-effective forms of invasive non-native species management. Chapter 3 therefore considers methods to support the prioritisation of non-native species pathway management using the impact data derived from chapter two. Pathways of introduction of all established non-native species in GB are identified and used, in combination with impacts data, to test different methods for ranking pathways in order of importance. The advantages and disadvantages of different ranking approaches are explored, as well as the implications for future pathway analysis and management.

Early detection and rapid eradication is one of the most effective forms of management after prevention; however, methods to support the prioritisation of eradication are largely lacking. Chapter 4 therefore explores a novel method for prioritising the eradication of invasive non-native species that takes into account not only the risk posed by species, but also the feasibility of their eradication. While eradication is the focus of this chapter, implications for the prioritisation of long term management as well as species specific prevention are also considered.

For prioritisation methods to be most useful they need to be widely applicable across taxa and at different scales. The prioritisation method developed in Chapter 4 is therefore applied, in Chapter 5, at a larger scale (to the European Union) and across a larger set of taxa. This chapter explores the value and challenges of applying the scheme at this scale.

The final chapter of this thesis (Chapter 6) brings together this research to consider how these methods combine within a wider invasive non-native species management prioritisation framework. The existing components of such a framework are discussed, as well as gaps and areas for further work. This is used to highlight the importance of systematic research into prioritisation and management methods; address the limitations of data availability and uncertainty; consider trends and patterns in invasion and management data; and, consider the implications for future work and policy direction in GB and further afield.

Chapter 2. Comprehensive biodiversity impact scores reveal taxonomic, environmental, geographic and temporal patterns of invasiveness in Great Britain

2.1 Introduction

Tens of thousands of non-native species have established worldwide (Seebens et al., 2018) without signs of saturation (Seebens et al., 2017); however, only a small proportion become invasive (Williamson and Fitter, 1996, Blackburn et al., 2014). Given the consequences of invasions (e.g. Butchart et al., 2010, Stigall, 2012, Bellard et al., 2016, Gallardo et al., 2016, Catford et al., 2018), a major research priority in invasion biology has been to identify the proportion of non-native species that become invasive in order to support the prioritisation of their management (Lodge et al., 2016, Cassey et al., 2018b). This is necessary given limited resources (Early et al., 2016, McGeoch et al., 2016) and to support precautionary yet proportionate action that reduces the impacts of invasive non-native species while allowing the legitimate use of more benign species that provide economic and societal benefits (Shine et al., 2000, Roy et al., 2018b).

Impacts caused by invasive non-native species can include economic and societal harm (Bacher et al., 2017); however, ecologists have often focussed on assessing their environmental, or more specifically biodiversity, impacts (e.g. Genovesi et al., 2015, Evans et al., 2016, Lavoie, 2017, Dueñas et al., 2018). These occur at different levels of ecological organisation, ranging from reducing the fitness of individual organisms (e.g. Brightwell and Silverman, 2010) to causing the global extinction of species (e.g. Wiles et al., 2003); and are caused by a range of mechanisms, including competition with native species, direct predation, the transmission of disease, hybridisation, poisoning / toxicity, biofouling, herbivory, grazing, browsing, chemical / physical / structural change and interactions with other species (Nentwig et al., 2010, Hawkins et al., 2017).

Much research has investigated what causes some non-native species to become invasive (have strong impacts) and, by extension, what proportion of non-native species are likely to become invasive (Rejmanek and Richardson, 1996, Williamson and Fitter, 1996, Simberloff et al., 2013, Lodge et al., 2016). This has benefited from an increasingly detailed understanding of the sequence of stages and barriers that define the invasion process

(Blackburn et al., 2011, Cassey et al., 2018b). Good progress has been made to understand the factors that influence the initial stages of this process, such as transport, establishment and spread (Lockwood et al., 2009, Lockwood et al., 2013, Cassey et al., 2018a). However, understanding of the latter stages, where established non-native species go on to exert strong impacts, is less well developed (Ricciardi et al., 2013).

Early studies suggested that approximately 10% of introduced non-native species would make their way into the wild, approximately 10% of those would establish and, finally, 10% of established non-native species would become invasive, known as the 'tens rule' (Williamson and Fitter, 1996, Williamson, 1999). However, over subsequent years the tens rule has been found not to hold true in many cases (Jeschke and Pysek, 2018), with its final stages in particular (i.e. the number of established non-native species that go on to cause impacts) considered to be underestimated (e.g. Jeschke and Strayer, 2005, Jarić and Cvijanović, 2012). Beyond the tens rule a number of hypotheses have been developed to explain the link between the impacts of a non-native species and variables such as species traits, ecosystem traits, trophic position, the presence or absence of natural enemies and phenotypic dissimilarity (reviews in Ricciardi et al., 2013, Jeschke and Heger, 2018a). Jeschke and Heger (2018b) reviewed 12 major testable invasion hypotheses using a hierarchy of hypotheses approach and found that support for most was mixed, often specific to particular taxa at a particular scale and in some cases declining (see also Jeschke et al., 2012). Nevertheless, six hypotheses were broadly supported by the evidence (shifting defence hypothesis, limiting similarity, plasticity hypothesis, invasional meltdown, disturbance, propagule pressure). Ricciardi et al. (2013) reviewed nineteen hypotheses that specifically attempted to explain variation in impact (as opposed to other stages of the invasion process) and found that each could explain at least some impact in some situations, despite poor validation. Examinations of the link between species traits and invasiveness occupies a large proportion of the literature, but Ricciardi et al. (2013) found evidence to support a link between species traits and impact was weak compared to other aspects of the invasion process (i.e. introduced, establishment and spread). Nevertheless, traits have been shown to correlate with impact in some cases, often in relation to specific taxa at difference scales (Keller and Drake, 2009, Pyšek et al., 2012, Yessoufou et al., 2014, Dawson et al., 2015, Gallagher et al., 2015, Pyšek et al., 2017).

While species life-history traits have been extensively investigated to predict invasiveness, it may be useful to consider whether a broader combination of variables (e.g. the environment in which a species occurs, its functional group and native origin) could contribute to predictions of non-native species impacts. For example, non-native species from more distant native origins may be less phylogenetically similar to native species in the invaded range and could, therefore, be expected to cause greater impacts (following Darwin's naturalisation hypothesis, Cadotte et al., 2018, Jeschke and Erhard, 2018). In terms of environmental differences, freshwater non-native species may be expected to cause greater impacts given that receiving freshwater ecosystems appear to be particularly vulnerable to invasion, possibly because they are isolated ecosystems similar to islands (Cox and Lima, 2006, Moorhouse and Macdonald, 2015).

A question that has received little attention to date is whether the proportion of non-native species that cause strong impacts (i.e. become invasive) is changing over time. The rapid increase in the accumulation of non-native species worldwide is well documented and accompanied by strong concerns about an equally rapid increase in the accumulation of invasive non-native species (Seebens et al., 2017, Seebens et al., 2018). However, the assumption that invasive non-native species accumulate at the same rate as non-native species has been largely untested and there are some grounds to suspect the two rates may differ. For example, the frequency, distance and types of trade, transport and travel that introduce non-native species around the world have changed dramatically over the past 200 years and particularly in the past 50 years (Hulme, 2009, Essl et al., 2011a, Essl et al., 2015, Seebens et al., 2015, van Kleunen et al., 2015, Dawson et al., 2017, Seebens et al., 2018). With major changes in these pathways it is feasible that the proportion of non-native species being introduced that case impacts could also be changing. This might be the case if, for example, more modern pathways have introduced species from further afield (which may be less phylogenetically related) or that are associated with other traits or variables that predicate impact (e.g. Pergl et al., 2017). If the proportion of non-native species that become invasive is not consistent over time, this could have important implications for how we interpret the threat from the increasing establishment of non-native species worldwide.

In order to explore these patterns and hypotheses, large datasets are required to assess not only the number of non-native species that have established over time, but also to evaluate their impacts (Lodge et al., 2016, Saul et al., 2017). Such assessments must be consistent and comparable, despite substantial variation in the types and severity of impacts that occur (Simberloff et al., 2013, Kumschick et al., 2015a, Roy et al., 2018b). Definitions of impact and methods for evaluation have been barriers on this front; however, good progress has been made to both better define impacts (Parker, 1999, Jeschke et al., 2014, Kumschick et al., 2015b, Kumschick et al., 2018) and develop generic methods for evaluating impacts across taxa (Blackburn et al., 2014, Hawkins et al., 2015, Kumschick et al., 2015a, Nentwig et al., 2016, Rumlerová et al., 2016, Bacher et al., 2017, Turbé et al., 2017, Roy et al., 2018b). Most recently this has included definitions aligned directly with levels of ecological organisation from the individual to community level (Blackburn et al., 2014, Evans et al., 2016, Hawkins et al., 2017, Kumschick et al., 2017).

A further problem is that for most non-native species there has been little research to explore impact and, even where there has, studies are rarely based on robust experimental trials (Parker, 1999, Pyšek et al., 2012, Hulme et al., 2013, Ricciardi et al., 2013, Roy et al., 2018b). The problem of limited data is not unique to invasion biology and is common in the field of conservation (Martin et al., 2012). Expert information is increasingly used to overcome this problem (e.g. Baker et al., 2008, Essl et al., 2011b, Ricciardi et al., 2017, Vanderhoeven et al., 2017, Roy et al., 2018a), ideally using structured elicitation techniques to reduce aspects of bias that can limit its use (Sutherland and Burgman, 2015). Expert elicited data does not replace more empirical data, but can be used to support analysis where these data are lacking (Roy et al., 2018b). It also provides a useful means of identifying where further research to gather additional evidence would be most useful.

Utilising the expertise of a wide range of invasive non-native species ecologists, this study set out to score the environmental (biodiversity) impact of all established non-native species in Great Britain (GB), benefiting from an existing database of non-native species (Roy et al., 2014c), recent advances in the scoring of impact (Hawkins et al., 2015) and application of expert judgement that also incorporates available evidence (Roy et al., 2014b). While many have considered patterns in non-native species within a given region (e.g. DAISIE, 2009) or invasiveness within a subset of non-native species (e.g. Kumschick et al., 2015a, Cameron et al., 2016, Evans et al., 2016, Measey et al., 2016, Rumlerová et al., 2016), there are few large scale assessments that include a complete dataset for a given region of all established non-native species and their impacts. This study therefore provides a novel opportunity to investigate and compare trends in the numbers and proportions of invasive non-native species across groups within a given region. A number of questions based on existing hypotheses in invasion biology are explored:

i. Do established non-native species in GB conform to the ‘tens’ rule?

ii. Do species native to Europe cause less severe impacts than those native to other regions? Following Darwin’s naturalisation hypothesis (Darwin, 1859) it is possible, given the close evolutionary history of British and European flora and fauna, that non-native species with native origins in Europe may cause less severe impacts than species native to other regions. However, the opposite may be true given that species native to Europe may be more suited to the climate and habitats found in Britain and therefore have a competitive advantage that other non-native species do not (Cadotte et al., 2018).

iii. Have higher impact (invasive) species accumulated over time at the same rate as lower impact (non-invasive) species? Assuming that the proportion of non-native species that cause impacts has remained consistent over time, no difference in the two rates would be expected.

iv. Has the taxonomic or environmental composition, or native origin, of species changed over time? Differences over time might be expected as pathways of introduction have changed, for example leading to species from more distant origins being introduced more frequently.

2.2 Methods

2.2.1 Species selection and screening

A list of all established non-native species in GB (n=1954) was extracted from the GB-NNSIP register (Roy et al., 2014c) on 3rd December 2015. This included all non-native species with self-sustaining populations in GB (Roy et al., 2014c), excluding microorganisms, parasites, parasitoides and macrofungi which were not comprehensively covered. Additional species metadata was also extracted from the NNSIP database, including: broad taxonomic group (plant, invertebrate or vertebrate); informal taxonomic group (bird, mammal, herptile, fish, insect, non-insect invertebrate, higher plant, lower plant); environment (freshwater, marine, terrestrial); functional group (predator, herbivore, omnivore, detritivore, filter feeder, parasite, land plant, algae), continent of native origin (Africa, Asian-temperate, Asia-tropical, Australasia, Europe, North America, South America, Pacific) and year of first record in the wild. Where a species spanned more than one continent of native origin, the nearest continent to GB was given. Where possible a single environment was allocated to each species: all amphibians were considered freshwater; all coastal plants were considered terrestrial; and all waterfowl were considered terrestrial. Only two species could not be classified into a single environment: Chinese mitten crab (freshwater and marine) and *Pseudamphistomum truncatum* (terrestrial and freshwater). Continent of native origin data was not available (missing from the NNSIP database) for 194 species. A further 99 species did not have a native origin as they were created for cultivation (predominantly plant species).

The full list of established non-native species in GB was screened to provide a subset of species with the potential to cause more than a 'minimal' biodiversity impact (see Table 2.1), or for which potential impact was uncertain and required further consideration. This screening was based on species flagged in the NNSIP database as having environmental impacts (n=190) and was augmented by cross checking other existing lists of invasive non-native species in GB (Wildlife and Countryside Act, 1981, Parrott et al., 2009, Thomas, 2010, UKTAG, 2015, Booy et al., 2015, GBNNSS, 2019) and consulting the experts involved in this study (Appendix A). Experts were guided to add any additional species considered likely to have the potential to cause more than minimal impacts or for which potential impact was uncertain. The result of this process was a final screened list of 238 species to be subjected to more detailed scoring. Species screened out at this stage (n=1716) were scored as minimal impact.

2.2.2 Criteria for scoring impact

Species were scored according to their ability to cause biodiversity impacts only. Current and maximum impact was scored on a five-point scale derived and modified slightly from EICAT (Table 2.1). Current impact was defined as the impact to date based on the species current distribution in GB. Maximum impact was defined as the impact that would be expected if the species were established in all suitable parts of GB (based on current biotic and abiotic conditions). All scores were accompanied by a written comment, citing relevant literature where available, and experts indicated whether the evidence used to support their score was: field observation, experiment, modelling, expert opinion and / or not from GB. Type of impact was recorded, separated into impacts on: species or habitats of conservation concern, species or habitats not of conservation concern, and /or ecosystem function. Impact mechanism (following Hawkins et al 2014) was also recorded separated into mechanisms that impact species, habitats and ecosystems. The full scoring guidance is available as supplementary information (Appendix B).

Table 2.1 Impact scoring definitions (modified from Hawkins et al. (2015)). Modifications to original definitions are underlined.

Minimal	Unlikely to have caused deleterious impacts on the native biota or abiotic environment.
Minor	Causes reductions in the fitness of individuals in the native biota, but no declines in native population sizes, and has no impacts that would cause it to be classified in a higher impact category.
Moderate	Causes declines in the population size of native species, but no changes to the structure of communities or to the abiotic or biotic composition of ecosystems, and has no impacts that would cause it to be classified in a higher impact category.
Major	Causes the local or population extinction of at least one native species, <u>and / or*</u> leads to <u>substantial but*</u> reversible changes in the structure of communities and the abiotic or biotic composition of ecosystems, and has no impacts that cause it to be classified in the MV impact category.
Massive	Leads to the replacement and local extinction of native species, and produces irreversible changes in the structure of communities and the abiotic or biotic composition of ecosystems.

2.2.3 Expert elicitation and consensus building

Scoring based on the criteria defined above was carried out by experts with experience in the invasion biology of the given species. In total, 36 different experts provided scores, separated into five groups based on taxonomic expertise: freshwater animals, terrestrial vertebrates, terrestrial invertebrates, marine species and plants (excluding marine plants). Each group comprised 5-8 members, with membership determined by the organisers in cooperation with an appointed group leader.

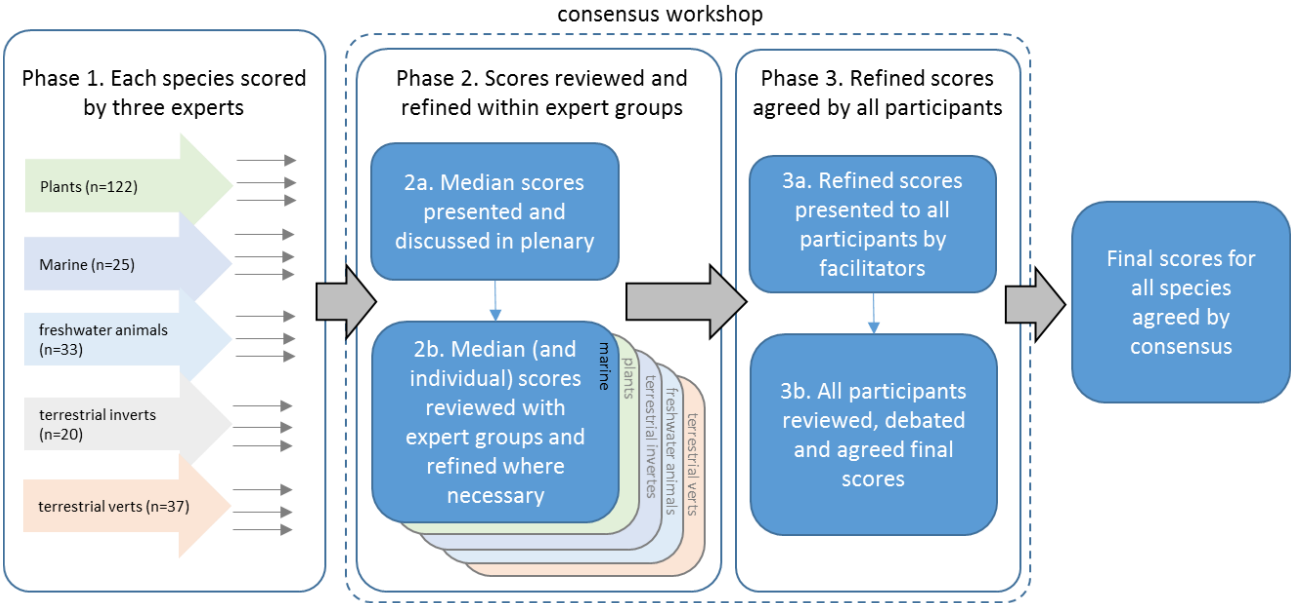


Figure 2.1 Overview of the GB impact scoring expert elicitation process. Species were divided into five groups based on taxonomy and environment (starting arrows, number of species in each group is indicated). In phase 1, each species was scored independently by at least 3 experts. Scores were collated and used as the basis for the consensus workshop. This started with phase 2, where median scores were presented and discussed by all participants (2a) followed by an opportunity for each expert group to review median and individual scores and make changes based on discussions (2b). In phase 3, refined scores were collated and presented to all participants in plenary (3a) who were invited, through facilitator led discussions, to review and make final changes (3b). The overall aim was to produce final scores agreed by the consensus of the group.

Scores were elicited through three distinct phases (Fig 2.1):

- Phase 1. At least three experts independently scored each species. Experts were guided to score current and maximum impact, indicate confidence in both scores, provide justification for both the response and confidence scores (with reference to literature where available) and complete additional fields (i.e. type of evidence, mechanism, type of impact). All scores were then collated anonymously and the median scores of current and maximum impact, as well as confidence scores, for each species calculated. These were circulated to expert groups, along with the individual scores and justifications. Additional data on type of evidence, type of impact, impact mechanism, etc. was also circulated; however, this was not subject to further review.

The next two phases took part during the consensus building workshop (27-28 May 2016) attended by 25 of the original experts, including all group leaders.

- Phase 2. Training was provided to group leaders to ensure consistency in application of guidance. Group leaders and organisers worked through the scoring criteria, including examples from each group, and to ensure there was consistent understanding across expert groups. All participants then met in plenary and group leaders presented median scores from phase one to all participants. Participants were encouraged to discuss and challenge scores and, in particular, to ensure there was consistent understanding and interpretation of the scoring guidance. Then all participants split into their original expert groups to review median scores and refine them, where necessary, in the light of the plenary discussion.

- Phase 3. The final stage of the scoring process was to agree the refined scores by consensus of all participants. All refined scores for each species were collated and presented back in plenary to all participants of the consensus building workshop by two facilitators (HR and OB). Participants were encouraged to review, discuss and challenge the scores, with final modifications made by the agreement of all participants.

2.3 Analysis

The maximum impact scores were used for analysis as these were most likely to be relevant when assessing species future impacts. Where practical, the five levels of impact (minimal to massive) were used for analysis. However, where it was necessary to distinguish between invasive species and 'non-invasive' species, species scoring more than minimal impacts were considered invasive and those scoring minimal impacts 'non-invasive'.

All coding was undertaken in R version 3.4.1 (R Core Team, 2017) primarily using the tidyr package (Wickham and Henry, 2018).

2.3.1 Taxa, environment, functional group, native origin and impact

To determine which species traits (taxa, environment, functional group and native origin) were important for predicting impact, the Random Forest algorithm (Breiman, 2001) was coupled with feature selection in the R package Boruta (Kursa and Rudnicki, 2010). The

Random Forest algorithm was used as it provides a powerful means of analysing data with multiple categorical and ordinal predictors and outcomes, while feature selection in Boruta provided a wrapper to identify statistically important features (variables) involved in the prediction of impact.

Feature selection in Boruta used repeated measures of variable importance (derived from Random Forest) to identify variables (features) that have significantly more predictive power than randomly permuted 'shadow' variables (Kursa, 2018). Variables that significantly outperformed the best 'shadow' variable were confirmed as important, while those that underperformed were rejected. Note that variable importance measured loss of accuracy of classification caused by random permutation of variable values between objects and was assessed separately for all trees in a forest that used a given variable for classification (Kursa and Rudnicki, 2010). In Boruta feature selection, variable importance was then expressed as z-scores - the average loss of accuracy of classification for all trees which use a variable for classification, divided by its standard deviation.

Before analysis with Random Forests was undertaken, it was necessary to balance the dataset as there were many more species scored as minimal impact than any other impact category. This type of class imbalance can be problematic when attempting to predict minority classes (i.e. minor to massive impact) using Random Forests, as the algorithm works to reduce overall error rate and therefore tends to focus on predicting the majority class. Data were therefore balanced using the package UBL (Branco et al., 2016) to randomly over-sample the least populated classes (i.e. minor to massive impacts). The importance of classes and their respective over-sampling percentages was calculated automatically using two different strategies 'balance' (which balances the frequency of all classes) and 'extreme' (which inverts the frequency of classes). The strategy that resulted in the smallest error when confusion matrices were compared was then chosen for analysis (Branco et al., 2016).

For this part of the analysis, records were removed where continent of native origin was 'unknown' or given as 'cultivated', leaving n=1690 records.

2.3.2 Changes in the number of invasive and 'non-invasive' species overtime

To explore whether invasive non-native species (i.e. those that scored more than minimal impact) accumulated at a similar rate to 'non-invasive' species (those that scored minimal impact) the number of species in each group that established over time was modelled. Year of first record from the NNSIP database was used as a proxy for establishment date and aggregated into twenty year periods (e.g. 1981-2000). Ten and fifty year periods were also explored; however, twenty years provided a better compromise between detail (showing trends in the data) and grouping (to allow for analysis). Records after 2000 (n= 60) were excluded to reduce potential bias due to lag in identification and reporting of species (similar to the approach of Seebens et al. (2017)). A GLM was fitted to predict the number of species (log +1) establishing in each 20 year period and to determine if there were differences between invasive and 'non-invasive' groups. The model was Gaussian. The response variable was number of species and the explanatory variables were year of establishment in the wild (based on 20 year periods) and whether invasive (> minimal impact) or not.

2.3.3 Changes in taxa, environment and native origin over time

Changes in the taxonomic and environmental composition of established species, as well as continent of native origin, over time were explored using the first record in the wild for each species grouped in 20 year time bins. Both changes in the number and proportion of species establishing from each group were considered over time.

2.4 Results

2.4.1 Overall proportion of species with impacts in GB

Impact scores were reached by the consensus of experts for all 238 species included in the expert elicitation and consensus building process. These were added to the species already identified as being of minimal concern by screening (n=1716) to provide a comprehensive set of scores for all species established in GB. In total, out of the 1954 non-native species established in GB, 183 (9%) were invasive (i.e. scored more than minimal impact) based on current impact and 210 (11%) were invasive based on maximum impact.

2.4.2 *Characteristics of species that caused more than minimal impacts*

The variables broad taxa, environment, functional group and continent of native origin were all found to be important in determining the level of impact of a species, compared to shadow variables (Fig 2.2). A far larger proportion of established vertebrate species scored more than minimal impact (84%) than invertebrates (9%) and plants (8%) (Fig 2.3b). Within the vertebrates, similar proportions of mammals, herptiles, birds and fish had non-minimal impacts; however, a larger number of mammals had major impacts, followed by herptiles, fish and birds (Fig 2.3e). The group with the smallest proportion of species causing more than minimal impact was insects (2%), which was in contrast to non-insect invertebrates

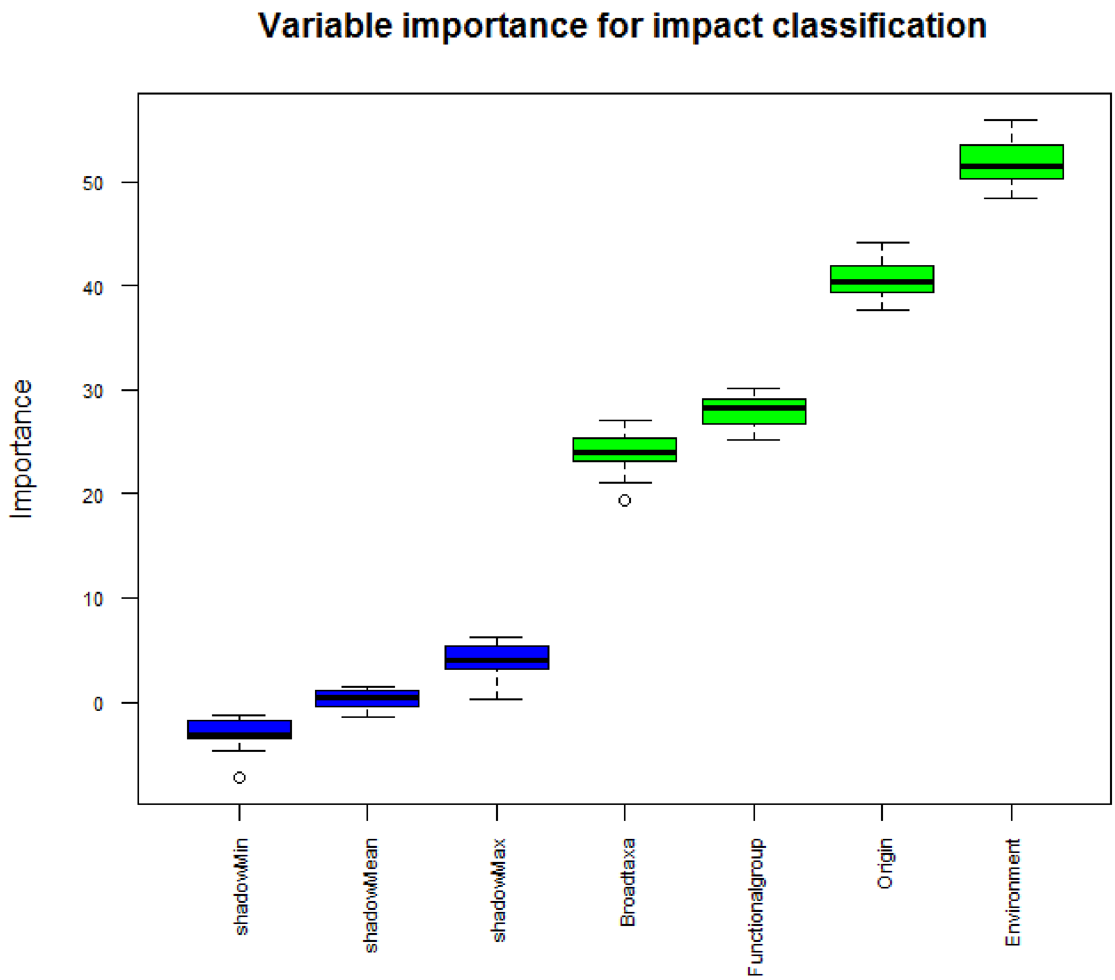


Figure 2.2 Random forest algorithm showing relative importance of each variable when determining the impact of a species. Variables that were important fell outside of the minimum and maximum shadow variables (blue boxes). In this case, all tested variables (broad taxonomic group, environmental group, continent of native origin and functional group) were important predictors of impact (green boxes).

(25%) (Fig 2.3e). Non-insect invertebrates also included the largest number of species scored as causing massive impacts (n=6). Eight percent of higher plants had non-minimal impacts, in contrast to lower plants (14%).

There were also substantial differences by environment, with a larger proportion of freshwater species (61%, n=52) scoring non-minimal impacts, compared to marine (29%, n=22) and terrestrial (7%, n=134) (Fig 2.3c). Taking taxa and environment combined, freshwater vertebrates (91%), terrestrial vertebrates (80%) and freshwater plants (65%) had the highest proportions of species causing more than minimal impacts, followed by freshwater inverts (43%) and marine inverts (33%) (Table 2.2). Terrestrial invertebrates had the lowest proportion (2%) followed by terrestrial plants (7%) and marine plants (18%) (Table 2.2).

The functional group with the largest proportion of species causing non-minimal impacts was filter feeders (37%, n=19), followed by predators (35%, n=30) and omnivores (28%, n=21) (Fig 2.3f). Herbivores, detritivores, land plants and algae had smaller proportions of species that caused non-minimal impacts (ranging from 5-16%).

A smaller proportion of species with native origin in Europe caused more than minimal impacts (8%, 69 out of 891), compared to species with native origins in the rest of the world (16%, 121 out of 776; Fig 2.3d). All continents except Antarctica and tropical Asia were associated with higher proportions of non-minimal impact species than Europe (Fig 2.3g). North America (27%) and the Pacific were sources of particularly large proportions of species that caused non-minimal impacts, although sample size for Pacific species was small (n=12). Temperate Asia (13%), South America (12%), Australasia (11%) and Africa (9%) had the next largest proportions. An unusually large proportion of terrestrial plants from North America scored non-minimal impacts (23%, n=96), compared to the next nearest group from temperate Asia (9%, n=204) and South America (9%, n=55) (Table 2.2). Fish, freshwater invertebrate and freshwater plant species from North America also comprised a high proportion of non-minimal impact species (Table 2.2). Of the terrestrial vertebrates that caused more than minimal impacts the majority were native to Europe, temperate Asia and North America, while freshwater vertebrates (i.e. fish) were native to Europe and North America (Table 2.2).

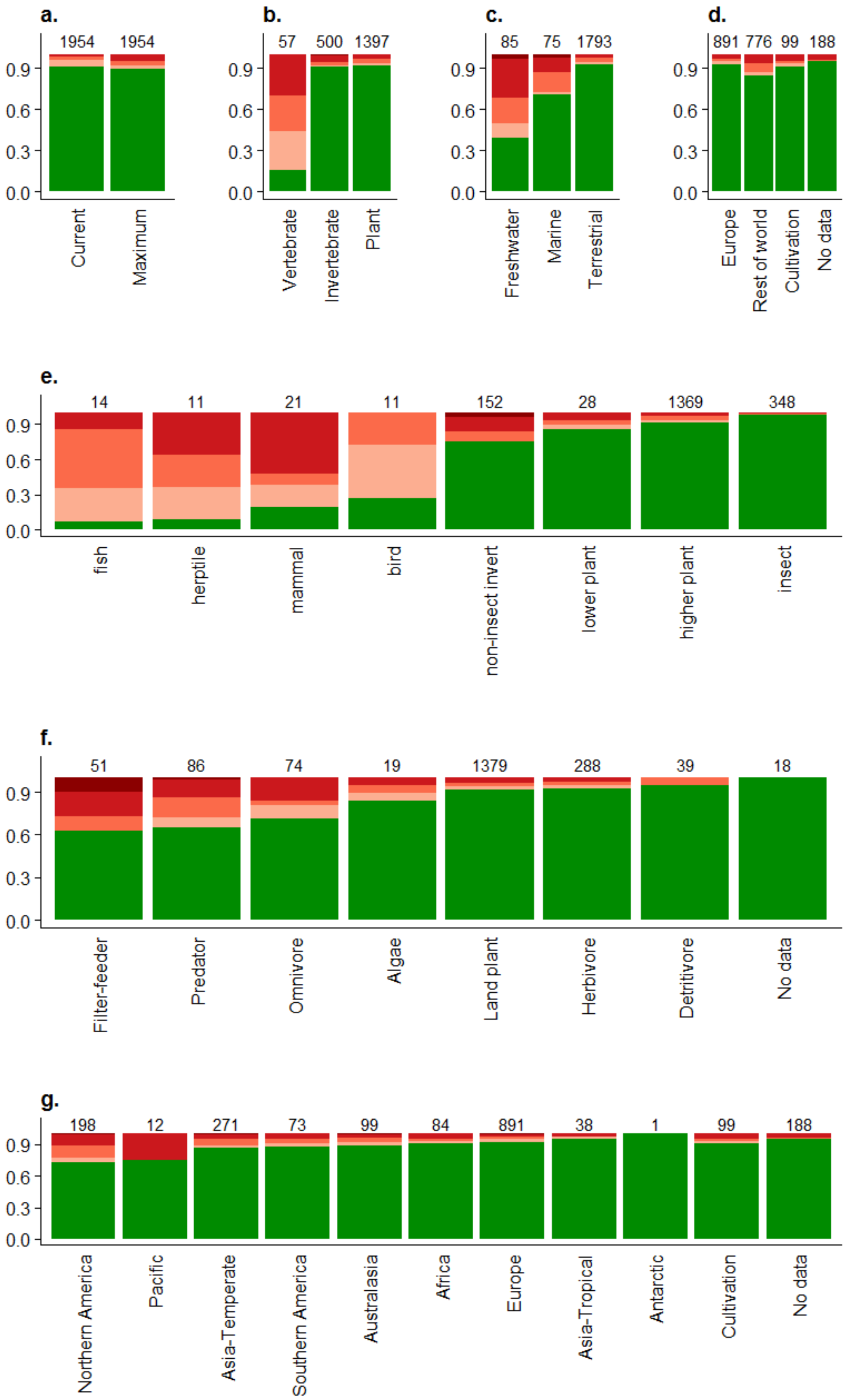


Figure 2.3 Proportion of species causing impacts in GB by (a) impact type, (b) broad taxa, (c) environment, (d) native origin (Europe vs Rest of world), (e) informal taxonomic group, (f) functional group and (g) continent of native origin. Colour indicates impact score from minimal (dark green) to massive (dark red).

Table 2.2 Proportion of species scoring more than minimal impact by broad taxa, environment and native origin (total count of all species is provided in brackets). Where total count of species <5 or there were no data (-), cells are shaded grey. The data included no marine vertebrates. Antarctica was excluded as only one (minimal impact) species was native to this continent. Am. (N) = North America; Am. (S) = South America; Asia (te) = Temperate Asia; Asia (tr) = Tropical Asia; Aust = Australasia.

	Freshwater			Marine		Terrestrial		
Native origin	**vert.**	**invert.**	**plant**	**invert.**	**plant**	**vert.**	**invert.**	**plant**
Africa	1.00 *(1)*	0.00 *(2)*	1.00 *(2)*	0.00 *(6)*	-	1.00 *(1)*	0.00 *(24)*	0.08 *(48)*
Am. (N)	0.83 *(6)*	0.44 *(16)*	0.82 *(11)*	0.27*(15)*	0.00 *(2)*	1.00 *(5)*	0.02 *(47)*	0.23 *(96)*
Am. (S)	-	-	0.67 *(3)*	-	-	1.00 *(2)*	0.00 *(13)*	0.09 *(55)*
Asia (te)	1.00 *(3)*	0.00 *(3)*	0.50 *(2)*	0.44 *(9)*	0.33 *(6)*	0.71 *(7)*	0.03 *(37)*	0.09 *(204)*
Asia (tr)	-	0.00 *(3)*	-	0.00 *(1)*	-	1.00 *(1)*	0.00 *(8)*	0.04 *(25)*
Aust.	-	1.00 *(1)*	1.00 *(1)*	0.60 *(5)*	0.00 *(2)*	0.00 *(1)*	0.05 *(41)*	0.08 *(48)*
Europe	0.92 *(12)*	0.67 *(12)*	0.00 *(2)*	0.25 *(8)*	0.50 *(2)*	0.81 *(16)*	0.02 *(174)*	0.04 *(664)*
Pacific	-	-	-	0.43 *(7)*	0.00 *(5)*	-	-	-
overall	0.91 *(22)*	0.43 *(40)*	0.65 *(23)*	0.33 *(57)*	0.17 *(18)*	0.80 *(35)*	0.02 *(402)*	0.07 *(1356)*

2.4.3 Changes in impact through time

The observed number of all 'non-invasive' non-native species establishing through time generally increased (Fig 2.4a, blue bar), albeit with two peaks at 1900 and 1960. In contrast, the observed number of all invasive non-native species establishing in GB increased up to 1920, but decreased thereafter (Fig 2.4a, red bar). Considering only terrestrial plants (Fig 2.4b), there was an increase in 'non-invasive' plants establishing up to 1900, but considerable variation in numbers establishing thereafter. This included a peak at 1960, which likely reflect the publication of the Atlas of the British Flora (Perring and Walters, 1962, Preston et al., 2002), and smaller numbers at other times, though more 'non-invasive' non-native terrestrial plant species established in each 20 year period after 1900 than before it (Fig 2.4b). The number of invasive non-native plants increased to a peak at 1900, but then declined through the rest of the 20th century despite the peak in 'non-invasive' non-native species in 1960 (Fig 4b). Considering all taxa except terrestrial plants (Fig 2.4c), both the number of 'non-invasive' and invasive non-native species establishing in GB generally increased throughout the 19th and 20th century (albeit with a dip in 1900 followed by a small peak in 1920).

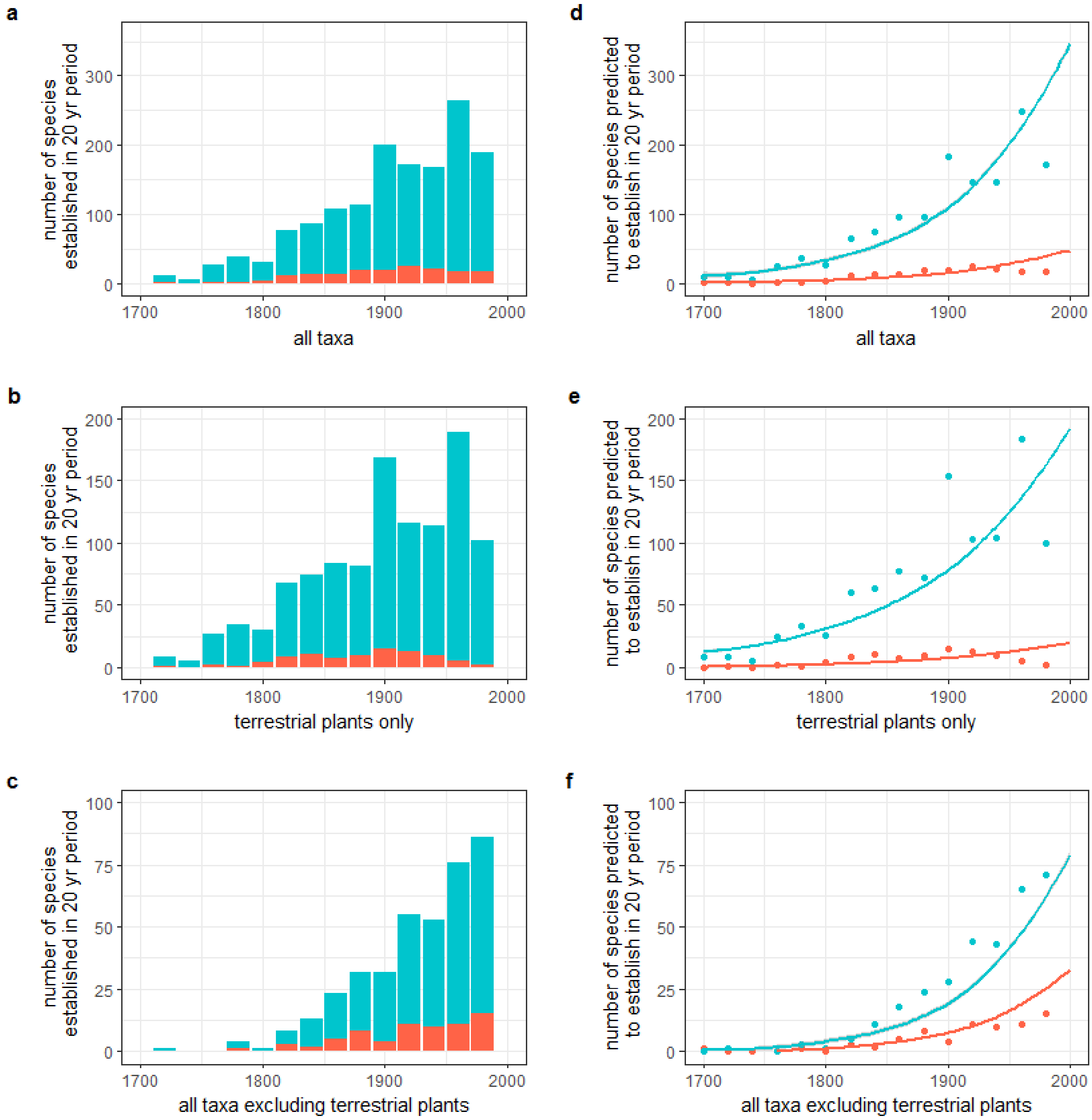


Figure 2.4 The number of invasive (red) and 'non-invasive' (blue) species that established in GB in 20 year time periods for (a / d) all species, (b / e) terrestrial plants only and (c / f) all taxa excluding terrestrial plants. GLM (log+1) predictions of the number of 'non-invasive' and invasive non-native species establishing in each 20 year period are presented (lines, plots d, e and f).

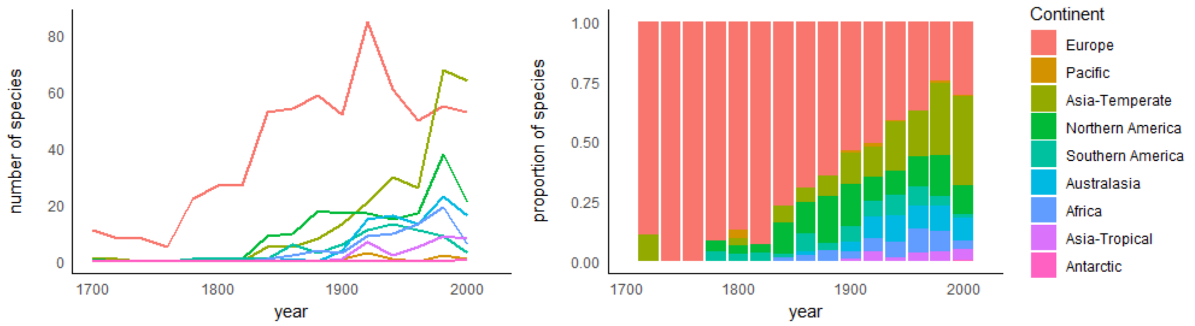
Predicting the number of invasive non-native species was challenging because the observed number in any 20 year period was low and variable. The model used a single data point for invasive and 'non-invasive' non-native species in each 20 year period and showed an interaction between year of introduction and impact status (whether invasive of not). This indicated that the rate of 'non-invasive' species establishing was significantly higher than invasive species (coefficient = 1.954, $p<0.001$). The model predicted an increase over time in the numbers of all groups (Fig 2.4d, e and f), despite the observed decrease in the number of terrestrial plants establishing (Fig 2.4b and d).

2.4.4 Changes in taxa, environment and native origin over time

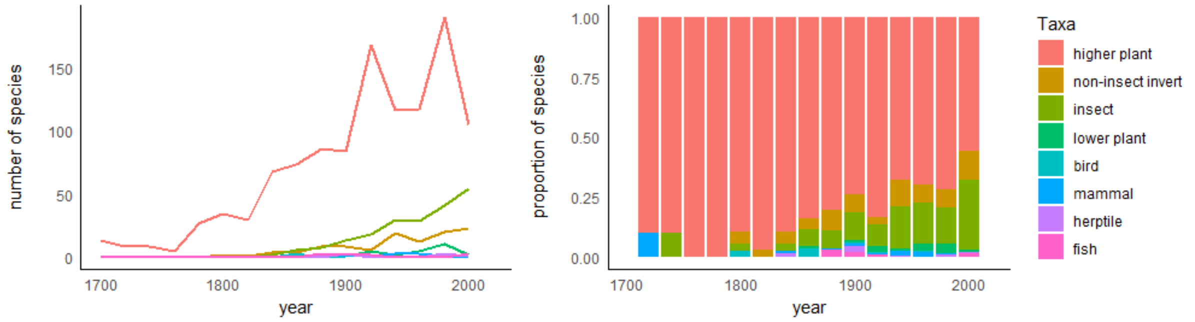
The composition of species from different taxonomic groups, environments, or native origins changed over time (Fig 2.5). The most marked changes were a rapid increase in the number of species establishing with native origins outside of Europe towards the end of the 20th Century (Fig 2.5a). However, there was also substantial increase in the proportion of invertebrates (both insects and non-insect invertebrates) established from the middle of the 1800s onwards (Fig 2.5b); and a small increase in freshwater and marine species compared to terrestrial species in the 20th century (Fig 2.5c).

While the majority of species establishing in GB had their native origins in Europe, this changed in the 1900s to a majority of species with native origins from other parts of the world. Indeed, it appears that European natives largely plateaued between 1900-2000, while species originating from outside of Europe increased rapidly. Temperate Asian species showed the most rapid increase (median first record = 1964) and overtook Europe as the main continent of native origin by the end of the 20th century (Fig 2.5c). North American species showed the next largest increase, followed by species with native origins in Australasia, Africa, Asia-tropical, South America and the Pacific. Environmental changes showed that, by comparison to terrestrial species (median year of first introduction 1913), freshwater (1961) and marine (1973) introductions were much more recent. There was a similarly recent and rapid increase in both insect (median year of first record 1968) and non-insect invertebrates (1957), compared to plants (1907) and vertebrates (1899).

a. Number and proportion of species establishing (20 year bins) by continent of native origin



b. Number and proportion of species establishing (20 year bins) by taxonomic group



c. Number and proportion of species establishing (20 year bins) by environment

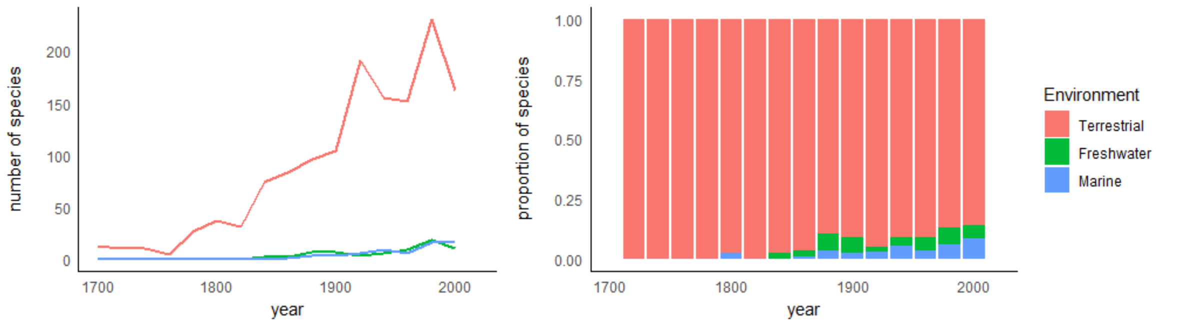


Figure 2.5 Trends over time in (a) continent of native origin (b) taxonomic composition and (c) environmental composition of established non-native species in GB based on first records. Line plots show counts of first records, stacked charts show change in the proportions of each group over time. All data are in 20 year time bins (e.g. 1981 – 2000).

2.5 Discussion

This study set out to assess biodiversity impacts of established non-native species in Great Britain (GB) and found, based on the consensus of a large group of experts, that 210 (11%) species had potential to cause more than minimal impacts. A limitation of large scale, cross-taxa studies on the impact of non-native species to date has been that standardised, empirical data on impacts are lacking, particularly given large datasets (Simberloff et al., 2013). Expert elicitation, combined with systematic approaches to defining and scoring impacts, provided a useful means of overcoming these limitations, following similar approaches used for invasive species horizon scanning (Roy et al., 2014b) and the evaluation of management feasibility (Booy et al., 2017). Expert judgment must be used carefully to reduce subjectivity and bias (Sutherland and Burgman, 2015). Clear guidance and definitions are particularly important to reduce linguistic ambiguity (Leung et al., 2012) so this approach was tailored using definitions derived from the EICAT scheme (Hawkins et al., 2015). Experts did not always agree on how to use and interpret these definitions and so it was necessary to spend some time as a group ensuring consistent interpretation and making two small modifications to the original definitions. While expert elicited results do not replace empirical data, they allow for initial analysis where data are lacking (Roy et al., 2018b). They also help to indicate where additional empirical data could be most useful, for example in this case identifying a subset of species that would benefit from further impact studies (in particular those that scored major and massive impacts but with low or medium confidence).

By updating and refining the biodiversity impact scores within the GB NNSIP this study helps to provide a dataset that is novel in a number of ways. Firstly, it provides the first systematic assessment of all established non-native species (excluding parasites, parasitoids, microorganisms and fungi) that cause negative impacts in GB. This can be used to help inform management, for example by prioritising individual species, identifying trends to inform prevention and surveillance approaches (for example trends in pathways that have introduced the most harmful species to support prevention effort) and developing indicators (e.g. Armon and Zenetos, 2015, Harrower et al., 2018b, Wilson et al., 2018). Secondly, it provides an unusually complete dataset with which to explore patterns in the invasiveness of all non-native species (across taxa and environment) within a large geographical region. While many have studied patterns in established non-native species globally (e.g. van Kleunen et al., 2015, Dawson et al., 2017, Pyšek et al., 2017, Seebens et al., 2017, Seebens et al., 2018) and regionally (Stohlgren et al., 2006, Lambdon et al., 2008, DAISIE, 2009, Roy et al., 2014c), or sampled subsets of species that cause negative impacts (e.g. Nentwig et al., 2010, Vilà et al., 2011, Pyšek et al., 2012, Nentwig and Vaes-Petignat, 2014, Iannone III et al., 2015, Evans et al., 2016, Gallardo et al., 2016, Measey et al., 2016, Galanidi and Zenetos, 2018), few datasets include all established non-native species and their impacts within a large region. By providing such a dataset, the GB NNSIP avoids sampling error (the data are not a sample but a complete set of all known species, although origin data were not available for 10% of species) and provides an opportunity not only to consider patterns across a broad range of highly different taxa, but also to consider the factors that affect invasiveness within a large geopolitical area.

2.5.1 The importance of taxa, environment and native origin for predicting impact

Many studies have considered differences in the impact of non-native species based on specific traits (e.g. Keller and Drake, 2009, van Kleunen et al., 2010, van Kleunen et al., 2011, Pyšek et al., 2012, Yessoufou et al., 2014, Dawson et al., 2015, Gallagher et al., 2015, Lodge et al., 2016, Dawson et al., 2017); however, few have comprehensively compared differences across broad taxa, functional groups, between different environments and from different native origins. All four of these variables were found to be important predictors of impact. In particular, vertebrates, aquatic species and species with native origins outside of Europe were associated with higher impacts than other groups in GB. The proportion of all established non-native species in GB that were found to be invasive (i.e. score more than minimal impact) conforms with the ‘tens rule’ (11%, which is comfortably within the bounds set by Williamson and Fitter (1996) of 5-20%); however, this masked substantial differences between taxa, functional group and environment. At a broad scale there were clear differences in the proportions of taxa that were invasive, as has been found by others (Jeschke and Pysek, 2018), with a far larger proportion of vertebrates scoring more than minimal impact than other groups. Within the invertebrates there was also a marked difference between insects and non-insect invertebrates, with the latter comprising a considerably larger proportion of species that caused more than minimal biodiversity impact. The reason for these differences cannot easily be discerned from these data and is worthy of further investigation; however, they suggest that at a broad level there were differences in the characteristics of species and their interaction with the environment that resulted in differing degrees of impact. Interestingly, a similar proportion of vertebrates were invasive across all relevant functional groups and in both freshwater and terrestrial environments (there were no marine vertebrates). This suggests that vertebrates were invasive irrespective of the environment in which they occur or the mechanism of their impact. By contrast, the environment in which invertebrates occurred was an important predictor of impact, with terrestrial invertebrates far less likely to be invasive than aquatic species. Overall, the functional group a species belonged to was also an important predictor of impact. This may be linked to trophic position (e.g. Gallardo et al., 2016, Walsh et al., 2016), given both predators and omnivores had disproportionately large proportions of invasive species. However, proportionally more filter feeders were invasive than any other functional group (and caused the most severe impacts) which suggests the role of trophic position may be more complicated and potentially connected to wider changes in ecosystem function and processes (e.g. MacIsaac, 1996). For example, filter feeding molluscs often drive ecosystem wide changes in nutrient and energy flows, as well as substantially changing the substrate (e.g. Higgins and Zanden, 2010, Herbert et al., 2016).

The likelihood of a species becoming invasive has been found by others to relate to both the characteristics of the species and the invasibility of the receiving habitat (e.g. Hui et al., 2016). Moorhouse and Macdonald (2015) argue that freshwater ecosystems are not only more likely to be invasible than terrestrial systems, but that impacts of invasive species in freshwaters are likely to be disproportionately severe. This is reflected in the results, with freshwater non-native species approximately six times more likely to be invasive than terrestrial species. The vulnerability of freshwater systems may be linked to their sensitivity to ecosystem scale changes, while also being biologically separate (or isolated, similar to island ecosystems) and susceptible to rapid secondary spread once colonised (Cox and Lima, 2006, Moorhouse and Macdonald, 2015, Thomaz et al., 2015).

Strikingly non-native species native to continents other than Europe were found to be twice as likely to be invasive than European natives; although, this was primarily because of lower levels of invasiveness in terrestrial plants native to Europe (by far the largest group within this dataset). Species in GB and continental Europe share a closer evolutionary history than species native to other continents and so this pattern could be explained by Darwin's naturalisation hypothesis, which predicts that non-native species should be more invasive where there is an absence of closely evolved native species (Daehler, 2001). This hypothesis has been challenged (Duncan and Williams, 2002, Cadotte et al., 2018) and other studies, including those on non-native plants in GB (Lim et al., 2014), have not found a link between invasiveness and phylogenetic distinctiveness. However, Jeschke and Erhard (2018) found that overall the balance of studies supported the naturalisation hypothesis when based on phylogenies rather than taxonomic groups. A further possible explanation is the enemy release hypothesis (Keane, 2002, Roy et al., 2011, Heger and Jeschke, 2014), which predicts that invasive species should benefit from an absence of enemies in their non-native range. It is plausible that the phylogenetic and geographic closeness of European non-native species in GB may make them more prone to natural enemies in GB than species native to other parts. However, more research is required to explore this further, particularly given that evidence to support the enemy release hypothesis is also mixed (Hega and Jeschke, 2018).

The increased invasiveness of species native to regions outside of Europe should be of particular concern, given that there has been a sharp increase in the number of species establishing from these regions over the past 200 years. There was a particularly rapid increase in species establishing from temperate Asia and North America, possibly reflecting major changes in trade and transport routes over this period.

2.5.2 Change in the proportion of species causing impacts over time

A question that appears to have received little attention to date is whether the proportion of non-native species that cause impacts is changing over time? Given that the numbers of non-native species establishing worldwide has increased rapidly over the past 200 years (Seebens et al., 2017), the answer to this question has important consequences for understanding potential future impact. It is possible that the ratio of invasive to 'non-invasive' (i.e. those causing minimal impact) non-native species has been a constant over time; however, the ratio could change if, for example, changes in pathways resulted in the introduction and establishment of different types of species from different native origins (e.g. Liebhold et al., 2016, Lodge et al., 2016, Zieritz et al., 2016, Dyer et al., 2017, Turbelin et al., 2017, García-Díaz et al., 2018). Using the initial scores for impact in the NNSIP database, Roy et al (2014) found that the proportion of species causing impacts in GB in fact decreased over time. With improved data this study found a similar result, showing that the number of new invasive non-native species establishing in GB increased until approximately 1920, but then decreased between 1920 and 2000; while the number of new 'non-invasive' species establishing continued to increase.

The most likely explanation for this decrease in the proportion of invasive non-native species establishing after 1920 was a lag in time between a species establishing in GB and its impacts being detected and reported (Roy et al 2014). It was hypothesised that if lag were the cause of this trend it would be most pronounced for terrestrial plants, which often have longer time lags than other species (Groves, 1999, Cunningham et al., 2003, Cunningham, 2004). This was the case, with a peak in the number of new invasive terrestrial plants establishing in GB at 1900 followed by a decline from 1900 to 2000, despite an increase in the number of new 'non-invasive' terrestrial plants establishing in the same period. This was in contrast to the model, which predicted an increase in invasive non-native terrestrial plants during this period. While the decline was prominent in terrestrial plants, it was far less pronounced in other taxa, which continued to increase in both new invasive and 'non-invasive' non-native species throughout the 20th Century, broadly in line with modelled predictions.

An alternative explanation for this decrease in invasive non-native species over time could be that changes in the taxonomic, environmental or native origin of species caused a reduction in the proportion that caused negative impacts. However, rather than contributing to a downward trend in the proportion of species causing impacts, changes in the types of species establishing were more likely to favour an increase in the proportion of invasive non-native species over time. This was principally because of the increase in the proportion of species with native origins outside of Europe, but also because of the increase in aquatics. By contrast, no other changes in environmental, taxonomic or origin data suggested a change in favour of lower impact species, with the exception of the increase in insects (only 2% of which were invasive) but this was counteracted by the increase in non-insect invertebrates (25% of which were invasive).

The confounding effect of lag in the detection of impact makes it difficult to be conclusive about how the proportion of non-native species causing impacts has changed over time. This is especially true given that terrestrial plants were by far the largest group in the dataset and the trend in this group has had an overriding, and potentially masking, effect on trend in invasiveness. However, given that the types of species establishing in GB over time is changing rapidly there is no reason to assume that the proportion of species causing impacts has remained constant. With the proportional increase in aquatics and species with native origins outside of Europe, it is plausible that the proportion of invasive non-native species could increase over time. Further work is needed to develop methods to explore this possibility that compensate for lag, possibly be modelling trends using species established before 1900 (i.e. the point where lag appears to be having little if any impact on this dataset); however, the small size of this dataset may limit the power of a such a model. Another possibility it to account for lag using data on the typical lag times of different taxonomic groups (e.g. Smith et al., 2018); however, such comprehensive data on lag is lacking.

Lag in the detection of impact in these data suggests that GB may be suffering from invasion debt (Essl et al., 2011a), in other words there are species established in GB that are currently considered benign, but will cause serious impacts in the future. This poses a problem for invasive non-native species managers who must attempt to identify and manage emerging threats (Simberloff, 2003a) and highlights the importance of the application of the precautionary principle in GB (i.e. taking action to manage potential threats even where there is considerably uncertainty) (Shine et al., 2000). As a result, the potential impact of emerging species, particularly terrestrial plants, should be closely monitored (e.g. Cunningham et al., 2003).

2.5.3 Consistency in impact assessment

A major challenge in invasion biology is to find methods to consistently score impact across taxa and environment (Ricciardi et al., 2013). Considerable steps forward have been made in this area, with the development of a range of generalizable scoring systems, including GISS and EICAT (Hawkins et al., 2015, Nentwig et al., 2016). EICAT in particular (which formed the basis of the criteria used in Chapter 2) aims to assign impact scores on the basis of change at different levels of ecological organisation (individual, population, community) that can be objectively determined and tested, therefore helping to ensure consistency between assessors (Hawkins et al., 2015). Despite this (Kumschick et al., 2017) found relatively low levels of consistency when comparing independent global impact assessments of amphibians using the EICAT system. Low levels of consistency have also been demonstrated more widely in the field of risk assessment, where different schemes were used to assess the same pool of species (González-Moreno et al., 2019). Consistency in impact scoring therefore remains a challenge in this discipline. The results presented here should be understood in this context. They represent the consensus view of a group of leading experts in GB; however, these scores should be subject to review, challenge and, where evidence indicates it is necessary, modification. To aid this it is important that the justification is transparent and available, which will be done in this case by publishing the scores (including justification) on the GBNNSS website. More broadly, if we are to achieve greater consistency in assessing the impact of invasive species, further work is required: (a) to provide detailed and clearly defined separation between scoring levels, (b) to undertake more primary research into the impacts of individual species and (c) to improve robust but practical expert elicitation and consensus building methods.

In terms of making scores generalizable across locations, it is the case that invasive non-native species may not have the same impact in all places where they occur. For example, a species considered invasive (i.e. causing negative impacts) in one country may be considered benign in another (e.g. Ruddy Duck in the UK and Spain). Even within a country a species may cause serious impacts in some locations, but not in others (e.g. American skunk cabbage in the UK).   This is why the focus of this study was limited to assessing the impact of species within GB. Individual impact scores are therefore not necessarily relevant to other countries, although the method for scoring impact is transferable. To address the potential for the same species to have different within-country impacts the ‘maximum impact’ score of each species was used for analysis. This follows approaches used elsewhere where maximum impact is used following the precautionary principle (Baker et al., 2008, Hawkins et al., 2015).

Chapter 3. Ranking the introduction pathways of non-native species in Great Britain: testing methods that incorporate impact, uncertainty and change over time

3.1 Introduction

The invasion process can be defined as a series of barriers that a species must overcome in order to enter, establish, spread and cause impacts in a new area (Blackburn et al., 2011). An introduction pathway is the means by which a species overcomes the first of these barriers (i.e. 'geography' and 'captivity or cultivation') and arrives in the environment in a new location as a result of human mediation (Essl et al., 2015). It can therefore be broadly defined as "any means that allows the entry" of a species (FAO 2007) encompassing a wide range of activities, routes and vectors (CBD, 2014b) including intentional and unintentional introduction as diverse as contaminants arriving attached to artificial marine debris (Therriault et al., 2018), plants escaping from gardens (Dehnen-Schmutz and Touza, 2008), animals released as part of religious practices (Everard et al., 2019) and quarry introduced for hunting (Scanes, 2018).

There are many different non-native species pathways, with the number, diversity and intensity in any given region linked to the diversity of its trade, travel and transport (Hulme, 2009, Essl et al., 2015, Seebens et al., 2015, van Kleunen et al., 2015). Pathways differ not only in the types of activities and vectors involved, but also the scale at which they operate, the routes that they take, the environments in which they move and the taxa that they introduce (e.g. Hulme et al., 2008, Copp et al., 2010, van Kleunen et al., 2015, Turbelin et al., 2017). This ultimately means that introduction pathways vary considerably in terms of their potential to introduce harmful invasive non-native species (Wilson et al., 2009, Pysek et al., 2011, Pergl et al., 2017, Saul et al., 2017).

With numbers of invasive non-native species increasing globally, preventing introductions by managing pathways is a priority (CBD, 2014b, Lodge et al., 2016) and one of the most cost-effective forms of management (Davies and Sheley, 2007, Pyšek and Richardson, 2010, Brancatelli and Zalba, 2018, Hulme et al., 2018). This has been demonstrated theoretically (e.g. Leung et al., 2014) and practically for a number of specific measures (Lodge et al., 2016), although evidence of the effectiveness of prevention can be limited by the availability of consistently collated data (Essl et al., 2015). Examples of the effectiveness of prevention include that of New Zealand where, after the introduction of stringent biosecurity legislation, the number of non-native mammal introductions reduced dramatically (Armon and Zenetos, 2015). Similarly, in Europe the introduction of pathway management measures appears to have resulted in a decline in the incident of new introductions through aquaculture (Katsanevakis et al., 2013a).

Managing introduction pathways can be complicated and expensive. For example, complex negotiations over 25 years have been required to bring the ballast water convention into force (IMO, 2004) and this is expected to require as many as 75,000 vessels to install ballast water management systems costing an estimated $640,000-$947,000 (USD) per vessel (David and Gollasch, 2015). With limited resources (Chapter 1), large numbers of pathways, high costs and considerable complexity, the management of introduction pathways must therefore be carefully prioritised (Mack et al., 2000, Hulme, 2009, Hulme, 2015, Lodge et al., 2016, McGeoch et al., 2016). This prioritisation must focus on those pathways likely to do the most harm (i.e. introduce the most species that cause serious impacts) and for which risk reduction is likely to be cost-effective (CBD, 2014c, Essl et al., 2015, Cassey et al., 2018b).

In order to prioritise introduction pathways for management it is first necessary to classify them (Hulme et al., 2008), ideally using consistent terminology to allow for comparative analysis across databases and other sources of relevant information (Harrower et al., 2018a). A number of different classification schemes have been developed (e.g. those used by UCN/ISSG GISD, CABI ISC, DAISIE, NNSIP and NOBANIS, discussed in the report of the Working Group on Invasive Alien Species (2018)). However, recent efforts have been made to adopt a single classification under the auspices of the Convention on Biological Diversity (CBD, 2014b), to which many major non-native species databases have been mapped (Saul et al., 2017, Tsiamis et al., 2017, Pagad et al., 2018). An advantage of this classification is that it utilises a hierarchy of pathways (following Hulme et al., 2008), which allows for analysis at different levels, starting with intentional and unintentional; then release, escape, contaminant, stowaway, corridor and unaided; before separating pathways into more detailed lower sub-categories (CBD, 2014b). It has also recently been accompanied by comprehensive guidance in an attempt to ensure pathways are clearly defined and easy to consistently apply (Harrower et al., 2018a).

Guidance for the prioritisation of pathways suggests criteria to take into account (CBD, 2014c, Essl et al., 2015). However, methods to support prioritisation are still at an early stage of development and yet to be broadly agreed (McGeoch et al., 2016). Different approaches have been used, for example based on an analysis of the volume, intensity and frequency of vectors that transport propagules (i.e. vector analysis and pathway risk analysis; Carlton and Ruiz, 2005, Copp et al., 2010, Leung et al., 2014, Lodge et al., 2016, Brancatelli and Zalba, 2018) or modelling approaches that incorporate proxies for propagule pressure (e.g. Bradie et al., 2015). However, one of the most common methods is to rank pathways based on numbers of past introductions (e.g. CBD, 2014b, Essl et al., 2015, Nunes et al., 2015, McGeoch et al., 2016, Zieritz et al., 2016, Saul et al., 2017).

Past introductions can be used to rank or assess pathways based on numbers of all non-native species (Katsanevakis et al., 2013b, CBD, 2014c, Nunes et al., 2014, Roy et al., 2014c, Turbelin et al., 2017); however, this does not take into account the very large differences in impact between species (e.g. Kumschick et al., 2015b). To do this a more limited number of studies have incorporated measures of species impact (McGeoch et al., 2016), usually based on the number of species introduced by pathways considered to be invasive (NOBANIS, 2015, Nunes et al., 2015, e.g. Saul et al., 2017). More comprehensive cross-taxa assessments of pathway impact are complicated because they require methods for comparing differing impact levels across taxa (Essl et al., 2015) and have rarely been completed (but see Madsen et al. (2014)). Indeed, Saul et al. (2017) stress the need for more rigorous assessments of impact to support pathway prioritisation. In addition to species impact, other variables may have an important effect on pathway ranking (Essl et al., 2015). For example, considerable uncertainty around which pathways introduced species could affect ranking (Scalera and Genovesi, 2016). Temporal changes in pathways may also have an important affect, given that the activity of pathways can change considerably through time (e.g. Faulkner et al., 2016, Zieritz et al., 2016, García-Díaz et al., 2018).

While some pathway ranking methods are more detailed than others, further work is required to develop comprehensive pathway ranking methods that incorporate species impacts, pathway uncertainty and temporal change (Essl et al., 2015). However, it is not clear whether such methods would improve upon more straightforward methods already developed. Given that different methods may be more or less practical to apply, but could have a substantial effect on the ranking of pathways and ultimately the prioritisation of management, it is important to investigate the differences, advantages and disadvantages of each approach. To do this a dataset is required that includes comprehensive information about non-native species, their impacts and other variables that may be of importance (such as year of introduction, continent of native origin and environmental information).

In Great Britain (GB), the Non-Native Species Information Portal (NNSIP) provides a comprehensive dataset of non-native species information, including introduction pathway (Roy et al., 2014c). To this has recently been added comprehensive environmental (biodiversity) impact scores for all established non-native species (Chapter 2). This provides a novel dataset with which to test different pathway ranking methods and explore the extent to which different methods result in different ranks. Pathways in the NNSIP database do not follow those of the CBD classification and so need to be mapped in order to provide consistency, in line with international initiatives. This therefore provides an opportunity to consider the practicalities of mapping the CBD classification to pathways at a national scale (one of the first national applications since adoption of the classification) and its use in supporting the prioritisation of pathway management in GB.

The main aim of this study is therefore to consider the implications of applying different pathway ranking methods to inform management, using GB as a case study. In doing so, a range of ranking methods will be developed and tested, including a comprehensive approach that incorporates cross-taxa impact assessment, pathway uncertainty and temporal change. The implications for pathway management in GB will be explored, as well as the practicalities of using the CBD classification at a national scale.

3.2 Methods

3.2.1 Mapping NNSIP and CBD pathways

NNSIP data were extracted (December 2015) for all established non-native species in GB (excluding microorganisms, parasites, parasitoids and fungi), providing for each species: taxonomic information, environmental group, continent of native origin, year of first record in the wild, introduction pathway and notes describing the introduction pathway for the majority of species. Recently added environmental (biodiversity) impact scores (Chapter 2) were also extracted, providing a maximum impact score for each species using a five-point categorical scale (minimal, minor, moderate, major and massive) designed to reflect impact at increasing levels of ecological organisation (from individuals to communities) (Chapter 2). NNSIP pathways were mapped to the CBD classification automatically where possible (coded in R) and manually where not (Fig 3.1). Automatic mapping was used where NNSIP pathways were the same as CBD pathways (e.g. NNSIP release biocontrol = CBD release biological control). In some cases pathways were not synonymous, but could be mapped directly using a series of rules based on the NNSIP pathway combined with taxonomic or environmental information (see supporting information, Appendix C). The large majority of species mapped in this way were correctly classified; however, a minority were not. All were therefore checked and manually corrected if necessary.

In some cases there was no direct match between an NNSIP and CBD pathway (e.g. the NNSIP 'stowaway marine' was split between seven CBD pathways: 'fishing equipment', 'ship excluding ballast or hull', 'machinery and equipment', 'ballast water', 'hull fouling' and 'other') (Fig 3.1). In these cases the NNSIP 'notes' field was reviewed and used to manually determine the most appropriate CBD pathway. This was straightforward in most cases, but where notes were lacking, further research using major databases (i.e. GISD, DAISIE, CABI ISC, NNSS portal) and the primary literature was carried out to determine the appropriate pathway(s). To support analysis, each pathway was codified (Table 3.1). The first letter of this code indicated the broad pathway (i.e. release, escape, contaminant, stowaway, unaided), followed by three or four letters indicating the CBD subcategory. Where an additional level of detail was added (i.e. to 'contaminants of plants' and 'contaminants of animals'), this was provided by adding an additional three or four letters after the subcategory.

Only the original pathway of introduction for each species was used; pathways of subsequent introduction and / or spread were excluded. Where the original pathway of introduction was unclear (i.e. it could have been one of multiple pathways) all possible introduction pathways were recorded for that species. In other words, if the introduction pathway was certain (or at least highly confident), then only that pathway was assigned to the species. Any subsequent pathways that may have introduced further populations after the species had established in GB were not included in this analysis. Only when the original introduction pathway was uncertain (i.e. it could have been one of multiple possible pathways) were multiple possible introduction pathways assigned.

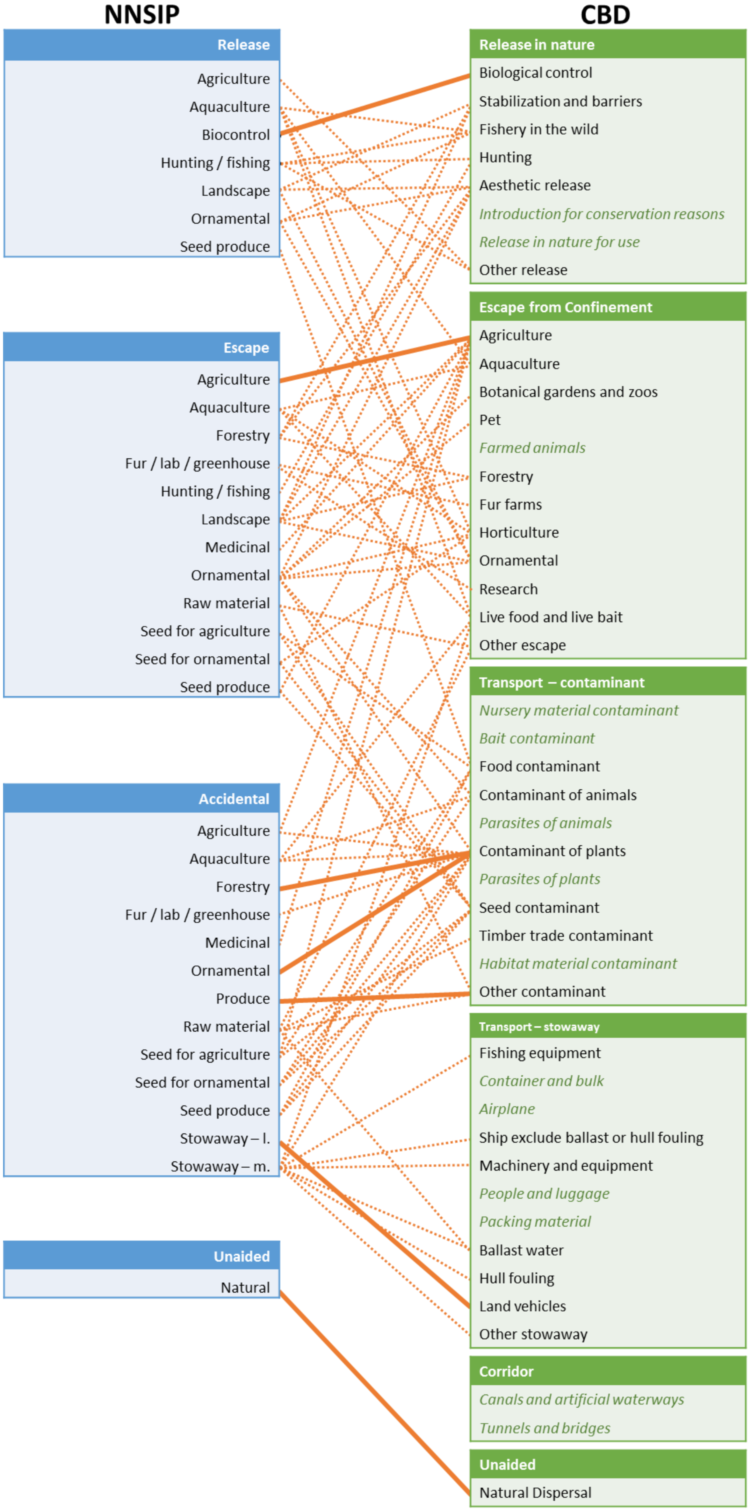


Figure 3.1 Alignment of pathways within the NNSIP and CBD classification schemes, based on pathways assigned within the NNSIP database to established non-native species in GB. Thick lines represent NNSIP pathways which align with a single CBD pathway. Thin dotted lines represent pathways that correspond to multiple CBD pathways. NNSIP pathways follow those of Roy et al. (2014c) CBD pathways follow those modified by the recommendations of Harrower et al. (2018a). CBD pathways in italics were those not represented in the NNSIP database.

Table 3.1 Pathway codes used in this study and the pathways to which they relate. Pathways are organised into a three level hierarchy (intent, broad pathway and subcategory). They follow the CBD classification as modified by (Harrower et al., 2018a) except for contaminant of animals and contaminant of plants which were further divided by sector (using sectors already defined for escape pathways). Only pathways active in GB are included in this table.

Intent	Broad pathway	Pathway sub-category	Code
INTENTIONAL	RELEASE	Biological control	R_BIO
		Stabilization and barriers	R_STAB
		Fishery in the wild	R_FHRY
		Hunting	R_HUNT
		Aesthetic release	R_AES
		Other release	R_OTR
	ESCAPE	Agriculture	E_AGRI
		Aquaculture	E_AQC
		Botanical gardens and zoos	E_ZOB
		Pet	E_PET
		Forestry	E_FOR
		Fur farms	E_FUR
		Horticulture	E_HORT
		Ornamental	E_ORN
		Research	E_RES
		Live food and live bait	E_LFB
		Other escape	E_OTR
UNINTENTIONAL	CONTAMINANT	Food contaminant	C_FOOD
		Contaminant of animals	C_ANI_AGRI, C_ANI_AQC, C_ANI_FISH, C_ANI_UNK
		Contaminant of plants	C_PLT_AGRI, C_PLT_AQC, C_PLT_FOR, C_PLT_ORN, C_PLT_UNK
		Seed contaminant	C_SEED
		Timber trade contaminant	C_TMBR
		Other contaminant	C_OTR
	STOWAWAY	Fishing equipment	S_ANG
		Ship ex. ballast or hull fouling	S_SHH
		Machinery and equipment	S_EQUIP
		Ballast water	S_BALL
		Hull fouling	S_HULL
		Land vehicles	S_LVEH
		Other stowaway	S_OTR
UNAIDED		Natural dispersal	U_NAT

3.2.2 Developing pathway scoring methods to support ranking

To compare methods for ranking pathways it was first necessary to score pathways. This was done using a range of different scoring methods that incorporated counts of species, impact, uncertainty and temporal change, each of which is described below and in Box 3.1.

Species count (Method 1)

This method (Method 1) scored pathways based on the total number of all possible non-native species recorded as being associated with each pathway (Box 3.1). This represented the maximum number of species recorded as being associated with each pathway and is therefore the same as Method 3c (see 'incorporating uncertainty' below).

Incorporating impact (Methods 2a and 2b)

Two different methods for incorporating impact into pathway scoring were used, based on categorical impact scores held by the NNSIP. The first (Method 2a) used a similar approach to Method 1, but counted only those species considered to be invasive (i.e. those that scored more than 'minimal' impact). The second method (Method 2b) converted the categorical impact score of all species into a value using a logarithmic scale (minimal = 0.01, minor = 0.1, moderate = 1, major = 10, massive =100). This logarithmic scale was used as it was considered to most closely reflect the increasing levels of ecological organisation used by the categorical impact scores (e.g. i.e. minor impacts affected individuals whereas moderate impacts affected populations etc.). Pathways were then scored based on the sum of species impact values for each pathway (Box 3.1).

Box 3.1 Scoring methods used to rank pathways. Methods were divided into those that incorporated species count (Method 1), impacts (Methods 2a and 2b), uncertainty (Methods 3a, 3b and 3c), temporal changes (not specifically listed) and a combination of methods (Method 4). All pathways were scored and ranked using each method.

Method 1. Count of all species Every non-native species associated with a pathway is scored 1 (regardless of the number of other pathways that could have introduced the species). The sum of these scores is calculated for each pathway.
Method 2a. Count of invasive non-native species Every invasive non-native species (i.e. those that have more than minimal impact) is scored 1. The sum of these scores is calculated for each pathway. Non-native species that have minimal impact are not included.
Method 2b. Sum of impact scores Every non-native species is allocated an impact value based on its categorical impact score, as follows: minimal = 0.01, minor = 0.1, moderate = 1, major = 10, massive = 100. The sum of these scores is calculated for each pathway.
Method 3a. Minimum count Every non-native species exclusively associated with a single pathway is scored 1. All other species (i.e. those associated with more than one possible original pathway of introduction) are excluded. The sum of these scores is calculated for each pathway.
Method 3b. Intermediate count A score of 1 for each non-native species is divided equally between the number of pathways by which it could have been originally introduced. For example, where a species has four possible introduction pathways, each pathway receives a score of 0.25 for that species. The sum of these scores is calculated for each pathway.
Method 3c. Maximum count This was the same as Method 1 (count method), i.e. all species were counted with a score of 1 regardless of the number of other possible pathways of introduction.
Method 4. Combined methods This method combines Method 2b, Method 3b and an element of time. To concentrate on recently active pathways, only non-native species introduced since 1950 are included. Each of these species is allocated an impact value based on its categorical impact score (Method 2b), which is then divided equally between possible pathways of original introduction (Method 3b). The sum of these scores is calculated for each pathway.

Incorporating uncertainty (Methods 3a, 3b and 3c)

Where the original pathway of introduction for a species was known with confidence, only a single pathway was listed for that species (because only one pathway could be the original introduction pathway). However, in some cases multiple pathways of introduction were listed because the original was uncertain. The minimum (Method 3a) number of species likely to have been introduced by a pathway was therefore determined by counting only those species for which a single pathway was given. Conversely, the maximum (Method 3c) number of species potentially introduced by a pathway was calculated by counting all species associated with the pathway, regardless of whether other pathways were also listed. This was therefore the same as Method 1. An intermediate (Method 3b) number of species introduced by a pathway was also calculated. This was done by dividing the score for each species evenly between the number of potential introduction pathways associated with it (Box 3.1). For example, if a species could have been introduced by three different pathways, the minimum method would score each pathway '0', the maximum would score each '1' and the intermediate would score each pathway '0.33' for that species. While applied here (Box 3.1) to counts of species (modified from Method 1), it could also be applied to calculate a minimum, intermediate and maximum impact score in combination with Methods 2a and 2b. This is demonstrated, in part, in Method 4 below.

Incorporating temporal change

To investigate whether pathway ranks changed over time, pathway scores were determined using the count method (Method 1) with species divided into four different 50 year periods (1800-1849, 1850-1899, 1900-1949 and 1950-2000) based on their year of first record in the wild in GB.

Combined methods (Method 4)

To produce a single method (Method 4) that incorporated impact, uncertainty and temporal change a number of methods were combined. Method 2b (sum of impact values) was combined with Method 3b (intermediate number of species) such that the impact value for each species was evenly divided between its potential pathways of introduction (Box 3.1). For example, where two different pathways were listed for a single species that had an impact value of 10, a score of 5 was allocated to each pathway. Temporal change was incorporated by only including species introduced since 1950. This cut off was used so that scoring was based on most recently active pathways.

3.2.3 Comparing ranks

The correlation between different pathway ranking methods was tested to explore the extent to which they resulted in similar or dissimilar lists of prioritised pathways. To do this, for each method pathway scores were used to rank pathways in order of importance, highest (rank position = 1) to lowest score; where ties occurred rank was assigned alphabetically by pathway name. The similarity, or difference, between these lists of ranked pathways was then compared using Kendall's tau (b) correlation coefficient. This compared the sequence of ranks in each list and determined the degree of concordance (pathways ranked in the same order) and discordance (pathways ranked in opposite order) between ranks. The correlation statistic was a number between -1 and +1, with numbers closer to -1 indicating strong negative correlation, those closer to +1 indicating strong positive correlation and those closer to 0 indicating no correlation.

The degree to which incorporating impact affected resulting ranks was investigated by comparing ranks produced using Methods 2a and 2b (impact methods) to those produced by Method 1 (count method). Ranks produced by Method 2a and Method 2b were also compared to each other, to investigate whether they produced similar or dissimilar results. The degree to which uncertainty affected the results of ranking was investigated by comparing ranks based on Method 3a (the minimum number of species), Method 3b (intermediate number of species) and Method 3c (maximum number of species – same as Method 1). To investigate the degree to which temporal change affected the results of ranking, ranks based on the count of species in each fifty year period were compared to each other. Finally, ranks produced by Method 1 (count method) were compared to ranks produced by Method 4 (combined methods) to investigate the degree to which incorporating a range of different approaches resulted in ranks that were different to the standard approach of using species count.

To explore which scoring method was likely to produce ranks that better align with the management objective of reducing impact, the cumulative impact of pathways ranked by different scoring methods (Methods 1, 2a, 2b and 4) was compared. Cumulative impact was determined based on the sum of impact values for species established after 1950. It was particularly important that the top ranking pathways reflected management priorities and so the cumulative impact of the top 5 pathways ranked by Method 1 was compared to that of Method 4.

3.2.4 Displaying uncertainty

Method 4 (combined methods) was used to rank GB pathways for further analysis. This used the sum of impact values for the intermediate number of species introduced after 1950 to rank pathways; however, there was uncertainty around these values. To visualise this the sum of impact values for the minimum and maximum number of species introduced after 1950 was also calculated. These were represented as either a range around the intermediate score (tables), error bars (point plots) or shading (line plots).

All coding was undertaken in R version 3.4.1. (R Core Team, 2017), primarily using the tidyr package (Wickham and Henry, 2018) and rworldmap (South, 2011).

3.3 Results

3.3.1 Availability of pathways data

Of the 1954 established non-native species in the GB NNSIP database, at least one possible pathway of original introduction was known for 1710 (88%); while the pathway of introduction was unknown for 168 (9%) and data were unavailable (NIL) for a further 76 (4%). The species with known pathways included a range of broad taxa (plants, n=1336; invertebrates, n=318; and vertebrates, n=56) from different environments (terrestrial, n=1561; freshwater, n=80; marine, n=68; marine and freshwater, n=1) and native origin (Africa, n=75; Asia-Temperate, n=253; Asia-Tropical, n=37; Australasia, 82; Europe, n=753; North America, n=171; Pacific, n=11; South America, n=66; no native origin, n=99; no data, n=163). All species for which the introduction pathway was unknown or unavailable caused minimal impact, these comprised mainly invertebrates (n=182), as well as plants (n=61) and one vertebrate.

Table 3.2 The number of established non-native species in GB associated with each broad CBD pathway category. Where more than one pathway was assigned to a species (because of uncertainty over which was the original introduction pathway) each species / pathway combination was counted, hence the total number of species in this table (n=2497) is more than the total number in the NNSIP database (n=1954). In some cases no pathway way known for the species (unknown), whereas for others pathway data was missing from the NNSIP database (NIL).

Broad pathway category	**Number of species associated with each broad pathway**
Release	194
Escape	1230
Contaminant	640
Stowaway	152
Corridor	0
Unaided	37
Unknown	168
NIL	76
TOTAL	2497

3.3.2 Ability to map NNSIP pathways to CBD classification

All introduction pathways were mapped to the CBD classification and hierarchy (Fig 3.1). This resulted in 2,497 species / pathway combinations (including NIL and unknown) (Table 3.2), with multiple pathways allocated to some species where the original introduction pathway was uncertain (1 pathway, n=1208 species; 2 pathways, n=379; 3 pathways, n=86; 4 pathways, n=32; 5 pathways, n=4; 6 pathways, n=1). Automatic rules were used to fit 1596 (64%) pathway entries (for 1416 species), of which 281 were manually corrected (for 230 species). The remaining 894 pathway entries (for 538 species) were fitted manually. In two cases CBD pathways were split to provide additional detail. ‘Contaminants of plants’ was divided into five pathways to reflect the purpose of importing the plant (agriculture, aquaculture, forestry, ornamental and unknown). ‘Contaminants of animals’ was divided into four pathways (aquaculture, agriculture, fish imports and other). While in many cases the majority of species within an NNSIP pathway mapped directly to a CBD pathway, it was rare that the pathways matched exactly. This resulted in many cross links between NNSIP and CBD pathways (Fig 3.1).

In total, established non-native species in GB were introduced by 31 (out of 45) different pathways from the CBD classification. Fourteen pathways were not represented (release: conservation in wild, release in nature for use; escape: farmed animals; contaminant: nursery material contaminant, bait contaminant, parasites on animals, parasites on plants, habitat material contaminant; stowaway: container and bulk cargo, airplane, packing material, people and their luggage; corridor: canals and artificial waterways, tunnels and bridges) and are therefore not included further in analysis. The number of pathways increased to 38 when the split in plant and animal contaminant pathways was taken into account. Of the known pathways, 63% were intentional, 35% unintentional and 2% unaided. At sub-category level the escape pathway was largest (55%), followed by contaminants (28%), releases (9%) and stowaways (7%); no species were introduced via the corridor pathway (Table 3.2).

3.3.3 Comparing pathway scoring methods for ranking pathways

Different pathway scoring methods produced different ranks (Table 3.3, Table 3.4, Fig 3.2). When both impact methods (Method 2a and 2b) were compared to the count method (Method 1) there were considerable differences in the resulting ranked lists of pathways ($\tau = 0.37$, and $\tau = 0.28$ respectively) (Table 3.3a, Table 3.4a, Figs 3.2a, 2b). However, ranks produced by each impact method were more similar to each other ($\tau = 0.71$) (Table 3.3a, Table 3.4a, Fig 3.2c).

There were fewer differences between pathways ranked by each uncertainty method, with ranks based on Method 3b (intermediate number of species) similar to those based on Method 3a (minimum number of species) and Method 3c (maximum number of species – same as Method 1) (Table 3.4b). However, when ranks based on Method 3a and Method 3c were compared to each other, there was a higher degree of dissimilarity ($\tau = 0.67$) (Table 3.3b, Table 3.4b).

Pathway ranks changed over time, with a tau score no greater than 0.65 between any 50 year period (Table 3.4c). The similarity between ranks reduced as the gap between periods increased, for example dropping to $\tau = 0.41$ between 1801-1850 and 1951-2000.

Combining methods into a single approach (Method 4) resulted in pathway ranks that were the least similar to the count method (Method 1) ($\tau = 0.26$) (Table 3.3a, Table 3.4d, Fig 3.2d). This was largely because Method 1 ranked pathways higher that introduced large numbers of species, even when few of these species caused significant impacts (e.g. seed contaminants and agricultural escapes). In total, half of the top ten pathways ranked by Method 1 were absent from the top ten priorities identified by Method 4 (Table 3.3a). Where there were pathways common to the top ten ranks produced by each scoring method, the rank position of these pathways differed markedly (Table 3.3a). For example, hull fouling was identified by Method 4 as the highest ranking pathway, but only the eighth rank using Method 1. This difference was due, in part, to the large proportion of hull fouling species that caused significant impacts; however, it also related to the recent increase in the introduction of harmful species via this pathway.

The cumulative impact curve for pathways ranked by Method 4 (combined methods) was steeper than for pathways ranked by other methods (Fig 3.3); while pathways ranked by Method 1 (count method) produced the shallowest curve. The cumulative impact of pathways ranked by Method 1 was close to half (54%) that of pathways ranked by Method 4 (Fig 3.3, inset table).

Table 3.3 Comparison of top 10 pathways ranked by different pathway scoring methods. Pathways marked ***** are unique and do not appear in the other ranked lists. For full explanation of pathway codes refer to Table 3.1.

a. Count (Method1), impact (Method 2a and 2b) and combined (Method 4) methods compared.

Rank	Method 1	Method 2a	Method 2b	Method 4
1	E_HORT	E_HORT	E_HORT	S_HULL
2	C_SEED*****	R_AES	S_HULL	E_HORT
3	R_AES	S_HULL	S_BALL	C_PLT_ORN
4	E_AGRI*****	S_BALL	R_AES	S_BALL
5	C_PLT_ORN	C_ANI_AQC	C_ANI_AQC	S_ANG
6	C_FOOD*****	E_PET*****	E_AQC	S_OTR
7	C_OTR*****	E_ORN	C_PLT_ORN	C_ANI_AQC
8	S_BALL	C_PLT_ORN	S_OTR	E_ORN
9	S_HULL	E_AQC	S_ANG	E_AQC
10	U_NAT*	R_FHRY	R_FHRY	E_LFB

b. Pathways ranked by different levels of certainty (minimum number of species (3a), intermediate number of species (3b) and maximum number of species per pathway (3c)).

Rank	Method 3a (minimum)	Method 3b (intermediate)	Method 3c (maximum)
1	E_HORT	E_HORT	E_HORT
2	C_PLT_ORN	C_SEED	C_SEED
3	C_SEED	C_PLT_ORN	R_AES
4	E_AGRI	E_AGRI	E_AGRI
5	C_FOOD	R_AES	C_PLT_ORN
6	U_NAT	C_FOOD	C_FOOD
7	C_OTR	C_OTR	C_OTR
8	C_TMBR*****	S_BALL	S_BALL
9	C_PLT_FOR*****	S_HULL	S_HULL
10	R_AES	U_NAT	U_NAT

Table 3.4 Kendall's tau b correlation coefficients indicating concordance between pathways ranked by different scoring methods (for method descriptions refer to Box 3.1). Method 1 (count of all non-native species), Method 2a (count of invasive non-native species) and Method 2b (sum of impact values) were compared to each other (a). Uncertainty methods (Methods 3a, 3b and 3c) were compared to each other (b). Methods used to rank species in different time periods were also compared to each other (c). Finally, Method 4 (combined methods) was compared to Method 1 (d). Rank ties were handled alphabetically. Where pathways were absent from one scoring method but not the other the rank was set to the lowest position.

a. Concordance between pathways ranked by count of all non-native species (Method 1), count only of invasive species (Method 2a) and sum of impact values of all species (Method 2b).

	Method 1	**Method 2a**	**Method 2b**
Method 1	1.00	0.37	0.28
Method 2a	-	1.00	0.71
Method 2b	-	-	1.00

b. Concordance between pathways ranked by counts of minimum (Method 3a), intermediate (Method 3b) and maximum (Method 3c – note this is the same as Method 1) number of species associated with each pathway.

	Method 3a	**Method 3b**	**Method 3c**
Method 3a	1.00	0.80	0.67
Method 3b	-	1.00	0.87
Method 3c	-	-	1.00

c. Concordance between pathways ranked (using Method 1) based on species that were introduced in different fifty year time periods.

	1951-2000	**1901-1950**	**1851-1900**	**1801-1850**
1951-2000	1.00	0.51	0.55	0.41
1901-1950	-	1.00	0.65	0.56
1851-1900	-	-	1.00	0.60
1801-1850	-	-	-	1.00

d. Concordance between pathways ranked by the total number of non-native species per pathway (Method 1) and combined methods (Method 4).

	Method 1	**Method 4**
Method 1	1.00	0.26
Method 4	-	1.00

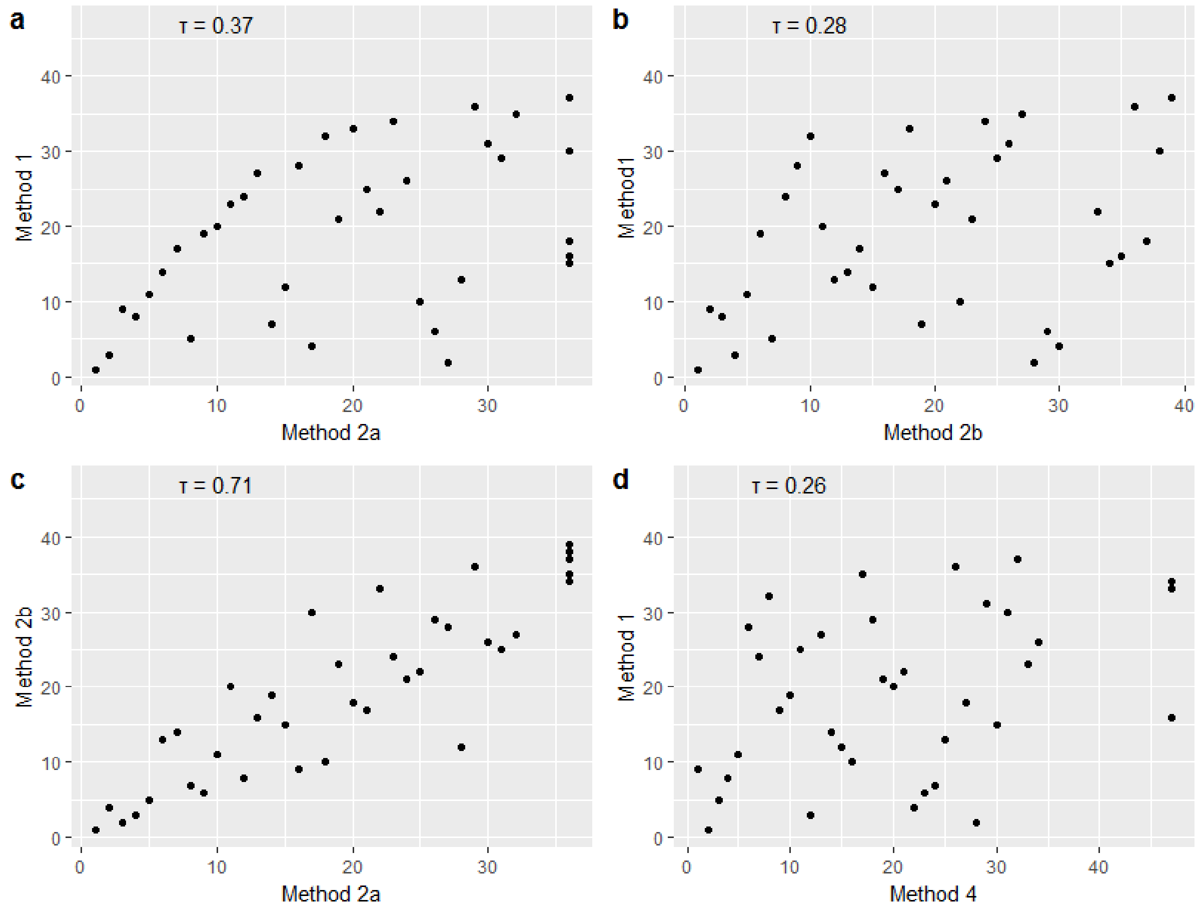
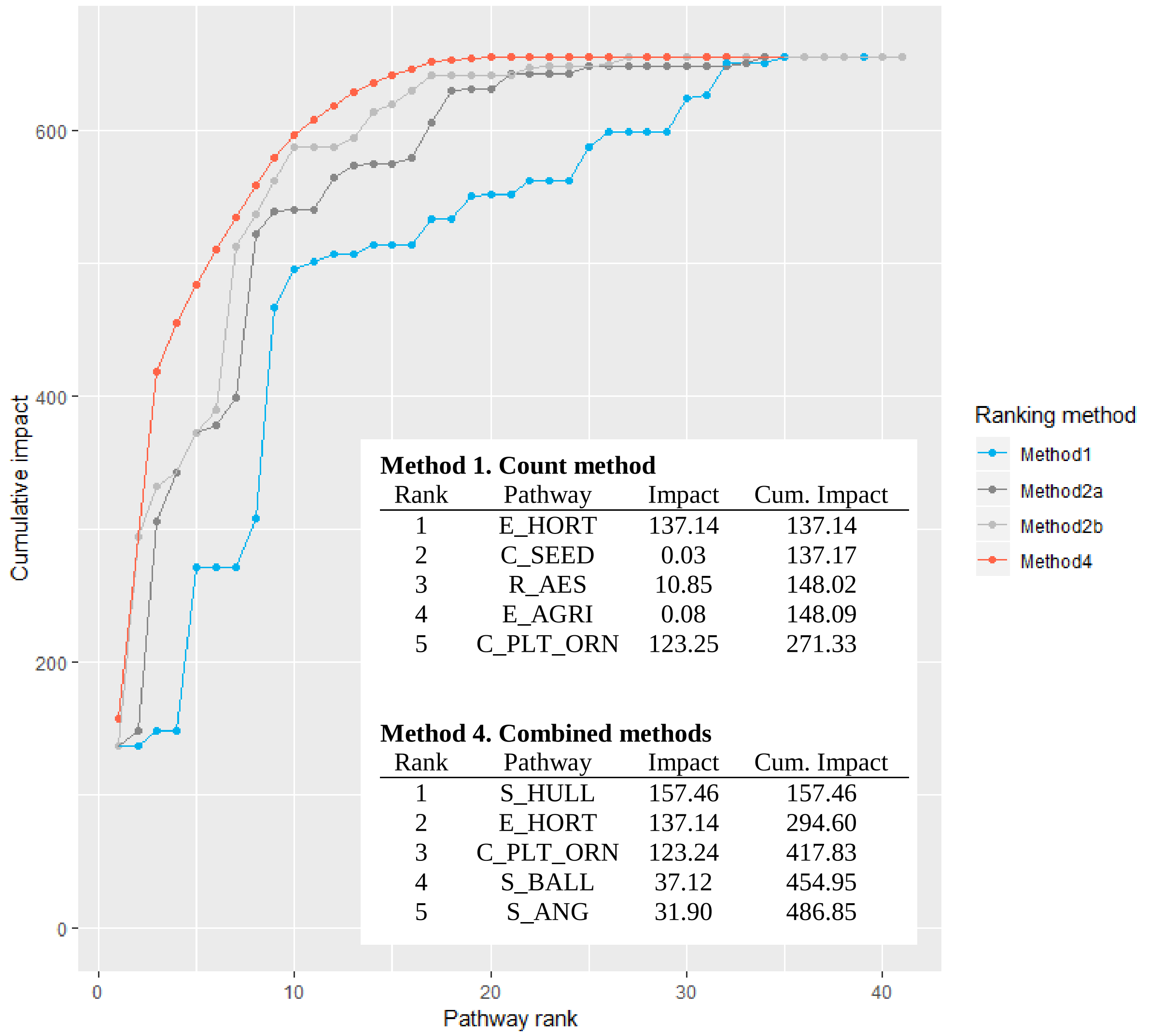


Figure 3.2 Plots illustrating the concordance, or lack thereof, between pathways ranked by selected scoring methods. Low levels of concordance were found between ranks produced by impact scoring methods (Methods 2a and 2b) and count method (Method 1) (panels a and b). However, ranks produced using impact scoring methods were more closely correlated with each other (panel c). The lowest level of concordance was found between ranks produced by combined methods (Method 4) and count method (Method 1) (panel d).



Method 1. Count method

Rank	Pathway	Impact	Cum. Impact
1	E_HORT	137.14	137.14
2	C_SEED	0.03	137.17
3	R_AES	10.85	148.02
4	E_AGRI	0.08	148.09
5	C_PLT_ORN	123.25	271.33

Method 4. Combined methods

Rank	Pathway	Impact	Cum. Impact
1	S_HULL	157.46	157.46
2	E_HORT	137.14	294.60
3	C_PLT_ORN	123.24	417.83
4	S_BALL	37.12	454.95
5	S_ANG	31.90	486.85

Figure 3.3 Difference in cumulative impact of pathways ranked by different scoring methods:Method 1 (count of all non-native species established in GB), Method 2a (count of invasive species only), Method 2b (sum of impact values) and Method 4 (combined methods). Points denote the cumulative impact (based on logarithmic scale applied to categorical scores) for pathways in rank order. Inset table indicates sum of impact values for species introduced by each pathway (Impact) and cumulative impact of pathways ranked by each method (Cumulative Impact). For full explanation of pathway codes refer to Table 3.1.

3.3.4 Ranking pathways of non-native species introduction in Great Britain

To explore pathway ranks further in GB, Method 4 (combined methods) was used (Fig 3.4), with additional detailed provided for the top ten pathways based on species established since 1950 (Table 3.5) and all established species (Table 3.6). Hull fouling (S_HULL) was identified as the highest ranking pathway (Fig 3.4). While this introduced relatively few species overall, those established after 1950 had larger combined impacts than species introduced by other pathways (Table 3.5). This was qualified by considerable uncertainty, with 86% of post-1950 species associated with this pathway also associated with at least one other pathway. Even so, the minimum number of potential introductions (bottom error bar, plot Fig 3.4) indicated this was still an important pathway and the maximum number indicated it could be substantially higher impact than others (top error bar, plot Fig 3.4). The majority of species introduced by this pathway were marine invertebrates, with smaller numbers of freshwater invertebrates and marine plants (Table 3.5). However, the freshwater species introduced by this pathway were particularly impactful (e.g. *Corbicula fluminea*, *Dreissena bugensis*, *Dikerogammarus villosus*, *Dikerogammarus haemobaphes*, *Rangia cuneata*). A number of high impact marine species were also introduced by this pathway (e.g. *Styela clava, Didemnum vexillum, Sargassum muticum* and *Undaria pinnatifida*). There has been a rapid increase in the impact of this pathway since 1950 (Fig 3.6), with species introduced that have native origins from all over the world (Table 3.5, column entitled 'Species origin'). However, impactful species recently introduced by hull fouling originated primarily from Europe (Table 3.5, column entitled 'Impact origin').

By contrast, horticultural escapes (E_HORT), the next highest ranking pathway, introduced by far the largest number of species (in total and since 1950) with low uncertainty (Fig 3.4). However, a smaller proportion of these species caused substantial impacts (Table 3.5). This pathway mainly introduced terrestrial plants, but also a small number of freshwater plants (Table 3.5). The number of species introduced by this pathway has been growing since the late 1700s and, while the proportion of species introduced by other pathways has increased, it is still the dominant pathway in terms of numbers of species introduced (Fig 3.5). However, in terms of recent impacts this pathway is less dominant (Fig 3.6). Including the most recent introductions, it appears that the impact of the horticultural escape pathway may be stabilising or even decreasing (Fig 3.6); however, this may be an artefact of lag in the ability of experts to detect impact (Chapter 2). Large numbers of horticultural escapes came from across the globe, with particularly large numbers from native origins in Europe and temperate Asia (Table 3.5, column entitled 'Species origin'). However, in terms of impact since 1950, species with native origins in North America have caused the most impact (Table 3.5, column entitled 'Impact origin').

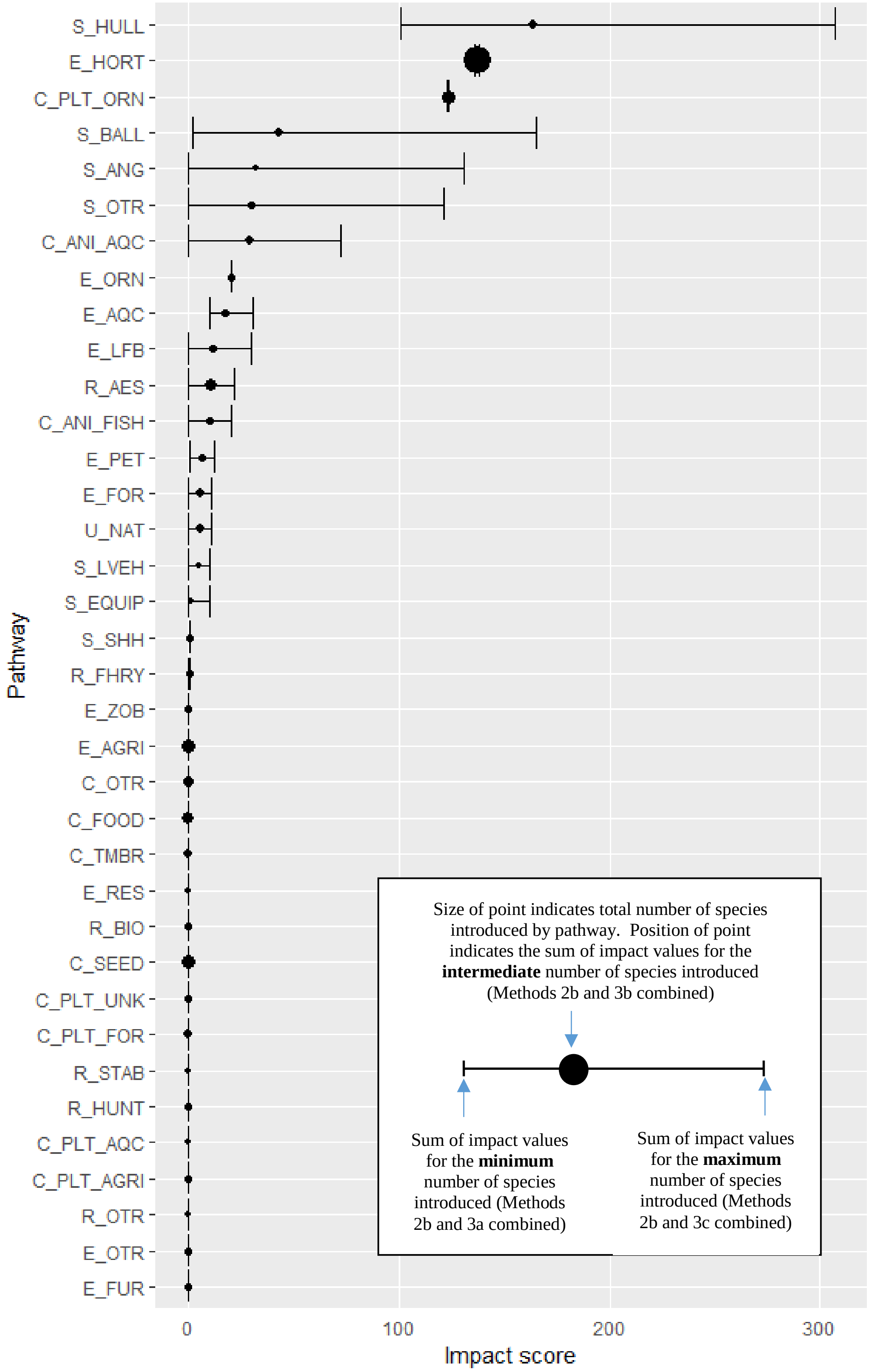


Figure 3.4 Pathways ranked by combined methods (Method 4), indicating potential priorities in GB. Point size indicates total number of species introduced since 1950, while position of points with error bars indicates the sum of impact values for the minimum, intermediate and maximum impact of species introduced by each pathway since 1950 (illustrated by inset). For full explanation of pathway codes refer to Table 3.1

The ornamental plant contaminant pathway (C_PLT_ORN) was the fifth largest in terms of total numbers of species introduced, but was ranked third by combined methods (Fig 3.4). Nine percent of species introduced by this pathway since 1950 have caused impacts, some of which have been particularly severe (e.g. *Arthurdendyus triangulatus*). There was a high degree of certainty in the species associated this pathway as there was often a clear trophic relationship between the non-native species and its plant host (e.g. *Arge berberidis* the berberis sawfly, *Cameraria ohridella* the horse-chestnut leaf miner and *Otiorhynchus crataegi* the privet weevil). This pathway primarily introduced terrestrial invertebrates; however, it also introduced freshwater invertebrates, terrestrial plants and possibly one freshwater plant (Table 3.5).

Of the remaining pathways a group of four (ranks 4-7) stood out as having more potential impact than others (ballast water stowaways, S_BALL; angling stowaways, S_ANG, 'other' stowaways, S_OTR and contaminants of aquaculture animals, C_ANI_AQC), albeit with considerable uncertainty (note S_OTR primarily related to stowaways on equipment such as pumps and water sports equipment used in freshwaters abroad). These all occupied a similar position, given their intermediate impact scores and wide error bars; although, ballast water (S_BALL) scored slightly higher. The pathways angling stowaways (S_ANG) and 'other' stowaways (S_OTR) scored similarly as they were associated with the same small group of particularly high impact species (including *Dikerogammarus haemobaphes*, *D. villosus*, *Dreissena bugensis* and *Hemimysis anomala*). The contaminant of aquaculture animals pathways was associated with a larger number of species introduced since 1950 (n=17), most of which were marine (n=14), but with lesser impacts. A further group of 10 pathways caused more than negligible impacts (ranks 8-17), with ornamental escapes (E_ORN), aquaculture escapes (E_AQC), life food and bait (E_LFB), aesthetic release (R_AES) and contaminants of fish (C_ANI_FISH) scoring higher than others (but with high uncertainty in all cases). Nineteen pathways were associated with little if any impact based on species introduced since 1950, despite relatively large numbers of species introduced in some cases (e.g. agricultural escapes, E_AGRI; contaminants of food, C_FOOD; and seed contaminants, C_SEED).

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Table 3.5 Top 10 pathways ranked by combined methods (Method 4) and associated statistics, based on non-native species that established in GB after 1950. No. NNS = total number of species introduced by pathway. No. INNS = total number of species with more than minimal impacts. Impact = sum of species' impact scores. Impact = sum of impact scores. Origin number = line weight indicates number of species; origin impact = line weight indicates sum of impact scores. The number of species from each broad taxa (P = plant, I = invertebrate, V = vertebrate) and environment is given. In all cases the intermediate number (or impact) of species is given followed by the minimum and maximum figures in brackets. For full explanation of pathway codes refer to Table 3.1.

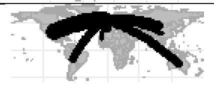
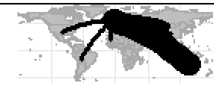
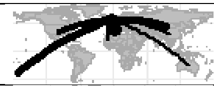
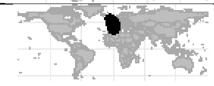
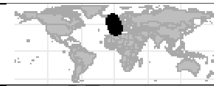
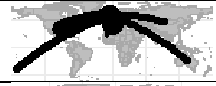
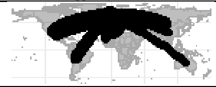
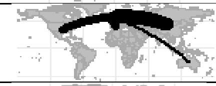
Pathway	No. NNS	No. INNS	Prop. INNS	Impact	Species native origin	Impact native origin	Terrestrial	Freshwater	Marine
S_HULL	17.1 (5-36)	8.2 (2-19)	0.48	163.5 (101-307.2)				I 2.2 (1-6)	I 13.8 (4-27) P 1.1 (0-3)
E_HORT	317.4 (284-352)	20.5 (19-22)	0.06	138.2 (136.8-139.6)			P 305.4 (273-339)	P 12 (11-13)	
C_PLT_ORN	73 (65-81)	6.5 (6-7)	0.09	123.3 (122.7-123.8)			I 60 (53-67) P 5.5 (5-6)	I 7 (7-7) P 0.5 (0-1)	
S_BALL	13.2 (5-28)	4.8 (2-12)	0.36	43.2 (2-165.2)			I 1 (1-1) P 1 (1-1)	I 1.2 (0-5)	I 9.8 (3-20) P 0.2 (0-1)
S_ANG	1.2 (0-5)	1.2 (0-5)	1	31.9 (0-131)				I 0.8 (0-4)	I 0.2 (0-1)
S_OTR	1 (0-4)	1 (0-4)	1	30.2 (0-121)				I 1 (0-4)	
C_ANI_AQC	7.2 (1-17)	4 (0-10)	0.55	29.2 (0-72.2)				I 0.2 (0-1) V 0.8 (0-2)	I 4.1 (0-10) P 2.1 (1-4)
E_ORN	4 (4-4)	3 (3-3)	0.75	20.1 (20.1-20.1)			V 1 (1-1)	I 2 (2-2) V 1 (1-1)	
E_AQC	3.7 (2-6)	2.2 (1-4)	0.59	17.2 (10-31)				I 2.5 (2-3)	I 1.2 (0-3)
E_LFB	2.5 (1-5)	1.2 (0-3)	0.47	11.7 (0-30)			I 1 (1-1)	I 1 (0-2)	I 0.5 (0-2)

Table 3.6 Top 10 pathways ranked by combined methods (Method 4) and associated statistics, based on all established non-native species in GB. No. NNS = total number of species introduced by pathway. No. INNS = total number of species with more than minimal impacts. Impact = sum of species' impact scores. Impact = sum of impact scores. Origin number = line weight indicates number of species; origin impact = line weight indicates sum of impact scores. The number of species from each broad taxa (P = plant, I = invertebrate, V = vertebrate) and environment is given. In all cases the intermediate number (or impact) of species is given followed by the minimum and maximum figures in brackets. For full explanation of pathway codes refer to Table 3.1.

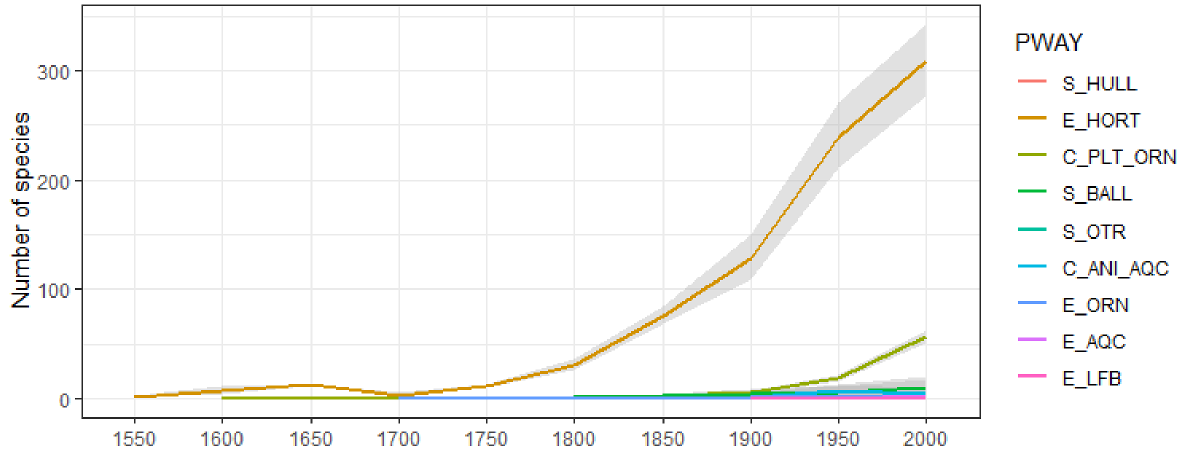
Pathway	No. NNS	No. INNS	Prop. INNS	Impact	Species native origin	Impact native origin	Terrestrial	Freshwater	Marine
S_HULL	26.4 (5-58)	10.5 (2-24)	0.40	207.5 (101-428.4)				I 5.2 (1-12)	I 18.4 (4-39) P 2.8 (0-7)
E_HORT	865 (762-980)	86.9 (75-100)	0.10	474.3 (417.6-539.9)			P 844 (743-957)	P 21 (19-23)	
C_PLT_ORN	115.8 (104-129)	7.5 (7-8)	0.06	133.7 (133.1-134.3)			I 94 (87-102) P 10.3 (8-13)	I 10.5 (10-11) P 0.5 (0-1) V 0.5 (0-1)	
S_BALL	29.6 (11-59)	9 (3-20)	0.30	107 (12.1-316.5)			I 1 (1-1) P 8.5 (6-12)	I 5.2 (1-12)	I 14.2 (3-32) P 0.6 (0-2)
S_ANG	1.2 (0-5)	1.2 (0-5)	1	31.9 (0-131)				I 1 (0-4)	I 0.2 (0-1)
S_OTR	4.5 (3-8)	3 (2-6)	0.67	41.3 (11-132)			I 1 (1-1) P 1 (1-1)	I 2 (1-5)	I 0.5 (0-1)
C_ANI_AQC	17.4 (6-36)	6.7 (1-15)	0.38	64.5 (1.1-175.4)				I 0.2 (0-1) V 1.2 (0-3)	I 10.8 (4-22) P 5.2 (2-10)
E_ORN	7.2 (4-11)	5.7 (3-9)	0.79	31.3 (20.1-43.2)			V 4.2 (1-8)	I 2 (2-2) V 1 (1-1)	
E_AQC	6.2 (3-10)	4.7 (2-8)	0.76	72.3 (10.1-141.2)				I 2.5 (2-3) V 2 (1-3)	I 1.7 (0-4)
E_LFB	3.3 (1-7)	1.2 (0-3)	0.35	11.7 (0-30)			I 1 (1-1)	I 1 (0-2)	I 1.3 (0-4)

3.3.5 Taxonomic, environmental and temporal patterns

Pathways changed over time in terms of numbers of species introduced (Fig 3.5), but particularly in terms of impact (Fig 3.6). While the numbers of species introduced by the horticultural escape (E_HORT) pathway increased rapidly throughout the 19th and 20th century (Fig 3.5a), in terms of impact it has plateaued in recent years (Fig 3.6a). Numbers and proportions of species introduced by the contaminant of ornamental plants pathway (C_PLT_ORN) increased towards the end of the 20th century, as did those introduced by hull fouling (S_HULL) and ballast water (S_BALL) to a lesser degree (Fig 3.5a and b). However, in terms of impact there has been a considerable increase in hull fouling (S_HULL) and contaminants of ornamental plants (C_PLT_ORN) (Fig 3.6a and b).

Twenty-five pathways introduced terrestrial non-native species that have established since 1950, of which 13 introduced plants, 13 introduced invertebrates and 5 introduced vertebrates (Fig 3.7a). In terms of impact, the key terrestrial pathways were horticultural escapes (E_HORT; terrestrial plants) and contaminants of ornamental plants (C_PLT_ORN; terrestrial invertebrates). In the freshwater environment, 14 pathways introduced species that have established since 1950, of which 2 introduced plants, 10 introduced invertebrates and 6 introduced vertebrates (Fig 3.7b). By far the largest uncertainty was associated with freshwater invertebrate pathways, which was also the group associated with the largest impacts. Key freshwater pathways in terms of impact included horticultural escapes (hull fouling (S_HULL), ballast (S_HULL), angling (S_ANG) and other (S_OTR) stowaways, for invertebrates; horticultural escapes (E_HORT) for plants). Few freshwater vertebrates have been introduced since 1950; these were introduced mainly as contaminants of fish stocks (C_ANI_FISH), escaped pets (E_PET) or contaminants of other aquaculture animals (C_ANI_AQC). Ten pathways introduced marine non-native that have established since 1950, of which four introduced marine plants and ten introduced invertebrates (no marine vertebrates have been introduced since 1950, or indeed at all in GB) (Fig 3.7c). The majority of marine impacts since 1950 have been caused by hull (S_HULL) and ballast (S_BALL) stowaways, as well as contaminants of aquaculture animals.

a. Number of species introduced



b. Proportion of species introduced by pathways over time.

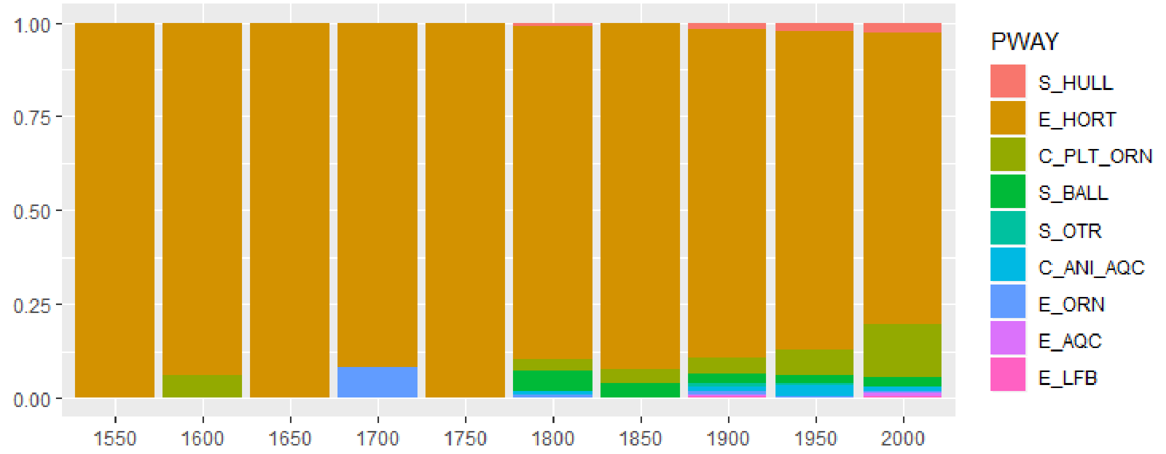
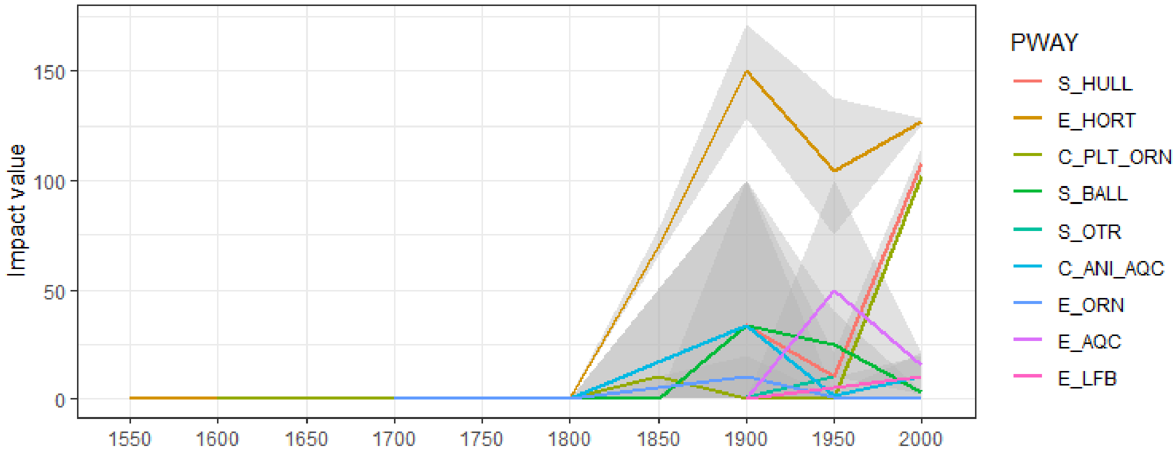


Figure 3.5 Number (a) and proportion (b) of species for each of the top ten pathways over time (50 year periods). Trends are based on intermediate number of species that have established in GB (Method 3b). Shading (panel a) indicates minimum (Method 3a) and maximum number of species (Method 3c). For full explanation of pathway codes refer to Table 3.1.

a. Impact of introduced species by pathways over time.



b. Proportion of impact caused by species introduced by pathways over time.

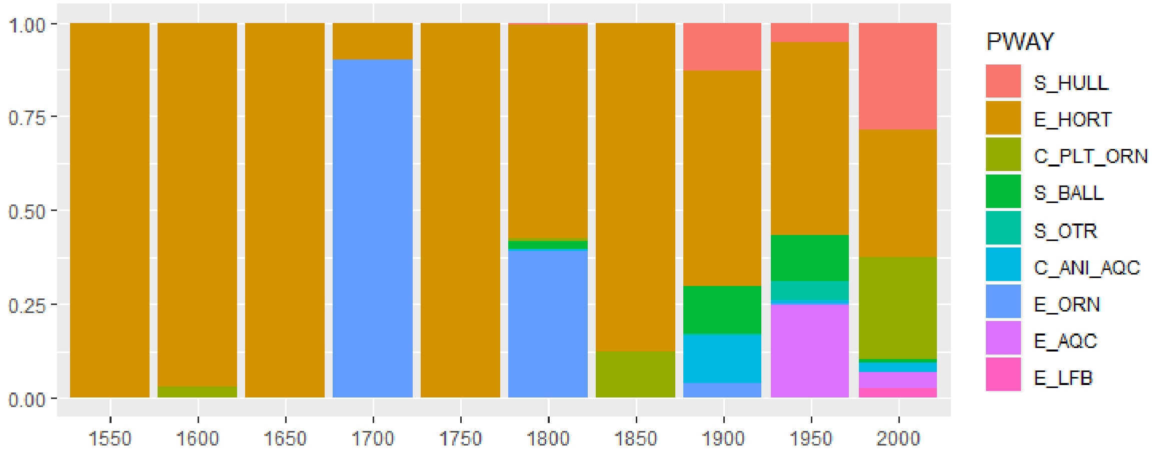
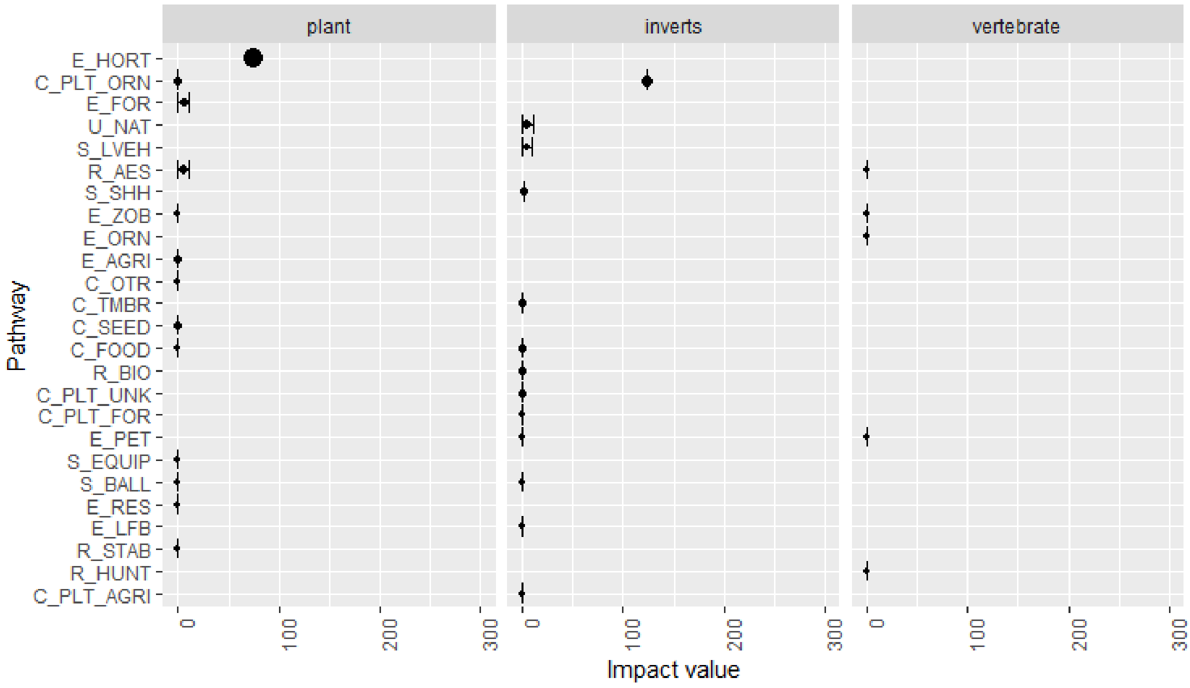
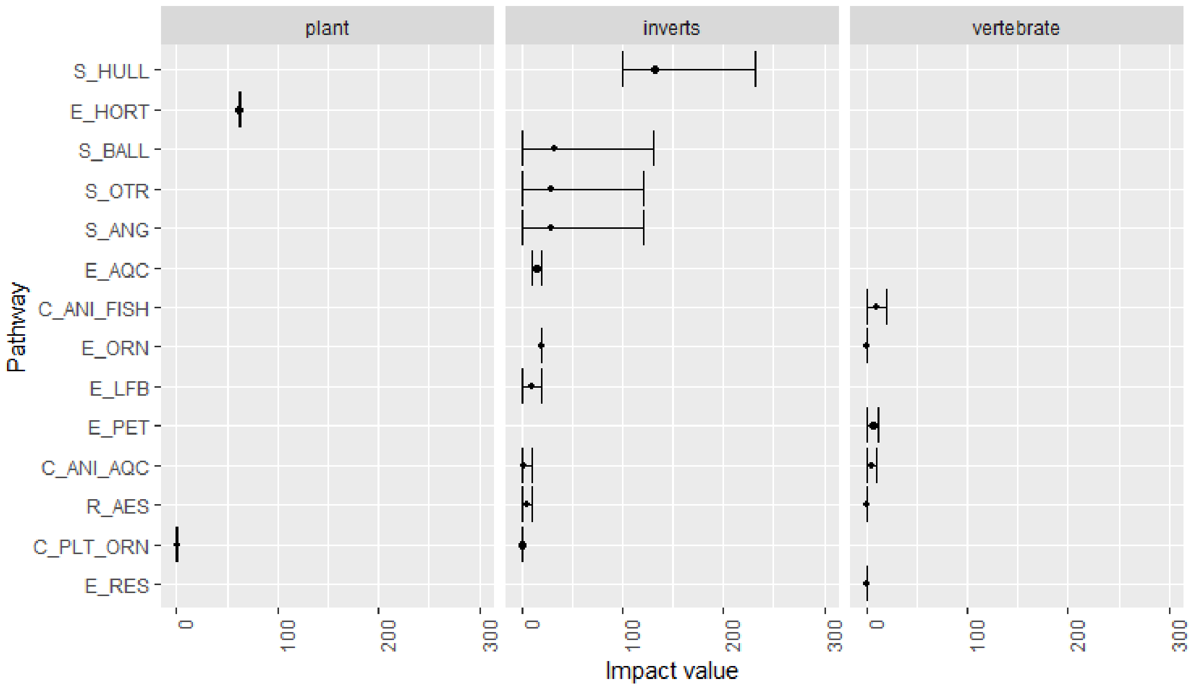


Figure 3.6 Impact (a) of and proportion of impact (b) of species introduced by each of the top ten pathways over time (50 year periods). Trends are based on the sum of impact values for species associated with each pathway, using the intermediate number of species introduced (Method 2b combined with Method 3b). The minimum (Method 3a combined with Method 2b) and maximum impact (Method 3c combined with 2b) of each pathway is indicated (shading in panel a). For full explanation of pathway codes refer to Table 3.1.

a. Terrestrial environment



b. Freshwater environment



c. Marine environment

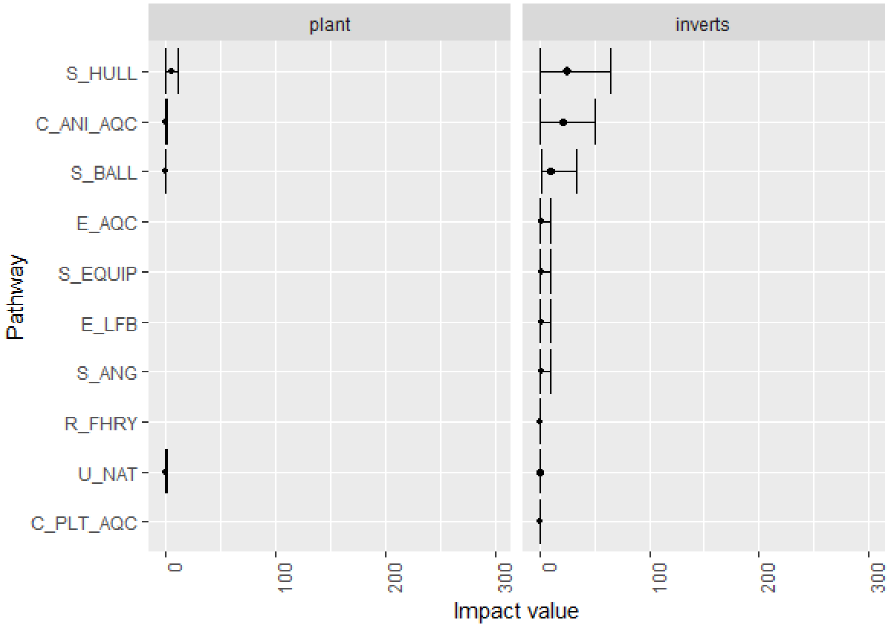


Figure 3.7 Pathways ranked by combined methods (Method 4) separated by taxa and environment. Point position indicates the intermediate impact of each pathway, while error bars indicate the minimum (Method 2b combined with Method 3a) and maximum (Method 2b combined with Method 3c) impact. Point size indicates the number of species introduced (using the intermediate method, Method 3b). All methods included only species that established in GB after 1950. Wide error bars indicate low certainty in pathway impact. For full explanation of pathway codes refer to Table 3.1.

3.3.6 Relationship between number of species associated with each pathway and proportion invasive

The total number of all non-native species introduced by each pathway was plotted against the proportion of those species that were invasive (i.e. caused more than minimal impact) (Fig 3.8). The relationship between the total number of species and invasive proportion appeared to be negative, with pathways either introducing many species or a large proportion of invasive species, but not both. Higher ranked pathways were those further from the bottom left corner of this plot (i.e. low numbers of species and small proportion invasive).

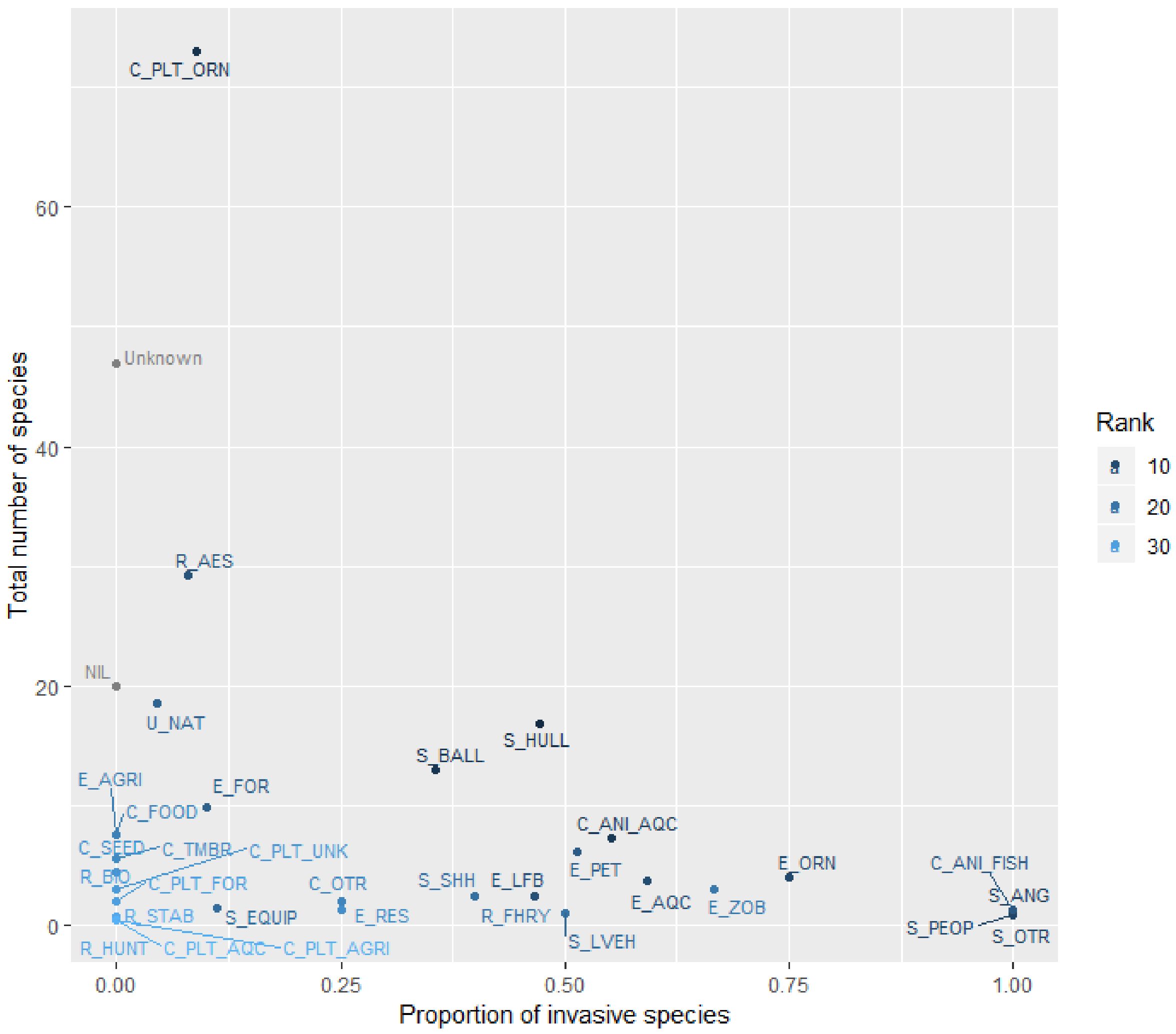


Figure 3.8 Comparison of total number of species introduced by each pathway and the proportion that were invasive (i.e. scored more than minimal impact), based on species established in GB since 1950. Colour indicates rank determined using Method 4 (combined methods), with darker colours indicating higher rank. To aid visualisation the horticultural escape pathway has been excluded from this figure (total number of species 317.4, proportion invasive 0.06). For full explanation of pathway codes refer to Table 3.1.

3.4 Discussion

This study found that pathway ranks differed substantially depending on the scoring method used. Given that the ultimate aim of pathway ranking is to identify management priorities (CBD, 2014b, Lodge et al., 2016), this is important as it suggests that different pathways would be prioritised depending on the method used. While different ranking approaches have been developed, often based on number of all non-native species (e.g. Katsanevakis et al., 2013b, CBD, 2014c, Nunes et al., 2014, Roy et al., 2014c, Turbelin et al., 2017); and in some cases assessments of species impact (e.g. Madsen et al., 2014, NOBANIS, 2015, Saul et al., 2017), these have not been compared to consider the extent to which they differ. This study found that methods that accounted for impact performed better than those based on numbers of species alone. This is perhaps intuitive given that the objective of management is to reduce impact (Essl et al., 2015), but was also demonstrated by the cumulative impact reduction that would be expected by prioritising pathway management using different ranking methods. Incorporating uncertainty produced less pronounced differences, with relatively high levels of concordance between ranks produced using minimum, intermediate and maximum numbers of species. However, uncertainty did have a strong impact on the ranking of some pathways, particularly those ranked in higher positions. For example, five of the top ten pathways ranked by the combined methods (Method 4) would be ranked differently depending on whether the minimum, intermediate or maximum number of species was taken into account. Temporal change also affected pathway ranks, with both the number of species introduced by pathways and the impact of pathways changing over time. Overall, these findings suggest that a combined approach to pathway ranking, taking into account impact, uncertainty and temporal change is likely to perform better than other methods. They also demonstrate that uncertainty should be clearly documented and communicated to support decision-making.

Using combined methods (i.e. Method 4) to rank pathways provided much of the information needed to support pathway prioritisation (Essl et al., 2015); however, it did not include an assessment of the feasibility of pathway management. This is critical for prioritisation as the management of some pathways will be more feasible than others which, with limited resources, may influence management decisions (Lodge et al., 2016). For example, it may be relatively feasible to introduce measures to reduce the risk of zoo escapes in GB (e.g. restrictions on keeping, codes of practice, regulation of holding facilities) but much harder to prevent species arriving via the unaided pathway from continental Europe (e.g. Asian hornet (Marris et al., 2011) or Asian shore crab (Seeley et al., 2015)). Methods to assess the feasibility of pathway management are therefore required and should complement the scoring methods identified here. These could use similar criteria to those used to assess the feasibility of managing species such as the effectiveness, practicality, cost, negative consequences and acceptability of pathway management (Booy et al 2017). Indeed, the need to assess the feasibility of pathway management is similar to that required for the prioritisation of species management, which is discussed later in this thesis (Chapters 4 and 5).

Central to the ability to analyse and rank non-native species pathways is the use of robust pathway classification systems (Essl et al., 2015). While adopting the CBD classification (CBD, 2014b) helps to ensure consistency, it is likely to require updates and improvements as it continues to be applied (Harrower et al., 2018a). For example, with non-native species introduced to GB it was useful to add a level of detail to some of the particularly broad CBD pathways (i.e. plant contaminants and animal contaminants). In these cases, pathways were separated based on the sectors involved (e.g. agriculture, aquaculture, fisheries, forestry, ornamental) as it is generally at this level that management intervention would occur. Indeed, it may be useful in classification systems to consider breaking all pathways down to units at which management is likely to be feasible. It was possible to map NNSIP pathways to CBD pathways (similar to findings of Saul et al. (2017) for DAISIE and GISD pathway categories, as well as Tsiamis et al. (2017) for EASIN); however, it was rare that pathways mapped directly without at least some manual corrections. This was primarily because of differences in the way the NNSIP and CBD classifications were structured (e.g. NNSIP grouped all accidental introductions, while CBD separated contaminants and stowaways), the level of pathway detail used by each classification scheme and ambiguity in the interpretation of pathways. These findings highlight a challenge for the coordination of pathway management at an international scale. On one hand such schemes need to be consistently applied, but on the other they need to improve and develop as lessons are learned from their application. In addition, even with extensive guidance (Harrower et al., 2018a) pathway definitions can still be ambiguous. There is therefore a need to determine how such schemes can be updated and ambiguities clarified while maintaining consistency. This could potentially be done through the development of standards (in a similar way to International Standards for Phytosanitary Measures, https://www.ippc.int) which could be developed and maintained at an international level, for example via platforms such as the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services or the IUCN’s Invasive Species Specialist Group.

Interestingly, pathways that introduced many non-native species tended to be associated with lower proportions of invasive species (e.g. horticultural escapes), while pathways associated with high proportions of invasive species tended to introduce relatively few species overall (e.g. angling stowaways). No pathways introduced both many species and a high proportion that were invasive; although some of the high ranking pathways introduced relatively large numbers of both (e.g. hull fouling and ballast water). The relationship between number of species introduced and proportion invasive could help to indicate different management strategies to reduce risk. For example, pathways that introduce large number of species but few that cause severe impacts are likely to require selective management methods. These could include blacklisting for intentional (release and escape) introductions (e.g. Essl et al., 2011b) or for unintentional pathways (contaminant and stowaway) methods targeting specific high risk routes, origins, vectors or activities (EU, 2014a, e.g. Haack et al., 2014). On the other hand, broader interventions may be more appropriate for pathways that introduce few species of which a large proportion are invasive. This could include white listing for intentional pathways, which would focus on allowing only the relatively small number of low impact species to be kept / used (Hulme, 2015). While for unintentional introductions, broad biosecurity measures may be required to reduce risk across activities (e.g. Anderson et al., 2014).

This is one of the first pathway ranking studies to explicitly incorporate uncertainty in the original pathway of introduction in its outputs. In doing so, this study showed that pathway uncertainty was common for many (36% of) species and particularly so for aquatic species and invertebrates. Uncertainty had the potential to affect the results of ranking. For example the wide error bars on the pathway ranked highest by the combined methods (hull fouling) showed the maximum impact could be twice that indicated by the intermediate impact score, while the minimum impact would reduce its rank from first to at most third place. The method used to incorporate uncertainty here was based on the number of pathways listed for each species. This provided a useful means of both assessing and displaying uncertainty; however, there were a number of limitations. Where only one pathway was listed for a species it was assumed that this was the original introduction pathway. However, it is possible in some cases that even though a single introduction pathway was listed there was still uncertainty that it was the original pathway (e.g. *Potamopyrgus antipodarum* was most likely associated with drinking water barrels (Ponder, 1988) but it is difficult to know this with certainty). Similarly, where multiple pathways were listed for a species, it was assumed that each had an equal chance of being the original introduction pathway; however, some may have been more likely to be the original than others (e.g. *Dreissena bugensis* may have been introduced by numerous pathways; however some, such as hull fouling, seem more likely than others (Bij de Vaate et al., 2013)). Future studies should therefore consider ways of adapting the methods used here to assign specific confidence or probability scores (e.g. following those used by Mastrandrea et al., 2011, Hawkins et al., 2015) to species associated with each pathway.

In order to determine a minimum, intermediate and maximum number of species per pathway it was necessary to only consider the original (first) introduction pathway for these species, with subsequent pathways of introduction (i.e. those that introduced further individuals of the same species) excluded from analysis; although other studies have not made this distinction (e.g. Pysek et al., 2011, CBD, 2014c, Roy et al., 2014c, Pergl et al., 2017, Saul et al., 2017, Van Gossum and Rommens, 2017). This was, in part, because it was not known with confidence the degree to which subsequent pathways contributed to the establishment and spread of additional populations (and therefore the impact of the species). Subsequent pathways may be inconsequential, for example they may not lead to any further populations, or may add individuals to an already widespread population therefore causing little additional impact. On the other hand, subsequent pathways could cause considerable additional impact, for example the escape of fallow deer from deer parks has likely facilitated the spread of this species throughout GB, despite release for hunting being the original introduction pathway (Lever, 2009). This issue highlights some of the complexity involved in pathway analysis and the need to account for trends in the impacts of pathways overtime. Further development is needed to account for subsequent pathways, which would need to consider not only when these became active but the extent to which they have contributed to each species' impact.

Two different methods for incorporating species impact were tested in this study, with advantages and disadvantages to each. Method 2a ranked pathways by counting only the number of species that were considered invasive (i.e. scored more than minimal impact). This had the advantage of using invasive species as the unit of measurement, which is more intuitive and can be communicated clearly. However, a disadvantage was that information was lost, because differences in the severity of impact between invasive non-native species were not taken into account (i.e. minor impacts were treated the same as massive). The alternative, Method 2b, converted all categorical impact scores into values, with rank determined by the sum of these values. Applying post-hoc values to categories in this way can be problematic as they may not accurately reflect the distances between qualitative levels; however, a logarithmic scale was considered a good fit as the qualitative categories were designed to reflect impacts at increasing orders of ecological organisation (Blackburn et al., 2014, Hawkins et al., 2015). This approach also produced results that correlated closely with those produced using Method 2a, which indicated that both methods came to a similar conclusion. Method 2b was considered the more appropriate for use in the final analysis of pathways in GB as it used all species (rather than excluding a proportion) and provided a means of distinguishing between impact levels.

The importance of taking temporal changes into account when assessing pathways was demonstrated. The number of species introduced by pathways changed over time, as did the impact of pathways, resulting in considerably different pathway ranks between fifty year time periods. While the horticultural pathway historically introduced by far the most species over time and this was still the case by the end of the 20th century; in terms of impact the horticultural escape pathway appeared to plateau after 1900, while horticultural contaminants and hull fouling increased rapidly. This result should be treated with caution, as the plateau in the horticultural pathway may be the result of lag in detecting impact (Chapter 2); however, it highlights that the threat from some pathways is changing considerably and increasing rapidly in the case of horticultural contaminants and hull fouling. It is therefore important that change over time is incorporated into scoring methods used to rank pathways, as has been recommended (Wilson et al., 2009, Essl et al., 2015, Zieritz et al., 2016, García-Díaz et al., 2018). The combined approach used here (Method 4) did this relatively simply by limiting assessment to those species that established after 1950. This is a point after which technical and logistical improvements have resulted in the increased spread of species (Hulme, 2009, Essl et al., 2011a) and was considered sufficiently recent to include the most relevant modes of transport (by air, sea and over-land), while providing a large enough dataset on which to perform analysis. Future development should consider ways by which the trajectory of a pathway's impact over time (i.e. the rate at which impact is increasing or decreasing) could be further incorporated, perhaps by modelling predicted future pathway impacts (Lodge et al., 2016).

Historical trends in pathway impacts do not necessarily indicate future risk; nevertheless, recent trends in the impact of introduction pathways may provide some insight. The accuracy of this will only be tested over time; however, an indication may be provided by comparing the pathways identified here to those predicted to introduce future invasive species as identified by horizon scanning (Roy et al., 2014b). Twenty out of the top 30 species identified by horizon scanning were associated with high ranking pathways identified by this study (primarily hull fouling or horticultural escapes), suggesting a good alignment between the pathways identified here and those identified by horizon scanning; however, there were also differences. None of the top 30 species were predicted to be introduced as contaminants of ornamental plants (four were forestry contaminants, but these were not considered contaminants of ornamental plants), and the remaining ten species were associated with at least six different pathways (aquaculture escapes, contaminants of fish, unaided, forestry contaminants, contaminants of raw material and contaminants of produced). Overall, it would appear that using past trends in pathway impact is a potentially useful proxy for near future risk; however, for more long distance forecasts more predictive ways of modelling future risk would be required (Lodge et al., 2016). These would need to consider the effect of lag in the detection of impacts (Chapter 2) and the role of propagule pressure (Lockwood et al., 2009, Cassey et al., 2018a), which is linked to changes in global markets, demographics and climate (Hulme, 2015, Seebens et al., 2015).

This study helps to indicate potential pathway management priorities in GB, notwithstanding the need to assess the feasibility of pathway management. Despite numerous introduction pathways that introduced non-native species to GB (n=38 at sub-category level), relatively few were responsible for the majority of impacts since 1950 (three pathways were responsible for the majority of post-1950 impact). Nineteen pathways had negligible impact (i.e. introduced species that exclusively caused minimal impact), despite several introducing large numbers of species (e.g. agricultural escapes, food contaminants, seed contaminants). This demonstrates an immediate advantage of ranking, that relatively large and complex lists of pathways can be reduced to more manageable short lists. Of the higher impact pathways, hull fouling, horticultural escapes and contaminants of ornamental plants stood out as pathways that caused most impact, with the impact of hull fouling and contaminants of ornamental plants increasing rapidly in recent years. Both hull fouling and horticultural escapes have already been identified as priorities in GB (Defra, 2015), which is supported by this result. Both marine and freshwater hull fouling are a concern, with marine species being introduced from a wide range of native origins (but particularly the pacific ocean) and invasive freshwater species predominantly having their native origins in Europe (which are primarily assocaited with the Ponto-Caspian region of south-eastern, Gallardo et al., 2015). Horticultural escapes had native origins from across the globe; however, aquatic plants with native origins in North America appear to be associated with a disproportionate degree of impact (see also Chapter 2). While contaminants of ornamental plants is often considered a risk from the perspective of plant health (e.g. Halstead, 2011, Scrace, 2018), it has not received as much attention in relation to introducing invasive non-native species that pose a wider environmental threat. This may deserve more attention in GB, particularly in relation to terrestrial invertebrates with native origins in Australia. However, managing these may not be straightforward, given that they often go undetected in imports and can be difficult to control (Sluys, 2016). Beyond the top three pathways, ballast water, angling stowaways, stowaways 'other' and contaminants of aquaculture animals all caused relatively high impacts, but with considerable uncertainty. Further investigation may be necessary to attempt to reduce uncertainty here where feasible; however, following the precautionary principle these pathways could be considered high risk until proven otherwise. While these are the highest ranking pathways based on species established since 1950, this does not preclude the possibility that other pathways could be a priority, particularly when the feasibility of management is taken into account. For example, contaminants of imported fish have been identified as a serious threat in the past (Pinder and Gozlan, 2003) and it may be relatively straight forward to tighten existing controls on this pathway to reduce future risk.

This study shows the value of comprehensive datasets of all non-native species established in a given region for helping to identify pathways that have not only introduced the most species, but have had the most impact. The methods developed and tested here show that impact, temporal change and uncertainty should be taken into account when prioritising pathways. This finding has relevance to those responding to national (Defra, 2015), regional (EU, 2014a) and international commitments (CBD, 2014b) to prioritise prevention effort. It is hoped that the methods developed here will contribute to those initiatives. Nevertheless, further development is required, not only to improve our ability to forecast the risk of future pathways, but in particular to assess the feasibility of pathway management in order to support prioritisation. While this analysis used historical information to indicate which pathways may be a priority in the near future, it does not take into account potential future changes to pathways caused, for example, by the opening of new markets, changes in technology or climate change. It would be useful to consider how these may affect future trends and be ready to update analysis as substantial changes to pathways occur.

Declaration

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Chapter 4. Risk management to prioritise the eradication of new and emerging invasive non-native species

4.1 Abstract

Robust tools are needed to prioritise the management of invasive non-native species. Risk assessment is commonly used to prioritise invasive non-natives, but is of limited use because the feasibility of management is not considered. Risk management provides a structured evaluation of management options, but has received little attention to date. A risk management scheme is presented to assess the feasibility of eradicating invasive non-natives that can be used, in conjunction with existing risk assessment schemes, to support prioritisation. The Non-Native Risk Management scheme (NNRM) can be applied to any predefined area and any taxa. It uses semi-quantitative response and confidence scores to assess seven key criteria: Effectiveness, Practicality, Cost, Impact, Acceptability, Window of opportunity and Likelihood of re-invasion. Scores are elicited using expert judgement, supported by available evidence, and consensus-building methods. The NNRM was applied to forty-one invasive non-natives that threaten Great Britain (GB). Thirty-three experts provided scores, with overall feasibility of eradication assessed as 'very high' (8 species), 'high' (6), 'medium' (8), 'low' (10) and 'very low' (9). The feasibility of eradicating terrestrial species was higher than aquatic species. Lotic freshwater and marine species scored particularly low. Combining risk management and existing risk assessment scores identified six established species as priorities for eradication. A further six species that are not yet established were identified as priorities for eradication on arrival as part of contingency planning. The NNRM is one of the first invasive non-natives risk management schemes that can be used with existing risk assessments to prioritise invasive non-natives eradication in any area.

4.2 Introduction

There are many non-native species worldwide (e.g. 50,000 in the USA (Pimentel et al., 2005), 12,000 in Europe (DAISIE, 2009), 2,000 in Great Britain (Roy et al., 2014c)) and the number is increasing (Hulme, 2009, Roy et al., 2014c). Between 5 and 20% of these are invasive (McGeoch et al., 2016) causing serious negative environmental, economic and social impacts (Clavero and Garcia-Berthou, 2005, Pimentel et al., 2005, Pejchar and Mooney, 2009, Simberloff et al., 2013, D'hondt et al., 2015, Kumschick et al., 2015b, Bellard et al., 2016). Decision-makers are under growing pressure to respond (Hulme, 2006); however, invasive non-native species management is costly and, with limited resources, must be carefully prioritised (Early et al., 2016, McGeoch et al., 2016). Yet practical and robust tools to support the prioritisation of management are lacking (Hulme et al., 2009).

One way of prioritising invasive non-native species management is to use risk analysis, which traditionally includes two separate components: risk assessment, used to identify the threat or hazard posed by a species; and risk management, used to evaluate management options (FAO, 1995). It is the balance between these two that allows for prioritisation, with high risk species for which management is cost-effective being prioritised first and low risk species for which management is expensive and ineffective prioritised last. Both risk assessment and risk management are essential for prioritisation; however, while numerous invasive non-native species risk assessment schemes have been developed (for reviews see Verbrugge et al., 2010, Heikkilä, 2011, Leung et al., 2012, Roy et al., 2014a, Early et al., 2016) very few exist for risk management (Heikkilä, 2011). Of the schemes that do include elements of risk management, many only include one or few questions (e.g. Branquart, 2007, Essl et al., 2011b) or provide an evaluation of what is advisable, but not an indication of priority (Schmiedel et al., 2016). While more elaborate schemes are available for weed risk management and plant health pests, these are limited by being taxonomically or sector specific (e.g. Hiebert and Stubbendieck, 1993, Baker et al., 2005, Setterfield et al., 2010, Virtue, 2010, Auld, 2012, Kehlenbeck et al., 2012, Sunley et al., 2012, Drolet et al., 2014, Firn et al., 2015a, Firn et al., 2015b), consider only specific aspects of risk management (e.g. Darin, 2008, Hauser and McCarthy, 2009, Darin et al., 2011) or being time and resource intensive (e.g. Vander Zanden et al., 2010, Darin et al., 2011, Leung et al., 2012, McGeoch et al., 2016).

There is, therefore, a need for a practical risk management scheme that is compatible with existing risk assessment schemes in order to support prioritisation of invasive non-native species (D'hondt et al., 2015). Given the range of species that become invasive, such a scheme should be broadly applicable to any taxa (Nentwig et al., 2010) and, given large numbers of species involved, should be efficient to apply (Andersen et al., 2004, Hulme et al., 2009). It should be possible to complete the scheme even where data are lacking, with uncertainty taken into account, documented and justified (Leung et al., 2012).

This study set out to develop a scheme, known as the Non-Native Risk Management Scheme (NNRM), that meets these criteria and complies with international standards for risk management (FAO, 1995, FAO, 2006, OiE, 2017) as well as good practice for prioritisation (summarised by Heikkilä, 2011). It is focussed on assessing the feasibility of eradication (sensu Genovesi, 2000), acknowledging that eradication is the most effective management response after prevention (Genovesi, 2005). It is also the focus of the second tier in the hierarchical approach to invasive non-native species management (Guiding Principle 2, COP 6 decision VI/23, Convention on Biological Diversity) as well as an important component of Aichi Target 9 (UNEP, 2011). To ensure it could be practically applied and completed even where data were limited, the NNRM was designed to use expert judgement (Martin et al., 2012) to provide semi-quantitative scores (sensu Baker et al., 2008) which are justified by written comments, and supported by evidence where available. This follows similar approaches used for risk assessment (Baker et al., 2008, Essl et al., 2011b, Baker et al., 2012, Leung et al., 2012).

To demonstrate its use the scheme was trialled in GB, which has a well-developed and robust invasive non-native species risk assessment process but lacks a compatible process for risk management (Defra, 2015). The scheme was applied to a group of new and emerging invasive non-native species that pose a threat to GB, as these were considered most likely to be potential candidates for eradication. This study demonstrates how the scheme can be used, in combination with existing risk assessment scores, to indicate priorities for eradication and contingency planning; and examine the importance of risk management for prioritisation. While applied here to GB, the scheme can be applied at different scales and in different areas worldwide. Indeed, the scheme may have particular application in the EU where the recent adoption of Regulation No. 1143/2014 includes requirements for eradication of invasive non-native species listed on the basis of both risk assessment and risk management variables (EU, 2014b).

4.3 Materials and methods

4.3.1 Development

The Non-Native Risk Management Scheme (NNRM) was developed over a 2-year period from 2013 to 2015 in collaboration with invasive non-native species management and risk analysis experts from Great Britain (GB). Initial criteria were developed in consultation with this group taking into consideration existing literature on invasive non-native species risk analysis and eradication (in particular Hiebert and Stubbendieck, 1993, Rejmánek and Pitcairn, 2002, Cunningham et al., 2003, Simberloff, 2003a, Baker et al., 2005, Genovesi, 2005, Cacho et al., 2006, Hulme, 2006, Genovesi, 2007, Randall et al., 2008, Simberloff, 2008, Johnson, 2009b, Mehta et al., 2010, Virtue, 2010, Kehlenbeck et al., 2012, Sunley et al., 2012). Refinements were made to the scheme during an initial trial in March 2014 and subsequently the expert elicitation and consensus-building process described below. Decision-makers were engaged in the initial development of the scheme and at intervals throughout the process to ensure the relevance of the scheme for them as end-users.

4.3.2 The Non-Native Risk Management Scheme

The NNRM takes the form of a questionnaire supported by guidance (a modified version of which is provided at Appendix D), which is summarised in Box 4.1. Preliminary stages record the details of authors, the organism to be assessed, the risk management area and the objective of the assessment. The risk management area is user defined to allow any area to be assessed, but must be precisely defined. The objective of the assessment is set from the outset as the complete eradication of the organism from the risk management area (sensu Genovesi, 2000).

Box 4.1 Summary of guidance provided to complete risk management assessments; the full scheme is available (Appendix D)

1. Define the invasion scenario. For species that are already established this is the current extent of the species in the risk management area. For species on the horizon this is the most likely extent of the species in the risk management area at the point detection could reasonably be expected (based on existing surveillance).
2. Define the eradication strategy. Based on the defined scenario briefly describe the eradication strategy being assessed. This should be a realistic strategy you consider most likely to be effective in eradicating the species completely from the risk management area. The overall strategy could include multiple methods (e.g. use of pesticides, herbicides, trapping, shooting, etc.) and should include any other work that would be required such as surveys, logistics and monitoring.
3. Assess the eradication strategy:
    a) Effectiveness. How effective would the eradication strategy be? This relates to how effective the defined strategy would be if it could be deployed regardless of issues with practicality, cost, impact and acceptability.
    b) Practicality. How practical would it be to deliver the eradication strategy? This includes issues such as gaining access to relevant areas, obtaining appropriate equipment, skilled staff or pesticides. If there are any legal barriers to undertaking the work these are assessed here.
    c) Cost. How much would the eradication strategy cost? This is the total direct cost of the strategy including materials, staff time and any other direct costs. Indirect costs, such as loss of business, are taken into account under negative impact (3d).
    d) Negative impact. What negative impact would the eradication strategy have? Assess the impact that the eradication strategy itself would have on the environment, economy or society.
    e) Acceptability. How acceptable is the eradication strategy? Could the eradication strategy meet significant disapproval or resistance from the general public, key sectors or any other stakeholder?
4. Assess the window of opportunity for delivering the described eradication strategy. How quickly will the species spread beyond the point that eradication, using the defined strategy, would be effective?

5. Assess the likelihood of re-invasion following eradication. Unless the eradication strategy deliberately targets populations in containment or otherwise not in the wild (i.e. in gardens, zoos, etc.) introduction from these sources should be considered potential sources of re-invasion. If relevant, the eradication strategy could include pathway management measures in order to reduce this score.
6. Overall feasibility of eradication. Taking into account all preceding scores, provide an overall score for the feasibility of eradicating this species from the risk management area.

Once preliminary stages are complete, the assessment is started by documenting the invasion scenario (Box 4.1, step 1), which describes the extent of the invasive non-native species in the risk management area (see guidance in Appendix D for detail). The scenario may be based on known existing or predicted future invasions, as well as probabilistic scenarios such as best, most likely, or worst case scenarios; however, for assessments to be comparable the scenario selected must be consistent (to this end the most likely scenario was adopted for all assessments in the trial described below). Multiple scenarios may be considered for individual species, in which case each scenario is assessed separately. In all cases assessors should carefully document the scenario being considered, along with any assumptions made, to provide context for the results.

The eradication strategy is then defined (Box 4.1, step 2). This is a realistic strategy considered likely to achieve complete eradication of the species from the defined risk management area and can include any combination of individual methods (e.g. use of pesticides, herbicides, trapping, shooting, etc.). Multiple eradication strategies can be considered if necessary to allow for comparison between different approaches, in which case each strategy should be separately assessed. Assessors determine which strategy they consider likely to achieve eradication, avoiding being too conservative (i.e. no eradication possible despite techniques being available) or unrealistic (i.e. cost / damage caused vastly outweighs potential benefits). As with the invasion scenario, defining the eradication strategy at this point allows for assumptions to be documented and a clear basis for the rest of the assessment to be set.

The feasibility of eradication, based on the defined eradication strategy, is then assessed using seven key questions relating to Effectiveness, Practicality, Cost, Impact, Acceptability, Window of opportunity and Likelihood of reinvasion (Box 4.1, steps 3a-e, 4 and 5). Lastly, the assessor provides a single overall score for the feasibility of eradication (Box 4.1, step 6), which is based on their expert judgement taking account of the scenario and responses made in the previous steps. The overall score is not directly calculated from individual scores, because no appropriate weighting could be identified that would account for the wide range of taxa and criteria being assessed (Mumford et al., 2010). Instead expert judgement based on previous steps was used, which follows the approach used by the UK, EPPO and other risk assessment schemes (Mumford et al., 2010, Baker et al., 2012) and provides flexibility, while ensuring overall scores are supported by individual scores and documented justification.

Table 4.1 Assessment criteria for GB non-native risk management response scores, 1 is least favourable and 5 the most.

Criteria	**Response Score**				
	1	**2**	**3**	**4**	**5**
Effectiveness	Very ineffective	Ineffective	Moderate effectiveness	Effective	Very effective
Practicality	Very impractical	Impractical	Moderate practicality	Practical	Very practical
Cost	>£10M	£1-10M	£200k-1M	£50-200k	<£50k
Negative impact	Massive	Major	Moderate	Minor	Minimal
Acceptability	Very unacceptable	Unacceptable	Moderate acceptability	Acceptable	Very acceptable
Window of opportunity	< 2 months	2 months - 1 year	1 – 3 years	4-10 years	>10 years
Likelihood of reinvasion	Very likely	Likely	Moderate likelihood	Unlikely	Very unlikely
Conclusion (overall feasibility	Very low	Low	Medium	High	Very high

4.3.3 Response and confidence scores

For each of the seven questions and the overall conclusion a response and confidence score are required with justification provided by a written comment. Response scores are ordinal on a five-point scale with one being least favourable and five being most (Table 4.1). Each alternative response is predefined using descriptive terms (similar to those used in risk assessment schemes, e.g. Baker et al., 2008, Baker et al., 2012), except for Cost and Window of opportunity which is based on quantified bands. Bands for Cost scores were determined in consultation with decision-makers that hold national budgets for invasive non-native species control and reflect the range of costs associated with historical eradication attempts that have been made in GB (if applied to other countries / regions these bands may need to be recalibrated). Window of opportunity was quantified in consultation with risk management experts to reflect timescales likely to be relevant to a wide range of taxa. Confidence scores are explicitly recorded for every response using a three point scale (low, medium high) following Mumford et al. (2010), which in turn is based on a simplification of guidance provided by the Intergovernmental Panel on Climate Change (Mastrandrea et al., 2011).

Table 4.2 Establishment status and environment of species used to test the risk management scheme

Taxa	**Environment**			**Status in GB**	
	Terrestrial	**Freshwater**	**Marine**	**Established**	**Not established**
Plants	5	5	1	8	3
Vertebrates	10	3	0	6	7
Invertebrates	2	8	7	8	9
Totals	*17*	*16*	*8*	*22*	*19*

4.3.4 Applying the scheme to new and emerging threats to GB

The scheme was used to assess 41 new or emerging invasive non-native species that pose a threat to GB and represent a broad range of taxa and environments (Table 4.2). Twenty species were already established in GB at the time of assessment, but with limited distributions; a further 21 were horizon species, defined as species not established in GB at the time of assessment but considered likely to invade in the near future. The list of horizon species was based on the top 30 threats identified by Roy et al. (2014b), less nine species that were excluded. Species were excluded if they were primarily crop, forestry or fish pests and dealt with by established plant or fish health regimes in GB; or were species that had already established in GB by the time of assessment, in which case they were included as established species. The remaining established species were selected based on their limited distributions in GB and because they were being considered for potential eradication by decision-makers (N. Moore 2016, *pers comm*). The most likely scenario was used for all species, which for established species was defined as the species' current extent, and for horizon species was the most likely extent at the point of detection with existing surveillance.

Expert judgement (supported by evidence where available) was used to elicit scores, which is practical but must be used carefully to minimise the impacts of subjectivity, bias and group think (Burgman et al., 2011, Martin et al., 2012, Sutherland and Burgman, 2015). To this end the approach used by Roy et al. (2014b) was followed, which combines expert elicitation with review and consensus building to reduce these effects, while still being practical and efficient to apply. Techniques incorporated within this approach include: a) the structured use of groups rather than individuals to produce scores, b) independent initial scoring followed by review and consensus building; c) transparent, documented justification of all scores; d) initial presentations and discussion around the scoring method and terminology to reduce the potential for language based misunderstanding; e) open facilitator-led discussions to encourage all participants to listen to one another, asses each other's judgements and cross examine reasoning behind scores; f) breakout sessions to provide smaller and more informal space in which to express views; and g) agreeing final scores through a facilitator-led discussion where every participant was directly invited to comment on each score. No attempt was made to weight individual expert judgements because of practical problems associated with constructing reliable and valid weights (Bolger and Rowe, 2015).

In total, 33 experts were engaged in the elicitation process (Appendix E) divided into four groups comprising 7-10 experts each: freshwater animals; terrestrial animals; marine species; and plants, excluding marine plants. Experts were selected based on their proven experience of invasive non-native species management in GB and diversity of background (i.e. government, non-government, practitioners, academics and policy advisors). Initial risk management assessments were drafted over a period of 7 weeks by experts from each group using published or grey literature to support scores and expert judgement where other forms of evidence were lacking or inconclusive. The task of completing assessments was shared between experts, with each species being assessed by a single expert and then reviewed by all others in the same expert group.

Drafted risk management assessments were then used as the basis for a consensus building workshop held on 28 April 2015 and attended by 19 of the original experts (limited due to availability). The first phase of the workshop commenced in plenary with a presentation and discussion around the criteria and scoring approach, followed by presentations of initial scores by group leaders with all workshop participants invited to discuss scores and provide challenge. The aim of this exercise was to provide an opportunity to resolve any ambiguity about the process, encourage consistency in scoring between expert groups and review scores. After initial scores had been considered by all participants the expert groups were reformed to discuss and agree alteration of scores if necessary.

In the second phase of the workshop the facilitators presented the refined scores for all species in plenary to all participants. Participants were asked to review and modify these scores if necessary. By the end of this second phase, all response and confidence scores were agreed by the consensus of all participants.

4.4 Analysis

The individual relationships between overall score and the sub-scores were examined for the seven detailed risk management questions using polychoric correlations as the scores were ordinal from 1 to 5 (see Table 4.1.). Polychoric correlations measure the agreement between multiple ordinal variables and are usually used where the ordinal variables are judgements (or ratings) of an underlying (latent) continuous variable (for example, comparing judgements made using Likert scales). In this case the ordinal variables are the scores given by experts for effectiveness, practicality, cost, impact, acceptability, window of opportunity and likelihood of reinvasion. Polychoric are preferred to Pearson's correlation for such analysis as the latter requires quantitative variables measured at intervals, whereas the former is used for truly ordinal data and is not affected by the number of rating levels (Holgado–Tello et al., 2010).

The relationship between risk management component scores for all 41 species and overall feasibility of eradication was further examined using a factor plot and non–metric multidimensional scaling (nMDS). The appearance of these plots are similar to Principle Component Analysis (although with important differences, (Holland, 2008) in that they reduce multiple variables to two dimensions designed to create the largest possible separation of data. The direction of the axes in both plots is the same, although variables (factor plot) and individual observations (nMDS) are measured using different scales. The two plots can therefore effectively be overlaid, with the direction of the arrows used to explore the relationship between variables (risk management components) and individual observations. Given the differences in scale between plots, only the direction of the arrows is relevant (length is not).

Within the factor plot (also known as a variable correlation plot) the contribution of each variable (i.e. each risk management component) to the two dimensions is shown by the length of the arrow, with longer arrows (closer to the outer circle) contributing more (also shown by the colour gradient). The direction of each arrow indicates the correlation of the variable with the two dimensions of the plot. Variables that are grouped together are positively correlated, while variables positioned on opposite sides are negatively correlated. Within the nMDS individual observations (i.e. species) are projected using the two dimensions (similar species are therefore grouped). Each observation is coloured according to overall feasibility of eradication, with 95% confidence ellipses presented, centred on the mean of each group.

Changes to all confidence scores (i.e. for each of the seven risk management questions and the overall score) were assessed from the initial scores to final scores at the end of the second phase of the workshop.

To indicate priorities for eradication a matrix was used to compare overall risk management scores with existing risk assessment scores. Within this matrix, species that scored the highest risk and highest feasibility of eradication were given greatest priority, while species that scored less on either axis were lower priority. A symmetric relationship between risk assessment and risk management scores was assumed, assuming equal importance of both risk assessment and the feasibility of eradication scores, such that a species of ‘high’ risk and ‘medium’ feasibility of eradication received the same priority as a species of ‘medium’ risk and ‘high’ feasibility of eradication. Risk assessment scores were derived from published data, with the GB Non-native Risk Assessment scheme (Baker et al., 2008) providing data for established species (published at www.nonnativespecies.org) and Roy et al. (2014b) providing data for horizon species. These two schemes differ in that the GB Non-native Risk Assessment scheme provides an overall score of high, medium or low risk; whereas horizon species were all assessed as high risk by Roy et al. (2014b) and were then further sub-divided into the top 10, top 20 and top 30 threats. This difference is reflected in the two prioritisation matrices produced.

The correlation between overall risk assessment (including horizon scanning) and risk management scores was assessed to examine whether these variables measured similar underlying information. Polychoric correlations were used, as the data were ordinal, with overall scores for emerging and new species tested separately as the risk variable differed for these groups (i.e. for emerging species risk was categories as low, medium high; whereas for new species it was measured as top 10, top 20 or top 30 risk).

All coding was undertaken in R version 3.4.1 (R Core Team, 2017) primarily using the tidyr package (Wickham and Henry, 2018), polycor (Fox, 2016) and ecodist (Goslee and Urban, 2007).

Table 4.3 GB risk management scores. Species are grouped according to the overall feasibility of eradication from Great Britain. Colours and numbers reflect response scores (see Table 4.1) with overall feasibility of eradication scored from 1 (very low) to 5 (very high). Confidence, rated L (low), M (medium) and H (high), was recorded for all response scores, but for simplicity is only provided here for overall score. Broad taxonomic group (Invert. = invertebrate, Amp. = amphibian, Rept. = reptile, Mam. = mammal) is provided as well as main environment in which the species occurs (M = marine, F = freshwater, T = terrestrial)

a. Species already established in GB, but with limited distribution (emerging)

Species	Invasion scenario (brief summary)	Eradication strategy (brief summary)	Taxonomic group	Environment	Risk management scores: Effectiveness	Practicality	Cost	Impact	Acceptability	Window of opportunity	Likelihood of reinvasion	Overall feasibility of eradication	Overall confidence
Baccharis halimifolia (Sea Myrtle)	1 site on south coast of England	Hand removal and cut stump treatment with glyphosate	Plant	T	5	5	5	4	4	5	4	5	H
Procambarus acutus (White River Crayfish)	1 site, <4 angling ponds (lentic)	Remove specimen fish and treat with biocides	Invert.	F	4	5	4	5	4	3	4	5	H
Zamenis longissimus (Aesculapian Snake)	2 sites, each c. <100 individuals	Intensive capture and habitat manipulation	Rept.	T	5	5	4	5	3	4	4	5	H
Sarracenia purpurea (Purple Pitcher-plant)	14 populations in total covering <10 hectares	Combination of hand pulling and herbicidal treatment	Plant	T	4	4	4	4	5	5	3	4	H
Lacerta bilineata (Green Lizard)	1 population (<1000 individuals) in cliff terrain	Intensive capture and habitat manipulation	Rept.	T	4	3	3	4	4	4	3	4	H
Orconectes limosus (Spiny-cheek Crayfish)	3 pond and lake sites (lentic)	Biocides (as described by Peay et al, 2006)	Invert.	F	4	4	3	3	3	4	4	4	M

Cabomba caroliniana (Fanwort)	1 population (many colonies) in Basingstoke Canal (SSSI), total area c.800m^2 (lotic)	Repeated mechanical control (no herbicides available)	Plant	F	5	4	5	2	2	5	3	3	M
Hydropotes inermis (Chinese Water Deer)	2000+ individuals; core population in East Anglia, scattered elsewhere	Trapping and shooting	Mam.	T	5	3	2	5	3	4	4	3	M
Aponogeton distachyos (Cape Pondweed)	c. 80 scattered populations; well established in c. 75%, primarily in lakes and ponds (lentic)	Combination of manual and herbicidal treatment	Plant	F	4	3	3	4	3	5	3	3	H
Alopochen aegyptiacus (Egyptian Goose)	>5000 individuals over half in Norfolk others in Thames Basin and Midlands	Primarily shooting, supplemented by trapping	Bird	T	4	3	2	4	3	4	4	3	M
Ichthyosaura alpestris (Alpine Newt)	c. 40 populations, mainly in garden ponds (lentic)	Intensive capture and destructive techniques	Amp.	F	4	3	2	5	3	4	1	3	M
Podarcis muralis (Wall Lizard)	Well established in c. 40 sites with 20,000+ individuals	Intensive capture and habitat manipulation	Rept.	T	5	3	2	4	2	3	2	3	M
Egeria densa (Large-flowered Waterweed)	c. 95 scattered sites; well established in half, primarily in rivers, canals and ponds (lotic)	Repeated use of dyes and manual methods	Plant	F	3	2	1	2	3	5	3	3	H
Sagittaria latifolia (Duck Potato)	c. 40 populations in lentic and lotic systems, 50% well established	Combination of manual and herbicidal treatment	Plant	F	3	3	2	2	2	3	3	2	M
Orconectes virilis (Virile Crayfish)	1 population in the River Lee catchment (lotic)	High density trapping and male sterilisation	Invert.	F	2	2	2	2	1	4	3	2	M
Procambarus clarkii (Red-swamp Crayfish)	Populations in ponds, single river and canal (lentic and lotic)	Isolate waterbodies and treat with biocides	Invert.	F	2	2	2	2	1	3	2	2	M
Lysichiton americanus (American Skunk-cabbage)	c.800 populations across GB, 50% well established	Combination of manual and herbicidal treatment	Plant	T	4	2	1	3	2	1	1	2	M
Hemigrapsus sanguineus (Asian Shore Crab)	2 locations, potentially undetected elsewhere	Trapping supplemented by scuba collection	Invert.	M	1	2	2	4	4	3	2	1	H
Hemigrapsus takanoi (Brush-clawed shore crab)	2 locations, potentially undetected elsewhere	Trapping supplemented by scuba collection	Invert.	M	1	2	2	4	4	3	2	1	H
Dreissena bugensis (Quagga Mussel)	Single catchment, one tributary and main river	Biocides with damning / draw down where possible	Invert.	F	2	1	1	1	1	2	1	1	H

b. Horizon species (new species, not yet established in GB). Note that these species are not yet established in GB; the scenarios listed were considered the most likely invasion scenarios at the point of detection should the species invade in the future

					Risk management scores								
Species	**Invasion scenario (brief summary)**	**Eradication strategy (brief summary)**	**Taxonomic group**	**Environment**	Effectiveness	Practicality	Cost	Impact	Acceptability	Window of opportunity	Likelihood of reinvasion	**Overall feasibility of eradication**	**Overall confidence**
Nassella neesiana (Chilean Needle Grass)	2 small populations	Herbicide and follow up monitoring	Plant	T	5	5	5	4	5	4	4	5	M
Corvus splendens (House Crow)	1 population <10 birds	Shooting, supplemented by egg oiling	Bird	T	5	4	5	4	4	4	4	5	H
Tamias sibiricus (Siberian Chipmunk)	1 dispersed population, 10s of individuals	Trapping supplemented by shooting	Mam.	T	5	5	5	5	5	3	2	5	H
Procyon lotor (Raccoon)	1 population <10 individuals	Trapping	Mam.	T	5	5	5	5	4	3	2	5	H
Threskiornis aethiopicus (Sacred Ibis)	1 population (<10 birds)	Shooting, supplemented by trapping	Bird	T	5	4	5	4	3	3	2	5	H
Microstegium vimineum (Japanese Stiltgrass)	1 population	Hand pulling and herbicide	Plant	T	4	4	5	4	4	4	4	4	M
Ocenebra inornata (Japanese Sting Winkle)	1 population in a single oyster farm	Mechanical removal targeting eggs	Invert.	M	4	4	4	4	4	5	3	4	L
Nyctereutes procyonoides (Raccoon Dog)	1 population	Trapping	Mam.	T	4	4	4	5	4	3	2	4	M
Vespa velutina (Asian Hornet)	3 nests, some high in trees	Pesticide and nest destruction	Invert.	T	4	3	3	5	5	2	2	3	L

Homarus americanus (American Lobster)	2 well established populations establish off south coast	Trapping supplemented by scuba collection and possibly male sterilisation	Invert.	M	2	3	2	5	4	4	3	2	M
Linepithema humile (Argentine Ant)	1 population in garden	Application of insecticidal bait and post treatment monitoring	Invert.	T	3	3	5	4	3	3	2	2	M
Rapana venosa (Rapa Whelk)	Multiple populations in estuary (near oyster beds), partially accessible	Physical removal by whelk fishermen and volunteers	Invert.	M	2	3	3	5	4	3	2	2	H
Proterorhinus marmoratus (Tubenose Goby)	1 well established population in a salt and freshwater system	Rotenone based piscicides	Fish	F	2	2	2	2	3	2	2	2	H
Neogobius melanostomus (Round Goby)	1 well established population in a salt and freshwater system	Rotenone based piscicides	Fish	F	2	2	2	2	3	2	2	2	H
Corbicula fluminalis (Asian Clam)	1 well established population in a freshwater system (e.g. Norfolk Broads)	Application of potash	Invert.	F	2	2	2	3	3	2	1	2	M
Celtodoryx ciocalyptoides (A sponge)	Well established populations in multiple oyster farms	Hand removal and chemical treatment	Invert.	M	2	1	3	5	4	4	3	1	H
Myriophyllum heterophyllum (American Water-milfoil)	2 small populations on a canal (lentic)	Physical and chemical methods	Plant	F	1	2	3	3	3	3	2	1	H
Gracilaria vermiculophylla (Rough agar weed)	Well established populations discovered in multiple oyster farms	Mechanical removal	Plant	M	1	1	3	4	3	3	1	1	H
Echinogammarus ischnus (Bald urchin shrimp)	1 population in a freshwater system, not widely dispersed	Use of biocides	Invert.	F	2	1	3	1	1	2	2	1	H
Echinogammarus trichiatus (Curly haired urchin shrimp)	1 population in a freshwater system, not widely dispersed	Use of biocides	Invert.	F	2	1	3	1	1	2	2	1	H
Mnemiopsis leidyi (American Comb Jelly)	Well established population detected off south-east coast	No effective eradication methods	Invert.	M	1	1	1	1	1	1	1	1	H

4.5 Results

Risk management scores for all 41 established and horizon species were agreed by consensus (Table 4.3a, b). There was a broad spread of scores for overall feasibility of eradication, with 13-25% of the species falling into each of the five possible response categories (i.e. 1 - very low to 5 - very high).

The score for overall feasibility of eradication was most strongly correlated with the risk management components Practicality (polychoric correlation +/- standard error 0.97 +/-0.02), Effectiveness (0.93 +/- 0.03) and to a lesser extent Cost (0.64 +/- 0.1). The correlation was weaker between overall feasibility of eradication and Impact, Acceptability, Window of opportunity or Likelihood of reinvasion.

The factor plot (Fig 4.1) confirms that individual risk management components were broadly positively correlated with each other (grouped on one side of the plot) and none were negatively correlated. The data were too sparse to predict overall feasibility of eradication by modelling sub-scores (i.e. scores from each of the seven key risk management questions). However, accounting for inter-correlations the nMDS showed that overall assessment of feasibility of eradication broadly relates to the underlying sub-scores (Fig. 4.2). Dimension one of the nMDS correlated with overall feasibility of eradication, with minimal overlap of overall scores except between scores 1 and 2 (i.e. 'very low' overall feasibility and 'low' feasibility of eradication respectively). The orientation of the confidence ellipses (Fig 4.2) indicates that lower scores for overall feasibility of eradication may be more affected by impact and acceptability (i.e. the orientation of the ellipses for very low and low scores appears to be more in line with these two variables, Fig 4.1); whereas, higher scores may be more affected by effectiveness, practicality and cost.

Both response and confidence scores were refined during the workshop, with 26% of response scores and 58% of confidence scores modified during the first phase, and 5% of response and 2% of confidence scores further modified during the second phase. Confidence increased from the initial scores (proportion of all confidence scores: low=13%, medium=87%, high=0%) to the final scores at the end of the second phase (proportion of all confidence scores: low=8%, medium=39%, high=52%). A similar number of response scores increased as decreased. Changes in the response and confidence scores for the seven key risk management questions tended to result in similar changes to the scores for overall feasibility of eradication.

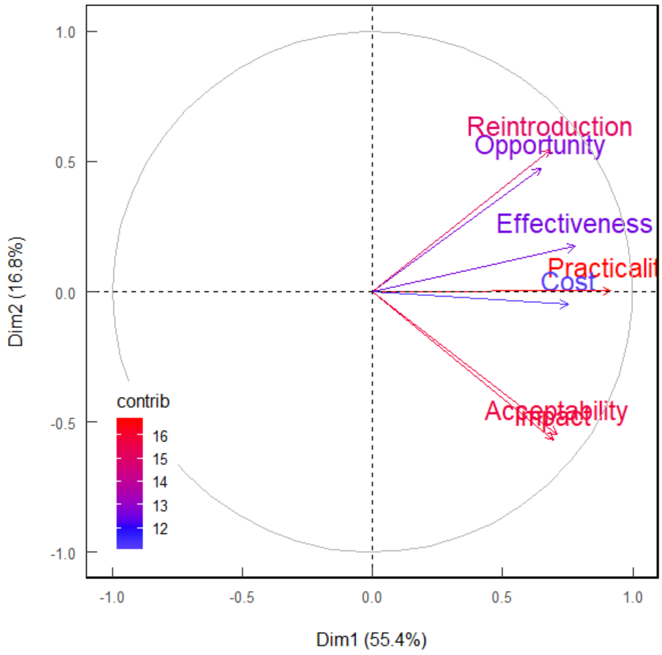


Figure 4.1. Factor analysis showing correlation between risk management sub-scores. The contribution of each factor to each dimension is represented by the length and colour of arrows and overall explain 72.2% of the variance in the data. Parallel arrows indicate correlation of factors. The direction of the axes in this plot is the same as that of the nMDS below.

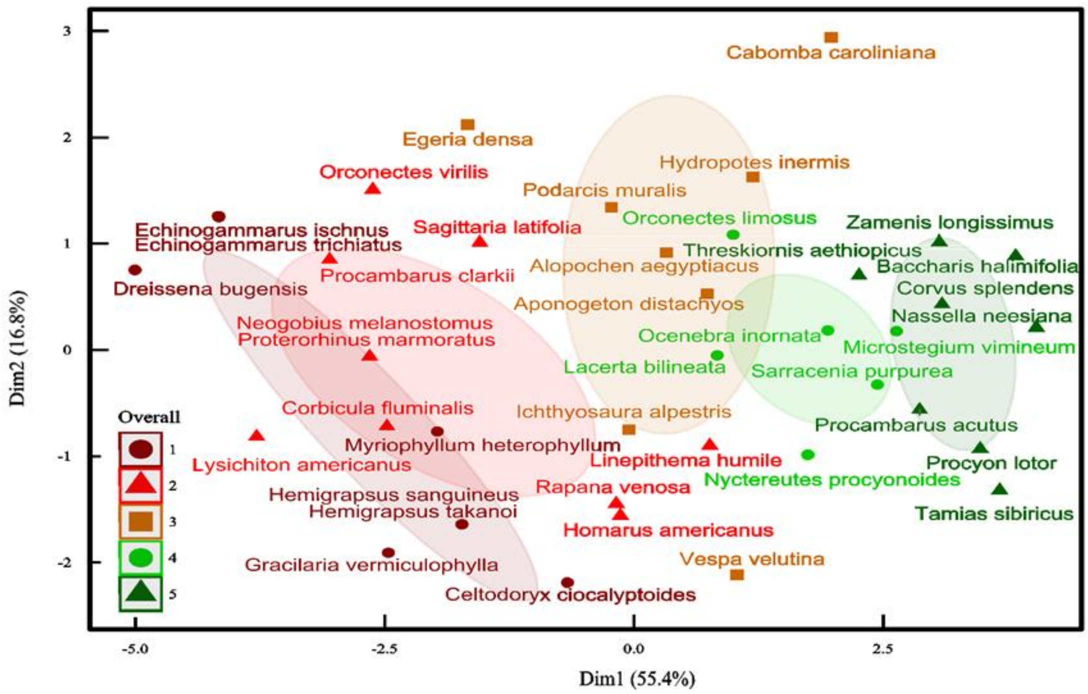


Figure 4.2. Non-metric multi-dimensional scaling (nMDS) of sub-scores with each species coloured by overall feasibility of eradication score. The shaded ellipses are a visual aid centred around the mean showing variation (scaled shape and size of the ellipse) of overall score. Dimension 1 correlated well with overall feasibility of eradication (1-5). The direction of the axes in this plot is the same as the factor plot above.

Differences in scores were found for overall feasibility of eradication between environments (Fisher's exact test, $p=<0.01$), with terrestrial species generally scoring 'very high', 'high' or 'medium' feasibility; freshwater species scoring 'medium' or lower feasibility; and marine species scoring 'low' or 'very low' feasibility (Fig. 4.3). Differences in scores were also found for overall feasibility of eradication between broad taxonomic groups (Fisher's exact test, $p=<0.01$), with more vertebrates receiving high scores, more invertebrates receiving low score and plants receiving a similar number of high, medium and low scores.

The scores for overall feasibility of eradication were combined with overall risk assessment scores to produce separate prioritisation matrices for established and horizon species (Fig. 4.4a, b). Overall, 12 of the 41 species assessed scored 'high', 'very high' or 'highest' priority for eradication. Established species were divided into four groups of differing priority ranging from 'very high' to 'low' priority with each group comprising 2-8 species and six species scoring 'high' or 'very high' priority. Horizon species were divided into seven groups of differing priority ranging from 'highest' to 'lowest' with each group comprising 1-5 species and six species scoring 'high', 'very high' or 'highest' priority.

There was no positive correlation between overall risk assessment and risk management scores. For established species the polychoric correlation (rho) between overall risk assessment and risk management scores was -0.6621 (estimated standard error of 0.169). For horizon species the polychoric correlation (rho) between horizon scanning scores (i.e. listed as top 10, top 20 or top 30 risk) and overall risk management score was 0.1631 with estimated standard error of 0.2599. This indicates a possible weak negative correlation in the first group and no correlation in the second. However, the weak correlation is likely to be because few low risk emerging aquatic species were included in the dataset (which are more likely to receive low scores for feasibility of eradication).

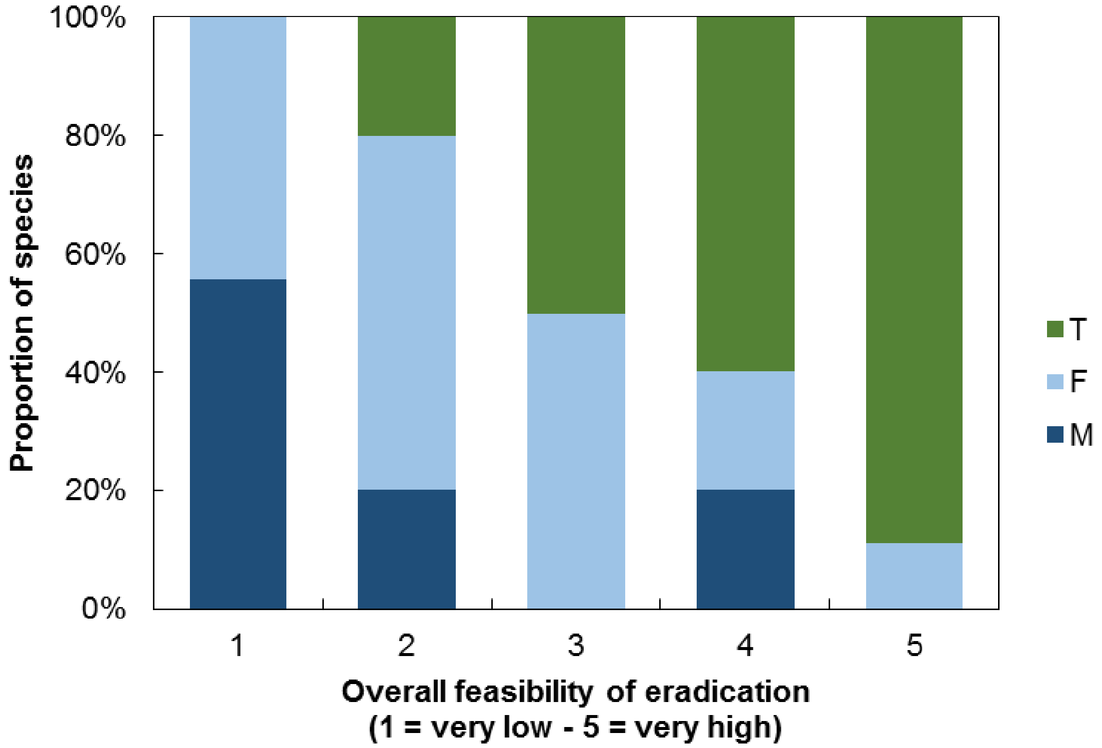


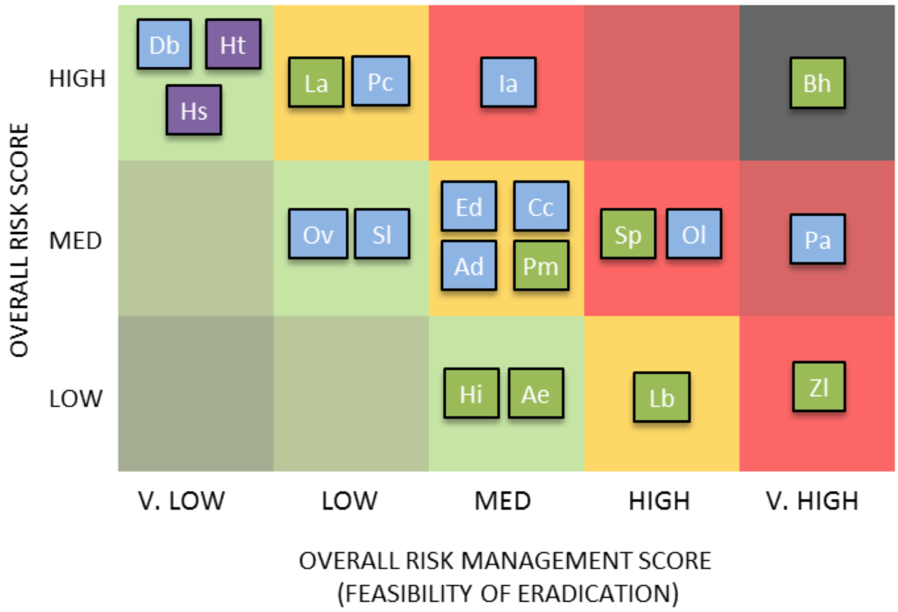
Figure 4.3. Overall feasibility of eradicating species based on environment: T = terrestrial, F = freshwater, M = marine. Overall feasibility of eradication is shown as a proportion of the species assessed

4.6 Discussion

This study demonstrates that the NNRM is a practical scheme that can be used to assess a wide range of taxa from different environments and directly compare them according to the overall feasibility of eradication. It complies with international standards for risk management (FAO, 1995, OiE, 2017) and good practice for non-native species prioritisation (summarised by Heikkilä, 2011) and is compatible with existing risk assessment schemes (Baker et al., 2008, Baker et al., 2012). In conjunction with risk assessment scores, the NNRM can be used to indicate priorities for eradication of existing and future invasive non-native species. With increasing legislative requirements to manage invasive non-native species, decision-makers require a rapidly applied, transparent and defendable process by which eradication actions can be prioritised for established species, and contingency plans developed for horizon species (Early et al., 2016). Not only does the NNRM facilitate risk based policy making in relation to the eradication of invasive non-native species, but also indicates other potential management actions where feasibility of eradication is low (e.g. targeted measures to prevent introduction or containment measures) as well as providing broad estimates of cost allowing for more effective budget management. While applied here to GB, the scheme can be applied to any defined area.

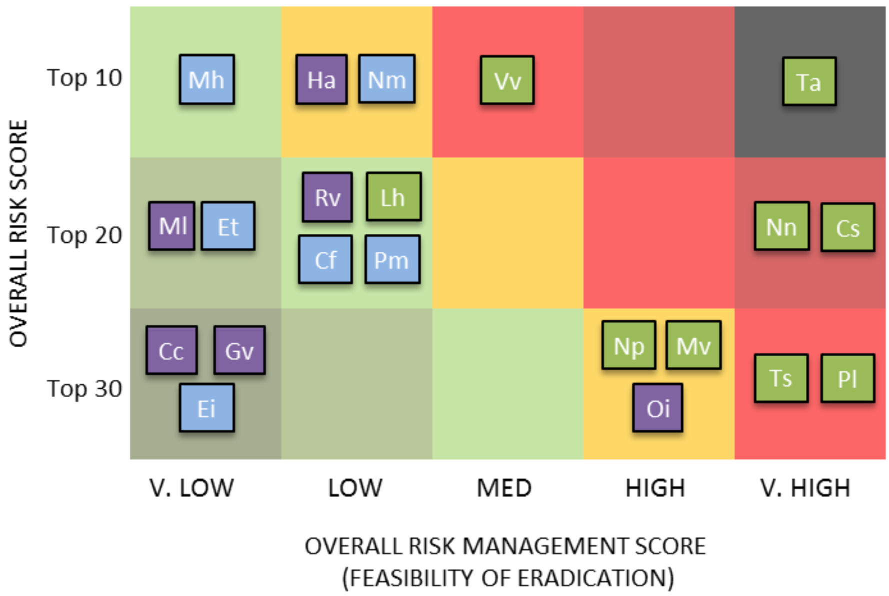
Expert scoring, based on predefined semi-quantitative scales, coupled with consensus building methods, was found to be a practical way of eliciting robust standardised risk management scores across taxa and environment, even where data were incomplete or uncertain. It was important to reduce the potential impact of subjectivity and bias, which was done following the approach of Roy et al. (2014b). This also provided additional benefits in the exchange of knowledge between a diverse group of experts that do not regularly engage, leading to the challenge of preconceptions about management in some cases. While this approach was found to be effective and practical, good practice in the use of experts continues to develop and should be considered in further applications of the scheme. This could include providing additional training steps for scorers using known data , using and evaluating scoring intervals and using multiple experts to independently score species before and after discussions (Martin et al., 2012, Sutherland and Burgman, 2015, Hanea et al., 2017).

a. Prioritisation matrix for eradicating species already established (emerging) in GB. Risk assessment scores from published risk assessments (available at www.nonnativespecies.org)



Priority	Species
Highest	*Baccharis halimifolia*
Very high	*Procambarus acutus*
High	*Zamenis longissimus*
	Sarracenia purpurea
	Orconectes limosus
	Ichthyosaura alpestris
Moderate	*Lacerta bilineata*
	Egeria densa
	Cabomba caroliniana
	Aponogeton distachyos
	Podarcis muralis
	Lysichiton americanus
	Procambarus clarkii
Low	*Hydropotes inermis*
	Alopochen aegyptiacus
	Orconectes virilis
	Sagittaria latifolia
	Dreissena bugensis
	Hemigrapsus sanguineus
	Hemigrapsus takanoi

b. Prioritisation matrix for eradication of horizon (new) species based on most likely scenario of invasion in GB. All horizon species were scored as high risk and further grouped into the top 10, top 20 and top 30 threats (i.e. upper 10/30; mid 10/30; and lower 10/30) (Roy et al., 2014b)



Priority	Species
Highest	*Threskiornis aethiopicus*
Very high	*Nassella neesiana*
	Corvus splendens
High	*Procyon lotor*
	Tamias sibiricus
	Vespa velutina
Medium	*Nyctereutes procyonoides*
	Microstegium vimineum
	Ocenebra inornata
	Homarus americanus
	Neogobius melanostomus
Low	*Rapana venosa*
	Linepithema humile
	Corbicula fluminalis
	Proterorhinus marmoratus
	Myriophyllum heterophyllum
Very low	*Mnemiopsis leidyi*
	Echinogammarus trichiatus
Lowest	*Celtodoryx ciocalyptoides*
	Gracilaria vermiculophylla
	Echinogammarus ischnus

Figure 4.4 Using overall risk management and risk assessment scores to indicate priorities for eradication of (a) established (emerging) species and (b) horizon (new) species. The background colour of the matrix indicates priority (from green = lowest, to black = highest). Note that these colours are different to those used in Table 4.3 to indicate feasibility of eradication (where red = lower feasibility and green = higher feasibility). Initials indicate the position of each species with coloured box representing environment (purple = marine, blue = freshwater, green = terrestrial). Where multiple species occur in one cell they have equal priority and are in no particular order. The accompanying tables show species lists in priority order.

A key aim of the consensus workshop was to provide an opportunity to refine scores based on knowledge exchange between experienced invasive non-native species managers and to ensure participants had a clear and consistent understanding of the guidance. This resulted in a number of changes to scores throughout the workshop, the majority of which were made during the first phase, which was the first opportunity participants had to make refinements following clarification of the guidance and extensive discussions within and between expert groups. The decrease in the number of changes made to assessment scores between the first and second phase of the workshop demonstrates consensus amongst the experts being achieved. Confidence scores increased throughout the workshop with the majority of scores increasing by one degree (i.e. from medium to high) during the first phase. While expert judgement often suffers from overconfidence (Hulme, 2012, Morgan, 2014), this suggests that individual assessors were initially cautious when providing draft scores, but confidence improved with clarification of the guidance and the benefit of collective experience. The increase in confidence was a consistent pattern across all expert groups, suggesting it was not driven by one or two individuals convincing others.

The strong correlation between overall score and Practicality, Effectiveness and to a lesser degree Cost indicates that these components are the most consistent factors when considering overall feasibility of eradication. The lack of correlation with Likelihood of reinvasion and Window of opportunity indicates that these components carry less weight in determining the overall feasibility of eradication; however, they do provide important additional information that may influence resource allocation and the timing of management. For example, while the purple pitcher-plant (*Sarracenia purpuria*) received a high score for overall feasibility of eradication, it received only a medium score for Likelihood of reinvasion, suggesting that if eradication were attempted, effort would be required to prevent reinvasion through further deliberate planting in the wild by carnivorous plant enthusiasts. Impact and Acceptability also did not correlate strongly with overall score, but did have a pronounced impact on the overall feasibility of eradication for some species. For example, while Carolina fanwort (*Cabomba caroliniana*) occurs in only one location in GB, the feasibility of its eradication was substantially reduced by high levels of impact and low levels of acceptability associated with repeated mechanical control (and potential dredging) where it occurs in an ecologically sensitive Site of Special Scientific Interest.

Systematic differences in feasibility of eradication across species were considered. There was a strong relationship between overall feasibility of eradication and environment, with terrestrial species receiving significantly higher scores than aquatics, which broadly reflects the findings of Genovesi (2005), Robertson et al. (2017) and Simberloff (2008). Freshwater species generally received low scores; however, eradication was more likely to be feasible if the species occurred in lentic (still) rather than lotic (flowing) systems. Eradication of marine invasive non-native species is notably difficult (Thresher and Kuris, 2004, Sambrook et al., 2014) and this group received lowest scores overall. However, eradication in the marine environment may still be feasible when specific conditions are met (e.g. Culver and Kuris, 2000, Bax et al., 2002, Wotton et al., 2004), and this is reflected in the result for Japanese Sting Winkle (*Ocenebra inornata*). There was also a relationship between taxa and overall feasibility of eradication, with terrestrial vertebrates generally receiving moderate or higher scores for feasibility of eradication and invertebrates receiving lower scores, which reflects experience from GB and elsewhere (Genovesi, 2005, Robertson et al., 2017).

When combined with existing risk assessment scores the results of this study demonstrate that the NNRM scheme can be used to prioritise the eradication of large numbers of non-native species across different taxa and environment. Twelve out of 41 species that pose a threat to GB were identified as 'high', 'very high' or 'highest' priority for eradication. These priorities are different from those that would result from either risk assessment or risk management alone, indicating that taking both into account provides a more refined approach to prioritisation.

Both established and horizon species can be assessed using the NNRM scheme, allowing for emerging species to be prioritised for eradication and contingency planning to be put in place for new species before they arrive. Six out of the 20 species established in GB were identified as 'high', 'very high' or 'highest' priority for eradication. For these, the extent of establishment appears to be an important factor in determining priorities in some cases (four of the six occurred in one or few small, isolated populations); however, it was not a reliable predictor of priority (three of the seven 'low priority' species were established in two or fewer populations, while two 'high priority' species were comparatively widespread). Of the horizon species, six out of 21 were prioritised as 'high', 'very high' or 'highest' priority for eradication in GB. Prioritising the eradication of these species in advance of an invasion allows for contingency plans to be developed that may increase the efficiency and effectiveness of a response, which is particularly important for species that have a short window of opportunity for eradication, such as the Asian hornet (*Vespa velutina*). Indeed, such plans are already in place in GB for three of the six priority horizon species identified (published at www.nonnativespecies.org).

Species that are not considered a high priority for eradication may be high priorities for other types of management action. For example, prevention is likely to be a particularly important for high risk species that are not yet established in GB and for which eradication on arrival is unlikely to be feasible. The results of this study indicate this is likely to be the case for most marine and many freshwater (particularly lotic) invasive non-native species, in particular broadleaf watermilfoil (*Myriophyllum heterophyllum*), American lobster (*Homarus americanus*) and round goby (*Neogobius melanostomus*). For established species, long term management (e.g. containment or control) may be a priority for those that score high risk and low feasibility of eradication, such as quagga mussel (*Dreissena bugensis*).

Care should be taken when considering the results of this work in the context of past eradications in GB, as the latter were not the result of a systematic and comprehensive prioritisation process but rather an ad hoc approach largely driven by particular stakeholders or specific political drivers (Sheail, 2003). However, some parallels can be drawn as well as exceptions highlighted. The results of this study indicate that terrestrial and lentic freshwater species are more likely to be priorities for eradication than marine or lotic freshwater species, and this already has been the case in GB where eradications, either complete or underway, have been instigated for terrestrial vertebrates (Himalayan porcupine, *Hystrix brachyuran;* coypu, *Myocastor coypus;* muskrat, *Ondatra zibethicus* (Baker, 2010); monk parakeet, *Myiopsitta monachus;* ruddy duck, *Oxyura jamaicensis* (Defra, 2015)) and lentic freshwater species (topmough gudegon, *Pseudorasbora parva;* fathead minnow, *Pimephales promelas;* black bullhead, *Ameiurus melas;* African clawed-frog, *Xenopus Laevis;* American bullfrog, *Lithobates catesbeianus;* and, water primrose, *Luwigia grandiflora* (Defra, 2015)). An important difference between the results of this study and experience from GB to date is that the NNRM scheme indicates terrestrial plants could be a high priority for eradication where limited to small populations; however, there are few recorded eradications of these species in GB, or indeed in Europe (Genovesi, 2005). This may be because terrestrial plants are often ‘sleeper weeds’ (Groves, 1999) being overlooked at the early stages invasion, with decisions to attempt management taken too late for eradication to be feasible or cost-effective. This indicates that greater care should be taken in the future to identify and eradicate potentially invasive terrestrial plants at the earliest opportunity.

This work could be developed in a number of ways. The focus of the scheme is on eradication; however, further tools (or an extension of this scheme) to prioritise species for prevention interventions and long term management are required. Advances have been made in this area in the field of pest and weed risk management (e.g. Johnson, 2009b, Setterfield et al., 2010, Virtue, 2010, FAO, 2011, Auld, 2012, Kehlenbeck et al., 2012) and similar approaches may be applicable to the broader field of invasive non-native species. To aid consistency and repeatability it is important that assessors can clearly define invasion scenarios, eradication strategies and distinguish between the predefined responses used in the semi-quantitative scoring scale. Guidance was provided for this purpose; however, further elaborations of the scheme may benefit from refining this further, in particular providing more prescriptive instructions for defining invasion scenarios based on population size and scale; testing the use of multiple scenarios and eradication strategies for individual species; and, further defining and calibrating the response and confidence scales. A simple assessment of confidence has been presented here, but novel methods have been developed to better utilise and communicate uncertainty that could further enhance the scheme (e.g. Holt et al., 2012). A symmetrical relationship between risk assessment and risk management scores was used within the prioritisation matrix, assuming decision makers consider these equally important. Further applications of the scheme may wish to explore this relationship further with decision makers and consider calibrating the matrix if necessary. While applied here at a national level, the scheme is designed for use at any scale from specific sites to continent wide. Indeed, it may be timely to apply the approach across the EU given the requirements for risk management included in the recently adopted Regulation for Invasive Alien Species (Genovesi, 2015).

Chapter 5. Prioritising the management of new and emerging invasive non-native species at a continental scale

5.1 Abstract

With large numbers of species and limited resources available for management, prioritisation of invasive non-native species is vital. Tools exist to support prioritisation, but most are based solely on assessments of risk or impact. Without evaluating the feasibility of management, these tools are of little use for prioritising active management. A new tool for evaluating invasive non-native risk management criteria has been used to support prioritisation at a national scale. However, to achieve greatest impact prioritisation is also required at larger, international scales, where it can facilitate co-operation between states and support more effective management across larger regions. Here the Non-Native Risk Management (NNRM) scheme was expanded to prioritise species at a continental scale using lists of new (not yet established) and emerging (established with limited distributions) invasive non-native species identified by horizon scanning as a threat to the EU. Thirty-four experts scored species against seven key risk management criteria, based on defined invasion scenarios and eradication strategies. The overall result was a score for the feasibility of eradicating each species from the EU, which was combined with existing risk scores (derived from horizon scanning) to identify priorities for action. Priorities were identified for the eradication of emerging species and contingency planning for new species; as well as potential priorities for prevention and long term management. Results show that risk management evaluated information that was otherwise absent from risk assessment and resulted in priorities that were different than those indicated by risk assessment alone. Patterns in the feasibility of eradication based on environment and spatial extent were also found that could be used to help inform management in the future. This study demonstrates the value of combining both risk assessment and risk management scores to support the prioritisation of invasive non-native species management at the continental scale. The implications of this for management in the EU are considered and the need for more use of risk management tools at large scales is highlighted.

5.2 Introduction

Invasive non-native species are establishing at unprecedented rates around the world and there is little sign of saturation (Seebens et al., 2017). In response, the international community has agreed to prevent the introduction of, control or eradicate non-native species that threaten ecosystems, habitats or species (CBD, 1992). In particular, parties to the Convention on Biological Diversity agree to prioritise prevention, followed by early detection and rapid action, with eradication being the preferred response (COP 6 Decision VI/23). However, with many thousands of non-native species and limited resources, decision-makers must carefully prioritise which species to manage and how (McGeoch et al., 2016).

A widespread approach to setting priorities is to use lists of harmful species, i.e. blacklisting (Early et al., 2016). Such lists are usually based on risk assessment (Essl et al., 2011b, Roy et al., 2018b) or impact assessment, which is a component of risk (Blackburn et al., 2014, Nentwig et al., 2016, Bacher et al., 2017). However, simply assessing the risk posed by species is of limited use for prioritising invasive non-native species management as it fails to take into account the feasibility of delivering an effective management response at an appropriate scale (Booy et al., 2017). Failure to account for these factors could result in sub-optimal allocation of resources, with species being prioritised that are unmanageable or for which management is unlikely to be efficient (Robertson et al., *in prep*). Evaluating the feasibility of management, including costs, effectiveness and potential negative consequences, is therefore essential when prioritising limited resources.

Traditionally within a risk analysis framework, the role of evaluating management measures is provided by risk management (Mehta et al., 2010, OiE, 2017, Vanderhoeven et al., 2017). However, while risk and impact assessment schemes have received considerable attention (Essl et al., 2011b, Vanderhoeven et al., 2017), invasive non-native species risk management schemes have received little (Heikkilä, 2011, Booy et al., 2017). A number of approaches have recently been considered to help address this. Purely economic cost-benefit and cost-effectiveness have both been used to assess the case for undertaking management (Born et al., 2005, Blackwood et al., 2010, Courtois et al., 2018). However, analyses that rely solely on economic costs and benefits have a number of limitations (Born et al., 2005, Binimelis et al., 2008). Any comprehensive assessment of non-native species or their management should also include consideration of the social, environmental, animal welfare and biodiversity consequences (Booy et al., 2017). Additionally, Schaefer et al. (2011) argue that cost-benefit analyses of any management option for non-native species must include the subjective valuation of species (Dudgeon and Smith, 2006, Evans et al., 2008, Sandler, 2010). The currencies used to measure all of these elements are difficult to monetise (Hoagland and Jin, 2006), although this approach has been used in some cases (Lupi et al., 2003, Nunes and van den Bergh, 2004). Bacher et al. (2017) conclude that attempts to quantify socio-economic impacts in monetary terms are unlikely to provide a useful basis for evaluating and comparing impacts of non-native taxa because they are notoriously difficult to measure and important aspects of human well-being are ignored.

Born et al. (2005) recommend multi-criteria decision to overcome these limitations and support non-native species policy and management. Multi-criteria decision-making approaches provide a method for identifying optimal solutions to complex problems where assessment criteria or data are measured in different or conflicting currencies, including when only incomplete or imprecise information is available, or where human evaluation is needed (Kahraman, 2008). By clearly structuring complex problems and explicitly evaluating multiple criteria, these techniques have the advantage of allowing the comparison of alternate options and can lead to more informed and better decisions (Liu et al., 2010, Liu et al., 2011a). By combining ecological knowledge and economic evaluation, multi-criteria evaluation opens up new ways of producing policy-relevant results rather than intensifying what Born et al. (2005) describe as the mono-dimensional approach of purely monetary evaluation.

A further disadvantage of purely economic cost-benefit and cost-effectiveness analysis, as well as some more complex multi-criteria decision-making schemes, is the data requirements needed to complete them. Given the numbers of non-native species and the complexity of criteria required to prioritise them, it is rare that there is sufficient formal empirical evidence to support data heavy, quantitative analysis for prioritisation within formal systems. Similar problems occur across the field of conservation ecology and yet management decisions are still required, even where data are limited (Sutherland and Burgman, 2015). Expert information is increasingly used to overcome problems with limited empirical data and there is a growing body of good practice in its application (Burgman et al., 2011, Martin et al., 2012, Hanea et al., 2017). It is already commonly used for invasive non-native species risk assessment (Essl et al., 2011b, Vanderhoeven et al., 2017) and it is increasingly applied to horizon scanning for invasive non-native species (the process of scoping for new invasive non-native species or invasive non-native species with limited distributions that pose a future threat) with expert information complemented by consensus building techniques (Roy et al., 2014b, Roy et al., 2015b, Ricciardi et al., 2017, Roy et al., 2017).

The Non-Native Risk Management (NNRM) scheme was designed to help fill the gap in invasive non-native species risk analysis and combines a multi-criteria approach with expert elicitation and consensus building to facilitate the prioritisation of invasive non-native species management (Booy et al., 2017). It takes a similar form to many invasive non-native species risk assessment systems, providing a structured method for evaluating key risk management variables and documenting associated uncertainties (Baker et al., 2008, Brunel et al., 2010, Mumford et al., 2010, Copp et al., 2016). The overall result of the NNRM is a score for the feasibility of eradicating an invasive non-native species based on a particular scenario and management strategy, which can be combined with the results of risk assessment to identify management priorities (Booy et al., 2017).

While the NNRM has been applied to date at the national scale, prioritisation is needed at different scales. Large-scale regional (e.g. continental) prioritisation of invasive non-native species management is particularly important, as the action of individual states can have important consequences for the larger region. Failure to take effective action in one state can have serious implications for the whole region. For instance, the emerald ash borer now threatens ash trees across the continent of Europe after its arrival and spread through Russia (Valenta et al., 2017), while the failure to eradicate grey squirrels in Italy threatens the survival of the native red squirrel in many part of Europe (Bertolino and Genovesi, 2003). Even where the threat from a species is not shared, cooperative invasive non-native species management may still be required in one state to protect another (Caffrey et al., 2014). For example, the ruddy duck poses little risk to northern Europe where it is currently established, but action is undertaken there to protect the remaining populations of white-headed duck in Spain (Robertson et al., 2015).

Priorities at regional scales may differ from those at more local scales. Identifying regional priorities should therefore help focus management across the region and encourage cooperative action (Early et al., 2016). This may facilitate states to take action in solidarity with each other and improve the effectiveness and efficiency of a response (Mumford, 2013, Caffrey et al., 2014). However, prioritisation at these scales is daunting given the large numbers of species involved, the diversity of taxa, heterogeneity of landscapes and differences in policy, legislation and perception between (Andersen et al., 2004, Hulme et al., 2009, Firn et al., 2015b). Prioritisation schemes used at these scales must therefore be particularly flexible (Early et al., 2016, McGeoch et al., 2016).

The spatial scale of an invasion is also likely to influence the choice and feasibility of management, with feasibility of eradication decreasing with the increasing extent and therefore a switch to long term management becoming more appropriate. However, feasibility of management at different scales is also likely to be influenced by other factors, such as taxa and the environment in which the species is established (Booy et al., 2017). A number of authors have assessed the feasibility of eradication at different scales for different taxa (Rejmánek and Pitcairn, 2002, Martins et al., 2006, Howald et al., 2007, Brockerhoff et al., 2010, Robertson et al., 2017); however, the relationship between feasibility and scale has rarely been considered across taxa and environment. Understanding the relationship between these variables should help to identify the circumstances in which eradication may be appropriate and where it may be necessary to switch to other responses.

Here the use of the NNRM for prioritising invasive non-native species management at large scales is explored by applying it to an existing list of new and emerging species that threaten the European Union (EU) (Roy et al., 2015b). In doing so, the aim was to test a method that identifies specific management priorities by combining both risk and risk management scores within an overall risk analysis framework. The implications of these results for invasive non-native species management in the EU are explored, as is the feasibility of eradication at different scales across taxa and environment. The wider application of this approach for large-scale invasive non-native species prioritisation elsewhere is considered, for example in low-income regions where the threat from invasive non-native species is predicted to increase rapidly but where resources are particularly limited (Early et al., 2016). Lastly, this study highlights the continued lack of risk management for invasive non-native species prioritisation and the implications this has for potentially inefficient resource allocation.

Table 5.1. Count of species by environment, establishment status in the EU and broad taxonomic group

Environment	Status	Plant	Vert	Invert	∑
Freshwater	Established	1	3	5	9
	Not established	0	10	4	14
Terrestrial	Established	6	10	4	20
	Not established	17	11	9	37
Marine	Established	0	1	5	6
	Not established	2	1	6	9
∑		26	36	33	

5.3 Methods

A list of 95 species were used that were identified as ‘high’ or ‘very high’ risk through horizon scanning Roy et al. (2015b). This comprised taxa that were either new to the EU (i.e. not yet established) or emerging (i.e. established with limited distributions) (Table 5.1). For each species, a risk management assessment was completed using a modified version of the Non-Native Risk Management (NNRM) scheme (Booy et al., 2017). Modifications included introducing a standardised method for documenting the extent of invasion scenario of the target species based on an alphanumeric code, with letters A-D representing the number of discrete populations (respectively 1-3, 4-10, 10-50, 50+) and numbers 1-6 representing the total combined area of all populations (respectively <1ha, 1-10ha, 10ha-1km^2, 1-10km^2, 10-100km^2, >100km^2). Species were included that had a range of extents (Table 5.2); however, as the focus of horizon scanning was on new and emerging species, most were at the low end of the scale (i.e. towards A1). The full, modified scheme and guidance is available as supplementary information (Appendix D).

Table 5.2. Count of species by scenario code for extent. Letters A-D represent the number of discrete populations (respectively 1-3, 4-10, 10-50, +50) and numbers 1-6 represent total combined area (respectively <1ha, 1-10ha, 10ha-1km2, 1-10km2, 10-100km2, >100km2). For emerging species (established with limited distributions) codes were used to define the current extent. For new species (not yet established), codes define the most likely extent at the point of detection.

		Area					
		1	**2**	**3**	**4**	**5**	**6**
Populations	**A**	22	23	3	5	5	2
	B	1	11	2	0	1	4
	C	1	6	3	1	0	1
	D	0	2	0	1	0	1

A combination of expert elicitation, review and consensus building methods were used to produce and validate risk management assessments following similar approaches to Roy et al. (2014b) and Booy et al. (2017). In total, 34 experts were engaged in the elicitation process (Appendix F) grouped into five specialisms: freshwater animals, terrestrial vertebrates, terrestrial invertebrates, marine species, and plants (excluding marine plants). Each group comprised 5-8 experts chosen by the organisers in cooperation with an appointed group leader based on proven experience of invasive non-native species management and representation of a range of EU member states.

Risk management assessments were first drafted by expert groups using the NNRM template. The invasion scenario and eradication strategy for each species was completed by the group leader, in consultation with other experts in their group as necessary. For emerging species the scenario was the current distribution of the species. For new species, the most likely invasion scenario was used, based on the likely extent of the species at the point of detection in the wild given current surveillance. Each species was then assessed independently by at least three different experts from each group, who provided response and confidence scores for seven risk management components (i.e. Effectiveness, Practicality, Cost, Impact, Acceptability, Window of Opportunity and Likelihood of Reintroduction) as well as Overall Score (i.e. overall feasibility of eradication). These were collated, anonymised and the scores returned to the expert group, along with the median response and confidence scores for each risk management component and Overall Score.

A two-day workshop (17-18 May 2016) was held to review, refine and ultimately agree scores by consensus. Twenty-eight of the original experts, including all group leaders, attended. The first session was for group leaders only to ensure they were consistently applying the scoring guidance and were clear on the requirements of the rest of the workshop. To aid in this, each group leader presented their group's initial scores, discussed any areas of potential ambiguity and agreed on clarifications. The main workshop then proceeded as follows:

1. All participants met in plenary to go through the scoring guidance with the organisers, resolve any issues or ambiguity and ensure consistency in application. Expert group leaders then presented an overview of the initial scores from their groups to all participants, who were encouraged to discuss and challenge the scores.

2. All participants then separated into their expert groups to review and refine the scores of their group, taking into account the discussions from session 1. Each group was provided with the median response and confidence scores for each of their species and asked to refine these scores, where necessary, based on the judgement of the group.

3. The final stage of the scoring process was to agree the refined scores by the consensus of all participants. All refined scores were collated and presented back to the workshop in plenary by two facilitators (OB and PG), with a focus on agreeing the final Overall Score for each species. Participants were encouraged to discuss and challenge the scores with any changes at this point made with the consensus of the whole group.

5.3.1 *Analysis*

Whether risk assessment and risk management scores were measuring similar underlying information was assessed using polychoric correlations as both scoring systems resulted in ordinal data. The interrelation between the component scores and Overall Score was examined in ordination space. A factor plot was produced to investigate how the components were related and non–metric multi-dimensional scaling (nMDS) used to explore how individual species component scores related to the Overall Score. All analysis was conducted using R (R Core Team, 2017).

To assess the relationship between Overall Score and environment, total area and number of populations a cumulative link model (CLM) was used in the R package 'Ordinal' (Christensen, 2018) as the Overall Score was ordinal. It was hypothesised that the Overall Score for each species would decline with increasing spatial extent (total area and number of populations) and be dependent on the environment in which the species occurred. Population categories 'C' and 'D' were pooled into one category (10+ populations) as were areas >10Ha (greater than category 3) owing to sparse data at these ranges. Ordinal regression assumes proportional odds (that the relationship between each pair of outcome groups is the same). Statistical tests for proportional odds have been criticised as they tend to falsely reject the null hypothesis, so proportionality was assessed using a graphical method following Bender and Grouven (1997) and Gould (2000). This method uses plots of predicted values derived from a series of binary logistic regressions to check the assumption that coefficients are equally separated across cut-points.

The CLM was used to predict the feasibility of eradication for every combination of environment, total area and number of populations. Model predictions were visualised using the R package ggplot2 (Wickham, 2009) and were expressed as the probability of the Overall Score being each of the five response levels (very high to very low). High feasibility was indicated by high probabilities for 'high' and 'very high' scores (green), while low feasibility was indicated by high probability for 'low' and 'very low' scores (red). Where the probability of 'low' and 'high' scores was roughly equal (similar proportions of green and red) this indicated combinations where the predictive power of the model was low (i.e. the model could not predict whether Overall Score was likely to be high or low).

To indicate priorities for eradication, the overall risk scores derived from the horizon scanning exercise (Roy et al., 2015b) were combined with the Overall Score from this risk management exercise in a prioritisation matrix (following Booy et al., 2017). As both the overall risk scores and Overall Score from this exercise used a five-point scale (very low to very high) the result was a 5x5 prioritisation matrix, with priorities ranging from lowest (1:1) to highest (5:5) (Table 5.3). However, as only species with risk scores of 'high' and 'very high' were included in this exercise, only positions in the top two rows of the matrix could be achieved, resulting in priorities ranging from medium-low (4:1) to highest (5:5).

The matrix was also used to investigate other priorities, including prevention and long-term management. For new species, prevention was likely to be a particular priority if the species posed a high risk and the feasibility of eradication after arrival was low. For emerging species, long-term management was likely to be a particular priority if the species posed a high risk and the feasibility of eradication was low. These priorities corresponded to the top left corner of the matrix and are marked: ++ highest, and + high priority for prevention / long-term management (Table 5.3).

Table 5.3. Priority matrix based on risk scores (derived from horizon scanning) and Overall Score from this risk management exercise (i.e. overall feasibility of eradication). Both scores use a 5-point scale (very low to very high); however, only species with risk scores of 'high' and 'very high' were included in this study (hence it was not possible for species to be placed in greyed out parts of the matrix). The matrix gives priority (for eradication) to species with the highest risk scores and highest feasibility of eradication. While focussed on prioritising eradication, the matrix can be used to consider potential priorities for prevention (new species that are high risk for which feasibility of eradication is low) and long term management (emerging species that are high risk for which feasibility of eradication is low); these prioritises are marked ++ highest priority and + high priority.

Risk score	Feasibility eradication				
	Very low (1)	Low (2)	Medium (3)	High (4)	Very high (5)
Very high (5)	Medium++	Med-high+	High	Very high	Highest
High (4)	Med-low+	Medium	Med-high	High	Very high
Medium (3)	Low	Med-low	Medium	Med-high	High
Low (2)	Very low	Low	Med-low	Medium	Med-high
Very low (1)	Lowest	Very low	Low	Med-low	Medium

5.4 Results

5.4.1 Correlation of scores

Overall risk management scores differed from those of risk assessment, there was no correlation between the two: polychoric correlation, rho= -0.281 +/- s.e. 0.136, Chi sq =0.519, p=0.89 (note rho is the test statistic where values near 0 indicate little agreement). This indicates the risk assessment and risk management schemes measuring different information.

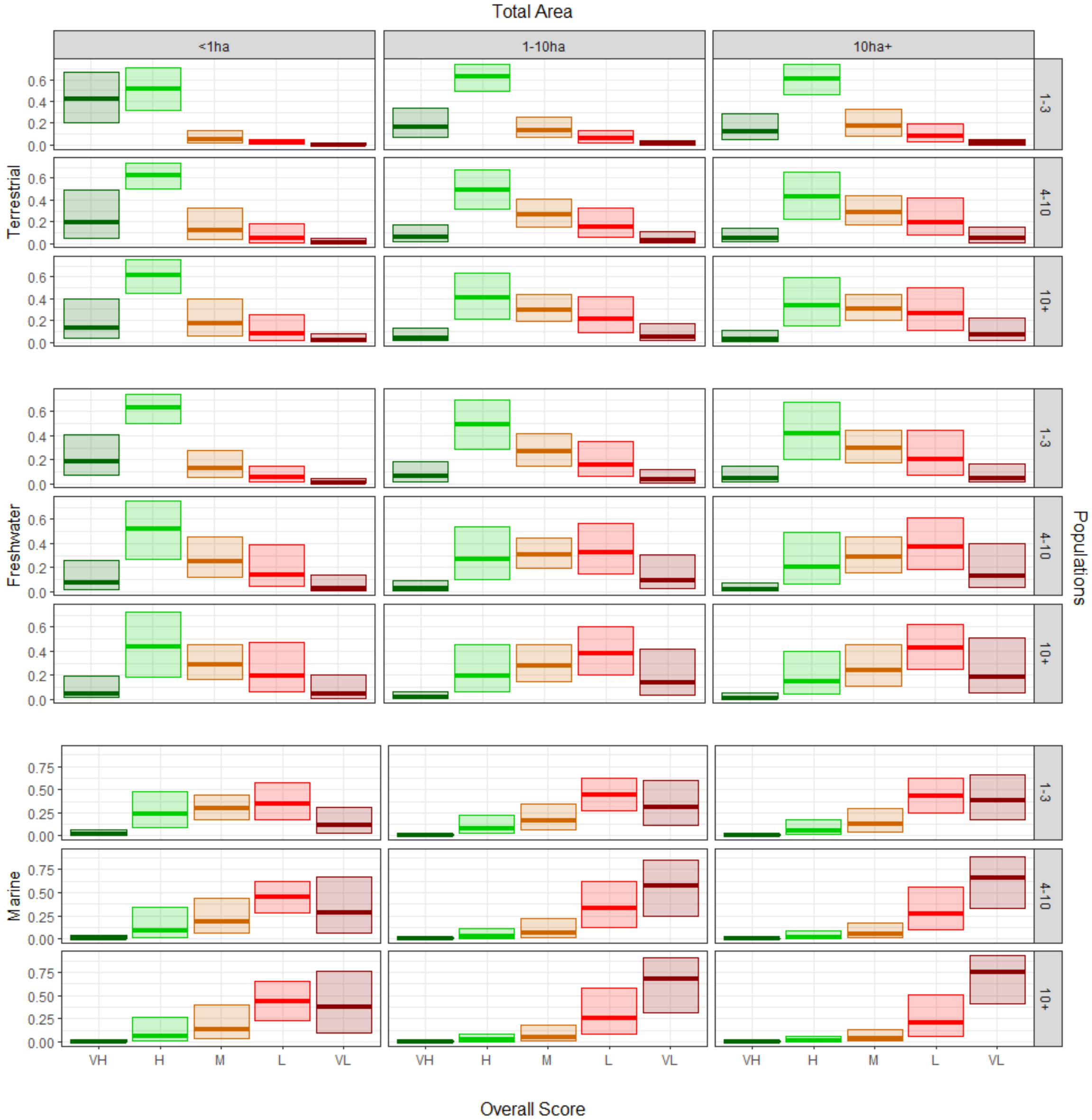


Figure 5.1. Cumulative Link Model predictions for the feasibility of eradication in different environments at different spatial scales. The probability of Overall Score (for feasibility of eradication) being each of the five response levels very high (VH) to very low (VL) is given (on the y axis) for each combination of variables, with 95% confidence intervals. Note that colours indicate feasibility of eradication (green = higher feasibility, red = lower feasibility), these are different to those used (e.g. in Table 5.3) to indicate priority (where red = higher priority and green = lower priority).

The first five risk management component scores (Effectiveness, Practicality, Cost, Impact and Acceptability) did correlate with Overall Score, while the last two (Window of Opportunity and Likelihood of Reinvasion) did not (Appendix G). This reinforces the suggestion that the latter two are of less importance when determining the Overall Score (Booy et al., 2017). Overall Score aligned in sequence with individual component scores (Appendix H) with some overlap; suggesting that individual scores were a good indication of Overall Score, but that it was not possible to consistently determine Overall Score based on individual components.

5.4.2 Modelling the effect of extent and environment on feasibility of eradication

The assumptions of proportionality were met in the cumulative link model as the thresholds (intercepts) for each covariate were broadly similar distances apart (Appendix I). All variables (environment, total area and number of populations) were significant predictors of the Overall Score (Appendix J). Marine species received significantly lower scores than freshwater species, while terrestrial species received significantly higher scores than freshwater species. Feasibility of eradication also decreased significantly with increasing area and number of populations.

For all environments, the model predicted that increasing total area and number of populations reduced the probability of ‘very high’ and ‘high’ Overall Scores and increased the probability of ‘very low’ and ‘low’ Overall Scores (Fig 5.1). For terrestrial species, ‘high’ scores were more probable than ‘low’ for every combination of extent (i.e. total area and number of populations). The probability of scoring ‘very high’ or ‘high’ was greatest at the smallest extent and there was high confidence at this extent that the score would not be ‘medium’, ‘low’ or ‘very low’ (indicated by the narrow confidence intervals at these level). At the largest extent (10ha+ and 10+ populations) it was more difficult to predict Overall Score as there was considerable uncertainty between the middle categories, although the probability of scoring ‘high’ was slightly greater.

For freshwater species, ‘high’ scores were most probable where either the total area was small (<1ha) or there were few populations (<1-3). Beyond this, the probability of ‘high’ scores dropped below those of ‘low’ scores, with ‘low’ scores considerably more probable at extents above the combination of 1-10ha and 4-10 populations. Confidence intervals for freshwater species indicated that ‘medium’, ‘low’ and ‘very low’ scores were not probable at the smallest extent, but relatively wide intervals at all other extents indicate considerable uncertainty in the freshwater scores.

For marine species, the model predicted an Overall Score of ‘low’ or ‘very low’ as most probable for all but the smallest extent (1-3 populations, <1ha) and there was considerable confidence in this pattern (relatively narrow confidence intervals for ‘very high’ and ‘high’ at all extents except where area was <1ha). At the smallest extent, although the probabilities were similar across the middle categories with wide confidence intervals, ‘low’ scores were slightly more probable than ‘high’ scores.

5.4.3 Prioritisation

Combining risk scores and risk management scores resulted in six levels of eradication priority: highest (1), very high (20), high (36), med-high (20), medium (14) and med-low (4). These were further divided into priorities for eradication of emerging species (Fig 5.2a) and contingency planning for new species (Fig 5.2b) as well as priorities for prevention and long term management (Fig 5.2a and b). Scores for all species are available as supplementary information (Appendix K and Appendix L). Detail on key priorities is provided below and in Tables 5.4 and 5.5.

5.4.4 Eradication priorities (of established species)

Of the 35 emerging species assessed, four were identified as ‘very high’ priority for eradication and a further ten were identified as ‘high’ priority (Table 5.4). The top four priority species were terrestrial vertebrates with risk scores of ‘very high’ and feasibility of eradication (i.e. Overall Scores) of ‘high’. At the time of assessment, these were considered to be established in no more than 3 populations, covering a minimum area of 1ha and maximum area of 100km^2 each. However, there was uncertainty about the status and extent of three of the four species (common myna, *Acridotheres tristis*; Berber toad, *Bufo mauritanicus*; red-vented bulbul, *Pycnonotus cafer*). Current populations of all four species were thought to be limited to Spain, except one population of *A. tristis* potentially in Portugal. The estimated cost of eradicating each species ranged from €1-50k (*B. mauritanicus*) to €0.2-1M (*A. tristis* and coati, *Nasua nasua*), with the total cost of eradicating all four species estimated to range between €0.45-2.25M. The key eradication methods identified included netting, trapping, manual capture and shooting, which were not considered to cause significant adverse environmental, social or economic harm. Acceptability scores were high, except for the *N. nasua*, which scored ‘medium’. The Window of Opportunity for all of these species was 1-3 years.

a. emerging species (priorities for long term management are marked highest^{++} and high^{+})

Emerging species			Feasibility of eradication				
			VL	L	M	H	VH
			7	8	8	12	0
Risk score	VH	13	3^{++}	4^{+}	2	4	0
	H	22	4^{+}	4	6	8	0
	M	0	-	-	-	-	-
	L	0	-	-	-	-	-
	VL	0	-	-	-	-	-

Species listed in priority order:

Very high - *Acridotheres tristis, Bufo mauritanicus, Nasua nasua, Pycnonotus cafer.* **High-** *Alternanthera philoxeroides, Axis axis, Botrylloides giganteum, Cherax destructor, Euonymus fortunei, Euonymus japonicus, Ligustrum sinense, Misgurnus anguillicaudatus, Rhea americana, Saperda candida.* **Med-high-** *Andropogon virginicus, Ehrharta calycina, Fundulus heteroclitus, Hypostomus plecostomus, Marisa cornuarietis, Wedelia trilobata, Callosciurus finlaysonii*$^{+}$, *Herpestes auropunctatus*$^{+}$, *Pomacea canaliculata*$^{+}$, *Pomacea maculata*$^{+}$. **Medium-** *Acridotheres cristatellus, Charybdis japonica, Pheidole megacephala, Psittacula eupatria, Arthurdendyus triangulates*$^{++}$, *Penaeus aztecus*$^{++}$, *Pterois miles*$^{++}$. **Med-low-** *Ashworthius sidemi*$^{+}$, *Bellamya chinensis*$^{+}$, *Macrorhynchia philippina*$^{+}$, *Pseudonereis anomala*$^{+}$.

b. new species (priorities for prevention are marked highest^{++} and high^{+})

New species			Feasibility of eradication				
			VL	L	M	H	VH
			1	8	11	30	10
Risk score	VH	14	1^{++}	2^{+}	3	7	1
	H	46	0^{+}	6	8	23	9
	M	0	-	-	-	-	-
	L	0	-	-	-	-	-
	VL	0	-	-	-	-	-

Species listed in priority order:

Highest- *Orconectes rusticus.* **Very high-** *Bison bison, Channa argus, Cryptostegia grandiflora, Gambusia affinis, Lampropeltis getula, Lonicera morrowii, Micropterus dolomieu, Misgurnus mizolepis, Oreochromis aureus, Oreochromis mossambicus, Oreochromis niloticus, Pachycondyla chinensis, Rubus rosifolius, Sirex ermak, Solenopsis invicta, Trichosurus vulpecula..* **High-** *Aeolesthes sarta, Albizia lebbeck, Amynthas agrestis, Boiga irregularis, Celastrus orbiculatus, Cherax quadricarinatus, Chromolaena odorata, Chrysemys picta, Cinnamomum camphora, Clematis terniflora, Crepidula onyx, Cyprinella lutrensis, Eleutherodactylus coqui, Gymnocoronis spilanthoides, Limnoperna fortunei, Lonicera maackii, Mytilopsis sallei, Prosopis juliflora, Prunus campanulata, Pycnonotus jocosus, Rhinella marina, Solenopsis geminata, Tetropium gracilicorne, Tilapia zillii, Triadica sebifera, Vespula pensylvanica..* **Med-high-** *Acanthophora spicifera, Cortaderia jubata, Cynops pyrrhogaster, Hemidactylus frenatus, Lygodium japonicum, Microstegium vimineum, Solenopsis richteri, Symplegma reptans, Codium parvulum*$^{+}$, *Homarus americanus*$^{+}$. **Medium priority-** *Eleutherodactylus planirostris, Gammarus fasciatus, Lespedeza juncea, Morone americana, Perna viridis, Potamocorbula amurensis, Plotosus lineatus*$^{++}$

Figure 5.2 Counts of species within the priority matrix for (a) emerging and (b) new species.The colour of the matrix reflects priority (derived from Table 5.3) ranging from highest (top right) to lowest (bottom left) priority. Note that species were not included in this study with lower than ‘high’ risk scores and so no species occupy the bottom three rows of each table.

The ten 'high' priority established species comprised three terrestrial plants, one freshwater plant, two terrestrial vertebrates, two freshwater animals, one insect and one marine tunicate (Table 5.4). These included species with primarily 'high' risk and 'high' feasibility scores; however, two species scored 'very high' risk with only 'medium' feasibility (alligator weed, *Alternanthera philoxeroides*; and the marine tunicate, *Botrylloides giganteum*). The majority of 'high' priority species were relatively well confined comprising 1-3 populations, although three plants had more (10-50 populations) as did the oriental weather-fish, *Misgurnus anguillicaudatus* (10-50 populations) and the apple tree-borer, *Saperda candida* (4-10 populations). The area covered by these species ranged from <1ha (common yabby, *Cherax destructor*; and *B. giganteum*) to >100km^2 (Indian spotted deer, *Axis axis*). Species were thought to be present in seven EU Member States, including: Italy (3), France (3), Germany (3), Spain (2), Croatia (1), United Kingdom (1) and Netherlands (1). The cost range for eradicating all ten species was in the region of €1M-5.5M. Barriers to eradication were identified for some species. For example, the eradication of the *M. anguillicaudatus* using electrofishing, fyke netting and piscicide was considered likely to cause moderate adverse environmental harm as well as low Acceptability. Both *Rhea americana* (greater rhea) and *A. axis* received only medium Acceptability scores; while the removal of *Ligustrum sinense* (Chinese privet) using mechanical means and herbicide had the potential to cause adverse environmental impacts. The Window of Opportunity for all of the ten 'high' priority species was 1-3 years, except *B. giganteum* which had a very short Window of Opportunity (<2 months) and *A. axis* with a longer window (4-10 years).

5.4.5 Contingency priorities (for eradication of species not yet established)

Of the 60 new species, *Orconectes rusticus* (rusty crayfish) scored the 'highest' priority for eradication, with both the overall risk and feasibility of eradication scoring 'very high' (Table 5.5). This was based on the most likely scenario at the point of detection of only 1-3 populations with a total area of <1ha and an eradication strategy of intensive trapping. Eradication based on this scenario was considered likely to cost no more than €50k with minimal impacts and high levels of acceptability. However, the Window of Opportunity was 'short' (2 months – 1 year) and Likelihood of Reintroduction 'high'.

A further 16 species not yet established in the EU were assessed as 'very high' priority for eradication if detected: seven freshwater fish, three terrestrial plants, three insects, two mammals and one reptile. Six fish scored 'very high' risk and 'high' feasibility, the other fish scored 'high' risk and 'very high' feasibility of eradication. All of the other species apart from *Lampropeltis getula* (common kingsnake) scored 'high' risk and 'very high' feasibility of eradication. As these species were not yet established in the EU, their extent was based on the most likely scenario of invasion. The majority of species were considered likely to be in 1-3 populations covering <1ha or 1-10ha at the point of detection. However, two species were considered likely to be in more than 1-3 populations (Asian needle ant, *Pachycondyla chinensis*; and Nile tilapia, *Oreochromis niloticus*) and three were likely to cover 1-10km^2 (American bison, *Bison bison*; brushtail possum, *Trichosurus vulpecula*; and *L. getula*). The bioregions that species could invade included the Mediterranean (13), Macronesia (12), Atlantic (8), Continental (7) and Steppe (6). Approximately thirteen different methods of eradication were identified, including: shooting, trapping, manual destruction, mechanical removal, herbicide, electrofishing, fyke netting, piscicide, draining, angling, poison baiting, insecticide and incineration. The total estimated cost of eradicating all 16 species was in the region of €0.5-2.6M. No significant adverse impacts were considered likely. All eradications had 'high' or 'very high' acceptability, except for *Gambusia affinis* (western mosquitofish) which scored 'moderate' because of potential negative reaction to the use of piscicides. The Window of opportunity for most species was short, 2m-1year, with two species <2m, six species 1-3 years and one species (*B. bison*) 4-10 years.

5.4.6 Long term management and prevention priorities

Eleven emerging species were identified as potentially high priorities for long term management, because they had high risk and low feasibility of eradication scores and were already established in the EU (Fig 5.2a; Appendix K). Three scored 'very high' risk and 'very low' feasibility of eradication, including *Arthurdendyus triangulates* (New Zealand flatworm), *Pterois miles* (lion fish) and *Penaeus aztecus* (northern brown shrimp). The remaining eight species scored 'high' risk and 'very low' feasibility or 'very high' risk and 'low' feasibility, including: two marine invertebrates (a hydroid, *Macrorhynchia philippina*; and a polychaete, *Pseudonereis anomala*), three freshwater invertebrates (Chinese mysterysnail, *Bellamya chinensis*; golden apple snail, *Pomacea canaliculata*; and giant apple snail, *Pomacea maculata*), one terrestrial invertebrate (a nematode, *Ashworthius sidemi*) and two terrestrial vertebrates (Finlayson's squirrel, *Callosciurus finlaysonii*; and small Asian mongoose, *Herpestes auropunctatus*).

Three new (not yet established in the EU) species were identified as potentially high priorities for prevention because they had high risk and low feasibility of eradication scores (Fig 5.2b; Appendix

L). *Plotosus lineatus* (striped eel catfish) scored 'very high' risk with 'very low' feasibility of eradication; while the *Homarus americanus* (American lobster) and *Codium parvulum* (a green algae) were both 'very high' risk with 'low' feasibility of eradication.

5.5 Discussion

The Non-native Risk Management scheme (NNRM) was applied to identify priorities for the eradication of new and emerging invasive non-native species in the European Union (EU). This not only indicated priorities for the eradication of emerging species and contingency planning for new species, but potential priorities for prevention and long term management as well. While the NNRM has previously been applied at a national scale (Chapter 4 and Booy et al., 2017), this is the first application of an invasive non-native species risk management scheme at continental scale. Despite increased complexity at this scale, particularly associated with landscape heterogeneity and a lack of information on the status of species in Europe, the scheme was successfully applied and used to identify priorities. However, given a lack of information on some species, particularly those that have recently established, further work is required to improve the confidence in these assessments. The results of this exercise should therefore be considered preliminary prior to further detailed analysis, which is similar to risk assessment where initial assessments are used to screen species for further detailed assessment (e.g. Tanner et al., 2017).

Risk management was applied to new and emerging invasive non-native species identified by horizon scanning as these are likely candidates for prevention, contingency planning and eradication given their absence or limited status in the EU (Roy et al., 2015b). They are also of particular concern currently in the EU which has recently adopted regulation 1143/2014 on invasive non-native species emphasising the importance of prevention and rapid eradication (EU, 2014b). While horizon scanning provides a useful method for reducing long lists of potentially thousands of species to a short list of those most likely to be threats (Roy et al., 2015b), it is of limited use for prioritising specific actions as it does not take into account the feasibility of management (Booy et al., 2017, Vanderhoeven et al., 2017). By applying risk management criteria, this study refined this list into specific management priorities, aligning with the guiding principles of the Convention on Biological Diversity (UNEP, 2011).

Table 5.4. Very high and high priority species established in the EU (n=14).

Priority	Scientific	English	RA	RM	Conf	Scen	MS	Methods	Effect.	Pract.	Cost min (1000s)	Cost max (1000s)	Impact	Accept.	Window	Reintro.
Very high	*Acridotheres tristis*	Common myna	VH	H	H	A5	ES, PT	netting, trapping, shooting	high	medium	€ 200	€ 1,000	minimal	high	1-3	medium
Very high	*Bufo mauritanicus*	Berber toad	VH	H	M	A2	ES	manual capture, netting	high	medium	€ 1	€ 50	minor	v. high	1-3	low
Very high	*Nasua nasua*	Coati	VH	H	M	A4	ES	trapping, shooting	high	high	€ 200	€ 1,000	minimal	medium	1-3	low
Very high	*Pycnonotus cafer*	Red-vented bulbul	VH	H	H	A5	ES	trapping, netting	high	high	€ 50	€ 200	minimal	high	1-3	medium
High	*Alternanthera philoxeroides*	Alligator-weed	VH	M	M	C2	FR, IT	mechanical, manual	medium	high	€ 200	€ 1,000	minor	high	1-3	medium
High	*Axis axis*	Indian spotted deer	H	H	H	A6	CR	shooting, sterilization	high	high	€ 200	€ 1,000	minor	medium	4-10	low
High	*Botrylloides giganteum*	a tunicate	VH	M	M	A1	IT	wrapping structures	medium	high	€ 200	€ 1,000	minor	high	<2 m	high
High	*Cherax destructor*	Common yabby	H	H	M	A1	ES	biocontrol, trapping	high	high	€ 1	€ 50	minimal	v. high	1-3	high
High	*Euonymus fortunei*	Winter Creeper	H	H	H	A2	FR	herbicide	high	low	€ 50	€ 200	minor	high	1-3	high
High	*Euonymus japonicus*	Japanese spindle	H	H	M	B2	UK	grubbing, mechanical, herbicide	high	high	€ 1	€ 50	minor	v. high	1-3	high
High	*Ligustrum sinense*	Chinese Privet	H	H	M	B2	FR	grubbing, mechanical, herbicide	high	high	€ 1	€ 50	moderate	v. high	1-3	medium
High	*Misgurnus anguillicaudatus*	Oriental weatherfish	H	H	H	C4	NL, DE, ES, IT	electrofishing, piscicide, fyke netting	v. high	medium	€ 200	€ 1,000	moderate	low	1-3	medium
High	*Rhea Americana*	Greater rhea	H	H	M	A5	DE	shooting, and other methods	v. high	high	€ 200	€ 1,000	minor	medium	1-3	medium
High	*Saperda candida*	Apple Tree Borer	H	H	H	B2	DE	manual destruction, felling of trees	high	high	€ 1	€ 50	minor	high	1-3	medium

Table 5.5. Highest and very high priority species not established in Europe (n=17).

Priority	Scientific	English	RA	RM	Conf	Scen	Regions	Main method	Effect.	Pract.	Cost min (1000s)	Cost max (1000s)	Impact	Accept.	Window	Reintro.
Highest	*Orconectes rusticus*	Rusty crayfish	VH	VH	M	A1	MED, ATL, CON, STE	trapping	v. high	high	€ 1	€ 50	minimal	v. high	2m-1	high
Very high	*Bison bison*	American bison	H	VH	H	A4	CON	shooting	v. high	high	€ 1	€ 50	minimal	high	4-10	v. low
Very high	*Channa argus*	Northern snakehead	VH	H	M	A2	MAC, MED, ATL, CON, STE	electrofishing, fyke netting	v. high	v. high	€ 50	€ 200	minimal	v. high	2m-1	medium
Very high	*Cryptostegia grandiflora*	None	H	VH	H	A1	MAC, ATL, MED	mechanical, herbicide	v. high	v. high	€ 1	€ 50	minimal	v. high	1-3	high
Very high	*Gambusia affinis*	Western mosquitofish	VH	H	H	A2	MAC, MED, ATL, CON, STE	piscicide	v. high	medium	€ 50	€ 200	minor	medium	<2m	medium
Very high	*Lampropeltis getula*	Common Kingsnake	VH	H	M	A4	MAC, MED	manual, trapping	high	medium	€ 200	€ 1,000	minimal	v. high	1-3	low
Very high	*Lonicera morrowii*	Morrow's Honeysuckle	H	VH	M	A2	ATL, CON, MAC, MED	manual, herbicide	v. high	high	€ 1	€ 50	minor	v. high	1-3	medium
Very high	*Micropterus dolomieu*	Smallmouth bass	VH	H	M	A1	MAC, MED, ATL, CON, STE	fyke netting, electrofishing	high	high	€ 50	€ 200	minor	high	2m-1	high
Very high	*Misgurnus mizolepis*	Chinese weather loach	H	VH	H	A1	MAC, MED, ATL, CON, STE	draining, piscicide	v. high	v. high	€ 1	€ 50	minimal	v. high	2m-1	low
Very high	*Oreochromis aureus*	Blue tilapia	VH	H	H	A2	MAC, MED	netting, angling	high	high	€ 50	€ 200	minimal	high	1-3	medium
Very high	*Oreochromis mossambicus*	Mossambique tilapia	VH	H	H	A2	MAC, MED	draining, piscicide	v. high	high	€ 1	€ 50	minimal	v. high	2m-1	medium
Very high	*Oreochromis niloticus*	Nile tilapia	VH	H	H	B2	MAC, MED	draining	v. high	high	€ 1	€ 50	minimal	v. high	1-3	low
Very high	*Pachycondyla chinensis*	Asian Needle Ant	H	VH	M	B1	MED, ATL, CON, STE, MAC	baiting, insecticide	v. high	high	€ 1	€ 50	minimal	v. high	2m-1	medium
Very high	*Rubus rosifolius*	Roseleaf Bramble	H	VH	M	A1	MAC	manual, herbicide	high	v. high	€ 1	€ 50	minimal	high	2m-1	low
Very high	*Sirex ermak*	Blue-black Horntail	H	VH	H	A1	CON, STE, BOR	incineration	v. high	v. high	€ 50	€ 200	minimal	v. high	<2 m	medium
Very high	*Solenopsis invicta*	Red Imported Fire Ant	H	VH	M	A1	MAC, MED	poison baiting	v. high	v. high	€ 1	€ 50	minimal	v. high	2m-1	high
Very high	*Trichosurus vulpecula*	Brushtail Possum	H	VH	H	A4	ATL, MED, CON, MAC	trapping	v. high	v. high	€ 50	€ 200	minimal	high	1-3	v. low

The results of this study demonstrate the value of incorporating both risk assessment and risk management criteria when prioritising invasive non-native species. There was no correlation between risk management and risk assessment scores, indicating that risk management evaluates information that is different to risk assessment. This additional risk management information is fundamental to decision-makers, who must take into account a wide range of criteria that go beyond risk (Simberloff, 2003b, Dana et al., 2014, Kerr et al., 2016). Indeed risk management is traditionally included along with risk assessment as part of an overall approach to risk analysis in other disciplines, such as plant health, animal health and food safety (Ahl et al., 1993, EFSA, 2010, FAO, 2013, OiE, 2017). Despite the important role of risk management, it is rarely used alongside risk assessment to prioritise invasive non-native species, particularly in the EU, where risk assessment alone has been the dominant method used to support prioritisation (Essl et al., 2011b, Heikkilä, 2011, Kerr et al., 2016, Turbé et al., 2017, Vanderhoeven et al., 2017, Roy et al., 2018b). The lack of invasive non-native species risk management is highlighted as an important gap in most existing prioritisation and listing at national and continental scales (Dana et al., 2014, Epanchin-Niell, 2017). It is recommended that systematic risk management methods, such as the NNRM, should be applied routinely as is commonplace in other biosecurity areas. While there are increasing calls for the application of risk assessment to more species (Carboneras et al., 2018), it is also suggested that there should be at least as much focus on risk management.

The results of this study can be used to explore the relationship between species traits and the feasibility of eradication at different scales. There was a clear effect of both environment and extent (total area and number of populations) on the Overall Score (for feasibility of eradication). In all environments, Overall Score decreased as extent increased (Fig 5.1). This was expected, as elements of feasibility, for example cost and resource effort, scale with extent (Brockerhoff et al., 2010, Howald et al., 2007, Martins et al., 2006, Rejmánek & Pitcairn, 2002, Robertson et al., 2017). However, until now it has been difficult to investigate the relationship between scale and feasibility as few report on or attempt eradications unless they consider them likely to succeed (Robertson et al., *in prep-b*).

Terrestrial species received highest scores for feasibility of eradication overall, followed by freshwater species and then marine species, which reflects the challenges of eradication in these different environments (Booy et al 2017). While the feasibility of eradicating terrestrial species was highest at smaller scales, it remained likely even at larger scales, albeit with lower confidence in high feasibility scores. This was not the case for freshwater species, where eradication was highly feasible at small scales, but quickly dropped at larger areas (1-10ha) and more populations (4-10). In the marine environment, feasibility was generally low, even at small extents; however, eradication may yet be feasible at the smallest extent (<1ha and 1-3 populations) in some circumstances. These results indicate that extent alone is not a good predictor of feasibility when comparing species from different environment. Both extent and environment need to be considered.

Early detection and rapid response will usually be most important where the feasibility of eradicating a species is initially high at small extents, but quickly drops as extent increases. The results of this study therefore suggest that early detection and rapid response may be particularly important for freshwater species and of lesser importance in the terrestrial and marine environments. While there are undoubtedly benefits to detecting and responding to terrestrial species at an early stage of invasion, the response may not have to be as rapid, give that feasibility is likely to remain high even at larger extents. The situation is different in the marine environment, where early detection and rapid response may not be a priority because there is little chance of delivering successful eradication even where species are detected at the smallest extents. This may not always be the case for marine species, for example successful responses have been delivered in specific circumstances (Bax et al., 2002, Wotton et al., 2004); however, if used, early detection and rapid response must focus on the rare circumstances in which eradication may be feasible.

The results of this study indicate species that are potential priorities for management in the EU. This includes species which are currently extant in the EU for which eradication now is considered feasible, based on expert judgement, and would result in significant benefits in terms of risk avoided. The top priorities for established species were terrestrial vertebrates with small population sizes and small areas (common myna, *Acridotheres tristis*; Berber toad, *Bufo mauritanicus*; Coati, *Nasua nasua*; red-vented bulbil, *Pycnonotus cafer*). This broadly reflects experience from the EU and elsewhere, where eradication campaigns have often targeted terrestrial vertebrates in small areas (Genovesi, 2005) and sometimes across wider extents (Robertson et al., 2015). However, the next ten priorities represented a much wider range of taxa including plants, invertebrates and fish, suggesting there may be scope to widen the taxonomic range of eradications in the EU. Results also indicate that eradication is not only feasible for the top fourteen species, but relatively inexpensive (total cost estimate to eradicate top four priorities was €0.45-2.25M, while total cost for the next ten was €1-5.5M) in comparison to EU funding for other invasive non-native species projects (Scalera, 2009). Lower scores for some risk management components suggest potential barriers to eradication that would need to be overcome, such as medium acceptability scores for eradicating the *N. nasua* (coati), *Axis axis* (Indian spotted deer) and *Rhea americana* (greater rhea) indicating a potential lack of public or stakeholder acceptance for this work. Acceptability was also a potential barrier for the eradication of *Misgurnus anguillicaudatus* (oriental weatherfish), but unlike the terrestrial vertebrates this was because of potential public concern over the use of piscicides rather than the charismatic nature of the species. Gaining access is a potential barrier to the eradication of some plant species, especially where they grow in difficult terrain. This was the case for *Euonymus fortunei*, which received a low practicality score because the most likely invasion scenario included the potential for its establishment on cliff edges. While these barriers are challenging and would have to be addressed as part of an eradication strategy, they were not considered insurmountable.

Contingency planning helps to ensure rapid eradication is delivered efficiently and effectively and is commonly used in the disciplines of plant and animal health (Wittenberg and Cock, 2001); yet it is rare outside of these areas for invasive non-native species in the EU. Indeed, the Asian hornet contingency plan of the UK is one of very few plans developed specifically for an invasive non-native species (i.e. not including plant and animal health pests and diseases) in the EU (Defra, 2017). The results of this study suggest that a total of 43 species are potential priorities for contingency planning, although 17 are a particularly high priority ('highest' and 'very high'). These priority species could establish in almost any region of the EU and would require a quick response (<1year) to improve efficacy and reduce cost. In addition, response teams would need to be capable of using a wide range of management techniques given that for the 17 high priority species at least 13 broad eradication techniques were identified. This suggests that for contingency responses to be effective, coordination across the EU would be vital to encourage the development and timely deployment of plans. A further challenge is that Member States would have to maintain or gain access to a broad range of management expertise and capacity, which may be lacking in some cases.

While the main role of the NNRM is to identify priorities for eradication and contingency planning, it also identifies potential priorities for long-term management and prevention. Long term management is likely to be a priority for established species where the feasibility of eradication is low. For example, *Arthurdendyus triangulatus* (New Zealand flatworm) for which the feasibility of eradication from its current EU distribution was considered very low, but for which slowing spread through phytosanitary measures may be feasible (Boag and Yeates, 2001). Similarly, the NNRM can identify potential prevention priorities for species that are not yet established where the feasibility of eradication is low. For example, should *Homarus americanus* establish in EU waters it is unlikely that eradication would be feasible and so prevention, perhaps by tightening control of its release and escape pathways (van der Meeren et al., 2016), should be considered a particularly high priority. A limitation of the NNRM is that it does not currently evaluate the effectiveness of long-term management or prevention measures. This is important because long term management may not always be feasible for species that cannot be eradicated, for example it seems unlikely that long term management would have much lasting impact on the spreading population of *Pterois miles* (lion fish), despite calls for its consideration (Kletou et al., 2016). Similarly, prevention may not always be feasible, as is likely to be the case for *Plotosus lineatus* (striped eel catfish) which seems set to establish in EU waters following its arrival through the Suez Canal (Edelist et al., 2012). Where considering future prevention and long term management priorities these factors need to be taken into account and this is a priority for further development of the NNRM.

The results of this study have application for policy and management. Given that much of the current focus of listing and management in the EU has been on widespread species (Lehtiniemi, 2016), they help to redress the balance and focus more attention on the eradication of species with limited distributions and contingency planning where this is feasible. EU regulation 1143/2014 (EU, 2014b) requires risk assessment to support the listing of invasive non-native species (Article 5); however, there are also elements of risk management in the regulation that cannot be provided by risk assessment. The approach used there helps to address these, including providing a method to assess the feasibility of eradication (Article 17), supporting the development of management plans (Article 19) and evaluating the potential benefits of listing (Articles 4.3e and 4.6). Regulation 1143/2014 offers a route to help deliver eradications and contingency planning across the EU, based on the requirements in Chapter 3 'early detection and rapid eradication'. However, Member States are only required to eradicate listed species if they were not already present at the time of listing (Articles 16 and 17), which means listing will not necessarily result in the eradication of emerging (i.e. already established) species. Even when listed species are detected for the first time in a Member State, eradication is only required if it is considered feasible (Article 18). However, there is no agreed method for determining whether eradication is feasible and so application is likely to be subjective and potentially inconsistent across the Union. These limitations suggest that listing alone may not be sufficient to drive EU wide eradication and contingency planning for species identified as priorities. Other mechanisms may be needed to drive this, for example specific eradication and contingency planning programmes under the EU LIFE funding stream. Such programmes would need to be coordinated across the EU and would benefit from sharing of expertise. In addition, systematic processes are required to evaluate whether eradication of new invasive non-native species is feasible or not to support the application of Article 18.

Given little attention has been given to invasive non-native species risk management to date, there are numerous opportunities for development and improvement. Both the risk assessment scores (derived from horizon scanning) and risk management scores used here were developed through rapid assessment. As such, further detailed analysis is recommended, as is common following screening exercises (D'hondt et al., 2015, Tanner et al., 2017). Tools are available to support more detailed analysis relating to some aspects of risk management, such as cost-benefit and cost-effectiveness analysis (e.g. Blackwood et al., 2010, Courtois et al., 2018); however, it may also be useful to develop more detailed and comprehensive risk management tools that expand upon the criteria identified here. For example, while short / rapid risk assessment (e.g. Defra, 2015, Tanner et al., 2017) is used to help identify initial priorities, these are often followed by more detailed and comprehensive risk assessments (e.g. Baker et al., 2008, EPPO, 2011). In a similar way, the criteria used here for assessing feasibility of eradication could be further expanded and divided into more specific and detailed questions to facilitate such analysis. A strength and limitation of this approach is that risk management scores are dependent on user defined scenarios. This allows assessments to be made for species that are not yet established, using the most likely scenario of invasion. It also allows for scenarios to be defined for established species where current extent is not certain. There is also potential to examine multiple scenarios to examine the effects of extent or the choice of control method on feasibility. However, in some cases there was uncertainty in the scenarios used. For example the current extent of *A. tristis*, .*B. mauritanicus*, *P. cafer*, three of the four top priorities for eradication, was not clear. It is recommended that scenarios are refined as part of future detailed analysis, using the most up to date understanding of current and potential future species distributions. Where uncertainty remains it may be useful to assess a range of potential scenarios for species to reflect this.

Only 'high' and 'very high' risk species were included in this exercise and potential priorities may have therefore been excluded (for example those with 'medium' risk but 'very high' feasibility of eradication = 'high' priority). It is recommended that, in future analysis, any species with the potential to result in more than minimal impacts should be screened using both risk assessment and risk management criteria. Also, only species with no or limited distributions were included in this assessment. Including species with a wider range of current distributions would improve understanding of the effect of extent on the feasibility of management and to investigate whether there are more widespread species in the EU for which eradication is both feasible and could be considered a priority.

Chapter 6. Discussion

6.1 Main findings

The number of non-native species establishing worldwide is increasing (Seebens et al., 2017). In Great Britain the numbers of established non-native species have increased rapidly since the industrial revolution (Roy et al., 2014c), with close to 2000 non-native species currently established and an average rate of establishment of 10.7 new species per year from 1950-2017 (NNSIP, 2017). A small proportion of non-native species cause substantial negative impacts and can therefore be considered invasive (Williamson and Fitter, 1996). This study provides, for the first time, a comprehensive assessment of the number of non-native species that cause biodiversity impacts in Great Britain (GB) and the severity of those impacts (Chapter 2). The proportion of species establishing with native origins outside of Europe was found to be increasing throughout the 20th century, as was the number of species establishing in aquatic environments (Chapter 2). This is of concern as both groups were found to be more likely to cause serious impacts than European natives or terrestrial species (Chapter 2). However, while the proportion of these groups increased over time, the overall proportion of established invasive non-native species declined after 1920 (Chapter 2). This may be the result of lag in the detection of impact, in line with the concept of invasion debt (Essl et al., 2011a), and suggests that future impacts in GB may be more severe than is currently understood (Chapter 2).

While the number and impact of non-native species in GB is increasing (Chapter 2), management is often expensive and resources limited (Kumschick et al., 2015b, McGeoch et al., 2016). Just eradicating a single species can be extremely expensive, for example the Ruddy Duck eradication programme in the UK has cost £5.79M (GBP) to date (I. Henderson 2019, *pers comm*). Failed management attempts are underreported in the literature (IUCN, 2018), but can also be costly. It is therefore essential to prioritise management carefully to ensure cost-effective resource allocation and reduce ineffective expense (Cassey et al., 2018b, Courtois et al., 2018). However, the sheer number of species involved and the wide range of possible management interventions means that identifying priorities is complex (Woodford et al., 2016). Support is therefore required to guide decision-making and the subsequent allocation of resources (McGeoch et al., 2016). To this end there has been much focus on the use of risk and impact assessment (e.g. Essl et al., 2011b, Hawkins et al., 2015, Bacher et al., 2017, Roy et al., 2018b); however, practical methods that relate specifically to management actions are largely lacking (Hulme et al., 2009, Heikkilä, 2011, Vanderhoeven et al., 2017). The aim of this study was therefore to develop and test methods to support the prioritisation of invasive non-native species management. This focussed on prevention and eradication interventions, as these were more likely to be cost-effective (Mack et al., 2000, Simberloff, 2003a, Genovesi and Shine, 2004); however, implications for long-term management were also considered.

The importance of comprehensive impact information was demonstrated (Chapter 2), which, in addition to supporting the analysis of patterns and trends in species' impact, was also essential for pathway analysis (Chapter 3). While pathway importance is often assessed based on counts of all non-native species (e.g. CBD, 2014c, Roy et al., 2014c), this study demonstrated that incorporating impact data, as well as uncertainty and change in pathway impact over time, produced substantially different results that were more likely to result in cost-effective management (Chapter 3). This highlights the importance of selecting prioritisation methods that reflect the objectives of management.

In addition to taking into account the impacts of invasive non-native species, managers must also consider the feasibility of management (Vanderhoeven et al., 2017). This study developed a novel risk management method to assess the feasibility of eradicating species and tested it across a wide range of taxa, environments and at different scales (Chapters 4 and 5). The results of these applications indicated that the feasibility of eradication can not only be successfully evaluated for diverse species and contexts, but that it can be combined with existing risk assessment scores (e.g. Baker et al., 2008, Roy et al., 2014b) to indicate potential management priorities. This approach incorporated management information that is essential to decision-makers (Simberloff, 2003b, Dana et al., 2014, Kerr et al., 2016, Epanchin-Niell, 2017), but was demonstrated not to be taken into account by risk assessment alone (Chapter 5). This is important as it indicates that priorities based on risk assessment are likely to align poorly with the priorities of decision-makers. Indeed, this has been borne out by a comparison of the cost and benefits of eradication based on priorities identified by risk assessment, risk management and a combination of the two (Robertson et al., *in prep*).

Using this novel risk management approach, priority species for eradication and contingency planning in GB and the EU were identified, as well as potential priorities for prevention and long term management. The successful application of this method at these two very different scales demonstrates the practicality and flexibility of the approach. This is important given the need for such methods at these scales (CBD, 1992, McGeoch et al., 2016, Scalera et al., 2016) and given the large numbers of species and differing contexts in which they have to be applied.

Methods to support pathway and species prioritisation developed here have relevance to decisions made at local, national, regional and international scales. For example, at the international scale signatories to the Convention on Biological Diversity (CBD) are committed to prioritising pathways of non-native species introduction and the management of species (CBD, 1992), while regionally the need for pathway analysis and risk management across the EU has been stressed (Tollington et al., 2017). This work has already had an impact at a national scale in GB where it has been used to inform pathway management, resourcing and eradication (Defra, 2015) and is being used to support the prioritisation of species of European Union Concern in Belgium (Adriaens et al., 2018). Developing standards for pathway and species prioritisation (including cost-benefit analysis), similar to those already produced for pathway classification and impact assessment (Hawkins et al., 2017, Harrower et al., 2018a, Roy et al., 2018b), could be an important next step for the international community. Indeed, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services recently approved the undertaking of a thematic assessment of invasive non-native species to consider, among other things, what methods are available for prioritizing invasive non-native species threats incorporating the cost / benefit of management (IPBES, 2019). Such a platform may be a useful place to develop and agree standards for pathway and species prioritisation. The methods tested here were developed to support such work.

6.2 A framework for prioritising species and pathway management

These pathway ranking and risk management methods contribute to a proposed overall framework for the prioritisation of non-native species management (Fig 6.1). Within this the risk posed by species and the feasibility of their management (based on differing management objectives) is assessed separately and then combined to indicate potential priorities for species management. Similarly, the risk posed by pathways would also be assessed and compared to the feasibility of management, to indicate potential pathway priorities. While methods to complete some of the components of this framework are well developed (e.g. risk assessment, see below), others are lacking or at the early stages of development. The components of the proposed framework are discussed in turn below (starting with species risk assessment as this is the most well developed part of the framework) as well as remaining challenges and potential issues with its implementation.

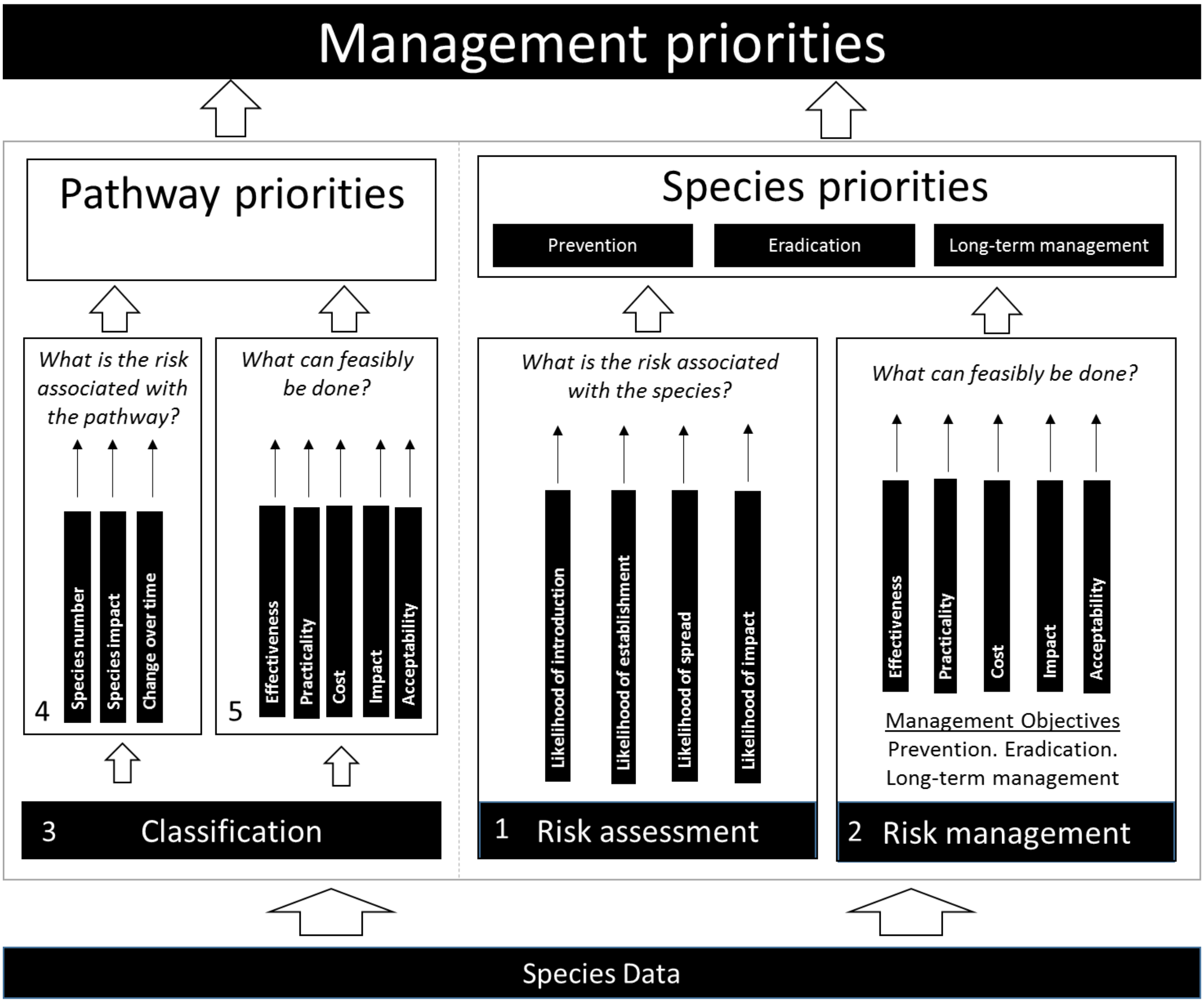


Figure 6.1 Proposed framework the prioritisation of invasive non-native species and pathway management based on risk analysis components.

6.2.1 Species risk assessment (1)

Non-native species risk assessment methods are well developed (reviews in Essl et al., 2011b, Leung et al., 2012, Kumschick and Richardson, 2013, Lodge et al., 2016, Vanderhoeven et al., 2017, Roy et al., 2018b) and provide a means of systematically evaluating each component of risk (e.g. introduction and/or entry, establishment, spread and impact) before determining an overall risk score (Vilà et al., 2018). These can be qualitative (Branquart, 2007, Peel et al., 2012, D'hondt et al., 2015), quantitative (e.g. Leung et al., 2002), or 'semi-quantitative' (e.g. Mumford et al., 2010, Sandvik et al., 2013) and more or less complex, depending on the purpose of the assessment and the level of evidence available (e.g. Brunel et al., 2010, Mandrak et al., 2012, Roy et al., 2014b, Tanner et al., 2017, Vilà et al., 2018). Tools have also been developed to support assessment of the individual components of species risk assessment, for example climate matching tools to support the assessment of establishment and various impact evaluation methods, such as the Generic Impact Scoring System (Nentwig et al., 2016), Invasive Species Environmental Impact Assessment (Branquart, 2007), Environmental Impact Classification of Alien Taxa (Blackburn et al., 2014), Socio-Economic Impact Classification of Alien Taxa (Bacher et al., 2017) and others (e.g. Poutsma et al., 2008, Holt et al., 2014). Non-native species risk assessment schemes have also been subject to testing and review (Hulme, 2012, Keller and Kumschick, 2017, Matthews et al., 2017, Vanderhoeven et al., 2017), the recommendations of which should be use to continue to refine approaches and achieve consistency. Indeed, development and testing has advanced sufficiently that international minimum standards for non-native species risk assessment have now been identified (Roy et al., 2018b).

6.2.2 Species risk management (2)

In contrast to risk assessment, few methods or standards have been developed to evaluate non-native species risk management criteria (Heikkilä, 2011, Tollington et al., 2017), hence the development of such methods was one of the main aims of this research. As with risk assessment, risk management can be used to assess the feasibility of management based on an evaluation of its component parts (Johnson, 2009a, Booy et al., 2017, Robertson et al., *in prep*). Over two separate tests of this method, at differing scales, this study found that the key components (effectiveness, practicality, cost, impact and acceptability), combined with clearly set management objectives, scenarios and management strategies, allowed for a robust account of management feasibility (Chapter 4 and 5). Risk management methods developed here were only applied to assess the feasibility of eradication and so it is important to develop this approach further to consider how it can be applied to assess different objectives, including prevention and various forms of long term management. For example, in Belgium the scheme has been developed to assess the feasibility of containing an established population in order to compare this to the feasibility of eradication (Adriaens et al., 2018). Further risk management development is required to bring these evaluations to a similar level to those of risk assessment. This could include elaborations of the current scheme to further define and attempt to quantify each risk management component (i.e. effectiveness, practicality, cost, impact, acceptability, window of opportunity and likelihood of reintroduction), expand the approach to include other management objectives (e.g. prevention, removal, suppression and containment, Robertson et al. (*in prep-a*)) and continue to apply and test the approach.

6.2.3 Pathway classification (3)

An important starting point for pathway prioritisation is to ensure that pathways are carefully classified using appropriate terminology and to a level of detail that is useful for managers (Essl et al., 2015). For example, very broad pathway categories may be useful for analysis (to increase sample size), but less useful for decision-makers determining which specific pathways to manage (McGeoch et al., 2016, Harrower et al., 2018a). Hierarchical pathway structures as currently used by the CBD pathway classification can therefore be useful (Hulme et al., 2008, Scalera et al., 2016). While other schemes are available (Leung et al., 2014, Madsen et al., 2014, NOBANIS, 2015), wide-scale adoption of the CBD classification should help to provide consistency and allow for comparison between studies (CBD, 2014c, Harrower et al., 2018a). However, the CBD classification requires testing and may need to be adapted as it is applied in different situations and at difference scales. This study found that the CBD classification could be applied at a national scale, but that in some cases additional pathway detail was required and pathway definitions were not always clear (Chapter 3). Given that additions or modifications may be required to the CBD classification, but that these may have implications for the major databases currently using the scheme, it is suggested that an international platform could help to oversee, agree and ensure consistency in proposed modifications.

6.2.4 Pathway ranking and risk assessment (4)

A number of studies have scored or ranked pathways (CBD, 2014b, Nunes et al., 2015, Saul et al., 2017), while in other cases broad guidance has been suggested indicating what variables may need to be considered to support prioritisation (Essl et al., 2015, McGeoch et al., 2016); however, formalised or widely adopted methods for non-native species pathway risk assessment are lacking (McGeoch et al., 2016, Tollington et al., 2017). It was therefore an aim of this study to devise and test a number of potential methods for ranking pathways (Chapter 3). This demonstrated that the method chosen had a substantial effect on the resulting ranks.

Given that the aim of management is to reduce future impacts, it is suggested that pathway ranking methods should, as a minimum, incorporate both the impact of species and change in pathways over time. Uncertainty could have an important effect on the ranking of pathways, including those in the higher ranked positions, and so should also be taken into account (Chapter 3). This study demonstrates methods for incorporating impact, uncertainty and temporal variables; however, further refinement and testing would be beneficial.

Methods used to rank pathways take a different approach to prioritisation than methods based on pathway risk / vector analysis (e.g. Carlton and Ruiz, 2005, Baker et al., 2008, Leung et al., 2014, Brancatelli and Zalba, 2018). The latter generally focus on individual pathways, separating them into separate components (e.g. origins, vector identity, vector intensity, vector tempo) and assessing each component (Carlton and Ruiz, 2005); rather than ranking methods, which compare across pathways and assess relative risk based on impact (in this case). Ranking and pathway risk assessment could therefore be complementary, in that ranking could be used to identify initial priorities. Pathway risk assessment / vector analysis could then be undertaken on high ranking pathways to provide a more detailed assessment of, for example, the origins, route, destination, tempo, volumes and vectors associated with each pathway (Carlton and Ruiz, 2005).

6.2.5 Pathway risk management (5)

There is little literature relating to schemes or tools designed to support pathway risk management, although examples relating to the management of individual pathways (examples given in Essl et al., 2015) and general guidance on pathway management (e.g. CBD, 2014b) are available. While the ranking of pathways according to their potential impact may provide much of the information needed for prioritisation, it is also important to consider the feasibility of management. For example, on one hand there may be pathways that are relatively small / low impact, but for which management is simple and can be implemented effectively to reduce the risk of further species being introduced. On the other hand there may be pathways that introduce high impact species, but for which little, if any, risk reduction can be achieved.

It may be possible to apply the same risk management criteria used for species (Chapters 4 and 5) to pathways. In this case a pathway management strategy would be defined and criteria used to assess its effectiveness (amount of risk likely to be reduced), practicality (ability to implement pathway management), cost (direct cost of implementing pathway management), impact (adverse effects / indirect costs to the environment, economy or society of implementing pathway management) and acceptability (whether pathway management would be resisted by stakeholders or the general public). If this approach were used, consideration would have to be given to how to evaluate the level of risk reduction that would be achieved by implementing the management strategy (i.e. a starting risk and a risk post management would have to be determined in each case). This is not a trivial challenge; however, schemes that measure risk reduction associated with pathway management have been developed in relation to plant health and could be informative (Baker et al., 2014).

6.2.6 Interchangeability

An advantage of this proposed framework is that different methods can be used to complete the different components. For example, this study demonstrates that different forms of risk assessment (e.g. Mumford et al., 2010, Roy et al., 2014b, Roy et al., 2015b) can be combined with risk management scores to indicate species priorities (Chapter 4 and 5). This means that where prioritisation is required at different scales and in different territories the most appropriate methods to that situation can be used. For example, simpler and more rapid methods (e.g. Daehler et al., 2004, Copp et al., 2009, D'hondt et al., 2015) could be used where time and resources are low, particularly if only an indication of potential priorities is required. Whereas, more detailed methods may be needed where a higher standard of evidence is required, for example when underpinning legal decisions (Shine et al., 2000, Baker et al., 2008, EPPO, 2011).

6.2.7 Data requirements

This framework helps to identify data and evidence needed to support the prioritisation of invasive non-native species management (e.g. taxa, environment, functional group, native origin, year of introduction, impact, pathway of original introduction, total area occupied and number of individual populations based on existing or potential future scenarios). Indeed, for some components there is an overlap in data requirements. For example, species risk assessment relies on pathway data (used in pathway assessment) to evaluate the risk of introduction; whereas, pathway assessment relies on species impact data, which is also included in risk assessment. A number of major international databases provide much of this data for many species (e.g. NOBANIS, DIAISE, GISD and NNSIP). For example, the CABI Invasive Species Compendium is one of the most comprehensive databases, providing full datasheets for 2565 non-native species (http://www.cabi.org/isc, *accessed January 2019*). However, while these databases often include valuable data to support species and pathway risk assessment, less data tends to be held on management interventions. Where management data is held, it tends not to be gathered in the same systematic way (i.e. broken into component parts) that is used for risk assessment data (which is broken down, for example, into data relating to entry, establishment, spread and impact). For example, the full datasheet used to collate data on invasive plants within the CABI Invasive Species Compendium contains 26 categories that relate to species risk and pathways; however, only one that pertains directly to management (‘prevention and control’), with two additional categories that are of potential indirect value (‘uses’ and ‘uses list’). Such databases are a vital source of information, but could be of more use for prioritisation by systematically gathering data on pathway and species management (e.g. effectiveness, practicality, cost, impact and acceptability).

6.2.8 Gaps

Gaps in the methods available to complete this framework have been discussed, in particular the need for species risk management methods to assess the objectives of prevention and long-term management, and pathway risk management methods in general. However, a limitation of this framework is that it does not indicate how to prioritise between the objectives of prevention (both pathways and species), eradication (species) and long term management (species). In other words, while for each objective the framework should indicate species and pathways that are likely to be priorities, it does not indicate how to prioritise between these. In order to do so it would be necessary to compare the cost-benefit of managing the species and pathways identified as priorities under each of these objectives and then select the combination that achieved the greatest impact reduction for least input (McGeoch et al., 2016). However, further research is needed to firstly develop the methods to complete the prioritisation framework and then compare across priorities to achieve the greatest impact reduction for least input. Despite this issue, there is agreement that prevention and eradication will generally result in greater cost-benefit than the long term management of an invasive non-native species (Mack et al., 2000, Genovesi and Shine, 2004, Jones et al., 2016). It is therefore likely that cost-benefit analysis would rank the majority of prevention and eradication priorities before those of more expensive and less effective long term management.

6.3 The importance of systematic research into species and pathway management and the use of these to support prioritisation

A key finding of this study was that evaluating the feasibility of management contributes important information that is critical for prioritisation, but it is often overlooked (Chapters 4 and 5). Using risk assessment alone to prioritise species for management is likely to result in an inefficient allocation of resources (Chapters 4 and 5). Incorporating risk management information is likely to produce different priorities that are more closely aligned to management objectives (Chapters 4 and 5). Indeed, a recent study using the risk management methods developed here found that prioritising species management by combining risk assessment and risk management produced significantly better cost-benefit than using risk assessment alone (Robertson et al., *in prep*). While it has not been possible to test pathway risk management methods as part of this study, it is likely that a similar relationship would be revealed.

A number of ongoing issues with the implementation of the EU invasive alien species regulation (EU, 2014b) highlight some of the practical implications of failing to take into account risk management criteria. While a substantial amount of work has been carried out to ensure the listing of species under the Regulation is supported by risk assessment (e.g. Roy et al., 2014a, Roy et al., 2015a, Roy et al., 2015b, Roy et al., 2018b), there is little relating to risk management (Tollington et al., 2017). This has led to concerns that too many widespread species are being listed, while higher priority prevention and eradication priorities may be overlooked (Lehtiniemi, 2016). There have also been concerns that the negative consequences of listing, such as implications to business and society, have not been adequately taken into account (Nielsen, Undated). Recognising this, the European Commission has recently started to place more emphasis on systematically gathering risk management information (IUCN, 2018) and some individual Member States have undertaken their own risk management evaluation of species of Union Concern (e.g. Adriaens et al., 2018).

Despite the importance of evaluating non-native risk management criteria, there is relatively little literature on this subject (reviewed in Chapters 4 and 5). Indeed, research has tended to focus on broader questions relating to invasion biology than in the application and implementation of management (Esler et al., 2010). This may be because management is not considered sufficiently novel, impactful or fundamental as a scientific research subject, which indeed appears to be the case based on the relatively low citation rate of papers on this subject (Pyšek et al., 2006). However, it may also reflect a lack of sufficiently close links between researchers and practitioners, the latter not being necessarily incentivised to publish their work in the primary literature and publishing grey literature instead (i.e. project reports, manuals, technical notes, etc.). Indeed, the gap between researchers and practitioners has been highlighted in other areas in conservation (Nature, 2007, Esler et al., 2010) and can lead not only to a lack of evidence being gathered that is useful to practitioners, but poor application of what evidence there is (Sutherland and Wordley, 2017). This study highlights the need for more research into the management of invasive non-native species and pathways and in particular investigation into how this data can be used to support prioritisation.

Such data should, ideally, be gathered systematically, helping to reveal trends and patterns in the effectiveness, practicality, cost, impact and acceptability of the management of pathways and species at different scales and in different situations. As is the case in other areas of conservation, there are likely to be considerable benefits in encouraging greater collaboration between practitioners and researchers (Young et al., 2014). There may also be benefits in making it easier to document and access evidence, for example through initiatives such as Conservation Evidence (Sutherland et al., 2004). Indeed, a synopsis of freshwater invasive non-native species management has been conducted by Conservation Evidence (Aldridge et al., 2017); however, this demonstrates the lack of management evidence in many cases. In future, such reviews would benefit from considering the valuable management information contained in grey literature and from evaluating management effectiveness against carefully defined management objectives.

6.4 Decision-making under uncertainty

Handling uncertainty is a common theme in this thesis and many studies relating to invasive non-native species as uncertainty occurs across the invasion process (Moffitt and Osteen, 2006, Baker et al., 2008, Essl et al., 2011b, Liu et al., 2011b, Leung et al., 2012, McGeoch et al., 2012, Copp et al., 2016, Lodge et al., 2016, Essl et al., 2018). It arises for different reasons, for example lack of information, conflicting evidence, context dependence or imprecise definitions and guidance (Vanderhoeven et al., 2017) and can be reducible (e.g. by gathering more information) or irreducible (e.g. natural variation that results in a probabilistic outcome) (Leung et al., 2012). Even where uncertainty is reducible, the sheer volume of species and pathways means it is unlikely that sufficient evidence could be gathered to provide high confidence in all aspects of prioritisation (McGeoch et al., 2016). However, management decisions must still be made despite uncertainty (Sutherland and Burgman, 2015). A key aim of this research was therefore to develop pathway ranking and risk management methods that could be used even where data are lacking or incomplete.

6.4.1 Using expert judgement to overcome data limitations and guide research

Expert scoring, based on predefined semi-quantitative scales, coupled with consensus building methods was used throughout this study to help overcome the limitations of incomplete data (Chapters 2, 4 and 5). This provided a practical means of eliciting standardised scores for large numbers of species across taxa, environment and situation where data were incomplete; which would not have been practically achieved using traditional methods. There were also benefits in the knowledge exchange created by bringing together a large and diverse group of experts that work in different areas and do not engage with each other on a regular basis (similar to Roy et al. (2014b)).

Expert knowledge is used to support decision-making in conservation biology in general (French, 2012, Martin et al., 2012) and frequently used in relation to invasive non-native species (e.g. Baker et al., 2008, Essl et al., 2011b, Ricciardi et al., 2017, Vanderhoeven et al., 2017, Roy et al., 2018a). However, it is vulnerable to a range of cognitive biases (Morgan, 2014, Sutherland and Burgman, 2015) and in at least one case has been found to be less accurate than more empirical evidence (Drolet et al., 2015). It is therefore important to limit cognitive bias using structured techniques (Sutherland and Burgman, 2015, Hanea et al., 2017) for which there is a growing body of good practice (e.g. Burgman et al., 2011, Martin et al., 2012, Hanea et al., 2017). To this end, this study followed and developed on the approach used by Roy et al. (2014b), whereby judgements were elicited independently at first from a wide and diverse range of experts (using structured guidance) and then subject to a consensus building process within and then between experts grouped by taxonomic expertise (similar to approaches used by Ricciardi et al., 2017, Roy et al., 2018a). It was useful to introduce a Delphi like process (Vanderhoeven et al., 2017) whereby multiple (at least 3 sets of) scores for each species were elicited independently and used to inform a final score decided upon by the wider group. It was also essential that experts documented their scoring justification and uncertainty, which not only allowed uncertainty to be reflected in the final results but also allowed experts to provide judgements even where data were limited. Other techniques used to structure and elicit expert judgement in this study included training to improve use and understanding of the guidance, presentations of scores that provided participants the opportunity to discuss and provide challenge, the use of facilitator-led discussions to encourage engagement and open discussion, and the use of smaller breakout sessions to provide smaller and more informal space in which to express views. While these approaches were adopted to limit bias in expert knowledge, good practice in this field is developing rapidly and so further evaluation and adaption is recommended (e.g. following the recommendations of Hanea et al., 2017, Vanderhoeven et al., 2017, Dias et al., 2018).

Despite expert judgement providing a useful means of carrying out analysis where data are incomplete, it does not replace experimental data. Experimental research to test expert judgements can therefore be useful. Indeed, given limited research budgets and large numbers of species, expert judgement could provide a useful means of identifying priority species or pathways on which to focus further experimental research. For example, it could be particularly fruitful to focus research where species/pathway impact or feasibility of management were high, but confidence was low (Chapters 2, 3, 4 and 5).

6.4.2 Consequences of uncertainty for decision making

Where it exists, uncertainty in the assessment of invasive non-native species and their pathways should ideally be explicitly recorded and reported (Vanderhoeven et al., 2017, Harrower et al., 2018a). To this end, all of the methods developed in this thesis included explicit methods for recording uncertainties associated with impact scoring (Chapter 2), pathway ranking (Chapter 3) and risk management (Chapters 4 and 5). Different methods can be used to record uncertainty associated with scores, which range in detail and complexity (e.g. Liu et al., 2011a, Liu et al., 2011b, Holt et al., 2012, Caton et al., 2018); however, in this case it was important that uncertainty was recorded efficiently and so a relatively simple approach was used, following Mumford et al. (2010), which in turn is based on guidance provided by the Intergovernmental Panel on Climate Change (Mastrandrea, 2011). These uncertainties have important implications for decision making. For example, different introduction pathways would be prioritised depending on the threshold of uncertainty used (Chapter 3). Whereas, in relation to impact scoring (Chapter 2), uncertainty could result in benign species being incorrectly identified as harmful (and therefore targeted for management) or harmful species incorrectly identified as benign. In Chapters 4 and 5 uncertainty in both risk assessment (or horizon scanning) and feasibility of eradication scores have implications for the degree of confidence that can be assigned to the identification of management priorities. It is therefore important that decision-makers are aware of and are able to correctly interpret the implications of uncertainty.

Invasive non-native risk assessment schemes used by decision makers often report uncertainty (or confidence) scores alongside risk scores; however, as the confidence scores are provided separately to the risk scores they can be easily overlooked or their implication misinterpreted. It is therefore useful to consider methods for incorporating uncertainty more directly and helping decision makers to interpret the implications of uncertainty. Holt et al. (2012) do this by transforming individual risk assessment scores into probability distributions, using uncertainty scores to calculate a beta distribution (see also Mumford et al., 2010, Holt et al., 2014). Similar approaches could be applied to visualise uncertainty in species prioritisation, taking confidence in both risk assessment and risk management scores into account (e.g. Fig. 6.2). Here, Monte Carlo simulations are used to present

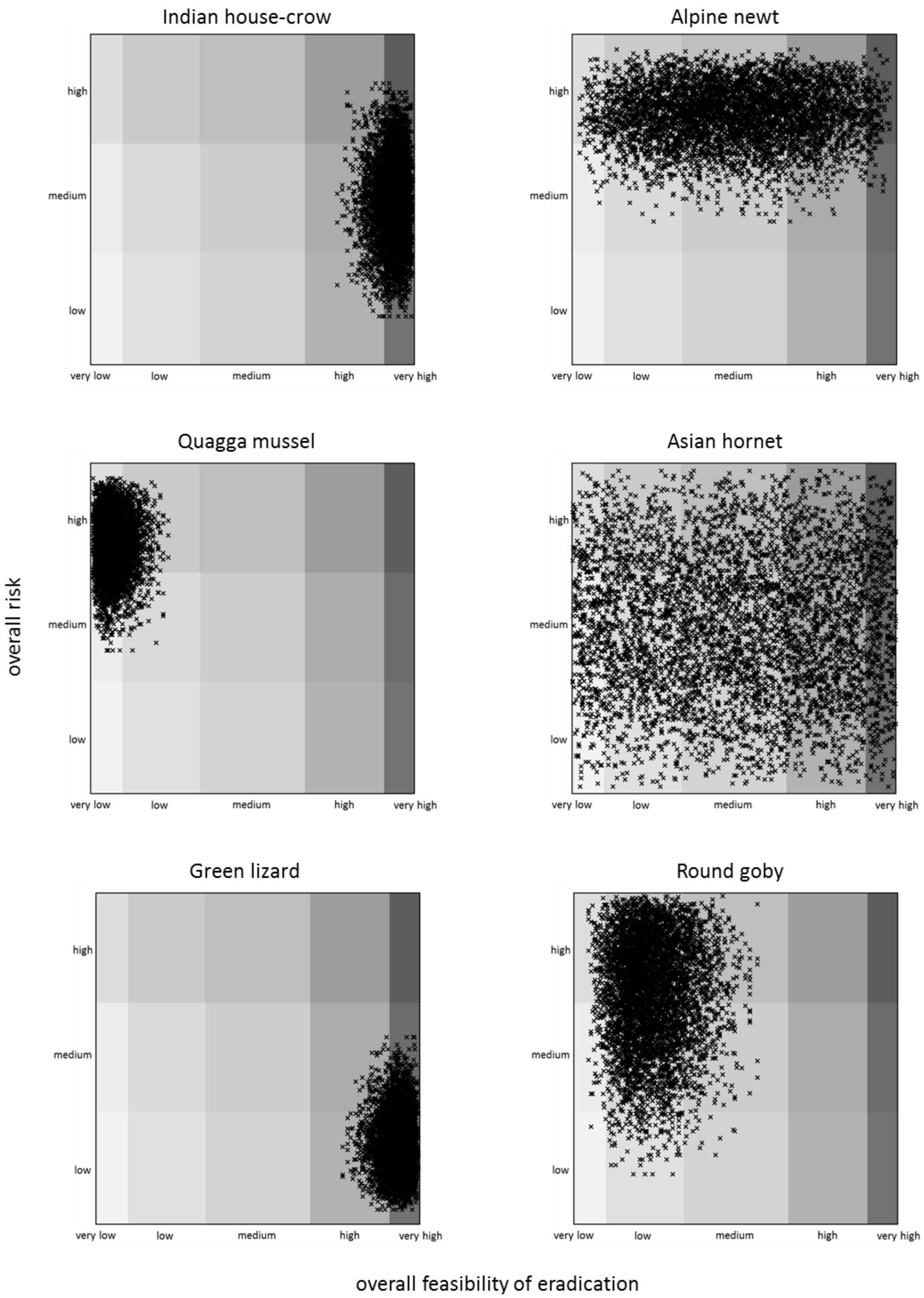


Figure 6.2 Illustrative plot visualising uncertainty within a prioritisation matrix. Assessments of confidence were used to transform individual scores of risk (y axis) and feasibility (x axis) into probability distributions using Monte Carlo simulations (1000 iterations). Closely clustered points indicate greater confidence.

priorities as cluster plots, with more compact clusters indicating higher confidence and more spread clusters indicating lower confidence (associated with either the risk assessment axis, risk management axis, or both). While this technique is a useful aid to visualising the implication of uncertainty, other methods could be considered for further incorporating uncertainty into decision making. For example, similar probability distributions could be modelled within a Bayesian network to assess the probability of different priorities being assigned to a species.

A further way to extend this analysis is to consider methods for optimising decision making while accounting for uncertainty, potentially taking a decision theory or information gap analysis approach. Decision theory relates to how agents chose between options and can be used to inform optimal (rational) decisions, or to assess how decision are made in reality (White, 2018). It is often used to explore probabilistic outcomes of different decisions and has been applied to invasive non-native species in relation to the optimisation of specific management actions (Mehta et al., 2010). Information-gap analysis provides a non-probabilistic means of assessing the robustness of decision making to uncertainty, assuming there are substantial knowledge gaps. It can be particularly useful for understanding how uncertainty may affect management decisions, and has been applied in this way to invasive species management (Burgman et al., 2010). The data requirements of such approaches mean they are particularly suitable for comparisons between specific management approaches for individual species and pathways, but may be less efficient at comparing across multiple species, pathways and management objectives. They could therefore be used to analyse in more detail initial management priorities identified by the methods developed in this thesis. A further and important application of the decision theoretic approach may be to help determine optimal allocation of resources between the broad management objectives of prevention, eradication and long term management. While priorities for each of these are considered in this thesis, optimising where and how to invest resource across these priorities could benefit substantially from decision theory (Epanchin-Niell, 2017).

6.4.3 Validating expert judgements

Validating expert judgement is difficult as it is used specifically where the information of interest is unknown. One approach is to validate judgements by assessing the performance of experts against test, or calibration, questions (e.g. Hanea et al., 2017). Calibration questions are those where the true value is known to those conducting the expert elicitation, but that are uncertain to the experts involved (i.e. the true values are not known or available to them). In the ‘classical model’ these questions can be used not only to validate the accuracy of experts, but to weight experts by their performance by combining 'statistical accuracy' and 'information' scores. Bolger and Rowe (2015) found that of the studies investigated, all performed better when weighted based on expert performance.

It would be useful to explore the use of calibration questions in future applications of the expert elicitation methods used in this thesis. For example, it may be possible to incorporate calibration questions on the impact of invasive species or the feasibility of eradication. However, there are also a number of challenges to overcome. Calibration questions must be derived from the expert's field, yet in this study experts were drawn from many different fields (e.g. invasive mammals, seaweeds, freshwater crustacea, etc.). Different calibration questions would therefore be needed for different expert groups, which could inhibit the consistent weighting of experts between groups. In addition, it may be difficult to identify sufficient calibration questions where the true value is known to those conducting the exercise but not the experts. For example, information about the impact of invasive species is available for a relatively small subset of species, and this information is likely to already be known by experts. It may be possible to overcome this by commissioning specific research, the results of which are not made known to experts; however, this would be resource intensive. The use of calibration questions would also require a different, and more involved, approach to the assessment of uncertainty, with experts providing scores at (for example) the 5th, 50th and 95th percentiles. While potentially useful, this may not be practical given the large numbers of species assessed.

A different approach would be to assess the judgements of experts against scores based solely on primary evidence available in published literature. While not strictly a form of validation, this would provide useful insight into the degree to which experts are aware of and utilising existing literature, as well as any gap between the perceived impacts of species and that which has been published. In the context of this study, it would be particularly interesting to compare impact scores derived from expert elicitation (Chapter 2) to those found solely in the literature, for example by following the EICAT protocol (Hawkins et al., 2015). Given that experts were expected to be aware of relevant literature, but also to draw on their own experience, we might expect expert impact scores to be consistently higher than those found in the literature. Where scores are lower, this would either reflect a lack of awareness of the published literature, or a rationale decision made by experts to downgrade evidence from the literature (perhaps because of conflicting research or judgement that impacts are no longer as severe).

While these approaches go some way towards validation, they are not a true test of whether the judgements made by experts are, in fact, accurate. For this, perhaps the only available approach is to revisit judgements once sufficient time has passed for predictions to be realised or management actions taken. For example, the GB non-native species horizon scanning exercise (Roy et al., 2014b) could now be validated by exploring which of the species predicted by experts to become invasive over a ten year period have subsequently done so. In the context of this research, experts have made predictions about the maximum impact of species that could be revisited in the future. It would also be interesting to revisit risk management scores in the case of species where eradication attempts are made in the future, to assess how effective, practical, costly, impactful and acceptable they have been. This could be done quantitatively in some cases (e.g. for cost) and qualitatively in others (e.g. acceptability), perhaps by interviewing those involved in the eradication.

While expert judgement provides a useful means of conducting analysis and supporting decisions where data are lacking, it does not replace empirical evidence. As Colson and Cooke (2018) state "expert judgment should not provide the final word on any issue; rather, it should guide future data collection, modelling, and analysis related to the topic." The scores and priorities identified by this study should therefore be used to guide empirical or experimental research to test the judgements made by experts. For example, a range of options are available for conducting studies into the impact of species at various levels of ecological organisation (e.g. Roy et al., 2012, Dick et al., 2013, Tanner and Gange, 2013, Cameron et al., 2016, Mathers et al., 2016, Lavoie, 2017). For the risk management scores, trials could be carried out to explore the effectiveness, practicality, cost, impact and acceptability of eradication (e.g. Coutts and Sinner, 2004, Martins et al., 2006, Estevez et al., 2015, Robertson et al., 2015). This would not only help to validate judgements, but could be used adaptively to update and revise assessments as new information comes to light. Indeed, judgements made by experts should be reviewed and updated at regular intervals as new information comes to light.

6.5 Trends and patterns in impact and management feasibility

A major aim of biological invasions research has been to predict which non-native species become invasive (Lockwood et al., 2013). Despite a long period of 'scientific drought' in this area, Cassey et al. (2018b) highlight the considerable progress that has been made in recent decades, particularly as a result of work on the unified framework for invasion biology (Blackburn et al., 2011).

However, despite advances in understanding the processes behind the introduction and establishment of non-native species (e.g. Cassey et al., 2018a), the ability to predict which species will exert strong negative impacts remains weak (Ricciardi et al., 2013). Many studies have explored the use of species traits (e.g. fecundity, body size, leaf area, height) to predict impact (e.g. Keller et al., 2007, van Kleunen et al., 2010, Gallagher et al., 2015), with varying success for some taxa and situations (Ricciardi et al., 2013). This study (Chapter 2) explored whether a broader range of variables (taxa, environmental group, functional group and native origin) correlated with established non-native species that were identified as invasive (i.e. that caused more than minimal impact). All were found to be important predictors of impact, with aquatic species, vertebrates and species not native to Europe more likely to cause impacts once established.

In addition to attempting to predict the invasiveness of non-native species, it is also useful to consider the means by which management feasibility could be predicted. This has been considered in relation to specific taxa and relating these to variables such as area occupied, number of populations and fecundity (e.g. Panetta and Timmins, 2004, Drolet et al., 2015, Robertson et al., 2017); however, rarely across taxa and environment. This study demonstrates that a species' environmental group, the area it occupies and number of discrete populations in which it occurs could provide a useful means of predicting eradication feasibility (Chapter 5). For example, eradication was more likely to be feasible for terrestrial species at most scales, whereas for freshwater species there was a relatively small window of feasibility (with eradication more feasible in lentic than lotic waters) and for marine species it was largely unfeasible at all scales. While intuitive, exploring and refining these differences in management feasibility should help inform management decisions, such as when and how to deploy early detection and rapid response. Indeed, methods developed by this study are currently being used to refine our understanding on the scale at which vertebrate eradication may be feasible (Robertson et al., *in prep-b*). It should be noted that relatively few marine species (n = 23) were included in the risk management and prioritisation components of this study (Chapters 4 and 5), compared freshwater (n=39) and terrestrial (n=74) species. While this does not affect the analysis presented, if looking to generalise further it would be useful to increase the number of marine species studies.

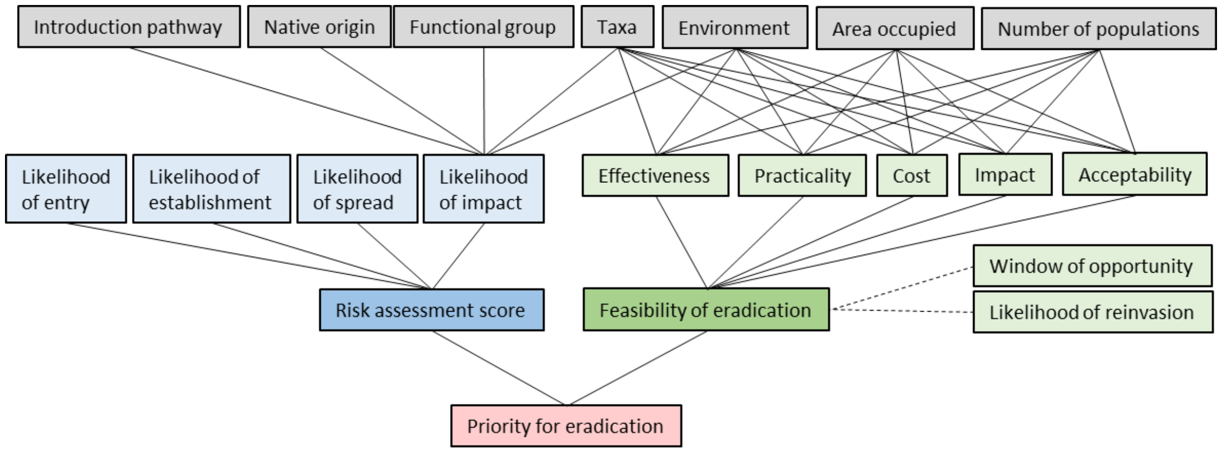


Figure 6.3 Theoretical Bayesian network illustrating the relationship between species variables and prioritisation components.

Given the correlations identified here between species variables and both the impact of established non-native species and the feasibility of their management, it may be useful to combine these within a Bayesian Network (BN). This could be used to support decision-making, as is common in other areas of conservation (Bower et al., 2018). For example, a BN could be created to replicate much of the proposed prioritisation framework (Fig 6.2). This could be used to consider the extent to which species data (e.g. taxonomic information, environmental group, native origin, area occupied, number of populations) might inform aspects of risk assessment (by predicting the likelihood of a species causing serious impacts) and risk management (by predicting the feasibility of eradication). This study gathered only limited data that could be used to explore the use of a BN for this purpose, which is illustrated in Figs 6.3 and 6.4; however, further development to add data and test the potential predictive power of this approach could be fruitful.

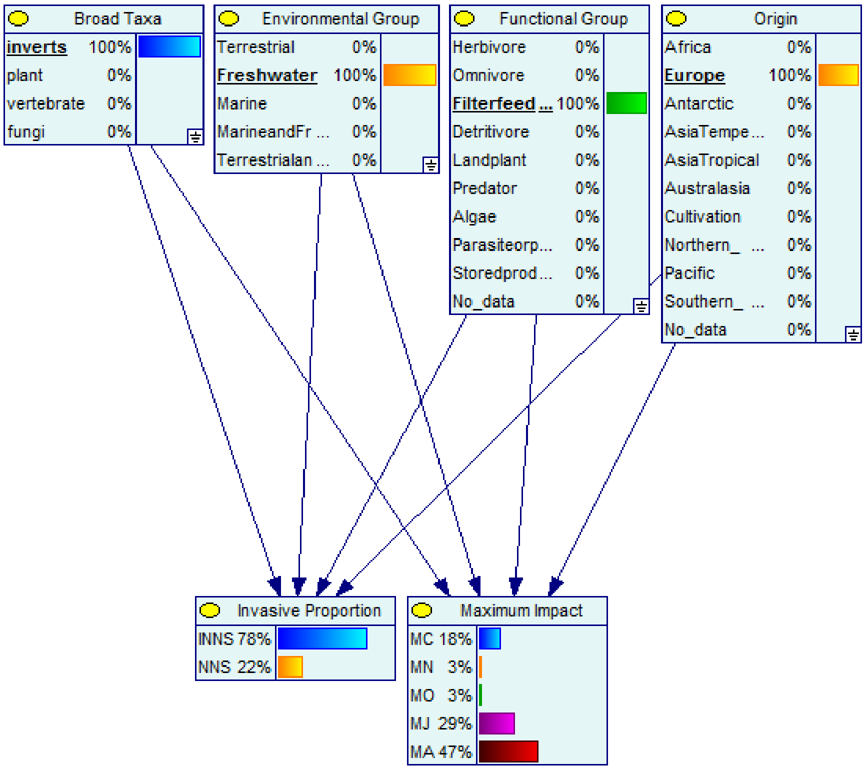


Figure 6.4 Bayesian network applied to established non-native species in GB to explore the relationship between species traits and impact. Example shown indicates the probability of a freshwater invertebrate filter feeder with European origins being invasive in GB (invasive proportion box) and the distribution of its possible impacts (provided by Maximum Impact box). Created using GeNIe 2.2 (a graphical user interface to the SMILE Engine which allows for interactive model building and learning). Explanatory and dependent variables were set as chance nodes, with user defined arcs (connections between nodes). Node properties were user defined, with parameters learned from the data (parameter initialisation was set to uniformise).

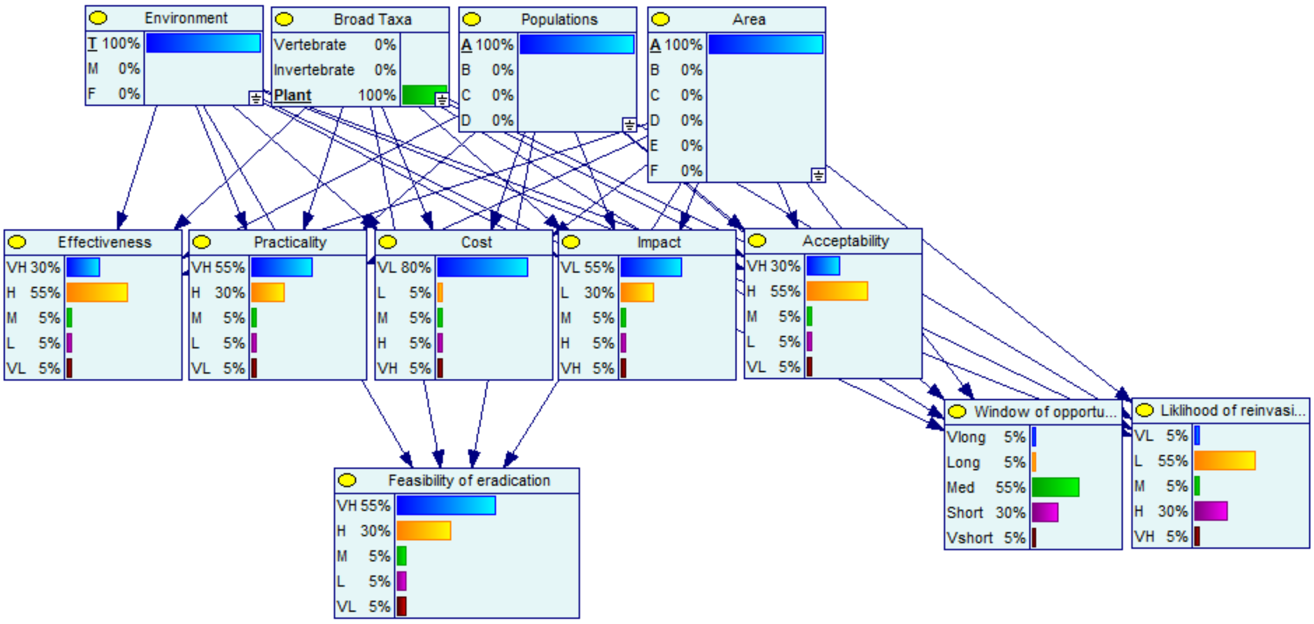


Figure 6.5 Bayesian network applied to horizon scanning species in the EU to explore the relationship between environment, taxa, extent and feasibility of eradication. Example shown is for a terrestrial plant established in <1ha and 1-3 populations. The predicted values for the seven (five + two) risk management components are shown as well as the probability of the overall feasibility of eradication. Created using GeNIe 2.2. All variables were set as chance nodes, with user defined arcs. Node properties were user defined (e.g. five point scale for risk management components), with parameters learned from the data (parameter initialisation was set to uniformise).

6.6 Scenarios for reducing future impact

The number of established non-native species in GB has increased rapidly over the past two hundred years, in line with global trends (Roy et al 2014; Chapter 2). Despite this increase in the numbers of non-native species, it appears that the rate of establishment of invasive non-native species (i.e. cause serious impacts) has decreased in recent decades (Chapter 2). This pattern was initially identified by Roy et al. (2014c) based on limited impact data and is confirmed by the more comprehensive impact assessment carried out as part of this study (Chapter 2). The most likely explanation for the decrease in the rate of invasive non-native species is that there is a lag in the detection of species impacts, particularly for terrestrial plants, which is in line with the concept of invasion debt (Essl et al., 2011a). It suggests that there are likely to be established species in GB that are currently considered benign, but that will cause more severe impacts in future. Paradoxically, these results also showed that there has been an increase in the proportion of high impact groups (i.e. aquatic species and those from non-European origins) establishing over time in GB, the effect of which on the overall trend in impact appears to have been masked by lag. It would be useful to explore this further, perhaps by modelling trends using only those species likely to be less affected by lag or by compensating for known lag times.

This study identified trends in the number and impact of non-native species in GB up to the beginning of the 21st Century. However, an important extension of this work would be to consider whether it is possible to model these trends into the future. The purpose of prioritising management is to reduce future impacts and so being able to make such predictions would provide a useful baseline against which to measure management success. If it were possible to model future impact, it may also be possible to model the cost-benefit of different management scenarios. Indeed, this could be one way to help prioritise resources between prevention, eradication and long term management objectives (discussed above). Attempting to model future numbers, impacts and management scenarios is likely to be challenging. While current trends can be extrapolated, the effect of lag would need to be taken into account. There is no certainty that past trends, for example in the role of introduction pathways, would continue beyond the near future and so such models would need to attempt to take into account for the complex variables that may affect pathways (such as changing markets, changing demographics and technological advancement).

6.7 Recommendations for the management of pathways and species

This analysis can be used by decision makers to inform pathway management priorities in GB. Some pathways are clear priorities given large impacts with high levels of confidence, such as hull fouling, horticultural escapes and contaminants of ornamental plants. The substantial increase in the impact of hull fouling and contaminants of ornamental plants since 1950 suggests these may be particularly important. Other pathways (e.g. those ranked 4-7) have a wide range of possible impacts that could mean they are low priorities; however, based on the intermediate and low confidence thresholds it is recommended that these be considered the next highest priorities for management. Table 3.5 can be used to tailor potential pathway management responses. For example, hull fouling species with native origins in continental Europe have had a disproportionately high impact compared to species introduced with other native origins. It would therefore be appropriate to particularly focus the management of this pathway on vessels traveling to GB from continental Europe. Similarly, inspections to reduce the risk of contaminants of ornamental plants may wish to focus on imports from Australia and New Zealand, which are likewise associated with disproportionately high impact.

Management priorities differed by environment. In the terrestrial environment horticultural escapes and contaminants of ornamental plants should be targeted as priority pathways. Horticultural escapes were exclusively associated with invasive plant species while contaminants of ornamental plants were almost entirely invertebrates, though also includes some plants. In the freshwater environment a wide range of pathways should be targeted, including hull fouling, ballast water and angling pathways (all of which were exclusively associated with invasive invertebrates), as well as horticultural escapes. In the marine environment, particular pathway priorities included hull fouling, contaminants of aquaculture animals and ballast water.

Decision makers should be aware of the range of confidence scores associated with these results. While there is considerable confidence in the top three pathways, the following four pathways have lower levels of confidence associated with them. Decision makers must consider whether to manage these pathways on the basis that they are likely to be priorities, but with some uncertainty, or consider methods for reducing uncertainty (for example, by commissioning more detailed research into the species specifically associated with these pathways). There were particularly low levels of confidence in pathways that introduced freshwater invertebrates (in most cases these were associated with a range of possible pathways, including angling, hull fouling and ballast water). It would therefore be useful to consider methods for more precisely identifying these. A possible method could be to randomly sample vectors associated with these pathways (e.g. boats, fishing nets, etc) to ascertain whether they contain invasive stowaways. Another approach could be to determine where species introduced in the past were first recorded and attempt to correlate this with the presence or absence of activities associated with each pathway (boating lakes, angling lakes, etc).

This study also indicates priorities for species management in both GB and the EU. It is recommended that the species identified as priorities in Chapters 4 and 5 be considered for management in GB and the EU respectively. These are divided into different priorities for: (i) prevention, (ii) early detection and eradication in the case of future invasion, (iii) eradication from areas where currently established, and (iv) long-term management. The separation into these management objectives is based on evaluations of risk and eradication feasibility (as per the framework in Table 6.1). The methods used here are designed particularly to identify species for early detection and eradication and eradication from areas where currently established. For prevention and long term management priorities it is recommended that consideration is given to further assessing the feasibility of these objectives. In addition to identifying priority species for management, the results of this study indicate factors to consider should decision makers decide to take action. For example, the detailed risk management scores can help indicate where there may be issues to take into consideration and potentially mitigate, such as potential negative environmental or socio-economic impacts of management, or practical barriers associated with land access or legislation. Such information could be used to help inform a management approach.

This study provides an effective means of rapidly identifying management priorities for large numbers of species based not only on risk assessment but also management feasibility. However, it does not provide all of the information required to develop a specific management programme for each species. It is therefore recommended that further investigation is carried out into priority species to inform a management programme. This could include undertaking specific cost-benefit or cost-effectiveness assessments (e.g. Coutts and Sinner, 2004, Reyns et al., 2018). More broadly, this should include developing a costed plan of action detailing the specific locations of all populations, the deployment of methods in the field and methods for overcoming any barriers or potential adverse consequences. These would then be evaluated before committing to further management.

Patterns in the feasibility of eradication identified by this research have important implications for decision makers. There were considerable differences found in the feasibility of eradication depending on the environment in which the species was established. The implications for decision makers are that early detection and rapid eradication is particularly important in freshwater environments, where feasibility of eradication is generally high at small scales, but can quickly drop if the species is allowed to establish over a wider area. This is less an issue in the terrestrial environment, where decision makers may have considerably more time (in some cases many years) before a species spreads beyond the point that eradication feasibility drops. Results suggest that eradication in the marine environment is generally unlikely to be feasible, even at small scales. This does not preclude decision makers attempting eradication; however, such attempts are only likely to succeed in situations where conditions are particularly favourable (e.g. Bax et al., 2002, Wotton et al., 2004). Of course, this does not mean that marine species should not be managed. Indeed, it emphasises the importance of managing pathways to prevent the introduction of marine invasive non-native species (as discussed in Chapter 3).

Table 6.1. Identifying management objectives (and priority) for invasive non-native species that are either established or not yet established in a given area. Risk scores, eradication feasibility scores and establishment status are combined to determine management objectives (EDRR = early detection and rapid response, LTM = long term management). Relative priority for each objective is given (in brackets).

		Simplified management priority	
Risk score	**Eradication feasibility**	**Not established**	**Established**
High	Low	Prevention (high)	LTM (high)
	High	EDRR (high)	Eradicate (high)
Low	Low	Prevention (lower)	LTM (lower)
	High	EDRR (lower)	Eradicate (lower)

6.8 Conclusion

The study of invasive non-native species is not only of considerable academic interest, but is also essential to combat the impacts of these species worldwide. There is therefore substantial benefit to be gained from building close links between the research community and those deciding on and implementing management decisions. A focal point for both communities is in the prioritisation of species and pathways for management; however, considerably more work and closer links are required. This study highlights the need for more research into the systematic evaluation of management feasibility and its incorporation into prioritisation. It demonstrates a way in which such information could be combined with existing tools to provide a comprehensive prioritisation approach to support policy makers faced with, at times, overwhelming complexity. Such methods are urgently required at local, national and global scales if we are to slow the threat from these species and the catastrophic effects they are having on global biodiversity.

Appendices

Appendix A. List of experts involved in Great Britain impact scoring workshop

Expert Name	Group based on area of expertise
Alison Dunn	Freshwater animals
Colin Bean	Freshwater animals
David Aldridge (lead)	Freshwater animals
Ian Winfield (lead)	Freshwater animals
Paul Stebbing	Freshwater animals
Rob Britton	Freshwater animals
Christine Maggs	Marine
Elizabeth Cook	Marine
Esther Hughes	Marine
Francis Bunker	Marine
Jack Sewell (lead)	Marine
John Bishop	Marine
Juliet Brodie	Marine
Roger Herbert	Marine
Fred Rumsey	Plants
Katharina Dehnem-Schmutz	Plants
Kevin Walker (lead)	Plants
Oliver Prescott	Plants
Pablo Gonzalez-Moreno	Plants
Pete Stroh	Plants
Trevor Dines	Plants
Alan Stewart (lead)	Terrestrial invertebrates
Chris Raper	Terrestrial invertebrates
Dick Shaw (lead)	Terrestrial invertebrates
Karsten Schonrogge	Terrestrial invertebrates
Martin Harvey	Terrestrial invertebrates
Aileen Mill	Vertebrates
Dave Parrot	Vertebrates
David Noble (lead)	Vertebrates
Jim Foster	Vertebrates
John Marchant	Vertebrates
John Wilkinson	Vertebrates
Kirsty Park	Vertebrates
Pete Robertson	Vertebrates
Robbie McDonald	Vertebrates
Tim Blackburn	Vertebrates

Appendix B. Scoring the biodiversity impact of invasive non-native species in GB: instructions for assessors

1. Select your species, add your name and the date

Use the pull down menus provided.

2. Score the current impact and confidence

Current impact is defined as the impact to date based on the species current distribution in GB. Use the predefined categories (minimal concern, minor, moderate, major, massive) to score your response. Definitions are provided for each category [refer to Hawkins et al. (2015)] – please make sure to use these – a useful decision diagram to help with scoring is also provided [refer to Hawkins et al. (2015)]. Indicate how confident you are in your response scores using the pull down menu provided. Guidance on scoring confidence is provided [refer to Hawkins et al. (2015)].

3. Score maximum impact and confidence

Maximum potential impact is defined as the impact the species would be expected to have in GB if it were established in all parts that are suitable (i.e. based on current biotic and abiotic conditions). Response and confidence scores should be determined in the same way as for current impact.

4. Provide a supporting comment

A supporting comment to justify the current and max impact response scores is required. You should cite relevant literature you are aware of to support your justification; however you are not expected to undertake a full search of new literature. Peer reviewed literature should be used if possible, but if not other forms of evidence is acceptable (e.g. grey literature, field observations). Where no evidence is available, expert judgement should be used to determine the scores and a reasoned argument provided as justification. Use short hand for references, e.g. Roy et al (2014), placing the full reference in the 'references' box below (full references can be provided in any format).

5. Type of evidence

Use these tick boxes to indicate the type of evidence used to determine the response scores. A space is provided next to each evidence type for you to list relevant references that you are aware of. Use short hand for the references here, e.g. Roy et al (2014), placing the full reference in the 'references' box below.

6. Score the impact type and mechanism

Use this section to score the type of impact (i.e. what is affected) along with the mechanism for that impact (i.e. how the impact comes about). These scores should be based on the maximum potential impact (not the current impact). As a guide, a species of conservation concern is generally defined as one that either has an international or domestic legal designation* or that is listed as a 'biodiversity list species' by JNCC (i.e. listed on NERC section 41 and 42, Scottish Biodiversity List or Northern Ireland Priority Species List). For reference, a full list of these species is provided as a separate spreadsheet. Habitats of conservation concern follows http://jncc.defra.gov.uk/page-5706, a full list is provided at Annex 3. The mechanism of impact is likely to be the same for habitats as it is for species, but please discuss with your group leads if you are uncertain. Note, some mechanisms may not appear relevant to habitats (e.g. predation, hybridisation), in which do not check them. A species or habitat NOT of conservation concern is one that is not covered in either of the definitions above. The different mechanisms of impact are defined [refer to Hawkins et al. (2015)].

*Bern Convention, Birds Directive, Convention on Migratory Species, OSPAR, Habitats Directive; The Wildlife and Countryside Act 1981, The Wildlife (Northern Ireland) Order 1985, The Conservation of Habitats and Species Regulations 2010, The Conservation (Nature Habitats) Regulations (NI) 1995, Protection of Badgers Act.

7. List the species and habitats of conservation concern that are affected

If species or habitats of conservation concern are affected please try to list them here. A pull down list of species of conservation concern is provided (multiple species can be selected); however you can also write the name of any species into the text box provided if this is easier (or a name is missing from the list). A separate copy of the list of species of conservation concern is also provided as a separate spreadsheet. For habitats of conservation concern please select from the list provided (Table B1), more than one habitat can be selected and broad habitats can be selected if desirable. There is also a free text box which can be used to list habitats that are not listed or provide more detail if necessary.

8. Socio-economic impacts

To double check the information already in the NNSIP system, use this box to flag species that have negative socio-economic consequences.

Table B1. Habitats of conservation importance

UK BAP broad habitat	UK BAP priority habitat
Rivers and Streams	Rivers
Standing Open Waters and Canals	Oligotrophic and Dystrophic Lakes
	Ponds
	Mesotrophic Lakes
	Eutrophic Standing Waters
	Aquifer Fed Naturally Fluctuating Water Bodies
Arable and Horticultural	Arable Field Margins
Boundary and Linear Features	Hedgerows
Broadleaved, Mixed and Yew Woodland	Traditional Orchards
	Wood-Pasture and Parkland
	Upland Oakwood
	Lowland Beech and Yew Woodland
	Upland Mixed Ashwoods
	Wet Woodland
	Lowland Mixed Deciduous Woodland
	Upland Birchwoods
Coniferous Woodland	Native Pine Woodlands
Acid Grassland	Lowland Dry Acid Grassland
Calcareous Grassland	Lowland Calcareous Grassland
	Upland Calcareous Grassland
Neutral Grassland	Lowland Meadows
	Upland Hay Meadows
Improved Grassland	Coastal and Floodplain Grazing Marsh
Dwarf Shrub Heath	Lowland Heathland
	Upland Heathland
Fen, Marsh and Swamp	Upland Flushes, Fens and Swamps
	Purple Moor Grass and Rush Pastures
	Lowland Fens
	Reedbeds
Bogs	Lowland Raised Bog
	Blanket Bog

Montane Habitats	Mountain Heaths and Willow Scrub
Inland Rock	Inland Rock Outcrop and Scree Habitats
	Calaminarian Grasslands
	Open Mosaic Habitats on Previously Developed Land
	Limestone Pavements
Supralittoral Rock	Maritime Cliff and Slopes
Supralittoral Sediment	Coastal Vegetated Shingle
	Machair
	Coastal Sand Dunes
Littoral Rock	Intertidal Chalk
	Intertidal Underboulder Communities
	Sabellaria alveolata reefs
Littoral Sediment	Coastal Saltmarsh
	Intertidal Mudflats
	Seagrass Beds
	Sheltered Muddy Gravels
	Peat and Clay Exposures with Piddocks
Sublittoral Rock	Subtidal Chalk
	Tide-swept Channels
	Fragile Sponge and Anthozoan Communities on Subtidal Rocky Habitats
	Esuarine Rocky Habitats
	Seamount Communities
	Carbonate Mounds
	Cold-Water Coral Reefs
	Deep-Sea Sponge Communities
	Sabellaria spinulosa Reefs
Sublittoral Sediment	Subtidal Sands and Gravels
	Horse Mussel Beds
	Mud Habitats in Deep Water
	File Shell Beds
	Maerl Beds

	Serpulid Reefs
	Blue Mussel Beds on Sediment
	Saline Lagoons

Appendix C. Automatic rules used to re-code NNSIP pathways to CBD classification

Automatic rules were coded in R. The following indicate which NNSIP pathways and other NNSIP criteria (i.e. NNSIP data on 'pathway method' and 'taxa') were used to determine a CBD category (based on codes in Table 3.1

NNSIP "Landscape" = "R_AES"

NNSIP "Ornamental" & NNSIP Method "Release" = "R_AES"

NNSIP "Hunting / fishing" & NNSIP Taxa !="fish" = "R_HUNT"

NNSIP "Hunting / fishing" & NNSIP Taxa "fish" = "R_FISH"

NNSIP "Agriculture" & NNSIP Method "Escape" = "E_AGRI"

NNSIP "Agriculture" & NNSIP Method "Accidental" = "C_AGRI"

NNSIP "Agriculture" & NNSIP Method "Release" = "R_OTR"

NNSIP "Medicinal" & NNSIP Method "Escape" = "E_AGRI_MED"

NNSIP "Medicinal" & NNSIP Method "Accidental" = "C_PLT_AGRI"

NNSIP "Aquaculture" & NNSIP Method "Escape" = "E_AQC"

NNSIP "Aquaculture" & NNSIP Method "Accidental" = "C_AQC"

NNSIP "Aquaculture" & NNSIP Method "Release" = "R_FHRY"

NNSIP "Ornamental" & NNSIP Method "Escape" & NNSIP Taxa "plant" = "E_HORT"

NNSIP "Ornamental" & NNSIP Method "Escape" & NNSIP Taxa "inverts" = "E_PET"

NNSIP "Ornamental" & NNSIP Method "Escape" & NNSIP Taxa "vertebrate" = "E_ORN_VRT"

NNSIP "Ornamental" & NNSIP Method "Accidental" = "C_PLT_ORN"

NNSIP "Forestry" & NNSIP Method "Accidental" = "C_FOR"

NNSIP “Forestry" & NNSIP Method “Escape" = "E_FOR”

NNSIP “Seed for agriculture" & NNSIP Method “Accidental" = "C_SEED”

NNSIP “Seed for ornamental" & NNSIP Method “Accidental" = "C_SEED”

NNSIP “Seed produce" & NNSIP Method “Accidental" = "C_SEED”

NNSIP “Seed for agriculture" & NNSIP Method “Escape" = "E_AGRI_SEED”

NNSIP “Seed for ornamental" & NNSIP Method “Escape" = "E_HORT_SEED”

NNSIP “Stowaway - land" = "S_LVEH”

NNSIP “Natural" = "U_NAT”

NNSIP “Aquaculture" & NNSIP Taxa "insect - moth" = "C_PLT_HORT”XX

Appendix D. The Non-native Risk Management scheme

This guidance is the current version of the NNRM scheme (as of 2019) which has been updated since its original use in GB (Chapter 4) as a result of the EU application (Chapter 5).

Guidance is provided, in full, for the use of the NNRM scheme for assessing the feasibility of eradication, including instructions to assessors and a template for recording scores.

<u>Guidance for the use of the Non-native Risk Management (eradication) Scheme (NNRM)</u>

1. Background

This guidance is provided to assess:

- non-native species already established in a defined risk management area, where options for eradication are being considered; and
- non-native species not yet established in a defined risk management area, where options for eradication following detection in the wild are being considered.

Aspects of risk management not related to eradication, i.e. prevention and long term management, are not dealt with here. The process for assessing risk management options is set out below and should be read in conjunction with the template at Fig 1. An example of a completed template is provided at Fig 2.

2. Preliminary sections

Define the risk management area. This can be any area at any scale, but must be clearly defined and understood from the outset of the assessment.

State the objective of the assessment. The objective is predefined as 'the eradication (defined as the complete removal of a species from a defined geographic area - *sensu* Genovesi 2000) of the target organism from the risk management area'.r

Define the target organism. The target organism can be any taxon but must be clearly defined.

Record the name(s) of assessors, date and version number of the assessment.

3. Assessment

Step 1 - Define the Scenario

The scenario should describe the extent of the species either based on its current distribution (if already established) or based on its most likely distribution at the point it is discovered (for species not already established).

For species that are already established in the wild - the scenario should be the current situation, i.e. the current level of establishment (estimated if necessary / existing information is weak).

For species not yet established in the wild - the scenario should be the most likely situation at the point the species is detected in the wild (based on current surveillance).

It is important to carefully define the scenario as it is fundamental to the rest of the assessment. While brief, the scenario should be sufficiently clear to facilitate subsequent scoring. Where multiple scenarios are feasible, the most likely scenario should be assessed. A lack of certainty should not prevent a scenario being defined; if there is doubt define the best scenario possible and make clear any uncertainty in the comments.

In defining the scenario you should consider (but only include if relevant):

- How widespread the species is (or will be at the point of detection) in the risk management area.
- The types of habitats / environments in which the species is (or will be) present.
- How many spatially distinct populations there are (or will be).
- What the size of the total population is (or will be).

A code should be provided for the scenario based on the number of discrete populations and total combined area of those populations using the table at Annex 3.

Step 2 – Define the eradication Strategy

The assessor should briefly describe a realistic strategy that could be used to eradicate the species entirely from the risk management area. This could include multiple methods (e.g. trapping, chemical use and mechanical removal); it should also include other elements, such as surveys, logistics and monitoring, if they are required in order to achieve eradication.

The strategy that is most likely to be successful should be described, avoiding being too conservative (i.e. no eradication possible despite techniques being available) or unrealistic (i.e. cost / damage caused vastly outweighs potential benefits). If no realistic strategy can be envisaged then it can still be useful to quickly assess extreme strategies.

The rest of the assessment (i.e. effectiveness, cost, etc.) will be based on the eradication strategy described here.

Step 3 – Assessing the eradication strategy

The eradication strategy should be assessed using the criteria defined under the headings below (3a to 3d).

The response score is a 5 point scale from 1-5 (Table 1). In all cases 1 is the least favourable and 5 the most. For example, a very effective eradication strategy scores 5, a very ineffective strategy scores 1; whereas a very inexpensive strategy (i.e. the cost favours taking action) scores 5, a very expensive one scores 1.

Table 1. Assessment criteria for response scores.

Criteria	**Response Score**				
	1	**2**	**3**	**4**	**5**
Effectiveness	Very ineffective	Ineffective	Moderate effectiveness	Effective	Very effective
Practicality	Very impractical	Impractical	Moderate practicality	Practical	Very practical
Cost	>£10M	£1-10M	£200k-1M	£50-200k	<£50k
Negative impact	Massive	Major	Moderate	Minor	Minimal
Acceptability	Very unacceptable	Unacceptable	Moderate acceptability	Acceptable	Very acceptable
Window of opportunity	< 2 months	2 months - 1 year	1 – 3 years	4-10 years	>10 years
Likelihood of reinvasion	Very likely	Likely	Moderate likelihood	Unlikely	Very unlikely
Conclusion (overall feasibility of eradication)	Very low	Low	Medium	High	Very high

A confidence rating should be provided for every response score. Confidence is recorded on a 3 point scale: 1 (low), 2 (medium), 3 (high). Even where evidence is lacking, assessors should make best judgements and use the confidence rating score to reflect uncertainty.

Step 3a - Effectiveness

This part of the assessment scores how effective the defined eradication strategy would be regardless of other issues, such as the practicality of deploying methods, costs, acceptability of methods, etc. which are taken into account elsewhere. For example, the eradication strategy for a non-native fish in a river could be to flood it with the piscicide rotenone – this would likely score 'very effective' despite low scores associated with practicality, impact and acceptability.

Points to consider:

- How effective has this approach proven to be in the past or in an analogous situation?
- How effective is the approach despite the biology / behaviour of the target organism?

Scoring scale:

- 5 – very effective
- 4 – effective
- 3 – moderate effectiveness
- 2 – infective
- 1 – very infective

Step 3b - Practicality

How practical is it to deploy the described strategy? In particular, consider barriers that might prevent the use of the strategy such as issues gaining access to relevant areas, obtaining appropriate equipment, skilled staff, chemicals, etc. If there are any legal barriers to undertaking the work these should be assessed here.

Points to consider:

- How available are the methods in the risk management area?
- How accessible are the areas required to deploy the eradication strategy?
- How easy would it be to obtain relevant licences or other approvals / permissions (e.g. access permission) to undertake the approach?
- How easy would it be to overcome legal barriers?
- How safe are the methods used in this approach (are there health and safety barriers)?

Scoring scale:

- 5 – very practical
- 4 – practical
- 3 – moderate practicality
- 2 – impractical
- 1 – very impractical

Step 3c - Cost

Cost relates to the total direct cost of eradicating the species from the risk management area using the defined eradication strategy. Total cost includes the cost of staff, resources, materials, etc. over the entire time period involved in the eradication and any required post eradication surveillance and follow-up. Note indirect costs (e.g. loss of business) are considered an impact and not recorded here.

In your comment, indicate the period over which costs would be occurred (i.e. number of years) and, if possible, indicate whether the cost would be evenly spread, frontloaded or back loaded.

Scoring scale:

- 5 - minimal - <£50k
- 4 - minor - £50-200k
- 3 - moderate - £200k-1M
- 2 - major - £1-10M
- 1 - massive - > £10M

Step 3d - Impact

Impact relates to the impact of the eradication strategy itself. It is important to note that any indirect economic impacts (i.e. economic consequences of the eradication strategy rather than the cost of the strategy itself) are recorded here and not under 'cost'.

Points to consider:

- How significant is the environmental harm caused by this approach?
- How significant is the economic harm caused by this approach?

  Examples of economic harm might include: reduction in the ability to trade or do business as a result of the management method; loss of earnings; reduction in tourism; reduction in house prices; etc.

- How significant is the social harm, including to human health, caused by this approach?

  Examples of social harm might be a reduction in a person's use or enjoyment (e.g. preventing them walking in a woodland or fishing in a river), disruptions of communities, etc.

Scoring scale:

- 5 - minimal
- 4 - minor
- 3 - moderate

- 2 - major
- 1 - massive

Step 3e - Acceptability

Acceptability relates to significant issues that could arise as a result of disapproval or resistance from individuals, groups or sectors. This does not include regulatory or legislative barriers which are considered under practicality.

- How acceptable is the approach likely to be based on environmental / animal welfare grounds?

  Note this question relates to likely criticism / resistance that the approach would meet based on environmental / animal welfare grounds.

- How acceptable is the approach likely to be to the general public?
- How acceptable is the approach likely to be to other stakeholders?

Scoring scale:

- 5 – very acceptable
- 4 – acceptable
- 3 – moderate acceptability
- 2 – unacceptable
- 1 – very unacceptable

Step 4 – Assessing the window of opportunity

The window of opportunity relates to how quickly the species will spread beyond the point that eradication, using the defined strategy, would be effective. It is linked to the mechanism and rate of spread, which is considered during the risk assessment.

Scoring scale:

- 5 - very long (10+ years)
- 4 - long (4-10 years)
- 3 - moderate (1 – 3 years)
- 2 - short (2 months - 1 year)

- 1 - very short (< 2 months)

Step 5 – Assessing the likelihood of re-invasion

Assuming the eradication is successful, i.e. there are no wild populations of the species left, how likely is it that re-invasion will occur? Note that unless the eradication strategy has deliberately targeted populations in containment or otherwise not in the wild (i.e. in gardens, zoos, etc.) introduction from these should be considered part of re-invasion.

Scoring scale:

- 5 – very unlikely
- 4 – unlikely
- 3 – moderate likelihood
- 2 – likely
- 1 – very likely

Step 6 – Determine the overall feasibility of eradication

This is the conclusion of the assessment. A score should be provided for the overall feasibility of eradication taking into account all other factors (i.e. 3a – 5). Assessors should provide a score they judge to be appropriate, taking other scores into account (but note the overall score is not necessarily the mean of other scores).

Scoring scale:

- 5 – very high
- 4 – high
- 3 – medium
- 2 – low
- 1 – very low

Figure 1. Template for Non-native Risk Management Assessment

Risk management area:	
Objective:	
Organism name:	
Assessor name(s):	
Date / version:	

Title	Response	Confidence	Justification
1. Define the scenario	*Input scenario and scenario code*		
2. Define the eradication strategy	*Input eradication strategy*		
3a. How effective is the strategy?	5 - V EFFECTIVE 4 – EFFECTIVE 3 – MODERATE 2 – INEFFECTIVE 1 - V INEFFECTIVE	3 – HIGH 2 – MED 1 – LOW	
3b. How practical is the strategy?	5 - V PRACTICAL 4 – PRACTICAL 3 – MODERATE 2 – IMPRACTICAL 1 – V IMPRACTICAL	3 – HIGH 2 – MED 1 – LOW	
3c. How expensive is the strategy?	5 (<£50K) 4 (£50-200K) 3 (£200K-1M) 2 (1-10M) 1 (> £10M)	3 – HIGH 2 – MED 1 – LOW	
3d. How much negative impact would the strategy have?	5 – MINIMAL 4 – MINOR 3 – MODERATE 2 – MAJOR 1 – MASSIVE	3 – HIGH 2 – MED 1 – LOW	
3e. How acceptable is the strategy?	5 - V ACCEPTABLE 4 – ACCEPTABLE 3 – MODERATE 2 – UNACCEPTABLE 1 - V UNACCEPTABLE	3 – HIGH 2 – MED 1 – LOW	
4. What is the window of opportunity for implementing the strategy?	5 (10+ YRS) 4 (4-10 YRS) 3 (1 – 3 YRS) 2 (2 MTHS - 1 YR) 1 (< 2 MTHS)	3 – HIGH 2 – MED 1 – LOW	
5. What is the likelihood of reinvasion?	5 – V UNLIKELY 4 – UNLIKELY 3 – MODERATE 2 – LIKELY 1 – V LIKELY	3 – HIGH 2 – MED 1 – LOW	
6. Conclusion (overall feasibility of eradication)	5 – V HIGH 4 – HIGH 3 – MEDIUM 2 – LOW 1 – V LOW	3 – HIGH 2 – MED 1 – LOW	

Figure 2. Example of a completed template for *Trichosurus vulpecula* (Brushtail Possum) eradication from the EU

Risk management area:	European Union (excluding outermost territories)
Objective:	Complete eradication
Organism name:	*Trichosurus vulpecula* (Brushtail Possum)
Assessor name(s):	[unspecified in example]
Date / version:	[unspecified in example]

Title	**Response**	**Confidence**	**Justification**
1. Define the scenario	*Not currently established in the risk management area (RMA). At the point of the detection, the most likely scenario is a single population in broadleaved woodland spread over 1-10km^2 and comprising 10-50 individuals (Scenario Code A2). This could occur in any of the temperate regions of the RMA.*		
2. Define the eradication strategy	*The strategy to eradicate this species would be trapping. Initial surveillance would be carried out in the 10km^2 area and a surrounding 2km buffer zone, including the use of camera traps / trained dogs / hair traps. Trapping would include live cage traps and kill traps (some of which may be at height).*		
3a. How effective is the strategy?	4 – EFFECTIVE	3 – HIGH	Not as effective as air dropping poison bait (as used in NZ); but still likely to be effective.
3b. How practical is the strategy?	5 - V PRACTICAL	3 – HIGH	Expect that population would be in accessible habitat (i.e. broadleaved woodland).
3c. How expensive is the strategy?	4 (€50-200K)	2 – MED	Cost estimate is based on experience with mammal trapping in GB; but medium confidence (score could be moderate) because there may be a shortage of fully trained staff.
3d. How much negative impact would the strategy have?	5 – MINIMAL	3 – HIGH	Possibly some short term restrictions on use of woodland during trapping – but of negligible consequence.
3e. How acceptable is the strategy?	4 – ACCEPTABLE	2 – MED	The methods are tested and considered humane (and used elsewhere in the world). Opposition to lethal control by a small number of the public is possible and varies across the EU. In some areas this may decrease acceptability (e.g. to moderate), hence only medium confidence.
4. What is the window of opportunity for implementing the strategy?	3 (1 – 3 YRS)	3 – HIGH	Spread is likely to be slow and new populations are unlikely to form. As such, the level of response required is unlikely to change for a number of years.
5. What is the likelihood of reinvasion?	4 – UNLIKELY	2 – MED	Risk of entry already considered low; risk of reintroduction after eradication therefore considered unlikely. However,

			if eradication is required then consideration should be given to closing down any active pathways.
6. Conclusion (overall feasibility of eradication)	5 – V HIGH	3 – HIGH	Based on the scenario only a single (small) population would need to be eradicated. Experience from elsewhere suggests eradication is highly feasible.

Figure 3. Table for codifying the scenario based on number of discrete populations and total area

Identify one box in the table to indicate the likely number of sites containing the species and the combined area of these populations. Populations are considered discrete if they would be unlikely to recolonise from other areas after removal. The total area is that from which the species would need to be removed, i.e. for three populations of a species each covering 10ha and each 100km apart, the total area is 30ha, not 100km+.

		Total combined area of populations					
		<1ha	1-10ha	10ha-1km2	1-10km2	10-100km2	>100km2
Number of discrete populations	1-3	A1 1-3 discrete populations estimated covering a total area of <1ha	A2 1-3 discrete populations estimated covering a total area of 1-10ha	A3 1-3 discrete populations estimated covering a total area of 10ha-1km2	A4 1-3 discrete populations estimated covering a total area of 1-10km2	A5 1-3 discrete populations estimated covering an area of 10-100km2	A6 1-3 discrete populations estimated covering an area of >100km2
	4-10	B1 4-10 discrete populations estimated covering a total area of <1ha	B2 4-10 discrete populations estimated covering a total area of 1-10ha	B3 4-10 discrete populations estimated covering a total area of 10ha-1km2	B4 4-10 discrete populations estimated covering a total area of 1-10km2	B5 4-10 discrete populations estimated covering a total area of 10-100km2	B6 4-10 discrete populations estimated covering a total area of >100km2
	10-50	C1 10-50 discrete populations estimated covering a total area of <1ha	C2 10-50 discrete populations estimated covering a total area of 1-10ha	C3 10-50 discrete populations estimated covering a total area of 10ha-1km2	C4 10-50 discrete populations estimated covering a total area of 1-10km2	C5 10-50 discrete populations estimated covering a total area of 10-100km2	C6 10-50 discrete populations estimated covering a total area of >100km2
	+50	D1 50+ discrete populations estimated covering a total area of <1ha	D2 50+ discrete populations estimated covering a total area of 1-10ha	D3 50+ discrete populations estimated covering a total area of 10ha-1km2	D4 50+ discrete populations estimated covering a total area of 1-10km2	D5 50+ discrete populations estimated covering a total area of 10-100km2	D6 50+ discrete populations estimated covering a total area of >100km2

Appendix E. List of experts involved in Great Britain risk management workshop

Name	Organisation	Group based expertise
Matt Brazier	Environment Agency	Freshwater animals
Tristan Hatton-Ellis	Natural Resources Wales	Freshwater animals
Alice Hiley	Environment Agency	Freshwater animals
Jo Long	Scottish Environment Protection Agency	Freshwater animals
Craig MacAdam	Buglife	Freshwater animals
Trevor Renals	Environment Agency	Freshwater animals
Paul Stebbing	Cefas	Freshwater animals
Mathilde Bue	Institute of Biological, Env. and Rural Sciences	Marine
Maggie Hatton-Ellis	Natural Resources Wales	Marine
Jan Maclennan	Natural England	Marine
Eiona Rodgers	RSPCA	Marine
Jack Sewell	Marine Biological Association	Marine
Stan Whittaker	Scottish Natural Heritage	Marine
Gabe Wyn	Natural Resources Wales	Marine
Richard Bullock	Wildfowl and Wetlands Trust	Plants
Camilla Morrison-Bell	British Ecological Society	Plants
Jonathan Newman	Centre for Ecology and Hydrology	Plants
Robin Payne	Scottish Natural Heritage	Plants
Mark Spencer	Natural History Museum	Plants
Kevin Walker	Botanical Society for British Isles	Plants
Simon Baker	retired	Terrestrial animals
Sam Bishop	Defra	Terrestrial animals
Steve Campbell	Science and Advice for Scottish Agriculture	Terrestrial animals
Dominic Eyre	Defra	Terrestrial animals
Jim Foster	Amphibian and Reptile Conservation Trust	Terrestrial animals
John Mumford	Imperial College London	Terrestrial animals
David Parrot	Animal and Plant Health Agency	Terrestrial animals
Helen Roy	Centre for Ecology and Hydrology	Terrestrial animals
Mike Sutton-Croft	Animal and Plant Health Agency	Terrestrial animals
Alastair Ward	Animal and Plant Health Agency	Terrestrial animals
Hannah Freemann	Wildfowl and Wetlands Trust	Observer
Niall Moore	Animal and Plant Health Agency	Observer
Pete Robertson	Animal and Plant Health Agency	Observer

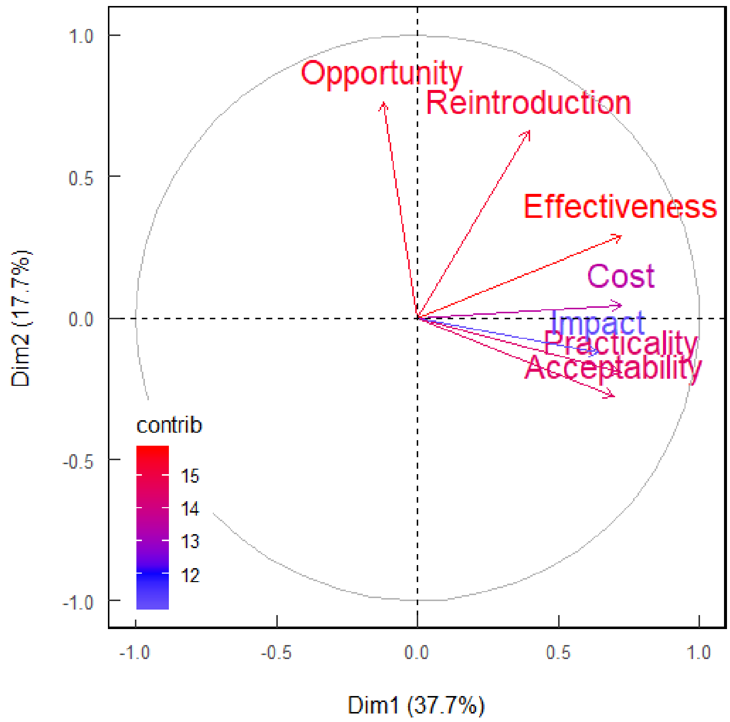
Appendix F. List of experts involved in European Union risk management workshop

Name	Organisation	Group
Elena Tricarico	University of Florence	Freshwater animals
Hugo Verreycken	Research Institute for Nature and Forest	Freshwater animals
Jamie Dick	Queens University Belfast	Freshwater animals
Joe Caffrey	INVAS Biosecurity	Freshwater animals
Eithne Davis	IT Sligo	Freshwater animals (observer)
Niel Coughlan	Queens University Belfast	Freshwater animals (observer)
Frances Lucy	IT Sligo	Freshwater animals
Gabe Wyn	Natural Resources Wales	Marine species
Francis Kerckhof	Belgian Institute of Natural Sciences	Marine species
Paul Stebbing	CEFAS	Marine species
Stuart Jenkins	Bangor Univeristy	Marine species
Olivier De Clerk	University of Ghent	Marine species
Stelios Katsanevakis	University of the Agean	Marine species
Johan van Valkenberg	Plant Protection Service Netherlands	Plants
Franz Essl	Environment agency Austria	Plants
Jonathan Newman	Centre for Ecology and Hydrology	Plants
Pablo Gonzalez-Moreno	CABI	Plants
Sonia Vanderhoeven	Belgian Biodiversity Partnership	Plants
Uwe Starfinger	Julius Kuhn Institute	Plants
Giuseppe Brundu	University of Sassari, Italy	Plants
Guillame Fried	ANSES (France)	Plants
Wolfgang Nentwig	University of Bern	Terrestrial invertebrates
Dick Shaw	CABI	Terrestrial invertebrates
Olivier Blight	Donana	Terrestrial invertebrates
Helen Roy	Centre for Ecology and Hydrology	Terrestrial invertebrates
Wolfgang Rabbitsch	Environment Agency Austria	Terrestrial invertebrates
Tim Adriens	Research Institute for Nature and Forest	Terrestrial vertebrates
Peter Robertson	Newcastle University	Terrestrial vertebrates
Frank Hysentruyt	Research Institute for Nature and Forest	Terrestrial vertebrates
Sandro Bertolino	Turin University	Terrestrial vertebrates
Sugoto Roy	IUCN	Terrestrial vertebrates
Dario Capizzi	Directorate Environment and Natural Systems, Italy	Terrestrial vertebrates
Jan Stuyck	Research Institute for Nature and Forest	Terrestrial vertebrates
Jim Casaer	Research Institute for Nature and Forest	Terrestrial vertebrates
Jess Ward	Newcastle University	Data support
Aileen Mill	Newcastle University	Data support
Olaf Booy	Newcastle University	Facilitator
Piero Genovesi	IUCN	Facilitator
Mike Sutton-Croft	Animal and Plant Health Agency	Observer

Niall Moore	Animal and Plant Health Agency	Observer
Maurits Vandegehuchte	Agentschap natuur &Bos	Observer
Myriam Dumotier	EU policy	Observer
Etienne Branquart	Belgian Biodiversity Partnership	Observer

Appendix G. Factor plot of risk management components applied to new and emerging invasive non-native species in the EU.

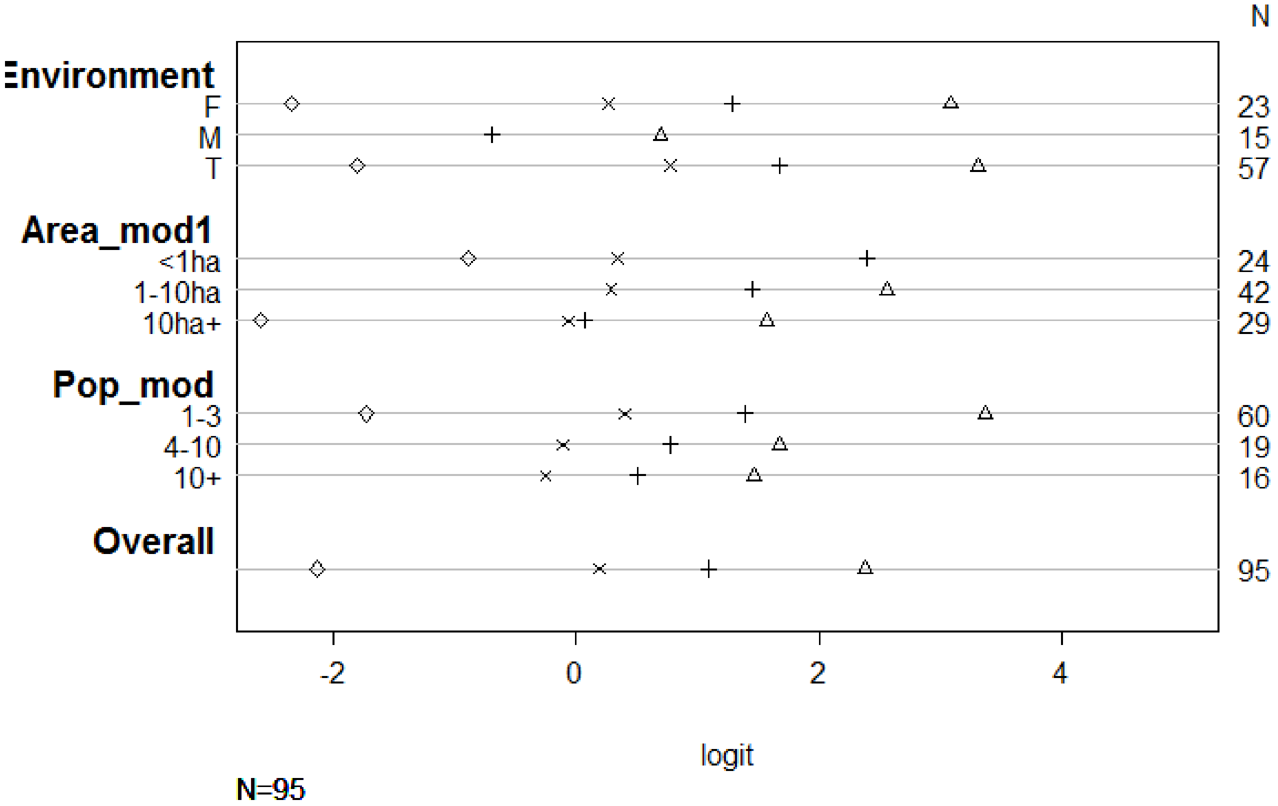
Factor plot of risk management components. Cost, Impact, Practicality and Acceptability were all highly correlated and were the main driver of dimension 1(37.8% variation) but these components did not correlate with Likelihood of reintroduction. Window of Opportunity had the highest correlation with Dimension 2 (17.3% variability).



Appendix H. nMDS ordination of risk management components applied to new and emerging invasive non-native species in the EU

nMDS (non-metric Multidimensional scaling) ordination of all species based on the component scores (Effectiveness, Practicality, Cost, Impact, Acceptability Window of Opportunity, and Likelihood of Reintroduction), coloured based on Overall Score. The axes of this plot are the same as those in the factor analysis above (Appendix G), with Dim 1 correlated with Effectiveness, Practicality, Cost, Impact and Acceptability, while Dim 2 is more closely correlated with Window of Opportunity and Likelihood of Reintroduction. The coloured ellipses are a visual aid to show the mean (large symbol) and variation (the scaled shape and size of the ellipse) of Overall Score. Overall Score aligns in sequence with Dim1 but with some overlap, or species out of sequence, particularly between scores 2, 3 and 4.

Appendix I. Pairwise separation of thresholds of each ordinal scale for each risk management covariate.



Appendix J. Cumulative link model summary for Overall Score (overall feasibility of eradication) predicted by environment, total area and number of populations

link	threshold	nobs	logLik	AIC	niter	max.grad	cond.H
logit	flexible	95	-111.45	242.90	6(0)	6.67e-11	6.3e+01

Coefficients:

Covariate	Estimate	Std. Error	z value	Pr(>|z|)
EnvironmentM	-2.5875	0.6801	-3.805	0. 000142 ***
EnvironmentT	1.1538	0.5232	2.205	0.027436 *
Area_mod11-10ha	-1.2732	0.5574	-2.284	0.022348 *
Area_mod110ha+	-1.6272	0.6051	-2.689	0.007166 **
Pop_mod4-10	-1.1217	0.5465	-2.052	0.040122 *
Pop_mod10+	-1.5621	0.5885	-2.654	0.007944 **

Appendix K. Management priorities for emerging (i.e. established with limited distributions) invasive non-native species in the EU.

Priorities for emerging (i.e. established with limited distributions) invasive non-native species in the EU (n=35): highest (0), very high (4), high (10), med-high (10), medium (7), med-low (4). Potential priorities for long term management based on high risk and low feasibility of eradication are denoted $^{++}$highest and $^{+}$very high priority. F^1 = scenario based on species in still (or slow flowing) freshwater; F^2 = scenario based on species in flowing freshwater.

Scientific Name	English Name	Env.	Group	Scenario code	Risk category	Feasibility of eradication	Priority
Acridotheres tristis	Common myna	T	Bird	A5	VH	H	Very high
Bufo mauritanicus	Berber toad	T	Amph.	A2	VH	H	Very high
Nasua nasua	Coati	T	Mammal	A4	VH	H	Very high
Pycnonotus cafer	Red-vented Bulbul	T	Bird	A5	VH	H	Very high
Alternanthera philoxeroides	Alligator-weed	F	V. Plant	C2	VH	M	High
Axis axis	Indian spotted deer	T	Mammal	A6	H	H	High
Botrylloides giganteum	a tunicate	M	Tunicate	A1	VH	M	High
Cherax destructor	Common yabby	F^1	Crust.	A1	H	H	High
Euonymus fortunei	Winter Creeper	T	V. Plant	A2	H	H	High
Euonymus japonicus	Japanese spindle	T	V. Plant	B2	H	H	High
Ligustrum sinense	Chinese Privet	T	V. Plant	B2	H	H	High
Misgurnus anguillicaudatus	Oriental weatherfish	F^2	Fish	C4	H	H	High
Rhea americana	Greater rhea	T	Bird	A5	H	H	High
Saperda candida	Apple Tree Borer	T	Insect	B2	H	H	High
Andropogon virginicus	Broom-sedge	T	V. Plant	C2	H	M	Med-high
Ehrharta calycina	Perennial Veldtgrass	T	V. Plant	B2	H	M	Med-high
Fundulus heteroclitus	Mummichog	F^2	Fish	B3	H	M	Med-high
Hypostomus plecostomus	Suckermouth catfish	F^2	Fish	A1	H	M	Med-high
Marisa cornuarietis	Giant ramshorn snail	F^2	Mollusc	A1	H	M	Med-high
Wedelia trilobata	Wedelia	T	V. Plant	B2	H	M	Med-high
Callosciurus finlaysonii	Finlayson's squirrel	T	Mammal	A6	VH	L	Med-high^{+}

Herpestes auropunctatus	Small Asian mongoose	T	Mammal	B6	VH	L	Med-high+
Pomacea canaliculata	Golden apple snail	F2	Mollusc	A2	VH	L	Med-high+
Pomacea maculata	Giant apple snail	F1	Mollusc	C3	VH	L	Med-high+
Acridotheres cristatellus	Crested Myna	T	Bird	B6	H	L	Medium
Charybdis japonica	Asian paddle crab	M	Decapod	A3	H	L	Medium
Pheidole megacephala	Big-headed Ant	T	Insect	D4	H	L	Medium
Psittacula eupatria	Alexandrine parakeet	T	Bird	B5	H	L	Medium
Arthurdendyus triangulatus	New Zealand flatworm	T	Platy.	D2	VH	VL	Medium++
Penaeus aztecus	Northern brown shrimp	M	Crust.	B6	VH	VL	Medium++
Pterois miles	Devil firefish, Lion fish	M	Fish	C6	VH	VL	Medium++
Ashworthius sidemi	None	T	Nematode	D6	H	VL	Med-low+
Bellamya chinensis	Chinese mysterysnail	F2	Mollusc	B2	H	VL	Med-low+
Macrorhynchia philippina	White stinger	M	Hydroid	B6	H	VL	Med-low+
Pseudonereis anomala	a polychaete	M	Poly.	A5	H	VL	Med-low+

Appendix L. Management priorities for new (i.e. not established) invasive non-native species in the EU.

Management priorities for new (i.e. not established) invasive non-native species in the EU (n=60): highest (1), very high (16), high (26), med-high (10), medium (7), med-low (0). Potential priorities for prevention based on high risk and low feasibility of eradication are denoted ++highest and +very high priority. F^1 = scenario based on species in still (or slow flowing) freshwater; F^2 = scenario based on species in flowing freshwater.

Scientific Name	English Name	Env.	Group	Scenario code	Risk category	Feasibility of eradication	Priority
Orconectes rusticus	Rusty crayfish	F^1	Crust.	A1	VH	VH	Highest
Bison bison	American bison	T	Mammal	A4	H	VH	Very high
Channa argus	Northern snakehead	F^2	Fish	A2	VH	H	Very high
Cryptostegia grandiflora	None	T	V. Plant	A1	H	VH	Very high
Gambusia affinis	Western mosquitofish	F^1	Fish	A2	VH	H	Very high
Lampropeltis getula	Common Kingsnake	T	Reptile	A4	VH	H	Very high
Lonicera morrowii	Morrow's Honeysuckle	T	V. Plant	A2	H	VH	Very high
Micropterus dolomieu	Smallmouth bass	F^1	Fish	A1	VH	H	Very high
Misgurnus mizolepis	Chinese weather loach	F^1	Fish	A1	H	VH	Very high
Oreochromis aureus	Blue tilapia	F^1	Fish	A2	VH	H	Very high
Oreochromis mossambicus	Mossambique tilapia	F^1	Fish	A2	VH	H	Very high
Oreochromis niloticus	Nile tilapia	F^1	Fish	B2	VH	H	Very high
Pachycondyla chinensis	Asian Needle Ant	T	Insect	B1	H	VH	Very high
Rubus rosifolius	Roseleaf Bramble	T	V. Plant	A1	H	VH	Very high
Sirex ermak	Blue-black Horntail	T	Insect	A1	H	VH	Very high
Solenopsis Invicta	Red Imported Fire Ant	T	Insect	A1	H	VH	Very high
Trichosurus Vulpecula	Brushtail Possum	T	Mammal	A4	H	VH	Very high

Aeolesthes sarta	City Longhorn Beetle	T	Insect	C3	H	H	High
Albizia lebbeck	Indian Siris	T	V. Plant	B2	H	H	High
Amynthas agrestis	Crazy snake worm	T	Annelid	C1	H	H	High
Boiga irregularis	Brown tree snake	T	Reptile	A2	H	H	High
Celastrus orbiculatus	Oriental Bittersweet	T	V. Plant	C3	H	H	High
Cherax quadricarinatus	Redclaw crayfish	F[1]	Crust.	A1	H	H	High
Chromolaena odorata	None	T	V. Plant	A2	H	H	High
Chrysemys picta	Painted turtle	T	Reptile	B3	H	H	High
Cinnamomum camphora	Camphor Tree	T	V. Plant	A2	H	H	High
Clematis terniflora	Leather Leaf Clematis	T	V. Plant	B2	H	H	High
Crepidula onyx	Onyx slippersnail	M	Mollusc	A2	VH	M	High
Cyprinella lutrensis	Red shiner	F[2]	Fish	A1	H	H	High
Eleutherodactylus coqui	Common coquí	T	Amph.	A2	H	H	High
Gymnocoronis spilanthoides	Senegal tea	T	V. Plant	A2	H	H	High
Limnoperna fortunei	Golden mussel	F[?]	Mollusc	A1	VH	M	High
Lonicera maackii	Amur Honeysuckle	T	V. Plant	A2	H	H	High
Mytilopsis sallei	Black striped mussel	M	Mollusc	A1	VH	M	High
Prosopis juliflora	Prosopis	T	V. Plant	C2	H	H	High
Prunus campanulata	Bell flower cherry	T	V. Plant	A2	H	H	High
Pycnonotus jocosus	Red-whiskered Bulbul	T	Bird	A5	H	H	High
Rhinella marina	Cane toad	T	Amph.	A4	H	H	High
Solenopsis geminata	Tropical fire ant	T	Insect	A1	H	H	High
Tetropium gracilicorne	Fine-horned spruce beetle	T	Insect	C2	H	H	High
Tilapia zillii	Redbelly tilapia	F[?]	Fish	B2	H	H	High
Triadica sebifera	Chinese Tallowtree	T	V. Plant	A1	H	H	High
Vespula pensylvanica	Western yellowjacket	T	Insect	C2	H	H	High
Acanthophora spicifera	a red alga	M	Alga	A1	H	M	Med-high
Cortaderia jubata	None	T	V. Plant	A2	H	M	Med-high
Cynops pyrrhogaster	Fire-bellied salamander	T	Amph.	A2	H	M	Med-high
Hemidactylus frenatus	House gecko	T	Reptile	A1	H	M	Med-high

Lygodium japonicum	Japanese Climbing Fern	T	V. Plant	A2	H	M	Med-high
Microstegium vimineum	Nepalese Browntop	T	V. Plant	B2	H	M	Med-high
Solenopsis richteri	Black Imported Fire Ant	T	Insect	D2	H	M	Med-high
Symplegma reptans	a tunicate	M	Tunicate	A1	H	M	Med-high
Codium parvulum	a green alga	M	Alga	A2	VH	L	Med-high^{+}
Homarus americanus	American Lobster	M	Crustacean	A3	VH	L	Med-high^{+}
Eleutherodactylus planirostris	Greenhouse frog	T	Amph.	A2	H	L	Medium
Gammarus fasciatus	Freshwater shrimp	F[1]	Crust.	A1	H	L	Medium
Lespedeza juncea ssp. sericea	None	T	V. Plant	C2	H	L	Medium
Morone americana	White perch	F[2]	Fish	A1	H	L	Medium
Perna viridis	Asian Green mussel	M	Mollusc	A2	H	L	Medium
Potamocorbula amurensis	Asian basket clam	M	Mollusc	A3	H	L	Medium
Plotosus lineatus	Striped eel catfish	M	Fish	A2	VH	VL	Medium^{++}

Appendix M. Annotated code used for analysis

Chapter 2. Comprehensive biodiversity impact scores reveal taxonomic, environmental, geographic and temporal patterns of invasiveness in Britain

```
# Fig 2.2 ----
# Which variables are most important determinants of impact?
# Data are categorigcal / ordinal and sparse at some levels - use random forest approach

# get data
imp1<-read.csv("chpt2.csv")

#structure the dataset
dat <- imp1
head(dat)
names(dat) #MI is the focus
dat<-as.data.frame(dat)
str(dat)
myvars<-c("broadtaxa","env","func","C1","year","MI")
dat<-dat[myvars]
dat[sapply(dat, is.character)] <- lapply(dat[sapply(dat, is.character)], as.factor)
head(dat)
na.omit(dat)
impDat<-na.omit(dat)

install.packages("randomForest")
library(randomForest)
impDat<-rfImpute(MI~.,dat)#imputing missing data

table(impDat$MI)

# data is inbalanced (many more MI) needs to be balanced
# exploring best balancing options, choose option with least error in confusion matrix

install.packages("UBL")
library(UBL)
oveBalan <- RandOverClassif(MI~., impDat, "balance")
table(oveBalan$MI)
oveInvert <- RandOverClassif(MI~., impDat, "extreme")
table(oveInvert$MI)
rf1 = randomForest(MI~.,impDat,ntree=500, sampsize=500)
rf1#look at the confusion matrix
rf2 = randomForest(MI~.,oveBalan,ntree=4000,sampsize=1000,strata=oveBalan$MI)
rf2#reduce the error in the confusion matrix
rf3=randomForest(MI~.,oveInvert,ntree=4000,sampsize=1500,strata=oveInvert$MI)
rf3#the error is higher than rf2 so we go with overbalance

#Random Forest classifier
install.packages("Boruta")
library(Boruta)
```

```
names(oveBalan)<-c("Broadtaxa", "Environment", "Functionalgroup","Origin", "Year", "Impact")
mod1<-Boruta(Impact~Broadtaxa+Environment+Origin+Functionalgroup,data=oveBalan)#missing
year and func
boruta_signif<-names(mod1$finalDecision[mod1$finalDecision %in% c("Confirmed",
"Tentative")])
print(boruta_signif)
plot(mod1,cex.axis=.7, las=2, xlab="", main="Variable importance for impact classification")


# Fig 2.4 ----
# Modelling invasive and non-invasive species through time

library(ggplot2)
library(tidyverse)
library(lattice)

#set data
inv<-imp1

#cut data into 20 year bins
fullyears<-seq(1,16, by=1) #needed for full_join below

inv20<-inv%>%
  filter(year>1680,year<2000)%>% #data before 1680 are too sparse and data after 2000 are
incomplete
  mutate(cutyear=cut(year,seq(1680,2010, by=20), labels=F))%>%
  group_by(invasive, cutyear)%>%
  count(invasive)%>%
  full_join(expand.grid(cutyear=fullyears, invasive=c("NNS", "INNS")))%>% #adds cutyear where
n=0
  as.data.frame

inv20[is.na(inv20)]<-0

#add actual year label to cutyear
actualyear<-seq(1700,2000, by=20) #the bin runs 1700-1720, etc.  1980 = 1980-2000
years<- as.data.frame (actualyear)
years$cutyear<- fullyears
inv20<-merge(inv20, years)

#model number of species per 20 year bin
bestbin20<- glm(log(n+1)~cutyear+invasive,  data=inv20) #interaction model
summary(bestbin20)
xyplot((exp(fitted(bestbin20))-1)+n~cutyear|invasive, data = inv20) #check model fit

#predict number of species using model
new.inv <- expand.grid(cutyear=c(1:17), invasive=c("INNS","NNS")) #creates empty grid with 17
rows each for NNS and INNS
new.inv$spp<-predict(bestbin20,newdata=new.inv,interval='confidence') #predict (log) number of
species for each case of cutyear and NNS or INNS
```

```
new.inv$expspp<-exp(new.inv$spp)-1 #exponentiate this number
new.inv<-merge(new.inv, years) #adds year labels

#plot the actual and predicted number of species per 20 year bin
ggplot()+
  geom_smooth(data=new.inv,aes(actualyear, expspp, group=invasive, col=invasive) )+
  geom_point(data=inv20, aes(actualyear, n, group=invasive, col=invasive))+
  xlim(1700,2000)+ ylim (0,300) + xlab("")+ ylab("")+theme_bw()+
  scale_colour_discrete(name="", labels=c("invasive", "non-invasive"))

##repeat for terrestrial plants only

#cut data into 20 year bins
fullyears<-seq(1,16, by=1) #needed for full_join below

inv20<-inv%>%
  filter(broadtaxa=="Plant" & env=="Terrestrial") %>%
  filter(year>1680,year<2000)%>% #data before 1680 are too sparse and data after 2000 are
incomplete
  mutate(cutyear=cut(year,seq(1680,2010, by=20), labels=F))%>%
  group_by(invasive, cutyear)%>%
  count(invasive)%>%
  full_join(expand.grid(cutyear=fullyears, invasive=c("NNS", "INNS")))%>% #adds cutyear where
n=0
  as.data.frame

inv20[is.na(inv20)]<-0

#add actual year label to cutyear
actualyear<-seq(1700,2000, by=20) #the bin runs 1700-1720, etc.  1980 = 1980-2000
years<- as.data.frame (actualyear)
years$cutyear<- fullyears
inv20<-merge(inv20, years)

#model number of species per 20 year bin
bestbin20<- glm(log(n+1)~cutyear+invasive,  data=inv20) #interaction model
summary(bestbin20)
xyplot((exp(fitted(bestbin20))-1)+n~cutyear|invasive, data = inv20) #check model fit

#predict number of species using model
new.inv <- expand.grid(cutyear=c(1:17), invasive=c("INNS","NNS")) #creates empty grid with 17
rows each for NNS and INNS
new.inv$spp<-predict(bestbin20,newdata=new.inv,interval='confidence') #predict (log) number of
species for each case of cutyear and NNS or INNS
new.inv$expspp<-exp(new.inv$spp)-1 #exponentiate this number
new.inv<-merge(new.inv, years) #adds year labels

#plot the actual and predicted number of species per 20 year bin
ggplot()+
  geom_smooth(data=new.inv,aes(actualyear, expspp, group=invasive, col=invasive) )+
```

```
  geom_point(data=inv20, aes(actualyear, n, group=invasive, col=invasive))+
  xlim(1700,2000)+ ylim (0,200) + xlab("")+ ylab("")+theme_bw()+
  scale_colour_discrete(name="", labels=c("invasive", "non-invasive"))

##repeat excluding terrestrial plants

#cut data into 20 year bins
fullyears<-seq(1,16, by=1) #needed for full_join below

inv20<-inv%>%
  filter(broadtaxa!="Plant" | env!="Terrestrial") %>%
  filter(year>1680,year<2000)%>% #data before 1680 are too sparse and data after 2000 are
incomplete
  mutate(cutyear=cut(year,seq(1680,2010, by=20), labels=F))%>%
  group_by(invasive, cutyear)%>%
  count(invasive)%>%
  full_join(expand.grid(cutyear=fullyears, invasive=c("NNS", "INNS")))%>% #adds cutyear where
n=0
  as.data.frame

inv20[is.na(inv20)]<-0

#add actual year label to cutyear
actualyear<-seq(1700,2000, by=20) #the bin runs 1700-1720, etc.  1980 = 1980-2000
years<- as.data.frame (actualyear)
years$cutyear<- fullyears
inv20<-merge(inv20, years)

#model number of species per 20 year bin
bestbin20<- glm(log(n+1)~cutyear+invasive,  data=inv20) #interaction model
summary(bestbin20)
xyplot((exp(fitted(bestbin20))-1)+n~cutyear|invasive, data = inv20) #check model fit

#predict number of species using model
new.inv <- expand.grid(cutyear=c(1:17), invasive=c("INNS","NNS")) #creates empty grid with 17
rows each for NNS and INNS
new.inv$spp<-predict(bestbin20,newdata=new.inv,interval='confidence') #predict (log) number of
species for each case of cutyear and NNS or INNS
new.inv$expspp<-exp(new.inv$spp)-1 #exponentiate this number
new.inv<-merge(new.inv, years) #adds year labels

#plot the actual and predicted number of species per 20 year bin
ggplot()+
  geom_smooth(data=new.inv,aes(actualyear, expspp, group=invasive, col=invasive) )+
  geom_point(data=inv20, aes(actualyear, n, group=invasive, col=invasive))+
  xlim(1700,2000)+ ylim (0,100) + xlab("")+ ylab("")+theme_bw()+
  scale_colour_discrete(name="", labels=c("invasive", "non-invasive"))
```

Chapter 3. Ranking the introduction pathways of non-native species in Britain

```
library(tidyverse)

# Table 3.4 ----
# Comparing different pathway ranking methods
# Use Kendall's tau to consider similarity (difference) between ranked lists
# IMPORTANT note pathway correlation have changed slightly because the pathway S_PEOP has
been removed

#get data
dat<-read.csv("read.csv("chpt3.csv")

# Table 3.4a ----
# comparing pathways ranked by methods 1, 2a and 2b
# to do this pathways need to first be ranked by the three different methods:
# simple count of all species, simple count of species that score > minimal impact,
# sum of all species moentised impact.  These then need to be tabulated and compared.

# METHOD 1 rank pathways by number of all non-native species introduced

dat %>% filter(PWAY!="Unknown" & PWAY!="NIL") -> dat

dat %>%
  group_by(PWAY) %>%
  summarise(SpCount=sum(max)) %>% #groups species by pathway and provides a count
  arrange(-SpCount) %>%
  mutate(method1=(1:36)) ->method1 # ranks from 1 to n

# METHOD 2a rank pathways by number of 'invasive' species introduced (i.e. > minimal impact)

dat %>%
  filter(Final.Maximum.Impact!="Minimal Concern") %>%
  group_by(PWAY) %>%
  summarise(MCexCount=sum(max)) %>%
  arrange(-MCexCount) %>%
  mutate(method2a=(1:31)) ->method2a

# METHOD 2b rank pathways by sum of impact values

dat %>%
  group_by(PWAY) %>%
  summarise(ImpCount=sum(maximp)) %>%
  arrange(-ImpCount) %>%
  mutate(method2b=(1:36)) ->method2b # sum of impact per pathway

# combine method 1, 2a, 2b ranks in a single table

method1 %>%
  left_join(method2a, by="PWAY") %>%
```

```
  left_join(method2b, by="PWAY") %>%
  select(PWAY, method1, method2a, method2b) ->methods12a2bpway # this table includes the
pway name for the write up

methods12a2bpway %>%
  select(method1, method2a, method2b) -> methods12a2b # pathway name removed here for 'cor'
analysis

methods12a2b[is.na(methods12a2b)] <- 36 #replaces NA with number outside of the rank range

# Correlations between pathways ranked by simple count and impact scoring methods.

library(Kendall)
cor(methods12a2b, method="kendall")

summary(Kendall(methods12a2b$method1, methods12a2b$method2a))
summary(Kendall(methods12a2b$method1, methods12a2b$method2b))
summary(Kendall(methods12a2b$method2a, methods12a2b$method2b))

# Table 3.4b ----
# Comparing pathways ranked by methods 3a, 3b and 3c
# to do this pathways need to first be ranked by the three different methods:
# minimum, weighted, maximum. These then need to be tabulated and compared.
# the following steps are here to the weighted score for species with multiple pathways
# to do this, the number of pathways per species is calculated and the weighted score
# for species with multiple pathways is given as 1/n (where n is the number of pathways);
# the minimum score is also calcualted by only scoring species where the number of pathways = 1,
# otherwise if n>1 the species scores 0 for minimum.
# at this point each species has been scored min, mid, max based on number of pathways

dat %>%
  select (name, PWAY, min, mid, max) %>%
  gather (minmidmax, Count,-PWAY, -name) %>%
  group_by (PWAY, minmidmax) %>%
  summarise (CumCount=sum(Count)) %>%
  spread (minmidmax, CumCount) -> unccount # sums the number of species in min, mid, max for
each pway

# now need to rank pways by each of min, mid and max

unccount %>%
  arrange(-mid) %>%
  ungroup() %>%
  mutate(method3b=(1:nrow(unccount))) %>%
  select(method3b, PWAY, mid)->method3b

unccount %>%
  arrange(-min) %>%
  ungroup() %>%
  mutate(method3a=(1:nrow(unccount))) %>%
```

```
  select(method3a, PWAY, min)->method3a

unccount %>%
  arrange(-max) %>%
  ungroup() %>%
  mutate(method3c=(1:nrow(unccount))) %>%
  select(method3c, PWAY, max)->method3c

method3b %>% #creates table of all 3 methods with ranks
  left_join(method3a, by="PWAY") %>%
  left_join(method3c, by="PWAY") %>%
  select(PWAY, method3a, method3b, method3c, min, mid, max) ->methods3a3b3cpway

methods3a3b3cpway %>%
  select(method3a, method3b, method3c) -> methods3a3b3c

# Correlations between pathways ranked by different uncertainty scoring methods and simple count.

cor(methods3a3b3c, method="kendall")

# Table 3.4c ----
# Comparing pathways ranked by 50 year time period
# to do this, species are first seperated into 50 year periods, then simple count is calculated
# the following code is needed to add year of first record into step 3
# note impact scores etc are included in this data in case of future need (not part of this analysis)
# ranking species based on simple count for each time period, each period is done seperately

#1951-2000

dat %>%
  filter(year_of_first_rec_in_wild>1951 & year_of_first_rec_in_wild<=2000) -> t1

t1 %>%
  mutate(ctr=1) %>%
  select(PWAY, ctr) %>%
  group_by(PWAY) %>%
  summarise(Count=sum(ctr)) %>%
  arrange (-Count)->t1tau

t1tau %>% mutate(RANK=(1:nrow(t1tau)))->t19512000

#1901-1950

dat %>%
  filter(year_of_first_rec_in_wild>1901 & year_of_first_rec_in_wild<=1950) -> t2

t2 %>%
  mutate(ctr=1) %>%
  select(PWAY, ctr) %>%
  group_by(PWAY) %>%
```

```
  summarise(Count=sum(ctr)) %>%
  arrange (-Count)->t2tau

t2tau %>% mutate(RANK=(1:nrow(t2tau)))->t19011950

#1851-1900
dat %>%
  filter(year_of_first_rec_in_wild>1851 & year_of_first_rec_in_wild<=1900) -> t3

t3 %>%
  mutate(ctr=1) %>%
  select(PWAY, ctr) %>%
  group_by(PWAY) %>%
  summarise(Count=sum(ctr)) %>%
  arrange (-Count)->t3tau

t3tau %>% mutate(RANK=(1:nrow(t3tau)))->t18511900

#1801-1850

dat %>%
  filter(year_of_first_rec_in_wild>1801 & year_of_first_rec_in_wild<=1850) -> t4

t4 %>%
  mutate(ctr=1) %>%
  select(PWAY, ctr) %>%
  group_by(PWAY) %>%
  summarise(Count=sum(ctr)) %>%
  arrange (-Count)->t4tau

t4tau %>% mutate(RANK=(1:nrow(t4tau)))->t18011850

# now to join in the same table and compare

t19512000 %>%
  left_join(t19011950, by="PWAY") %>%
  left_join(t18511900, by="PWAY") %>%
  left_join(t18011850, by="PWAY") %>%
  rename(yr19512000=RANK.x, yr19011950=RANK.y, yr18511900=RANK.x.x,
yr18011850=RANK.y.y) %>%
  select(PWAY, yr19512000, yr19011950, yr18511900, yr18011850)->tautimewithpway

tautimewithpway[is.na(tautimewithpway)] <- 29 #replaces NA with number outside of the rank
range

tautimewithpway %>% select(-PWAY)->tautime

# Correlations between pathways ranked at different time periods.
#Ranks are based on simple counts of species introduced in each 50 year period.
```

```
cor(tautime, method="kendall")

# Table 3.4d ----
# Comparing pathways ranked by method 1 and method 4
# firstly, get the combined methods:
# imapct score, minimp, midimp, maximp ordered by midimp
# then take a 50yr snapshop

# rank pathways by combined methods

#### combined methods with time cut and unknown pways excluded

dat %>%
  filter(year_of_first_rec_in_wild>1951 & year_of_first_rec_in_wild<=2015) -> method4step1
#first step - based on species recently introduced

method4step1 %>%
  select (PWAY, minimp, midimp, maximp) %>%
  gather (minmidmax,impscr,-PWAY) %>%
  group_by (PWAY, minmidmax) %>%
  summarise(CumImpact=sum(impscr)) %>%  #calculates impact score for each of min, mid, max
  spread(minmidmax, CumImpact, fill=0) ->method4step2

method4step2 %>%
  ungroup() %>%
  arrange(-midimp) %>% #only mid is used for combined method ranking of pathways (min and
max are used to plot uncertainty later)
  mutate(method4=1:nrow(method4step2))->method4

# add to simple count

method1 %>%
  left_join(method4, by="PWAY") %>%
  select(PWAY, method1, method4)->methods14pway

methods14pway[is.na(methods14pway)] <- 47

methods14pway %>% select(-PWAY)->methods14

#Table 3.4d Correlation between pathways ranked by combined methods
#(incorporating impact, uncertainty and time) and simple count.

cor(methods14, method="kendall")
summary(Kendall(methods14$method1, methods14$method4))

# Fig 3.2. ----
# PlotS illustrating correlation between pathways ranked by scoring methods
Fig.3.2.a<-ggplot()+geom_point(data=methods12a2b, aes(x=method2a, y=method1))+
  xlab("Method 2a")+ylab("Method 1")+
  geom_text(aes(x=10, y=47),label="\u03c4 = 0.38")
```

```
Fig.3.2.b<-ggplot()+geom_point(data=methods12a2b, aes(x=method2b,
y=method1))+xlab("Method 2b")+ylab("Method1")+
  geom_text(aes(x=10, y=47),label="\u03c4 = 0.32")
Fig.3.2.c<-ggplot()+geom_point(data=methods12a2b, aes(x=method2a,
y=method2b))+xlab("Method 2a")+ylab("Method 2b")+
  geom_text(aes(x=10, y=47),label="\u03c4 = 0.72")
Fig.3.2.d<-ggplot()+geom_point(data=methods14, aes(x=method4, y=method1))+xlab("Method
4")+ylab("Method 1")+
  geom_text(aes(x=10, y=47),label="\u03c4 = 0.29")
```

Chapter 4. Risk management to prioritise the eradication of new and emerging invasive non-native species

```
install.packages("polycor")
library(polycor)
library(tidyverse)

# Section 4.5 (para 2) ----
# Exploring relationship between individual risk management components
# and overall score using polychoric correlations as both variables are ordinal

#getdata
gbRMdat<-read.csv("chpt4.csv")
gbRMdat %>% select("Effect", "Pract", "Cost", "Imp", "Accept", "Oppo", "Rein", "Overall") -
>gbRMdat2

#set ordinal factors
gbRMdat2$Effect<-factor(gbRMdat2$Effect, levels = c("verylow", "low", "medium", "high",
"veryhigh")) #set RM levels
gbRMdat2$Pract<-factor(gbRMdat2$Pract, levels = c("verylow", "low", "medium", "high",
"veryhigh")) #set RM levels
gbRMdat2$Cost<-factor(gbRMdat2$Cost, levels = c("verylow", "low", "medium", "high",
"veryhigh")) #set RM levels
gbRMdat2$Imp<-factor(gbRMdat2$Imp, levels = c("verylow", "low", "medium", "high",
"veryhigh")) #set RM levels
gbRMdat2$Accept<-factor(gbRMdat2$Accept, levels = c("verylow", "low", "medium", "high",
"veryhigh")) #set RM levels
gbRMdat2$Oppo<-factor(gbRMdat2$Oppo, levels = c("verylow", "low", "medium", "high",
"veryhigh")) #set RM levels
gbRMdat2$Rein<-factor(gbRMdat2$Rein, levels = c("verylow", "low", "medium", "high",
"veryhigh")) #set RM levels
gbRMdat2$Overall<-factor(gbRMdat2$Overall, levels = c("verylow", "low", "medium", "high",
"veryhigh")) #set RM levels

#compare correlations, standard deviation and test for binomial normality
hetcor(gbRMdat2, ML = FALSE, std.err = TRUE, bins=4, pd=TRUE)

# Fig. 4.1 and 4.2 (and para 3, p 100) ----
# Factor Map and nMDS.
# Exploring the relationship between species and individual risk management components (nMDS)
# and among risk management components (FactorMap)

install.packages("FactoMineR")
install.packages("factoextra")

library(FactoMineR)
library(plyr)
library(MASS)
library(factoextra)
library(ggrepel)
```

```
library(ggplot2)
library(polycor)

#get data
riskx<- gbRMdat

#set factors to numeric for PCA
riskx$Effect<-as.numeric(as.character(revalue(riskx$Effect, c("veryhigh"="5", "high"="4",
"medium"=3, "low"=2, "verylow"=1))))
riskx$Pract<-as.numeric(as.character(revalue(riskx$Pract, c("veryhigh"=5, "high"=4, "medium"=3,
"low"=2, "verylow"=1))))
riskx$Cost<-as.numeric(as.character(revalue(riskx$Cost, c("veryhigh"=1, "high"=2, "medium"=3,
"low"=4, "verylow"=5))))
riskx$Imp<-as.numeric(as.character(revalue(riskx$Imp, c("veryhigh"=1, "high"=2, "medium"=3,
"low"=4, "verylow"=5))))
riskx$Accept<-as.numeric(as.character(revalue(riskx$Accept, c("veryhigh"=5, "high"=4,
"medium"=3, "low"=2, "verylow"=1))))
riskx$Oppo<-as.numeric(as.character(revalue(riskx$Oppo, c("veryhigh"=5, "high"=4,
"medium"=3, "low"=2, "verylow"=1))))
riskx$Rein<-as.numeric(as.character(revalue(riskx$Rein, c("veryhigh"=1, "high"=2, "medium"=3,
"low"=4, "verylow"=5))))
riskx$Overall<-as.numeric(as.character(revalue(riskx$Overall, c("veryhigh"=5, "high"=4,
"medium"=3, "low"=2, "verylow"=1))))

#structure and set factor
row.names(riskx)<- riskx[,1]
risk2<-riskx
risk2$Overall<- factor(risk2$Overall)
names(risk2)<- c("Taxa", "Effectiveness", "Practicality", "Cost", "Impact", "Acceptability",
"Opportunity", "Reintroduction", "Overall")

# try PCA
res.pca <- PCA(risk2, scale.unit=TRUE, ncp=5, quali.sup=c(1,9),  graph=F)

#data is non-numeric, try nMDS
mydata<-risk2[,2:8]
d <- dist(mydata) #distance matrix
d<-d+0.001# euclidean distances between the rows
fit <- isoMDS(d, k=2) # k is the number of dim
fit # view results

# plot solution
x <- fit$points[,1]
y <- fit$points[,2]
plot(x, y, xlab="Coordinate 1", ylab="Coordinate 2", main="Nonmetric MDS", type="n")
text(x, y, labels = row.names(mydata), cex=.7)

# put into the res.pca object for plotting
fit$points[,1]<-fit$points[,1]*-1
res.pca$ind$coord<-fit$points
```

```
# plot nMDS for individuals
nmDSplot<-fviz_pca_ind(res.pca, habillage = 9,label="ind",
                      addEllipses =TRUE, ellipse.level = 0.5,title = "", ellipse.type= "t", repel =T, labelsize
= 3, pointsize= 2) +
  scale_fill_manual(values=c("Dark Red", "Red", "Darkorange3", "Green3", "Dark
Green"),guide=FALSE)+
  scale_shape_manual(values=c(16,17, 15, 16, 17))+
  scale_color_manual(values=c("Dark Red", "Red", "darkorange3", "Green3", "Dark Green"))+
  scale_size(guide=FALSE)+
  guides(labels=FALSE)+
  theme_classic()+
  theme(legend.background = element_rect(),legend.position=c(.06, .19),
        panel.border = element_rect(colour = "black", fill=NA, size=1),
        text = element_text(size=8, family="Times"),
        axis.ticks.length=unit(-0.25, "cm"), axis.text.x = element_text(margin=unit(c(0.5,0.5,0.5,0.5),
"cm")),
        axis.text.y = element_text(margin=unit(c(0.5,0.5,0.5,0.5), "cm"))
  )

nmDSplot
dev.off()

# plot factor map showing risk management components
factormap<-fviz_pca_var(res.pca, col.var="contrib", title = "", labelsize = 6 )+
  scale_color_gradient2(low="light blue", mid="blue", high="red", midpoint = 12) +
  theme_classic()+
  theme(legend.background = element_rect(),legend.position=c(.1, .19),
        panel.border = element_rect(colour = "black", fill=NA, size=1),
        text = element_text(size=12, family="Times"),
        axis.ticks.length=unit(-0.25, "cm"), axis.text.x = element_text(margin=unit(c(0.5,0.5,0.5,0.5),
"cm")),
        axis.text.y = element_text(margin=unit(c(0.5,0.5,0.5,0.5), "cm"))
  )
Factormap
```

Chapter 5. Prioritising the management of new and emerging invasive non-native species at a continental scale

```
library(tidyverse)
library(plyr)
library(polycor)
library(FactoMineR)
library(factoextra)
library(MASS)

# get data and set levels
eudat0<-read.csv("chpt5.csv")
eudat<-eudat0
eudat$E<-factor(eudat$E, levels = c("VL", "L", "M", "H", "VH")) #set RM levels
eudat$P<-factor(eudat$P, levels = c("VL", "L", "M", "H", "VH")) #set RM levels
eudat$C<-factor(eudat$C, levels = c("VH", "H", "M", "L", "VL")) #note these run in opposite
direction to E and P
eudat$I<-factor(eudat$I, levels = c("VH", "H", "M", "L", "VL")) #note these run in opposite
direction to E and P
eudat$A<-factor(eudat$A, levels = c("VL", "L", "M", "H", "VH")) #set RM levels
eudat$W<-factor(eudat$W, levels = c("Vshort", "Short", "Med", "Long", "Vlong")) #set RM levels
eudat$L<-factor(eudat$L, levels = c("VH", "H", "M", "L", "VL")) #note these run in opposite
direction to E and P
eudat$RM<-factor(eudat$RM, levels = c("VL", "L", "M", "H", "VH")) #set RM levels
eudat$RM<-factor(eudat$RM, levels = c("VL", "L", "M", "H", "VH")) #set RM levels
eudat$RA<-factor(eudat$RA, levels = c("H", "VH")) #set RA levels

# Section 5.4.1. (para 1, p.118) ----
# Checking for correlation between RA and RM scores

#select data
eudat2<-as.data.frame (eudat[,c(1,17,30)])

#test polychoric correlation between RA and RM
polychor(eudat2$RA, eudat2$RM, std.err=TRUE, ML=FALSE) #polychoric correlation (rho) is -
0.2806 with estimated standard error of 0.1356

# Section 5.4.1 (para 2, p.119 - and Appendices G and H)----
# Factor plot and nMDS

#select and structure data
eudat3<-as.data.frame (eudat[,c(1,10:17)])
row.names(eudat3)<-eudat3[,1]
eudat3$RM<- factor(eudat3$RM)
eudat4<-eudat3

#convert to numeric
eudat4$E<-as.numeric(as.character(revalue(eudat4$E, c("VH"=5, "H"=4, "M"=3, "L"=2,
"VL"=1))))
```

```
eudat4$P<-as.numeric(as.character(revalue(eudat4$P, c("VH"=5, "H"=4, "M"=3, "L"=2,
"VL"=1))))
eudat4$C<-as.numeric(as.character(revalue(eudat4$C, c("VH"=1, "H"=2, "M"=3, "L"=4,
"VL"=5))))
eudat4$I<-as.numeric(as.character(revalue(eudat4$I, c("VH"=1, "H"=2, "M"=3, "L"=4, "VL"=5))))
eudat4$A<-as.numeric(as.character(revalue(eudat4$A, c("VH"=5, "H"=4, "M"=3, "L"=2,
"VL"=1))))
eudat4$W<-as.numeric(as.character(revalue(eudat4$W, c("Vlong"=5, "Long"=4, "Med"=3,
"Short"=2, "Vshort"=1))))
eudat4$L<-as.numeric(as.character(revalue(eudat4$L, c("VH"=1, "H"=2, "M"=3, "L"=3,
"VL"=5))))
eudat4$RM<-as.numeric(as.character(revalue(eudat4$RM, c("VH"=5, "H"=4, "M"=3, "L"=2,
"VL"=1))))
names(eudat4)<- c("Taxa", "Effectiveness", "Practicality", "Cost", "Impact", "Acceptability",
"Opportunity", "Reintroduction", "Overall")

# try PCA
res.pca <- PCA(eudat4, scale.unit=TRUE, ncp=5, quali.sup=c(1, 9),  graph=F)

#data are odinal so use nMDS
mydata<-eudat4[,2:8]
d <- dist(mydata) #distance matrix
d<-d+0.001# euclidean distances between the rows
fit <- isoMDS(d, k=2) # k is the number of dim
fit # view results

# plot solution
x <- fit$points[,1]
y <- fit$points[,2]
plot(x, y, xlab="Coordinate 1", ylab="Coordinate 2", main="Nonmetric MDS", type="n")
text(x, y, labels = row.names(mydata), cex=.7)

# put into the res.pca object for plotting
fit$points[,2]<-fit$points[,2]*-1
fit$points[,1]<-fit$points[,1]*-1
res.pca$ind$coord<-fit$points

# plot nMDS for individuals
nMDSplot<-fviz_pca_ind(res.pca, habillage = 9,label="ind",
                    addEllipses =TRUE, ellipse.level = 0.5,title = "", ellipse.type= "t", repel =T, labelsize
= 3, pointsize= 2) +
  scale_fill_manual(values=c("Dark Red", "Red", "Darkorange3", "Green3", "Dark
Green"),guide=FALSE)+
  scale_shape_manual(values=c(16,17, 15, 16, 17))+
  scale_color_manual(values=c("Dark Red", "Red", "darkorange3", "Green3", "Dark Green"))+
  scale_size(guide=FALSE)+
  guides(labels=FALSE)+
  theme_classic()+
  theme(legend.background = element_rect(),legend.position=c(.06, .19),
       panel.border = element_rect(colour = "black", fill=NA, size=1),
```

```
        text = element_text(size=8, family="Times"),
        axis.ticks.length=unit(-0.25, "cm"), axis.text.x = element_text(margin=unit(c(0.5,0.5,0.5,0.5),
"cm")),
        axis.text.y = element_text(margin=unit(c(0.5,0.5,0.5,0.5), "cm"))
  )

nMDSplot

dev.off()

factormap<-fviz_pca_var(res.pca, col.var="contrib", title = "", labelsize = 6 )+
  scale_color_gradient2(low="light blue", mid="blue", high="red", midpoint = 12) +
  theme_classic()+
  theme(legend.background = element_rect(),legend.position=c(.1, .19),
        panel.border = element_rect(colour = "black", fill=NA, size=1),
        text = element_text(size=12, family="Times"),
        axis.ticks.length=unit(-0.25, "cm"), axis.text.x = element_text(margin=unit(c(0.5,0.5,0.5,0.5),
"cm")),
        axis.text.y = element_text(margin=unit(c(0.5,0.5,0.5,0.5), "cm"))
  )

factormap

# Fig 5.1 ----
# Modelling the relationship between environment, area, number of populations and feasibility of
eradication

library(reshape2)
library(Hmisc)
library(gridExtra)
library(grid)
library(ordinal)

#get and structure data
eurm<-eudat0
eurm_scores<- eurm[,c(7,8,9,17,33)]
eurm_scores$Pop_mod<-revalue(eurm_scores$Pop_mod, c("A"="1-3", "B"="4-10", "C"="10+"))
eurm_scores$Area_mod1<-as.factor(eurm_scores$Area_mod1)
eurm_scores$Area_mod2<-as.factor(eurm_scores$Area_mod2)
eurm_scores$Area_mod1<-revalue(eurm_scores$Area_mod1, c("1"="<1ha", "2"="1-10ha",
"3"="10ha+"))
eurm_scores$Area_mod2<-revalue(eurm_scores$Area_mod2, c("1"="<10ha", "2"="10ha-10km",
"3"="10km+"))
eurm_scores$Overall<-factor(eurm_scores$RM, levels=c("VL","L", "M","H","VH"))

#fit model and test for separation of thresholds
xtabs(~Overall + Environment, data = eurm_scores)
eurm_scores$Scenario = paste(eurm_scores$Pop_mod, eurm_scores$Area_mod1)
table(eurm_scores$Scenario,eurm_scores$Environment )
table(eurm$Scen_code,eurm$Environment )
```

```
model2<-polr(Overall~Environment+Area_mod1+Pop_mod,data=eurm_scores)
clm2<- clm(Overall~Environment+Area_mod1+Pop_mod,data=eurm_scores)

#Consider the threshold coeffiecents
summary(clm2)
clm2$cond.H
clm2$Hessian

#test for separation of thresholds (as statistical tests are unreliable)
ctable <- coef(summary(model2))
p <- pnorm(abs(ctable[, "t value"]), lower.tail = FALSE) * 2

(ctable <- cbind(ctable, "p value" = p))# combined table

sf <- function(y) {
  c('Y>=1' = qlogis(mean(y >= 1)),
    'Y>=2' = qlogis(mean(y >= 2)),
    'Y>=3' = qlogis(mean(y >= 3)),
    'Y>=4' = qlogis(mean(y >= 4)),
    'Y>=5' = qlogis(mean(y >= 5)))
}

(s <- with(eurm_scores, summary(as.numeric(Overall) ~ Environment+Area_mod1+Pop_mod,
fun=sf)))

plot(s, which=1:5, pch=1:12, xlab='logit', main=' ', xlim=c(-2.5,5))

eurm_scores$Scenario = paste(eurm_scores$Area_mod1, eurm_scores$Pop_mod)
table(eurm_scores$Scenario,eurm_scores$Environment )

###
newData <- expand.grid(Environment=levels(eurm_scores$Environment),

Area_mod1=levels(eurm_scores$Area_mod1),Pop_mod=levels(eurm_scores$Pop_mod))

predict_overall<-cbind(newData, predict(clm2, newdata=newData)$fit)

predict_overall2<-cbind(newData, predict(clm2, newdata=newData, type="prob", interval=TRUE))
pred_o2<- melt(predict_overall2, id = c("Environment", "Area_mod1", "Pop_mod"))
pred_o2x<-pred_o2%>% separate(variable, c("point", "Code"),sep=4)%>% as.data.frame()
pred_o3<- dcast(pred_o2x, Environment+Area_mod1+Pop_mod+Code~point)
pred_o3$Code<-factor(pred_o3$Code,
                levels=c("VH", "H", "M", "L", "VL")) #("VL","L", "M","H","VH"))

predict_only<- cbind(eurm_scores,predict(clm2, newdata=eurm_scores, type="prob",
interval=TRUE))

predict_class<-cbind(eurm_scores, predict(clm2, newdata=eurm_scores, type="prob",
interval=TRUE))
compar<-cbind(eurm_scores$Overall, predict_class$fit)
```

```
predict_overall$Scenario = paste(predict_overall$Area_mod1, predict_overall$Pop_mod)
pred_over<- melt(predict_overall, id = c("Environment", "Area_mod1", "Pop_mod", "Scenario"))

###
#3 plots

Terr<-ggplot(subset(pred_o3,Environment=="T"), aes(x = Code, y=fit.,colour=Code)) +
  geom_crossbar(aes(ymin = lwr., ymax = upr., fill=Code ),fatten = 3,show.legend=FALSE)+
  scale_colour_manual(values=c("Dark Green", "Green3", "Darkorange3", "Red", "Dark Red"))+
#"Dark Red", "Red", "Darkorange3", "Green3", "Dark Green"
  scale_fill_manual(values=alpha(c("Dark Green", "Green3", "Darkorange3", "Red", "Dark Red"),
0.2))+
  ylab("Terrestrial") +
  xlab("")+
  theme_bw()+
  theme(plot.title = element_text(hjust=0.5)) +
  theme(axis.text.x=element_blank())+
  theme(axis.ticks.x=element_blank())+
  facet_grid(Pop_mod~Area_mod1) #switch="y")

Fw<-ggplot(subset(pred_o3,Environment=="F"), aes(x = Code, y=fit.,colour=Code)) +
  geom_crossbar(aes(ymin = lwr., ymax = upr., fill=Code ),fatten = 3,show.legend=FALSE)+
  scale_colour_manual(values=c("Dark Green", "Green3", "Darkorange3", "Red", "Dark Red"))+
#"Dark Red", "Red", "Darkorange3", "Green3", "Dark Green"
  scale_fill_manual(values=alpha(c("Dark Green", "Green3", "Darkorange3", "Red", "Dark Red"),
0.2))+
  ylab("Freshwater") +
  xlab("")+
  theme_bw()+
  theme(plot.title = element_text(hjust=0.5)) +
  theme(strip.text.x=element_blank())+
  theme(axis.text.x=element_blank())+
  theme(axis.ticks.x=element_blank())+
  facet_grid(Pop_mod~Area_mod1) #switch="y")

Marine<-ggplot(subset(pred_o3,Environment=="M"), aes(x = Code, y=fit.,colour=Code)) +
  geom_crossbar(aes(ymin = lwr., ymax = upr., fill=Code ),fatten = 3,show.legend=FALSE)+
  scale_colour_manual(values=c("Dark Green", "Green3", "Darkorange3", "Red", "Dark Red"))+
#"Dark Red", "Red", "Darkorange3", "Green3", "Dark Green"
  scale_fill_manual(values=alpha(c("Dark Green", "Green3", "Darkorange3", "Red", "Dark Red"),
0.2))+
  ylab("Marine") +
  xlab("")+
  theme_bw()+
  theme(strip.text.x=element_blank())+
  facet_grid(Pop_mod~Area_mod1)# switch="both")

grid.arrange(Terr, Fw, Marine, top="Total Area", bottom="Overall Score", left="",
right="Populations", ncol=1)
```

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