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Increasing landscape connectivity: evaluating the risks that this will encourage invasive non-native species

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Foreword

Natural England commission a range of reports from external contractors to provide evidence and advice to assist us in delivering our duties. The views in this report are those of the authors and do not necessarily represent those of Natural England.

Background

Increasing landscape connectivity to enhance the resilience of landscapes to climate change is a key adaptation principle, and is one of the most common forms of adaptation on the ground. Joining up landscapes was also one of the main findings of the Lawton Review as a means of ensuring coherent ecological networks.

At the same time an increasing threat from non-native invasive species has been identified as a developing threat to the natural environment and Natural England's objectives from climate change.

Research into the spread of non-native species has demonstrated that certain aspects of connectivity may lead to an increased risk of spread, whilst in certain circumstances isolation and the introduction of barriers to spread (ie reduced connectivity) can be used as a conservation measure.

Conversely increasing connectivity through enhancing the habitat matrix has been suggested as a means of reducing the risk of invasive species by enhancing the resilience of native communities.

This project was therefore commissioned to investigate the potential conflicts and trade-offs between increasing connectivity to enhance resilience through enabling species to track climate change and reducing resilience through encouraging the spread of invasive species.

The findings of this research will help inform the design of ecological networks which in turn will help determine how and where we prioritise interventions such as habitat creation under the New England Land Management scheme, and biodiversity offsetting.

Natural England Project Officer - Simon Duffield, Mail Hub Block B, Whittington Road, Worcester, WR5 2LQ **simon.duffield@naturalengland.org.uk**

Contractor - Sarah Knight, FERA, York **www.fera.defra.gov.uk/**

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Increasing landscape connectivity: evaluating the risks that this will encourage invasive non-native species




By: Sarah Knight*, Dr Larissa Collins*, Simon Conyers*, Dr Andrew Crowe*, Dominic Eyre*, Dr Dave Parrott*, Dr Sugoto Roy*, Kate Somerwill* Dr Jamie Williams[†] and Dr Nigel Boatman*

*Fera
[†]Environment Systems

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Executive Summary

1. Non-native species are those that enter regions outside of their natural range, along a number of different pathways, involving either deliberate or accidental release through human activity. Those species that spread and impose detrimental impacts on native species or the country's economy or health are called invasive non-native species.

2. Increasing connectivity of the landscape is known to be beneficial to the conservation of native species in many situations as it promotes coherent ecological networks that allow movement through the landscape. The importance of ecological networks increases when considered in light of climate change as they are believed to facilitate species range shifts.

3. It is expected that increasing the connectivity of the landscape will increase the resilience of England's biodiversity to climate change, however this increased connectivity can also reasonably be expected to improve the dispersal of non-native species, and increases the risk of them becoming invasive.

4. Invasive mammals and birds are less influenced by landscape connectivity because of (i) the scale of their natural movements, (ii) behavioural flexibility, and (iii) the ability to exploit anthropologically-disturbed habitat. However, high levels of connectivity may still facilitate dispersal, particularly for habitat specialists.

5. In general, non-native invasive species are able to exploit disturbed habitats to a greater extent than native species are. The group for which this is especially true is plants. Invasive plants are known to preferentially invade edge habitat (for the reasons stated above) particularly those near human-disturbed habitat as these tend to be sources of additional non-native species.

6. Large patches may benefit natives as they allow for heterogeneity and therefore diversity and are more likely to be more stable. This means that they are more buffered from the invasion process. However the reverse may be true of smaller patches as they may be easier to invade as the edge of these patches, the most vulnerable zone, represent a greater area of the patch.

7. The creation of new habitat that requires the creation of disturbance is the most vulnerable habitat to invasion. It will be important to ensure that these are nutrient-poor to reduce the advantage invasive non-natives may have in these environments.

8. Improving the condition of semi-natural habitat by reducing other stressors may not have an impact on invasive invertebrates as they often attack both healthy and stressed vegetation.

9. Invasive mammals, especially those that are generalists, are able to exploit disturbed habitats.

10. Invasive amphibians often fare better in poor quality and disturbed habitats in comparison to native species so improving the quality of sites may help favour natives over invasives.

11. Habitat specialists irrespective of taxa are more influenced by habitat quality and connectivity than generalists.

12. Additionally, whilst landscape connectivity (including habitat coridors, patches and mosaics) is of huge importance for some species (such as many species of plants and insects) it becomes less important for others (mainly birds and mammals). Where connectivity is important, patch size and quality are important factors in addition to distance between them. Understanding trade-offs, likely impact of landscape design and prioritisation of invasive species that have a high potential to cause damage are important when considering design of future landscapes.

1. Introduction

Invasive non-native species (INNS) are those that have been introduced to a country, either deliberately or accidentally. They represent one of the leading threats to natural ecosystems and biodiversity in the UK and globally, along with habitat loss (Atkinson 1996; Diamond 1984; Vitousek *et al.* 1997).

Generally non-native species in the UK cause no harm and survive alongside native species without any need for intervention (Manchester and Bullock 2001). However, a subset has become widespread as well as becoming sufficiently abundant to interfere with either natural or managed systems such that action is required to mitigate these effects. The impacts of INNS can be grouped into five categories: consumption through predation or herbivory, resource competition, introduction of diseases, interbreeding and disturbance of the environment (White & Harris 2002). These impacts in turn lead to a loss of biodiversity through direct loss of species or hybridisation. Along with their impacts on biodiversity, INNS also have major economic, agricultural and health impacts. Climate change is likely to increase the impact of invasive species through more species arriving in the UK, facilitation of range expansion in those already present and through both new and established species becoming invasive (Natural England 2012).

This project was commissioned by Natural England to broadly assess the ease with which landscapes and changes facilitate the development and movement of populations of invasive species, or conversely, facilitate their management. Conectivity is a crucial factor influencing movement through landscapes. The greater the connectivity, the greater the ease of movement is likely to be, though this varies between different taxa. Connectivity may be provided by continuous habitat ‘corridors’, small sites acting as ‘stepping stones’, or a mosaic of habitats (Lawton *et al.* 2010). Patch size and quality are also important factors within ecological networks. The influence of landscape on dispersal will depend on a number of traits that are likely to be largely similar for species within a taxon. For example, mammals and birds may not require intimate or immediate connection of suitable habitats as they possess greater vagility than invertebrates or plants.

The potential to use landscape to manage non-native invasive species in England has not been fully studied previously. Landscape scale gap creation (i.e. large breaks in the preferred habitat) may have potential to manage invasive species (With 2002; 2004), particularly plant and insect populations, but could have limited impact on the movement of invasive mammals and birds. In these cases, an understanding of landscape pattern, and the potential for policy tools to modify this pattern may permit the development of strategies to slow spread, minimise the locations at which control operations are necessary and increase the likelihood of maintaining cost-effective policies of zero-density management at key sites of conservation concern.

The aims of this project are:

- To identify which non-native species have the propensity to become invasive in the light of climate change in England.
- To identify the land management practices (especially those that are supported by Environmental Stewardship options), landscape design and the design of ecological networks that might facilitate invasiveness of species.
- To explore the practical considerations of the potential trade-offs that might need to be made by reference to a small number of case-studies.
- To provide recommendations for the design of networks to facilitate the conservation of native species while reducing the potential impacts of non-native invasives.

In order to achieve our goals a four-stage approach is taken, consisting of; i) a literature review (Work Package 1), ii) a species level analysis of key dispersal traits (Work Packages 2 and 3), iii) a simple analysis of English landscapes focussing on connectivity (including the impact of Environmental Stewardship schemes) (Work Packages 4, 5 and 6), and iv) a report to integrate these threads (Work Package 7). The literature review and species-level analysis are summarised below and presented in full in appendices A and B respectively.

2. Summary of literature review and species selection process for case studies

The main results from the literature review and species selection are discussed below. The summary is split in to six sections (landscape, climate, key traits, species selection, additional factors and conclusions).

2.1. Landscape

The impacts of landscape vary between and within all of the species groups considered in this project. Between groups, birds and mammals seem to be less influenced by landscape connectivity in terms of distance between habitat patches, with plant and herptile species more affected. Additionally, it is apparent that rather than being distinct independent factors of ecological networks; bigger, better, more and joined up work together to create a connected landscape.

2.1.1. Bigger

For birds, mammals and herptiles the larger the size of the fragment the larger the population that fragment can support. This is true of native and non-native species and is likely caused by the link between increased fragment size allowing increased heterogeneity and a greater chance of a species finding a preferred habitat. There is also some evidence, particularly for birds that smaller sites, that by definition have a lower area to edge ratio than larger sites, also have higher risks of predation. This is especially true if the fragment edge is near highly productive land that supports generalist predators as is the case in landscapes associated with human activity (e.g. agriculture). This could benefit some non-native generalist predators allowing smaller sites to be more readily invaded.

Invasive plant species are also found to be more common in edge habitats due to the increased light, space and soil moisture found in such areas and so are more able to invade smaller sites than larger. Additionally, larger sites are thought to be more ecologically stable due to their increased heterogeneity. This stability makes large sites more suitable for native species than non-native that typically prefer disturbed habitats.

Invertebrate species that are host specific will benefit from increases in the habitat type that supports their host plants; however this is not the case for more generalist species able to use a wide variety of plants as hosts. These species are limited by the distribution of a diverse range of host plants and not by a particular habitat type, so any increase in habitat size is unlikely to have a large impact on population numbers.

2.1.2. Better

In general, non-native invasive species are able to exploit disturbed habitats to a greater extent than native species are. The group for which this is especially true is plants. Invasive plants are known to preferentially invade edge habitat (for the reasons stated above) particularly those near human-disturbed habitat as these tend to be sources of additional non-native species. This is also true of some invasive bird species that are able to thrive in such human-disturbed landscapes. Invasive plants are able to disperse through the landscape using disturbed habitat corridors (railways, roads etc.). Landscape configuration with its matrix of transport corridors and edges is of primary importance to the presence and establishment of alien plant species while local scale factors such as vegetation structure and soil resources are of key importance for their population growth (Vila and Ibanez, 2011). It may be possible to use ecological restoration to decrease the invadability of some degraded habitats.

For mammals and herptiles the degree to which a species can exploit degraded habitat is dependent on their degree of specialism. Habitat specialists need good quality, connected habitats in their native ranges and the lack of this for some species in the UK may explain their slow spread (e.g. Chinese water deer). Generalist species are able to exploit degraded habitat and may show no habitat preference at all, while intermediate species benefit from high quality habitat but are able to use low quality areas. However, the effects of quality on herptiles are difficult to disentangle from the impacts of predator densities in disturbed and non-disturbed habitats.

Invasive insects are also more able to exploit degraded habitats than native species and in general insects can use highly disturbed, productive land to increase their populations before moving to areas of higher quality. However, for the species selected in this project, quality is unlikely to have a large impact on their distribution.

2.1.3. More

Mammals, herptiles and birds show similar reactions to an increase in the number of habitat patches available; more sites would lead to an increase in the number of individuals in a population. Meta-population dynamics are impacted by an increase in the number of sites available and this is particularly true for amphibians. For birds, there is some evidence showing that population persistence is more influenced by isolation from surrounding habitat patches than by patch size alone possibly because the high mobility of this group allows individuals to incorporate several habitat patches into their home range.

It is difficult to extract the impacts of increasing the number of sites away from increasing connectivity between sites as additional sites can act as stepping stones allowing spread across the landscape (as is the case in invertebrates with low dispersal distances).

The creation of new habitat, that requires a disturbance event, is the time at which an area is most at risk of being invaded by disturbance-loving non-native plant species. It would be important to ensure that any new areas of habitat are nutrient poor to reduce the competitive advantage of non-native species.

2.1.4. Joined Up

Due to their high mobility birds and mammals are less constrained by distance between patches than the other taxa considered. In the UK invasive birds are largely unconstrained but this is not the case with native forest specialists in tropical regions (and with some native woodland specialists in the UK). The impacts of landscape connectivity on non-native invasive species present in the UK have not been fully researched but, due to their large dispersal distances, are likely to be minimal. While not limited by connectivity, mammal species would benefit from habitat connectivity particularly habitat specialists as it would facilitate spread. Additionally, habitat corridors that join patches are sometimes used as prey-rich habitat and so may attract non-native predatory mammals.

Herptiles are highly limited by fragmentation but as invasive non-native species are highly adaptable, they can cross and / or persist in a wide range of habitats. Invertebrates are also limited by fragmentation specifically the distance between patches of suitable habitat / or host plants. However, the importance of fragmentation decreases with increasing dispersal distance of the species. Increasing landscape connectivity will have some benefit both these groups of invasive species.

Plants are perhaps most influenced by landscape connectivity. In general, better connected habitats benefit native species and corridors do not necessarily have to be vulnerable to invasion by non-native species if the quality of them is sufficiently high (high quality corridors allow increased colonisation by increasing seed deposition and within patch recruitment and potentially by altering seed predation). Poor quality corridors, particularly those with a close association to human activity, promote the spread of non-native species.

2.2. Climate

Climate change is likely to cause an increase in the number of invasive species in the UK. For all groups considered, climate change is expected to increase the northern range edge of warm adapted species and increase over-winter survival rates. However, for many plant species a cold period during winter will still be needed to allow seeds to germinate. Additionally, an increase in temperature could cause an increase in breeding potential, particularly in herptiles where non-natives, currently unable to breed in the UK, may start to produce viable eggs.

Several groups are likely to be impacted by alterations in trade, with new exotics entering the trade routes meaning more species are likely to escape or be deliberately released. It may also become easier to keep certain exotic pets outside, further increasing the chances of accidental release.

Aquatic amphibians and mammals are expected to benefit from the increase in precipitation in winter, through increasing food availability and habitat connectiveness (allowing greater movement through the landscape). Alterations in movement may also be seen in invertebrates, where there may be an increase in the number of days that are suitable for migration.

Many mammal and bird species will adapt to any changes in climate and so any impacts are expected to be of a smaller scale than those seen for other taxa; however extreme weather events may reduce seasonal survival parameters of some species.

2.3. Key traits

Across all groups, 21 key traits were highlighted from the literature and these are shown in Table 1. The traits are highlighted if they are considered **key** for the species within a group to become invasive. The most important of these in terms of the impact of ecological networks is dispersal ability. Others that may be relevant include behavioural flexibility, high fecundity, and r-slected species (good at colonising).

Table 1. Key traits that impact the invasiveness of non-native species.

Key Trait	Birds	Plants	Invertebrates	Mammals	Herptiles
Aesthetically attractive to Humans		■			
Behavioural flexibility	■				
Body mass (large body masses)	■				
Multiple broods per season	■				
High dispersal ability		■			■
Ecoclimatic match	■	■	■	■	■
Existing species assemblages			■		
Fast growth/Competitive ability		■			
Germination without pre-treatment		■			
High fecundity		■			
Human commensalism / activity / trade	■	■	■	■	■
Juvenile development (short juvenile development time)	■	■		■	■
Large geographic native range	■			■	
Migratory behaviour (non migratory)	■			■	
Phenotypic plasticity		■			
Propagule pressure	■	■	■		
r – selected species*				■	
Self-compatibility		■			
Sexually selected traits/ plumage type	■				
Successfully invaded elsewhere				■	
Water availability					■

2.4. Species selection

All of the selected invasive avian species have the capacity to move *across* landscapes at the scale of the project's case studies (and larger-scale). The factor determining whether any of these species would settle, establish and spread *within* a defined area would be the availability of suitable habitat to encourage the initial settlement of dispersing or prospecting individuals. For example, cavity nesting species (e.g. ring-necked parakeet) would require the presence of mature trees in order to establish breeding sites. In the absence of mature trees with suitable cavities, parakeets might utilise the area for alternative functions such as foraging or roosting (dependent on suitable habitat) or else pass over the area to locate such resources elsewhere.

All of the selected avian species are considered generalists either through possessing a wide diet and/or being able to utilise a range of different habitats. Although a number of the species are generally sedentary they do illustrate dispersal behaviour (which can be extensive) in response to seasonal cues (e.g. movement between breeding and moulting areas) and/or in response to the availability of water or other resources, or disturbance.

There are many more potentially invasive species of plant than any other taxa considered in this report, several of them having the potential to radically alter any ecosystem they were to enter. The example species selected here are representative of the different non-native plant species present in Britain. Several are likely to be heavily impacted by any changes in landscape or climate. There are a disproportionate number of trees and shrubs but these have been shown to represent the largest group of invasive plants amongst environmental weeds (Hulme *et al.,* 2012). They may also represent the largest threat to biodiversity as they may be early scrub/woodland successionists and therefore able to invade grassland and heathland. However in field surveys they are the most difficult categorise as it is not always obvious whether they were planted (Hill *et al.*, 2009).

Most of the invertebrate species included in the trait matrix are incapable of dispersing over great distances without human assistance. They generally move a few kilometres each year at most. However, some moth species, including the oak processionary moths have been recorded as migrants, having flown to mainland Britain from continental Europe. Generally however, human movement pathways, particularly of timber and wood products and garden plants, are likely to result in much more rapid spread than the natural spread these species could achieve unassisted. The Oak Lace Bug is assisted in its spread by motorways. It is moved along on the air currents caused by traffic.

All of the herptiles species and many mammal species selected for this study are involved in the pet trade and this is their main transport pathway into Britain. With the exception of Chinese water deer none of the mammal species selected are present in the UK outside of captivity and two species, the coypu and Muskrat have been eradicated in the last century.

The majority of herptiles species selected were in category A of the Belgian risk scheme. The Alpine newts and wall lizards were selected because of their susceptibility to habitat alteration and climate change. All of the species (except Alpine newts) are expected to benefit from increased climate change with turtle and amphibian species benefiting from increased precipitation.

2.5. Additional factors

As globalisation increases, the number of invasives species in a region also increases, and in fact it has been shown that the number of invasive species can be linked directly to a proxy for the increase in global trade such as a country's measures of GDP (Hulme 2009).

Invasive species often do not spread alone. For example, species may be transported together, as with Alpine newt eggs being spread accidentally through the trade in aquatic garden plants, as discussed elsewhere. Also, invasive species often spread invasive diseases with them too (Crowl *et al.* 2008). Similarly, many invasive plant species facilitate the spread of other invasive plant species (Simberloff & Von Holle 1999). Through even more complex interactions, in parts of the world where native plant species cannot tolerate predation by invasive herbivores, herbivory-tolerant invasive pants are favoured and facilitated. This is particularly true of oceanic islands where mammals have been absent until recently (Asner *et al.* 2008).

Something that cannot be predicted or measured is deliberate movement of animals by people within a country. This is believed to be how invasive edible dormouse have been spread around Britain, as people have trapped them in their roof spaces and released them far from where they were caught (Morris & Morris 2011).

The confounding effects of invasive species exploiting manmade structures designed to ameliorate against climate change is difficult to predict. Cane toads for example are able to spread through the landscape in otherwise inhospitable areas of Australia by exploiting water storage devices (Florance *et al.* 2011).

As discussed earlier, there are some elements of the way invasive species spread which are complex, hard to map and difficult to assess. The pet trade, in particular Internet trade, is hard to assess spatially as items are sold across the world regardless of provenance. In addition the water plant trade for garden ornamentals may harbour the eggs of certain amphibian species causing an undetected spread and it is believed that some of the Alpine newt populations in the UK spread thus (Fisher *et al.* 2009) (Trent Garner, Evolution & Molecular Ecology Theme Leader, Zoological Society of London, Pers. Comm.).

As global climates get warmer, the number of species that would become easier to keep and trade will increase, in turn increasing the range of new, potentially invasive species that may threaten the UK's biodiversity (Salinas *et al.* 2011) and only frequent horizon scanning exercises such as the one carried out by Parrott *et al.* (Parrott *et al.* 2009) can keep abreast of this.

Finally, non-native species are known to show atypical reactions when placed in new areas. For example, striped skunks, unlike other skunk species, in their native range often avoid edges and disturbed habitats. In areas they have been introduced to they can react atypically and can often invade urban and other disturbed habitats (Lariviere & Messier 2000; Anon. 2009; Ordenana *et al.* 2010). Predicting the reactions of non-native species to changing situations is therefore difficult due to the adaptability they often exhibit.

2.6. Conclusions

Introductions mediated by man are either deliberate (as with many plant, mammal and bird species) or accidental, as in the case of several insect invasives, and similar biotic, abiotic and in some cases behavioural traits impact the ability of a non-native species to disperse through the countryside. In many cases, differences in these traits between taxa and species mean that a landscape feature that would be a barrier in one species is a corridor in another (for example, a major road may be a barrier to certain amphibian species, but could be a 'corridor pathway' for some invasive plant species).

Examination of the ecological traits of the selected invasive non-native avian and mammal species indicates that potential future changes in the nature and extent of landscape connectivity (increases in area, quality and connectivity of conservation areas) does not impose a barrier to their movements and spread; especially at the spatial scale of the case studies in the present investigation (1500km^2). Three traits in particular enable invasive birds and mammals to overcome potential barriers in the landscape: (i) the scale of natural movements, (ii) behavioural flexibility, and (iii) the ability to exploit anthropologically-disturbed habitat.

Insects are not limited by the size of areas as they are reliant on host plants rather than habitats; they are able to use any habitat the host plant is present in. They can exploit degraded as well as high quality habitat due to their adaptable nature and are not limited by habitat connectivity.

Plants and herptiles seem to be impacted by landscape factors to a greater extent than other species but sometimes in opposing ways. Larger areas would be a benefit to herptiles but would restrict the spread of invasive plants (due to a lack of edge effects). Increasing the quality of an area benefits herptiles to varying degrees depending on their habitat specificity but has a negative impact on invasive plants that prefer lower quality habitats. More sites will benefit herptiles and plants as will increased connectivity. Plants may benefit additionally from habitat creation if an element of disturbance is required.

Propagule pressure is thought to be of importance across several species groups and so landscape characteristics that enhance this are likely to increase the spread of invasive species. Such characteristics would be high area to edge ratios (many species invade habitats from the edges inwards) and proximity to human disturbed areas (often a source location for invasive species due to trade / human activity).

Several recommendations have already started to emerge from the literature review. These are as follows:

- Bigger:
  - Larger reserves increase native species numbers whilst edge habitat is at a higher risk of invasion. Therefore larger areas, with lower area / edge ratios are more able to reduce non-native spread than several smaller areas.

- Better:
  - Increasing the quality of sites, through ecological restoration could increase the sites resilience to invasion.
  - Ensure connecting corridors are of a high quality to reduce the risk of invasion by non-native plants.

- More:
    - Ensure that any habitat creation that requires disturbance is carried out on nutrient poor areas to reduce the competitive edge that non-native species may have.

- Joined up:
    - Any additional sites or corridors should be placed away from human disturbed areas and disturbance corridors such as railways, roads and canals as these disturbed areas are more easily exploited by invasive rather than native species. This has implications for linking urban and semi / non-urban areas.

- General:
    - Continued surveillance of ecological networks to monitor for presence of invasive species

Invasive species are highly adaptable and can exhibit atypical reactions when introduced to new areas. This makes invasive species' reaction to novel situations unpredictable. This review has summarised current knowledge and expert opinion on the likely reactions that the selected non-native species will have to changes in climate and landscape. However they should be viewed in light of this high level of adaptability. Understanding trade-offs, likely impact of landscape design and prioritisation of invasives that have a high potential to cause damage are inevitable when considering design of future landscapes.

3. Trade off table and species selection

3.1. Environmental Stewardship

The use of agri-environment schemes was highlighted as a crucial way to enhance landscape connectivity in England (Lawton 2010) through increasing the quality of available habitat and creation of new habitat patches and corridors. However, their impact on invasive species is not well researched. Donald & Evans (2006) suggest that increasing connectivity through agri-environment schemes rather than by creating artificial corridors may potentially mitigate the effect of increasing the ease of dispersion for invasive plant species by creating a coherent ecological network, free from the disturbance favoured by non-native invasive species.

Environmental Stewardship (ES) contains options both for the creation and maintenance of important semi-natural habitats. We made use of data from the Natural England funded project ‘Use of Environmental Stewardship Options to Support Ecological Networks’ where all Entry Level Stewardship (ELS) and Higher Level Stewardship (HLS) options had been assessed with regards to their impact on the following 24 habitat types:

- Bog
- Acid Grassland
- Broadleaf and Mixed Woodland
- Calcareous Grassland
- Coniferous Woodland
- Cropped Areas
- Ditches
- Farm Buildings
- Fen Marsh Swamp
- Field Margins and Uncropped Farmland
- Heathland
- Hedgerows
- Improved grassland
- Lowland Meadows
- Moorland
- Neutral Grassland
- Ponds
- Rivers and Streams
- Salt Marsh
- Sand Dunes and Coastal Cliffs
- Scrub
- Stone Wall and Earth Bank
- Traditional Orchards
- Upland Hay Meadows

Options could have either a positive or negative impact on the quality or quantity of habitats. For example converting coniferous woodland to heath would have a positive impact on heath but a negative impact on coniferous woodland habitat. Using the information gathered in the literature review and opinions from the taxa experts, species and ES exprets scored the impact that these positive or negative changes in habitat had for each of the 53 species selected. A scoring range of 3 to -3 was used: 3 indicating that the species will majorly benefit from this option, 0 that the species will be unaffected by it, and -3 indicating the species will be majorly and detrimently impacted by it. This feeds into the case studies to provide insight into how ES will change the invasion process in different landscapes.

This table also contains the corresponding data regarding native species which have been provided by the Project Team of the Defra project BD50101: 'Synthesising Review' of the Use of Environmental Stewardship for restoring, maintaining and enhancing a coherent ecological network in England. This provides information on the trade-offs between native and non-native species in ES management.

Specific options covering supplementary bird feeding could be of disproportionate benefit to certain bird species.This is incorporated in the scoring for the habitats within which these options are present. Invasive species are highly adaptable as has been shown throughout the literature review. Due to this they are likely to be capable of exploiting many sources of food including those provided by these Environmental Stewardship options. The impact of these options on invasive bird species should be assessed as it may be that removal of the options has little impact on non-native species due to their adaptable nature, but could have a marked impact on more specialised native birds.

3.2. Birds

The categories of ES with the highest positive impact on non-native birds are improving cropped areas, field margins and uncropped farmland, and rivers and streams. Others which will have a large positive impact on a single or small number of species include woodland, salt marshes and moorland. As previously discussed, habitat improvement will have only a limted affect on non-native bird species as generally they are able to disperse to areas that are more favourable.

3.3. Plants

The impact of habitat improvement on plants has a much more complex pattern. For every category in the trait table for improving a habitat via ES, some species are positively impacted, others are negatively impacted. This impact is often a very strong response to a change in habitat (a score of 3 or -3). For example, by improving heathland there will be a very strong negative impact on Himalyan balsam but a very strong positive impact on Maritime pine and Holly oak (though a requirement that these species be controlled within an Environmental Stewardship option could mitigate this impact). The ideal situation is one where an improvement in habitat causes a negative impact on non-native species; this occurs to an extent by implementing ES options that have a positive impact on cropped areas. This has a negative impact on all of the species in the trait table apart from *Senecio inaequidens* and Giant goldenrod.

3.4. Invertebrates

These non-native invertebrates are largely unaffected by the improvements to habitat brought about by ES. The two categories that will affect this group are woodland and hedges – an improvement to these will have a positive impact on over half of the invertebrates in the table. Therefore any ES option that has a negative impact on woodland and hedges will have a negative impact on these non-native invertebrates.

3.5. Herptiles

There are several habitats that, if improved, will have very strong positive impacts on the non-native herptiles in this study. These are bog, ditches, ponds, rivers and streams, and saltmarshes. These are obviously the habitats characterised by a larger proportion of water alongside land. The Burmese python and the Wall lizard are exceptions that will both benefit from improvements in scrub too. To avoid increasing the possibility of increasing populations of non-native herptiles, ES options that do not improve moist habitats should be implemented.

3.6. Mammals

Like plants, mammals are much more varied in their response to habitat improvements across the taxa. Habitats that strongly impact mammals are woodland, cropped areas, field margins and uncropped farmland, scrub and orchards.

3.7. Summary

Overall, the habitat improvements that will most benefit non-native species (total score of 30+) across the entire trait table are cropped area, ditches, field margins and uncropped farmlands, hedgerows, rivers and streams, scrub and traditional orchards. Therefore to reduce the potential for invasion by non-native species, improvements should occur in other habitat types, or a prioritised trade-off between benefits and likely invasion needs to be considered.

Implementing ES options to improve cropped areas provides a positive impact on all taxonomic groups in this study, with the exception of plants, where it has a negative impact on the majority of species, and herptiles where they remain indifferent. Therefore to reduce the likelihood of invasion by non-native birds and mammals, implementing ES options that negatively impact cropped areas would be suggested.

Implementing ES options that have a negative impact on habitats with a high water content (e.g. rivers, ponds, bogs) will reduce the potential invasiveness of most herptiles. However that will provide habitat that is suitable for the invasive plants examined in the project. The same applies to native species. For example, 38 native species will benefit from improvements to ponds. A trade off would have to be made depending on the geographic location and the habitats and species present.

Improvements to hedgerows increase the invasiveness of many non-native species in this study. However it is also an important habitat for native species, so a prioritisation and trade-off must occur. For example, many species that are found in hedgerows also live in woodlands. Improving woodlands may benefit native species whilst having less of a favourable impact on non-natives.

Overall, there are no ES options that completely favour native species and are completely unfavourable for non-natives. For specific detailed examples using the selected case study species see Section 4.6. The trait table and trade-off table provide a useful summary for the evaluation of trade-offs between different invasive species and also between native and non-natives.

4. Case Study

4.1. Study site

The Dorset Area of Outstanding Natural Beauty (AONB) was selected as a case study site to test the findings of the literature review and trait based analysis. This region was identified as being important for conservation and contains habitats and species of relevance for this project. The region has full National Vegetation Classification Phase 1 data coverage and Environment Systems have previously worked with Dorset AONB on the cross-border Cordiale land management project. Dorset is a county with a rich biodiversity, landscape and quality of life, reflected in the AONB status given to the coastal part of the county. The county also contains parts of the Cranborne Chase and West Wiltshire Downs AONB.

The variety of habitats and land covers includes areas of woodland, agricultural land, urban regions and coastal zones. Much of the biodiversity is linked to a range of habitats across the county, which, whilst previously much more extensive, remain as a core of high biodiversity and the basis of a functioning ecological network.

The county also contains part of the Dorset and East Devon Coast World Heritage Site, the Dorset section being 114km long. In addition there are 76kms of Heritage Coast, 141 SSSIs, 49 Local Nature Reserves and 1267 Sites of Nature Conservation Interest. The county also includes areas that are designated as Special Areas of Conservation (SAC), Special Protection Areas for Birds (SPAs) or Ramsar sites under European and International legislation.

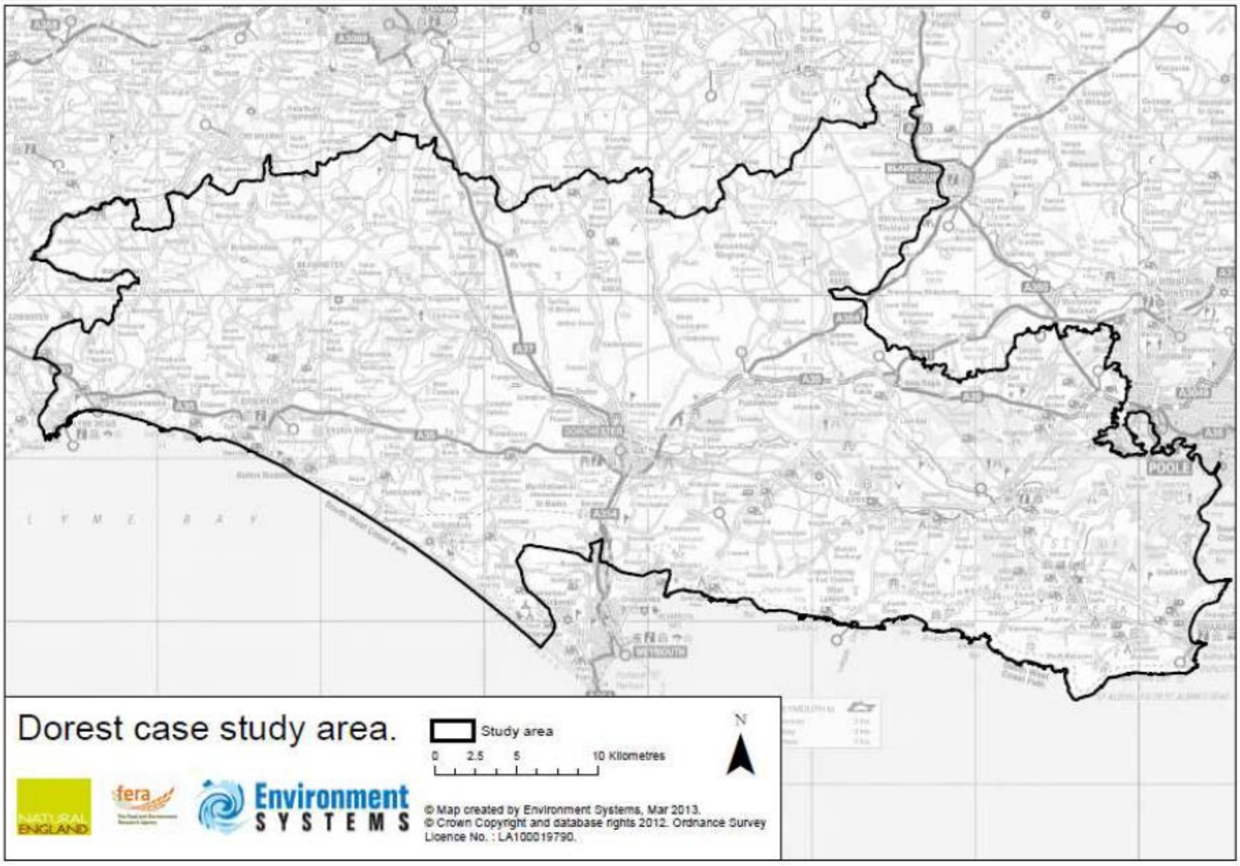


Figure 1. The Dorset AONB study site.

The project aims, selected species and methodologies applied in the project were shared with staff from Dorset AONB. This was to gather their opinions regarding the project and to ensure that the network selections were suitable.

4.2. Species selection

To identify a set of suitable species to analyse in this section, a principcal component analysis (PCA) was conducted. This was carried out using all of the data in the trait table excluding habitat preferences; inclusion of this trait strongly groups the species and overshadows any other possible influencing factors.

The species have been grouped by the PCA into 4 distinct groups (see Figure 2). The results from the P.C.A. were plotted using a biplot, which plots the P.C.A. loading (as arrows). The length of the arrows represents the variability in a component. The angles between arrows represent the correlation between those species. If the arrows are in the same direction, the species are positively correlated, if they are in opposite directions, they are negatively correlated.

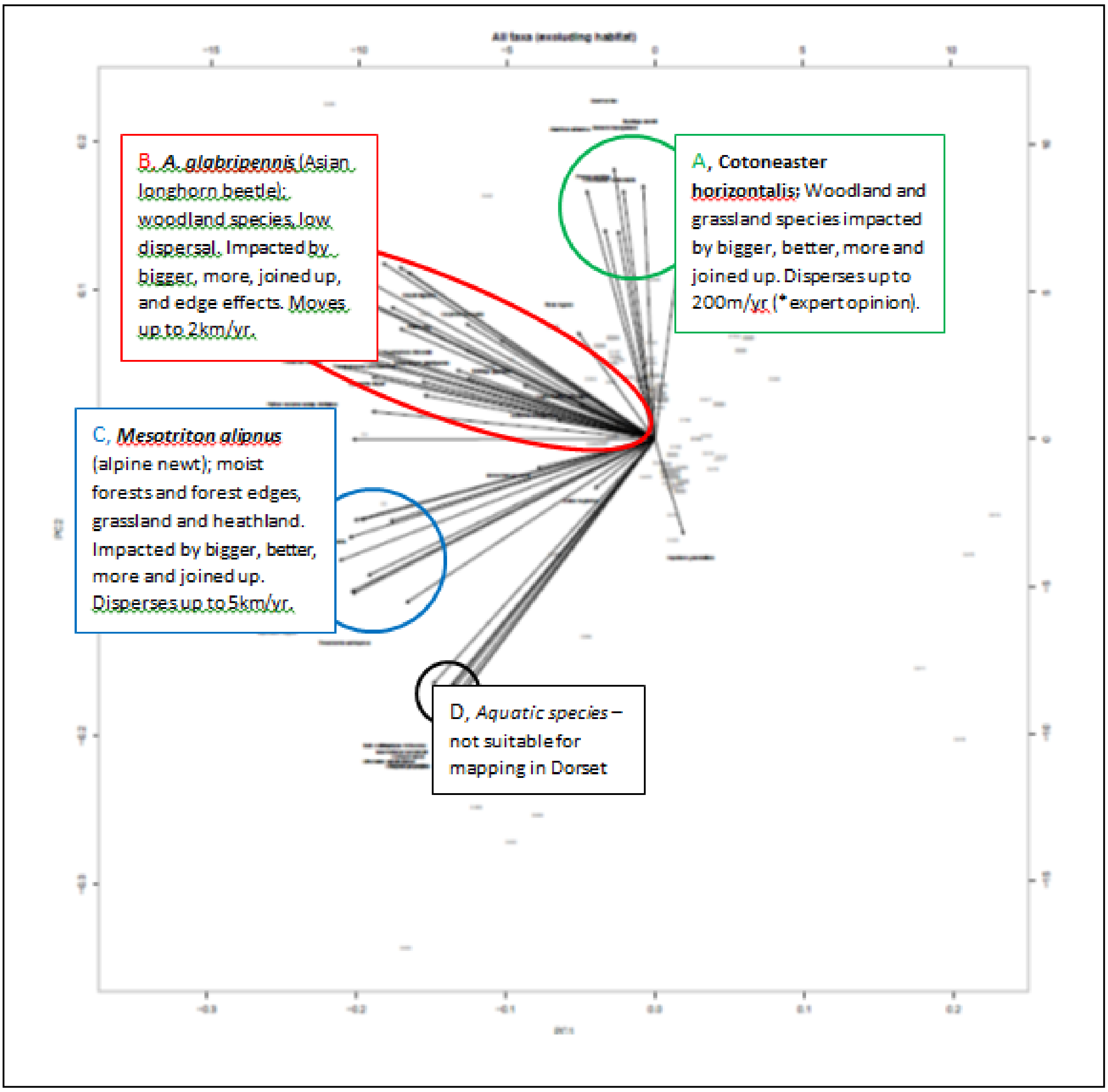
The plot groups the species which prefer traits of similar type and provision. For example, *Pecari taiacu* and *Procyon lotor* are very closely positioned on the plot, therefore are likely to prefer similar traits of similar provision. *Macrochelys temminckii* and *Pinus pinaster* are not closely positioned and are likely negatively correlated, therefore traits that favour *Macrochelys temminckii* are likely not to favour *Pinus pinaster.*

Of the four groups emerging from the analysis, Group A are influenced by e.g. habitat corridors, such as motorways and train tracks, group B are influenced by e.g. improvements to woodland and hedgerows, group C are influenced by e.g. improved habitat quality and group D are influenced by e.g. proximity to water and improvement in glasslands. The nature of these distinct groups suggests that a one indicator species could suitably represent the traits of that group, for modelling purposes (as highlighted in the boxes).

A selected species for 3 of these groups was chosen given the selection critera below (one plant, one insect and one herp). Mammals and birds were excluded as their dispersal distances are too great (Chinese water deer excluded because there are not many ways to increase a wetland habitat in Dorset). The 4th group was excluded because all of these were aquatic species which are not suitable for mapping in Dorset.

Criteria for species selection are:

- small movement distances
- good indicator species (derived from the PCA)
- we have lots of knowledge of
- impacted by habitat connectivity / ecological networks / climate change



B, *A. glabripennis* (Asian longhorn beetle); woodland species, low dispersal. Impacted by bigger, more, joined up, and edge effects. Moves up to 2km/yr.

A, **Cotoneaster horizontalis**; Woodland and grassland species impacted by bigger, better, more and joined up. Disperses up to 200m/yr (*expert opinion).

C, ***Mesotriton alipnus*** (alpine newt); moist forests and forest edges, grassland and heathland. Impacted by bigger, better, more and joined up. Disperses up to 5km/yr.

D, *Aquatic species* – not suitable for mapping in Dorset

Figure 2. The PCA groupings, the selected species and the reasons for this.

4.3. Modelling methodology

The process of modelling the potential spread of invasive species through the landscape involved two steps. Firstly, the creation of three different ecological networks that follow the Lawton principles. Secondly, modelling the predicted spread of a given indicator species for each of the three ecological network scenarios.

Stage 1 & 2 – All networks

All of the networks were based on the habitat map previously created by Environment Systems for the Dorset AONB. This habitat map was created using a combination of ecological knowledge of the target land cover classes and remote sensing expertise, to create a computer algorithm to identify and delineate the land cover classes within an image. The algorithm relies heavily upon the distinct spectral reflectance of the classes found within earth observation data. The target land cover classes that are prioritised for delineation are outlined in Table 2.

Table 2. The habitat classes mapped in the survey. Broad habitat classes were identified, with the grassland split into several classes based on vegetation productivity, species richness and landscape function.

Target land covers	Factors pertaining to importance	Identification
Improved temporary grass lays	Too productive for good opportunities for habitat creation	Identified using satellite imagery values
Old improved grassland / Permanent grass lays (less productive)	Good opportunities for habitat recreation – the less productive the better	Identified using satellite imagery values
Small fields and margins	Less productive and more heterogeneous small fields and arable margins, likely to be more permeable than intensive improved grassland or arable	Identified using satellite imagery values and field size information
Potentially interesting grassland	Land most likely to be semi-improved and or species rich grassland identified from remote sensing	This class is made up of land on steep south and north slopes with low productivity values as well as those fields with indicators of semi-improved characteristics from the remotely sensed imagery
Wet grassland	Can be good habitat in its own right or suitable for opportunities for wet woodland	Identified using satellite imagery values, however, only larger patches will be easy to identify
Semi-improved grassland	Existing habitat type so not included in opportunities layer	Identified using satellite imagery + data already existing for most important sites.
Arable land	Too productive for good habitat recreation opportunities This will be split according to crop features if the data allows	Identified using satellite imagery
Coniferous woodland	Can be part of woodland network – Focal network	Identified using satellite imagery for small blocks and ribbon woodlands but data already exists for FC sites
Broadleaved woodland	Core network – cannot tell planted from semi-natural	Small and planted woodlands identified + data already exists for semi-natural sites. Using satellite data boundaries much less well defined as shadow can increase woodland edge by 10m

Target land covers	Factors pertaining to importance	Identification
Scrub	Gorse Woody	Identified using satellite imagery, however, it is only possible to map large patches and manual checking will be needed from RGB aerial photography
Wide hedges / ribbon woodlands	Part of Core network	Identified using satellite imagery, however, they will have a larger minimum mapping unit as the boundaries are much less well defined and shadow can increase woodland edge by 10m.
Orchards and parklands	Orchards and parklands have similar characteristics, to that of the woodland network and are a component of the network.	Identified from previous studies and large commercial orchards identified from their appearance on the imagery
Lowland heath	Links to woodland, sandy, acidic, nutrient poor soil, cool, moist climate.	Data already exists for semi-natural sites. Outside known sites satellite imagery allows for the identification of larger patches with more than 30-50% cover of ericoid species
Coastal habitats	Part of existing coastal network	Data already exists

Stage 3 & 4 – 'More'

The original network previously created for the Dorset ANOB was used to represent a '*more*' opportunities network, as described in the Lawton principles. The network offers a combination of ecological opportunities through identifying linkages between habitats as well as movement costs through each habitat (Catchpole, 2007).

The methodology employs geoinformatic approaches to calculate landscape permeability and highlight areas of greater opportunities for enhancing habitat connectivity and ecosystem resilience. The ecological network aims to prioritise areas for targeted habitat creation and protection based upon findings through the production of detailed maps for woodland and grassland networks.

The creation of this network involved the identification of the following areas;

- Core: existing areas of semi-natural habitat,
- Potential: areas of potential network expansion,
- Permeable: areas which do not restrict species movement.

Areas of ‘target’ land cover classes (areas which will readily support a native community, and from which ‘potential’ areas are selected) are outlined in Table 2. The network specific land cover classes were extracted from the habitat map and combined with additional datasets, such as the Integrated Habitat Systems dataset which, in turn, form the base data for the network creation.

The base data is then classified into core, potential and permeable areas of either woodland or grassland classes. Habitat networks were created by combining the locations of suitable target areas with the permeability of the surrounding landscape. Buffer zones were created around these core networks using a hierarchical buffering approach to represent the distance native species would move through the surrounding land cover. Restoration within the boundaries of these buffer areas will have the most benefit by enhancing the existing network. This approach is conceptualised in Figure 3 and represented in Figure 4.

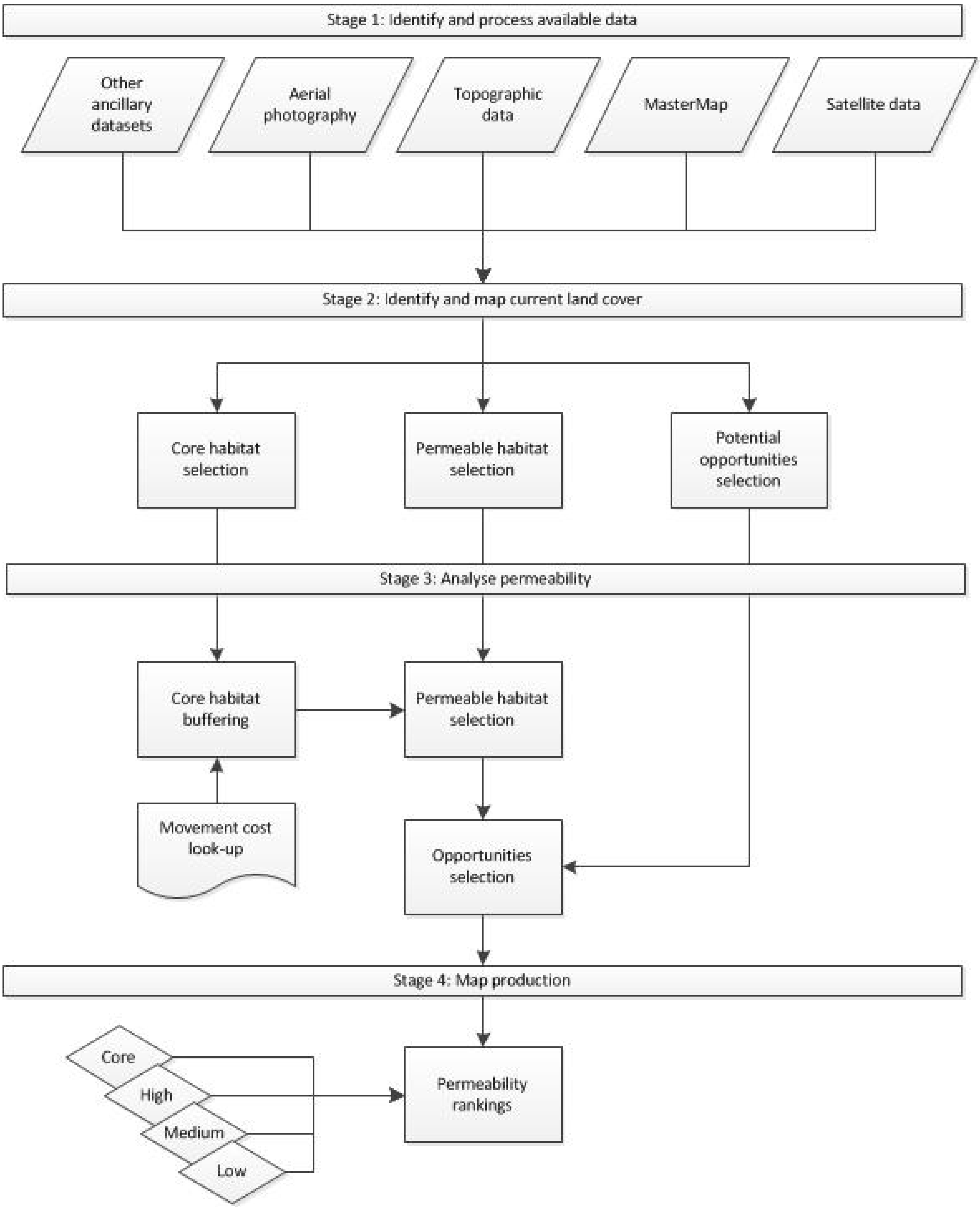


Figure 3. Workflow for the creation of the more opportunities network.

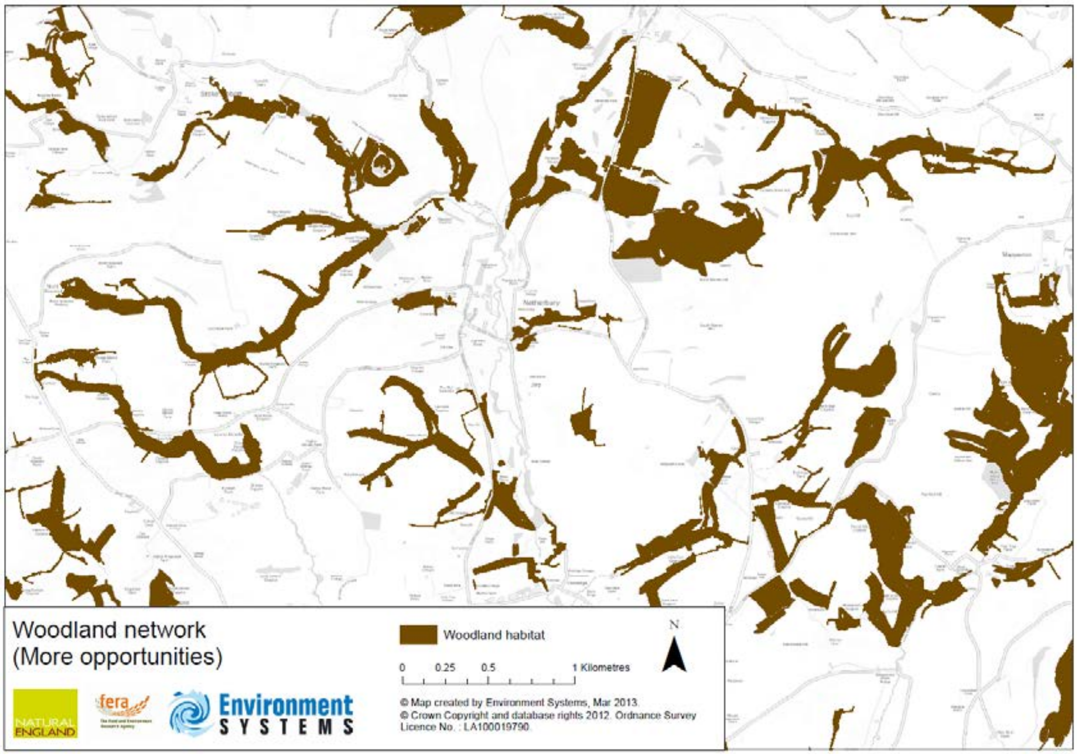


Figure 4. Hypothetical woodland network – More.

Stage 3 & 4 – ‘Bigger’

A modification to the ‘*more*’ network methodology allowed the creation of a ‘*bigger*’ opportunities network, as described in the Lawton principles. The creation of this ‘*bigger*’ network involves the identification of core semi-natural habitat and, in the case of the woodland network, ‘sub core’ areas (areas of coniferous woodland and scrub features) were also identified to allow greater areas for network creation. Areas of target land cover classes are outlined in Table 2.

The network specific land cover classes were extracted from the habitat map to form the base data. This base data is then classified into core, sub-core (where applicable) and target areas of either woodland or grassland classes. Habitat networks were created by selecting target and sub-core areas that were adjacent to core areas, thus creating bigger patch sizes. This approach is conceptualised in Figure 5 and represented in Figure 6.

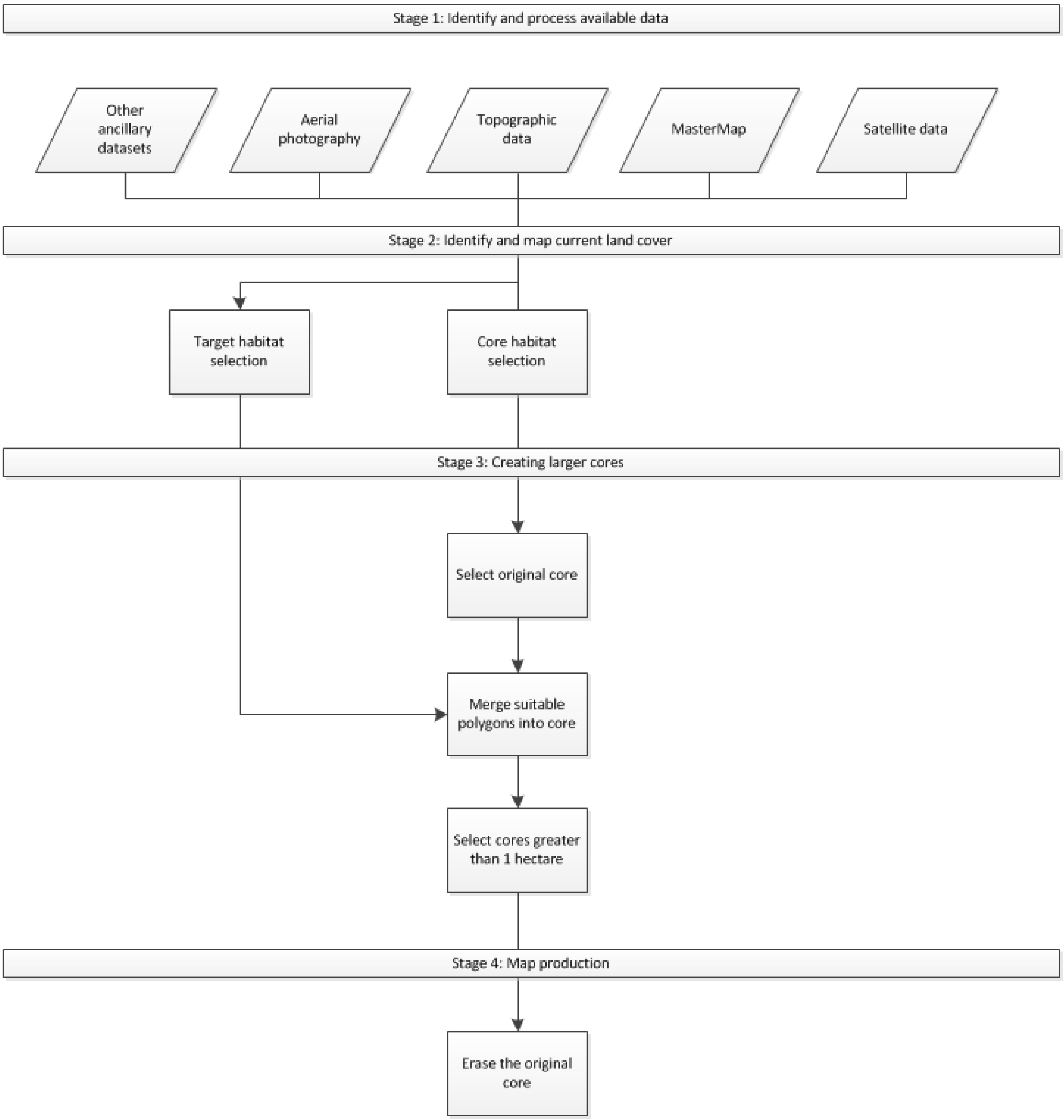


Figure 5. Workflow for the creation of the bigger patch size network.

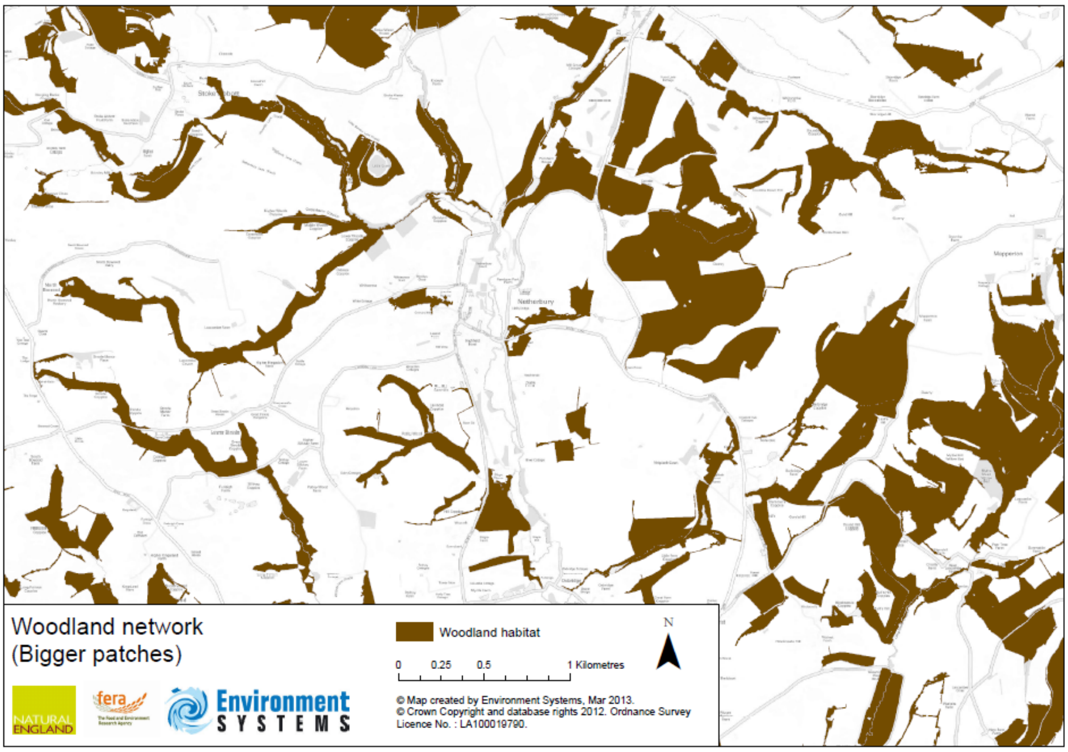


Figure 6. Hypothetical woodland network – Bigger.

Stage 3 & 4 - Joined up

A modification to the '*more*' network methodology allowed the creation of a '*joined-up*' opportunities network, as described in the Lawton principles. The creation of a network which is more '*joined-up*' involves the identification of core habitat along with target and permeable areas of the target land cover classes outlined in Table 2.

Core habitats greater than or equal to 2 hectares were selected as a starting point. Remaining core areas smaller than this were attributed as target. The core layer was subsequently buffered by 15 metres and then target and permeable habitats within the 15 metre buffer were attributed as step one. The step one polygons were buffered by 10 metres and then target and permeable habitats within the 10 metre buffer were attributed as step two. Step two polygons were buffered by 5 metres and polygons within the 5 metre buffer were attributed as step three, and so on, until the pre-determined number of growth stages had been reached. This approach is conceptualised in Figure 7 and represented in Figure 8.

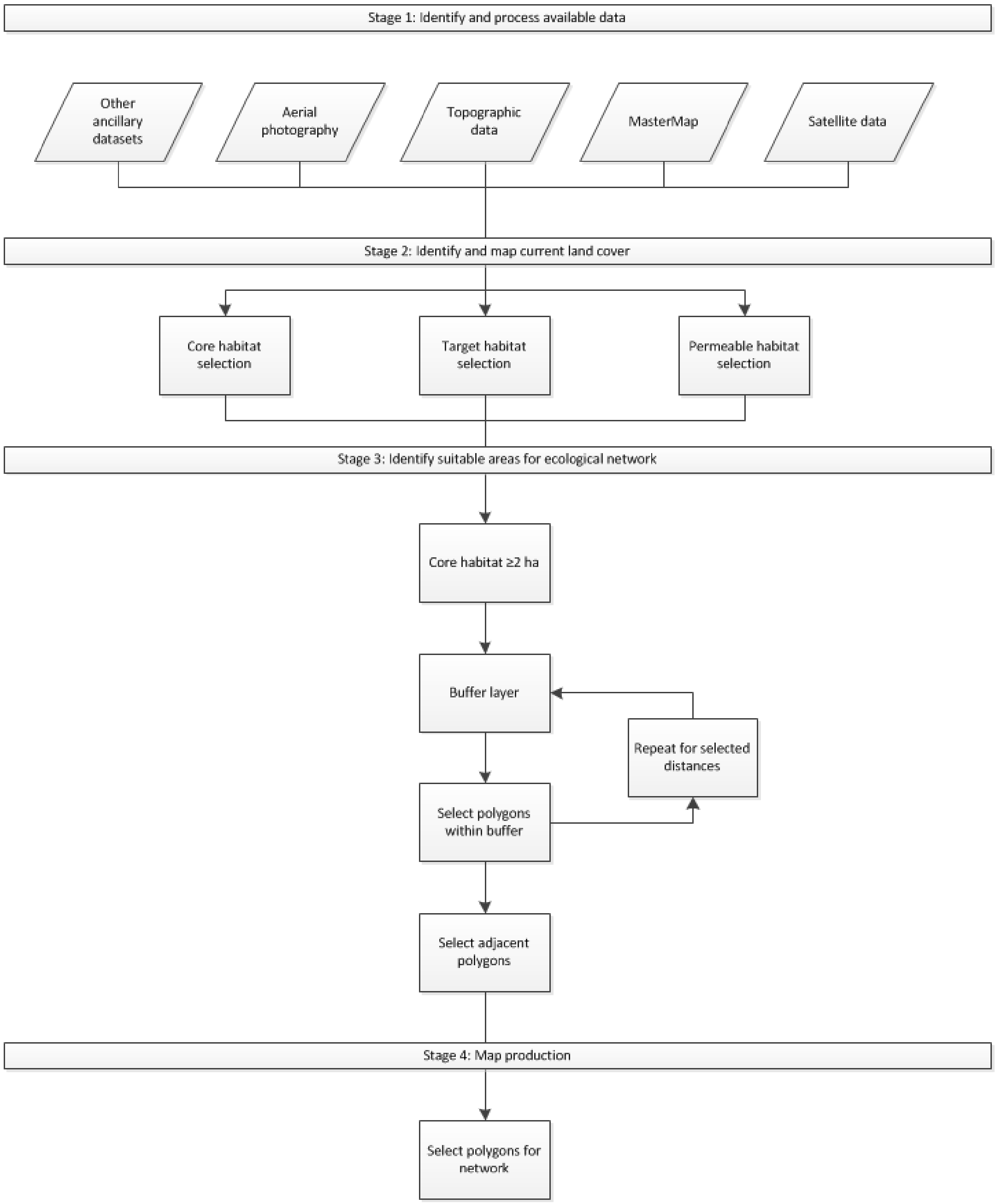


Figure 7. Workflow for the creation of the more joined up network.

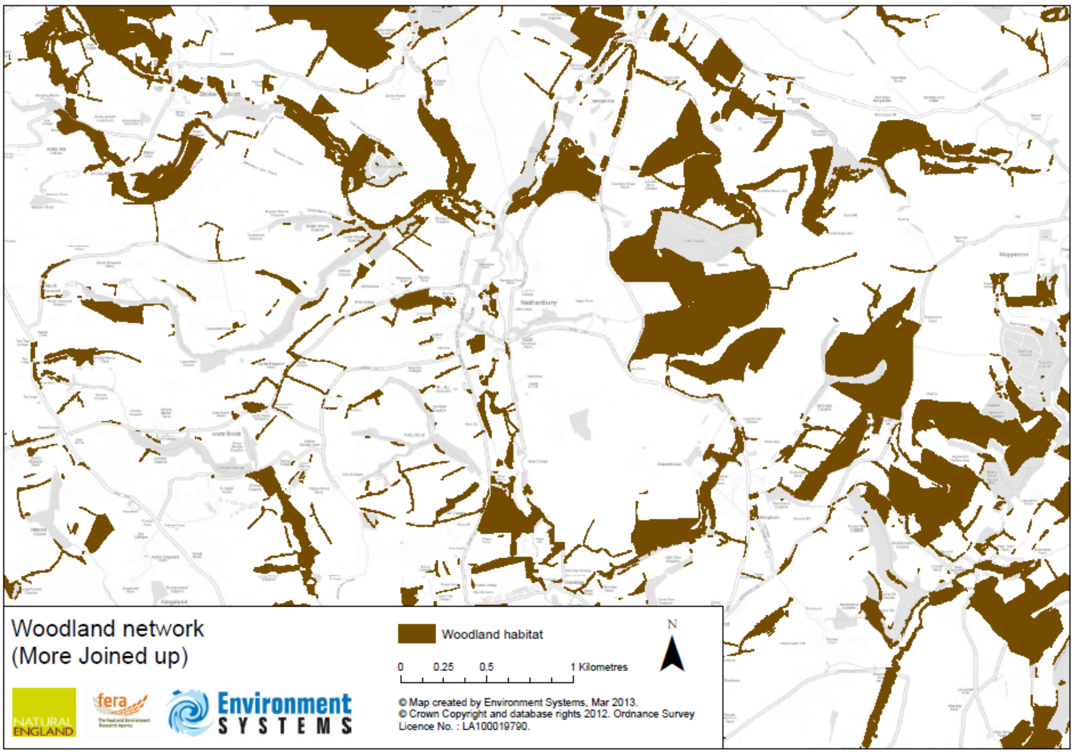


Figure 8. Hypothetical woodland network – Joined up.

STEP 2: Modelling Predicted Spread

Hypothetical areas of current invasion

Hypothetical areas of current invasion of three indicator species were used in the modelling process. These areas were identified with expert guidance (Table 3).

Table 3. The hypothetical areas of invasion of the indicator species.

Indicator species	**Hypothetical area of current invasion**
Cotoneaster horizontalis	Calcareous grassland near a built up area
Anoplophora labripennis	Currently not present in the UK, so anywhere within its potential habitat, perhaps at the edges on the North east of case study area
Mesotritonalpestris	Current sparse presence in the UK so anywhere within its potential habitat

To reduce this initial broad selection of areas, only areas of habitat greater than 0.25 hectares were selected to better represent sufficient establishment to facilitate spread. From this, 50 locations were chosen at random using a GIS software algorithm.

Categorise invasive traits

It is mostly possible, with existing knowledge and literature, to categorise the invasive species traits relevant to each dataset into those of high to low significance. Even with less than ideal data and a less than exact knowledge about the interactions between the habitats, its location, management and the service, it is possible to grade the importance into simple categories, from low (-3) to high importance (3).

Formulate the rule-base

For each invasive species a rule-base was required that identified:

- The specific elements of each dataset important for that species
- The score assigned to each element
- Any weighting required when datasets are combined or merged

Scores within the rule-base were generally based on the trait table and adapted to the most suitable attributes within the datasets available Figure 9.

Theme	Attributes	*Ailanthus altissima* (Tree of heaven)	Buddleja davidii	Cotoneaster horizontalis	*Impatiens glandulifera* (Himalayan Balsam)	Pinus pinaster (Maritime pine)	Prunus serotina (Black cherry)	Quercus ilex (Holm oak or Holly oak)	Rosa rugosa (Japanese rose)	Senecio inaequidens	Solidago gigantea (giant goldenrod)	*Monochamus sartor* (Sawyer Beetle)	*Cyphostethus tristriatus* (Juniper Shieldbug)	*Lymantria dispar* (Gypsy Moth)	*Thaumetopoea processionea* (Oak Processionary Moth)	*Anoplophora glabripennis* (Asian longhorn beetle)	*Corythucha arcuata* (Oak Lace Bug)	*Anoplophora chinensis* (Citrus Longhorn Beetle)	*Agrilus planipennis* (Emerald Ash Borer)	*Lasius neglectus* (an invasive garden ant)	*Gelechia senticetella* (cypress groundling)	Pelophylax ridibundus (Marsh frog)	Xenopus laevis (African Clawed Toad)	Macrochelys temminckii (Alligator Snapping Turtle)	Lithobates catesbeianus (American Bullfrog)	Python molurus subsp. bivittatus (Burmese Python	Bufo marinus (Cane Toad)	Chelydra serpentina (Common Snapping Turtle)	Trachemys scripta (Red-eared Terrapin)	Mesotriton alpestris (Alpine Newt)	Podarcis muralis (Wall Lizard)	Myocastor coypus (Coypu)	Ondatra zibethicus (Muskrat)	Pecari tajacu (Collared Peccary)	Alopex lagopus (Arctic Fox)	Felis (Prionailurus) bengalensis (Leopard Cat)	Hydrochaeris hydrochaeris (Capybara)	Nyctereutes procyonoides (Raccoon Dog)	Procyon lotor (Raccoon)	Pudu puda (Southern Pudu)	Hydropotes inermis (Chinese Water Deer)	Mephitis mephitis (Striped Skunk)	Nasua nasua (Coati)	Prairie dog
Habitat_agri_environment_+ve	Coniferous woodland	1	-3	-3	-3	1	1	1	-3	-3	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	2	0	0	0	3	0	2	0	3	3	3	1	1	1	0
Habitat_agri_environment_+ve	Dry heath	1	1	1	-3	3	1	3	2	2	1	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	2	0	0	0	0	1	0	0	0	0	0	0	0	0	1
Habitat_agri_environment_+ve	FenSwampFlush	1	-3	-3	2	-3	1	-3	-3	-3	1	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	2	0	0	0
Habitat_agri_environment_+ve	Gardens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Habitat_agri_environment_+ve	Hedgerows	1	1	2	-3	-2	1	1	3	1	1	0	0	1	1	2	1	2	1	0	0	1	1	0	1	2	1	0	0	2	1	0	0	0	0	0	0	1	1	0	0	1	1	1
Habitat_agri_environment_+ve	Improved grassland	1	-1	0	0	-2	1	-1	1	1	2	0	0	0	0	0	0	0	0	0	0	1	1	1	1	1	1	1	1	1	0	1	1	0	0	0	1	0	0	0	1	0	0	3
Habitat_agri_environment_+ve	Inundation vegetation	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Habitat_agri_environment_+ve	Lowland meadows	2	0	1	-3	-3	1	0	1	2	1	0	0	0	0	0	0	0	0	0	0	2	2	1	2	0	1	1	1	3	0	1	1	0	0	0	0	0	0	0	3	0	0	0
Habitat_agri_environment_+ve	Marshy grassland	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Habitat_agri_environment_+ve	Mixed woodland	1	-3	-3	-3	-3	1	1	-3	-3	-3	0	0	1	1	1	1	1	1	0	0	0	0	0	0	1	0	0	0	2	0	0	0	3	0	2	0	3	3	3	1	1	1	0
Habitat_agri_environment_+ve	Neutral grassland	2	1	1	-3	-3	1	0	1	2	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	2	0	0	2
Habitat_agri_environment_+ve	Orchard	0	0	0	-3	0	0	0	0	0	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	2	1	3	0	0	1	3	3	3	3	2	3	1
Habitat_agri_environment_+ve	Quarry	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Habitat_agri_environment_+ve	Roadbeds	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Habitat_agri_environment_+ve	Ribbon_Woodland	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Habitat_agri_environment_+ve	Saline lagoon	-3	-3	-3	-3	-3	-3	-3	1	-3	1	0	0	0	0	0	0	0	0	0	0	3	3	2	3	1	3	3	3	3	0	2	1	0	0	0	0	0	1	0	2	0	0	0
Habitat_agri_environment_+ve	Saltmarsh	-3	-3	-3	-3	-3	-3	-3	-3	-3	-3	0	0	0	0	0	0	0	0	0	0	2	2	2	2	1	2	2	2	1	0	3	2	0	0	0	2	0	0	0	3	0	0	0
Habitat_agri_environment_+ve	Sand dune	1	2	2	-3	2	1	2	3	3	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	3	0	0	0	3	1	0	0	3	0	0	0	0	0
Habitat_agri_environment_+ve	Scrub	2	2	2	-3	1	2	2	2	1	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	3	0	0	0	2	3	0	0	3	2	3	0	3	3	1	2	3	2	1
Habitat_agri_environment_+ve	Sea cliff	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Habitat_agri_environment_+ve	Semi-improved chalky grassland	1	1	2	-3	-3	1	1	1	1	1	0	1	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	2	0	0	2

Colour	Value
🟩	3
🟩	2
🟩	1
⬜	0
🟥	-1
🟥	-2
🟥	-3

Figure 9. Example section from the rule-base; and scoring key.

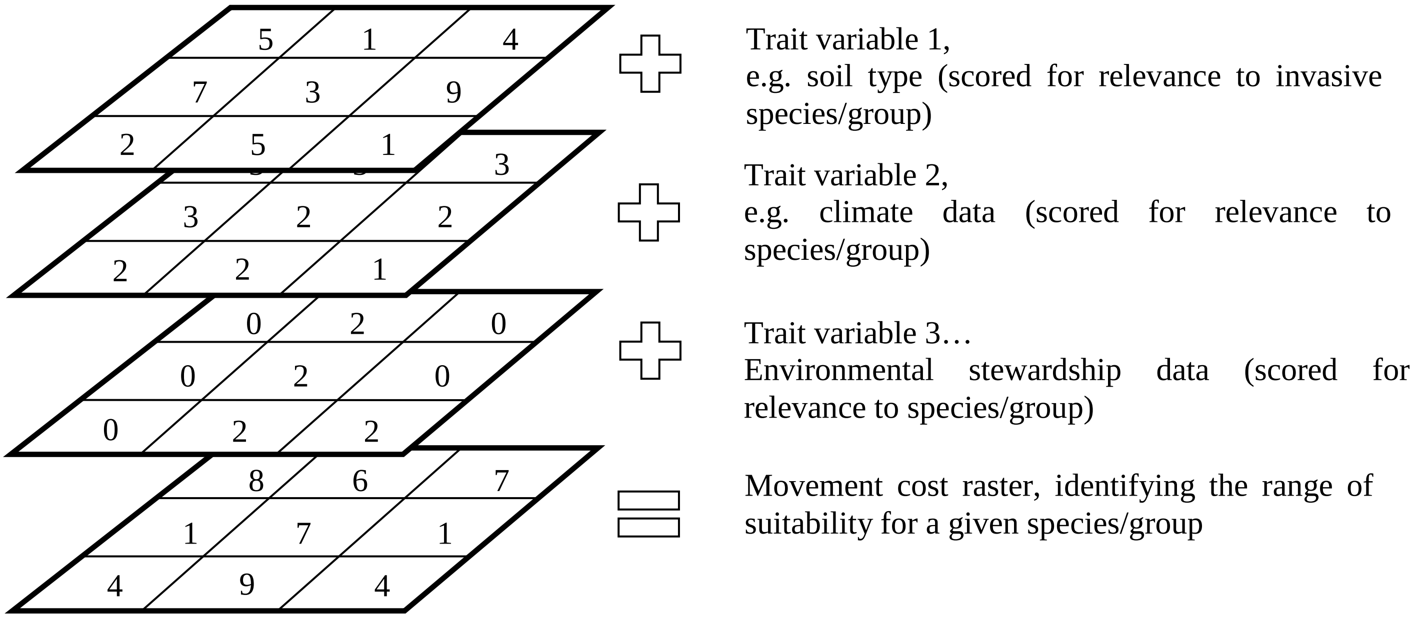
Raster modelling

The rule-base was then used to transfer these trait score across to the spatial raster data using a GIS. This resulted in multiple raster data layers, each layer representing the spatial distribution of the influence of a particular trait e.g. influence of soil type or land cover. These layers were then combined in a GIS using an additive raster model, to produce a single raster layer or ‘movement cost’ landscape data layer (Fig. 14). These data represent the cost to movement across the surface of the landscape, from one point (or cell) to the next, for a given species. This process was repeated for each invasive species to be modelled.

The method for modelling of the predicted spread was cost-distance analysis. This process calculates the accumulative cost distance for each cell to the nearest hypothetical areas of current invasion, over the movement cost landscape data layer.

This process continues until a threshold is reached, which is based on the extent to which that species will move through the landscape (the dispersal distances within the trait table). The resulting data then shows the predicated spread and an estimate of probability, ranging from highly probable to improbable, across the study area.

A cumulative sum was then plotted of the proportion of area at a given probability, for the total area of study. The steeper the line gets, the greater the increase in area over that particular likelihood range.



Trait variable 1,
e.g. soil type (scored for relevance to invasive species/group)

Trait variable 2,
e.g. climate data (scored for relevance to species/group)

Trait variable 3…
Environmental stewardship data (scored for relevance to species/group)

Movement cost raster, identifying the range of suitability for a given species/group

Figure 10. Conceptual diagram of an additive raster model.

4.4 Study sites

The starting points (indicating arrival or presence of invasive) for each species in the case study were considered and decided for using expert judgement as follows:

- *Cotoneaster horizontalis* (Wall cotoneaster) – calcareous grassland near a built up area
- *Anoplophora glabripennis* (Asian Longhorn beetle) – currently not present in the UK so anywhere within its potential habitat, perhaps at the edges on the NE
- *Mesotriton alipnus* (alpine newt) – current sparse presence in the UK so anywhere within its potential habitat.

Three study sites were selected within the Dorset AONB to reflect a more detailed pattern of results. Figure 11 shows where these sites are in relation to the whole Dorset region.

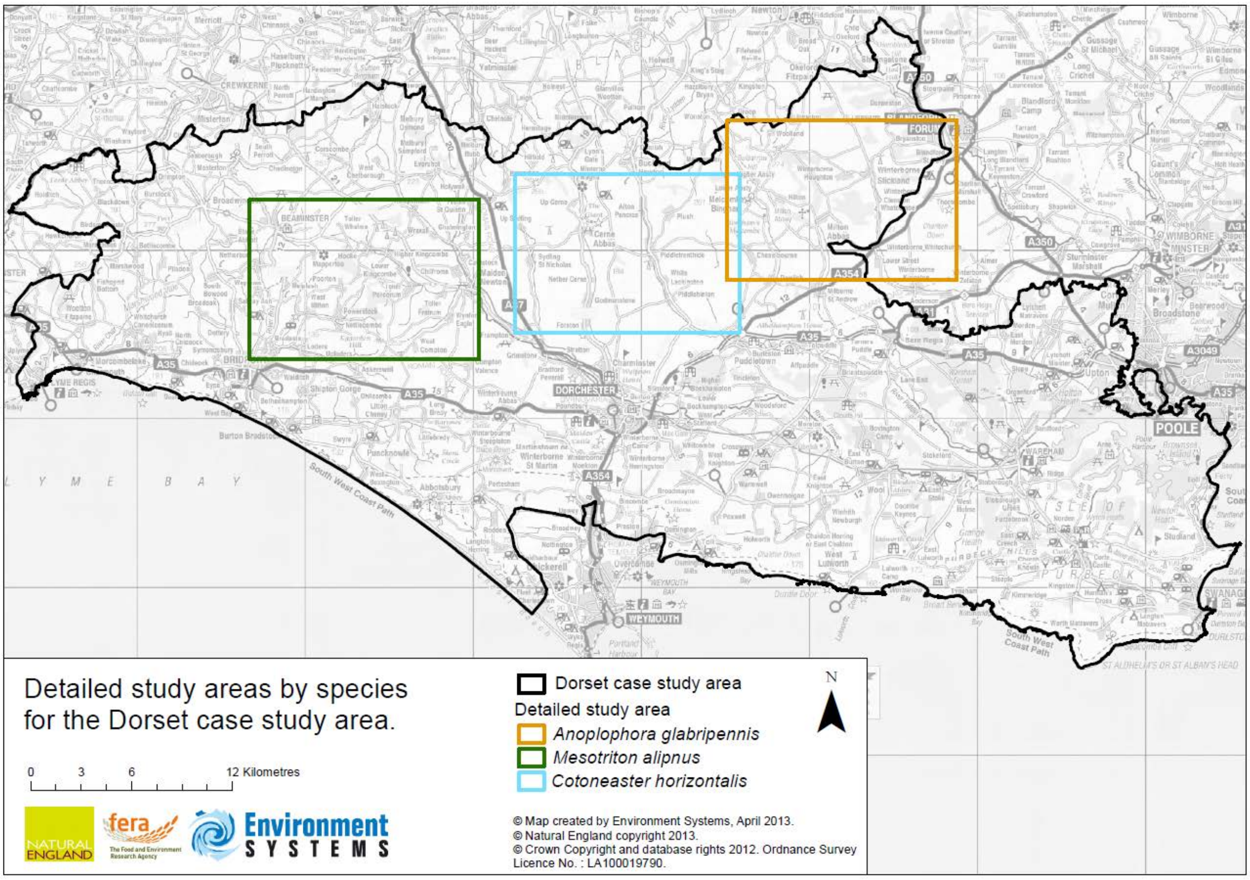


Figure 11. The study sites for selected within the Dorset AONB region.

To reduce this initial broad selection of areas, only areas of habitat of greater than 0.25 hectares were selected. From this, 50 areas were chosen at random, using GIS software. The modelling consisted of least cost distance analysis. This process calculates the least accumulative cost distance for each cell to the nearest hypothetical start points over a cost surface data layer. This process continues until a threshold is reached (the extent to which that species will move – based on the dispersal distances within the trait table). The resulting data then shows the predicated spread and an estimate of probability, ranging from highly probable to improbable, across the study area. A cumulative sum was then plotted of the proportion of area at a given probability, for the total area of study. The steeper the line gets, the greater the increase in area over that particular likelihood range.

4.5 Modelling Results

Figures 12-14 show thumbnail maps of the modelling results. These show both 1 year and 10 year time intervals for all network scenarios:

- Control – no change from present day
- More – more patch sizes
- Bigger – bigger patch sizes
- Joined Up – more continuity between patches
- Climate Change – using climate degraded habitat
- Agri-environment schemes (negative) – using those schemes that will have a negative impact on the species
- Agri-environment schemes (positive) – using those schemes that will have a positive impact on the species

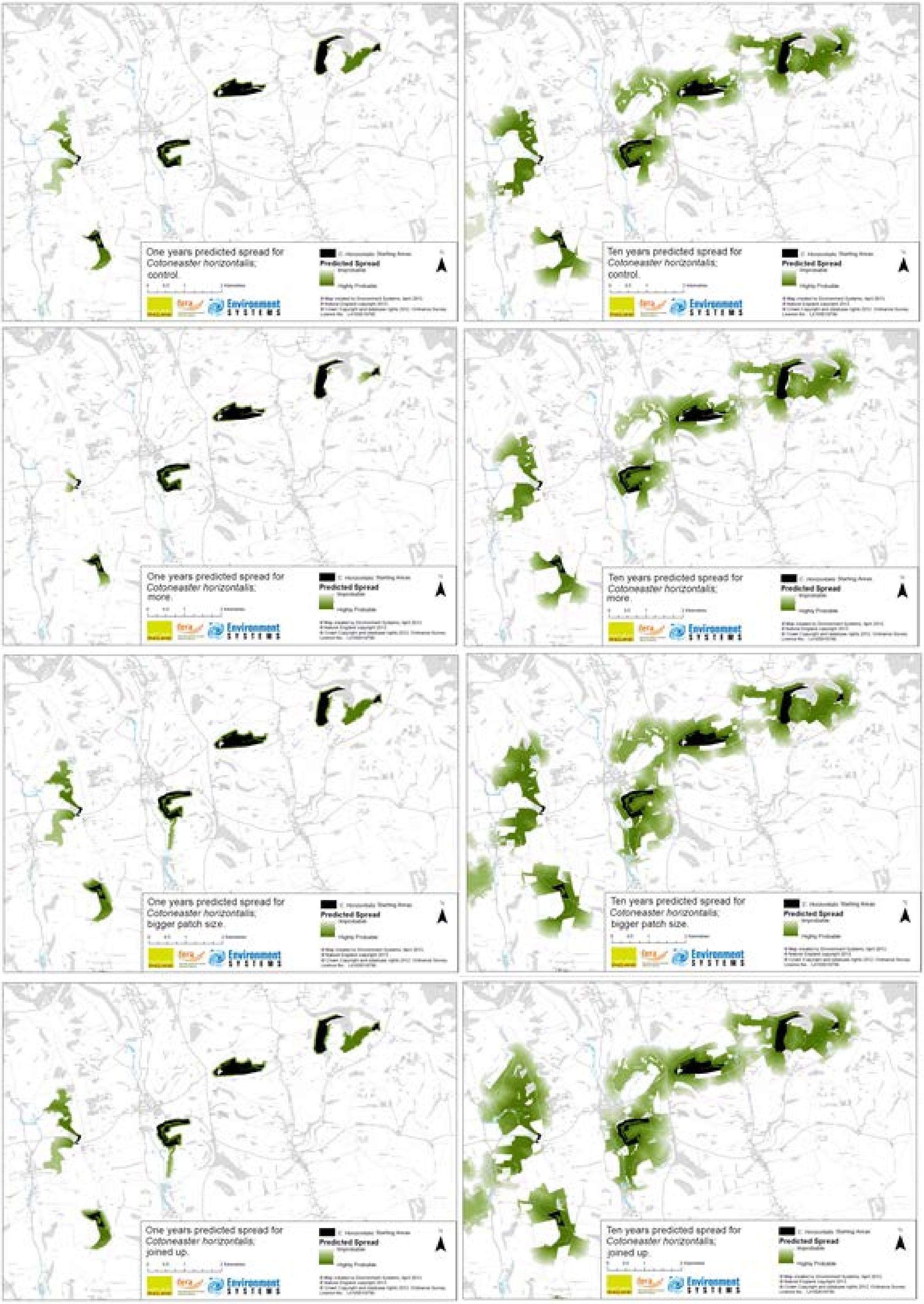


Figure 12a. Spread map thumbnails for Wall cotoneaster.

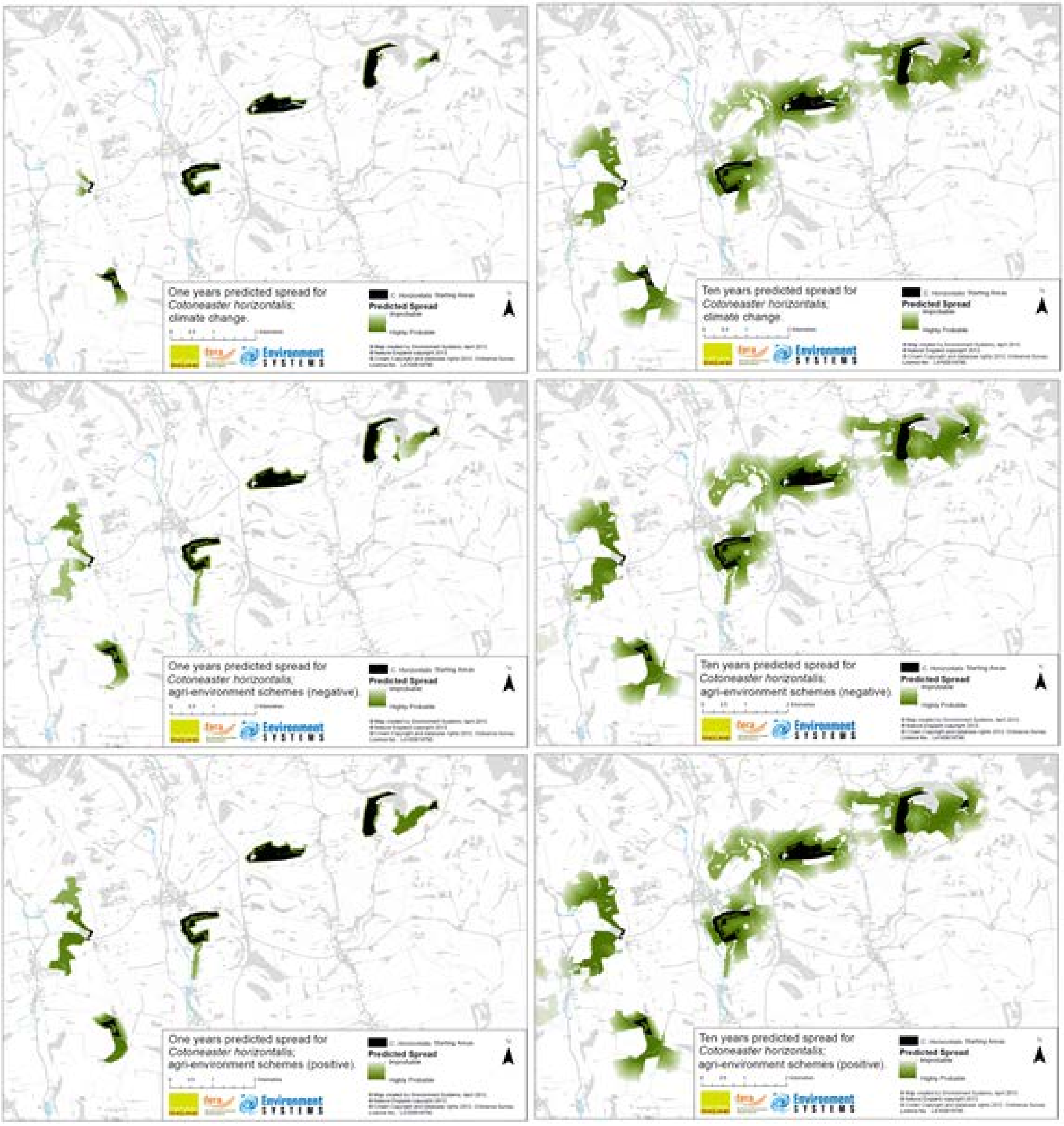


Figure 12b. Spread map thumbnails for Wall cotoneaster.

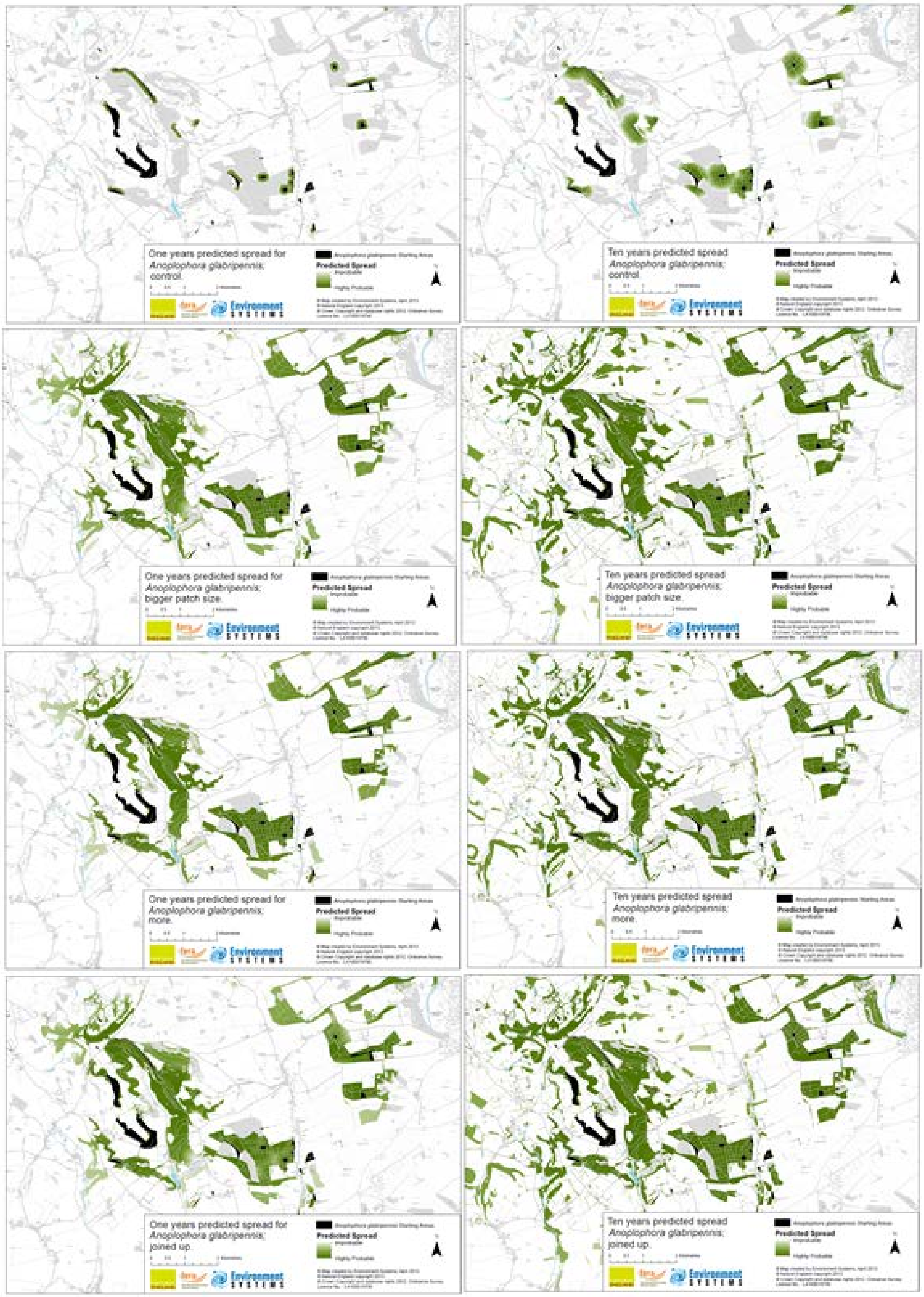


Figure 13a. Spread map thumbnails for Asian Longhorn Beetle.

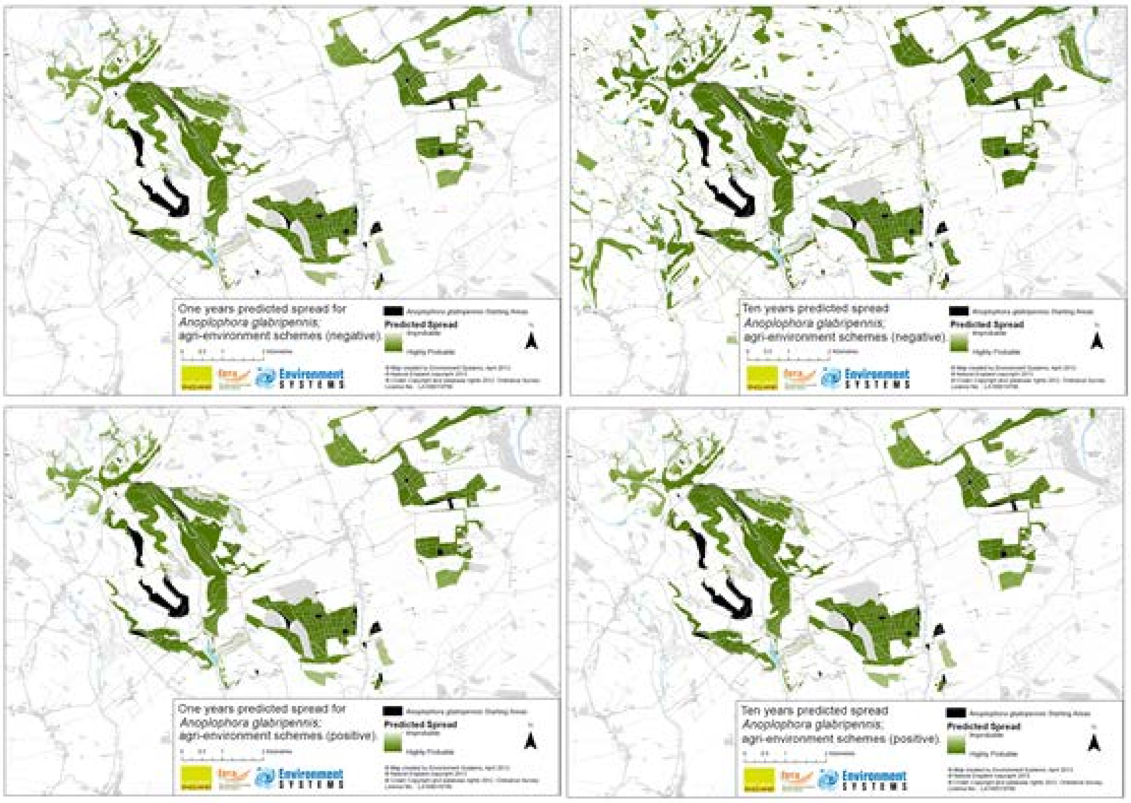


Figure 13b. Spread map thumbnails for Asian Longhorn Beetle.

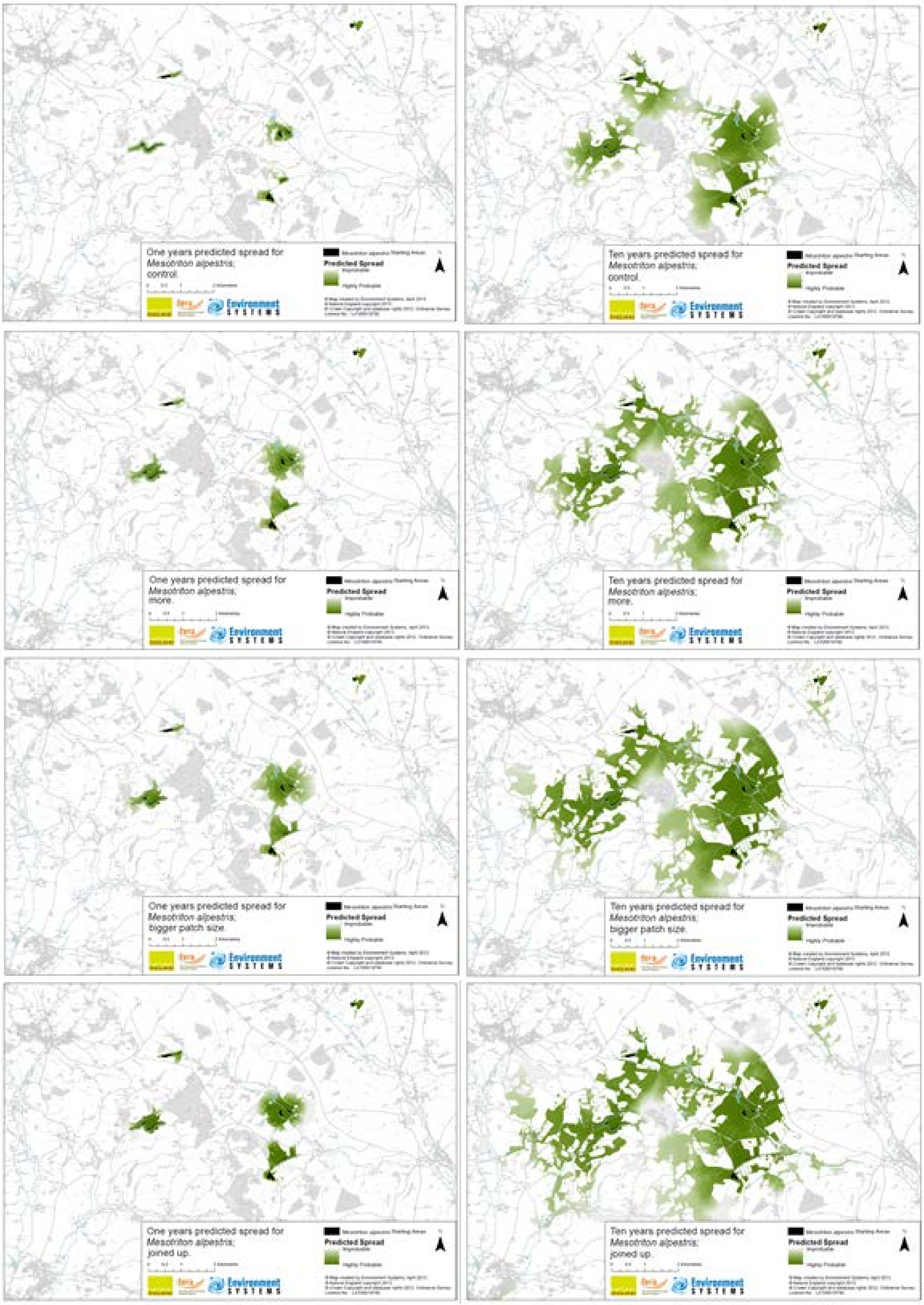


Figure 14a. Spread map thumbnails for Alpine Newt.

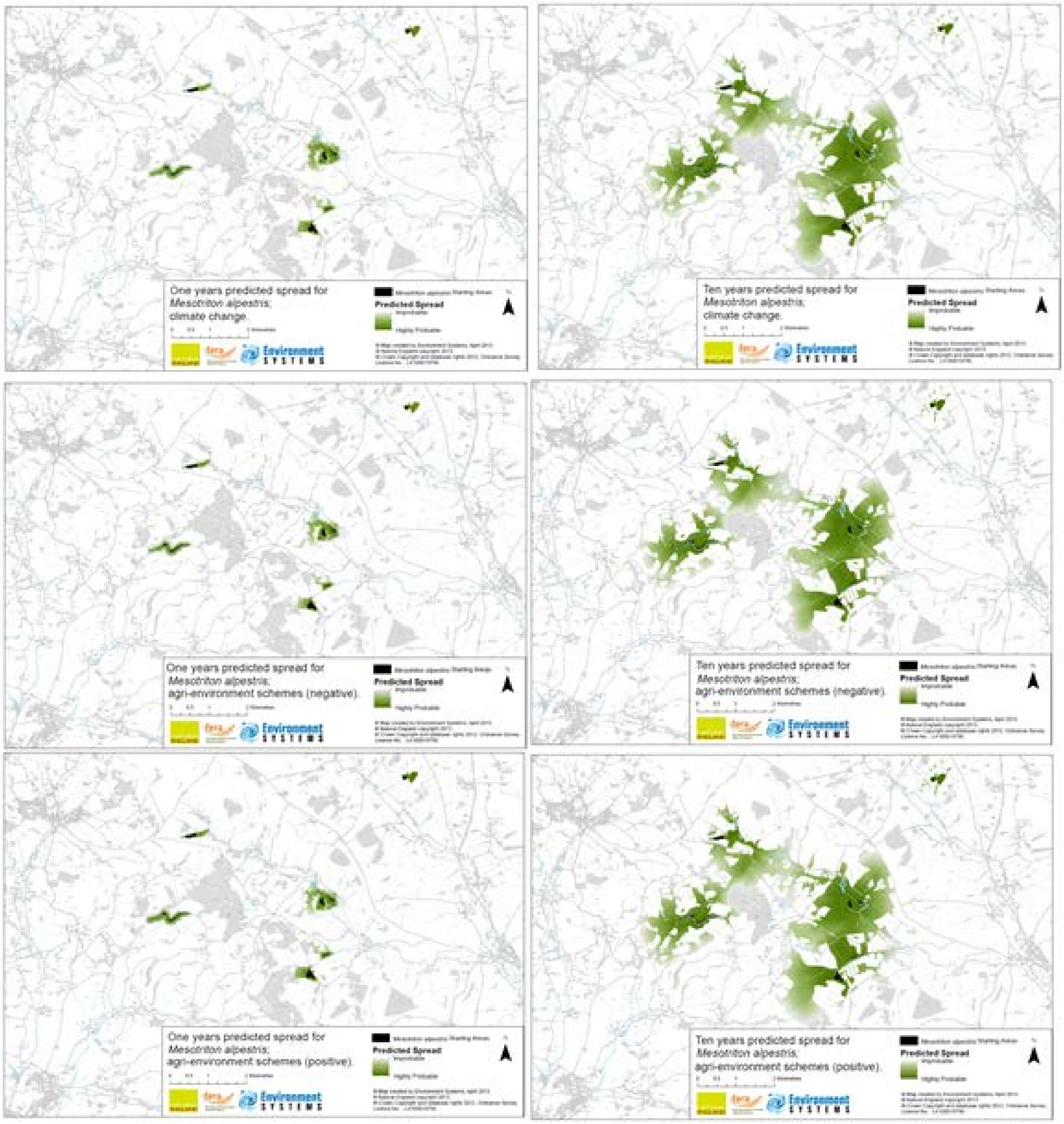


Figure 14b. Spread map thumbnails for Alpine Newt.

Figures 15-17 show a selection of the cumulative plots.These plots show for each of the three species under the range of network scenarios the proportion of probabilities of spread in relation to the entire Dorset AONB after 10 years from introduction. These support the maps in Figures 12-14 by quantifying the proportions of area at different probabilities of spread. WC and AN have linear graphs, indicating that in this case study the spread of probabilities is relatively equal across the area. The graphs for ALB however have a relatively large proportion of the area of spread at the highly probable end of the x-axis (darker green), with a subsequent plateau at the middle - low probabilities. This pattern occurs across the network scenarios in contrast to the control scenario.

This indicates that ALB will be successful at spreading across the habitat networks in this area, whereas WC and AN have larger areas of a lower probability. It must also be noted that the proportion of potential spread across the Dorset AONB is still relatively small, even after 10 years (maximum of 5% of the total area).

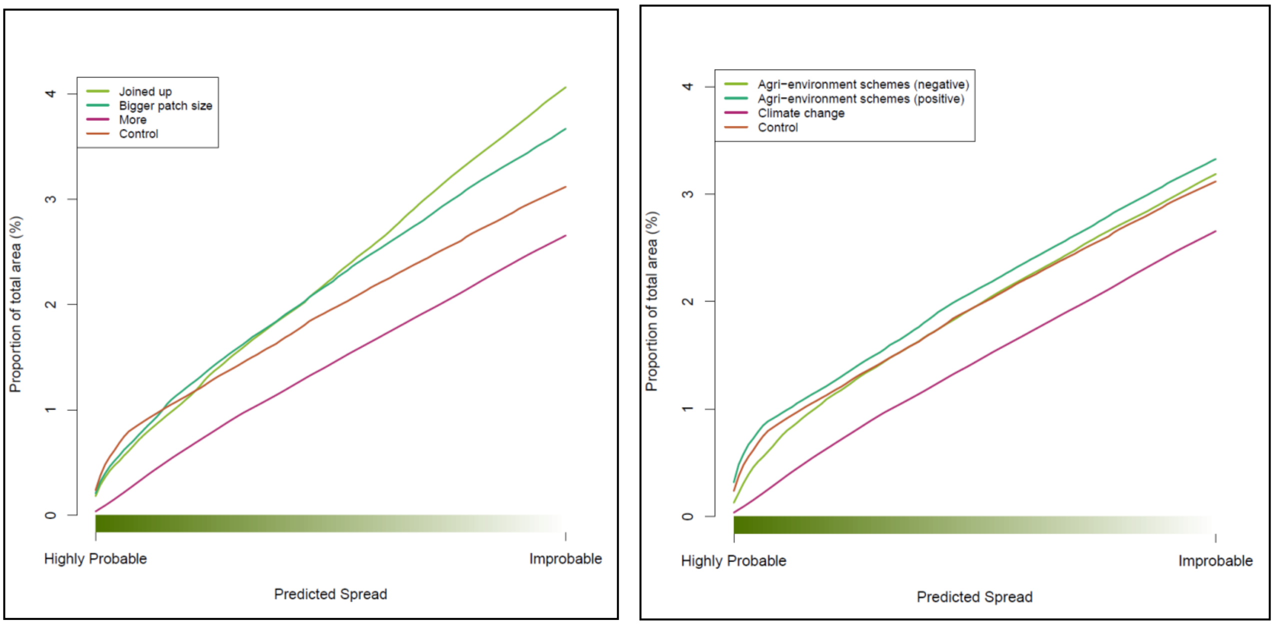


Figure 15. Wall cotoneaster – 10 years over Dorset.

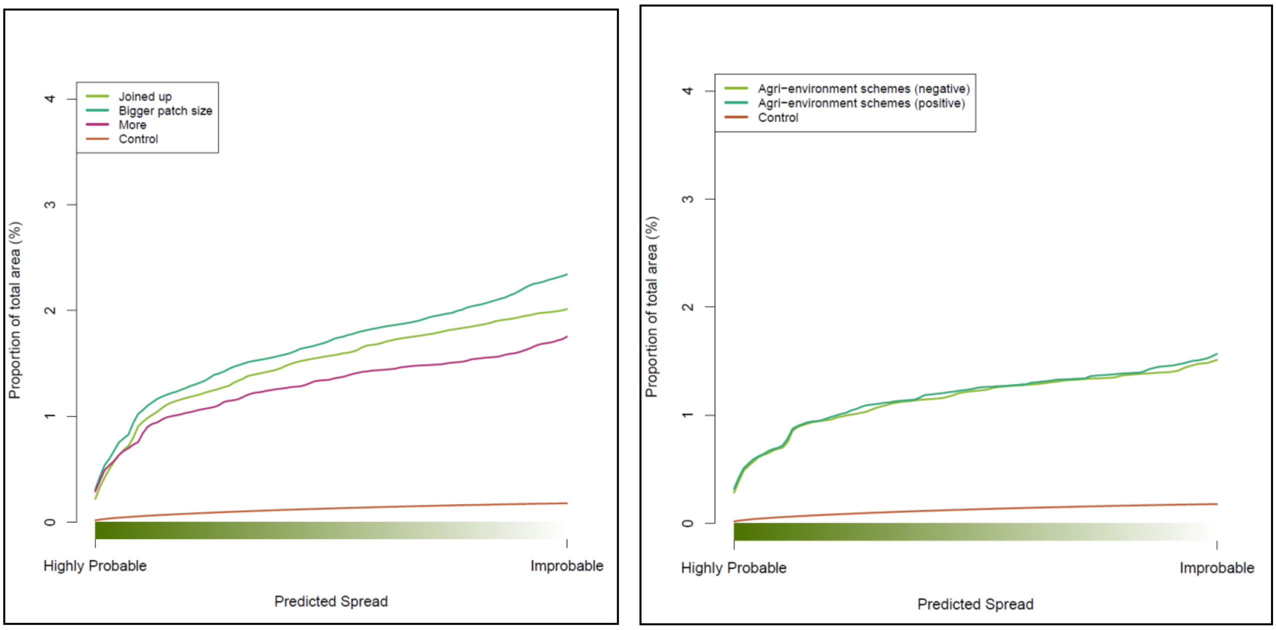


Figure 16. Asian Longhorn Beetle – 10 years over Dorset.

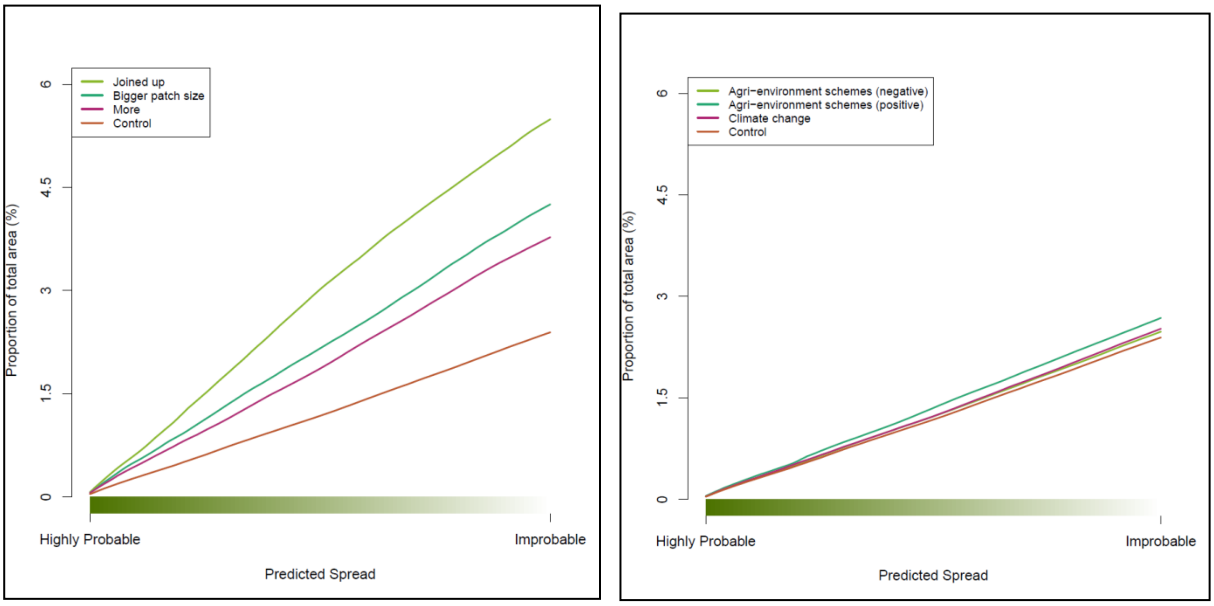


Figure 17. Alpine Newt – 10 years over Dorset.

Total areas were calculated to numerically show the areas of spread for each network at different probability levels. These are also represented as proportions of the total study area of each example and of the total Dorset AONB area (see Appendix G for full tables).

Wall cotoneaster

The maps show that in this area WC will predominantly move through calcareous, improved and semi-improved grasslands and scrub. The network design with the greatest proportion of potential invasion is Joined Up, with over 4% of the Dorset AONB area containing a probability of invasion after 10 years. 25% of this area has a very high probability. The scenarios with the lowest area of probability are the More patches and the climate change impacted scenarios. These have under 3% of the Dorset area containing a probability of invasion after 10 years. In contrast to the Control results, these two scenarios appear to have the effect of decreasing the area of spread.

Asian Longhorn Beetle

The maps show that in this area ALB will move predominantly through a variety of grasslands and woodland. The network designs with the greatest proportion of potential invasion are Bigger patches and again Joined Up, with over 2% of the Dorset AONB area containing a probability of invasion after 10 years. In this instance the results show that the Control scenario has the smallest area of potential invasion and therefore suggests that any activity involving ecological network alteration or climate change will increase potential invasiveness of ALB. This species also has the smallest difference between the 1 year and 10 year scenarios which implies that any outbreak will require an immediate response.

Alpine Newt

The maps show that in this area AN will predominantly spread through a variety of grasslands and woodlands. The network designs with the greatest proportion of potential invasion are the same as ALB – More patches and Joined Up, with approximately 4-5% of the Dorset AONB area containing a probability of invasion after 10 years. The distribution of the probability levels is relatively equal from very high to very low probability.

4.6 Trade-off table results for case study species

Wall cotoneaster (WC)

WC would potentially benefit (score of 2 in the trait table) by an improvement in a variety of habitats (calcareous grasslands, hedgerows, sand dunes and coastal cliffs, scrub and stone wall and earth banks). However, Environmental Stewardship options have within their prescriptions an action to target and reduce/remove invasive species, and WC would be relatively easy to remove especially if early action was taken before it became too widespread. Therefore timely control should limit impacts on habitats managed under Environmental Stewardship. By improving these habitats a variety of native species will also be positively impacted. These include 127 native species for calcareous grasslands (e.g. Brown Long-eared Bat, Marsh Fritillary and Purple Milk-vetch and 66 for hedgerows (e.g. Dormouse, Crester Cow-wheat and the Scarlet Malachite Beetle). For full list see the Trade-Off Table.

Asian Longhorn Beetle (ALB)

ALB will be positively impacted by one habitat type – hedgerows and only slightly positively impacted by another – broadleaf and mixed woodland.

Alpine Newt (AN)

AN will majorly benefit (score of 3 in the trait table) by an improvement in three habitats (ditches, lowland meadows, ponds) and positively impacted (score of 2 in the trait table) by an improvement in a range of habitats (bogs, broadleaf and mixed woodland, fen marsh swamp, healthland, hedgerows, moorland, rivers and streams, scrub, stone wall and earth banks and upland hay meadows).

Summary

Overall, the habitat improvements with the largest positive scores for impact (total score of 30+) across the entire trait table are cropped area, ditches, field margins and uncropped farmlands, hedgerows, rivers and streams, scrub and traditional orchards. As stated above, the improvement of hedgerows for example also benefits 66 native species. Therefore either a prioritisation of species needs to be devised, or an alternative habitat if available needs to be improved. For example, the native Dormouse will benefit from an improvement in both hedgerows and broadleaf and mixed woodland. Overall, improving the latter has a lower positive impact across the species in the trait table than that of hedgerows, so this could be a potential conservation plan for the Dormouse.

5. Discussion

Increasing connectivity of the landscape promotes coherent ecological networks that allow movement through the landscape. The importance of ecological networks increases when considered in light of climate change as they are believed to facilitate species range shifts. High levels of connectivity may also facilitate dispersal of non-native species, particularly for habitat specialists. However, non-native species that are invasive tend to be less influenced by landscape connectivity, especially mammals and birds, so the scope for restricting their invasiveness by manipulating connectivity is limited. On the plus side, efforts to improve connectivity to facilitate spread of native species are likely to have little impact on the spread of many invasive species.

When considering the impact of improving ecological networks on invasive species therefore, the traits of the species concerned need to be carefully considered. Plants, invertebrates and herptiles are more likely to be limited by connectivity, especially if they are habitat specialists. Many invasive non-native species are better able to exlpoit disturbed habitats than are natives, so the level of disturbance is a key consideration to take into account when creating or enlarging ecological networks.

When considering the types of landscapes that will result from the development of ecologican networks, several general principles emerge. Small habitat patches with low area to edge ratio (and therefore more edge habitat per unit area) are more likely to provide suitable conditions for invasion by alien species. Therefore larger habitat patches are preferable where invasives are a concern. Similarly, as noted above, invasive species are more able to exploit disturbed and degraded habitats. Maintaining high quality habitats is more likely to promote the maintenance of a diverse community of native species, resistant to invasion by aliens.

For non-mobile species, especially herptiles, number of habitat patches may be a consideration, but it is difficult to separate this from the effect of distance between patches. Isolation may be a key factor for some species or groups, e.g. birds.

Plants are most influenced by the degree of fragmentation, and habitat corridors that increase connectivity may encourage dispersal of invasive species, especially if the habitat quality is not high.

Improved connectivity within ecological networks is applied to improve dispersal of native species of conservation significance in order to allow them to move to areas that are climatically suitable as climatic conditions change. However, it is also likely that climate change will lead to an increase in the number of invasive species as conditions become more suitable for them. The challenges of designing sustainable, effective ecological networks without facilitating the spread of invasives are therefore significant.

Among traits that promote invasiveness, the most importat with respect to the impact of ecological networks is dispersal ability. Others that may be relevant include behavioural flexibility, high fecundity, and r-slected species (good at colonising). Bearing these traits in mind, species from each major grouping were selected as case studies to illustrate the impact of different aspects of network design. The likelihood of an invasive species reaching the UK through both natural and assisted dispersal through human-aided transport was also taken into consideration.

The potential impact of Environmental Stewardship was con sidered in terms of specific habitats and options available to manipulate their management. It was concluded that the habitat improvements most likely to benefit non-native species were cropped area, ditches, field margins and uncropped farmlands, hedgerows, rivers and streams, scrub and traditional orchards. Therefore to reduce the potential for invasion by non-native species, it was suggested that improvements should occur in other habitat types, or a prioritised trade-off between benefits and likely invasion needs to be considered. However, there are no ES options that completely favour native species and are completely unfavourable for non-natives, so the risks need to be considered for each case.

The potential for landscapes to be invaded by individual species selected to represent plants, insects and herptiles (the groups most likely to be influenced by the structure of ecological networks) was investigated by modelling their potential spread through selected areas with in the Dorset Area of Outsnading Natural Beauty. The modelling considered the inpact of varying the different elements of ecological network design defined in the Lawton princoples: ‘More’, ‘Bigger’, and ‘Joined up’. The impact of climate change and agri-environment schemes were also considered.

Overall, the modelling indicated that the network design with the highest impact on potential area of spread was Joined Up, followed by Bigger, then More. This implies that landscape management practices that increase the connectivity of the landscape to protect native species will also be beneficial for non-native species. This will be particularly relevant for plants as they are less mobile than other taxonomic groups. Creating bigger patches and more of them may be a better solution at mitigating against the invasion of non-natives. However as defined in the literature review, many non-native species will remain ambivalent to connectivity as they tend to be mobile generalists and will thrive regardless.

Creating bigger patches and having more of them will reduce the impact of edge effects which are the vulnerable areas to invasion. Many invasive species explored in this study also favour lower quality disturbed areas so ensuring these areas are also of a high quality through ecological restoration will help to favour native species.

The Asian Longhorn Beetle had the highest proportions of invasion area in the very high probability band, with values of over 50% of the Dorset AONB after 10 years with a probability of invasion. Alpine Newt had the highest proportional total area that had a probability of being invaded, with a maximum of 5.49% of Dorset AONB. Asian Longhorn Beetle had the lowest, with a maximum of 2.34%. Asian Longhorn Beetle has also had the smallest difference between the 1 year and 10 year areas of spread, indicating that any impact of network change will occur more rapidly for this species than for Wall Cotoneaster and Alpine Newt.

Conclusions

- Creating an ecological network to increase the joining up of patches will have the largest impact on the invasiveness of species in an area of grassland and woodland.
- Increasing the number of patches has the least affect on the invasiveness in this area.
- Climate change by itself will impact and increase the invasivity of non-native species.
- Many non-native species will remain ambivalent to changes in landscape connectivity as they are mobile generalists and will therefore thrive regardless of design.
- Ecological network design will have the biggest impact on plants.

Key Findings

1. In general, non-native invasive species are able to exploit disturbed habitats to a greater extent than native species are. The group for which this is especially true is plants. Invasive plants are known to preferentially invade edge habitat (for the reasons stated above) particularly those near human-disturbed habitat as these tend to be sources of additional non-native species.
2. Large patches may benefit natives as they allow for heterogeneity and therefore diversity and are more likely to be more stable. This means that they are more buffered from the invasion process. However the reverse may be true of smaller patches as they may be easier to invade as the edge of these patches, the most vulnerable zone, represent a greater area of the patch.
3. The creation of new habitat that requires the creation of disturbance is the most vulnerable habitat to invasion. It will be important to ensure that these are nutrient-poor to reduce the advantage invasive non-natives may have in these environments.

4. Improving the condition of semi-natural habitat by reducing other stressors may not have an impact on invasive invertebrates as they often attack both healthy and stressed vegetation.
5. Invasive mammals, especially those that are generalists, are able to exploit disturbed habitats.
6. Invasive amphibians often fare better in poor quality and disturbed habitats in comparison to native species so improving the quality of sites may help favour natives over invasives.
7. Habitat specialists irrespective of taxa are more influenced by habitat quality and connectivity than generalists.

Recommendations

- Bigger:
    - Larger reserves encourage stability, have higher heterogeneity than smaller patches and the edge represents a small proportion of the total area. Therefore larger areas, with lower area / edge ratios are more able to reduce non-native spread than several smaller areas.

- Better:
    - Increasing the quality of sites, through ecological restoration could increase the sites resilience to invasion
    - Ensure connecting corridors are of a high quality to reduce the risk of invasion by non-native plants.
    - Improving the condition of semi-natural habitat by reducing other stressors will not necessarily have an impact on invasive invertebrates.

- More:
    - Ensure that any habitat creation that requires disturbance is carried out on nutrient poor areas to reduce the competitive edge that non-native species may have.

- Joined up:
    - Any additional sites or corridors should be placed away from human disturbed areas and disturbance corridors such as railways, roads and canals as these disturbed areas are more easily exploited by invasive rather than native species. This has implications for linking urban and semi / non-urban areas.
    - In this instance this type of network design has the biggest impact on increasing potential invasiveness. Therefore this type of network could be implemented on habitats that have the least positive impact on invasives (e.g.bog improvement)

- General:
    - Certain habitats have a lower risk of invasion by invasive species than others. Prioritise these for improvements.
    - Continued surveillance of ecological networks to monitor for presence of invasive species.
    - Continued and improved strict inspection protocols at import sites to detect non-natives.

6. Acknowledgements

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8. Appendix A Literature review

8.1. Aims

The literature review addresses the following questions:

a) Do landscape characteristics affect the invasiveness of non-native species?
b) What are the likely impacts of climate change on the process of invasiveness?
c) What are the traits of non-native species that influence their invasiveness

The review is structured as follows:

- General discussion of invasive species and the invasion process (Section 2.2).
- How landscape influences the invasiveness of non-native birds, plants, insects, mammals and herptiles (Section 2.3).
- How climate change influences invasiveness of non-native birds, plants, insects, mammals and herptiles (Section 2.4).
- Other traits which influence invasiveness of non-native species (Section 2.5).
- Summary of the key traits

8.2. Introduction

Non-native species are those that enter regions outside of their natural range, along a number of different pathways, involving either deliberate or accidental release through human activity (Ruiz & Carlton 2003; Hill *et al.* 2005; Copp *et al.* 2007). Not all non-native species introduced outside their natural range are invasive (i.e. spread and impose detrimental impacts). Those species that are invasive, however, can have major impacts on native biodiversity and ecosystems, economics, agriculture and health (both animal and human) (White & Harris 2002).

It is widely accepted that only a small proportion of introduced non-native species become established, and that a small proportion of those established species then spread or become invasive. The idea holds that around one in ten imported species is introduced to the wild, one in ten of these introduced species becomes established in the wild and one in ten established species spreads and becomes invasive. This 'tens rule' was originally derived as a rule of thumb in relation to Angiosperms introduced into Britain (Williamson 1993) but has subsequently been broadly applied across other taxonomic groups. The application of the rule that around one in one hundred introductions will become established and one in one hundred of those will become invasive became enshrined in non-native species thinking. More recent work, however, has suggested that the ‘tens rule’ does not hold for vertebrates. For introductions of mammals, birds and freshwater fish between Europe and North America, one study found that around 50% of introduced species became established and around 50% of established species spread (Jeschke & Strayer 2005). Alternative studies, however, have argued that these higher estimates for the rate of establishment success can be over-estimates due to a lack of information on failed introductions (Rodriguez-*Cabal et al.* 2012).

There is an extensive and growing body of literature that has attempted to identify the traits that characterise successful and unsuccessful invasive species (Ehrlich 1989; Williamson & Fitter 1996). Despite intensive study, however, it is still difficult to predict the effects of a new species on a new environment. It is recognised that in addition to the species inherent life history and ecological traits, phenotypic plasticity (behavioural morphological and physiological flexibility), the characteristics of the receiving environment (including landscape design) and introduction effort (propagule pressure) can be as important as the invasive species attributes in influencing success (Lodge 1993).

Some of the most common traits for animal species in particular are as follows (Brook 2008; Hayes & Barry 2008; Blackburn *et al.* 2011; Essl *et al.* 2011):

- Climate matching
- Propagule pressure
- Mobile species are able to overcome geographical barriers and therefore are more likely to become invasive.
- History of invasive success elsewhere.
- Large distributional area in its native range.
- Human affiliation.

It should also be noted that many of these factors interact with each other. For example, many invertebrates are spread as a result of the pet trade which in turn is closely linked to human affiliation and results in a large propagule pressure as many animals are released at multiple sites over a broad geographical area over a long period of time. Indeed, the human element in landscape change (e.g. land use change and disturbance), in relation to invasive species ecology is often under reported but is increasingly important (Essl *et al.* 2011). Human affiliation has been shown to be one of the most consistent and strongest factors associated with a species becoming invasive. In a study carried out across Europe and North America(Jeschke & Strayer 2006) that teased out all of the determinants of invasion relative success game species and domestic species were the most successful invaders. Figure A1 is taken from this study. Also animals with a large native geographical range are likely to be adapted to a broad spectrum of climatic conditions, resulting in the species being pre-adapted climatically to novel areas they are being introduced to

(Brook 2008). Thus most authors would now agree that simply looking at biological traits of invasive species and the invasibility of recipient habitats is too simplistic as most factors work together additively or synergistically (Didham *et al.* 2007).

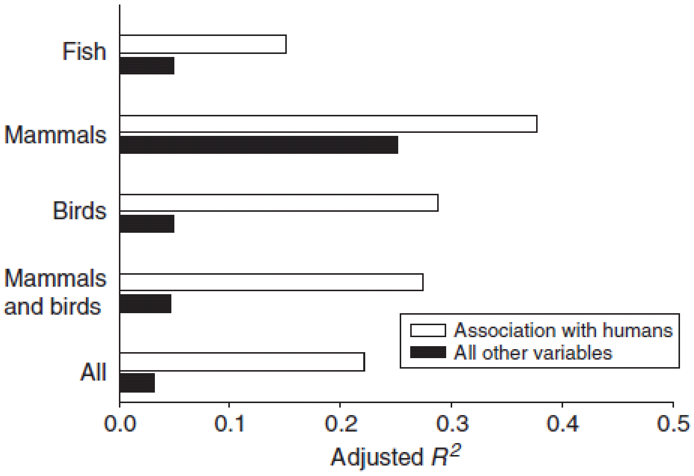

Figure A1. From (Jeschke & Strayer 2006) showing the importance of human affiliation with invasion success.

There are four principal stages of a non-native species invasion: (i) transport, (ii) introduction, (iii) establishment and (iv) spread (Kolar & Lodge 2001; Hellman *et al.* 2008; Blackburn *et al.* 2011). Transport involves the entrainment of a species in its native range within a transport pathway (e.g. indirectly within a ship's ballast water; directly via international trade as a commodity). Introduction involves the initial entry and colonisation of a new habitat. Establishment involves the initiation of breeding and the formation of a self-sustaining population. Spread occurs with the increase and expansion of the invader's range.

Between each sequential step of the invasion process there are barriers to a successful invasion (Hellman *et al.* 2008; Blackburn *et al.* 2011). In Blackburn *et al.'s* (2011) 'unified framework' for biological invasions, six barriers are recognised: (i) geography, (ii) captivity or cultivation, (iii) survival, (iv) reproduction, (v) dispersal and (vi) environmental. For example, for species in captivity or cultivation there is the barrier of containment between the species and the wild; for establishment there is the barrier of population growth – a species introduced into the wild may be able to survive and reproduce but may fail to establish due to negative long-term population growth.

Movement through each of the invasion stages is influenced by different life history strategies of the invading species (Kolar & Lodge 2001). Key traits at introduction are dispersal ability (which links to landscape structure) and competitiveness (competitiveness is involved in many stages of the invasion process and is an important determinant of invasion success (Valosaari 2008)).

Dispersal ability and structure of the landscape are also important factors in determining how far a species can spread as is reproductive efficiency.

8.3. Landscape

Modern human activities are considered the major source of non-native species introductions, either deliberately or accidentally, through trade and travel (Hulme 2009). Further, human shaping of the environment through land use patterns and habitat disturbance may facilitate the establishment, spread and invasiveness of non-native species.

Animal movement is required for survival and/or reproduction, including the avoidance of predators and other dangers, the acquisition of nutritional resources, to seek mates and reproductive opportunities, social interactions and the avoidance of intraspecific competition. Movement involves a trade-off between these potential benefits and the potential costs (e.g. increased energy expenditure and mortality risk); the relative risks and benefits being influenced by the structure of the landscape (Fahrig 2007). Different habitat types of varying quality in the landscape and the spatial relationship between habitat and surrounding matrix present different levels of risk and benefit. These risks and benefits will vary between species and hence different species will exhibit variation in their movement parameters, i.e. their probability of movement, the distance moved, propensity to cross boundaries and the shape of their movement path.

Wilson *et al.* (2009) distinguished six general symptomatic types of extra-range dispersal pathway: leading edge dispersal, corridors, jump dispersal, extreme long-distance, mass dispersal and cultivation.

Increasing connectivity of the landscape is thought to be beneficial to the conservation of native species as it develops an ecological network that allows movement through the landscape (including migratory, dispersal and genetic movement) often between designated sites and can also be provide access to resources which are limited at a specific location such as foraging or nesting resources. The importance of ecological networks has been recognised at a European level (EU Habitats Directive, 1992[1]), and for England within the Natural Enviornment White Paper: "We want to promote an ambitious, integrated approach, creating a resilient ecological network across England" (HM Giovernment 2011) and the National Planning Policy Framework (2012[2]), where it states that:

"The planning system should contribute to and enhance the natural and local environment by minimising impacts on biodiversity…contributing to the Government's commitment to halt the

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[1] http://ec.europa.eu/environment/nature/legislation/habitatsdirective/index_en.htm

[2] https://www.gov.uk/government/publications/national-planning-policy-framework--2

overall decline in biodiversity, including by **establishing coherent ecological networks** that are more resilient to current and future pressures"

The importance of ecological networks increases when considered in light of climate change. For any particular species there are limits to the abiotic conditions that it can tolerate; all species are restricted in range and are found associated with certain climates and environments but not with others (Willamson 1996). Shifts in species ranges, consistent with the effects of climate change have already been recorded (Morecroft and Speakman (2013) Austen & Rehfisch 2005, Franco *et al.* 2006, Hickling *et al.* 2005, Thomas *et al.* 2001) and more are expected to occur in Britain (Lawton *et al.* 2010). These range shifts can only occur if the species in question can travel through the landscape to new areas and subsequently establish viable populations here.

Since the formulation of the theories of island biogeography (MacArthur & Wilson 1967), conservation managers have attempted to maximise the benefits yielded by following the broad principles of increasing reserve size, connectivity and habitat heterogeneity to maximize the number of species housed within. Although some authors believe the generic rule to be over simplistic (Simberloff 1984), and underestimating the influence of human disturbance (Chown, Gremmen & Gaston 1998), there has been a resurgence in adopting these rules as pressure on wild landscapes increase (Lomolino *et al.* 2006). For England, this has culminated in a review carried out for Defra looking at how best to manage the existing network of reserves so that they are resilient and maximise biodiversity (Lawton *et al.* 2010).

Lawton *et al.'s* 2010 report 'Making Space for Nature' strongly recommended creation of such an ecological network in England. Any network would consist of five components; i) core areas (areas of high conservation importance due to the habitat or wildlife present), ii) corridors and stepping stones (areas that connect the network and allow movement between core areas), iii) restoration areas (areas planned to become core areas once management has taken place), iv) buffer zones (act as protective area around components i to iii) and v) sustainable use areas (areas used for economic gain whilst maintaining ecosystem services). A number of ecologists have also recognised the need to connect even lower grade habitats in order to increase their potential benefits to biodiversity (Kelly & Sullivan 2010).

Landscape barriers to dispersal are important and will vary depending upon spatial scale. Roads, rivers, hedgerows, train lines, mountain ranges are all barriers to dispersal, but their impact will vary between species and taxa. For example a feature that is a barrier to a plant (e.g. a fence or a hedge) may not be a barrier to a bird species. It must also be recognised that what represents a barrier to one species may be a pathway or corridor for another. Barriers can mean that a species' realised (actual) geographical distribution is smaller than its potential (predicted) geographical distribution i.e. it cannot disperse to the geographical limits of its ecological niche.

Landscape management such as deliberate fragmentation may be a useful strategy under some circumstances to halt the spread of invasive species (With 2002; 2004) especially if the species are reliant on high-quality habitats to disperse and spread, and are at an early stage of the invasion process. Indeed, for some habitat specialists such as wallabies (Potter *et al.* 2012), habitat connectivity has been shown to be far more important than any physiological traits for the survival and long-term viability of their populations.

Figure A2 is taken from With (2002), and highlights how each stage of the invasion process is influenced by the landscape ecology of the species in question. This ranges from the indirect effects of landscape at the production stage, including how the frequency of introduction could be influenced by human presence and levels of disturbance (for example in urban environments), all the way through to the direct influences of habitat fragmentation and connectivity influencing the species spread across the landscape as a whole.

England’s Environmental Stewardship schemes have a key role to play in the management of land in order to facilitate creation of this network. Lawton (2010) highlighted the Higher Level Stewardship (HLS) as the single most important method of land management that would be needed to develop any network. The impact of Environmental Stewardship on invasive species will be considered in Work Package 4 and reported on in the final report.

It is expected that increasing the connectivity of the landscape will increase the resilience of England’s biodiversity to climate change by creating routes of suitable habitat in landscapes through which species can disperse, by increasing patch size to increase the viability of populations and number of propagules able to disperse, or by increasing the quality of habitat. This increased connectivity can also reasonably be expected to improve the dispersal of non-native species, allowing them to become invasive (Donald & Evans 2006). For example, habitat corridors can be effective for both “r” selected species and “K” selected species in the short and long term respectively (Hudgens & Haddad 2003), regardless of whether or not the species that is spreading is of conservation concern or is problematic to conservation such as an invasive species.

Assessing the way in which invasive species use the landscape may highlight potential methods to control their spread whilst allowing native flora and fauna to disperse, or to aid in identifying the trade-offs between the benefits to native wildlife and the risks from invasive species.

The different aspects of ecological network design from Lawton *et al.* 2010 can be placed into four large categories as follows:

- Bigger (increasing the size of existing areas or adding buffer zones)
- Better (increasing the quality of existing sites)
- More (increasing the number of sites)
- Joined up (creating connections between existing sites)

The landscape section of this report will focus on these four aspects and how they impact on each group of invasive species (birds, plants, invertebrates, mammals and herptiles).

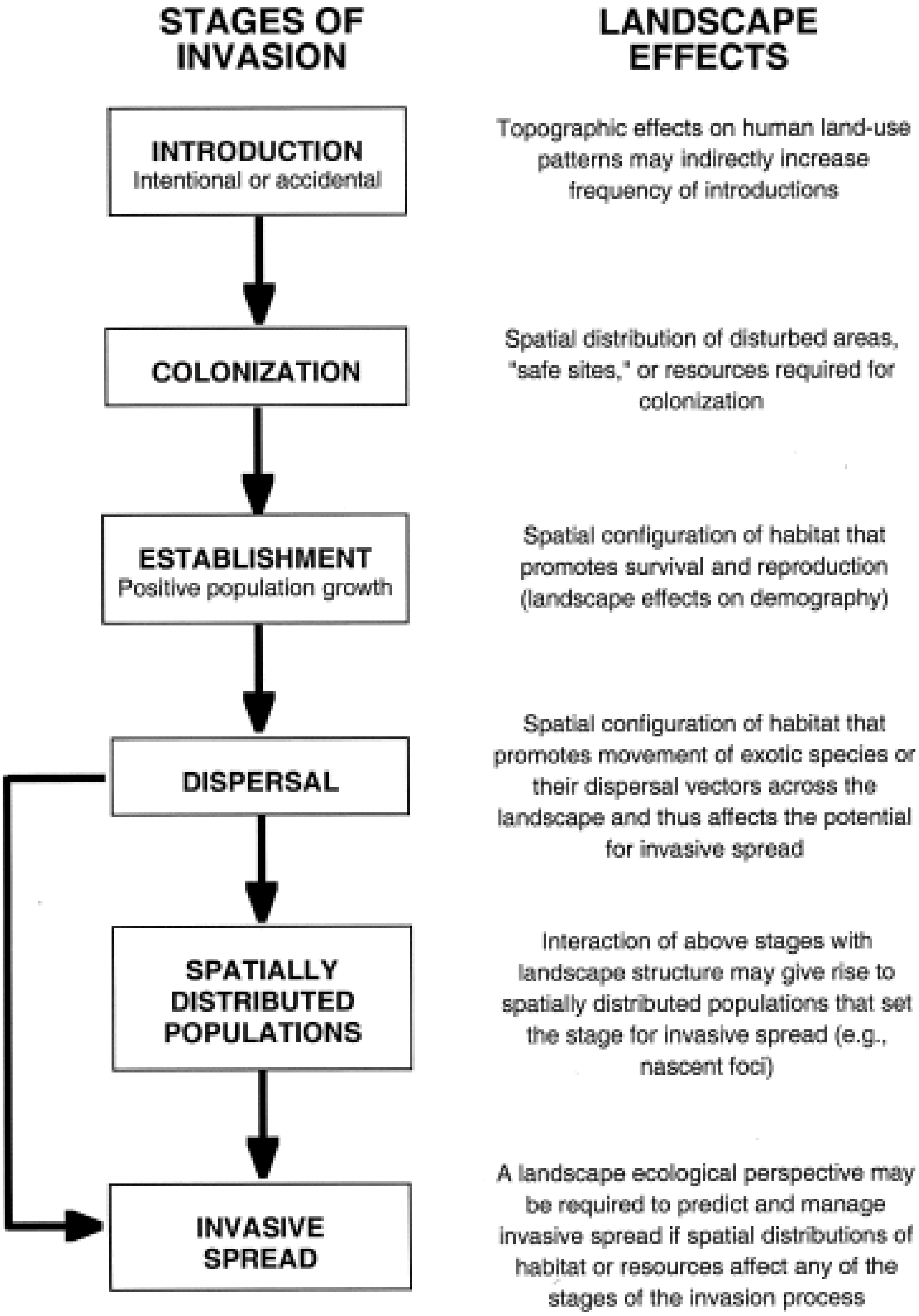


Figure A2. The influence of habitat fragmentation on invasion taken from With 2002. The transport stage occurs before the introduction stage in this diagram.

8.3.1. Landscape - birds

The occurrence of avian species and their ability to spread through a particular landscape is influenced by the size, quality and inter-connectivity of preferred habitat patches (Opdam 1991, Williams 1964 cited in Opdam 1991, Hinsley 1996, Prugh et al. 2008, Shake *et al*. 2012 . This encompasses populations within individual patches at the local scale (sub-populations) and metapopulations at the landscape scale - metapopulations consist of distinct sub-populations separated by space or barriers and connected by dispersal movements (Opdam 1991).

Fragmentation of habitat will affect different species in different ways. In woodland, for example, for woodland-interior species the areas between fragmented woodland patches will be unsuitable or even inhospitable; and may act as a barrier. For woodland edge species, however, the fragmentation will increase the availability of preferred habitat (Opdam 1991).

Bigger

In studies of metapopulations of holarctic breeding birds in fragmented agricultural landscapes, the probability of occurrence and the local extinction rate in sub-populations depended on factors that included the size of the habitat fragments (Opdam 1991). Patch size can affect occurrence through a relationship between patch area and heterogeneity: the larger the fragment the greater probability that the species will find its preferred habitat (Williams 1964 cited in Opdam 1991). A greater probability of extinction in smaller patches could be related to the relationship between patch size, carrying capacity and minimum viable population; smaller populations are more at risk from demographic and environmental stochasticity.

Another factor associated with smaller habitat patches is an increase in the relative length of edge habitat (compared to interior habitat) and the potential for higher risk of predation. An increased nest predation rate owing to edge effects is often cited as an explanation for declines in bird populations in fragmented landscapes; although alternative studies have not supported this interpretation (Evans 2004). In Sweden, Andren (1992) investigated the abundance and distribution of corvids and predation rates on dummy nests, in relation to forest fragmentation across a landscape gradient from an agricultural land dominated landscape to a forest dominated landscape. The density of corvids and predation rates increased as the proportion of agricultural land increased, i.e. predation was higher in smaller woodland fragments. Angelstam (1986) suggest that the main factor affecting the rate of predation in fragmented environments is the steepness of the productivity gradients between a habitat island and the surrounding matrix rather than the patch size itself. That is, predation rates will be higher in areas where fragmented woodland, for example, is surrounded by habitat with high general productivity (e.g. a high proportion of agricultural land) and thus more food for generalist predators. High densities of general predators are often considered to be a consequence of human activities, such as agriculture (Angelstam 1986).

Chalfoun *et al*. (2002) reviewed nest predator responses to habitat fragmentation. The review found that nest predators were more likely to show a positive response to fragmentation (increased abundance, activity, or species richness) when fragmentation was measured at the landscape scale than at finer scales. Responses at local scales were more variable and dependent upon the landscape context within which the study was conducted. Chalfoun *et al*. (2002) concluded that the response of nest predators to fragmentation was both taxon and context specific. Consequently management measures to preserve declining species may need to be customised to the local predator assemblage and nature of the landscape mosaic.

Stephens *et al*. (2003) reviewed the effects of habitat fragmentation on nesting success. Overall, the proportion of studies detecting relationships between habitat fragmentation and nesting success increased as the scale at which habitat fragmentation was measured increased from edge, to patch, to landscape scale. The study concluded that habitat fragmentation at larger scales may affect nesting success more than fragmentation at smaller scales. One recognised consequence of this was that management of avian populations will have to consider management of human development in order to minimise human-induced fragmentation.

If fragmented habitat, particularly that in relative proximity to human activity, does provide enhanced nest predation opportunities this would be beneficial to some of the selected invasive non-native birds, such as the Indian house crow and common mynah.

Estimates of species area requirements have been examined in a number of different habitats. For shrubland birds, in the USA, the probability of occupying an individual patch of habitat increased with patch size (Shake *et al*. 2012). Two species, Yellow-breasted Chat *Icteria virens* and Prairie Warbler *Setophaga discolour* were shown to be area-sensitive with minimum requirements of 2.3 ha and 1.1 ha respectively. The study recommended that when shrubland habitat is created or managed that patches should be >5 ha to ensure habitat use by the entire community of shrubland birds. It was also considered, however, that this area only addressed the issue of species' occupancy of a patch and did not consider a patch's ability to support a high density and productive breeding birds; consequently it was considered likely that patches may need to be even larger.

For grassland birds, in the USA, the perimeter–area ratio, which reflects both the area and shape of a patch, was the strongest predictor of both individual species presence and overall species richness (Helzer & Jelinski 1999). It was concluded that species richness is maximized when patches are large (>50 ha) and shaped so that they provide abundant interior areas, free from the impacts of edges.

In English woodlands different species showed different probabilities of breeding in relation to woodland area (Hinsley 1996). Species (including tree creeper *Certhia familiaris* and marsh tit *Parus palustris*) largely dependent on woodland (occurring only rarely, if at all, in habitats such as hedgerows or gardens) the probability of breeding only approached 100% for woods >10ha. Species less dependent on woodland (e.g. blackbird *Turdus merula* and robin *Erithacus rubecula*) the probability of breeding approached 100% for woods of 1ha or less. For some species the relationship between the probability of breeding and woodland area changed with changes in regional abundance. In chiffchaff *Phylloscopus collybita*, for example, when regional abundance was reduced the probability of breeding shifted away from smaller woods so that only larger woods were occupied and returned to include smaller woods when regional abundance recovered.

Better

In Britain, the effects of structural gaps (physical spaces in the tree canopy) and functional habitat gaps (differences in the quality of different plant species as foraging habitat) on parental energy expenditure and breeding success in great tits *Parus major* and blue tits *Cyanistes caeruleus* was investigated by comparing breeding pairs in urban parkland and continuous woodland (Hinsley *et al.* 2008). The presence of gaps increased travel time and distances of foraging trips as a result of both greater distances to suitable foraging habitat and also potentially to anti-predator flight patterns (e.g. non-direct crossing of gap). The study found: both species reared fewer young in the park; great tits in the park had higher daily energy expenditure than those in the wood; in the park female tits with more patchy habitat around their nest boxes worked harder; similarly great tits in the wood with less oak around their boxes worked harder.

More

The relationship between the incidence of three forest bird species and patch characteristics and landscape context was investigated in fragmented Brazilian Atlantic Forest; involving eighty forest patches in four landscapes (Boscolo & Metzger 2011). The presence of all three species in a forest patch was in general positively affected by the amount of surrounding habitat and negatively affected by inter-patch distance. Local species presence is in general more influenced by isolation from surrounding forests than by patch size alone. In highly fragmented landscape birds that cannot find patches large enough to settle may be able to overcome short distances through the matrix to include several nearby patches within their home ranges (Boscolo & Metzger 2011).

In North America, pectoral sandpipers *Calidris melanotos* exhibited more frequent movements between individual wetlands as the proportion of the landscape composed of wetlands increased (Farmer & Parent 1999). The greater proximity of wetlands allowed the birds to exploit more feeding opportunities without incurring increased search costs.

The effect of having more sites is inter-linked with the size of those sites and the degree of connectivity between sites. Prugh *et al*. (2008) conducted a meta-analysis of 785 animal species globally including 370 avian species examining the effect of patch area and isolation on species occupancy of fragmented terrestrial systems. Patch area and isolation were indeed important affecting the occupancy of many species but properties of the intervening matrix was also important. It was recommended that improving matrix quality may lead to higher conservation returns than manipulating the size and configuration of remnant patches for many of the species that persist in the aftermath of habitat destruction.

The creation of more patches, therefore, needs to be considered in the context of their size, shape, connectivity and surrounding landscape.

Joined up

Some forest dwelling species are impacted by habitat connectivity particularly in tropical regions. The relative permeability of three landscape types (open habitat, shrubby secondary vegetation and wooded corridors) to movement by the Chucao Tapaculo *Scelorchilus rubecula* (a territorial bird preferring dense forest understory habitat) was investigated using a translocation experiment in South America (Castellon & Sieving 2006). Subjects remained in release patches surrounded by open habitat significantly longer than subjects released into patches surrounded by dense shrubs or adjoining wooded corridors.

Ibarra-Macias *et al.* (2011) carried out a dispersal challenge to investigate the gap-crossing proclivities of six species of tropical forest birds in a fragmented forest in Mexico**.** Individuals were released in a cattle pasture at different distances from a forest edge. Movement and behaviour following release were measured in terms of latency, orientation and direction. Gaps as narrow as 50m affected movements; gaps 100m and larger presented a threshold distance beyond which birds are less likely to attempt and successfully navigate. Three forest-restricted species (white-breasted wood-wren *Henicorhina leucostica*, black-faced ant-thrush *Formicarius analis*, Woodcreeper *Glyphorhynchus spiurus*) showed greater latency to cross gaps compared to forest-unrestricted species (red-throated ant-tanager *Habia fusticauda*, orange-billed sparrow *Arremon aurantiirostris*, ochre-bellied flycatcher *Mionectes oleagineus*).

Variation in Neotropical bird communities was investigated in limestone forest patches embedded in three different human-dominated matrix types (agriculture, peri-urban development and bauxite mining) relative to patches in continuous forest in central Jamaica (Kennedy *et al.* 2010). The abundance of almost 70% of species differed between patches embedded in the different landscape types. Species with lower abundances in patches embedded in human-dominated landscapes than in continuous forest were insectivores, frugivores, canopy nesters, understorey and canopy foragers and forest-restricted species. In contrast, nectivores, omnivores, granovores, ground and multi-strata nesters, ground foragers and forest-unrestricted species were least sensitive to forest fragmentation.

A more fragmented habitat is associated with increased edge habitat. As discussed earlier, increased nest predation rate at habitat edges is often cited as an explanation for declines in bird populations in fragmented landscapes. In a review of 55 empirical studies Lahti (2001), however, concluded that although an edge effect on nest predation occurs in some sites it is not a valid generalisation. An edge effect has been exhibited slightly more often in landscapes with high fragmentation than low. It was concluded that studies exhibiting an edge predation effect explained their results by species-specific predator behaviour and its relation to habitat and landscape features rather than habitat edges *per se*

Factors influencing the movement of birds through the landscape are reviewed in Appendix F. Here we consider invasive species specifically.

For the majority of invasive bird species landscape connectivity is less important than for less mobile species as many invasive birds possess the potential to undertake large dispersal distances allowing them to easily cross any potential gaps in habitat. An example of an invasive species bridging unfavourable habitat is the ring-necked parakeet. Parakeet distribution is primarily governed by the amount of older forests, parks and built-up area in the landscape, reflecting the parakeets' requirement for suitable nesting cavities and reliance upon urban areas to forage. However, despite its reliance on some specific habitat features, in England the species is undergoing a period of rapid population growth. The observed growth rate over recent years is amongst the highest observed in current British bird populations, Breeding Bird Survey data indicates that the population grew by as much as 600% between 1995 and 2007 (Baillie *et al.* 2009) and almost 700% over the 13 years 1995–2008 (http://www.bto.org/birdtrends2010). Butler (2003) observed that parakeets were now being seen in rural areas rather than the previously typical urban and semi-urban areas, and that it may not be long before they spread into the British countryside (Butler 2003). Between 2004 and 2009, the mean rate of spread of parakeets out of London was 2.89km per year (analysed along 15 vectors – 'directions'?) (Parrott *et al.* 2011). In this analysis there was considerable variation in both annual spread and the overall distance moved along each of 15 individual vectors. Even vectors with slow net rates of progress over the five-year period showed significant annual advances - the maximum annual advances along any of the fifteen vectors ranged between 4km and 13km with a mean of 8.1km. This suggests that parakeets have the potential to make considerable annual advances across apparently inhospitable landscapes even where mean rates are low. A recent report published by the Arthur Rylah Institute for Environmental Research (Morris 2012) found that waterbirds, due to their large dispersal distances, are unconstrained by landscape connectivity (although distance between foraging, roosting and nesting habitat may be of importance during certain life stages). This is also true of species that are likely to impact England.

8.3.2. Landscape - plants

The British Landscape is very heterogeneous and a spreading species will face multiple establishment events under ever broader range of environmental conditions. The spread is controlled by the size and distribution of suitable habitats patches, the distance between these patches, and the population characteristics, growth rates and dispersal ability of the non-native plant (Theoharides and Dukes, 2007). Each new patch requires the species to go through the establishment stage again and surmount the barriers of survival and reproduction, before spread can continue. The invasion range is limited by the extent of suitable environments encountered. Invasions can fail even after spread reflected in ‘boom and bust’ dynamics that can even lead to extinction (Simberloff and Gibbons, 2004).

Bigger

Humans have fragmented landscapes into discrete patches within a matrix of human land use. Large patches may benefit natives as they allow for heterogeneity and therefore diversity and are more likely to be more stable. This means that they are more buffered from the invasion process. However the reverse may be true of smaller patches as they may be easier to invade as the edge of these patches, the most vulnerable zone, represent a greater area of the patch (Ohlemuller *et al.*, 2006). A literature review by Vila and Ibanez (2011) of the influence of the surrounding landscape on the local level of plant invasions has shown that alien plant species are more abundant at fragment edges than in the interior and that the decline in density, richness and cover towards the interior is exponential and sharp near the boundary. The maximum distance of edge influence ranged from 10 to 225 m in 17 studies (e.g. (Watkins *et al.*, 2003; Hansen and Clevenger, 2005; Cilliers *et al.*, 2008). These edge effects are more pronounced in small patches and increased light, space and soil moisture may favour non-natives (Parendes & Jones, 2000), though this can be lower in edge habitats in woodland (Riutta *et al* 2010). Edge areas also experience more propagule pressure.

Better

Non-natives can capitalise on disturbance events that eliminate their native competitors. Alterations of natural landscapes may favour species that have coevolved with human land use and disturbances (D'Antonio *et al.*, 2000; Vermeij, 2005). The edge effects discussed above are further exacerbated if the habitat patches are near developed edges as these are a major source for non-natives. Improving habitat quality through ecological restoration could increase the resistance of a landscape to invasion by non-native plant species. Bakker and Wilson (2004) suggested that resistance to invasion depended on the species in the native community and that restoration seed mixes may be tailored to constrain non-natives. They chose their restoration grassland seed mix from the same functional group as the non-native grass and it comprised representatives of later successional stages. The results of the study showed that the non-native cover was much more strongly negatively correlated with planted grass cover than with species richness. This supports the idea that successful invaders replace native species in their niches rather than occupying empty niches (Shea & Chesson, 2002).

Non-native species disperse best through strips of human-disturbed habitat such as road verges and railway embankments. Assessing the risk of invasive spread in fragmented landscapes requires understanding of the relative effects of its structure on the processes that contribute to invasive spread. Colonisation success is predicted to be highest when greater than 20% of the landscape has been disturbed particularly if these are large or aggregated in space (With, 2004). Roads, railways and building construction continue to take place, and will continue to do so, and therefore will constantly increase the urban/countryside interface (Theobald & Romme, 2007).

More

A range of distinctive plant traits i that enhance competitive performance, reduce niche overlap or increase herbivory resistance increase the chances that non-native plants will disrupt, intensify or suppress the natural disturbance regime along with anthropogenic activities (Hobbs, 2000; D'Antonio and Hobbie, 2005). Enhanced physical attributes such as fast growth, fecundity, leaf size, stem height, vegetative reproduction, self compatibility and flowering phenology all improve the success of establishment (Goodwin *et al*., 1999; Lloret *et al*., 2005 Dietz & Edwards 2006; Richardson *et al* 2000a), along with production of secondary chemicals such as root exudates, the formation of novel mutualisms with native-nitrogen-fixing bacteria that increase establishment success and change the availability of soil nutrients (Callaway *et a*l., 2004) and pollination by generalists along with high floral resources (Richardson *et al.* 2000a)..

Changes that increase resource availability or change landscape pattern can promote non-native spread by creating favourable patches (Burke & Grime, 1996; D'Antonio *et al.*, 2000). Fire-adapted Old World grasses have come to dominate many North American grasslands; as a result of greatly increased frequency and intensity of fires, they have eliminated or reduced the abundance of natives over large areas (D'Antonio and Vitousek, 1992).

The creation of new habitat requires disturbance and therefore leaves the new area vulnerable to invasion. It will be important to ensure that soils are nutrient-poor to reduce the advantage invasive non-natives may have in these environments. It is also important to realise that invasive non-natives are likely to exhibit the same set of traits as do invasive native species (Thompson *et al*, 1995), that non-natives and natives exhibit similar successional dynamics (Meiners, 2007) and that natives of disturbed fertile habitats represent as great a threat as that from aliens of similar habitats (Leishman *et al.* 2010).

Joined up

For the spread stage the invasive non-native will typically be one with a competitive ability, high fecundity, fast growth and generation time, the efficient use of resources, phenotypic plasticity, small seed size, and an effective mode of dispersal both long- and short-distance (reference?). Connectivity of suitable patches influence dispersal of non-natives and their metapopulation dynamics and is important in the spread and persistence across the landscape (Theoharides and Dukes, 2007).

Corridors between patches provide transport for natives and non-natives across unfavourable landscape facilitating interactions between populations (With, 2002). Humans play an important role, and roads and railway networks are useful means for plants to move within their new range. They form "corridor pathways" that act as invasion conduits into both fragmented and undisturbed landscapes (With, 2002; Hulme *et al.*, 2008).

The scheme of Blackburn *et al.* (2012) allows non-natives to be simply categorised and compared based on how far along the unified framework they have progressed. Species can have several populations at different locations on the framework. There is a constant interaction of invasion barriers with the traits of the invader, and these are further affected by the number of introduction events and the characteristics of the ecosystem (Theoharides and Dukes, 2007). The emergence of populations in new locations depends on establishment success, the connectivity of viable habitat patches, and the mode and pattern of dispersal. The rate at which a species spreads depends on many system- and species-specific factors, and these factors make it difficult to form broad generalizations. Nevertheless, the presence of human-disturbed habitat or disturbance corridors across the landscape can be important in terrestrial systems (D'Antonio *et al.*, 2000). These include roads, railways and canals and can facilitate rapid non-native dispersal due to the presence of disturbed spoil, high light, altered hydrology and the destruction of the native seed bank (D'Antonio *et al.*, 2000; Trombulak & Frissell, 2000). Native plants often require wide undisturbed corridors of intact habitat (D'Antonio *et al.*, 2000; Parendes & Jones, 2000).

The pattern of the landscape is less important if the species is capable of the occasional long-distance dispersal event. Levey *et al.* (2008) used a spatially explicit model to predict the distribution of seeds dispersed long distances by birds into habitat patches of different shapes, which had been validated in a previous study of seed dispersal in real landscapes. When landscapes are comprised of different habitat types, the behaviour of the bird is modified upon encountering habitat boundaries and dispersal distances become irregular. This difference is likely to have been caused by dispersers being attracted to patches and spending more time in them than in the matrix, thereby reducing the distance seeds are dispersed. However long-distance dispersal, in this case flights of >150 m, characterised by seed movement well beyond the bounds of the parent population, still constituted 50% of the total movements. Therefore this work has shown that heterogeneity provided by connectivity does limit some long range dispersal but does not prevent all large scale movement between patches.

Damschen *et al.*, (2006) have shown through a large-scale replicated study that these corridors need not be vulnerable to invasion. The study areas were patches of rich herbaceous understorey surrounded by dense pine plantations. Each landscape consisted of a central patch (100 x 100 m) and four surrounding patches 150 m away. One of the four surrounding patches was connected to the central patch by a corridor 150 m by 25 m. Their results showed that habitat patches connected by corridors retain more native plant species than isolated patches, that the difference increases over time and those corridors do not promote invasion by non-natives. The reasons for their success may be that corridors promote colonisation by increased seed deposition (Tewkesbury et al 2002; Levey et al 2005), promote within-patch recruitment by increasing pollen movement (Tewkesbury et al 2002; Townsend and Levey, 2005) and alter foraging by seed predators that could benefit species otherwise more likely to be excluded by seedling competition (Orrock *et al.* 2006).

8.3.3. Landscape - invertebrates

Bigger

The relatively small size of invertebrates means that populations can survive in relatively small areas, therefore the presence of larger blocks of potential habitat is less important than it is likely to be for vertebrates. Bigger areas of habitat will lead to larger populations from which invasive invertebrates could spread, however, the impact of this factor is considered to be minor in most cases. Nine of the ten invertebrate species selected for the case studies in this project (Section 3.5) are phytophagous. The potential distribution, rate of spread and population size for these species will be determined by the distribution of their host plants whether or not they are in particular habitats. All of the phytophagous species with the exception of juniper shieldbug, *Cyphostethus tristriatus* and cypress groundling,*Gelechia senticetella* have very widely distributed hosts. Therefore having bigger patches of host is unlikely to have much of an impact. The hosts of *C. tristiatus* and *G. senticetella* are junipers, which are not widely distributed in the environment, and Lawson's cypress, *Chamaecyparis lawsonoiana* which are commonly grown in gardens. If there were larger patches of juniper in moorland habitats, this could lead to greater population of these species. The one species chosen for a case study that is not phytophagous is *Lasius neglectus*, an ant. Ants are omnivores and this species is not specialised to any habitat.

Better

Changes in habitat quality are unlikely to have any impact on the potential invasiveness of the ten species selected, they are not reliant or not known to be reliant upon having hosts in any particular condition. Insects can use new, highly productive habitat to increase their population size and therefore their dispersal into fragments of natural habitat (Evans *et al.* 2011). Persistence of invertebrate species in habitats is another factor which affects their invasiveness. The persistence of a species can vary between habitats or they may be persistent in a wide range of habitats. For example, non-native *Cheiracanthium* spiders are better adapted to disturbed habitat than their native relatives and this increases their dispersal rate (Hogg, 2011).

"Better' includes factors that can be altered by habitat management. In the example of oak trees and their susceptibility to gypsy moth this is not relevant, and susceptibility in Sharov et al. 1999 seems to refer to species composition. Regarding the example of *A. planipennis*, it 'will preferentially colonize stressed ash trees (McCul-lough et al. 2009a, 2009b), but healthy trees growing under optimal conditions are also attacked and killed (copied from McCullough and Mercader, 2012, Int. J. Pest. Management, 58: 9-23). Therefore, it will attack healthy and stressed trees, and the factors that stress trees in non-urban environments are unlikely to be factors that can be altered by habitat management. In the instance of *Anoplophora*, it can attack healthy and stressed trees.

More

Having more habitat patches could be important for species with low natural dispersal ability as these areas would be able to act as ‘stepping stones’ allowing the species to move across the landscape. However, all ten species selected in Section 3.5 either have a high natural dispersal ability or their hosts / habitats are very widespread already, therefore increasing the number of areas of habitat is unlikely to be significant. Having more areas of habitat is only likely to be significant for speciesthat have a low natural dispersal ability and which attack hosts which are widely dispersed.

Joined up

The extent of fragmentation of the landscape is important for these taxa: insects exploit both natural habitat fragments and fragments of new/disturbed habitat, however these fragments can only be invaded if they are within the range of the species’ dispersal distances. For the species selected in Section 3.5, joining up habitat patches is unlikely to have a large impact due to their high dispersal ability or the widespread of their hosts / habitats. As above, joining up areas of habitat is only likely to be significant for species that have a low natural dispersal ability and which attack hosts which are widely dispersed.

8.3.4. Landscape - mammals

Bigger

Increasing the area available to a species leads to an increase in population size ;the larger the species in question, the more space it needs (Connor, Courtney & Yoder 2000). Most invasive species however, are able to withstand a degree of habitat fragmentation and range constriction, and in some cases have even adapted to urbanization in their native ranges as with muskrats (Cotner & Schooley 2011). Indeed, often other factors such as hunting or disturbance pressures are to blame for limiting population growth in species such as coypu (Guichon & Cassini 1999; Leggieri, Guichon & Cassini 2011), coati (Beisiegel & Mantovani 2006; Sanchez, Sanchez-Palomino & Cadena 2008) and even habitat specialists such as pudu (Beisiegel & Mantovani 2006; Meier & Merino 2007). Leopard cats for example are extremely adaptable and are unusual among the wild Felidae in their ability to occupy extremely small home ranges by exploiting a greater dietary width (Watanabe 2009).

Better

Invasive mammals, especially those that are generalists, are able to exploit a range of habitats, including disturbed habitats. By default many invasive mammals are generalists, and are thus able to exploit a range of food and habitat resources and are thus less likely to be limited by patch size. A good example of this is the raccoon which can achieve much higher densities in human disturbed areas such as urban parks, than in natural and semi-natural landscapes (Riley, Hadidian & Manski 1998). However, species behaviour in novel habitats can be unpredictable. Striped skunks, unlike other skunk species for example, in their native range often avoid edges and disturbed habitats, while in areas where they have been introduced to can often invade urban and other disturbed habitats (Lariviere & Messier 2000; Anon. 2009; Ordenana *et al.* 2010).

Species that are intermediate between generalists and specialist are able to exploit disturbed habitats, but also benefit from habitat of good quality. Raccoon dogs for example need berries and fruits at certain times of the year (prior to hibernation) and thrive in areas of high quality with the right mix of animal and vegetable food (Melis *et al.* 2010). Specialist species that are still invasive in Britain need a good degree of habitat quality, connectivity and area in order to thrive in their native ranges (Jung, Shimizu & Omasa 2011), which could explain their slow rates of spread across the UK. Other habitat specialists, the aquatic rodents, muskrats, capybaras and coypus require quality habitats with adequate accessibility to high quality food plants in their native ranges (Guichon *et al.* 2003; Schooley & Branch 2009; Corriale, Arias & Quintana 2011).

More

As with a number of other taxa, simply having more sites will benefit most species, regardless of whether or not it is invasive. Also it is hard to extract the effects of having more sites, from the effects of having better connectivity between sites. Due to their mobility, trying to understand meta-population dynamics is more difficult than it is for other taxonomic groups such as the amphibians, and finding meta-populations in nature is generally rare (Fronhofer *et al.* 2012). However there is evidence that the maintenance of a meta-population of a species is beneficial to populations persisting, growing and expanding their range. This has been seen with invasive grey squirrels in Italy (Martinoli *et al.* 2010), and with raccoons in Japan (Kotani *et al.* 2009). There is little evidence of this effect on other invasive mammals in the scope of this project, within the literature.

Joined up

Although highly mobile, habitat connectivity is beneficial to the spread of invasive species. This is particularly the case for habitat specialists such as the Chinese water deer (Ward 2005). Though difficult to tease out from the compounded effects of size, raccoon dogs in their native range benefit from areas of both improved size and connectivity (Soga & Koike 2013), and meso-predators (which would include coatis, skunks, arctic foxes and raccoons as well as raccoon dogs) are adept at exploiting the broad range of prey species using corridors (Frey & Conover 2006). Here, the corridor itself is seen as a valid, prey-rich habitat in its own right, as well as a "highway" to prey patches though which movement is faster. This has also been seen in raccoons in their native range (Barding & Nelson 2008), where the edge habitat is a "corridor habitat" in its own right.

Due to their high levels of mobility, mammals are less affected by landscape barriers than some other taxa. Features such as roads and motorways, high altitudes and rivers are easily traversable as are fragmented landscapes. All of the species selected in Section 3.6 are able to spread through the landscape regardless of such barriers.

8.3.5. Landscape - herptiles

Most authors combine reptiles and amphibians together as herptiles as their patterns of colonisation and movement are usually quite similar (Ferreira *et al.* 2012).

Bigger

As with all species increasing the area available to a species leads to an increase in population size (Connor *et al.* 2000). Amphibian densities in particular are closely linked with size of habitat patches (Myers *et al.* 2000) although there is a paucity of data on this (Martin & Murray 2011).

Better

Invasive amphibians often fare better in poor quality and disturbed habitats in comparison to native species. This has been in seen in bullfrogs in America (D'Amore *et al.* 2010). Indeed, they are able to exploit features in habitats that enable them to persist even in very low grade habitats that other amphibians cannot survive in, as in the case of *X. laevis* in Sicily (Faraone *et al.* 2008), thus facilitating their spread, and even opportunistically capitalizing on made features such as irrigation wells, as in the case of cane toads in (Florance *et al.* 2011).

Reptiles on the other hand vary in their degree of habitat specialism and ability to invade disturbed or low grade habitats. Some tropical lizard species (though not one of the species selected for this project) are able to withstand changes in climate if habitats are of a higher quality (Huang & Pike 2011), while *P. muralis* actually seeks out lower grade and smaller habitat patches with greater edge effects, in its native range in order to avoid encounters with predators (Rugiero & Luiselli 2006; Maura *et al.* 2011). Indeed, refugia from predators is the most important factor in their choice of habitat (Amo *et al.* 2007) and in its native range at least the species is able to exploit a number of different habitat types due to a highly catholic diet (Capula *et al.* 1993). In areas where the species has been introduced, such as in Ohio (Brown *et al.* 1995), the species modifies its density to optimise use of available niches.

The need to disentangle the effects of habitat quality from the presence of predators is apparent. As with wall lizards, *T. scripta* also show differences in ecology in different habitats, but this is more closely linked to the presence of predators, in this case alligators, rather than habitat size or quality (DeGregorio *et al.* 2012). Apart from its need to be close to water, In its native range the species is a habitat generalist and is competitively aggressive and able to out compete conspecific species, and in mainland Europe, the species has often out competes more specialist native freshwater turtles (Cadi & Joly 2003; Aresco 2010). It has also been suggested that *T. scripta* may be showing unpredictable niche shifts in areas where it has invaded, not conforming to distribution patterns as manifested by the species in its native range, something that has been suggested for a number of invasive species in novel habitats (Roedder *et al.* 2009). The larger turtle on the list is the alligator snapping turtle, whose distribution is also driven more by thermal conditions (Fitzgerald & Nelson 2011), followed by prey availability (Shipman & Riedle 2008) and habitat type (Harrel *et al.* 1996), rather than disturbance or habitat quality itself. Again this is information based on the species in its native range, and there is little information on the species in its introduced range.

Perhaps the most adaptable of the reptiles are Burmese pythons, with their spread limited mainly by temperature rather than habitat quality where the species is invasive (Pyron *et al.* 2008; Mazzotti *et al.* 2011; Jacobson *et al.* 2012). In its native range (for a similar species, the Indian rock python), prey availability rather than habitat disturbance, appears to have the greatest influence over distribution (Goursi *et al.* 2012).

More

If there are more sites available to any species to live in, then that species will do well. This is an ecological truism that does not warrant further investigation. However, that population only really does persist if these sites are interconnected and the population is then a meta-population with dispersal events linking them together (Fronhofer *et al.* 2012). Thus the benefits of having “more” sites cannot be disentangled from the benefits of improved connectivity. However empirical meta-populations are rare in nature. One of the few taxonomic groups that do present meta-populations in nature are amphibians (Bauer *et al.* 2010) and thus having more sites will benefit the establishment, growth, expansion and spread of all invasive amphibians. The only main difference between invasive amphibians over native species is that, as discussed in other sections, they are able to make use of sub-optimal habitats in order to move from one site to another, as is the case with alpine newts (Emaresi *et al.* 2011).

Joined up

For herptiles, roads act as barriers that will negatively influence the slower moving species. From the species selected in Section 3, it is believed that only cane toads and wall lizards benefit from road networks as they spread quickly over open rocky ground. Some less mobile reptiles and amphibians in particular are highly limited by habitat fragmentation (Ficetola *et al.* 2007; Driscoll *et al.* 2012). Elevation is also important for some species such as the alpine newt as they are adapted to living at high altitudes in continental Europe, and as a result are believed to be competitively favoured at higher altitudes.

Habitat preferences also limit the degree to which herptiles can move across the landscape. Turtles favour sandy habitats in which to lay eggs, wall lizards are tied to rocky areas whilst amphibians and reptiles in general are tied to areas close to water. Most species however are capable of persisting in a wide range of different habitat types.

8.4. Climate change

Anthropogenic climate change represents a growing threat to the environment and society in the coming years. It expresses a warming climate, more extreme weather events, rising sea levels, as well as unpredictability in crop success and increasing vulnerability of populations. Invasive species represent a current global threat that could be exacerbated by future climate change; for example under the warmer and wetter conditions predicted for the UK (Hulme 2010; HR Wallingford 2012; Natural England 2012).

The geographical range or biological impact of existing non-native species may be currently limited by climate and therefore go relatively unnoticed at present. Under future climate change, however, these species may be able to flourish, dispersing within and beyond their current range, leading to increased impact and economic cost (Natural England 2012).

Climate change has the potential to affect the invasion process through influencing both the compatibility of the invading species with its new environment (e.g. climate matching) and by altering the mechanics of the invasion stages. Hellman *et al.* (2008) specified five possible consequences of climate change on invasive species: (i) altered mechanism of transport and introduction, (ii) altered climate constraints, (iii) altered distribution of existing invasive species, (iv) altered impact of existing invasive species, and (v) altered effectiveness of management strategies. For example, climate change could alter the demand for certain non-native species commodities (e.g. plant nursery species) or the quantity of movements along existing or new international commercial and/or tourist routes, thereby increasing the numbers and/or diversity of non-native species transported. Climate change in the invaded area may facilitate progress of non-native species through the invasion pathway barriers, for example by enhancing conditions for survival and reproduction.

Early trends of this are already being seen in species that have a small thermal tolerance such as invertebrates (Knop & Reusser 2012). It is becoming increasingly apparent that climate is a major factor in the distribution of invasive species (Simberloff 2000). Warmer places around the globe have the highest concentration of invasive species (Simberloff 2000) and competition between species alters at different climates (Carmona-Catot *et al.* 2013). Thus any long-term planning to prevent the spread of invasive species should be integrated with current knowledge of the processes of climate change (Pyke *et al.* 2008).

Synergy between invasive species and climate change is predicted to lead to a reduction in biodiversity (Cooling *et al.* 2012). Climate change, species invasions and a more fragmented, non-natural landscape will lead to unpredictable interactions and homogenisation of biota (Thuiller

2007). Opportunities may arise that enhance the reproductive capacity, ability to survive and competitive edge of non-native species over natives that lead to increased invasiveness.

Conversely however, climate change may make current ‘at risk’ areas less suitable for invasive species as the abiotic parameters become too extreme (Bradley *et al.* 2008). This will be particularly important to plants as they are less able to disperse as fast as other taxonomic groups.

Climate change will lead to both direct and indirect changes in landscape character (Mitchell *et al.* 2007). Direct changes are those that are imposed on the environment as a consequence of altered weather patterns, including increases in average annual temperatures and rainfall and more frequent extreme weather events. Seven of the 32 priority habitats in the UK Biodiversity Action Plan were assessed to be at high risk from the direct impacts of climate change – montane habitats, standing waters, floodplain and grazing marsh, saltmarsh, maritime cliffs and slopes, saline lagoons and open seas. Indirect changes will arise as a consequence of changes in socio-economic drivers, practices, policies and resource use in the various land use sectors. For example, in agriculture, there are predicted changes in crop type and regional patterns of crop planting and harvesting, e.g. an increase in vineyards and other soft fruits currently grown further south, and regional changes in cereal production. For freshwaters and wetlands, in addition to the direct impacts of climate change on hydrological regimes, will be affected by catchment management, including water abstraction and flood management. These changes in landscape use will affect the ability of species (native and non-native) to utilise and move through landscapes.

Invasives have been shown to be more flexible and adaptable than natives; therefore their potential niche is much larger and more dynamic (reference?). For example, invasive plant species are more effective at tracking seasonal temperature variation than natives, and they show a greater ability to shift their flowering time (Willis *et al.* 2010). Climate change may also disrupt native communities and ecosystems (Byers 2002) which will have an indirect effect on non-natives. Native species may be host species for invasive pests, and therefore a climate-driven change in their distribution, health and dynamics will affect the ability for a non-native to invade. Disturbances such as fire and flooding have also been recorded to have a positive impact on invasion success (Smith *et al.* 2004) by creating new pathways for invasive species to disperse without the competition from native species.

However, empirical and experimental data remains relatively scarce when observing the link between climate change and invasive species (Brook 2008), relying on theoretical observation and bioclimatic modelling. Moreover, the link between climate change, invasive species and biodiversity loss has been contested. It is suggested that invasive species will not necessarily cause the demise of native species or loss of biodiversity, with the exception of island and lake environments (Davis 2009, Davis *et al.* 2011Invasive species populations have also been seen to collapse with time (Cooling *et al.* 2012) and therefore colonisation may not be permanent. This has an obvious positive economic impact on management strategies.

It is clear that the synergies between invasive species and climate change are complex, with direct and indirect implications for native species and management strategies. The success of these strategies will also be affected by climate change and therefore must take its complexities into consideration (Pyke, *et al.* 2008).

8.4.1. Climate - birds

Avian introductions are significantly more likely to succeed when there is a close match between the latitude (Blackburn & Duncan 2001), climate (Duncan *et al.* 2001) or habitat (Kolar & Lodge 2001) of a species' origin and its introduction. Climatic match is one of the few consistent predictors of the establishment success of non-native species (Forsyth *et al.* 2004). However, some studies have found that contrary to commonly held beliefs, establishment in birds is not related to native range area, or to the diversity of climates inhabited in their native range (Kolar & Lodge 2001).

There are a number of exceptions to climate/environmental matching including cases where species have naturally colonised new climatic regions, and cases in which species have been deliberately introduced into new climates (Williamson 1996). Natural colonisation includes the spread of the serin *Serinus serinus* from Mediterranean areas to northern Europe, and the collared dove *Streptopelia decaocto* north-westwards from Turkey and the Balkans across north-western Europe. Deliberate introductions include the ring-necked parakeet *Psittacula krameri* from Africa and Asia into a number of European countries, including Britain; the Indian house crow *Corvus splendens* has been accidently introduced from the Indian sub-continent to North West Europe

Climate change is unlikely to affect the transport and introduction pathways of non-native avian species. Non-native species enter England as commodities in the pet trade, ornamental and zoological collections and the game bird industry; birds may also enter via accidental ship-assisted transfer. Climate change is unlikely to significantly alter the demand, either qualitatively or quantitatively, in these trade sectors (transport), or the propensity for captive birds to escape or be deliberately released into the wild (introduction). Under the present climatic conditions psittacines (includes parrots, macaws, conures, parakeets, cockatoos, cockatiels, lories, budgies, lovebirds), which are largely native to tropical and sub-tropical regions, are already the most commonly sold exotic birds in UK (Parrott *et al.* 2009).

Climate change, however, has the potential to affect the invasion success of released non-native species through the latter stages of the invasion process (establishment and spread), through: (i) improving the climatic match between England and its native range and (ii) through direct (e.g. thermal effects) and indirect changes (land management) to habitats and land use.

As global minimum temperatures have increased the ranges of many bird species have shifted towards higher latitudes and altitudes; allowing the species to survive in new locations (Root *et al.* 2003). The consequence of this shift on the overall global range varies between species. For some species shifts in climatic conditions are leading to an expansion of the overall range. For other species, however, for example those more limited by habitat availability, climatic shifts are leading to range contractions. Distributions are also being affected on a finer scale, for example in Sweden, the 1998-2009 national population trends of breeding birds were positively correlated with predicted changes in climatic suitability by 2050 (Jiguet *et al.* 2012).

In England, climate warming would be expected to enhance the thermal suitability for invasive avian species that are native to warmer regions, and whose introduced range is beyond the northern limits of their natural range (e.g. Indian house crow, sacred ibis, ring-necked parakeet). Climate warming will facilitate the spread of such species to more northerly regions of England and to higher altitudes – providing suitable resources are available. The action of climate warming on the life history traits of such species, however, is not necessarily straightforward. For example, sacred ibis have higher breeding performance in temperate Western France than those estimates published for its warmer native Africa (Clergeau & Yesou 2006).

For ring-necked parakeets, conversely, establishment success is lower in colder regions across Europe (Strubbe & Matthysen 2009). Although parakeets can succeed in establishing self-sustaining populations in relatively cold regions (e.g. in central Europe), breeding performance is not as good as in warmer regions. In colder regions, the spread of established parakeet populations could be impaired by low reproductive success. Global warming may reduce the climate mismatch and facilitate invasive spread of ring-necked parakeets. In England, the population growth of ring-necked parakeets over recent years is already amongst the highest observed in current British bird populations; despite lower reproductive success than in its native range. Climate warming, therefore, has the potential to improve breeding success and help promote even greater population growth and spread.

Climatically, Indian house crows are well-suited to a wide range of tropical and subtropical climate types, from high-rainfall coastal areas to the arid zone. They are abundant across these climate types overseas (Csurhes S. 2010). The species, however, has established a breeding population in The Netherlands (since the late-1990s), in temperate north-western Europe – an ecoclimatic zone similar to the UK. The Dutch birds have survived winter temperatures as low as -8°C (Ryall 2002). The availability of food scraps is probably a more important influence on abundance and distribution (Csurhes 2010). Global warming is likely to enhance conditions for the species in more northern latitudes, such as the UK.

For many avian species (non-native and native) a consequence of recent climate warming has been the advancement of birds' breeding season. Long-term studies on the date of laying (or hatching) conducted in 68 species has shown a significant advance of laying date in the majority of species (Moller *et al.* 2010). Birds are advancing their laying dates by a mean of 0.13 days per year and are laying 2.4 days earlier for every degree centigrade warmer. Based on global warming projections, Moller *et al.* (2010) estimate that birds might be expected to lay 15 days earlier by the end of the century. A benefit to earlier breeding is that there is more time for fledglings to develop and adults to replenish lost reserves prior to the onset of winter. This would facilitate improved survival over winter and subsequent population growth and spread. This process is in agreement with Arim *et al.*'s (2006) analysis of the spread dynamics of invasive species, where spread was considered to be regulated by a non-reproductive leading front that delays spread until local population growth produces new propagules to sustain the wave of advancement.

Two principal land use changes that might affect avian species are associated with freshwaters and wetlands and agricultural land. Wetland connectivity is a dynamic process with cyclical periods of high connectivity following periods of high precipitation and more fragmented following periods of drought. In Australia, the drying and flooding of wetlands strongly influences waterbird dispersal (Kingsford & Norman 2002; Chambers & Loyn 2005 both cited in Morris 2012). In inland south-eastern Australia, during summer, many waterbird species become concentrated in large swamps as smaller ones dry. As densities increase a proportion of the population disperses (most following rivers); returning inland during the winter. The geographical arrangement of wetlands in the landscape also influences the distribution of waterbirds. In Iowa, USA, clustered wetlands contained more species than isolated ones. In the Great Plains of North America, the movements of pectoral sandpipers *Calidris melanotos* were associated with the extent of wetland connectivity (Farmer & Parent 1999). Birds captured at wetland feeding sites were radio-tagged and released. Over 40% of birds made no inter-wetland movements and over 90% moved less than 10km. As the distance between wetlands decreased, and the proportion of the landscape composed of wetlands increased, individual birds moved between wetlands more frequently and moved longer distances. Movement patterns were consistent with a more connected landscape allowing shorebirds to exploit more feeding sites with reduced search costs. As the distance between individual wetlands decreases there is a point at which birds begin to perceive the complex as though it were one large wetland. In England, reductions in the spatial and temporal availability of wetlands caused through a warmer, drier climate and associated altered land management practices (abstraction, irrigation) may similarly limit the spatial and temporal availability of this habitat to wetland species; altering their distribution, residency and movement patterns. The scale of spatial connectivity between neighbouring wetlands at which such effects may occur in England are not known.

In agriculture, predicted changes in crop type and regional patterns of crop planting and harvesting will alter the landscape for birds in terms of resource availability. For example, there may be an increase in the growing of grapes, other soft fruits and produce (e.g. sunflowers) currently concentrated in warmer, drier south-eastern regions. Increase in the coverage of such crops in the south and their introduction further north will provide enhanced foraging opportunities for birds, including invasive species such as the ring-necked parakeet which already forage on these crops in their present range. Parakeets forage up to 20km from their roost sites and an increase in preferred foraging patches will facilitate their on-going population expansion.

For those species present in the wild, establishment and spread may be facilitated for those currently experiencing reproductive productivity lower than that in their native range. Alterations to the landscape in respect to spatial and temporal changes in the distribution and availability of preferred habitat are unlikely to represent significant barriers to spread, due to the species' inherent mobility. Inherent movement and dispersal traits will enable birds to overcome changes in the availability of preferred habitat as a consequence of climate warming. In addition, all of the species possess behavioural flexibility that will facilitate the utilisation of altered habitat (e.g. changes in nesting or feeding behaviour) and/or allow dispersal to more suitable habitat.

Table A1 shows the potential influence of climate change on invasive avian species.

Table A1. Potential influence of climate change on invasion pathway for selected invasive avian non-native species.

Invasion Phase	Invasion Pathway	
	Present	**Effect of Climate change**
Transport	Trade: – Pet trade – Ornamental/zoological collections – Gamebird industry Other: – Accidental ship transfer	 – No obvious effect on trade and hence transport – No obvious effect on trade and hence transport – No obvious effect on trade and hence transport – No obvious effect
Introduction	– Escapes – Deliberate release	– No obvious effect – No obvious effect
Establishment	– Survival – Reproduction – Further escapes/releases	– Climate warming may increase the survival of species at/outside the northern end of their native range; especially over-winter survival. – Climate warming may increase the reproductive success of species at/outside the northern end of their native range. – No obvious effect.
Spread	– Natural movements – Escapes/releases in new areas	– Enhanced reproductive success and associated increased population growth would lead to increased rate of spread. – Climate warming is unlikely to have a significant effect on invasive species movements through the landscape. – Natural movements and behavioural flexibility will allow the selected species to utilise altered habitats and/or disperse to preferred habitat. – No obvious effect

8.4.2. Climate - plants

Anecdotal evidence suggests that climate change is not likely to substantially decrease the impact of current invasive species because many of them already span a range of environmental conditions (Qian & Ricklefs, 2006). Mutiple sources in the invasion of Common Ragweed, *Ambrosia artemisiifolia* into France have resulted in an admixture meaning that levels of allelic diversity in the native and invaded range are similar and therefore have lost none of the traits that allow it to be such a successful invader (Genton *et al*., 2005). However the invading species will only succeed under abiotic conditions that provide a suitable growing environment. *Prunus serotina,* a gap-phase tree species (Curtis, 1959), is unable to invade waterlogged or calcareous soils whereas it successfully colonises well-drained, nutrient-poor soils in northern France (Closset-Kopp *et al.,* 2011).

Climate change is a subtle and long term process and it will require more research to understand how specific invasive species may behave in the changing conditions and which new species will emerge as the new threat (Hellmann *et al.*, 2008). The current levels of plant invasions in Europe have been mapped (Chytry *et al.,* 2009) and fututure species invasions will be driven by climate (Walther *et al.,* 2009) and land use but the effect of climate on the level of invasion of an area is much weaker than the effect of land use (Chytry *et al*., 2008a). Chytry and co-workers (2012) projected trends in furure plant invasions under changing land-use and climate scenarios, and found that while plant invasions depend on an interaction between land-use change, socio-economic factors and climate, they are most closely linked to land-use change.

Hulme (2012) states that climate change is virtually certain to contribute to increased probability of non-native establishment. However the magnitude of any direct climate effect on the number of new species will be small relative to the role of accidental and deliberate introductions. Therefore it is still important to develop invasive plant management and ecological restoration strategies in the context of changing environments (Drenovsky *et al.*, 2012). Extreme climatic events such as unusual heatwaves, freak storms, floods and droughts that are predicted under climate change may further promote the transport and introduction of non-native species (Diez *et al*., 2012). Non-native vines benefitted from tree canopy loss after Hurricaine Andrew in Florida in 1992, and spread widely across that state despite the presence of native vines (Horvitz *et al*., 1998).

With this in mind it will also be essential to develop means to identify potentially harmful invasive plants (Drenovsky *et al.*, 2012). The decreasing frequency of lethal cold temperatures will allow pole-ward spread of plant species (Simberloff, 2000). Increased atmospheric carbon dioxide may expand warmer range boundaries of some species. Changes in rainfall patterns may also increase opportunities. Plant species that cannot shift ranges quickly enough to maintain populations may decline (Simberloff, 2000), thus traits such as rapid dispersal, high fecundity and small seed mass may have disproportionately high benefits for non-natives (Dukes & Mooney 1999). There may be both expansion and contraction of the range of invasive plants due to climate change leading to increased risk or opportunities for restoration (Bradley *et al.*, 2010).

Climate change may affect patterns of trade and introduction and success of a different range of non-natives. This may be particularly relevant for potential novel agricultural weeds (Theoharides and Dukes, 2007). Above all the horticultural trade will be prepared to make available new plants from previously untapped parts of the world as supply and demand creates considerable motivation for novel species and poses the greatest risk of a new wave of plant invasions (Bradley *et al.*, 2012). It will also increase the intentional introduction of non-native plants for biofuel production, and therefore possible invasive species as many of the traits are comparable for both. Continued changes in land use or land cover are likely to affect ecosystems as profoundly as climate (Richardson, 2006; Hulme, 2009).

Table A2 shows the potential influence of climate change on invasive plant species.

Table A2. Potential influence of climate change on invasion pathway for selected invasive plant non-native species.

Invasion Phase	Invasion Pathway	
	Present	Effect of Climate change
Transport	Trade:	
	– Horticulture	– Allow trade with regions that were not previously used
	– Timber	– Allow trade with regions that were not previously used
	– Biofuel	– Allow trade with regions that were not previously used
	– Amenity	Allow trade with regions that were not previously used
	– Other: Accidental	– May increase as general trade with all regions increasing. Most likely from China and Asia
Introduction	– Deliberate planting	– Increased chance of survival and vegetative spread
	– Throw outs	– Increased chance of survival
Establishment	– Survival	– Climate warming may increase the survival of species at/outside the northern end of their native range; especially over-winter survival.
	– Reproduction	– Climate warming may increase the reproductive success of species at/outside the northern end of their native range.
	– Further Throw outs	– Increased chance of flowering and setting seed
Spread	– Natural movements	– Enhanced seed production and associated increased population growth would lead to increased rate of spread. – Climate warming is unlikely to have a significant effect on invasive species movements through the landscape. – Seed set/Increased seed production will allow the selected species to utilise altered habitats and/or disperse to preferred habitat.
	– Further Throw outs in new areas	– Increased range and chance of establishment

8.4.3. Climate - invertebrates

Temperature is an important driver of invasion in insects (Dukes and Mooney, 1999). For some species, temperatures at different times of year are important. For example *Bemisia tabaci*, the tobacco whitefly is frost sensitive (Bosco & Caciagli, 1998) tending to be found in areas with few frosts per year, whereas summer temperature allowing increased population growth is critical for determining development rate for others e.g. *Anoplophora glabripennis. A. glabripennis* can complete its development in a year in the southern part of its native range in China, but its lifecycle might be two or three years in the north of its range (Hu *et al.*, 2009).

Precipitation is critical for some insects species but not for others. For example insects which form colonies and nests (e.g. oak processionary moth) in trees are susceptible to heavy rainfall events but the larval and pupal stages of wood boring species are protected from these. . Heavy rain can kill or remove insects from crops, for example onion thirps (Reiners and Petxoldt, 2005). It can also increase the mortality of insects which have larval stages in the soil (Vincent et al 2003). In addition, precipitation can raise humidity levels increasing mortality by entomopathogenic fungi (Petzold et al, , such as *Beauvaria bassiana.*Precipitation can also influence the tendency of invertebrates to disperse, for example, it can limit the dispersal of spruce budworm, *Choristoneura fumiferana* (Sturtevant *et al.,* 2013) and stimulate the flight of European corn borers, *Ostrinia nubialis* (Keszthelyi, 2007).

Increased CO_2 levels can also affect the suitability of habitat for invasive insects. Increased CO_2 decreases the nutritional quality of host plants so that herbivorous insects must compensate by increasing consumption (Dermody *et al.* 2008). This may affect predator/prey population dynamics and decouple tritrophic interactions. In addition, fertilizer use can affect tritrophic interactions. For example, nitrogen availability alters the types and quantities of plant defensive compounds (Chen *et al.* 2010).

As average temperatures increase, many insect species are extending their ranges northwards or increasing their range in altitude. There are many examples of this in the literature, for example the pine processionary moth, *Thaumetopoea pityocampa* (Stastny *et al.,* 2006). Forest pests, particularly defoliating species such as gypsy moth and wood borers such as the *Anoplophora* sp., could reach outbreak levels in new areas (Vanhanen *et al.,* 2007). This would be brought about by a combination of more favourable conditions such as the absence of extremely cold winter temperatures facilitating increases overwintering survival rates or warmer summer temperatures increasing development rates. There is also the possibility of a shift to new host plants. This is more likely under outbreak conditions, for example the oak processionary moth will readily move onto other tree species if there is a dense population (Evans, 2007). This phenomenon has also been seen for diamond back moth, *Plutella xylostella* which can become more polyphagous as population density rises. This makes ecological sense because the capacity to use a broader range of food plants may lead to higher local carrying capacity (Bigger & Fox, 1997). It has become apparent that even the specialists, such as the juniper shieldbug, can move to novel host plants. For this reason, it is not possible to state with certainty what the range of a non-native species may be if the climate constraints on its spread are lessened. It is possible that some species will then be limited by habitat, assuming this has not also increased in range, but it is also possible, particularly if large populations are present, that the insects will move to new host plants.

Table A3 shows the potential influence of climate change on invasive insect species.

Table A3. Potential influence of climate change on invasion pathway for selected invasive invertebrate non-native species.

Invasion Phase	Invasion Pathway	
	Present	**Climate change**
Transport	Trade: – Timber and wood products	– New trade patterns may result from shifts in the areas which are optimal for timber production of different species
	– Garden plants	– Changing environmental conditions may affect the species of plant imported for gardens and horticulture. This may lead to an increase in the number of incidences of non-native species being imported without interception and, therefore, a greater propagule pressure for these species.
Introduction	– Escapes – Deliberate release	– Not relevant for most invasive invertebrates
Establishment	– Survival	– Low winter temperatures can be fatal for some insects such as *Bemisia tabaci.* Damp conditions can increase the mortality of juvenile stages of some pests and also decrease activity and hence chances of survival for others.
	– Development rate	– The development rate of some invertebrates such as *Anoplophora* is wholly dependent on summer warmth. This can mean that one generation can take one or more years longer in the UK than in a warmer climate.
Spread	– Natural movements	– Enhanced reproductive success and associated increased population growth would lead to increased rate of spread. – Climate change may affect the number of days that are suitable for migration, for example some species tend to fly in warm and dry conditions.

8.4.4. Climate – mammals

In comparison to other taxonomic groups, mammals are less influenced by abiotic factors such as precipitation or temperature. The only exceptions are the habitat specialists such as Chinese water Deer, muskrats, capybara and coypu which are restricted to aquatic, riparian habitats (Nowak 1999). In addition the literature also reveals that some species do not favour deep snow and therefore high altitude, such as raccoon dogs in Finland (Melis *et al.* 2010). Overall, generic studies at large landscape scales have shown that North West Europe will get warmer and wetter and the European mammal fauna will extend their ranges northwards (Flojgaard *et al.* 2009)

Most European climate change models suggest that rainfall will increase, flooding will increase and temperature will increase by 3 to 4°C (Flojgaard *et al.* 2009). A study of Danish fauna for example is found that the number of species such as coypu and muskrats, raccoon dogs and Chinese water Deer will benefit from climate change.

The effects of climate change and the impacts of invasive species have a social dimension that is hard to quantify and assess. This is a two way process as humans are constantly influenced by both and actively seek to manage both. In addition, the effects of climate change and the impacts of invasive species also have an interaction with each other. A good example of this is the research carried out by Pyke *et al.* (2008) which shows how, as a result of climate change in America, an increasing number of man-made coastal defences are being built. These in turn are exploited by invasive coypus, which are using them to spread through the landscape. The same is true of a number of aquatic invasive species, which across a number of taxonomic groups have been shown to benefit from reduced water quality that may result from climate change (Rahel & Olden 2008).

Table A4 shows the potential influence of climate change on invasive mammal species.

Table A4. Potential influence of climate change on invasion pathway for selected invasive mammal non-native species.

Invasion Phase	**Invasion Pathway**	
	Present	**Climate change**
Transport	Trade: – Accidental transport of small species – Deliberate movement of species around the country	– As trade patterns shift globally, novel species may come into UK ports – New species may become available through pet trade. – Changing environmental conditions may affect longevity and pre-establishment over-winter survival
Introduction	– Escapes – Deliberate release	– Increase in exotic pets and collections will lead to more escapes and releases
Establishment	– Survival	– As thermo-regulators- not as affected by climate change. Some enhancement of over-winter survival – Wetter weather could mean greater food availability. Aquatic species will benefit directly
	– Reproduction	– Climate warming may increase the reproductive success of species at/outside the northern end of their native range, and enhance juvenile survival of species with large litters.
Spread	– Natural movements	– Aquatic species may benefit directly in a wetter regime – Food availability aseasonally/ of novel species may enhance foraging movements and survival during dispersal.

8.4.5. Climate - herptiles

Herptiles are strongly affected by their immediate environment and are thus sensitive to climate as it influences reproductive timing which in turn influences the distributional limits of any species. This is particularly true of small ectotherms (McKay & Phillips 2012). In addition, amphibians are usually constrained by the availability of water for breeding and foraging, and this additionally limits their distribution (Morris 2012). Although behavioural unpredictability is not as strong as in the higher vertebrates such as mammals and birds, even reptiles and amphibians have manifested behavioural adaptation to changes in climate through making use of unusual sources of water in the case of cane toads or adapting burrowing habits appropriate to local climatic conditions in the case of snakes (Holway & Suarez 1999; Tingley & Shine 2011; Tingley *et al.* 2012), so once again these unpredictable behaviours are hard to assess.

The climatic factors that influence herptiles are as follows. These climatic traits were used to assess the species selected in Section 3:

- *Temperature*: With the exception of Alpine newts, all species benefit from increased temperature. Alpine Newt have been shown to restrict themselves to cooler Upland regions of continental Europe due to temperature (frog life pers. comm.).

- *Precipitation*: all turtle and amphibian species benefit from increased precipitation, and disadvantaged by reduced precipitation. Reduced precipitation in the breeding season is likely to have particularly damaging impacts. None of the remaining species are assessed to be disadvantaged by increased or decreased precipitation.

A number of native amphibians and reptiles may benefit by allowing earlier spawning and breeding(Araujo *et al.* 2006). However, any competitive advantages this confers may be negated by the fact that a number of invasive species will also benefit, as is the case with cane toads (Tingley & Shine 2011; Tingley *et al.* 2012)., epecially if the climate is wetter rather than drier (Brown *et al*. 2011)

> Prolonged periods of warmth could benefit a number of reptile species. *T. scripta* for example could benefit from a longer, warmer dry period in summer as this is one of the few factors that have prevented this already widespread species from breeding (Kikillus *et al.* 2010), and certainly restricts the distribution of Pythons in America (Dorca *et al.* 2011).

Generally, it is believed that through climate change, much of North West Europe will become warmer and wetter with greater and more frequent extremes (Flojgaard *et al.* 2009), although at smaller spatial scales, changes in rainfall in particular, may vary widely, as it is influenced by local geographical and topological features (Jenkins *et al.* 2009). One aspect of climate change is localized flooding. This is particularly true for Western parts of the country and especially true in winter. Flood events may speed up the spread of invasive species that are tied to aquatic environments, such as amphibians (Rahel & Olden 2008). Certainly a lack of connectivity otherwise has a mixed influence over the distribution of the species selected. Some species such as wall lizards are habitat specialists and limited by a lack of connected habitats whilst others are competitively favoured by habitat fragmentation and are able to spread over large areas of the seemingly unsuitable habitat – such as cane toads and African clawed toads (Faraone *et al.* 2008; Florance *et al.* 2011).

Table A5 shows the potential influence of climate change on invasive amphibian and reptile species.

Table A5. Potential influence of climate change on invasion pathway for selected invasive amphibian and reptile non-native species.

Invasion Phase	Invasion Pathway	
	Present	Climate change
Transport	Trade:	
	– Accidental transport of small species	– As trade patterns shift globally, novel species may come into UK ports
	– Deliberate trade as pets	– New species may become available through pet trade.
	– Easier to keep captive populations	– Changing environmental conditions may affect longevity and pre-establishment over-winter survival
		– Easier to keep and transport
	– More availability of exotics	– More likely to be transported with novel plant species
Introduction	– Escapes	– Increase in exotic pets and collections will lead to more escapes and releases
	– Deliberate release	– Some species easier to keep outside in gardens
Establishment	– Survival	– Greater survival and rates of growth to reach sexual maturity earlier
		– Extreme weather may kill species (such as prolonged snow)
		– Direct benefit to aquatic species
	– Reproduction	– Successful breeding and viable eggs for many species such as terrapins
Spread	– Natural movements	– Aquatic species may benefit directly in a wetter regime
		– Food availability aseasonally/ of novel species may enhance foraging movements and survival during dispersal.

8.5. Other factors influencing of invasion success

Biotic environment

Communities differ in their susceptibility and response to invasion. Community invasibility has been considered to be dependent on the number of species present, with species rich sites being more resistant to invasion (Elton 1958). Conversely, it has been argued that sites with high species diversity are more susceptible to invasion (Stohlgren *et al.* 2006).

Simulation of species invasions within complex ecological networks revealed that, in general, an invader has a high probability of success if it is a generalist, an herbivore or an omnivore, and is relatively invulnerable to predation (Romanuk *et al.* 2009). In regard to the latter factor, the 'predator release' hypothesis predicts that, when introduced to a new habitat that is lacking in its natural enemies, a species is freed from predation and can then increase its population growth and spread (Dayan & Simberloff 1994).

Environments that have been degraded, usually by anthropogenic factors, are considered more susceptible to invasion. This is due to the disruption of the ecological networks, with the removal or weakening of native species and/or interactions, providing opportunities for exploitation by an invader (Elton 1958).

Several plant species such as *Avena barbata, Brassica nigra, Lolium perenne* and *Trifolium pratense* are successful alien colonists of these highly transformed human landscapes as shown by specific quantitative impact studies (Hulme *et al*., 2012). Recent studies of plant invasion in Europe at a regional scale revealed that the proportion of alien plants is mainly determined by land use or habitat type (Chytry et *al*., 2008a). Land-use scenarios used by Chytry and co-workers (2012) together with projections of future climate change have shown that the levels of invasion in north-western and northern Europe may increase under regimes where biofuel crop plantations replace grasslands. Eastern and some southern areas may decrease with the abandonment of agricultural land that are particularly suitable for the spread of alien plant species (Pysek *et al*., 2005; Chytry *et al*., 2008b).

Terrestrial animals that are able to live in close association with humans tend to become invasive more often than species less tolerant of people (Jeschke & Strayer 2006). Examples of species that can reach high population densities in urban and peri-urban areas are the Norway rat *Rattus norvegicus*, rose-ringed parakeet *Psittacula krameri* and the harlequin ladybird *Harmonia axyridis* (Daisie 2009). Others include the plants *Buddleia davidii* and *Impatiens glandulifera*.

Behavioural flexibility

Behavioural flexibility is a trait common to a number of successful invasive vertebrate species, and can be expressed in a number of ways, including habitat tolerance, diet plasticity or innovation frequency (Clergeau & Yesou 2006). Behavioural flexibility confers a rapid response to new or changing environmental conditions and facilitates the exploitation of a wide range of ecological resources (Wyles *et al.* 1993). For example, a species that readily investigates novel food resources is more likely to adapt to a new environment compared to a species that is inflexible in its foraging behaviour and diet (Sol & Lefabvre 2000).

Propagule pressure

The term propagule refers to an individual or group of individuals arriving in a place (Simberloff 2009). Propagule pressure has two key components – propagule size (the number of individuals) and propagule number (the rate at which propagules arrive per unit time). Propagule pressure, therefore, encompasses the size and number of propagules and the spatial and temporal pattern of their introduction. Studies have indicated the positive effect of propagule size and/or propagule number on the probability of invasion success for both mammal and bird species (Kolar & Lodge 2001; Simberloff 2009).

Introduced populations are usually small and so are at greater risk from extinction due to stochastic events (i.e. random population and environmental effects - see below) and Allee effects (e.g. inability to find mates at low population densities). In addition, in small founder populations there is the increased likelihood of inbreeding between closely related individuals and resulting negative impact on population fitness (Roman & Darling 2007). Invasions that involve a greater number of individuals will ease the risks associated with small founder populations. In addition, multiple introduction events at a number of different sites will increase the probability that at least one propagule will encounter favourable conditions (e.g. habitat, seasonal climate).

Several studies have suggested that propagule pressure is *the* most important factor in determining establishment success, over and above species traits and ecosystem characteristics (Simberloff 2009). It has been argued that studies implicating the importance of species and ecosystem traits failed to account for propagule pressure. Cassey *et al*. 's (2004a) analysis of the geographic and taxonomic distribution of introduction effort found that introduction effort is both the strongest correlate of introduction success and correlated with a large number of variables previously cited as influencing success. The principal finding was that, except for habitat specialisation, all factors were significantly correlated with propagule size.

Although propagule pressure is recognised as having a key role in the likelihood of establishment of introduced species, limitations to its effect are also recognised. It cannot be concluded that a propagule sufficiently large in size and/or number will ensure the establishment of any introduced species (Simberloff 2009). For any species there are limits to the abiotic and biotic conditions that it can tolerate. Another factor that will influence the effect of propagule pressure is the condition of the introduced individuals. Poor condition or ill health will exacerbate the effect of demographic stochasticty and decrease the likelihood of establishment.

Demographic and environmental stochasticity

Factors that interact with propagule pressure are demographic stochasticity and environmental stochasticity (Simberloff 2009). Demographic stochasticity is the random fluctuation of population size associated with a finite number of individuals present in the population and with random fluctuations in the demographic traits such as birth rate, death rate and sex ratio. In a founder population of a very few individuals, there is a greater probability that all will die by the next time period than in a population with many more individuals. The factor has very important implications in relation to the minimum viable population size and to the founding of new populations following an introduction. Environmental stochasticity involve events such as unusual weather, floods and fires. The occurrence of such extreme environmental incidents can completely eliminate even a very large propagule, if it coincides with the introduction event.

The interaction between propagule pressure and demographic and environmental stochasticity differs for the two key components. Environmental stochasticity has an inherent temporal component - adverse events, such as sub-zero temperatures or floods occur at specific points in time and can eliminate entire propagules, even large ones. However, as the adverse effect passes, subsequent propagules will not be affected. Consequently, the effect of environmental stochasticity will be ameliorated by propagule number. Conversely, demographic stochasticity exerts a constant interaction with the numbers of individuals present and random fluctuations in demographic traits. The influence of environmental stochasticity, therefore, is reduced, by increased propagule size.

Failed invasions

When reviewing the traits of successful invasive species, note should be taken of a very recent addition to the literature on biological invasions. Zenni & Nunez (2013) have highlighted that the field of invasion biology has tended to focus on the interspecific comparison of traits between successful and unsuccessful invading species. Whereas, failed invasions are a common outcome of species introductions and species exhibit intraspecific variation in invasive success. Five mechanisms were associated with species' failure to invade: low propagule pressure, abiotic resistance, biotic resistance, limited or inappropriate gene pool and lack of mutualists. Zenni &

Nunez (2013) argued that the estimates of the intrinsic invasiveness of species may be biased if there is failure to take into account the number of introduction attempts and intraspecific differences between invasive and non-invasive populations.

8.5.1. Other influencers of invasion success - birds

Life history and ecological traits affecting invasiveness

A number of biological and ecological traits have been cited as factors in determining the propensity for mammal and bird species to become invasive. Early studies indicated that successful vertebrate invaders are characterised by: *high abundance* in a *large native range*, *large size*, *vagility*, *wide dietary range*, *short generation times*, *much genetic variability*, *gregariousness*, *ability of females to colonise alone*, *able to tolerate a wide range of physical conditions* and have a *close association with humans* (Ehrlich 1989).

Studies on avian introductions have suggested important variables influencing invasion success include migration, body mass, clutch size, sexually selected traits, latitude of origin, larger geographic range, climate/environmental matching and taxonomy. For example, the probability of a successful establishment was found to be higher in species that were non-migratory (Veltman et al. 1996), had larger body mass, laid smaller clutches (Green 1997), had more broods per season (Kolar & Lodge 2001) and were sexually monochromatic (i.e. sexes have similar plumage) as opposed to dichromatic (Sorci et al. 1998). Other important attributes cited are behavioural flexibility (Sol & Lefabvre 2000) and mode of juvenile development (Sol & Lefabvre 2000). Species possessing behavioural flexibility have been associated with relatively large brains and a high frequency of foraging innovations; whilst nidifugous (precocial) rather than nidicolous (altricial) species tend to be better invaders. Finally, studies have indicated a key role of introduction effort or propagule pressure (i.e. the number of individuals introduced and/or the number of introduction events) on increasing the probability of invasion success (Duncan 1997; Green 1997; Kolar & Lodge 2001; Simberloff 2009).

For many of these traits, however, there is contradictory evidence from different studies (Duncan *et al.* 2003; Hayes & Berry 2008). In a review of bird introductions, Duncan *et al.* (2003) highlighted, for example, equivocal evidence for a positive effect of clutch size, body mass and generation times. In an analysis of 49 studies across seven biological groups, Hayes & Barry (2008) found that the importance of individual factors can be contradictory both within and between different studies, dependent on taxonomic and statistical treatments; specific variables can be confounded by others (Hayes & Barry 2008). For birds, some traits from the same data sets were inconsistently reported as important for establishment success/failure using different analytical methods (diet breadth/type, plumage dichromatism) or analysed using the same method (geographic range size, no. release/arrival attempts, migratory tendency).

Biotic environment

Examination of broad-scale patterns of species richness and species density for native and non-native birds in North America, concluded that native species richness was positively and primarily associated with optimal environmental conditions, while non-native species were positively associated with native species richness, optimal environmental conditions, and secondarily with human population and factors related to land use change (Stohlgren *et al.* 2006). In their review, Duncan *et al.* (2003) concluded that there is little evidence from avian introductions that species-poor locations are easier to invade than species-rich locations.

Terrestrial animals that are able to live in close association with humans tend to become invasive more often than species less tolerant of people (Jeschke & Strayer 2006). Examples of avian species that can reach high population densities in urban and peri-urban areas are the European starling *Sturnus vulgaris*, ring-necked parakeet *Psittacula krameri* and the Indian house crow *Corvus splendens*.

Behavioural flexibility

A wide diet spectrum and an ability to use various habitats are considered to have enabled the sacred ibis *Threskiornis aethiopicus* to establish feral colonies in France and Italy (Clergeau & Yesou 2006). Conversely, however, some studies have shown that diet breadth in birds was not related to establishment (Kolar & Lodge 2001). Different innate responses in the closely related common starling and Asian crested mynah *Acridotheres crsitatellus* appear to explain their contrasting invasion success in North America – following establishment starlings spread continent-wide whilst mynas established but failed to spread significantly (Ehrlich 1989; Sol & Lefabvre 2000). Studies of sympatric populations showed that mynas retained breeding habits appropriate to their native range that were less appropriate for the North America climate – including a lack of attentiveness to incubating eggs.

Propagule pressure

The establishment success of non-native birds has been shown to increase with the number of individuals released (Kolar & Lodge 2001; Duncan *et al.* 2003; Hayes & Barry 2008) and the number of release events (Kolar & Lodge 2001). For birds introduced to Australia, on average the larger the propagule the greater the probability of establishment (Newsome & Noble 1986). In New Zealand, most of the established introduced bird species had been introduced ten or more times, whilst all but three of the unsuccessful introductions had been introduced fewer than ten times

(Veltman *et al.* 1996). However, there are documented cases of invasions originating from very small propagules (Simberloff 2009). For example the establishment of the Eurasian house sparrow *Passer domesticus* in North America, followed numerous previous unsuccessful introductions.

Key traits

The literature has cited a number of variables that may play important roles in influencing the invasion success of avian species. Most studies have focussed on species' life history and ecological traits; with little information on species' landscape use and spatial dynamics. Factors considered to be important in promoting invasive success of non-native avian species, are shown in Table A6 along with the likely impact climate change will have on these.

Table A6. Key traits for non-native avian species and their possible reactions to climate change.

Key Trait	Reaction to climate change	Notes
Ecoclimatic matching; A good climate match between the native range and introduced area.	Increase	Assuming invasive species coming from areas of the world that experience warmer weather with drier summers and wetter winters.
Large geographic native range; Species exposed to a broad range of conditions and/or variable environment are more likely to establish.	Variable	Native range is likely to increase at northern edge (and higher altitudes) but may decrease at southern edge (and lower altitudes).
Migratory behaviour; Non-migratory species.	Variable	If the UK climate becomes unsuitable for certain non-migratory non-native species they may adopt some level of migratory behaviour. Conversely, the climate may become more suitable and so currently migrating species released into the wild could become non-migratory as the drivers for dispersal are reduced.
Sexually selected traits/ plumage type; Sexually monochromatic species.	No change	Climate change is unlikely to affect the relative costs and benefits of the presence or absence of sexually selected plumage traits.
Broods per season; Multi-brooded.	Increase	Increased temperatures could result in the advancement and extension of the breeding season allowing more time for multiple broods per year.
Body mass; Larger body mass.	No change	Climate change is unlikely to enhance the benefit of larger body mass on invasiveness.
Behavioural flexibility; Relatively large brain size, wide diet spectrum, high frequency of foraging innovations	No change	Climate change would not affect a species' inherent ability to utilise a wide range of habitats and/or resources.
Juvenile development; Nidifugous (precocial) rather than nidicolous (altricial).	No change	Climate warming is likely to enhance the breeding success and survival of both precocial and altricial young; therefore there would be no significant change in the relative benefits of the different juvenile development types.
Human commensalism; Species that live in close association with humans in anthropogenically-modified habitats.	No change	Climate change is unlikely to lead to a significant increase in the numbers of exotic birds kept as pets as under the present climatic conditions the largely tropical and sub-tropical psittacines are already the most commonly owned pet birds.
Propagule pressure; Greater number of individuals and/or release events.	No change	Climate change is unlikely to lead to significant increases in releases/escapes of non-native birds as climate change is unlikely to increase the quantity of non-native birds introduced for pets, collections or quarry.

8.5.2. Other influencers of invasion success - plants

Life history and ecological traits affecting invasiveness

Biological invasions are the consequences of the ever-increasing movements of plant species around the world as a result of the growing volumes of international trade, environmental change and assisted migration (Blackburn *et al.*, 2011). Many species have been moved beyond the limits of their geographic range by human activity throughout history and this trend continues. The European Plant Protection Agency (EPPO, 2013) has estimated that 80% of the invasive alien plants found in Europe have been voluntarily introduced for ornamental purposes, and the international trade in plants is increasing yearly. Over the period 1990-2007 records of non-native species in samples of birds mammals, plants and marine life rose by 23%, but except for mammals, the absolute proportion was still only about 1% of the total (Hill *et al.*, 2009).

Characteristics that allow plants to achieve such an effect outside their native range include broad climatic tolerances and large geographic ranges (Rejmanek, 1995; Goodwin *et al.*, 1999; Qian & Ricklefs, 2006). They often have characteristics that facilitate rapid range shifts, such as low seed mass and short time to maturity (Rejmanek & Richardson, 1996). Several frameworks have been developed to try and understand the stages, means and barriers to plant invasion (Elton, 1958; Vermeij, 1996; Williamson, 1996; Richardson *et al.* (2000b; 2011); Theoharides & Dukes, 2007; Crowl *et al.*, 2008; Wilson *et al.*, 2008) and these have been unified within a general framework for all human-mediated invasions by Blackburn *et al.*, (2011).

Drenovsky *et al.* (2012) suggested that plant functional traits are important. ‘Hard’ plant traits directly influence plant function but are difficult or time consuming to measure such as dispersal distance, relative growth rate (RGR), and competitive effect and growth. ‘Soft’ Plant Traits on the other hand are more readily quantifiable, for instance Leaf mass/unit area and specific leaf area, and the latter correlates well with RGR (Lambers and Poorter, 1992), photosynthetic rate (Wright *et al.*, (2004), leaf N content (Reich *et al.*, 1997) and leaf lifespan (Westoby *et al.*, 2002). However expecting one trait to be universal of invasive potential is unrealistic (Grotkopp *et al.*, 2010). The traits that are advantageous in resource-rich environments may be disadvantageous in resource-poor environments where slow growth and tissue retention are important (Drenovsky *et al.*, 2012).

Flexible traits are also important and the plasticity of key functional traits may be particular beneficial during the invasion process and to all plants facing a changing environment and climate (Richards *et al.*, 2006; Nicotra *et al.*, 2010). Phenotypic plasticity is the ability of a particular genotype to express range of phenotypes across different environments (Richards *et al.*, 2006). Phenotypic plasticity may lead to rapid evolution of novel traits facilitating invasiveness in changing environments (Drenovsky *et al.*, 2012). There is evidence that that invasives are more plastic in morphological and physiological effects than related native non-invasives (Davidson *et al* 2011). However a review of 80 introduced plant, animal and fungal species, the average loss of molecular-level diversity from the native range to the non-native was found to be substantial (Dlugosch and Parker 2008).

Biotic traits

Blackburn *et al.*, (2011) recognises that the invasion process can be divided into a series of stages. The success of a non-native plant species is controlled by a series of barriers that need to be overcome to progress to next stage in the invasion sequence. The first stage in the process is the transport to the non-native country and therefore seed longevity is an important attribute to overcome the barrier of geography. Historically these have been accidental transports via the wool trade (*Abutilon theophrasti, Senecio inaequidens, Stipa neesiana*), as a grain contaminant *(Argemone mexicana, Iva xanthiifolia*), bird seed (*Amaranthus spp., Ambrosia artemisiifolia, Solanum rostratum*) contaminants or vegetable food imports such as soyabean waste (*Cenchrus echinatus, Sida spinosa, Solanum carolinense*). This process is happening faster and the storage conditions on route are better than before, increasing the likelihood of seed survival. This is taking place from more distant regions, primarily due to the ever growing global commerce and travel (Mack *et al*, 2000; LeMaitre *et al.*, 2004).

Introduced species tended to be ruderal, capable of fast growth and high resource uptake (Mack & Lonsdale, 2001). Many seeds would not survive such transport (Perrings *et al.*, 2005) and once in the non-native area, climate sets the broad limit to plant distribution and productivity and may cause immediate failure (Sakai *et al.*, 2001). However higher numbers of propagules, seeds, arriving would increase the likelihood of survival (Lockwood *et al.*, 2005). This is further enhanced if these come from many places within the native range of the species thereby increasing the genetic variation and opportunities for new genotypes through recombination, as in *Phalaris arundinacea* introduced to North America from many areas within Europe (Lavergne & Molofsky, 2007). If the introductions take place across a wide range of receiving country then it is more likely that the ideal match will be achieved (Lockwood *et al.*, 2005). There are other relationships with native origin as abundance were similar for species in their native and introduced sites, therefore home abundance is a useful predictor of the potential invasiveness of a species (Firn *et al.*, 2011). This may occur as non-native communities are regulated by processes similar to those in the native area, particularly because of the strong interaction between invasion success and human disturbance (La Sorte *et al.*, 2007; HilleRisLambers *et al.*, 2010).

Other means of introduction are those that are deliberate where plants are introduced for a purpose such as for food, fuel, forage, lumber, and medicine. By far the most common route is for the horticultural trade. Here there is a deliberate selection of plants from parts of its native range most similar to the new country, or even from an area where cold hardiness is assured (Theoharides and Dukes, 2007). Plants are preferentially selected for large size, flower size and profusion, as well as fruit crop size that are all factors (Mack & Lonsdale, 2001). These are all attributes that may mean these non-natives will be able to survive and compete within their new range if they can spread into the wider environment. Plants are often assisted directly into their new environment via planting in gardens or for amenity (Blackburn *et al.*, 2011).

This helps to facilitate the next stage, the Introduction. Most plants fail to cross physical barriers of fence or hedge, however they may become pollen donors to form hybrids with natives or even non-natives already present in the wider environment. Cultivation is more likely to help plants into the next stage, Establishment, by assisting the non-natives across the next barrier, Survival. These species may suffer less as a result of environmental stochasticity and low population size (Mack, 1995; 2000). Important traits in achieving establishment include phenotypic plasticity, wide abiotic tolerances, fast growth/short juvenile period, self-compatibility and germination without pretreatment (Goodwin *et al.*, 1999; Sakai *et al.*, 2001). The next barrier to full establishment is reproduction and for this there is a need for population growth. This is dependent on the existence of a self-sustaining population over a period of time that would involve multiple generations. Biotic filters come into play and may be the most important constraint on population growth (Theoharides and Dukes, 2007). If the natives present in the community share resource acquisition traits they are likely to compete strongly however if the non-native is in a functional group that is not present, it may encounter reduced competition (Melbourne *et al.*, 2007). Advantageous traits such as allelopathic root exudates, fast growth and high fecundity are important attributes help to overcome this barrier (Richardson & Rejmanek, 2004; Dietz & Edwards, 2006)

Dispersal is the barrier that has to be overcome to achieve final stage, Spread, in the new environment. It has been suggested that diverse native communities provide greater competitive resistance, probably due to reduced resource availability for the invader and the greater likelihood of competitive natives (Hooper *et al.*, 2005). Herbivores, parasites, pathogens, soil biota pollinators and dispersal agents also influence non-native establishment. The may be other factors that may benefit the non-native in the new environment. The enemy release hypothesis suggests that non-natives benefit from transport outside the range of their natural enemies (Elton, 1958; Carpenter & Cappuccino, 2005) and may improve competitive ability and explain disproportionate success of non-native plants in new ranges. Availability of abiotic resources such as increased light, moisture and soil nutrients have been shown to increase non-native success (Burke & Grime, 1996; Davis & Pelsor, 2001) and it is important that any gains made during these periods of high resource availability must be retain through storage organs such as rhizomes so that this advantage is maintained (Melbourne *et al.*, 2007). However studies of species by measuring the rate of establishment and spread have shown that there is no correlation between the mechanism of invasion and the subsequent impact to the invaded ecosystem (Ricciardi & Cohen, 2007). It is estimated that as few as 50% of non-native plants in general can be classified as ecologically harmful (Richardson *et al.*, 2000b)

There is evidence that soil communities favour non-natives over natives (Reinhart *et al.*, 2003; Wolfe & Klironomos, 2005). Mutualism with native pollinators and seed dispersal agents in introduced region are also vital to ensure establishment (Richardson *et al.*, 2000a). It is unlikely that plants with a very closely coevolved pollinator or disperser will find a replacement. However plants that are pollinated by generalists, possess vegetative reproductive, or are self-compatible or even have two or more of these traits may have significant advantages (Richardson *et al.*, 2000a). Indeed non-native plants with simple flowers and abundant nutrients may draw native pollinators away from native species causing reduced seed set.

Key traits

The key factors considered to be important in promoting invasive success of non-native plants species are shown in Table A7 along with the likely impact climate change will have on these.

Table A7. Key traits for non-native plant species and their possible reactions to climate change. (Goodwin *et al.*, 1999; Sakai *et al.*, 2001; Richardson & Rejmanek, 2004; Dietz & Edwards, 2006; Thompson & Davis, 2011).

Key Trait	Reaction to climate change	Notes
Ecoclimatic matching; A good climate match between the native range and introduced area. Most likely to occur if species has a large native range leading to wide abiotic tolerances	Generally Increases	Assuming invasive species coming from areas of the world that experience warmer weather the future UK climate is likely to benefit these non-native species. However, a cold period (below 5^{o}C) during winter would be needed to allow seed germination.
Phenotypic plasticity; Ability of a particular genotype to express range of phenotypes across different environments	No change	There would be no change in this trait, but invasive species would be able to adapt to the changing conditions due to their already high phenotypic plasticity.
Fast growth/Competitive ability; Higher leaf nutrients, higher specific leaf areas, efficient use of resources, able to take advantage of resources e.g. capitalise on disturbance events, allelopathy	No change	There would be no change in this trait, but invasive species would be able to adapt to take advantage of climate-degraded habitats.
Short juvenile period generation time; Germinate quickly, flower in the first year from germination	Increase	Warmer summers likely to increase probability of this occurring as long as there is adequate rainfall.

Key Trait	Reaction to climate change	Notes
High fecundity; Each fertilisation produces a viable seed, devote more resources to reproduction, produce more seeds	Increase	An increase in temperature would result in more seeds setting. Particularly relevant in species with late flowering periods where warmer winters may allow the seed to sett before being degraded.
Self-compatibility; No need for pollinator	No change	
Germination without pretreatment; No need for stratification or scarification	Decrease	An increase in winter temperature could result in a lack of stratification and so a reduction in the number of seeds germinating.
Effective mode of dispersal both long- and short-distance; small seed size, mechanism for dispersal e.g. wind, rhizome production	Increase	Warmer summers likely to increase growth rate of vegetative structures long as there is adequate rainfall as well. They may also allow the completion of seed maturation for certain species e.g. *Acanthus mollis.*
Aesthetically attractive to Humans; Species that are chosen by humans for horticulture due to colour size, number of blooms, constancy of flowering	Possible increase	Warmer summers may allow plants to flower for a longer period, and possibly have a second flowering period.
Propagule pressure; Greater number of individuals and/or release event	Increase	For certain species, seedling survival may be increased due to a decrease in late frosts (e.g. *Impatiens glandulifera*, providing a cold period in winter is present). *Pinus pinaster* would see an increase in the number of seeds shed from cones after warmer summers. Survival of seedlings would be increased in some species of tree (e.g. *Ailanthus altissima* and *Quercus ilex*).

8.5.3. Other influencers of invasion success - invertebrates

Life history and ecological traits affecting invasiveness

Some of the ecological factors that may allow introduced species to spread include: i) lack of predators, competitors and parasites, ii) ability to adapt to new hosts, iii) ability to be an effective predator, iv) availability of artificial or disturbed habitats, v) high adaptability to novel ecosystems, vi) efficient dispersal (Moore, 2005; Pimentel et al , 2000). Five factors that have been found to be associated with failed invasions are low propagule pressure, abiotic resistance, biotic resistance, limited or inappropriate gene pool and a lack of mutualists (Zenni & Nunez, 2013). Most invertebrates are very dependent on appropriate environmental conditions, especially temperature, especially in temperate climates such as northern Europe. Most of the invertebrate invaders in Britain arrive from warmer climates and they are consequently at or close to their climatic limits in the country. Successful insect invasions are wholly reliant on the presence of suitable host plants or prey within the area they are invading, the impact of this dependence is determined by the range of hosts of the invader and the distribution of those hosts. Some invasive species are reliant on one host species or Genus, such as *Agrilus planipennis*, wheras others, such as *A. glabripennis* are polyphagous and able to develop within a range of hosts.

After habitat and climate, propagule pressure and human activity are the two factors which are most important in determining the invasiveness of an insect species (Evans *et al.* 2011). The initial propagule pressure can be determined by anthropogenic factors including trade routes, plant and produce movements, agricultural and quarantine practices (Worner and Gevrey, 2006) as well as successful migrations into Britain. The number of successful migrations into the UK will be determined by the number of potential migrants within range, the ability and tendency of the species to migrate and the prevalence of suitable climatic conditions such as south-east winds. The visibility / invisibility of invading invertebrates can influence invasion success. For example larger beetles such as *Anoplophora* are more likely to be noticed when they first invade than smaller beetles such as *Agrilus planipennis,* which means that *A. planipennis* is likely to have greater opportunities to spread before an eradication campaign is launched. There is also some evidence from work on spider communities that invasive species are more successful in altered landscapes. Hogg and Daane (2011, 2013) found that an exotic spider was more successful in a vineyard habitat and that its native competitors were more successful in the existing native habitat. The authors suggest that it may be possible to limit the spread of invasive species by increasing the availability of habitat for native species. However we have found in our own analysis of the species included in this report that this is not always the case and that by providing more suitable habitat for the invasive species we would simply give it a further advantage. This strategy would have to be well researched for each species in question since it must depend on the relative competitiveness of the invasive and native competing species.

Biotic environment

Insect communities themselves can control the movement of invasive species and this effect may be enhanced if the existing communities have already been disturbed by human activity (Carroll, 2008). Existing communities can reduce the populations of invasive species through competition (Paini *et al.*, 2008; Walters and Mackay, 2005), predation and parasitism. The effectiveness of the defence mechanisms of the invasive species against the existing insect community can also be important in determining whether the invasion is successful (e.g. Lundgren *et al.*, 2010).

It is possible to predict the invasiveness of a species within a region by comparing the pest species assemblages in the region with assemblages where the invasive species is already established. Species assemblages are assessed using self-organizing map, an artificial neural network algorithm. The similarities between assemblages with and without the invasive species in question are then assessed. It is more likely that the invasive species will be able to establish in the new area if species with which it is closely associated in its current range are present (Worner and Gevrey, 2006).

Key traits

It is hard to generalise key traits across all invertebrate invasives as they tend to sit along a range of values for each key trait. However, the factors shown in Table A8 could be considered important in promoting invasive success of non-native invertebrate species:

Table A8. Key traits for non-native invertebrate species and their possible reactions to climate change.

Key Trait	Reaction to climate change	Notes
Ecoclimatic matching; A good climate match between the native range and introduced area.	Generally increase	Assuming invasive species coming from areas of the world that experience warmer weather.
Propagule pressure; Greater number of individuals and/or release event	Increase	For species already present in the UK the warmer weather will lead to increased survival rates and so increased population sizes. For species not currently in the UK, no impact on propagule pressure is predicted when anthropogenic movements are considered. Climate change could increase the number of potential migrants from continental Europe.
Human activity; e.g. trade	Unknown	Human activities will alter due to climate change, but the impacts of this on invasive species are difficult to assess. New trade patterns may result from shifts in the areas which are optimal for timber production of different species.
Existing species assemblages; competitive pressure	No change	No major changes predicted that would impact invasive success.

8.5.4. Other influencers of invasion success - mammals

Like birds, mammals through a combination of behavioural adaptation and thermoregulation are disproportionately more likely to establish as invaders (Clout & Russell 2008; Hill *et al.* 2009). Even looking at the fossil evidence in history mammals have successfully invaded large swathes of Europe (Terry *et al.* 2011) through a combination of both migration and in situ changes in abundance. Figure A3 below is taken from (Hill *et al.* 2009) and shows the proportion of species in Great Britain that are made up of non-native species. It highlights how invasive species make up a large proportion of extant vertebrate taxonomic groups in Great Britain, especially the mammals and birds.

In the literature a large number of species risk assessments have been carried out and the species selected are almost identical to the ones outlined in Section 3 as part of this study (e.g.(Nentwig, Kuehnel & Bacher 2010)).

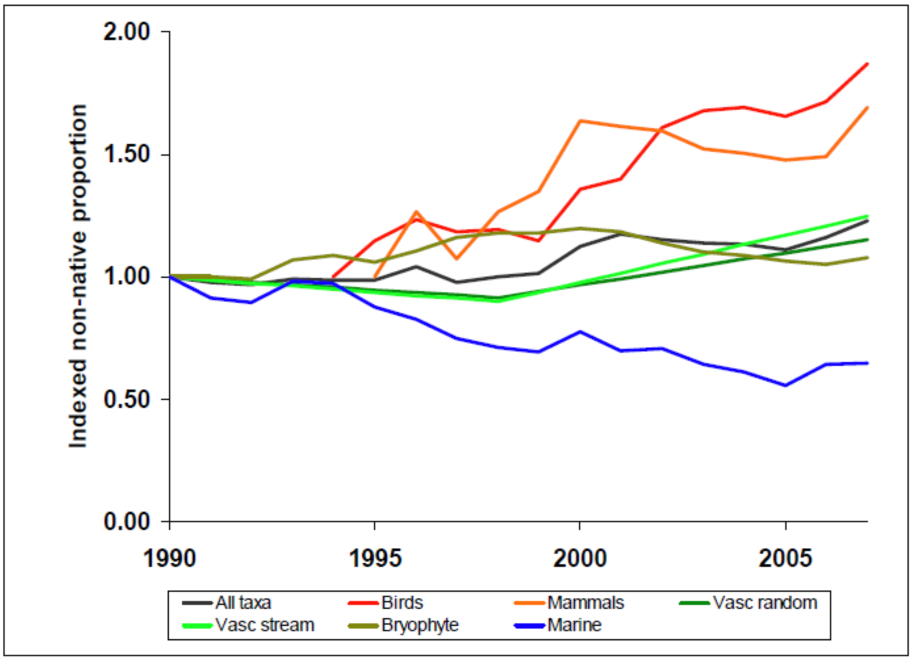


Figure A3. The abundance of successful invasive species based on taxonomic group (Hill *et al.* 2009).

Biotic traits

The following biological traits were assessed for the species selected in Section 3:

- Dispersal: none of the mammal species selected in Section 3 are sedentary, although the Chinese water Deer, capybara and coypu has been assessed as having only minor dispersal ability.
- Diet specificity: the aquatic rodents and herbivores are habitat and diet specialists, the remainder of the species are generalists, both in terms of habitat and diet.
- Large litters: Only the Leopard cat, the raccoon and the deer species have a fixed litter size. All of the other species have a flexible litter size and when habitat, climate and food availability is high litter sizes can also be high.
- Weaning age: all species have a low weaning age and are able to breed within their first year.
- Litters/year: the coypu, capybara, Muskrat, peccary and raccoon dog can have multiple litters a year. All other species are limited to 1 litter per year.
- Behavioural adaptations: the capybara and raccoon dogs exhibit evidence of behavioural adaptation to climate. Both species are able to attain a level of dormancy to overcome cold weather.
- Climate matching and native range: only the capybara and pudu are not climatically matched to the UK. All species have a large native range.

Sociological factors

Only two of the species, the raccoon and skunk are found frequently in urban habitats close to humans. Both of these species are also traded as pets (Parrott, Roy & Fletcher 2008). A proxy for the pet trade could be the proximity of urban habitats as these are centres of human population from where animals are traded. However it is difficult to quantify spatially the trading animals through the Internet.

Missing elements

There are number of missing elements in this study. For example, mammals often behave unpredictably in the way they move and invade or colonise new areas. For example in a study on whitetail deer (Long *et al.* 2005) it was found that habitat fragmentation actually has the capacity to speed up colonisation as animals move rapidly over on suitable areas until they find adequate woodland habitat. This may be true of a number of mammals and birds. Human mediated movement of animals is also under recorded and difficult to map and assess, and the pet trade itself is huge in the UK.

Fundamentally, across all taxa the influence of one invading species on another is complex. Non-native species may for example spread the seeds of non-native plant species (Lopez, altieri). This is hard to quantify for the purposes of this study.

Key traits

Traits that make mammals successful as invaders are shown in Table A9 (information on key traits synthesised from Forsyth & Duncan 2001; Forsyth *et al.* 2004; Clout & Russell 2008):

Table A9. Key traits for non-native mammal species and their possible reactions to climate change.

Key Trait	**Reaction to climate change**	**Notes**
Large geographic native range; Species exposed to a broad range of conditions and/or variable environment are more likely to establish.	Variable	Native range is likely to increase at northern edge (and higher altitudes) but may decrease at southern edge (and lower altitudes).
r – selected species; Species with short life spans, early weaning age and early age of first breeding	Increase	Increased temperatures could result in earlier weaning ages.
Migratory behaviour; Non-migratory species.	No change	
Ecoclimatic matching; A good climate match between the native range and introduced area.	Increase	Assuming invasive species coming from areas of the world that experience warmer weather with drier summers and wetter winters.
Successfully invaded elsewhere	No change	

8.5.5. Other influencers of invasion success - herptiles

Biotic traits

The following biological traits were assessed for the species selected in Section 3:

- *Dispersal*: with the exception of the turtles and wall lizards, which sedentary, all species are capable of maintaining a significant, albeit minor rate of dispersal.
- *Generalists/specialist*: All species are generalists in terms of their diet and habitat needs. Wall lizards are the only species to have highly specific habitat requirements.
- *Clutch size and age to sexual maturity*: All species have a large clutch size, and with the exception of turtles and Burmese python, all are able to breed within the first two years of hatching.
- *Evidence of behavioural adaptation*: Species show remarkable behavioural adaptations to changes in climate. Some pythons (which are particularly susceptible to climate (Dorcas, Willson & Gibbons 2011)) have been shown to modify their burrows appropriate to the local temperature. Cane toads and African clawed toads are particularly adaptive, and are capable of making use of man-made habitats to augment their water requirements. Both species are capable of migrating large distances overland (Fouquet & Measey 2006; Faraone *et al.* 2008; Florance *et al.* 2011). Invade managing man-made water sources to prevent the spread of cane toads has already been discussed as an option (Florance *et al.* 2011).
- *Climate matching*: The native range of only Alpine newts and marsh frogs climatically match that of the UK. All other species live in warmer climates and benefit from global warming (Roedder 2009; van Wilgen, Roura-Pascual & Richardson 2009). Indeed, the widespread is believed that the red eared Terrapin is unable to breed successfully in the UK due to a lack of consecutive days with temperatures above a minimum threshold ((Kikillus, Hare & Hartley 2010).
- *Large native range*: All species selected have a large distribution in their native range and have the capacity to adapt quickly as a result, as discussed earlier (Fisher-Reid, Kozak & Wiens 2012).

Sociological factors

All of the species selected (in Section 3.8) are important components of both the pet trade (Tingley *et al.* 2010), and research laboratories such as the African clawed toad. As a result it is important to integrate sociological factors into the drivers of species distribution. All species are capable of living in urban environments, and with the exception of the turtles and the Burmese python, all species are found living in close proximity to humans in other countries. Indeed many of the smaller amphibian species do well in garden ponds where water regimes are regulated and more predictable (Riley *et al.* 2005; Ernst, Massemin & Kowarik 2011).

Missing elements

As discussed earlier, there are some elements of the way invasive species spread which is complex, hard to map and assess. The pet trade, in particular the Internet trade is hard to assess spatially as items are sold across the world regardless of provenance. In addition the water plant trade for garden ornamentals may harbour the eggs of certain amphibian species causing an undetected spread and it is believed that some of the Alpine newt populations in the UK spread thus (Trent Garner, Evolution & Molecular Ecology Theme Leader, Zoological Society of London, pers. comm.).

It is also difficult to generalise across species. Some species such as turtles and terrapins are restricted by a lack of habitat connectivity and indeed this may be a good way of limiting their spread, whilst others such as bullfrogs actually spread quicker from patch to patch through fragmented habitats (D'Amore, Hemingway & Wasson 2010).

Key traits

The traits most influential for this taxonomic group are shown in Table A10, synthesised from a number of authors (Ficetola, Thuiller & Miaud 2007; Roedder, Sole & Boehme 2008; Semlitsch 2008; van Wilgen, Roura-Pascual & Richardson 2009; Fujisaki *et al.* 2010; Kolbe, Kearney & Shine 2010; Driscoll, Whitehead & Lazzari 2012):

Table A10. Key traits for non-native herptile species and their possible reactions to climate change.

Key Trait	Reaction to climate change	Notes
Ecoclimatic matching; A good climate match between the native range and introduced area.	Increase	Assuming invasive species coming from areas of the world that experience warmer weather with drier summers and wetter winters.
Water availability; Especially important for amphibians	Variable	Varies across the country due to predicted differences in rainfall. Overall it is likely to decrease in summer and increase in winter.
Human commensalism; Species that live in close association with humans in anthropogenically-modified habitats.	Increase	Warmer temperatures could lead to an increase in the number of tropical pets kept which increases the risk of non-native release (Essl 2011).
Juvenile development; A short time to maturation (van Wilgen, Roura-Pascual & Richardson 2009)	Increase	Increased temperatures could result in faster maturation.
Dispersal; High dispersal ability	Increase	Increased temperatures could allow herptiles to cross larger distances.

9. Appendix B Species analysis

9.1. Non-native species selection

The selection of invasive non-native species (INNS) for the scoring of traits and subsequent mapping process was undertaken using a two-stage process. First, using a combination of information sourced in the literature review and expert opinion a shortened list of known problem invasives taken from the GB non-native species secretariat portal was created. These species encompassed a broad range of ecology and landscape uses. Second, those species for which there was evidence of known invasiveness were then prioritised for their relative risk and the potential impact of climate change on the invasibility of species was also considered using expert opinion and information from the literature review. Those scoring the highest risk values, and those species likely to become more invasive under climate change were added to a short-list of species for scoring of traits.

In some cases, however, the latter stage was modified through the inclusion of species with lower risk scores, undertaken in order to ensure that the final group of species encompassed a broad range of ecology, habitat use and movements. A further step undertaken to ensure a broad range of species ecology was to include a number of important invasive species that were presently absent from England, but were considered to have a good probability of future introduction.

9.2. Assessment of species relative risk

A number of INNS risk assessment schemes exist. Most of these schemes, however, are too detailed (e.g. UK Non-Native Risk Assessment Scheme) and labour-intensive, or have been designed for specific taxonomic groups (e.g. EPPO scheme for invasive alien plants in Europe).

A more rapid scheme, particularly useful for the rapid screening of numerous species, is the Harmonia information system guidelines for Environmental Impact Assessment and List Classification of Non-Native Organisms in Belgium, developed by the Belgian Forum on Invasive Species

(http://ias.biodiversity.be/documents/ISEIA_protocol.pdf).

The Belgian risk assessment scheme allocates species to different list categories based on a simplified environmental impact assessment referred to as the Invasive Species Environmental Impact Assessment (ISEIA). The ISEIA assesses environmental risk only (not economic) and is relatively straightforward and time-efficient, requiring responses to ten questions in four categories or parameters (Table B1). The approach uses documented evidence from invasion histories in other areas to assess the potential for imposing adverse environmental effects in England. Non-native species that have impacted detrimentally on native species and ecosystems elsewhere are also likely to impose such effects in England.

The four variables that are evaluated are: (i) the potential for spread, (ii) colonisation of natural habitats, (iii) adverse impacts on native species and (iv) adverse impacts on ecosystems. Species are assessed against the four parameters on a three-point scale: 1, 2 or 3 corresponding to low, medium and high. Variables for which data are limited cannot be scored in this way and thus are assessed as 'unlikely' (=1) or likely (=2), using expert opinion. If data completely data deficient for a variable, it is scored as DD (=0).

The total risk score for a species is the sum of the risk rating scores from the four parameters. The total risk score range is 4–12, and is used to allocate species into three risk categories (Table B2). It should be noted that as the total risk score is derived from the cumulative scores of a suite of parameters a high impact score in one category of impact (e.g. predation/herbivory) would not automatically elevate that species into an overall high risk category. The full details of this scoring system for each of the species assessed in this way is provided in the separate document titled 'Non-native risk scores'.

In addition to the allocation of species to environmental risk (or impact) categories, species were also categorised according to one of four invasion stages (Table B3): (i) absent, (ii) absent from the wild but restrained in enclosed environments (e.g. zoological collections), (iii) scarcely established (isolated populations), and (iv) established and frequent locally (locally established). This represents a modification of the ISEIA scheme which uses only three invasion stages (absent, scarce and locally established).

The protocol, therefore, assigns species to a list system designed as a two-dimensional ordination (environmental impact x invasion stage) (Figure B1). High risk species that are already present in England (A_2 and A_1 species) pose the greatest threat and comprise a Black List; high risk species that are absent or confined to enclosed environments ($A_{0.5}$ and A_0 species) represent an Alert List; and medium risk species (B_2, B_1, $B_{0.5}$, B_0) form a Watch List.

From the results of the above method, 53 species were selected to be carried through to the next stage of the project (assessment of the likely interaction with changes to ecological networks (bigger, better, more joined up etc) and climate change).

The traits for each species selected using the methodology described above were scored according to their impact on the establishment of the species'. The matrix produced from this is supplied as a separate document to this report titled 'Trait Matrix for Selected Invasive Species'. Several 'mappable' traits are included in this matrix such as climatic and topographical preferences and are scored as follows from -3 to +3 where:

-3 = major negative impact (the species will decline)

-2 = negative impact

-1 = slight negative impact

0 = no impact

1 = slight positive impact

2 = positive impact

3 = major positive impact (the species will benefit)

Biotic traits are marked as either present (x) or not present (0) for each species.

Potential impacts of climate change were assessed in the trait matrix via the inclusion of climate degraded habitat. Descriptions of 'climate degraded habitat' came from the Defra report 'England Biodiversity Strategy – Towards adaptation to climate change' (Mitchell *et al.* 2007) and describe how much habitats will be affected by future climate change. The descriptions can be found at Appendix C. Each species was assessed against these descriptions using expert knowledge. The likely impact that these climate degraded habitats would have (e.g. either increase the use of that habitat by the invasive species considered [positive], decrease the use [negative] or have no effect on habitat use) was stated.

The evaluations for the presence or absence of the traits associated with increased invasion success, using published literature, are shown in Appendix D.

Table B1. Scoring system

Score	Dispersal potential	Colonization of high-value habitats	Species Impact Score (predation/herbivory, competition, disease and genetic pollution)	Ecosystem Impact Score (nutrients, hydrology, destruction of nursery areas, modification of succession or food webs)
1	Not spreading in the environment, with either or both poor dispersal or low reproduction	Restricted to man-made habitats	Data from invasion histories suggest that the negative impact on native populations is negligible	Impact is considered negligible
2	Not spreading by more than 1 km per generation unless spread by humans; may become locally abundant because reproducing strongly in situ	Usually confined to habitats with low or medium conservation value, but may occasionally colonize high-value habitats	Non-native species is known to cause local changes (< 80%) in population abundance, growth or distribution of one or several native species	Impact is moderate and easily reversible
3	Fecund and readily spreading by more than 1 km per generation	Often colonizes high-value habitats; at least one type of high-value habitat is readily colonized when a source population is in the vicinity	Non-native species often causes local severe (>80%) population declines in native species (includes exotic plants forming dense mono-specific stands, even where potential for replacement is poorly documented)	Impact is strong and difficult to reverse

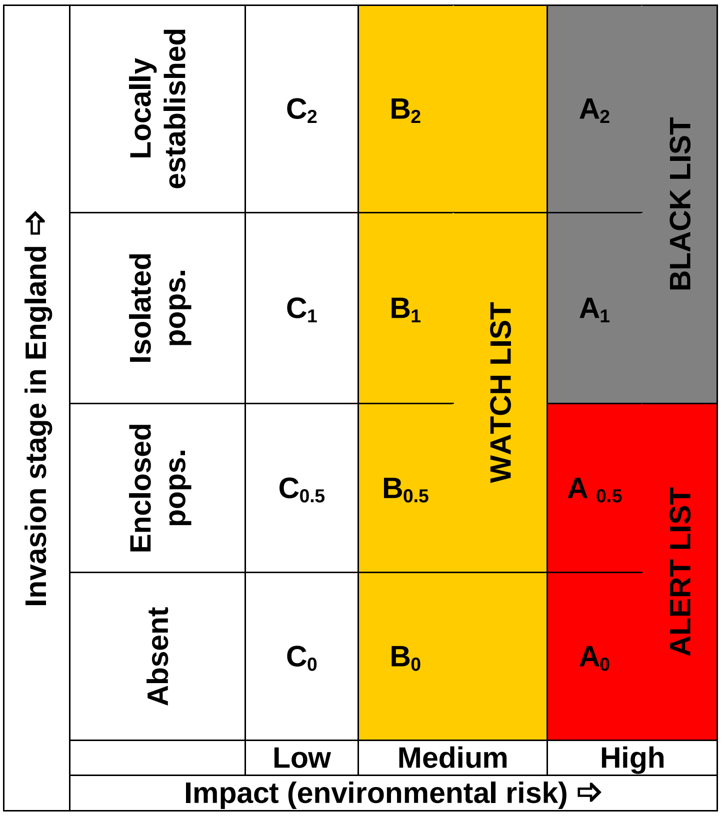
Scoring of adverse impacts on native species and ecosystems involves scoring in each of four sub-categories:
Adverse impact on native species: (i) predation/herbivory, (ii) interference and exploitation competition, (iii) transmission of diseases to native species, (iv) genetic effects such as hybridisation or introgression with native species.
Adverse impact on ecosystem function: (i) modifications of nutrient cycling or resource pools, (ii) physical modifications of the habitat, (iii) modification of naturalsuccessions, (iv) disruption of food webs.
Species Impact score = maximal score recorded for predation/herbivory, competition, disease and genetic effects.
Ecosystem Impact score = maximal score recorded for nutrient cycling, physical modifications, natural successions and food webs.

Table B2. Total risk score and list category.

Total risk score	List category	Environmental risk
11–12	A	High
9–10	B	Medium
4–8	C	Low

Table B3. Invasion stage.

Score	Category	Mnemonic
0	Not present in England	Absent
0.5	Absent from the wild but restrained in enclosed environments, e.g. zoological/botanical collections, or kept as domestic pets.	Enclosed
1	Present in England and either not established or with isolated populations that have not spread more than 10 km from their source	Isolated Populations
2	Local populations present in less than 10% of England, with some having arrived from further than 10 km from their source; or if more widespread then populations scattered and sparse	Locally Established



	Low	Medium	High
Locally established	C_2	B_2	A_2
Isolated pops.	C_1	B_1	A_1
Enclosed pops.	$C_{0.5}$	$B_{0.5}$	$A_{0.5}$
Absent	C_0	B_0	A_0

Figure B1. List system categorising non-native species by potential environmental risk and invasion stage in England. Figure adapted from the Invasive Species Environmental Impact Assessment (ISEIA) (http://ias.biodiversity.be).

9.3. Birds

Of the 324 avian species listed in the GB INNS portal, 23 were assessed (using expert opinion) as potentially invasive and thus evaluated using the ISEIA risk assessment scheme (Table B4).

Table B4. Impact scores and invasion stage of selected known invasive non-native avian species evaluated using the Belgian ISEIA environmental risk assessment scheme. Highlighted species were taken forward for the scoring of traits to support the mapping process.

Scientific Name	Common Name	Impact category	Invasion Stage
Alopochen aegyptiacus	Egyptian goose	A	2
Bubo bubo	eagle owl	A	1
Acridotheres tristis	common mynah	A	0.5
Threskionis aethiopicus	sacred ibis	A	0.5
Corvus splendens	Indian House Crow	A	0
Psittacula krameri	ring-necked parakeet	B	2
Cygnus atratus	black swan	B	1
Anser indicus	bar-headed goose	B	1
Tadorna ferruginea	ruddy shelduck	B	1
Branta leucopsis	barnacle goose	B	2
Myiopsitta monachus	monk parakeet	B	1
Bubulcus ibis	cattle egret	B	1
Cairina moschata	Muscovy duck	B	1
Chen caerulescens	snow goose	B	1
Netta rufina	red-crested pochard	B	1
Nycticora nycticorax	night heron	B	0.5
Chloephaga picta	upland goose	B	0.5
Acridotheres cristatellus	crested mynah	B	0.5
Aix galericulata	Mandarin duck	C	1
Aratinga acuticaudata	blue-crowned parakeet	C	0.5
Branta sandvicensis	Hawaiian goose	C	0.5
Psittacula eupatria	Alexandrine parakeet	C	0.5
Pycnonotus cafer	red-vented bulbul	C	0.5

On the basis of the ISEIA impact scores, ten avian species were selected for the evaluation of traits – species highlighted in Table A9.

This group of species comprised the five species scoring the highest environmental risk scores (i.e. impact category A – high risk), four (of 11) species scoring medium risk (impact category B) and one (of 6) species scoring low risk (impact category C). The rationale behind this group selection was to include those species scoring the highest risk whilst also encompassing a wide-range of avian species groups and ecology, i.e. species exhibiting a diverse range of habitat use and movement.

For all of the selected invasive avian species, inherent traits are likely to enable them to overcome changes in the availability of preferred habitat as a consequence of climate warming. All of the species possess behavioural flexibility that will facilitate the utilisation of altered habitat (e.g. changes in nesting or feeding behaviour) and/or allow dispersal to more suitable habitat.

9.3.1. Species descriptions

The following section summarises the selected invasive species ecology, including (where information was available) details on habitat preferences and movements.

Eagle owl

The range of the eagle owl is North Africa, most of Europe (except some western and northern parts), to eastern Asia (except south-east), India, the Middle East and North Africa (REF). The species inhabits forests, steep rock and ravine regions and mountain cliffs but is known to occupy a range of habitats for breeding, using woodland, heathland, mixed agricultural landscapes and even urban habitats within the western European component of its known range (Cramp *et al.* 1985). Habitat use varies between individuals in the same population with respect to social status (i.e. breeders and floaters) (Campioni *et al.* 2012b). Breeders and floaters prefer forest stands with a different vertical structure - breeders prefer more mature stands characterised by higher trees. The habitats/climatic conditions in England are similar to those seen elsewhere within the species' known range (Martinez *et al.* 2003).

A study of radio-tagged eagle owls (n=34) showed that both external (i.e. habitat structure and composition) and internal (i.e. sex and health state) factors explained a substantial amount of the variation in home range behaviour (Campioni *et al.* 2012a). At the broader temporal scale, home range and core area size were negatively correlated with landscape heterogeneity. Males showed (i) smaller home range and core area sizes, (ii) more complex home range internal structure and (iii) higher rates of movement. The better the physiological condition of the individuals, the simpler the internal home range structure.

They are a territorial and largely sedentary species and although dispersal rates are low (Olsson 1997), dispersing youngsters may undertake significant movements, as revealed by the recovery data from two British ringed youngsters (moving 160km and 218km respectively) and from satellite tracking work carried out in Switzerland (Aebischer *et al.* 2007). The species has a diet consisting of a range of small mammals, game birds, wildfowl, gulls, other birds (including raptors), snakes, lizards, amphibians, fish, invertebrates.

Eagle owls have a relatively low annual reproductive output; single brooded; 2-3 eggs per clutch and around 2 young fledged per nest. However, birds are long-lived (up to 26 years in the wild) and lifetime production can be relatively high - e.g. during 1996-2005 a pair bred annually in North Yorkshire successfully rearing a total of 23 young.

Indian house crow

Indian house crows are found in the Indian sub-continent in sub-tropical and tropical lowlands and hills. Although native to India and present in Africa and the Middle East, the house crow is now also breeding in The Netherlands, in north-west Europe. The transfer of house crows into new countries is mainly ship-assisted, with birds often travelling over thousands of kilometres.

Records on the distribution of the house crow indicate that populations are largely restricted to coastal areas (Bijlsma & Meininger 1984, Ryall 1994, 2002); having spread from their original introduction site. Where there are suitable resources, however, house crows can also be found further inland (e.g. Yemen, Ryall 1994; Utrecht, Netherlands, Ottens & Ryall 2003). House crows are most abundant in areas where they can benefit from improper human food and refuse handling, such as commercial areas, public housing areas and urban parks (Lim *et al.* 2003). No populations are known to live independently of man (Nyari *et al* . 2006).

House crows are highly social, staying in a locality for perhaps their whole life. Movement into new areas is often as a result of major disturbances (e.g. attempted unorganised control) on nest or roost sites. Adult birds, however, regularly travel up to 20km per day to known feeding areas. They are single brooded (in Kenya can be double) and lay 3 to 5 eggs per clutch. It is likely that reproductive output is lower in temperate Northern Europe than in its tropical and sub-tropical range.

The species is omnivorous with a wide-ranging and opportunistic diet, consuming a variety of plants and animal species. Closely associated with people, inhabiting urban/semi-urban areas; takes advantage of scavenging opportunities provided by discarded food items and refuse dumps.

Ring-necked parakeet

The ring-necked parakeet is found in India and sub-Saharan Africa. It has successfully established breeding populations in 35 countries across five continents as a result of introductions by man (Butler, 2003). The species is found in a wide variety of environments including urban and semi-urban areas (Lever, 1987). They are secondary cavity nesters preferring higher cavities but not limited by tree species and even choosing to nest in buildings (Ali & Ripley, 1969). Parakeet numbers are strongly associated with tree cavity density, suggesting that cavity availability is a limiting factor (Strubbe & Matthysen 2007). Across Europe the establishment of ring-necked parakeet populations has been more successful in areas more densely populated by humans, probably because of the high food availability in human-dominated areas (Strubbe & Matthysen 2009). Parakeet distribution is primarily governed by the amount of older forests, parks and built-up area in the landscape, reflecting the parakeets' requirement for suitable nesting cavities and reliance upon urban areas to forage (Strubbe & Matthysen 2007, 2011). Ring-necked parakeets prefer to forage in city parks, gardens and orchards rather than forests or agricultural areas.

Parakeets are largely sedentary species (Juniper & Parr 1998) and introduced populations appear to spread slowly during establishment (Butler 2003). In England, the rates of spread of the well-established ring-necked parakeet population in London were calculated between the year 2004 and 2009 (Parrott *et al.* 2011). The mean rate of spread of parakeets was 2.89km year-1 (averaged over 15 vectors). There was, however, considerable variation in both annual spread and the overall distance moved when considering each individual vector; the temporal pattern of spread was also erratic. It was notable that even vectors with slow net rates of progress could show significant annual advances. This suggests that even where mean rates of spread are relatively low parakeets have the potential to make considerable annual advances across apparently inhospitable landscapes.

Nesting success in its native range is reported between 1.4 and 3.1 fledged young and in the UK it was found to be 1.9 (Butler 2003). More recently, (Parrott *et al.* 2011) found 2.2 ± 0.23 (n=39) fledglings per breeding pair and 2.6 ± 0.20 (n=33) fledglings per successful pair. The parakeets are able to produce second clutches within one season and are also long lived.

Parakeets are a generalist feeder, consuming a variety of cereals, weed and tree seeds, fruit, nuts and flowers (Juniper & Parr 1998). A total of 58 different plant species have been recorded as consumed (reviewed in Parrott *et al.* 2010).

Sacred ibis

The sacred ibis is spread throughout Africa from south-western Mauritania, Senegal and Gambia East to Somalia, and Ethiopia and South to South Africa; south-eastern Iraq. Non-native breeding populations are established in France, Italy, the Netherlands and the USA

(Banks *et al* . 2008, Clergeau & Yésou 2006, Herring & Gawlik 2007*,* Ottens 2006, Yésou & Clergeau 2005), which have similar ecoclimatic conditions to England. They are found in a wide variety of habitats in both its native and introduced range, including meadows, rubbish dumps, marshes, reedbeds, seashore and ploughed fields (Clergeau & Yésou 2006 and are a gregarious species, forming large colonies near waterways. Sacred Ibises are very adaptable and will feed in a variety of man-made habitats including rubbish tips, farmyards, ploughed fields and even slurry pits (Clergeau & Yésou 2006).

In France, the sacred ibis, which escaped from zoos during the 1990s, is well established and has spread along the Atlantic and Mediterranean coasts (Clergeau & Yesou 2006). Breeding in natural habitats was first recorded at three locations 25km, 70km and 150km respectively from the sources of introduction. Dispersal of the species has mostly occurred on wetlands along the Atlantic coast but some birds have been observed hundreds of kilometres away and appear to have dispersed along the river Loire. Overall, inland records are rare.

Breeding colonies occur near large waterbodies, most often in trees, but also in low scrub or on the ground, particularly on islands. In its introduced range, all the breeding colonies in France have formed near the coast, but in Italy the species breeds at an inland site. In its native range this species often breeds at inland sites near water. Sacred Ibises lay 2-4 eggs per brood, but not all young survive with 1-2 fledglings per pair on average (Cramp & Simmons 1977). They are long lived (up to 21 years in the wild).

The species is omnivorous, but largely predatory, feeding on amphibians, crustaceans, small rodents, molluscs, fish, earthworms, insects and the eggs and chicks of other bird species (Cramp and Simmons 1977, Kopij *et al* . 1996, Clergeau & Yésou 2006).

Bar-headed goose

The bar-headed goose is found in central and southern Asia. During the breeding season, bar-headed geese live near mountain lakes and prefer areas with short grass. From their native breeding grounds they migrate over the Himalayas to over-winter in India and northern Burma. In winter they graze in areas cultivated for wheat, barley and rice crops; the diet occasionally includes crustaceans and invertebrates.

In the UK, the species is commonly kept in ornamental waterfowl collections and birds seen in the wild are all escapees. Currently there are around 100 widely dispersed individuals in the UK.   Around 10 pairs breed annually producing a single clutch of 4-6 eggs per pair.

Black swan

Black swans are found throughout Australia with the exception of Cape York Peninsula; and are more common in the south. In 2006/07 WeBS, black swans were recorded at 73 sites across GB; the majority of records were of single birds; 16 sites held peak counts of three or more birds. They prefer larger salt, brackish or fresh waterways and permanent wetlands.

In permanent suitable habitat species is largely sedentary; in ephemeral habitat young and adults wander extensively. The species is territorial and stay in solitary pairs when mating but are known to occasionally mate in colonies. They are herbivorous, eating aquatic vegetation, also terrestrial plants in pastures or on farm land.

Ruddy shelduck

The species is found in northern Africa, eastern Mediterranean and Asia. In England there are occasional records. Highest monthly count in 2006/07 WeBS was 10, five of which were on North Norfolk coast. They are less dependent upon large water bodies for resting and feeding than most other Anatidae and can congregate into larger flocks during the autumn and winter, but are more characteristically found in scattered small flocks.

Ruddy shelducks are dispersed in pairs during the breeding season, although may form small nesting groups and are omnivorous eating grain, vegetable shoots, tubers, aquatic insects, molluscs, worms, small fish, amphibians and reptiles.

Egyptian goose

The Egyptian Goose is a widespread species throughout its natural range - Western, Eastern and Southern Africa (Brown *et al.* 1982). Egyptian Geese have been widely introduced in western Europe, and have successfully formed established self-sustaining populations in a number of countries - Belgium, Denmark, France, Germany, The Netherlands, Spain, Switzerland and the UK (Banks *et al.* 2008).

The species occupies a broad range of wetland habitats including lakes, ponds, reservoirs, estuaries, sewage works, swampy woodland and meadows. On farmland, birds can sometimes be found on arable fields some distance from water, and on silage clamps.

As summarised in Gymesi & Lensink (2010) and Marchant (2012) Egyptian geese nest in a wide variety of locations: on the ground, often on small islands, but also in burrows, in caves, among reeds, in trees in old nests of other birds and on buildings, at up to 3km from water (but more usually several hundred metres). In trees, they may use large holes or old nests, such as buzzard *Buteo buteo*, goshawks *Accipiter gentilis*, magpie *Pica pica* and crow *Corvus corone*.

The species is largely sedentary over much of its range although it may make seasonal nomadic or dispersive movements related to water availability. Individuals occur solitarily or in pairs, but following breeding annual dispersal to moult sites occurs where birds gather in large aggregations of hundreds or even thousands (Del Hoyo *et al.* 1992; Kear 2005). In The Netherlands, dispersal can be hundreds of kilometres away from the breeding area (Lensink 1999).

In England the reproductive success of Egyptian Geese is relatively low (Sutherland & Allport 1991), however in the Netherlands the species has a six-month breeding season with much higher productivity leading to more rapid population expansion (Lensink 1999). It is unclear why this difference between two populations in apparently similar habitats and similar climates exists.

In The Netherlands, dry summers negatively influence survival: the amount of summer rainfall was positively correlated to the survival of both adult and young animals; probably through promoting grass growth, and hence food abundance (Gymesi & Lensink 2010). In addition to grasses, the Egyptian goose has a wide diet of seeds, leaves, grasses, and plant stems, occasionally locusts, worms, or other small animals.

Red-vented bulbul

The species is found in southern and south-eastern Asia, from Pakistan, east through India and Sri Lanka to the Vietnam region. They live around secondary growth and shrub, cultivation, parks and gardens. They are also found in forest and agricultural areas. Bulbuls are found in pairs or small flocks in the non-breeding season and are sedentary. They consume fruits (e.g. bananas, lychees and papaya), berries, insects, flower nectar, seeds and buds.

Common mynah

This species is found in India and the Middle East in open countryside, close to human establishments. Prefers modified habitat; communal roosts; cavity nester; pairs stay together returning to the same territory each year. Common mynah are sedentary throughout the year but can travel up to 12 km between roosts and feeding areas. They are adaptable omnivorous scavengers consuming invertebrates, fruit, grain, birds’ eggs, small reptiles and food scraps; predominantly ground feeders.

9.4. Plants

Using expert opinion, non-native plants that were known agricultural weeds were removed as they are limited to the arable environment and can be controlled through standard crop protection practices.. Also, aquatic plants were not considered as there is a plethora of knowledge about them, with a disproportionate number of risk assessments that have been produced. Although their threats maybe greater to the habitats where they occur, their effect on the landscape and ES is limited to a few options. The remaining terrestrial plant species listed on the INSS were further reduced using several other reviews and assessments of plants that currently, or may in the future, threaten biodiversity within Europe and the main candidates from these exercises were selected for this process. These were as follows:

- The European and Mediterranean Plant Protection Organisation (EPPO) EPPO activities on Invasive Alien Plants EPPO (2013).
  - Provided a list of 32 species within the region and a few outside the region that were recommended for regulation as quarantine pests.

- ‘Delivering Alien Invasive Species Inventories for Europe’ (DAISIE, 2012).
  - This added a further 17 species from their top 100 invasive species list that included animals and plants.

- European Network on Invasive Alien Species (NOBANIS). Concentrates on north and central Europe. Although this may seem outside the Atlantic and Continental bio-geographical zone that includes England, there are representative countries within the same zone such as Eire, Denmark, Belgium and the Netherlands.
  - This had 22 species of plants amongst its 100 invasive species list.

- Horizon scanning for new invasive non-native animal species in England (Parrott *et al.*, 2009).
  - This had risk categories attached to the 93 species that equated to the threat that they posed to biodiversity.

- Developing an indicator of the abundance, extent and impact of invasive non-native species. Hill, M. O. *et al.* (2009).
  - This had risk categories attached to the 25 species that equated to the threat that they posed to biodiversity.

- Non-Native Species in Great Britain: establishment, detection and reporting to inform effective decision making. Roy, H. *et al.* (2012).
    - This had 66 plant species amongst its invasive species list.

- Here today, here tomorrow? Horizon scanning for invasive non-native plants. Plantlife (2010).
    - This had 44 terrestrial with critical ranking, the highest threat posed to biodiversity.

- IUCN list of 100 of the World's Worst Invasive Alien Species (Lowe *et al.*, 2000).
    - As this is a global system, most of the species were tropical however it did provide a few temperate species that may pose a threat with climate change.

The lists were combined, duplicates were removed and this gave a combined total of 174 species. These were checked against the NNSC list and 35 species were missing from this list. These were not present in Stace (1997) or the Online Atlas of the British and Irish flora (Plant Atlas, 2012). These are not yet present in this country (invasion stage 0) but represent a real threat to biodiversity if introduced especially in the future with climate change. The remaining species were researched using Stace (1997) and the Plant Atlas (2012), together with the fact sheets provided by NNSC, EPPO, NOBANIS, DAISIE and IUCN.

The extent of their population development within England was assessed using the criteria from Blackburn *et al.* (2012) and the risk assessment as described in Section 3.2. Ten species were chosen to be included in the mapping exercise (Table B5). These were representatives of the growth types (3 trees, 4 shrubs and 3 herbs) and strategies of the different non-natives plant species present in England. Their descriptions were put together from various web resources and these are listed in Appendix E. The other species reviewed for this exercise are found in the separate document ‘Non-native risk scores’.

Table B5. Impact scores and invasion stage of selected known invasive non-native plant species evaluated using the Belgian ISEIA environmental risk assessment scheme and invasion stage (Blackburn et al 2012).

Scientific Name	Common Name	Impact category	Invasion Stage
Ailanthus altissima	Tree of heaven	B	C3 Localized
Buddleja davidii	Summer lilac, Butterfly-bush, or Buddleia	A	E Widespread
Cotoneaster horizontalis	Wall Cotoneaster	A	D2 Widespread
Impatiens glandulifera	Himalayan Balsam	A	E Widespread
Pinus pinaster	Maritime pine	A	D2 Localized
Prunus serotina	Black cherry	A	D2 Localized
Quercus ilex	Holm oak or Holly oak	A	E Widespread
Rosa rugosa	Japanese rose	A	D2 Localized
Senecio inaequidens	Narrow-Leaved Ragwort	C	C3 Localized
Solidago gigantea	Giant goldenrod	A	E Widespread

9.4.1. Species descriptions

The following section summarises the selected invasive species ecology, including (where information was available) details on habitat preferences.

Ailanthus altissima Tree of heaven

Ailanthus altissima is native to subtropical and warm temperate climates in Asia and northern Oceania, but is also able to invade climates ranging from cool temperate to tropical (Cronk and Fuller, 1995). It is a small to medium-sized broadleaved tree, often 6-10 m tall but up to 30 m. *Ailanthus altissima* seed was first brought to the UK in 1751. It is a seed or vegetatively propagated tree. Flowers are unisexual, produced in large panicles at the end of the branches and up to 1 million seeds are produced per tree. Established trees also produce numerous suckers from the roots and sprout vigorously from cut stumps and root fragments, growing up to 3 cm per day. *Ailanthus altissima* is very difficult to remove once the tap-root is established and seedlings should be removed as early as possible. With larger plants, two cuts per year will be required over several years (Anon., 2002). It may be confused from a distance with a number of other trees such as *Fraxinus* and *Juglans.*

Ailanthus altissima has become a weed in cultivated, pasture, forestry, wasteland and urban areas. Seedlings and root suckers grow rapidly and spread prolifically and thus quickly outcompete many native species for light and space. Prolific fruiting, ready germination, adaptability to infertile sites and a rapid growth rate make *A. altissima* a noxious weed in many countries where it has been introduced. It also produces a toxin in its bark and leaves which accumulates in the soil inhibiting the growth of other plants.

Buddleja davidii Buddleia

It is native of temperate central and southwestern China. It is a medium to large perennial shrub especially prevalent on urban and disturbed sites. *Buddleja davidii* rapidly forms dense thickets becoming the dominant vegetation, outcompeting natives. It tolerates a wide range of soil types and conditions, but prefers dry, open sites as it is shade intolerant and is frost tolerant. It was introduced as an ornamental plant and has rapidly spread through lowland England. It is readily pollinated by native insects and a standard plant can produce up to 3 million seeds per year. The seeds show lengthy dormancy, remaining in seed bank for several years.

Buddleja davidii is increasing its range and frequency and it does this with seed adapted for wind dispersal. These can be distributed over long distances by wind currents. This dispersal ability is well suited for movement along transport networks such as roads and railways. *Buddleia* is able to colonise a new zone in one to two years from seeding. A shrub can flower and bear fruit in the first year. *Buddleja davidii* is a coloniser of short lifespan. The largest densities of invasion would normally be observed in the first ten years. It is often associated with habitation, and is found on waste ground, walls, railway banks, quarries and areas of scrub.

Cotoneaster horizontalis Wall Cotoneaster

This is a deciduous to semi-evergreen procumbent shrub, native of western China. It was first introduced to gardens in 1880. Each plant produces many dark red fruits, 1 cm in diameter, in autumn. These fruits are very attractive to birds, thrushes and finches, and these are its main means of dispersal. Each fruit has three seeds which germinate as a clone of the parent plant as they are produced without fertilisation. The seeds need up to six weeks of cold stratification to germinate. Once established it spreads rapidly and its prostrate growth form smothers other vegetation. It is tolerant of dry, nutrient-poor sites and therefore is able to invade semi natural habitats of high conservation value such as dry limestone grassland, limestone pavement and cliffs. AIs range is associated with light soils in southern England.

Impatiens glandulifera Himalayan Balsam

Impatiens glandulifera is a glabrous annual, 50 to 250 cm tall. Roots are up to 15 cm deep, and it often forms numerous adventitious roots from the lower nodes. The native range is the western Himalayas, and it was first introduced to the UK in 1839 as a garden ornamental. The spread was enhanced by beekeepers who released the plant into the wild on many occasions. Each plant produces numerous nectar-rich and scented flowers that attract many more pollinators than native plants, and thus has a negative effect on the fitness of the natives (Chittka and Schürkens, 2001). Each resulting seed capsule contains from 4-16 seeds (Beerling and Perrins, 1993). Its spread is rapid, with seeds transported over large distances along rivers and aided by humans. In the UK, the rate of spread was estimated at up to 38 km per year (Perrins *et al.*, 1993).

It grows in moist, semi-shaded damp areas, sparse woodland and is particularly dominant on the banks of watercourses. Associated flora is often reduced in vigour and numbers due to the superior competitive strength of *I. glandulifera*, though not excluded, as being an annual, is not present in the vegetation for the whole growing season. It germinates in spring and reaches dominance in the summer, and plants completing their life cycle in spring or early summer are little affected. *I. glandulifera* has been noted as leading to increased riverbank erosion in places as it leaves soils bare when it dies back in winter, although in general, it is often integrated in perennial vegetation.

I. glandulifera has spread rapidly in many parts of Europe and North America after its introduction as an ornamental and spread is likely to continue. Due to its ability to form dense stands it has been blamed for negative biodiversity effects. The spread is likely to continue with global warming to more northerly or montane areas (Beerling, 1993).

Pinus pinaster Maritime pine

This is an evergreen coniferous tree. It can grow to 30 m and is a native of the Iberian Peninsula, south and western France and Morocco. In its native range it grows from sea level to 600 m. It was widely planted in southern England for shelter belts and small plantations. It has been known in known in the wild in England since 1850. It disperses via wind-dispersed seeds. Each female cone takes two years to ripen, and the seeds are released over a several years. High temperatures induced by fire or hot weather cause the cones to open and this accelerates the release of the seeds. Germination is higher in mineral soil than litter, and occurs profusely after fire. Extensive patches develop rapidly that exclude other native vegetation.

Pinus pinaster prefers open dry habitats and alters heathland in particular as it shades out the natives, hastens succession to woodland. The tree has become naturalised in warm, sandy areas such as heathland close to the sea and on sandy heaths in Dorset and Hampshire.

Prunus serotina Black cherry

Prunus serotina is a deciduous, single-stemmed, medium- to large-sized tree, native to North America. It is associated with areas with a warm to hot wet summer and a cool to cold, wet winter, and is hardy to zone 3. In Central Europe, *P. serotina* is mostly a shrub. In Europe *P. serotina* occurs in both forests and in open vegetation. It reaches the highest cover values in stands of shade-intolerant trees such as oak, pine, or birch but does not grow well in dense shade. It tolerates a wide range of moisture conditions and can invade wetlands, such as bogs and their degeneration stages. At the other extreme it also colonises dry grassland that is too dry for most other woody plants (Nobanis)

Prunus serotina produces cylindrical racemes, each with approximately 30 flowers. The following fruits are purple-black drupes, 8-10 mm diameter (Weber, 2003). *P. serotina* seeds are dispersed by birds, with their digestion increasing germination (Smith, 1975), and mammals including foxes and wild boar (Starfinger, 2004). Seed is produced in large numbers from 30-100 years old trees (Cronk and Fuller, 1995), beginning at approximately 7 years old (Starfinger, 1991). Seeds require cold stratification to germinate, and germination is higher in litter than in mineral soil. Seeds may remain viable for at least 5 years (Wendel 1972). In forests with shade tolerant climax trees, *P. serotina* is a gap-phase species (Curtis, 1959), capable of germinating and establishing, and rapid growth commences if a gap occurs. *Prunus serotina* grows well in temperate and moist climates and occurs on a variety of soils except in extremely wet or dry sites. It prefers lower east or north-facing slopes, and is most frequent on very acid and relatively infertile soils (Marquis, 1990). In the invaded range, *P. serotina* colonizes soils too dry for most woody species, such as dry grasslands, as well as very wet soils such as degenerated bogs. It is also found in semi-natural or managed woodland particularly on acid sandy soils, often following a disturbance event (Cronk and Fuller, 1995).

Quercus ilex Holm oak

Quercus ilex is a large spreading evergreen tree to 25m that is hardy to -15°C. It prefers deep fertile loam but succeeds in all soils, including shallow chalky and sandy soils, except those that are poorly drained. Young plants tolerate reasonable levels of shade. It is very resistant to maritime exposure and these were the areas where it was planted in southern England, along with urban sites. It is frequently self-sown in urban and brown-field sites, but also colonises semi-natural habitats such as chalk grassland, coastal habitats and lowland heath from its coastal plantings. Its range is increasing and is frequent in England south of a line from the Severn to the Humber. Its main dispersal agent is the Jay. Normally the tree is unable to withstand frost, which would normally prevent it from spreading north, but with climate change, it has successfully penetrated these areas. It is drought tolerant but undoubtedly its growth is improved by the increased moisture levels found in its more northerly range. This also increases the survival rates of its seedlings whose establishment are usually a limiting factor to population growth in its native range.

Rosa rugosa Japanese rose

It is a small multi-stemmed shrub that forms dense thickets, mainly in coastal habitats and is native to East Asia. It was established in gardens 1870's It has hermaphroditic flowers which are insect pollinated. Self-fertilisation does also occur. Seeds are developed in large rose hips that are very attractive to birds, the most important long-distance vector, and small mammals. On average each fruit contains 60 seeds with 85-95% viability. Seeds in the soil can remain viable for several years; dormancy is broken by a long cold period. The plants also reproduce by rhizomes. This leads to the formation of dense thickets occurs. It shows a preference to open, dry habitats. It can colonize acidic and basic soil alike, and is able to invade nutrient-poor habitats. Habitats occupied in its invaded range are altered to monospecific stands, with greatly reduced light availability and decreased number of native species. It can even outcompetes native relatives with similar habitat requirements such as *Rosa spinosissima*. These include coastal dunes and sandy shores, rock cliffs, ledges and shores, woodland fringes, clearings and tall forb stands, temperate shrub heathland and hedgerows. Invaded areas become impenetrable to humans due to the spiny thickets.

The native climate of Rosa rugosa is comparable to the climate in the region, and it thrives well in its new range. Rosa rugosa has photosynthetic leaf characteristics (in comparison to other roses) that may support the occurrence in more northern regions (Ueda *et al.* 2000). Furthermore, the species has several ways of spreading seeds and rhizomes, all of which have contributed to the naturalisation of the species. Rosa rugosa grows successfully in coastal areas, especially dunes, because the species is salt tolerant (Dirr 1978), and is adapted to moderate sand cover (Belcher 1977). The establishment in yellow dunes is supported by arbuscular mycorrhizae, which occurs in Ammophila species (Gemma and Koske 1997). Moreover, propagule pressure due to urban areas enhances spreading. hips of Japanese rose are extraordinarily buoyant and can float up to 40 weeks in both fresh water and seawater. After this the hips would disintegrate revealing the seeds inside. The seeds, however, are also buoyant for several weeks on their own, due to special tissues in the cell walls of the seeds.

Senecio inaequidens Narrow-Leaved Ragwort

Senecio inaequidens is a short-lived perennial herbaceous or woody shrub from South Africa. It is up to 100 cm tall, rising from a shallow taproot (Hilliard, 1977). Initial sites of introduction were associated with the wool trade. In Europe, the species has two main periods of flowering, one starting in July and one in September, lasting until January (Heger *et al.*, 2005). Its flowers are visited by many types of native insects (Ernst, 1998). Vegetative propagation also occurs by the rooting of stems that touch the ground (Ernst, 1998). Individual *S. inaequidens* plants vary greatly in the numbers of achenes that they produce. On average, 10,000 seeds are produced per plant each year. Achenes may remain viable for at least 2 years if they remain dry; dormancy lasting several months has also been observed (Ernst, 1998). They are mainly dispersed by wind over large distances. They can also be dispersed by birds and mammals. Germination occurs all year long, with the highest rates being spring and autumn. *S. inaequidens* is a prolific seed producer and has vigorous growth. It adapts to a wide range of environments.

Senecio inaequidens is associated with areas with a warm to hot, wet summer and a cool winter (dry or wet). It is hardy and well adapted to zone 7 (-18 to –12°C). The movement of seed by the motion of road and rail vehicles are considered an important transport pathway for long distance movement (Ernst, 1998). The plant is dispersed by colonizing communication paths, and then spreads to other habitats such as pastures and natural areas. In Europe, *S. inaequidens* grows on warm and dry ruderal sites, mostly with gravelly or sandy soil. It can also be found in tall herbaceous vegetation and among young shrubs, though never in deep shade (Bornkamm 2002). It is also found in natural environments such as dunes and cliffs in littoral areas, and temporary ponds in France (Brunel, 2003). It is toxic to livestock and humans and in its native South Africa it is a crop weed and repeatedly finds its way into bread (Bromilow, 1995). It is very actively spreading in Europe at present, and is likely to continue to spread, particularly along roads and railway tracks. The plant has adverse impact on crops and plant biodiversity (Bromilow, 1995).

Solidago gigantea Giant goldenrod

Solidago gigantea is a native to the USA and Canada between 30°N and 55°N. It was introduced into Europe as an ornamental plant in the mid 1700s. It is an erect, 50-150 cm tall, rhizomatous perennial with annual aboveground shoots and persistent belowground rhizomes. Seeds of European plants do not show dormancy and do not need scarification or stratification (Voser-Huber, 1983). Intensive shoot growth starts in April and increases nearly linearly until the end of July when final height is achieved. Inflorescences are formed from June onwards. Seeds are easily dispersed by wind, but in dry weather conditions only. Seeds are produced in very large numbers and long-distance dispersal is by wind, aided by dry conditions. It grows in a wide range of different soil conditions but is not shade tolerant (Ellenberg *et al.*, 1992). *Solidago gigantea* associated with areas with a cool to warm wet summer and a cool to cold winter (wet or dry). It is said to be hardy to zone 6 (-23 to –18°C). It invades poorly managed pasture and can be a considerable weed in forest nurseries Where introduced, *S. gigantea* is a principle weed in managed forests, and a serious weed in wetlands, on riverbanks/canal sides, on rail/roadsides, wastelands, urban areas and forest edges.

9.5. Invertebrates

Expert opinion was used to reduce the number of species listed by excluding insects recorded on the INNS portal as native or extinct from GB. None of the Ar*anea, Collembola, Hymenoptera Nematoda* or terrestrial *Mollusca* were regarded as high impact species on rapid screening. The only species of concern of the Platyhelminthes are New Zealand Flatworm (*Arthurdendyus triangulate*) which is already widespread and well established and Australian Flatworm (*Australoplana sanguinea*) which only affects garden centres and adjacent land (Santoro & Jones 2001; Jones *et al.* 2001). In addition, insects which have widespread naturalized populations (covering >10% of the country) or those with only indoor populations (such as storage pests) or those only known as migrants. The lists in Appendix F detail insect species excluded from the scoring exercise for other reasons. Impact scores were assessed for the 195 remaining species and these are shown in the separate document the separate document 'Non-native risk scores'. Ten species were chosen and these are detailed in Table B6.

Table B6. Impact scores and invasion stage of selected known invasive non-native insect species evaluated using the Belgian ISEIA environmental risk assessment scheme.

Scientific Name	**Common Name**	**Impact category**	**Invasion Stage**
Monochamus sartor	Sawyer Beetle	B	Localized
Cyphostethus tristriatus	Juniper Shieldbug	C	Widespread
Lymantria dispar	Gypsy Moth	A	Localized
Thaumetopoea processionea	Oak Processionary Moth	B	Localized
Anoplophora glabripennis	Asian longhorn beetle	A	Localized
Corythucha arcuata	Oak Lace Bug	C	Absent
Anoplophora chinensis	Citrus Longhorn Beetle	A	Absent
Agrilus planipennis	Emerald Ash Borer	A	Absent
Lasius neglectus	an invasive garden ant	B	Localized
Gelechia senticetella	cypress groundling	B	Localized

9.5.1. Species descriptions

The following section details examples of the use of habitat and movements by the selected insect species. Our rational for selecting each species from the list of those for which we calculated an Impact Factor is also given. These species were chosen based on their impact factor and distribution.

Anoplophora beetles

This species was chosen because it has a high impact factor and is not widely distributed in the UK. *Anoplophora glabripennis* (Asian longhorn beetle) and *Anoplophora chinensis* (Citrus longhorn beetle) are both invasive species from Asia. They have both been moved in trade from Asia to Europe and North America leading to numerous interceptions and a number of damaging outbreaks. Adult females lay their eggs in grooves cut into the bark of host trees. The larvae initially feed just below the surface of the bark, but within weeks, tunnel further towards the centre of the tree where they feed on the sapwood until they are ready to pupate. This lifecycle means that the juvenile stages are a) difficult to spot on imported trees and also on established trees when there are outbreaks; b) difficult to control chemically or with biological control agents. Both species are treated as quarantine pests across the European Union, member states are obliged to prevent their introduction and spread. *A. glabripennis* and *A. chinensis* have overlapping host ranges which include numerous native and exotic deciduous trees. In outbreaks of these pests in North America and Europe a large proportion of the trees infested have been Acers (maples). The lifecycle of both beetles takes at least one year to complete, but it can be two or three years in cooler climates. Both species are relatively cold tolerant, winter conditions are unlikely to be the most important factor in determining the survival of these beetles in the UK. At low population densities, adult ALB and CLB tend to either stay on the trees they have emerged from or disperse short distances (<100m), but both species have been shown to be capable of dispersing over 2km (Adachi, 1990; Bancroft & Smith, 2005; Haack *et al,* 2010).. Larger patch areas and a greater number of patches of broadleaved woodland would provide opportunities for greater populations of *Anoplophora* in the UK, depending upon the species composition of that woodland. Increasing the quality of patches would be unlikely to have an impact.

Anoplophora glabripennis, Asian longhorn beetle

This species was chosen because it has a high impact factor. The native range of Asian longhorn beetle (ALB) is China and Korea. ALB spread northwards (2°), eastwards (2°) and westwards (15°) in China during the twenty years from 1984-2004. This spread was linked to widespread reforestation programs that began in eastern China in the 1960s using mostly native *Populus dakuanensis* (Haack, et al, 2010). ALB has been spread from Asia to other parts of the world in infested wood packaging material such as pallets. The first outbreak of the beetle outside Asia was discovered in 1996 in New York. Since then numerous outbreaks have been discovered in North America and Europe. Over $400 million has been spent on eradication programmes for this pest in the USA. Under field conditions, 1,264 degree-days (DD) above a threshold of 13.4°C are required to complete development of (Yang *et al.*, 2000). An outbreak of ALB was discovered near Paddock Wood in Kent in March 2012. Over 2000 trees have been removed to eradicate the outbreak.

Anoplophora chinensis, citrus longhorn beetle

This species was chosen because it has a high impact factor. The native range of citrus longhorn beetle (CLB) is China, Korea, Japan, Malaysia, the Philippines, Myanmar, Taiwan and Vietnam. It is a destructive pest of citrus in Japan and lowland areas of China. It has been moved around the world in dwarfed trees and hardy ornamental nursery stock, especially in Japanese maples, *Acer palmatum*. It is not known to be present in the UK, but has been included in this study because it is a current known threat to our trees. There is a large outbreak of CLB close to Milan in Lombardy, Italy which was first discovered in 2000. The total number of degree days between egg laying and the date on which the maximum number of eggs are laid by a female developing from this egg is around 1900DD in excess of 11.2°C (Adachi, 1994). In 2008, there 12 confirmed and 11 suspected finds of the beetle in private gardens around the UK following the sale of thousands of *Acer palmatum* by mail order (Eyre et al, 2010).

Agrilus planipennis, emerald ash borer

This species was chosen because it has a high impact factor. The native range of emerald ash borer (EAB) is the far east of Asia, including parts of Russia, China, Japan, Korea, Mongolia and Taiwan. It is currently absent from the EU, but it has been included in this project because introduced populations in the USA have caused massive damage. EAB is thought to have killed more than 30 million trees in North America. In 2007, there was an official report of EAB in the Moscow area and now the pest is known to be present 230km east of the city. In the USA it attacks a range of ash species including, European ash, the ash tree native to the UK, *Fraxinus excelsior*. The larvae feed on the phloem, cambium and shallow sapwood beneath the bark. It is very difficult to detect and control the pest until it is already well established. The lifecycle is usually one year in the USA, but can be two years. Trees in urban and forested areas start to die within 1-4 years of first infestation by EAB. In Michigan, the pest and symptoms of the pest have been noted to spread at between 6-10km per year in the USA (Sargent *et al.*, 2010). EAB are cold tolerant and winter cold is unlikely to restrict their distribution in the UK. *A. planipennis* adults favour sunlight but are active in the canopy under strong sunlight and high temperature conditions (> 25° C) (Wang et al, 2010). Summer warmth is likely to be a limiting factor for *A. planipennis* in the UK. Larger patch areas and a greater number of patches of broadleaved woodland including ash trees would provide opportunities for greater populations of *A. planipennis* in the UK,. Increasing the quality of patches would be unlikely to have an impact.

Corythucha arcuata, Oak Lace Bug

This species was chosen because it is capable of establishing in the UK under current climatic conditions and is capable of surviving on UK native species. The Oak Lace Bug is native to the USA and is not yet present in the UK but is spreading rapidly elsewhere and has been reported from northern Italy, southern Switzerland and Turkey (Anderson, 2007). Its primary hosts are several species of oak tree and American Chestnut (*Castanea dentata* (americana)). The Oak Lace Bug is also occasionally found on found on *Acer* (maple), *Malus* (apple) and *Rosa* (roses) (Barnardinelli, 2006). In Italy *C. arcuata* has been found on *Quercus robur* (common, or English oak), *Q. petraea* (sessile oak) and *Q. pubescens* (downy oak) (Bernardinelli and Zandigiacomo, 2000). *Q. robur* and *Q. petraea* are native to England. Based on the spread so far of *C. arcuata* there is a risk of this species entering the UK by natural spread at some point in the future. The Oak Lace Bug is much more readily transported by human activity (movement of timber and living trees) than by natural spread, which is slow. The adults overwinter under the loose bark scales of oak tree trunks and larger branches (Connell and Beacher, 1947). Based on the geographical distribution in the USA, this species would be capable of establishing and overwintering in the UK. At the northern edge of its range, the Oak Lace Bug has two and a partial third generation per year (Connell and Beacher, 1947). At the southern edge of its range this species can complete three generations a year and have a partial fourth (Bernardinelli, 2000). The Oak Lace Bug spreads slowly by natural processes but its movement is increased by road traffic (Bernardinelli, 2000). This species is found in natural woodland, amenity park land and gardens. This species would benefit from larger areas of habitat which contain oak trees. It would colonise more oak trees if they were present in a larger patch of habitat. An increase in habitat quality would not affect this species unless that increase leads to the presence of more of its host tree. This species would benefit from more numerous patches of its preferred habitat.

Cyphostethus tristriatus, Juniper Shieldbug

This species was chosen because it is spreading through threatened juniper habitat in the UK. This species has been included because it is present in juniper habitat but is thought to be currently expanding its range and host plants. The juniper shieldbug overwinters as an adult and emerges and mates in early spring. Its traditional host is juniper (*Juniperus communis*). The larvae feed on the berries and new adults begin to appear at the end of August. The juniper shieldbug was present in England prior to 1951 but was then not recorded in a survey of juniper-living fauna in Surrey by 1968. This disappearance may have been due to degradation in the juniper habitat. Older patches of juniper, containing older plants, were found to support a more varied insect fauna, including the juniper shieldbug (Ward and Lakhani, 1977). The Juniper Shieldbug was previously confined to southern juniper woodlands, although it has recently been discovered on stands of juniper in northern England and Scotland. It has also been found on Lawson's Cypress (*Chamaecyparis lawsoniana*). It is now common across southern and central England due to the widespread planting of juniper and cypress in gardens. It is probably expanding its range (British Bugs). This species is in a position to take advantage of bigger areas of habitat in which its host plant, juniper, is present. Increased quality of habitat would not affect this species. The factor affecting its presence is the presence of juniper. This species would benefit from more numerous patches of its preferred habitat. It is capable of colonising new patches of juniper-containing habitat,

Gelechia senticetella, cypress groundling, also known as the southern juniper moth

This species was chosen because it has a moderate impact factor and is spreading through threatened juniper habitat in the UK. This moth was recorded in the UK for the first time in 1988 in a light trap in Essex and is now reported to be established in southeast England (Agassiz, 1989). The caterpillars of *Gelechia senticetella* feed on various species of juniper, including *Juniperus communis* (which is native to the UK) as well as cypress (*Chaemaecyparis* spp., *Cupressus* spp). *J. communis* is currently threatened by a pathogen, *Phytophthora austocedrae* which has been confirmed at some moorland sites in England. *G. senticetella* has been recorded from Russia, Ukraine, Bulgaria, Slovakia, Spain and Belgium. Between 1998 and 2002 the moth was found to be spreading in juniper forests along the Black sea coast line in the Kasnodar region of southern Russia which led to research in appropriate control measures (Gninenko, 2002). In addition, damage to *J. excelsa* in Bulagaria has led to research into effective chemical and biological control methods (Mirchev, 2001; Georgiev, 2000). This species has been selected because its distribution in the UK is restricted and it is a potential threat to native junipers. The cypress groundling will increase in population if bigger areas of habitat in which its host plant, juniper, is present are created. Increased quality of habitat would not affect this species. The factor affecting its presence is the presence of juniper. This species would benefit from more numerous patches of its preferred habitat. It is capable of quickly colonising new patches of juniper-containing habitat.

Lasius neglectus, an invasive garden ant

This species was chosen because it has a moderate impact factor and is not yet widely distributed in the UK. *Lasius neglectus* was first reported as being present in England in 2009 when it was found at the National Trust's Hidcote Manor in Gloucestershire (Fox, 2010). The ant is polygynous (each nest has many queens) and adjacent nests do not compete with each other. Once established it can form super colonies of interconnecting nests containing many queens. It is thought to originate in the Asian part of Turkey, but since being discovered in Budapest in 1974 it has been found in 19 European countries including France, Germany, Belgium and the

Netherlands. Evidence from Gloucestershire and from other parts of Europe show that it can exclude or severely reduce the populations of other species of ants. Efforts are being made to eradicate the ant from buildings at Hidcote Manor, but not from the gardens. It is possible that it may have spread in association with plants sold at the site. This species has been selected for this study because it has only been reported from one site in the UK and could have a significant impact on native ant species. Natural spread is believed to take place when new nests are built and this takes place over between a few and 89m a year (Esplander et al, 2007). This species lives in many different habitat types including woodland, grassland, dry dwarf shrub heath, cultivated and disturbed land, hedges and built-up areas. Therefore it is unlikely that any changes in the size or quality of particular habitat types will affect the spread of this species. It will simply colonise any new habitat type it encounters.

Lymantria dispar, Gypsy Moth

This species was chosen because it has a high impact factor and is spreading in the UK. Gypsy moth is native to Europe and Asia and within this range outbreaks sometimes occur. Outbreaks can result in severe defoliation and reduce growth, but tree mortality is occasional. Gypsy moths have a very wide host range of trees, including *Acer*, *Betula*, *Prunus*, *Carpinus, Fagus, Fraxinus, Larix, Malus, Picea, Pinus, Populus and Salix,* but are most damaging to *Quercus* (CPC, 2013). European gypsy moth was introduced into Massachusetts in the 1860s and it is now one of the most damaging forestry pests in North America. Gypsy moth populations are typically eruptive in North America; in any forest stand densities may fluctuate from near 1 egg mass per ha to over 1,000 per ha (US Forest Service, 2013). There are believed to be European and Asian strains of the gypsy moth. Females of the European strain are flightless or poor flyers and the primary means of dispersal is the wind dispersal of young larvae which can carry them ‘several miles’ (Waring *et al.* 2003), females of the Asian strains are capable of flight and this is likely to be the primary dispersal phase. However, the degree to which these strains are geographically separated and their potential for hybridisation is not clear. In Michigan, between 1985 and 1994 gypsy moth was reported to disperse an average distance of 15.8km per year. Passive dissemination, by the movement of egg masses on motor vehicles is considered to be the most common form of spread of the European strain in North America (Cannon et al, 2003). Gypsy moth was well established in the fens of Cambridgeshire and Norfolk in the first half of the 19th century, but this population, which was probably a distinct sub-species, was last recorded in 1907. The decline in this area was thought to be the result of the clearing and draining of the land. In 1995 an outbreak of gypsy moth was discovered in South Woodford in North-east London. This population has since been monitored by Defra and the Forestry Commission. It remained at very low levels until 2003 (Cannon, et al, 2003) but has since been found more widely. Larger patch areas and a greater number of patches of oak woodland or woodland containing oak would provide opportunities for greater populations of *L. dispar* in the UK, Increasing the quality of patches would be unlikely to have an impact.

Monochamus sartor, Sawyer Beetle

This species was chosen because it is only climatic factors which have prevented it from establishing in the UK and if the climate changes it may well establish here. The native range of the sawyer beetle is across Europe from eastern France to the Ukraine. The sawyer beetle has been intercepted entering England on numerous occasions, mostly in association with imported wood such as wooden packaging or wood products such as furniture. It prefers a mountainous continental climate and its main host is the pine *Picea abies*. Minor hosts include *Pinus sylvestris, P. cembra, P. mugo* and *Abies alba*. Mated females lay their eggs singly in small holes in the bark of a tree. The larvae feed under the bark for a month and then burrow into the wood and construct a pupal chamber from which adults will emerge. There is generally one generation per year, but there may only be one generation every two years. The Sawyer Beetle is not considered an aggressive pest but there is the possibility that it could vector the pinewood nematode, *Bursaphelenchus xylophilus.* The sawyer beetle has never established in the UK. The reason for this probably relates to its climatic and habitat requirements. It is a central and eastern European species adapted to a continental climate and is found mainly in mountainous regions (Anderson, 2009). The climate and habitats available in the UK are different to those in its native area, especially at places in the UK where it has been found or intercepted. *Monochamus* adult flight is generally quite short (tens to hundreds of metres) and this reduces its ability to disperse naturally, although longer flights up to 3km have been recorded (Evans *et al.*, 1996). There is nothing in the literature to suggest that this species would not be in a position to take advantage of bigger areas of coniferous forest habitat if the climatic conditions were favourable. Increased quality of habitat is not an issue for this species – as long as its coniferous host species are present, it will infest them. This species would benefit from more numerous patches of its preferred habitat. It would infest new sites if the climatic conditions were favourable.

Thaumetopoea processionea, oak processionary moth

This species was chosen because it has a high impact factor and is spreading in the UK. Oak processionary moth (OPM) is native to central and southern Europe. Its distribution spread northwards during the 20th century and is now firmly established in Belgium and the Netherlands. Oak trees (*Quercus* spp.) are the main hosts, but other broadleaved trees such as hornbeam (*Carpinus* spp.), hazel (*Corylus* spp.), beech (*Fagus* spp.), sweet chestnut (*Castanea* spp.) and birch (*Betula* spp.) have also been attacked by this pest, mainly when they are grown next to severely defoliated oaks (FAO). OPM can cause significant and occasionally complete defoliation of oak trees; it is also a human health pest because the hairs of the caterpillars can cause severe allergic reactions. In mainland Britain there were 19 records of OPM as a migrant in the last two decades of the 20th century and in 1984 it became established in Jersey (Waring *et al.*, 2003). In 2006, OPM was found on oak trees in south-west London, since then it has become established in large parts of west London and there are on-going outbreaks in Bromley, Croydon and Pangbourne in West Berkshire. It was found in Leeds and Sheffield in 2009, but is not thought to have become established there. It is considered likely that OPM could breed in much of England and Wales. Adult moths have been found in southern Sweden, but there is no evidence that it has successfully bred there (Evans, 2007). Larger patch areas and a greater number of patches of oak woodland or woodland containing oak would provide opportunities for greater populations of *L. dispar* in the UK, Increasing the quality of patches would be unlikely to have an impact.

9.6. Mammals

The following species (Table B7) were selected from the NNSS list of invasive mammals using the methodology described in Sections 3.1 and 3.2. In addition species specific parameters were sourced from major encyclopaedias (Harris & Yalden 2008).

Table B7. Impact scores and invasion stage of selected known invasive non-native mammal species evaluated using the Belgian ISEIA environmental risk assessment scheme.

Scientific name	Common name	Impact category	Invasion Stage
Myocastor coypus	Coypu	A	Extinct
Ondatra zibethicus	Muskrat	A	Extinct
Pecari tajacu	Collared Peccary	A	Captivity
Alopex lagopus	Arctic Fox	A	Captivity
Felis (Prionailurus) bengalensis	Leopard Cat	A	Captivity
Hydrochaeris hydrochaeris	Capybara	A	Captivity
Nyctereutes procyonoides	Raccoon Dog	A	Captivity
Procyon lotor	Raccoon	A	Captivity
Pudu puda	Southern Pudu	B	Captivity
Hydropotes inermis	Chinese Water Deer	B	Localized, established
Mephitis mephitis	Striped Skunk	B	captivity
Nasua nasua	Coati	B	captivity
Cynomys ludovicianus	Prairie dog	B	Localized

Species were selected if they scored highly and were in category A of the Belgian scheme of risk assessment as interpreted by UK studies (Hill *et al.* 2009; Parrott *et al.* 2009). Species were also selected if they had habitat specificity and were limited through it, such as Chinese water Deer, or were prevalent as potential novel non-natives soon to become a potential problem in the UK (parrot). Similarly species were not selected if they are already widespread in the UK such as the mink *Neovison vison*.

With the exception of Chinese water Deer none of the species selected are present in the UK outside of captivity and two species, the coypu and Muskrat have been eradicated in the last century. The remainder of the species had been identified by Parrott, Roy & Fletcher (2008) as potential invaders in the future.

9.6.1. Species descriptions

Information from the following section has been obtained from the following sources(Corbett & Hill 1992; Harris & Yalden 2008), (Mitchell-Jones 1999; Nowak 1999).

Coypu *Myocastor coypu*

The coypu is native to some tropical and temperate South America. However it has been introduced to North America large sections of Europe Asia and Africa for the fur trade. The species lives primarily in self-excavated burrows adjacent to rivers and lakes. It is primarily vegetarian and eats aquatic plants and roots, but is also an agricultural pest and will raid crops.

In its native range, in the wild, the species has a lifespan of up to 3 years and can reach sexual maturity within the first six months of life. Litter sizes can vary widely from 1 to 13 offspring. The species has no fixed breeding season but instead can breed throughout the year, with up to 3 litters per year.

This species was selected because although it was once widespread in Norfolk, and was successfully eradicated, it has become invasive over much of mainland Europe and could easily re-enter the UK through trade movements or deliberate release.

Muskrat *Ondatra zibethicus*

This species is native to North America, Northern Mexico and Canada, but is widespread as an introduced species in Europe Asia and South America. Given the broad latitudes covered by its native range it can survive in a wide range of climatic conditions and also habitat types. The species is broadly herbivorous, although they will also eat crustaceans amphibians and even small aquatic reptiles and birds. They are mostly nocturnal although in some parts of the world they have become crepuscular.

The species is able to withstand extreme temperatures through sheltering in burrows or even beaver like lodges that they create. The species has no fixed breeding season, but instead can be up to 3 times a year, each time raising a litter from 6 to 8 young. Breeding can occur within the first six years of an animal's life.

This species has been selected because although it was widespread throughout the UK, and then eradicated in the 1930s, it is still widespread throughout continental Europe and could easily be reintroduced.

Raccoon *Procyon Lotor*

Native to North, Central and South America, the raccoon is a nocturnal species that has an omnivorous diet of birds, eggs, fish, amphibians, nuts and berries. It will also scavenge. It prefers woodlands close to water. The species can withstand a broad range of temperatures, and in extreme cold can become dormant. It is a good climber, and often dens in holes in trees. It is fairly sedentary, though can show occasional dispersal covering large distances of up to 20km. Mating usually occurs between January and March resulting in 2-5 young 2 months later.

This species was selected because it has become a widespread invasive in continental Europe (Long) and there have been approximately 13 reports of the species in Britain (Long 2003). Therefore the species could become a devastating invader in England.

Raccoon dog *Nyctereuctes procyonoides*

Native to Eastern Asia and Japan, the raccoon dog is a nocturnal species that prefers wooded hills. It either excavates dens or uses natural cavities, and is able to hibernate in winter. It has a plastic social systems, living singly, in pairs or in small groups. Highly omnivorous, it eats a wide range of small mammals, reptiles, birds and eggs, fish, amphibians, invertebrates, acorns, nuts, fruits, berries, grain and roots. Mating occurs February to April and results in 6-16 pups after 60-70 days.

This species was selected because though generally considered absent from the UK, it is highly invasive in continental Europe. It is also become more prevalent in the pet trade, and one was sighted in Berkshire in 2005.

Striped skunk *Mephitis mephitis*

The skunk can exploit a broad range of different habitats in its native range of North and Central America, including forest, woods, plains, desert, agricultural land, river valleys, and suburban areas, using dens excavated by other animals. Although it does not hibernate, it can become dormant during cold periods. It is omnivorous, and crepuscular, feeding on a range of small vertebrates, bird eggs, carrion, insects, insect larvae, fruits and berries. It should be noted that skunks are no longer classed as a must delete but are in a family of their own.

Breeding takes place between February and April, outside of which the species is solitary. Four to eight young are born up to 2 months later.

There have been several attempts to introduce skunks into Europe, but these have generally not been successful and as a result the species is not as widespread as raccoons or raccoon dogs. However it is becoming more prevalent in the pet trade and could become invasive in the future. During 2001-2003, the RSPCA responded to a total of 25 incidents involving skunks.

Coati *Nasua nasua*

The coati has a native distribution ranging from the southern United States and over much of South America. It is primarily a woodland animal and is diurnal. An adept climber it is an omnivore taking a wide range of prey both at ground level and the tree canopy. Highly sociable it lives in groups although males are often solitary outside of the breeding season.

Coatis generally agreed between January and March although breeding is highly flexible depending on food availability Denning occurs in tree holes or within boulder piles, and after a 2 1/2 months gestation period a litter of between 3 to 7 young are born.

This species was selected because although coatis have not been found breeding in the wild in England. However a large number are kept in captivity and there are frequent escapes particularly in the North West of England during 2003 and 2004. These animals have generally been caught back successfully.

Black-tailed prairie dog *Cynomys ludovicianus*

Black-tailed prairie dogs have a native distribution throughout North America from Texas to the Canadian boundary. It is primarily a grassland prairie animal. It is highly colonial and excavated zone burrows to form an intricate network of interconnected subterranean tunnels and passages. The species is diurnal and will eat grass, herbs, roots, leaves and stems. Occasionally insects are also consumed. Due to its borrowing habits the species can withstand a broad variation in climatic conditions by sheltering.

Prairie dogs can breed within their first year if food is abundant. Breeding occurs from February to April and after 34 days of gestation 3 to 6 young are born. The species breeds only once a year.

In England prairie dog colonies have existed in Cambridge, Cornwall and the Isle of Wight. All have originated from nearby captive companies. These have died out, although a more recent colony in Somerset has survived into the late 1990s.

Collared peccary *Pecari tajacu*

The peccary is a New World pig that is native to the south-western USA, Central and South America. Like all pigs the species is omnivorous with a diet consisting of a broad range of plant species and plant parts, together with small vertebrates, invertebrates and carrion. The species is diurnal and highly sociable and lives in large herds or sounders, sometimes consisting of up to 20 animals. It can survive in a broad range of habitat types ranging from deserts and shrub land through to grasslands and forests. They also adapt incredibly well to human modified habitats and are found in South American cities.

Breeding occurs within the first year and after a gestation period of five months to young are born.

The species does not have a history of being invasive, although it has been introduced successfully to Cuba. The species was selected however because there are several populations in captivity in zoological parks e.g. Marwell, In Britain peccaries have been recorded living in the wild (n=4) for unknown periods of time. Also in mainland Europe several New World game species have been introduced into the wild for hunting purposes. This species was therefore chosen as an indicator species of this trend.

Arctic fox *Alopex lagopus*

This species has a natural distribution in the Arctic Circle and is well adapted to cold environments. It has a broad diet and will eat carrion small vertebrates invertebrates as well as seabirds and their eggs. The species forms monogamous pairs during the breeding season that lasts from September to May. After a gestation period of approximately 50 days, litters of anything up to 25 may be born. Arctic foxes have been highly prevalent in fur farms throughout Europe and this is the reason for their selection. Although there has never been a long-term population living in the wild in Britain there have been a number of escaped individuals (27) that are subsequently been recaptured.

Leopard cat *Felis (Prionailurus) bengalensis*

This species is native to South and East Asia with several subspecies endemic to some of the islands of the Philippines, Indonesia, Japan and China. It is highly adaptable, and can live in a broad range of habitat types and eat a range of prey species such as small vertebrates and invertebrates. The species is also heavily persecuted in the wild for the fur trade. It is however widespread and common.

Like many cat species, leopard cats do not have a fixed breeding season but instead can breed throughout the year. However in more northern latitudes breeding is focused around March April and May when food is more plentiful. After 60 to 70 days up to 4 kittens are born and in part of their range sexual maturity is reached within the first year.

Although leopard cats themselves are not common in the pet trade, there have been four recorded escapes in Britain. Also, hybrids of leopard cats and domestic cats, commonly known as Bengal cats are becoming more prevalent, hence the selection of the species for this study.

Capybara *Hydrochaeris hydroc*haeris

This species is related to coypus and is native to savannas and dense forests close to water in large regions of South America. It is highly sociable and lives in groups of up to 20 individuals. It is prized for its meat and hide. Like the coypu it eats aquatic vegetation but will also raid crops. The species is not strictly diurnal or nocturnal but can vary its behaviour depending on food availability and levels of disturbance.

Breeding occurs within the first year. After a gestation period of up to 150 days a litter size ranging from 1-8 is produced. There is no strict breeding season, although breeding can refocus to periods when food is more plentiful.

A number of introduced populations occur around the world as a result of escapes from captivity, such as in Florida and California. In Britain 19 individuals have been recorded living in the wild for anything up to 3 months

Chinese water deer *Hydropotes inermis*

The Chinese water Deer is native to eastern China and Korea. It is a habitat specialist and prefers to live near rivers and lakes with reeds and tall grasses. In Britain it also likes woodland habitats and open grassland. The species is mostly solitary and the species can be active during day or night.

Making mainly occurs during December and after a gestation period of up to 210 days females usually give birth to 2 young. However in parts of China up to 7 foetuses have been recorded. Sexual maturity is reached within the first year of life.

In Britain the species having escaped from zoological collections in the 1950s, has well-established populations in Bedfordshire Cambridgeshire Norfolk and Suffolk. The UK population is currently estimated at 1500 animals. It has also been introduced to France.

Southern pudu *Pudu puda*

Pudus are the smallest deer in the world, and are native to South America. The southern species is native to Chile and Argentina. In its native range the species is classified as Vulnerable by the IUCN. Primarily a forest species, the Pudu can inhabit a broad range of forest habitat types, such as montane brush, and dense woodland. The species is solitary and highly elusive. Behaviour patterns are not fixed and the species can be both diurnal and nocturnal depending on levels of disturbance. It has a broad vegetarian diet and will eat leaves, sprouts, herbs, flowers and fruit.

In the southern hemisphere mating occurs from April to May and after a gestation period of up to 220 days a single young is born. Pudu are sexually mature within their first year.

The species is not common in captivity but has been included in this selection because it is a small and elusive species that will act as a good proxy for others of this behavioural type. Also, because of its uniquely small size, it may enter the pet trade and also become more prevalent in zoological collections (www.brc.ac.uk).

9.7. Herptiles

The species shown in Table B8 were chosen from the NNSS website. In addition species specific parameters were sourced from major encyclopaedias (Behler & Wayne King 1979, Halliday & Adler 2002).

Table B8. Impact scores and invasion stage of selected known invasive non-native herptile species evaluated using the Belgian ISEIA environmental risk assessment scheme.

Scientific name	Common name	Impact category	Invasion Stage
Pelophylax ridibundus	Marsh frog	A	localized
Xenopus laevis	African Clawed Toad	A	localized
Macrochelys temminckii	Alligator Snapping Turtle	A	captive
Lithobates catesbeianus	American Bullfrog	A	localized
Python molurus subsp. bivittatus	Burmese Python	A	localized
Bufo marinus	Cane Toad	A	absent
Chelydra serpentina	Common Snapping Turtle	A	captive
Trachemys scripta	Red-eared Terrapin	A	widespread
Mesotriton alpestris	Alpine Newt	B	localized
Podarcis muralis	Wall Lizard	C	localized

Species were selected if they scored highly and were in category A of the Belgian scheme of risk assessment as interpreted by UK studies (Hill *et al.* 2009; Parrott *et al.* 2009). With the exception of Alpine newts and wall lizards, which were selected because of their susceptibility to habitat alteration and climate change (Trent Garner, Evolution & Molecular Ecology Theme Leader, Zoological Society of London, pers. comm.). All of the species selected are of the highest categories and scores as interpreted by previous schemes.

9.7.1. Species descriptions

Information for this section has been taken from the following sources encyclopaedias (Behler & Wayne King 1979, Halliday & Adler 2002) together with Internet sources (www.surrey-arg.org.uk and www.iucnredlist.org).

Marsh frog *Pelophylax ridibundus*

The Marsh frog is the largest European frog and like most frogs is aquatic and can live in a broad range of freshwater habitats including large lakes and rivers. As a carnivore it will eat aquatic insects and insect larvae, molluscs and worms. They will also consume smaller amphibians, small rodents and fish when they become large. It has a native range across most of continental Europe and parts of Asia. It is not well adapted to colder climates.

The species can hibernate from November through to March and begins breeding from May to June with several hundred eggs being laid mostly in June. Tadpoles become adults in September.

The species was first introduced to Britain in 1935 and a population still exists in Kent and East Sussex. It also occurs in and around London. There is also a population in Humberside.

African clawed toad *Xenopus laevis*

The African clawed toad is native to sub Saharan and West Africa. The species however is highly prevalent both in the pet trade and as a laboratory species and is therefore widespread in captivity. It is highly invasive around the world. The species is predatory and an ambush predator and lives mostly in still waters. It can also tolerate brackish and estuarine waters, where it feeds on aquatic invertebrates and small vertebrates. The species has been observed undertaking overland migration of several kilometres.

In part of its range to species can breed year-round and can lay between 1020 7000 eggs per breeding event. Sexual maturity can be achieved within one year depending on temperature. In captivity the species can live up to 20 years.

This species is known to have localised populations in Britain were breeding has occurred successfully. Colonies exist in the Isle of Wight and in South Wales. Through climate change breeding will only become more successful in warmer weather and thus this species has been selected for the study.

Alligator snapping turtle *Macrochelys temminckii*

The alligator snapping turtle is one of the largest freshwater turtle species in the world. It is native to the eastern and southern states of America and spend most of their time in water. The species is highly carnivorous and as an ambush predator will lay in wait for anything that will fit into its mouth, including a large number of terrestrial species that enter or approach the water to drink. The species will also take carrion.

The species become sexually mature at 12 years of age and breeding occurs from March to May. The female will lay up to 50 eggs after two month gestation period. Eggs hatch after 100 4250 days.

The species can live up to 120 years and therefore despite the long time to sexual maturation, it is believed that the species could persist at low densities were long periods of time in the wild. The species is highly prevalent and available in the pet trade. Although populations of the species have not been recorded in the wild, individuals have been encountered occasionally, probably due to release by reptile keepers (Driver 2010). This, together with their lifespan is the reason for their selection as part of the study.

American bullfrog *Lithobates catesbeianus*

This species is native to North America but is invasive around the world due to its prevalence in the pet trade. It is highly invasive in Europe with populations found particularly in northern Italy. It can live in most water bodies but prefers calm and slow-moving water. The species can reach large sizes and is highly predatory, feeding on a large range of native amphibians but also small birds and snakes and invertebrates.

Breeding takes place in June and July and up to 20,000 eggs are laid. Tadpoles take up to 2 years to mature to adult form. As tadpoles the species can move large distances.

This species has been selected because it is highly prevalent in the pet trade, and the species in the wild is more likely to breed if appropriately warm temperatures are achieved.

Burmese python *Python molurus* subsp. *bivittatus*

The Burmese Python is a subspecies of *Python molurus*. It is native to South and Southeast Asia. It is highly prevalent in the pet trade and as a result has introduced populations in Puerto Rico and the United States, although occasional individuals have been found free living throughout Europe. It is nocturnal and predatory and is able to live in a broad range of habitats ranging from forests through to urban and suburban gardens and buildings. In colder regions of its range to species can survive low temperatures by occupying underground burrows.

Sexual maturity is reached by the fourth year of age, and up to 100 eggs may be laid after a 90 day gestation period. The species can live for up to 25 years in the wild and breeding can occur all year round that is mostly targeted to the warmer seasons of the local environment.

The species is not found in the wild in Britain, but is highly prevalent in the pet trade and therefore escapes are likely. Although the British climate is unlikely to lead to breeding, long-term climate change may alter this and therefore the species had been included in the selection.

Cane toad *Bufo marinus*

The cane toad is recognised as one of the world's worst amphibian invasive species. The species has been introduced around the world as a biological pest control agent to control invertebrate pests in sugarcane fields. It also reacts to chemical changes in the environment rapidly and therefore has been used in the past as a human pregnancy test. The species is highly adaptable and is able to exploit a broad range of habitat types, both aquatic and terrestrial, and exploits a large number of man-made structures in order to complete its life-cycle.(Florance *et al.* 2011). It is highly predatory and will eat any terrestrial animal species that it can fit into its mouth.

In the northern hemisphere, cane toad breed between April and September and female cane toads produce up to 2 clutches of between 8000 to 35,000 eggs each. These hatch within 72 hours, and tadpoles achieve adult form within seven weeks. Breeding can take place within the first year.

The species is absent from Britain and much of Europe, though it is highly prevalent in the pet trade and has therefore been included in this study.

Common snapping turtle *Chelydra serpentina*

The common snapping turtle is native from Canada through to Mexico. It is therefore able to survive and breed in cold climates, able to survive even under ice. The species prefers shallow ponds lakes, streams and brackish estuaries. It is nocturnal and omnivorous and will eat both plant and animal material as well as scavenging on carrion. Unlike the alligator snapping turtle, it is also able to travel large distances overland.

The species can live for up to 40 years. Mating occurs from April to November, and the female is able to store sperm. Females lay a single clutch of up to 40 eggs a year.

The species, like the red eared Terrapin and alligator snapping turtle is highly prevalent the pet trade and invasive populations are becoming established in part of mainland Europe, such as in Italy. This together with its long lifespan, and ability to survive and breeding cold climates cold climates suggests that this species has the potential to become invasive in Britain, hence its inclusion in the study.

Red eared terrapin *Trachemys scripta*

Additional information for this species has been sourced from Kikillus *et al.* (2010) and the IUCN (www.issg.org). This species is extremely widespread both globally and within Britain due to its prevalence in the pet trade. The red eared Terrapin is native to North America, where it prefers large deep still water bodies. In it introduced range the species is found in a wide range of parkland ponds, canals and rivers. The species is omnivorous and will eat both plant and animal material. It will also take Carrion. The species is mostly carnivorous when it is younger. In cold temperatures they can become less active though they do not truly hibernate.

Whether species is able to read mating takes place from March to July, after which the female will lay up to 5 clutches each consisting of between two and 30 eggs. Eggs hatch up to 90 days later and sexual maturity is reached in the second or third year depending on local temperatures. Breeding events in Britain however have not been formally recorded due to cold climatic conditions.

This species has been selected because of its prevalence in the pet trade, its widespread distribution throughout Britain and the potential for the species to breed successfully should the climate become warmer.

Alpine newt *Mesotriton alpestris*

Alpine units are native to continental Europe where they prefer upland areas. This species is able to exist in a broad range of habitat types including moist forests and forest edges, grassland and heathland.

The species is dormant in the winter, and after a breeding period from March to May tadpoles achieve adult form in August and September. Up to 200 to 300 eggs are laid.

In Britain it now exists in both upland and lowland regions, and populations have been recorded in Birmingham, Brighton, London, Cornwall, Shropshire and Sunderland.

Wall lizards Podarcis muralis
The wall lizard is common to continental Europe where the species can exist in a broad range of habitat types ranging from parkland is to woodland and scrub, particularly those associated with rocks and boulders. It is a fast moving predator and its diet consists of a broad range of invertebrates. The species will also eat fruits and berries.

The species is active from November to February when it hibernates, and breeding occurs from April through to August. Several clutches are laid by a female over the course of the breeding season, with each clutch consisting of between three and 17 eggs. Eggs hatch after a month and sexual maturity is reached by three years of age.

In Britain the species has localised introduced populations, mostly associated with man-made habitats such as stone walls, buildings and warmer south facing facades. Colonies exist on the Isle of Wight and in Dorset. There is some dispute as to whether the species is native to Jersey in the Channel Islands. The species will benefit from a warming of the climate and this is why it has been selected as part of the study.

10. Appendix C: Climate degraded habitat

Table C1. Descriptions of degraded habitat used to assess the impacts of climate change for each of the selected species. Table from Mitchell *et al.* 2007 as provided by Natural England as part of this project.

Habitat	Description
Coastal Grazing Marsh	**EBS notes -** Inundation due to sea level rise, increased flooding in winter, more variable water levels leading to changes in species, changes in species distribution due to increased winter temps, increased drying in hotter summers.
Coastal Saltmarsh	**EBS notes -** Inundation due to sea level rise, coastal squeeze, changes in erosion rates, changes in species distribution due to increased winter temps, increased drying in hotter drier summers. MONARCH - mixed results for species climate space modelling.
Lowland Raised Bog	**EBS notes -** Sensitive to changes in temp and precip. Increased CO_2 leads to changes in community composition. Rain fed wetlands more sensitive than ground fed (but it is more complex than this). Changes in inundation leads to changes in floodplain wetlands. Floodplain wetlands dependant on marked flow peak and snow melt are especially sensitive. Seasonal changes in precip will impact. Changes in temp affect species composition. Increased pollution risk. Accelerated transition to terrestrial habitats. Changes in phenology. Increased fire and flood risk. Release of emissions due to increased temps and drying (severe drought), changes in hydrology and changes in species diversity. Peat loss and contraction of bogs in the south and east. Drier bogs liable to succession. Western mires possibly more secure. Some changes in plant and soil fauna species composition. Accelerated transition to terrestrial habitats. Changes in phenology. Increased fire risk.
Maritime Cliff and Slope	**EBS notes -** Increased erosion, changes in species due to defences, increase invasive species.
Montane (new national habitat)	**EBS notes -** Montane habitats - probably the most vulnerable habitats, species have nowhere to go, loss of montane species from Pennines, Lake District and North York Moors by 2050. **NCA template notes** - increased winter temperature = Loss of species e.g. dwarf willow (Salix herbacea) and the trailing azalea (Loiseleuria procumbens) predicted to disappear from upland areas such as the by 2050, Competition from replacement by faster growing species of grass species, Fragments of habitat and individual arctic-alpine species will become restricted to smaller and smaller pockets – i.e. the highest and shadiest ground (north-facing slopes), Damage to the plant's root growth, Larger fruit - improved reproductive success, loss of associated species e.g. 'montane' lichens. MONARCH - montane heath = all species lose, most vuln habitat.
Saline Lagoons	**EBS notes -** Inundation due to sea level rise and storms, increased freshwater inundation due to flooding events in winter, hyper salinity in summer.

Habitat	Description
Standing Water (freshwater)	**EBS notes -** Impacts on species numbers and communities, increased cyanobacterial dominance, increased disease, eutrophication, negative impact on clear water biodiversity, impacts of high water temps and high nutirent loads on phytoplankton, changes to breeding, development and emergence dates (e.g. toads, mayflies), surface temp increase, period of stagnation increase, ice cover decrease, changes in stratification and mixing, Changes in phenology. Increased densities of aquatic grazers, encroachment of marginal emergent vegetation, shallow waterbodies may become seasonally wet.
Upland Hay Meadows (new national habitat)	**EBS notes -** Mixed response, change in species composition but some studies say this will be due to management, some species projected to change range, others not, possible replacement of distinctive upland meadow with lowland type. CC suggested as a driver for species composition change in neutral grasslands (decreasing northern species, increasing southern species) (Briathwaite, Ellis and Preston 2006). Cairngorms National Park website - UHM one of their most sensitive habitats to climate change. Possibly greater influence of management practices. Changes in species composition are expected (Climate NE paper - Climate Change and Biodiversity in North East England - Mike Harley (AEA), Keith Buchanan (KBA), Pam Berry (ECI) and Nikki Hodgson (AEA)) - expect that change in temp and precip leads to this. MONARCH - changing species composition. R Jefferson pers comms - populations of certain boreal-montane species which are characteristic of the habitat such as Geranium sylvaticum, Troillius europaeus, various Alchemilla species etc. These are likely to be vulnerable to climate change and a study suggested that the first named could lose climate space under CC scenarios. It has also been in decline in UHM and one explanation has suggested longer growing seasons and warmer springs have put it at a competitive disadvantage.
Floodplain Grazing Marsh	**EBS notes -** Sensitive to changes in temp and precip. Increased CO_2 leads to changes in community composition. Rain fed wetlands more sensitive than ground fed (but it is more complex than this). Changes in inundation leads to changes in floodplain wetlands - species communities and composition/habitat type e.g. MG8 to MG5 due to summer drought. Floodplain wetlands dependant on marked flow peak and snow melt are especially sensitive. Seasonal changes in precip will impact. Changes in temp affect species composition. Increased pollution risk. Accelerated trasition to terrestrial habitats. Changes in phenology.Changes in overwinteringbird populations. Altered sediment input times, grasses increase due to wet winters, increased flooding leads to increased watterlogging, accumulation of CO_2 etc, altered thermal conditions, changes in soil structure, decreased soil inverts, changes in belowground communities e.g. fungi to bacteria.
Purple Moor Grass and Rush Pasture	**EBS notes -** Sensitive to changes in water table and flooding- very specific conditions required. Changes in rainfall will impact. Decline in condition and extent may result. Accelerated trasition to terrestrial habitats.
Coastal Vegetated Shingle	**NCA template notes** - Sea level rise - continued and potentially accelerating reduction in sediment supply to shingle beach systems, gravel beach systems are at risk of breaking down or are moving more rapidly landwards.

Habitat	**Description**
Lowland Meadows (wet)	**EBS notes -** Sensitive to changes in temp and precip. Increased CO_2 leads to changes in community composition. Rain fed wetlands more sensitive than ground fed (but it is more complex than this). Changes in inundation leads to changes in floodplain wetlands - species communities and composition/habitat type e.g. MG8 to MG5 due to summer drought. Floodplain wetlands dependant on marked flow peak and snow melt are especially sensitive. Seasonal changes in precip will impact. Changes in temp affect species composition. Increased pollution risk. Accelerated trasition to terrestrial habitats. Altered sediment input times, grasses increase due to wet winters, increased flooding leads to increased watterlogging, accumulation of CO_2 etc, altered thermal conditions, changes in soil structure, decreased soil inverts, changes in belowground communities e.g. fungi to bacteria.
Reedbeds	**NCA template notes -** Drying out in summer, accelerated trasition to terrestrial habitats, inundation at the coast, flooding elsewhere, Reduction in water quality due to less dilution of pollutants – can constrain ecosystem functioning. Changes in phenology.
Blanket Bog (new national habitat)	**EBS notes -** release of emissions due to increased temps and drying (severe drought), changes in hydrology and changes in species diversity. Peat loss and contraction of bogs in the south and east. Drier bogs liable to succession. Western mires possibly more secure. Some changes in plant and soil fauna species composition. Accelerated trasition to terrestrial habitats. Changes in phenology. Increased fire and flood risk. **NCA template notes -** Drought = Increase in risk of wildfire – increase intensity and frequency, Increase intensive of rainfall = Erosion of peat, increase in bare peat – erosion of surfaces, increased summer temps - Increased plant growth - species competitiveness alters, community compositional changes – e.g. heather might out-compete sphagnum. MONARCH - little change in most currently widespread species, but southern species gain and northern species lose, some elements of this habitat are sensitive to climate change.
Coastal Sand Dunes	**EBS notes -** Increased erosion due to sea level rise. MONARCH - some species gain others lose climate space.
Intertidal Mudflats	**EBS notes -** loss of 8-10,000ha of mudflat predicted by 2013 (UKBAP 2005), changes in species distribution, higher nutirent levels and lower oxygen content due to summer drought, lower salinity due to wetter winters, changes in erosion and become more sandy due to slr, influx of freshwater from increase flooding, morphological changes due to storm events.
Lowland Fen	**EBS notes -** release of emissions due to increased temps and drying (severe drought), changes in hydrology and changes in species diversity. Peat loss and contraction of bogs in the south and east. Drier bogs liable to succession. Western mires possibly more secure. Some changes in plant and soil fauna species composition. Wetlands info - Sensitive to changes in temp and precip. Increased CO_2 leads to changes in community composition. Rain fed wetlands more sensitive than ground fed (but it is more complex than this). Changes in inundation leads to changes in floodplain wetlands. Floodplain wetlands dependant on marked flow peak and snow melt are especially sensitive. Seasonal changes in precip will impact. Changes in temp affect species composition. Increased pollution risk. Accelerated trasition to terrestrial habitats. Changes in phenology. Increased fire and flood risk.

Habitat	**Description**
Heathland	Joint classification for both wet and dry heathland.
Rivers	**EBS notes -** Changes in water temperature leading to impacts on fish species e.g. Atlantic salmon, affecting rivers in the South and East of England most (River Medway predicted to be affected by 2020). Flooding - changes in frqeuancy will lead to changes in erosion, sediment transportation and depostion and therefore to changes in habitat structure, changes in peak flow timing. nearer the coast - alteration in estuarine ecotone and sediment pattern. Projections of river flow regimes vary, more severe on some but les severe on other rivers, no distinct regional pattern, localised impacts. Low flows - higher temps and reduced rainfall will lead to reduced flows, but unsure about the magnitude of impacts. Impacts can be - increased temp, reduce dissolved oxygen, increase light penetration, increase pollution concentration, impacts on species that require specific depth and flow velocity, reduction of available habitat in summer, changes in species communities but differential responses over diff regions/catchments etc. 3 main impacts on species - reduced oxygen concentrations, changes in quality, changes in availability. Changes in phenology. Changes in floodplain habitat type e.g. MG8 to MG5 due to summer drought.
Upland Heathland (new national habitat)	**EBS notes -** More diverse communities - increase in generalist species over typical upland species, likely to increase with greater climate change. Changes in microbial activity. Upland heath and montane habitats - probably the most vulnerable habitats, species have nowhere to go, loss of montane species from Pennines, Lake District and North York Moors by 2050. **NCA template notes -** increased above ground biomass, Litter fall tended to decrease - changes in soil chemistry with possible impacts on soil fauna, Changes in community composition, acceleration of spring growth, Lower plant replacement rate, Increased risk of wildfire, Increased spring leaching of nitrate in winter.
Limestone Pavements (new national habitat)	Altered species composition, changes to rainfall = impacts on geomorphalogical processes, temp changes = impacts on chemical processes, Climate change is likely to pose the greatest threat to tufas in Yorkshire, which are already often subject to erosion in many areas (MONARCH). Increased runoff and changes in water quality from pollution (JNCC report 450). Peak District Biodiv Action Plan - Climate change risks – Possibly favourable as extreme weather events remove topsoil and expose pavements. Otherwise low with risk to shift in species composition.
Lowland Meadows (Dry)	**EBS notes -** Sensitive to changes in temp and precip. Increased CO_2 leads to changes in community composition. Rain fed wetlands more sensitive than ground fed (but it is more complex than this). Changes in inundation leads to changes in floodplain wetlands - species communities and composition/habitat type e.g. MG8 to MG5 due to summer drought. Floodplain wetlands dependant on marked flow peak and snow melt are especially sensitive. Seasonal changes in precip will impact. Changes in temp affect species composition. Increased pollution risk. Accelerated trasition to terrestrial habitats. Altered sediment input times, grasses increase due to wet winters, increased flooding leads to increased watterlogging, accumulation of CO_2 etc, altered thermal conditions, changes in soil structure, decreased soil inverts, changes in belowground communities e.g. fungi to bacteria.

Habitat	**Description**
Deciduous Woodland	**EBS notes -** Issues comparing upland woodlands (ash and oak L), beech and yew woodlands M, lowland deciduous woodland L, wet woodland M - perhaps we should go with Low to avoid dominance of woodland. **Broadleaved, mixed and Yew woodland** (all the BAP poriority habitats in England) - temp rises = changes in species composition e.g. loss of drought sensitive species in SE and changes in dominant species, phenology e.g. leafing dates and loss of synchrony for predator prey, loss of drought sensitive species e.g. Beech, increased fire risk, wind throw (probably positive). Increased pest survival through winter and new pest species. **Lowland Beech and Yew -** beech sensitivty to drought, decline in SE, changes in composition and distribution, yew unlikely to be affected. MONARCH - Beech loses suitable climate space from southern and central England. **Upland Mixed Ash Woods -** minimal impact, species likely to persist where currently present but species composition of woodlands may change. **Upland Oak woods -** restricted distribution, species such as ferns, bryophytes and lichens (species of particular importance for this habitat) may be particularly sensitive to changes in temp and humidity. MONARCH - changes in species composition. **Wet Woodland -** possible impacts from changes in seasonal precipitation although may depend on local and regional factors.
Lowland Calcareous Grassland	**EBS notes -** Potential increase in climate space, although limited to chalk so constrained. Change in composition with lowland species spreading to upland communities at the expense of northern species and a reduction in grasses. Management has a greater impact than climate change. Older grasslands are more resilient. Affected by increase in winter temps - less species diversity, temp and rainfall impacts inverts. MONARCH - species modelled show little response to climate change.
Lowland Dry Acid Grassland	**EBS notes -** fairly resilient, potentially replacs wet heath in the south east, some loss of species due to drought but also some increase in less widespread species, possible impacts in upland areas from increased temps and growing season due to low nutrients, some changes in species composition. MONARCH - drought prone acid grassland thought to be a relatively sensitive habitat but species could gain space.
Upland Calcareous Grassland (new national habitat)	**EBS notes -** Potential increase in climate space, although limited to chalk so constrained. Change in composition with lowland species spreading to upland communities at the expense of northern species and a reduction in grasses. Management has a greater impact than climate change. Older grasslands are more resilient. Affected by increase in winter temps - less species diversity, temp and rainfall impacts inverts.

11. Appendix D: Invasion pathways and traits

Table D1. Summary of bird species status in England, invasion pathways and traits facilitating (•) or constraining (○) establishment and spread; plus overall assessment of the effect of landscape and climate warming on invasiveness.

		Invasion Stage				**Effect on Establishment/Spread**	
Species	**Status**	**Transport**	**Introduction**	**Establishment**	**Spread**	**Landscape**	**Climate Warming**
Eagle owl	–Local breeder –In captivity	–Pet trade/keepers/ –collections	–Escapes/ releases	• Varied habitat use • Wide diet • Long-lived o Low annual productivity	• Long distance natal dispersal • Varied habitat use o Sedentary	0 to -1	0
Indian house crow	–Absent	–Ship-assisted	–Ship assisted	• Wide diet • Very social • Human commensalism o Urban/semi-urban	• Human commensalism • Behavioural flexibility • Foraging flights up to 20km o Sedentary	0 to -1	1
Ring-necked parakeet	–Widespread breeder	–Pet trade/keepers/ –collections	–Natural spread –Escapes/ releases	• Wide diet • Human commensalism o Urban/semi-urban but spreading to agric.	• Human commensalism • Behavioural flexibility • Foraging flights up to 20km	0 to -1	1
Sacred ibis	–Absent in wild –In captivity	–Waterfowl collections	–Natural spread –Escapes/ releases	• Varied habitat use • Wide diet • Long-lived o Coast, wetlands	• Varied habitat use • Behavioural flexibility • Foraging flights > 30km • Can disperse long distances linked to water availability	0 to -1	0 to -1

		Invasion Stage				Effect on Establishment/Spread	
Species	**Status**	**Transport**	**Introduction**	**Establishment**	**Spread**	**Landscape**	**Climate Warming**
Bar-headed goose	–Local breeder –In captivity	–Waterfowl collections	–Natural spread –Escape/ releases	• Wide diet o Freshwater, wetlands	• Can disperse long distances linked to water availability	0 to -1	0 to -1
Black swan	–Local breeder –In captivity	–Waterfowl collections	–Natural spread –Escapes/ releases	• Wide diet o Freshwater, wetlands	• Can disperse long distances linked to water availability	0 to -1	0 to -1
Ruddy shelduck	–Local breeder –In captivity	–Waterfowl collections	–Natural spread –Escapes/ releases	• Wide diet • Less dependent on large water bodies	• Can disperse long distances linked to water availability	0 to -1	0 to -1
Egyptian goose	–Widespread breeder –In captivity	–Waterfowl collections	–Natural spread –Escapes/ releases	• Varied habitat • Wide diet • Less dependent on large water bodies	• Can disperse long distances linked to water availability	0 to -1	0 to -1
Red-vented bulbul	–Absent in wild –In captivity	–Pet trade/keepers/ –collections	–Escapes/ releases	• Varied habitat use • Wide diet	• Varied habitat use	0 to -1	0 to +1
Common mynah	–Absent in wild –In captivity	–Pet trade/keepers/ –collections	–Escapes/ releases	• Wide diet • Behavioural flexibility o Urban/semi-urban	• Human commensalism • Behavioural flexibility • Foraging flights up to 12km o Sedentary	0 to -1	0 to +1

Table D2. Summary of plant species status in England, invasion pathways and traits facilitating (•) or constraining (○) establishment and spread; plus overall assessment of the effect of landscape and climate warming on invasiveness.

		Invasion Stage				Effect on Establishment/Spread	
Species	**Status**	**Transport**	**Introduction**	**Establishment**	**Spread**	**Landscape**	**Climate Warming**
Ailanthus altissima	Naturalised	Human collection	Amenity planting	• anthropochory[3], suckering o Climate	• Large seed production, anemochory[4] o Climate	Zero/Very low	High
Buddleja davidii	Naturalised	Human collection	Garden introduction	• anthropochory, anemochory o Climate, disturbed ground	• Roads, railways, water o Climate, disturbed ground	Very Low	Medium
Cotoneaster horizontalis	Naturalised	Human collection	Garden introduction	• anthropochory, self-fertilisation o Shade	• zoochory[5] o Shade	Very Low	Low
Impatiens glandulifera	Naturalised	Human collection	Garden introduction	• Shade out competitors, o Climate, Frost sensitive seedlings	• 700-800 seeds/plant, vigorous seed dispersal o Dry soils, full sun	Medium	Medium
Pinus pinaster	Naturalised	Human collection	Shelter belt, plantation	• anthropochory, fire o Climate, Time to maturity	• Fire o Habitat, anemochory	High	High
Prunus serotina	Naturalised	Human collection	Garden introduction	• anthropochory[6], shade o Climate	• anthropochory, zoochory[7] o Climate	Low	Very low

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[3] Anthropochory: the dispersal of plants by man.

[4] Anemochory: the dispersal of plant seed by the wind.

[5] Zoochory: the distribution of fruits and seeds by animals, either on animal surfaces; in animal digestive tracts; and by storing fruits or seeds.

[6] Anthropochory: the dispersal of plants by man.

[7] Zoochory: the distribution of fruits and seeds by animals, either on animal surfaces; in animal digestive tracts; and by storing fruits or seeds.

		Invasion Stage				**Effect on Establishment/Spread**	
Species	**Status**	**Transport**	**Introduction**	**Establishment**	**Spread**	**Landscape**	**Climate Warming**
Quercus ilex	Naturalised	Human collection	Garden introduction	• anthropochory, o Climate	• anthropochory, zoochory o Habitat, Climate	Very Low	High
Rosa rugosa	Naturalised	Human collection	Garden introduction	• anthropochory o Open, well-drained habitat	• zoochory, rhizomes o Open, well-drained habitat	Medium	Low
Senecio inaequidens	Naturalised	Wool trade, natural	Wool contaminant	• anemochory[8] o Climate	• anemochory o Climate	High	High
Solidago gigantea	Naturalised	Human collection	Garden introduction	• anemochory anthropochory, rhizomes o Habitat, shade	• anemochory o Habitat, shade	Medium	Low

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[8] Anemochory: the dispersal of plant seed by the wind.

Table D3. Summary of invertebrate species status in England. Invasion pathways plus an overall assessment of the effect of landscape and climate warming on invasiveness.

		Invasion Stage				Effect on Establishment/Spread	
Species	**Status**	**Transport**	**Introduction**	**Establishment**	**Spread**	**Landscape**	**Climate Warming**
Monochamus sartor	Absent	Wood packaging material	NA	Possible in warmer parts of UK with climate change	Natural and movement of wood	Low	High
Cyphostethus triatus	Established but still generally absent or at most occasional	Natural migrant	Unknown	Already in northern England and Scotland	Natural migration	High via hosts	Moderate
Lymantria dispar	Established but still generally absent or at most occasional	Possibly hitchhiker on freight (as egg masses)	Found in 1995	Established in and around London	Natural	Low	High
Thaumetopoea processionea	Established but still generally absent or at most occasional	On plants for planting	Introduced around 2007	In London area	Natural	Low	High
Anoplophora glabripennis	Not or scarcely established	Wood packaging	Outbreak discovered 2012	Under eradication	Natural and movement of wood and nursery stock	Low	High
Anoplophora chinensis	Absent	Plants for planting	Has only been found in association with imported plants	NA	Natural and movement of nursery stock	Low	High
Corythucha arcuata	Absent	Plants for planting	Not introduced	NA	Natural and movement of nursery stock	Low	High
Agrilus planipennis	Absent	Plants for planting, as hitchhiker	Not introduced	NA	Natural and movement of nursery stock	Low	Moderate

		Invasion Stage				Effect on Establishment/Spread	
Species	**Status**	**Transport**	**Introduction**	**Establishment**	**Spread**	**Landscape**	**Climate Warming**
Lasius neglectus	Not or scarcely established	Plants for planting	Introduced in 2009, possibly with imported plants	One small outbreak in Gloucestershire	Natural spread and movement of plants	Low	Moderate
Gelechia senticetella	Established but still generally absent or at most occasional	Natural	Recorded in light trap in 1988	Established in south-east England	Natural spread movement of plants	Low	Moderate

Table D4. Summary of mammal species status in England, invasion pathways and traits facilitating (•) or constraining (○) establishment and spread; plus overall assessment of the effect of landscape and climate warming on invasiveness.

		Invasion Stage				Effect on Establishment/Spread	
Species	**Status**	**Transport**	**Introduction**	**Establishment**	**Spread**	**Landscape**	**Climate Warming**
Myocastor coypu	Eradicated	Deliberate human movement	Escapes/ release	o Wetland restricted o Temperature restricted • Early weaning and breeding age	o Slow disperser o Habitat restricted	High	High
Ondatra zibethicus	Eradicated	Deliberate human movement	Escapes/ release	o Wetland restricted • Early weaning and breeding age	• Reasonable disperser o Habitat restricted	High	Medium
Pecari tajacu	In captivity	Deliberate human movement	Escapes/ release	• Early weaning and breeding age • Broad habitat, climate and diet	• Large dispersal possible • Moves in groups- always near mates	Medium	Medium
Felis bengalensis	In captivity and the pet trade	Deliberate human movement	Escapes/ release	• Broad habitat, climate and diet • Regular occurrence in captivity	• Large dispersal possible • Flexible habitat and diet	Medium	Medium
Hydrochaeris hydrochaeris	In captivity	Deliberate human movement	Escapes/ release	o Wetland restricted o Temperature restricted	o Slow disperser o Habitat restricted	High	High
Nyctereutes procyonoides	Captivity and the pet trade	Deliberate human movement	Escapes/ release	• Early weaning and breeding age • Broad habitat, climate and diet • Able to hibernate o Needs a mixed habitat with varying food	• Large dispersal possible • Flexible diet	Medium	Medium

		Invasion Stage				Effect on Establishment/Spread	
Species	**Status**	**Transport**	**Introduction**	**Establishment**	**Spread**	**Landscape**	**Climate Warming**
Procyon lotor	Pet trade, some localized individuals sighted/caught in the past	Deliberate human movement	Escapes/ release	• Early weaning and breeding age • Broad habitat, climate and diet • Can live in cities	• Large dispersal possible • Flexible diet • Can thrive in cities	Low	Low
Pudu puda	Captivity	Deliberate human movement	Escapes/ release	o Restricted to woods • Breeds early o Influenced by habitat disturbance	• Habitat specialist but rapid colonizer in native range	Medium	Medium
Hydropotes inermis	Localized population in South East	Deliberate human movement	Escapes/ release	o Habitat restricted • Breeds prolifically and from early age	o Slow spread • Can utilize different habitat types	Medium	Low
Mephitis mephitis	Pet trade, some localized individuals sighted/caught in the past	Deliberate human movement and some potential stowaways on lorries etc.	Escapes /release	• Early weaning and breeding age • Broad habitat, climate and diet • Can live in cities	• Large dispersal possible • Flexible diet • Can thrive in cities	Low	Low
Nasua nasua	Captivity with escapes from zoos which are caught back	Deliberate human movement	Escapes/ release	• Early weaning and breeding age • Broad habitat, climate and diet o Woodland specialist	• Large dispersal possible • Flexible diet	Medium	Medium
Cynomys ludovicianus	Localized populations	Deliberate human movement	Escapes/ release	o Habitat specialist • Early reproduction • Early weaning	• Large dispersal possible	High	Medium

Table D5. Summary of herptile species status in England, invasion pathways and traits facilitating (•) or constraining (○) establishment and spread; plus overall assessment of the effect of landscape and climate warming on invasiveness.

		Invasion Stage				Effect on Establishment/Spread	
Species	**Status**	**Transport**	**Introduction**	**Establishment**	**Spread**	**Landscape**	**Climate Warming**
Pelophylax ridibundus	Localized	Deliberate human movement Accidental vegetation movement	Escapes/ release Accidental in waste water and aquatic plants	o Wetland restricted o Temperature restricted • Early weaning and breeding age • Good competitor • Large clutch size	o Slow disperser o Habitat restricted	High	High
Xenopus laevis	Localized	Deliberate human movement Accidental vegetation movement	Escapes/ release Accidental in waste water and aquatic plants	o Wetland restricted • Good disperser and colonizer of sub optimal habitat • Broad-predatory diet • Large clutch size	• Reasonable disperser • Exploits sub optimal habitats	High	Medium
Macrochelys temminckii	Absent- in captivity	Deliberate human movement	Escapes/ release	• Prevalent in pet trade • Broad-predatory diet • Large clutch size • Long-lived • Needs to eat only infrequently	• Can remain undetected for a long time	Medium	High

		Invasion Stage				Effect on Establishment/Spread	
Species	**Status**	**Transport**	**Introduction**	**Establishment**	**Spread**	**Landscape**	**Climate Warming**
Python molurus	Absent- in captivity	Deliberate human movement Stow aways on transport systems, e.g. boats and rubble and lorries	Escapes/ release	• Broad -predatory diet • Needs to eat only infrequently • Large clutch size • Regular occurrence in captivity • Can adapt burrows to temperature	• Large dispersal possible • Flexible habitat and diet • Exploits disturbed and human habitats	High	Medium
Bufo marinus	Absent- in captivity	Deliberate human movement Stow aways on transport systems, e.g. boats and rubble and lorries	Escapes/ release	• Wetland restricted, but can exploit manmade environments o Temperature restricted • Can adapt burrows to temperature • Large clutch size	• Exploits disturbed and human habitats	High	Medium
Chelydra serpentina	Absent- in captivity	Deliberate human movement	Escapes/ release	• Large clutch size • Long-lived • Needs to eat only infrequently	• Can remain undetected for a long time	Medium	High
Trachemys scripta	Widespread	Deliberate human movement	Escapes/ release	• Long-lived • Broad habitat, climate and diet • Can live in cities • Needs to eat only infrequently	• Large dispersal possible • Flexible diet • Can thrive in cities	High	High

		Invasion Stage				Effect on Establishment/Spread	
Species	**Status**	**Transport**	**Introduction**	**Establishment**	**Spread**	**Landscape**	**Climate Warming**
Mesotriton alpestris	Localized	Deliberate human movement Accidental vegetation movement	Escapes/ release Accidental in waste water and aquatic plants	• Not as wetland restricted • Broad habitat needs	• Large dispersal movements have been recorded	Medium	Medium
Podarcis muralis	Localized	Deliberate human movement	Escapes/ release	o Habitat restricted	o Slow spread • Can utilize different habitat types such as cities o Climate restricted	High	High

12. Appendix E: References used in plant species analysis

Ailanthus altissima
https://secure.fera.defra.gov.uk/nonnativespecies/factsheet/downloadFactsheet.cfm?speciesId=101
http://www.eppo.int/QUARANTINE/Pest_Risk_Analysis/PRAdocs_plants/draftds/05-11828%20DS%20Ailanthus%20altissima.doc

Buddleja davidii
https://secure.fera.defra.gov.uk/nonnativespecies/factsheet/factsheet.cfm?speciesId=581
Global Invasive Species database -
http://www.issg.org/database/species/ecology.asp?fr=1&si=650

Cotoneaster horizontalis
https://secure.fera.defra.gov.uk/nonnativespecies/factsheet/downloadFactsheet.cfm?speciesId=964

Impatiens glandulifera
https://secure.fera.defra.gov.uk/nonnativespecies/factsheet/downloadFactsheet.cfm?speciesId=1810
http://www.eppo.int/QUARANTINE/Pest_Risk_Analysis/PRAdocs_plants/draftds/05-11831%20DS%20Impatiens%20glandulifiera.doc

Pinus pinaster
https://secure.fera.defra.gov.uk/nonnativespecies/factsheet/factsheet.cfm?speciesId=2727

Prunus serotina
https://secure.fera.defra.gov.uk/nonnativespecies/factsheet/downloadFactsheet.cfm?speciesId=2858
http://www.eppo.int/QUARANTINE/Pest_Risk_Analysis/PRAdocs_plants/draftds/05-11834rev%20DS%20Prunus%20serotina.doc
http://www.nobanis.org/files/factsheets/Prunus%20serotina.pdf

Quercus ilex
https://secure.fera.defra.gov.uk/nonnativespecies/factsheet/factsheet.cfm?speciesId=2958

Rosa rugosa

https://secure.fera.defra.gov.uk/nonnativespecies/factsheet/downloadFactsheet.cfm?speciesId=3041

http://www.europe-aliens.org/pdf/Rosa_rugosa.pdf

http://www.nobanis.org/files/factsheets/Rosa_rugosa.pdf

Senecio inaequidens

https://secure.fera.defra.gov.uk/nonnativespecies/factsheet/downloadFactsheet.cfm?speciesId=3225

http://www.eppo.int/QUARANTINE/Pest_Risk_Analysis/PRAdocs_plants/draftds/05-11836rev%20EPPO%20DS%20SENIQ.doc

http://www.nobanis.org/files/factsheets/Senecio_inaequidens.pdf

Solidago gigantea

https://secure.fera.defra.gov.uk/nonnativespecies/factsheet/downloadFactsheet.cfm?speciesId=3324

http://www.eppo.int/QUARANTINE/Pest_Risk_Analysis/PRAdocs_plants/draftds/05-11839%20DS%20Solidago%20gigantea.doc

13. Appendix F: Insects excluded from analysis

Invertebrates excluded from scoring

Invertebrates marked as native to or extinct from GB were not included in the scoring exercise unless they were considered a current threat.

Spread by human activity and their habitats are heated food stores or premises:

American Seed Beetle - Acanthoscelides obtectus
Lesser Mealworm Beetle - Alphitobius diaperinus
Black Fungus Beetle - Alphtobius leavigatus
Australian Carpet Beetle - Anthrenocerus australisCarpet Beetles - Attagenus speciea
Blatta orientalis
Blattella germanica
Lentil Seed Beetle - Bruchus ervi
Pea Beetle - Bruchus pisorum
Bean Seed Beetle - Bruchus rufimanus
Adzuki Beanseed Beetle - Callosobruchus chinensis
Indian Stick-insect - Carausius morosus
Broad-Nosed Grain Weevil - Caulophilus oryzae
Larder Beetles - Dermestes species
Broad-horned Flour Beetle - Gnatocerus cornutus
House Longhorn - Hylotrupes bajulus
Coffee berry borer - Hypothenemus hampei
Long-Headed Flour Beetle - Latheticus oryzae
Pharo's ant - Monomorium pharaonis
Monomorium salomonis
Pentarthrum huttoni
Perigona nigriceps
Australian Spider Beetle - Ptinus tectus
Museum Nuisance - Reesa vespulae
Reticulitermes lucifugus
Dark Mealworm Beetle - Tenebrio obscurus
Rust-red Flour Beetle - Tribolium castaneum
Confused Flour Beetle - Tribolium confusum
Globular Spider Beetle - Trigonogenius globules
Warehouse beetle - Trogoderma variabile

Specific host – ornamental, crop plantsor honey bees:

Arenophilus peregrinus
Corsican Stick-insect - Bacillius rossius
Berberis Sawfly - Arge berberidis
Brachypterolus vestitus - Antirrhinum Beetle
Corticeus linearis
Colorado Beetle - Leptinotarsa decemlineata
Lily Beetle - Lilioceris lilii
Otiorhynchus species
Acericerus ribauti
Aphis oenotherae
Arocatus longiceps
Cacopsylla fulguralis
Closterotomus trivialis
Conostethus venustus
Cypress Aphid - Cinara cupressi
Crypturaphis grassii
Rhodedendron Whitefly - Dialeurodes chittendeni
Dicyphus escalerae
Edwardsiana nigriloba
Eremocoris fenestratus
Ligurian leafhopper - Eupteryx decemnotata
Fieberiella florii
Fieberiella septentrionalis
Laburnum sucker - Floria variegate
Rhododendron Leafhopper - Graphocephala fennahi
Homotoma ficus
Cottony cushion scale - Icerya purchasi
Illinoia azaleae
Illinoia goldamaryae
Illinoia lambersi
Illinoia morrisoni
Impatientinum asiaticum
Leucaspis podocarpi
Liguropia juniperi
Macrolophus melanotoma
Macrosiphum albifrons

Neotoxoptera formosana
Southern Green Shield Bug - Nezara viridula
Tamarix Leafhopper - Opsius stactogalus
Azalea whitefly - Pealius azaleae
Pineus similis
Placotettix taeniatifrons
Cypress mealybug - Planococcus vovae
Hydrangea Scale - Pulvinaria hydrangeae
Hottentot fig scale - Pulvinariella mesembryanthemi
Rhododendron Lacebug - Stephanitis rhododendri
Andromeda Lacebug - Stephanitis takeyai
Takecallis taiwana
Takecallis taiwanus
Toxoptera aurantii
Bay Sucker - Trioza alacris
Pittosporum Psyllid - Trioza vitreoradiata
Tupiocoris rhododendri
Tuponia brevirostris
Tuponia mixticolor
Asian Hornet - Vespa velutina
Zygina nivea

Found on ornamental indoor or glasshouse plants:
Soft scale - Coccus hesperidum
An orchid scale insect - Coccus pseudohesperidum
A palm scale insect - Colobopyga kewensis
Frankliniella occidentalis
Hemiberlesia palmae
Green shield scale - Pulvinaria psidii
Olive hemipherical scale - Saissetia oleae

Insufficient information is available to score the following species:
Canada Thistle Beetle - *Altica carduorum*
Palm seed borer - Coccotrypes dactyliperda
Cryptops anomalans
Gronops inaequalis
Hadrognathus longipalpis

Hypothenemus eruditus
Leistus (Pogonophorus) rufomarginatus
Lithocharis nigriceps
Macrorhyncolus littoralis
Magdalis (Magdalis) memnonia
Plain shortwing beetle - Nathrius brevipennis
Scybalicus oblongiusculus
Thecturota marchii
Brachycaudus amygdalinus
Acericerus heydenii
Anoterostemma ivanhofi
Brachynotocoris punctipennis
Nothogeophilus turki
Lamyctes emarginatus
Lithobius peregrinus
Pachymerium ferrugineum
Stigmatogaster souletina
Arcitalitrus dorrieni
Metatrichoniscoides leydigii
Cacyreus marshalli
Heteropterus Morpheus
Rapala schistacea
Anisolabis maritime
Andricus species
Aphelonyx cerricola
Crematogaster scutellaris
Hypoponera punctatissima
Ponera coarctata
Clitarchus hookeri
Oxidus gracilis
Unciger foetidus

14. Appendix G: Statistical tables of spread maps

Table G1. Cotoneaster horizontalis (proportion of total study area).

Scenario	Time lag (yrs)	Total prediction area (ha)	Proportion of total area (%)	Very high probability (%)	*	High probability (%)	*	Medium probability (%)	*	Low probability (%)	*	Very low probability (%)	*
Control	1	1972600	1.56	0.59	37.82	0.26	16.67	0.25	16.03	0.33	21.15	0.13	8.33
Control	10	10525400	8.35	2.25	26.95	1.6	19.16	1.42	17.01	1.48	17.72	1.59	19.04
More	1	1023400	0.81	0.24	29.63	0.14	17.28	0.14	17.28	0.14	17.28	0.15	18.52
More	10	9690200	7.69	1.64	21.33	1.64	21.33	1.44	18.73	1.53	19.90	1.44	18.73
Bigger	1	2500800	1.98	0.73	36.87	0.35	17.68	0.28	14.14	0.41	20.71	0.21	10.61
Bigger	10	13969000	11.08	3.06	27.62	2.05	18.50	2.26	20.40	2.03	18.32	1.69	15.25
Joined up	1	2505400	1.99	0.7	35.18	0.35	17.59	0.29	14.57	0.42	21.11	0.24	12.06
Joined up	10	17353800	13.77	3.21	23.31	2.42	17.57	2.94	21.35	2.72	19.75	2.47	17.94
Climate change	1	1023400	0.81	0.24	29.63	0.14	17.28	0.14	17.28	0.14	17.28	0.15	18.52
Climate change	10	9690200	7.69	1.64	21.33	1.64	21.33	1.44	18.73	1.53	19.90	1.44	18.73
Agri-environment schemes (-ve)	1	1320000	1.05	0.3	28.57	0.19	18.10	0.19	18.10	0.18	17.14	0.19	18.10
Agri-environment schemes (-ve)	10	10096300	8.01	2.06	25.72	1.55	19.35	1.5	18.73	1.39	17.35	1.51	18.85
Agri-environment schemes (+ve)	1	2021400	1.6	0.9	56.25	0.3	18.75	0.14	8.75	0.13	8.13	0.13	8.13
Agri-environment schemes (+ve)	10	10633700	8.44	2.29	27.13	1.6	18.96	1.43	16.94	1.51	17.89	1.61	19.08

Table G2. Cotoneaster horizontalis (proportion of Dorset AONB).

Scenario	Time lag (yrs)	Total prediction area (ha)	Proportion of total area (%)	Very high probability (%)	*	High probability (%)	*	Medium probability (%)	*	Low probability (%)	*	Very low probability (%)	*
Control	1	27758000	0.86	0.37	43.02	0.18	20.93	0.13	15.12	0.11	12.79	0.06	6.98
Control	10	100994900	3.12	1.13	36.22	0.53	16.99	0.53	16.99	0.48	15.38	0.45	14.42
More	1	10242500	0.32	0.06	18.75	0.06	18.75	0.06	18.75	0.07	21.88	0.07	21.88
More	10	86014000	2.66	0.63	23.68	0.54	20.30	0.51	19.17	0.49	18.42	0.49	18.42
Bigger	1	24541300	0.76	0.32	42.11	0.15	19.74	0.1	13.16	0.1	13.16	0.09	11.84
Bigger	10	118841900	3.67	1.21	32.97	0.67	18.26	0.63	17.17	0.61	16.62	0.55	14.99
Joined up	1	23003500	0.71	0.29	40.85	0.14	19.72	0.09	12.68	0.1	14.08	0.1	14.08
Joined up	10	131602100	4.06	1.1	27.09	0.77	18.97	0.72	17.73	0.78	19.21	0.7	17.24
Climate change	1	10242500	0.32	0.06	18.75	0.06	18.75	0.06	18.75	0.07	21.88	0.07	21.88
Climate change	10	86014000	2.66	0.63	23.68	0.54	20.30	0.51	19.17	0.49	18.42	0.49	18.42
Agri-environment schemes (-ve)	1	21695500	0.71	0.22	30.99	0.17	23.94	0.12	16.90	0.09	12.68	0.1	14.08
Agri-environment schemes (-ve)	10	97710000	3.19	1.09	34.17	0.57	17.87	0.54	16.93	0.5	15.67	0.48	15.05
Agri-environment schemes (+ve)	1	28738500	0.94	0.48	51.06	0.19	20.21	0.12	12.77	0.1	10.64	0.05	5.32
Agri-environment schemes (+ve)	10	101989300	3.33	1.21	36.34	0.59	17.72	0.54	16.22	0.51	15.32	0.47	14.11

Table G3. Asian Longhorn Beetle (proportion of total study area).

Scenario	Time lag (yrs)	Total prediction area (ha)	Proportion of total area (%)	Very high probability (%)	*	High probability (%)	*	Medium probability (%)	*	Low probability (%)	*	Very low probability (%)	*
Control	1	1292600	0.99	0.44	44.44	0.18	18.18	0.14	14.14	0.12	12.12	0.12	12.12
Control	10	4604700	3.54	1.47	41.53	0.75	21.19	0.56	15.82	0.43	12.15	0.33	9.32
More	1	13125200	10.09	7.24	71.75	1.14	11.30	0.81	8.03	0.7	6.94	0.21	2.08
More	10	16885700	12.98	11.58	89.21	1.08	8.32	0.1	0.77	0.09	0.69	0.13	1.00
Bigger	1	14444000	11.11	7.41	66.70	1.42	12.78	1.4	12.60	0.44	3.96	0.44	3.96
Bigger	10	19446600	14.95	13.06	87.36	1.32	8.83	0.33	2.21	0.09	0.60	0.16	1.07
Joined up	1	13906200	10.69	6.17	57.72	1.7	15.90	1.44	13.47	0.92	8.61	0.46	4.30
Joined up	10	19720900	15.16	12.74	84.04	1.83	12.07	0.37	2.44	0.14	0.92	0.09	0.59
Agri-environment schemes (-ve)	1	12426200	9.56	6.83	71.44	1.37	14.33	0.65	6.80	0.28	2.93	0.43	4.50
Agri-environment schemes (-ve)	10	15763700	12.12	10.9	89.93	0.97	8.00	0.1	0.83	0.09	0.74	0.05	0.41
Agri-environment schemes (+ve)	1	12473200	9.59	7.16	74.66	1.24	12.93	0.51	5.32	0.51	5.32	0.18	1.88
Agri-environment schemes (+ve)	10	15879900	12.21	11.2	91.73	0.69	5.65	0.09	0.74	0.09	0.74	0.14	1.15

Table G4. Asian Longhorn Beetle (proportion of Dorset AONB).

Scenario	Time lag (yrs)	Total prediction area (ha)	Proportion of total area (%)	Very high probability (%)	*	High probability (%)	*	Medium probability (%)	*	Low probability (%)	*	Very low probability (%)	*
Control	1	1751000	0.05	0.02	40.00	0.01	20.00	0.01	20.00	0.01	20.00	0.01	20.00
Control	10	5753700	0.18	0.08	44.44	0.04	22.22	0.03	16.67	0.02	11.11	0.02	11.11
More	1	24444800	0.75	0.39	52.00	0.14	18.67	0.1	13.33	0.06	8.00	0.06	8.00
More	10	56794000	1.75	1.03	58.86	0.23	13.14	0.17	9.71	0.1	5.71	0.21	12.00
Bigger	1	31317400	1.02	0.43	42.16	0.17	16.67	0.16	15.69	0.07	6.86	0.19	18.63
Bigger	10	71799500	2.34	1.27	54.27	0.31	13.25	0.26	11.11	0.21	8.97	0.3	12.82
Joined up	1	29262300	0.9	0.33	36.67	0.17	18.89	0.13	14.44	0.09	10.00	0.18	20.00
Joined up	10	65246100	2.01	1.17	58.21	0.28	13.93	0.22	10.95	0.17	8.46	0.17	8.46
Agri-environment schemes (-ve)	1	21093900	0.69	0.39	56.52	0.14	20.29	0.08	11.59	0.04	5.80	0.04	5.80
Agri-environment schemes (-ve)	10	46361300	1.51	0.95	62.91	0.19	12.58	0.13	8.61	0.09	5.96	0.15	9.93
Agri-environment schemes (+ve)	1	21391000	0.7	0.42	60.00	0.13	18.57	0.07	10.00	0.05	7.14	0.03	4.29
Agri-environment schemes (+ve)	10	48081100	1.57	0.97	61.78	0.22	14.01	0.1	6.37	0.1	6.37	0.18	11.46

Table G5. Alpine Newt (proportion of total study area).

Scenario	Time lag (yrs)	Total prediction area (ha)	Proportion of total area (%)	Very high probability (%)	*	High probability (%)	*	Medium probability (%)	*	Low probability (%)	*	Very low probability (%)	*
Control	1	207300	0.16	0.04	25.00	0.03	18.75	0.03	18.75	0.03	18.75	0.03	18.75
Control	10	10472100	8.06	1.84	22.83	1.39	17.25	1.52	18.86	1.68	20.84	1.63	20.22
More	1	2515300	1.94	0.35	18.04	0.46	23.71	0.42	21.65	0.36	18.56	0.35	18.04
More	10	17610700	13.55	3.41	25.17	2.51	18.52	2.96	21.85	2.38	17.56	2.29	16.90
Bigger	1	3200200	2.46	0.43	17.48	0.59	23.98	0.51	20.73	0.48	19.51	0.45	18.29
Bigger	10	19844900	15.27	3.98	26.06	3.13	20.50	2.27	14.87	2.92	19.12	2.96	19.38
Joined up	1	3467000	2.67	0.39	14.61	0.55	20.60	0.64	23.97	0.55	20.60	0.53	19.85
Joined up	10	26508500	20.39	4.77	23.39	3.62	17.75	2.78	13.63	3.79	18.59	5.43	26.63
Climate change	1	1340100	1.03	0.18	17.48	0.2	19.42	0.22	21.36	0.21	20.39	0.22	21.36
Climate change	10	11176300	8.6	2.02	23.49	1.45	16.86	1.61	18.72	1.85	21.51	1.67	19.42
Agri-environment schemes (-ve)	1	1145400	0.88	0.15	17.05	0.16	18.18	0.17	19.32	0.21	23.86	0.19	21.59
Agri-environment schemes (-ve)	10	10165200	7.82	1.77	22.63	1.41	18.03	1.43	18.29	1.61	20.59	1.6	20.46
Agri-environment schemes (+ve)	1	1252000	0.96	0.19	19.79	0.16	16.67	0.19	19.79	0.2	20.83	0.22	22.92
Agri-environment schemes (+ve)	10	11364400	8.74	1.91	21.85	1.5	17.16	1.65	18.88	1.81	20.71	1.87	21.40

Table G6. Alpine Newt (proportion of Dorset AONB).

Scenario	Time lag (yrs)	Total prediction area (ha)	Proportion of total area (%)	Very high probability (%)	*	High probability (%)	*	Medium probability (%)	*	Low probability (%)	*	Very low probability (%)	*
Control	1	4270700	0.13	0.03	23.08	0.03	23.08	0.02	15.38	0.02	15.38	0.02	15.38
Control	10	77477700	2.39	0.48	20.08	0.47	19.67	0.47	19.67	0.49	20.50	0.5	20.92
More	1	12790400	0.42	0.1	23.81	0.08	19.05	0.08	19.05	0.08	19.05	0.07	16.67
More	10	115746400	3.78	0.77	20.37	0.72	19.05	0.76	20.11	0.79	20.90	0.73	19.31
Bigger	1	14748400	0.48	0.1	20.83	0.1	20.83	0.09	18.75	0.1	20.83	0.08	16.67
Bigger	10	130390700	4.25	0.88	20.71	0.85	20.00	0.81	19.06	0.89	20.94	0.82	19.29
Joined up	1	17591400	0.57	0.13	22.81	0.12	21.05	0.12	21.05	0.11	19.30	0.1	17.54
Joined up	10	168335200	5.49	1.15	20.95	1.22	22.22	1.16	21.13	1.02	18.58	0.94	17.12
Climate change	1	8915900	0.28	0.07	25.00	0.06	21.43	0.05	17.86	0.05	17.86	0.05	17.86
Climate change	10	81667200	2.52	0.51	20.24	0.47	18.65	0.49	19.44	0.52	20.63	0.53	21.03
Agri-environment schemes (-ve)	1	8287600	0.27	0.07	25.93	0.06	22.22	0.05	18.52	0.05	18.52	0.05	18.52
Agri-environment schemes (-ve)	10	75987500	2.48	0.5	20.16	0.49	19.76	0.48	19.35	0.51	20.56	0.51	20.56
Agri-environment schemes (+ve)	1	9135100	0.3	0.08	26.67	0.06	20.00	0.06	20.00	0.05	16.67	0.05	16.67
Agri-environment schemes (+ve)	10	82210000	2.68	0.53	19.78	0.53	19.78	0.55	20.52	0.54	20.15	0.54	20.15

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