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ARE REINTRODUCTIONS AN EFFECTIVE WAY OF MITIGATING AGAINST PLANT EXTINCTIONS?

Systematic Review

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Summary

1. Background

Re-introductions are considered by some conservation practitioners to be a controversial management option for mitigating threatened plant declines. The use of translocations (including re-introductions) has been criticised for the lack of monitoring and central recording, inappropriateness of the action due to genetic considerations, a lack of knowledge of the demography of the donor populations and inadequate information on the habitat requirements of the species. Despite these arguably justified criticisms, re-introductions are growing in use as practitioners see no other option for meeting management plan targets. Re-introductions have been proposed as options for overcoming habitat loss, habitat fragmentation and reproductive isolation. An extension of this increasingly interventionist approach, often termed assisted colonisation, is being considered as a potential method for preventing extinctions due to climatic shifts too rapid to allow corresponding species' distribution changes.

This review evaluates the effectiveness of re-introductions as a conservation tool by using the available evidence to determine in what context plant translocations have improved the status of threatened species.

2. Objectives

To evaluate the effectiveness of re-introductions as a method for mitigating extinctions of plant species by answering the following question: are re-introductions an effective method of increasing the viability of endangered or vulnerable plant species?

3. Methods

Ten electronic databases were searched using ten sets of search terms. The library databases of the Joint Nature Conservation Committee, Natural England, Scottish Natural Heritage and the Countryside Council for Wales were searched on behalf of the review team by staff at each of the agencies using search terms provided. The IUCN's Re-introductions Specialist Group and the Center for Plant Conservation have both published volumes on re-introductions which were used to identify cases for inclusion and practitioners that might be contacted for details on specific re-introduction attempts. The Botanical Society of the British Isles and Plantlife also provided databases on plant re-introductions in the UK.

4. Main results

Using systematic review protocols, we identified peer-reviewed and grey literature that provided evidence that re-introductions (using various definitions which some practitioners might describe as conservation introductions) have been attempted, or are planned, for approximately 700 taxa in 32 countries. The USA is the biggest advocate and implementer of re-introductions (228 taxa) and when considered in combination with the high use of the technique in Europe (particularly the UK), this explains why more than 300 taxa are associated with temperate forest biomes. National conservation protection was afforded by 28 countries to 440 taxa although

we could not identify levels of protection for about 200 taxa and most (618) had not been evaluated using IUCN Red List protocols making it difficult to discern any links between the perceived extinction threat and use of re-introduction as a management strategy. Threats to target taxa were recorded and agriculture, grazing, competition from invasive or aggressive plants and urban or industrial development were most often cited accounting for 60% of stated causes of decline. When the level of endemism was used to categorise re-introduction targets, approximately one third could be classed as narrow endemics, one third had, at least formerly, wide distributions (cross-continental), and one third fell in between these extremes. This, together with the high level of national protection, indicates that many re-introductions are undertaken because the taxon is declining in part of its range rather than being undertaken as a 'last resort' to mitigate species-level extinctions.

Metadata analysis of 301 attempted re-introductions of 128 plant taxa generated relative measures of re-introduction success based on propagule survival, population persistence and potential for recruitment of progeny. We found that attempting to summarise re-introduction success based on the results reported in the literature may erroneously imply that re-introductions are mostly successful. This is due to early reporting of outcomes in the literature: the average monitoring time prior to publishing the outcome of a study is about 3 years. Even for annual species this time period is insufficient to judge whether a re-introduction has been successful as populations may succumb to inter-annual variation over longer time scales. Our treatment of the data to discern population persistence (whether extant or extinct at specified time points) is a coarse measure, but illuminating: for those projects that were monitored for more than 10 years, most re-introduction attempts failed i.e. no plants had survived at the last survey. Further, the vast majority of projects initiated more than 5 years prior to this review are unknown in outcome indicating that there potentially exists a vast pool of data that could be used to better evaluate re-introductions if made publicly available.

We used covariates associated with the target organism and intervention (the methods used to re-introduce the target) to discern patterns in the success of re-introduction derived from the proportion of surviving propagules. We found that many factors that might be expected to confer lower risk to a project could not be linked to increased success of threatened plant re-introductions. These factors included removing the cause of original species decline from a site prior to propagule introduction, ensuring the site is within the historic range of the species and sourcing propagules from wild, rather than *ex situ*, populations.

5. Conclusions

A narrative synthesis of speculative causes for failure and the absence of empirical evidence that the factors mentioned above can enhance success, are combined to support calls for amended guidelines for future re-introduction projects. Further monitoring and improved dissemination of results of existing re-introduction projects is needed. Plus, more rigorous project design using treatment and site replication in addition to improved monitoring of individuals and populations is required to conclusively elucidate the causes of failure in this increasingly utilised restoration technique.

1. Background

Twenty years ago the IUCN's Re-introduction Specialist Group devoted most of an issue of their newsletter 'Re-introduction News' to plant re-introductions (Maunder, 1991). In the editorial, Maunder (1991) noted that compared with translocations of animals, plant re-introductions were 'relatively under-studied and little debated'. Today, this situation has changed greatly – translocations of plant seeds or individuals are now widely discussed in the scientific and conservation press and form the subject of an increasing volume of research work.

In the following year, Maunder (1992) wrote that plant re-introductions could 'at present only be regarded as experimental'. Again, there has been a significant change since then, as re-introductions have now been absorbed into the 'toolkit' available to plant conservationists and population establishment (although not always by re-introduction) is advocated in 41 of the 63 species action plans for vascular plants under the UK Biodiversity Action Plan (UK Biodiversity Group, 1999). Re-introductions have been proposed as options for overcoming habitat loss, habitat fragmentation causing reproductive isolation (Quinn et al., 1994), and as a potential method for preventing extinctions of dispersal-limited species due to rapid climate change (Hulme, 2005).

Despite increased discussion and expansion out of the purely experimental domain into applied conservation, plant re-introductions are still questioned in the literature (e.g. Hodder and Bullock, 1997; Pearman and Walker, 2004; Strahm, 2003; Sutherland et al., 2006). The Botanical Society of the British Isles' 2006 conference highlighted the rise in application of plant introductions without an equivalent increase in success rate. The use of translocations, including re-introductions, has been criticised for the lack of monitoring and central recording, inappropriateness of the action due to genetic considerations, a lack of knowledge of the demography of the donor populations and inadequate information on the habitat requirements of the species.

This review aims to evaluate whether re-introductions should be advocated as a conservation tool by using available evidence to determine in what context plant translocations can improve threatened species' status and which situations the technique might be inappropriate.

2. Objectives

The objective of this review is to evaluate the effectiveness of re-introductions as a method for mitigating plant species extinctions by answering the following question:

Are re-introductions an effective method for increasing the viability of endangered or vulnerable plant species?

To do this we have attempted to identify threatened plant species that have undergone translocation in order to mitigate further decline in individual or population numbers. We state the number of re-introduction successes and failures. The inclusion of covariates related to the species, its status and management intervention forms part of the evaluation process. We use these covariates to distinguish subgroupings within the dataset and identify patterns in the data that can form the basis of recommendations to practitioners embarking on re-introduction projects.

3. Methods

3.1 Question formulation

Question formulation was initially driven through a process of informal correspondence with plant conservation practitioners and instigated formally by distribution of the question setting proforma in August 2006. The proforma consisted of a set of standard questions asking respondents to comment on the utility of the proposed systematic review before asking for opinions on what subjects, interventions and outcomes should be included and what factors might be reasons for heterogeneity in the resulting dataset. The proforma also asked the respondents to suggest key sources of evidence and who they thought the stakeholders were. Draft review protocols were produced in response to the proforma and distributed to two re-introduction practitioners with experience in the UK and overseas. Their comments were incorporated into the final protocol and contributed to the formulation of the primary research question above.

3.2 Definition of terms

Several terms have come to be associated with the technique of re-introductions and ambiguous use of words such as 'translocation', 'augmentation' and 're-establishment' is one of the contributing factors to some confusion around the ongoing debate. The following section explains the different usage of terms and defines what is meant by those which are used in the remainder of the report. It should be emphasised that due to the variable definitions that accompany each term, they are not necessarily mutually exclusive and often refer to specific stages or aspects of the intervention.

Translocation is an overarching term and has been used by the IUCN to refer to movement of living organisms from one area with free release in another (IUCN, 1987). It has been more precisely defined as deliberate and mediated movements of wild individuals or populations from one part of their range to another (IUCN, 1998). However, the term has been used more broadly by practitioners in the UK (pers. obs.) and Australia (Leonie Monks pers. comm.) to include movements of species via *ex situ* institutions as well as direct transport of wild-sourced individuals to recipient sites. American practitioners make the distinction between moving seeds and whole plants from an *in situ* site to recipient sites: translocations pertain to the latter and often refer to rescuing or salvaging plants from a localised threat which will destroy a population. This report uses the UK and Australian interpretation to mean movement of seeds or whole plants from wild and *ex situ* sources to any other sites. Translocations include the acts of introduction, re-introduction, and re-stocking.

Introduction is the intentional or accidental dispersal by human agency of a living organism outside its historically known native range (IUCN, 1987) and includes translocations that have occurred for reasons associated with (for example) food provision, horticulture and biocontrol. The negative impacts of non-native introductions are well documented in the scientific literature and beyond the scope of this project.

Re-stocking and associated terms including enhancement, augmentation, supplementation and re-enforcement, refer to the addition of individuals to extant, but often threatened, populations of the same species (Falk et al., 1996; IUCN, 1987, 1995).

Re-introductions are defined here as elsewhere, as attempts to establish a species in an area which was once part of its historical range but from which the species has been made (at least, locally) extinct. The reason for the extirpation from a given site may be natural or anthropogenic, but the attempted re-establishment is entirely deliberate and with the intention of enhancing the long-term survival of the species. Re-introductions may be undertaken with other objectives in mind such as promoting conservation awareness or reinstating keystone species and the ecosystem-level processes they drive, but these objectives are typically secondary to enhancing the demographic status of a species. The literature of re-introductions includes references using the hyphenated and non-hyphenated form of the term; it is the reviewers' opinion that there is no difference in meaning between these versions.

Assisted migration, assisted colonisation, managed relocation and other similar terms refer to translocations that aim to mimic the expected range change a species may experience under current and predicted climatic fluctuations. It involves the movement of propagules (any plant material used for translocation, including whole plants) to a site not within the species' current or historic range and by definition, is treated as a form of conservation or benign introduction. For this reason it is not included in the scope of this study. There are however, many commonalities between the rationale and practicalities of re-introductions and assisted migrations; many of the lessons learnt through attempted re-introductions may have relevance to the growing field of climate-responsive conservation practice.

3.3 Search strategy

The literature search strategy used the following electronic databases: ISI Web of Knowledge including ISI Web of Science (Science Citation Index expanded 1945-present) and ISI Proceedings (Science and Technology Proceedings 1990-present), JSTOR, Index to Theses Online (1970-present), Digital Dissertations Online, Dogpile Meta-search (internet search), Google Scholar (internet search), COPAC, Scirus, Scopus and ConservationEvidence.com. The following search terms were used (an asterisk denotes a wild card search term allowing for several permutations of each intervention type): plant* AND re-introduc*, plant* AND reintroduc*, plant* AND introduc*, plant* AND translocation*, plant* AND establish*, plant* AND re-establish*, plant* AND restor*, plant* AND reinstat*, plant* AND regenerat*, plant* AND assisted migration.

The libraries of Natural England, Scottish Natural Heritage, the Countryside Council for Wales, and the Joint Nature Conservancy Council were searched, by sending the search terms to the libraries' curators. In the case of the Countryside Council for

Wales the authors were given remote access and conducted the search using the search terms described above before requesting any relevant literature. The (re-) introduction records of the Botanical Society of the British Isles (BSBI) and Plantlife were also kindly made accessible to the authors and were incorporated into the literature search.

The IUCN Species Survival Commission's Re-introduction Directory (Soorae and Seddon, 1998) contains a list of practitioners that have undertaken re-introductions and the species they have worked with. This was used to identify studies and if literature could not be identified through the database search outlined above, the practitioners were contacted directly. The Center for Plant Conservation has produced a volume on re-introductions including chapters on particular aspects of practice and case studies (Falk et al., 1996). This was also used to identify suitable studies and practitioners.

3.4 Study inclusion criteria

The literature search was used to identify studies suitable for addressing the question of re-introductions initially by title but if there was any doubt of the studies relevance, the abstract was also acquired and judged by the following criteria. The study subject had to be a vascular plant which had undergone a deliberate translocation of individual plants or seeds to sites that were unoccupied at the time of translocation but previously supported extant populations or were judged to be within the former or historic range of the species. Although there is some discussion as to whether this is a strict definition of a re-introduction, for the remainder of this document we refer to these studies as re-introductions. Augmentations of extant populations were only included if it was possible to follow the survivorship of the translocated individuals. Habitat translocations or species translocations for reasons of habitat restoration were not included.

Outcomes were identified in the systematic review protocol as follows: survival of translocated populations for more than 5 years (may be restricted by time lapsed since intervention, translocations less than 5 years old will still be included in the review but not necessarily in the meta-analysis) measured either as number of individuals per population and/or proportion of population reproducing, abundance of species expressed as numbers of individuals per population and numbers of populations, successful recruitment, increased genetic diversity of population (measured as proportion of heterozygosity or polymorphism in a population, or number of genotypes) as compared to that of the species as a whole.

Types of study which would be accepted for review were also described in the protocol: quantitative studies with pre-intervention comparators and/or site comparisons were pre-requisites for meta-analysis, field evidence in the form of descriptive studies/reports were also collated.

3.5 Study quality assessment

Full text articles were considered for inclusion and if suitable, were included in a narrative review and quantitative meta-analysis based on the availability of data on propagule number and type (seed, juvenile or adult plant), number of propagules translocated, and number surviving over a monitoring period described in the study. It was our intention to run sensitivity analyses separating “high” from “low” quality studies. A high quality study was expected to include re- introduced populations at multiple sites which had been surveyed after an appropriate time for the population to establish and stabilise. However, after a preliminary search yielded few studies that included monitoring periods over three years and even fewer containing data on comparative sites, this was abandoned. Therefore, all studies that were found to be yielding a measure of propagule survival over a reported time period were included in the review and meta-analysis. Instead of the score-based assessment described in the review protocol, the impact of specific reported quality co-variates (length of monitoring period and number of propagules translocated) were explored in sensitivity analyses (see section 3.7 for more detail).

3.6 Data extraction

A single reviewer (SED) extracted information from all relevant studies; a summary of the species and intervention descriptors can be found in Appendix 4. The data were included in subsequent analysis if they met the study inclusion criteria and the minimum requirements for meta-analysis of propagule survival over time and/or the authors specified whether progeny recruited from the translocated plants had occurred. Propagule survival is derived from surveys of population size reported in the reviewed literature. It is here defined as the number of individuals at each survey expressed as a proportion of the number of propagules introduced. It can be a number greater than 1 as the study authors typically report the number of individuals within the population without discriminating between translocated propagules and their progeny. Proportional propagule survival and recruitment must therefore be aggregated into a combined measure and are referred to as ‘population size expressed as a percentage of propagule input’.

3.7 Data synthesis

Where appropriate data were available, the outcomes pertaining to proportional propagule survival (population size expressed as a percentage of propagule input), population persistence (i.e. whether any individuals have survived at a given time point), and potential for or achievement of recruitment were summarised in a narrative synthesis (see section 4.3). As discussed in section 3.6, where populations had undergone recruitment of progeny, these individuals were usually incorporated into surveys of population size, therefore measures of population size expressed as a percentage of the initial propagule pressure can provide an indicator of population growth or decline.

There were three deviations from the protocol with respect to data extraction and synthesis. It was initially intended that reproductive ability within each population could be expressed by the proportion of a population reproducing. However, this was often unreported and had to be adapted to instead report whether reproductive maturity and recruitment had been achieved. This outcome has therefore become a binary measure of recruitment presence or absence per attempt rather than a variable that could be placed on a scale of 0 to 1. The outcome describing the relative species’ abundance before and after the intervention was not used at all as it proved too difficult to get data on the number of ‘wild’ populations as a comparator. Genetic diversity comparisons between wild and re-introduced populations were also not included again because of the paucity of data derived from the search terms described above. Another systematic review is ongoing specifically on the genetic impacts of translocations (Raj Whitlock, pers. comm.), we refer any interested parties to this document. It was initially intended to exclude studies from the meta-analysis that had been reported with less than 5 years of monitoring prior to publication however, the paucity of available data prompted this threshold to be abandoned. Data on propagule survival was incorporated into the meta-analysis regardless of the length of monitoring period reported, time was alternatively included in the analysis as described below.

3.7.1 Meta-analytical techniques

Re-introductions of plants use seeds, juvenile plants or adults as the propagule type. Due to the increased risk of mortality when using seeds, re-introduction attempts of seeds, juveniles and adults were analysed separately. The proportion of re-introduced plants surviving at last census was used as an effect size with weights based on standard errors derived by assuming 50% mortality in a hypothetical control group (Cohen, 1977). We adopted this approach to overcome the problem that direct analysis of proportions underestimates confidence intervals and overestimates heterogeneity across effect sizes, especially when the observed proportions are very high or very low (Lipsey and Wilson, 2001). This distortion is due to compression of the standard error when the proportions approach one or zero (these extremes are given high weight in the analysis). The direct analysis is therefore only recommended in the rare circumstance when mean proportions are expected to be between 0.2 and 0.8 and only the mean is of interest. A standard alternative is use of logit proportions. Unlike the proportion, which is constrained to values between zero and one, the logit can take any numerical value and is approximately normal with a mean of zero. It therefore has appealing statistical properties. However, it now arbitrarily gives high weight to studies with mid-range proportions. Here we avoid the weighting distortions inherent with both standard approaches, and use risk ratios to provide meaningful effect measures. The larger the risk ratio, the higher the probability of death of individuals and hence failure of the re-introduction programme. DerSimmonian and Laird Random effects meta-analysis was used to generate pooled estimates of effect.

Formal meta-analysis of time to event outcomes could not be undertaken using standard techniques as this analysis requires the monitoring of survival of individual plants over time and therefore survival curves could not be derived for individual studies. However, time was included as an important co-variate using meta-regression (Higgins and Thompson, 2004). All analyses were performed in Stata version 11.0 (StataCorp USA).

3.7.2 Subgroup analysis

Subgroup analyses were used to explore the impact of key factors associated with the species or intervention type on re-introduction success. The risk ratios generated for each re-introduction attempt were combined for each subgroup; these pooled risk ratios and associated 95% confidence intervals were used to determine whether differentiation occurred between subgroups and if parameters could explain relative success of re-introduction attempts. The information needed to categorise these subgroups was taken from the studies (Appendix 1) or supplementary sources from reputable authorities on plant species taxonomic status, conservation threat status and distribution (Appendix 2). The subgroup types are summarised in Appendix 3 and justification for the choice of parameters defining the subgroups is as follows.

Endemism was included as a subgroup parameter in order to ascertain whether species with narrow ranges and therefore, narrow habitat tolerances, were any more difficult to re-introduce than those with much broader ranges. Subgroups were based on size of distribution and were categorised as global (where a species occurred on more than one continent), continental, regional (where the species occupies a major biome within a continent but could not be said to be widespread across a continental area), national, local or single site. The categories of regional and national are subjective and we considered merging them to avoid potential overlap and misleading subgroup allocation. However, the status of 'national endemic' is one of the few indicators of endemism that is often reported and therefore was retained where accurate descriptions of range size were absent and precluded reliable use of other subgroup categories.

The continued presence of the original cause of a species decline should normally prevent the inclusion of a site as a candidate for re-introduction. Indeed, the removal of the cause of decline is strongly recommended in many guidelines for conducting re-introduction programmes. However, in some cases the cause of decline may return to a site (e.g. herbivores or disease) or it may have been impossible to remove entirely prior to the re-introduction (e.g. periodically unfavourable climate). We considered this to be an important potential predictor of re-introduction success and consequently incorporated the presence or absence of the original cause of species decline in the subgroup analysis.

The IUCN Red List was used to acquire information on the level of threat of extinction of taxa forming the focus of re-introduction projects. We hypothesised that the more vulnerable a taxon was to extinction, the more difficult it might be to successfully re-introduce it due to too few or genetically depauperate propagules or lack of suitable habitat. If this turned out to be the case it might be sensible to allocate efforts and resources to less threatened species that would have a better chance of benefiting from a re-introduction programme. The ability to make strategic recommendations such as this, is an important aspect of the systematic review process and so was included in the subgroup analysis.

The status of the re-introduction site relative to the species' range was incorporated into the subgroup analysis by defining the site as: one which previously supported an extirpated population, one which is in the historic range of the species, or a site which is outside the historic range. There are some problems in applying these categories to the dataset mainly derived from different interpretations of range: some authors may not make a distinction between sites associated with extirpated populations and sites within the historic range. It is expected that some of the latter subgroup could be reclassified as sites of extirpated populations but without being able to corroborate this with the authors, we adopted the more conservative definition. Further, defining a site as outside the species' historic range may mean that the site has no recorded presence of the species or alternatively the site may occur outside the range 'polygon' which describes the most inclusive area of range and is defined by the outermost occurrence of a species. In all cases we primarily took the description provided by the authors of each study as our definition of site status relative to range. Where this was ambiguous or absent we attempted to determine site status using other sources on species' distribution.

The provenance of propagules in terms of sourcing from wild populations as opposed to *ex situ* propagation and cultivation is another parameter that features in guidelines for re-introduction programmes. It is generally recommended that wild individuals are preferable to those from *ex situ* sources if the donor population will not be adversely affected by their removal. However, for many threatened plants, the extant populations cannot support the removal of individuals especially if they are being moved to a potentially more risky site. Given this and the absence of welfare issues that animal conservationists must consider, *ex situ* involvement is perhaps a viable option despite risk of genetic bottlenecks and pest or pathogen transfer from cultivation. To test this, we included in our dataset a category denoting whether the propagules were translocated directly from wild populations or whether they had been propagated and/or cultivated in *ex situ* institutions.

The number of donor populations, i.e. the wild populations from where propagules were sourced, was included in the subgroup analysis because there are conflicting views about whether a single or multiple populations are better in a re-introduction context. Using a single donor population has advantages because threatened plant species often persist in isolated fragments of habitat and become selected for those specific conditions. If the site for re-introduction is thought to be ecologically similar to the donor site, it is prudent to avoid outbreeding depression of fitness which could result from disrupting co-evolved and adaptive traits. Multiple donor populations may offer benefits in some situations for the very reason that they may be detrimental in others; propagules from multiple populations would be expected to have much higher genetic diversity and in situations where the re-introduction site was very heterogeneous or suspected to be slightly different to the donor populations, greater genetic diversity would maximise survival.

4. Results

4.1 Review statistics

The literature search was carried out according to the description above by two reviewers between 21st June 2007 and 9th January 2008. The numbers of articles identified and processed at each stage of the systematic review are given in Table 1. Figures are not available for the number of references identified in the initial search stage; the numbers of 'returns' from many of the search terms was so large as to necessitate the selection of relevant articles by title and article prior to downloading to reference storage software. A total of 168 articles were selected for full text viewing. In addition, information was taken from the Botanical Society for the British Isles Introductions database and Plantlife records on past and current re-introduction attempts. Four practitioners responded to an email questionnaire providing articles or references on re-introduction projects. Of these, three also provided updates on the current status of the species in question. These data are included in further analysis and the reference is cited as a personal communication from the respondent alongside the published article. Several articles could not be acquired and were not included in the study.

Table 1. Number of articles identified at each stage of the literature search and review process.

Stage of review process	Number of articles
Relevant references identified from search of online databases after removal of duplicates and relevance assessment using title	377
References identified from other sources	5
References remaining after relevance assessment using titles, abstracts and full text	168
Articles judged relevant for meta-analysis after full text viewing	67
Relevant articles excluded from further analysis	101

4.2 Description of studies

The systematic literature search generated many thousands of document links using the search terms described earlier. The literature search continued until December 2008; any studies that were identified after that date were not included in further analysis (described below). The search results were filtered for relevance using title and abstracts to generate a document list of 168. The literature search combined with the BSBI and Plantlife databases and IUCN documents generated evidence for the attempted re-introduction of 708 taxa. In many cases the rationale for undertaking the re-introduction was ambiguous or non-existent. This made the decision to include these examples in the dataset difficult but ultimately, the reviewers followed the terminology of the study authors or practitioners coordinating the projects. For example, the IUCN's Re-introduction Practitioners Directory (Soorae and Seddon, 1998) was obtained and species listed within this were deemed to be a true re-introduction despite no other information being available on the majority of species from either the literature or attempted direct contact with the practitioner.

Whilst the literature search generated a great number of taxa undergoing or intended for re-introduction, relatively few of these projects were recorded in enough detail to include in quantitative analysis. The process of selection is described as follows: of the list of 708 taxa, reports of the re-introductions of only 128 taxa satisfied the minimum requirements for inclusion within the meta-analysis. For many of the 128 taxa, re-introduction projects involved translocation of propagules to several sites, from this point onwards we will refer to each site-based re-introduction as an 'attempt'. Once the separate re-introduction attempts had been incorporated into the dataset for meta-analysis, the 128 taxa then represented 304 re-introduction attempts. However, due to the need to make sensible interpretations from the data, attempts were categorised according to whether seeds, juvenile plants or adults plants were used as propagules. In some instances, this information was not available so these were omitted from the dataset. The final number of re-introduction attempts included in the analysis was 301, 47 of which were attempts using seeds, 134 used juvenile plants and 115 used adult plants.

Further analyses of subgroupings of reintroduction attempts are described in subsequent sections of this report. However, subsets of re-introduction attempts are used where information on the analysed parameters were not available.

The 128 taxa and their attempted re-introductions are thought to be broadly representative of the global effort to restore or create populations of threatened plant species (see Appendix 3 for a summary of attempted re-introductions). At a higher taxonomic level, 44 families are represented in the dataset of which Orchidaceae is the most common. Using the Worldwide Fund for Nature classification of major biomes, each re-introduction was assigned to one of 14 biome types; only tundra, cold winter deserts and tropical grasslands and savanna were not represented in the dataset. Re-introductions included in this review have been undertaken in 14 countries across all inhabited continents; however, Africa is very poorly represented. Taxa have distributions ranging from single site endemics to being, at least formerly, extremely widespread across several continents.

The literature search captured reports on true re-introductions, i.e. movement of individuals (seeds or whole plants) to previously extirpated locations and sites within the species' historic range, augmentations of extant but vulnerable populations, and conservation introductions, i.e. to a site outside the known historic range of the species (Table 2). As discussed above (section 3.7.2), the description of re-introduction site status relative to the species' range is often ambiguous due to ill-defined use of the 'range' concept. We accept that in some cases our classification of sites may be erroneous despite making every effort to check site location against published distributions. Effectively this means that our category described in Table 2 as 're-introduction: attempted creation of population within species' historic range' is essentially a blanket term and may contain examples of restoration of extirpated populations where the authors applied a particularly stringent use of the range concept. Additionally, some of these might arguably be classified as conservation introductions; for example, if a population has been created in an area of suitable habitat within the range 'polygon' but where no specific records of species presence exist, some practitioners would classify this as conservation introduction whilst others refer to it as a 'true' re-introduction.

Many studies described their attempt as a trial translocation with the aim of investigating the feasibility of translocations as a conservation tool. In all cases these attempts were under field conditions and the description of 'experimental' or 'trial' used by the author did not result in the attempt being treated differently to 'full' re-introductions in subsequent analysis. Experimental re-introductions often involved the comparison of different treatments which make comparisons problematic; further discussion of the implications of this can be found in section 5.3. From this point onwards all interventions are referred to using 're-introduction' as an umbrella term unless a more specific use of the word is stated.

Table 2 Frequency of taxa associated with intervention types according to reviewer's interpretation of study descriptions. Studies used totalled 67 and incorporated 123 taxa. Categories are not mutually exclusive, some studies describe interventions that can be classified several ways, therefore frequencies do not equate to total number of taxa.

Intervention	**Frequency**
Mitigation of specific threat e.g. development of site supporting extant population	43
Re-introduction: attempted restoration of extirpated population	17
Re-introduction: attempted creation of population within species' historic range	90
Augmentation of extant population	7
Conservation introduction: attempted population creation outwith historic range	7
Experimental (re-)introduction e.g. trial to investigate feasibility of future attempts	5
Experimental (re-)introduction investigating effects of propagule type on success	13
Experimental (re-)introduction investigating habitat requirements	9
Experimental (re-)introduction investigating intervention-related management	21

The time period over which re-introduction projects were monitored varied considerably between studies although the mean time period over which the survival of seed, juvenile and adult propagule types has been reported are very similar – somewhere around 3 years (Table 3). Figure 1 shows range and median time period values for seeds, juvenile and adult re-introductions. One author reported the survival of juvenile plants after only one month since outplanting whilst at the other extreme, the results of a mixed seed and adult plant translocation were recorded by the Botanical Society of the British Isles 32 years after the attempt. It is not possible to predict if a longer monitoring period might have changed the proportion of surviving propagules but it is assumed that the longer the monitoring time period, the more reliable the reported outcome.

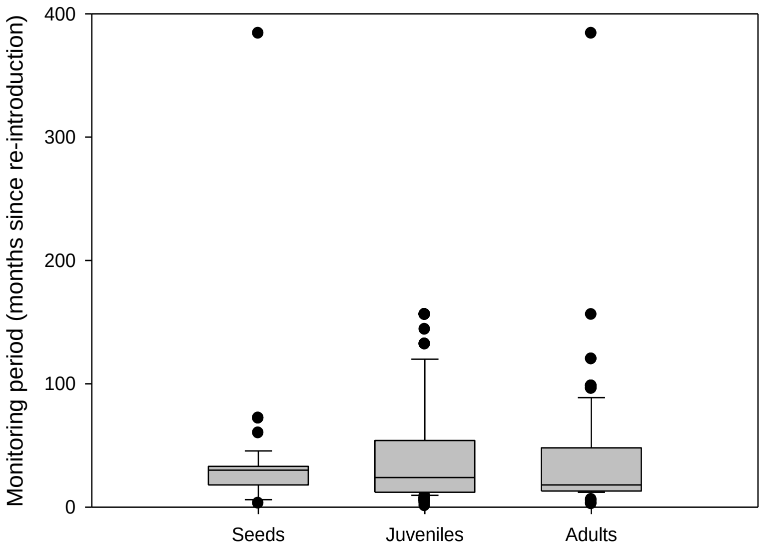

Figure 1. Time period of re-introduction monitoring in months between re-introduction attempt and last survey prior to publication. The boundary of the box closest to zero indicates the 25th percentile, the line within the box marks the median, and the boundary of the box farthest from zero indicates the 75th percentile. Error bars above

and below the box indicate the 90th and 10th percentiles. Black circles represent outlying values.

4.3 Narrative synthesis

4.3.1 Summary of outcome measures

The studies can be summarised in terms of outcomes identified in the search protocol, in this case propagule and population survival (where each attempted re-introduction constitutes a population), attainment of reproductive stages of the lifecycle and achieving *in situ* recruitment of an offspring generation. The latter three measures only require that the authors note survival, reproductive state or recruitment of one individual per attempted creation of a population. Reproductive maturity is evidenced by the presence of flowering or fruiting bodies. Recruitment of an offspring generation includes individuals resulting from vegetative and sexual reproduction. Although these are coarse measures of success, they were identified in the protocol as being key outcomes with which to evaluate re-introductions and similar interventions. They are summarised in Table 3.

Table 3. Summary statistics describing key parameters used to assess effectiveness of re-introductions, 'n' refers to number of attempts categorised by propagule type (seeds, juvenile or adult plants), means are shown ± 1 standard error.

Summary parameters	**Seeds** n = 47	**Juveniles** n = 134	**Adults** n = 115
Mean monitoring period (months)	34.34 ± 7.93	41.16 ± 3.66	36.89 ± 4.23
Monitoring period range from point of re-introduction (months)	3 - 384	1 – 120	2.5 - 384
Mean number of surveys (range in parentheses)	1.38 (1-4)	1.38 (1-5)	2.23 (1-12)
Mean number of propagules	5640.62 ± 2007.51	157.30 ± 30.85	111.17 ± 21.55
Mean population size as a percentage of initial propagule input	4.6% ± 1.4	65.0% ± 4.7	998.5% ± 730.7
Number of attempts to re-introduce annuals	25	2	3
Number of attempts to re-introduce biennials	0	0	3
Number of attempts to re-introduce perennials	22	132	109
Percentage of unsuccessful attempts (extinct at last survey)	36.1%	9.0%	15.7%
Percentage of 'successful' attempts (extant at last survey)	63.8%	91.0%	84.3%
Percentage achieved reproductive maturity	48.9%	18.7%	34.8%
Percentage of attempts where offspring recruited	46.8%	5.2%	20.9%

Re-introductions using seed had by far, the highest mean number of propagules (5640.62 ± 2007.51) and the lowest mean propagule survival at only 4.6% (averaged across all 47 attempts). Seeds carry an inherent risk of mortality and therefore re-introductions using seed are expected to perform worst when measured in this way. However, despite this very low propagule survival and relatively low population survival, a much higher proportion of attempts using seed reach reproductive maturity (48.9%) and produce an offspring generation (46.6%) than using whole plants. This result is partly explained by the high proportion of annuals that are introduced to a site as seed. If the practitioner or researcher sowing the seed has managed to select favourable microsite conditions that allow persistence for just a few months, there is a high probability that the resulting plants can reach reproductive maturity and set seed leading to recruitment of an offspring generation in the following year. Of the attempts using seed that resulted in *in situ* recruitment of offspring, only two were attempts to re-introduce perennials, all others were annual species.

Juvenile plants are defined as those not yet achieving reproductive maturity and include propagules described as seedlings, saplings and in some cases cuttings (depending on the part of the plant), by the study authors. The mean number of propagules used in each attempt is 157.30 (± 30.85); this is much lower than seed-based re-introductions reflecting both the greater resource requirement to produce juvenile plants through propagation and the lowered risk of mortality in individuals that have developed beyond a seed. Propagules of juvenile plants do indeed have a very promising survival of 65.0% implying that they overcome much of the mortality experienced by seed propagules. The juvenile-based projects also have the lowest population extinction and the highest number of extant attempts at the point of the last recorded survey, but whilst the longest mean monitoring period is associated with juvenile introductions, it is still only 41.16 months and it is not possible to say whether this encouraging survival of both propagules and populations might confer longevity over longer timescales. This problem of short monitoring times is a probable explanation for why reproductive measures are so low. The description of only 134 attempts recorded whether reproductive structures such as flowers or fruits were present at the latest census. Of these only 18.2% attempts resulted in individuals producing flowers or fruits, whilst just 5.2% showed *in situ* recruitment.

Adult plants used for re-introductions have the lowest mean number of propagules per re-introduction attempt (111.17 ± 21.55) and the highest mean propagule survival which actually increases by several orders of magnitude compared to seed- or juvenile-based projects. However, this measure of re-introduction success is strongly influenced by attempts to re-introduce *Aldrovanda vesiculosa*, a vegetatively spreading aquatic species which in one extreme case was able to increase from 60 propagules to 50,000 in 6 years (Ademec and Lev, 1999; Ademec, 2005). If that species is removed from the dataset, propagule survival decreases to 84.6% ± 23.9 (1 standard error) and although this is still higher than for other propagule types, the standard error value indicates that a great deal of variance exists in this dataset. Within reported timescales re-introductions using adult propagules have a high success rate in terms of population survival – 84.3% of attempts are still extant at the last survey. Attainment of reproductive maturity and recruitment of offspring were seen in 34.8% and 20.9% of attempts, respectively. Although this is low and might suggest that many attempts do not result in the creation of self-sustaining populations, these are higher than the equivalent figures for juvenile propagules suggesting that within the short timescales reported by study authors, adult plants are further along in reproductive development and therefore a higher proportion have recruited into the next generation.

4.3.2 Population survival over time

Figure 2 shows the proportion of population survival within a timescale reflecting the number of years prior to data analysis in 2009. The bars show how many studies were initiated in a given number of years preceding 2009 and are split between three categories: whether an attempted re-introduction is extant or extinct at the last survey and if they are unknown in outcome. In addition to demonstrating that attempts to re-introduce threatened plants have mainly been undertaken in the last ten years, it shows that, overall, the status of most attempts is unknown because we have not been able to update the outcomes since the authors published their studies. This information is more insightful than showing population survival at last survey alone (Table 3), because it reflects how many cases are of undetermined status and acts as a warning that using the published results without follow-up surveys should be used with caution: they cannot convey whether re-introductions are a reliable tool for mitigating plant declines because timescales for publication are normally much shorter than generational timespans of the species of interest.

The first bar representing our knowledge of the status of all 301 attempts at 6 months since transplanting, indicates that unsurprisingly, most attempts are still extant. However, a small proportion (n = 41) are classed as unknown because the latest survey of these attempts was undertaken within 6 months; many of these studies focussed on *in vitro* propagation methods and reported the success of outplanting only as a final stage of their project. With increasing time prior to 2009, the number of studies that can be included decreases but the ‘unknown’ proportion remains large relative to the extinct or extant categories. An important threshold is crossed at the transition between 5 and 10 years: at 5 years since re-introduction, 46 attempts are still extant whilst 26 have gone extinct; at 10 years, only 20 are extant whilst 30 are recorded as being extinct. It could be argued that at some point between 5 and 10 years, re-introductions go from being a successful intervention to an unsuccessful intervention. However, the proportion of attempts which have an undetermined status is relatively large and conclusions are therefore drawn from a small subset of the potentially available data. The small size of the final bar showing outcome of attempted re-introduction at 25 years reflects the fact that very few re-introductions were initiated more than 25 years ago. At this point the number of extant and extinct attempts are equal but there are only 4 in each category.

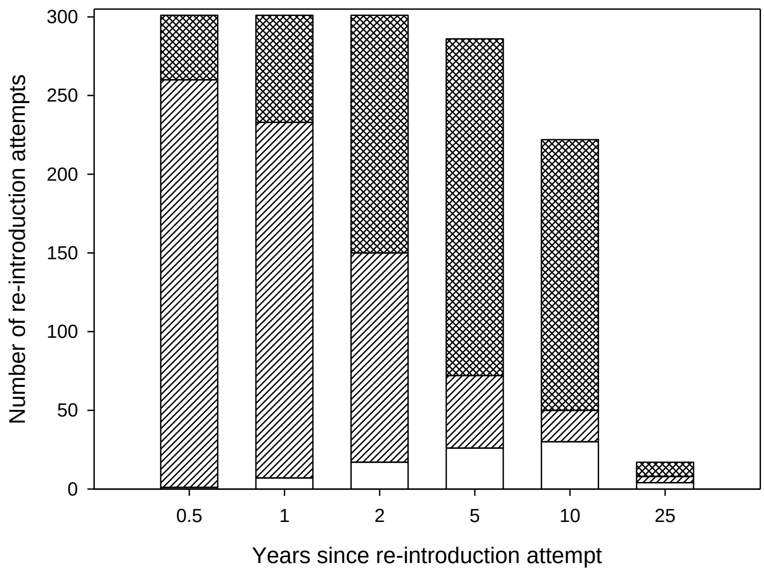

Figure 2. Survivorship of attempted re-introductions over time period since propagules transplanted to site (initial n = 301). Colourless stacks represent failed attempts (all transplants and any resulting progeny dead), diagonally-hatched stacks represent extant attempts, cross-hatched stacks represent attempts of unknown fate. Decreasing total number of attempts reflects how many attempts were undertaken in each time period preceding analysis in 2009.

4.3.3 Explanations for failure

In cases where attempted re-introductions resulted in very few or no surviving individuals, many authors offer explanations for the failure of their projects. Although these are often speculative, the insight gained from practitioners with experience of working with different species in different situations is a valuable component of amassing an evidence-base upon which to inform future use of a technique.

Re-introductions often fail due to unfavourable habitat conditions despite most practitioners selecting the re-introduction site by matching conditions with those associated with extant, wild populations. Specifically, causes such as drought, particularly in the first few years after outplanting (Jusaitis, 2005; Batty et al., 2006), inappropriate disturbance regime including too much and not enough disturbance (Drayton and Primack, 2000; Leonard, 2006 a, b; Maschinski and Dusquenel, 2007) and unsuitable substrate texture (Fiedler and Laven, 1996) have been cited. In some studies where authors deliberately included marginal habitat types, these were unsurprisingly shown to be less suitable for propagule survival (Arnold et al., 2005; Jusaitis, 2005). Competition from invasive plant species confounded several re-introduction attempts (e.g. Jusaitis, 2005: comparison of establishment of propagules in weeded and non-weeded plots); in one case this was because non-native weeds responded more positively to post-translocation management than the target species (Mehrhoff, 1996).

Other common causes for failure are linked to the species' development and reproductive biology including propagules being outplanted at too early a stage in their development (Ruth Aguraiuja, pers. comm.; Batty et al., 2006). In some cases the authors admitted that too few propagules had been introduced to overcome demographic and environmental stochasticities leading to loss of all individuals (Dalrymple and Broome, 2010). In other studies the transplanted individuals might survive for the duration of the reported monitoring period but pollen limitation of flowering adults has been cited as the reason for an absence of subsequent recruitment (Drayton and Primack, 2000).

4.4 Meta-analysis

The pooled risk ratio from all 301 re-introduction attempts is 1.184 (95 % CI 1.147 to 1.223, test of RR = 1: z = 10.40, p < 0.001) indicating that overall re-introductions of threatened plants tend to have high levels of mortality. I^2 (variation in the risk ratio attributable to heterogeneity) was 96.6%. To make further interpretation reliable, re-introductions using seeds, juvenile plants and adult plants were then separated.

4.4.1 Meta-analysis of seed-based re-introductions

The pooled risk ratio of re-introductions using seed is 1.937 (95% CI 1.906 to 1.968, test of RR= 1: z = 80.65 p <0.001) and between study heterogeneity is very high (Heterogeneity chi-squared = 452.32, d.f. = 46, p <0.001, I^2 = 89.8%).

Subgroup analyses were undertaken to investigate whether heterogeneity can be explained by variation in covariate groupings. The analyses indicate that although risk ratios are lower and therefore, re-introduction success is higher where the cause of decline is no longer present, there is a large overlap of confidence intervals (risk ratio 1.911 95% CI 1.867 to 1.955; risk ratio of subgroup with cause of decline still operating at the site 1.937 95% CI 1.906 to 1.968, Figure 3).There is significant heterogeneity within the former subgroup but not the latter (Heterogeneity chi-squared =298.50, d.f.=38, p <0.001, I^2 87.3%). The former includes 39 re-introduction attempts where the cause of decline was removed prior to outplanting; the latter includes only 8 attempts where the original cause of decline was still acting at the site. The low level of heterogeneity may simply be a factor of small numbers of attempts and that five of the eight were from the same study.

Endemicity also explained some variation with regional endemics (i.e. those with sub-continental distributions) experiencing higher mortality than local endemics (risk ratio of regional endemics = 1.974, 95% CI 1.931 to 2.018, risk ratio of local endemics = 1.817, 95% CI 1.718 to 1.915). This does not conform to expectations and apparently has no biological significance as the endemicity was used as an indicator of range size: it was expected that those species with very narrow ranges might have specific habitat requirements making it more difficult to successfully select suitable sites for re-introduction. However, regional endemics are associated with higher mortality than the narrow endemics indicating that assumptions of broader habitat tolerances making for simpler re-introduction site selection may be ill-founded. As practitioners, we might assume more widely distributed species can tolerate more variable conditions but the results from the meta-analysis suggest that we may be overlooking key factors which determine survival and hence, experience low success rates in our re-introduction projects. Threatened species with very narrow ranges might qualify for greater conservation effort resulting in a more rigorous and successful re-introduction attempt.

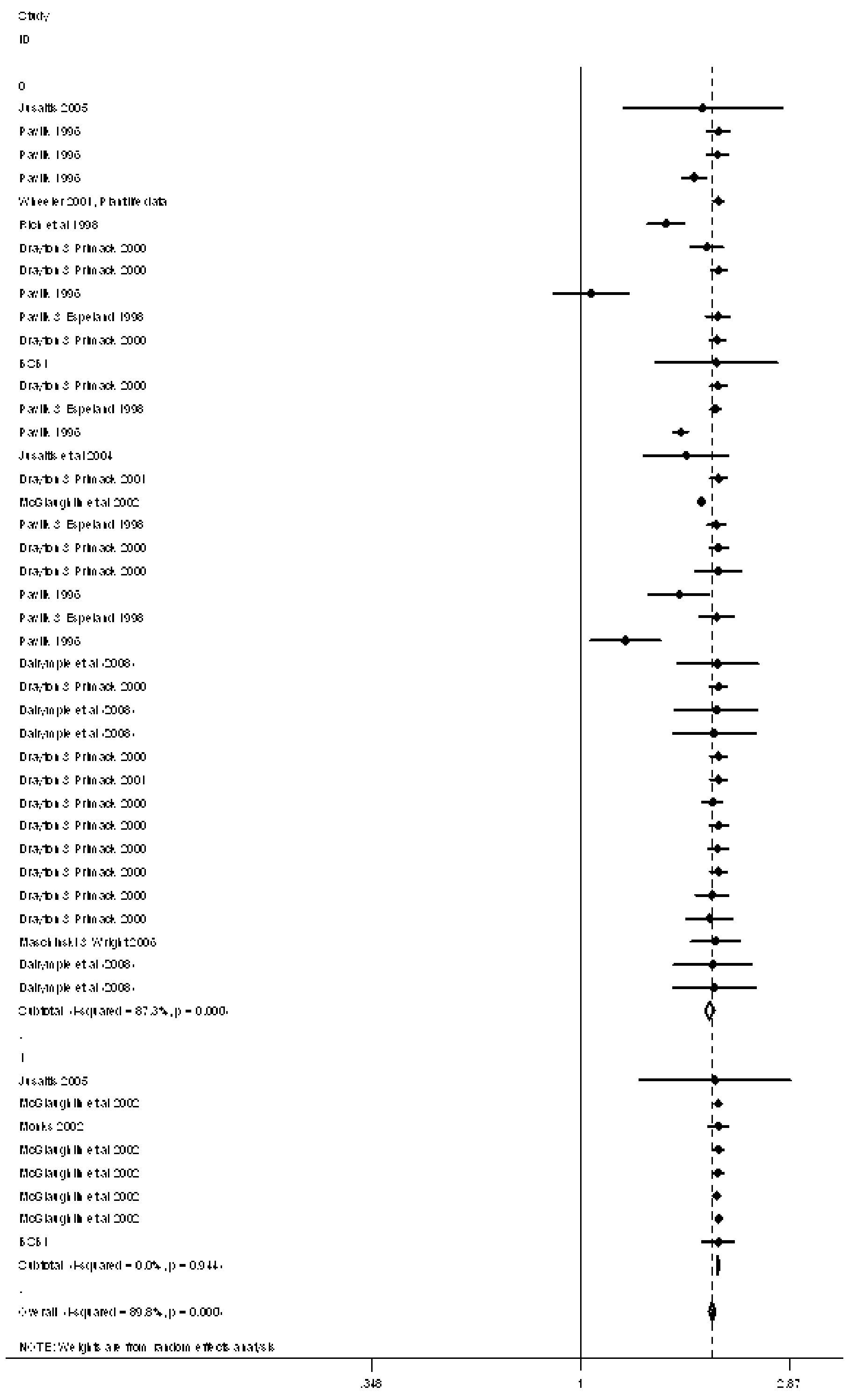

Figure 3 Risk ratios generated from survival of propagules as a proportion of number of seed introduced. Solid boxes represent the individual risk ratios; box size is derived from sample size; error bars are 95 % confidence intervals; open diamond and dotted line indicate the pooled effect size for subgroups generated using random effects meta-analysis. Subgroups refer to whether the cause of decline is still affecting the re-introduction site, '0' = no, '1' = yes.

No other differentiation based on covariate subgroups (IUCN threat category, re-introduction site status relative to species distribution, provenance of propagules or number of donor populations) could be discerned, nor were the proportion of seedlings surviving related to time (Figure 4, p 0.145, SE 0.0035, 47 observations, 10000 permutations). The funnel plot shown in Figure 5 indicates that there is no publication bias as although there are some studies which are relatively small in terms of number of propagules and report higher survival rates, there are very few of these compared to those studies which report very high mortality.

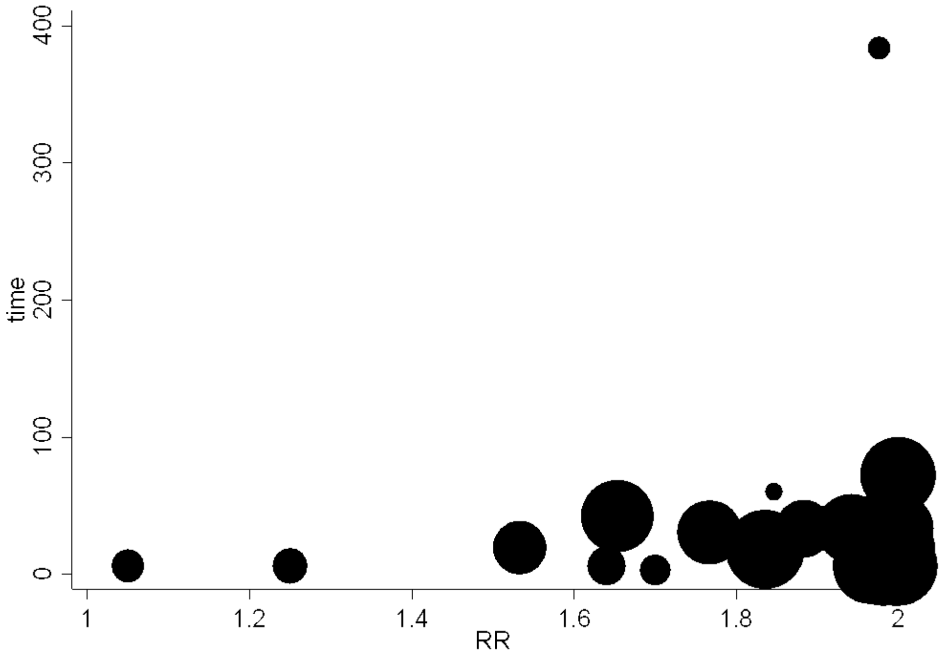

Figure 4 Risk ratio derived from proportion of seed propagules surviving to last survey against time between outplanting and last survey. Size of circle indicates analytical weight in the random effects analysis.

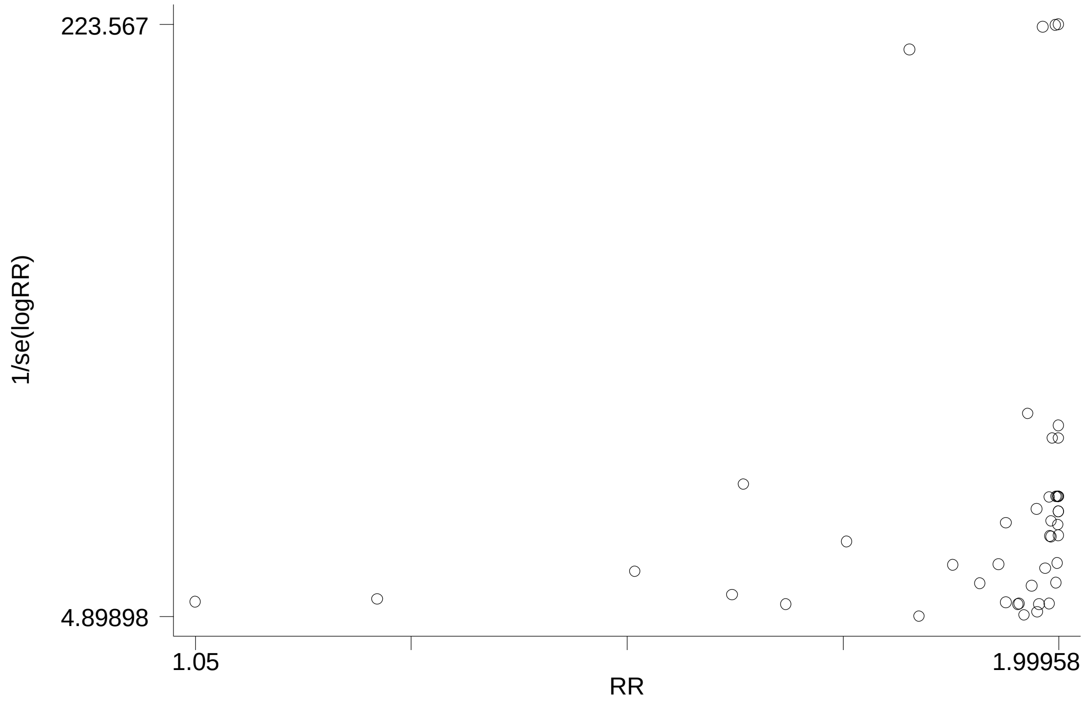

Figure 5 Funnel plot to assess the evidence of publication bias in studies using seed as a propagule for re-introduction attempts. Large studies (with small variance) have high *Y*-axis values, while small studies have low *Y*-axis values. Publication bias may be suspected if small studies reporting low propagule mortality are present but small studies reporting high mortality are absent.

4.4.2 Meta-analysis of re-introductions using juvenile plants as propagules

The pooled risk ratio of re-introductions using juvenile propagules is 0.607 (95% CI 0.525 to 0.701, test of RR= 1: z = 6.79 p = 0.001) confirming that survival of juvenile propagules is much higher than seeds. Between study heterogeneity is very high (Heterogeneity chi-squared = 4613.21, d.f. = 131, p <0.001, I^2 97.2%), multiple factors contribute to this heterogeneity as described below.

The extent of species' distribution contributes to between study heterogeneity: local and national endemics having much higher survival than either global, continental or regional endemics (Figure 6). As with the endemicity of seed-based attempts, this is converse to our expectations but strengthens our hypothesis that re-introductions of species with large natural ranges may be overlooking a fundamental factor in habitat requirements thus explaining lower re-introduction success.

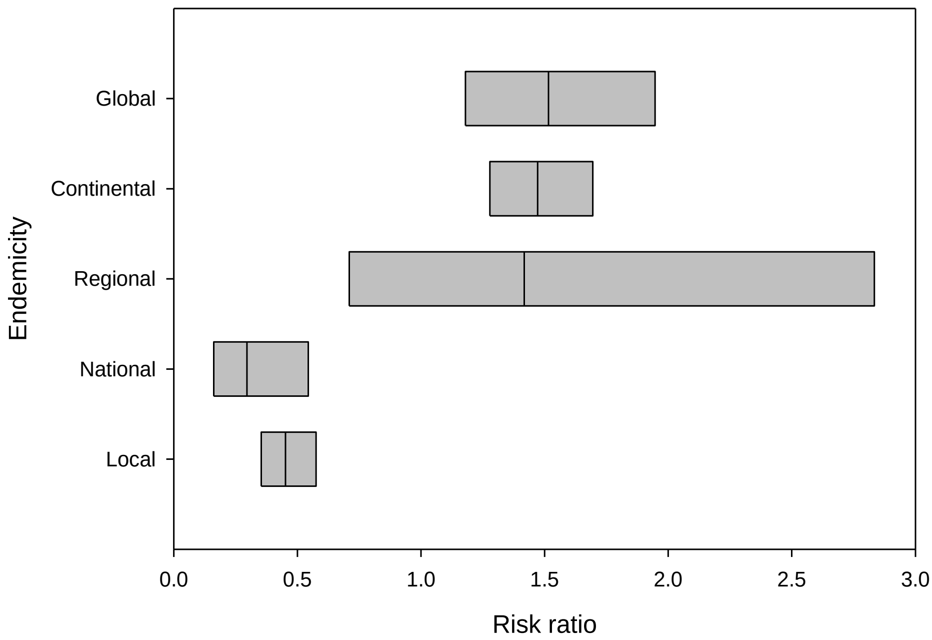

Figure 6 Box plot showing pooled risk ratio (central vertical line in each box) and 95% confidence intervals for each subgroup categorised by level of endemicity. Numbers of attempts in each subgroup varied widely (global n = 8, continental n = 10, regional n = 4, national n = 16, local n = 94).

IUCN threat status seems to explain some heterogeneity but this is not a meaningful result because the subgroup with highest survival are those species which have not been evaluated. Subsequently, this output tells us nothing about how extinction risk might be related to re-introduction success. In addition, this subgroup accounts for 121 of the 134 attempted re-introductions using juvenile propagules. Any other relationship which may be discerned between threat level and re-introduction success is therefore statistically dubious due to the very low number of attempts in each subgroup.

The status of the re-introduction site relative to the species' range is noteworthy but again, this is because the analysis output contradicts hypothesised outcomes. We predicted that attempts to establish a population at a site which was outside the historic range would incur a higher mortality of propagules because the habitat was more likely to be unsuitable. Instead, projects that have used sites outside the species' historic range have a remarkably low risk ratio: the pooled risk ratios of these attempts is 0.177 (95% CI 0.053 to 0.588, n = 7). Compare this with re-introductions to sites confirmed as supporting previously extant populations (risk ratio = 0.827, 95% CI = 0.646 to 1.059, n = 23) and those within the historic range of the species show very similar success levels in terms of propagule survival (risk ratio = 0.665, 95% CI = 0.578 to 0.763, n = 99). It should be emphasised that the confidence intervals of all subgroups overlap and the number of attempts in each subgroup is very unbalanced. Many more studies detailing the results of attempts to move a species outside its historic range would be needed to strongly suggest that this intervention is more successful than true re-introductions.

Of the 134 attempts to re-introduce threatened species using juvenile plants, we could reliably only classify 66 attempts according to whether a single donor or multiple donor populations were used. This is in part because very few authors describe where the original material was sourced if the propagules for re-introduction were raised in *ex situ* facilities. Meta-analysis of the two subgroup's risk ratios show that using multiple donor populations leads to higher survival (risk ratio = 0.253, 95% CI = 0.129 to 0.498) whilst relying on propagules from only one population has a higher mortality (risk ratio = 0.472, 95% CI = 0.323 to 0.691). However, given that the confidence intervals overlap between these subgroups, we cannot conclusively make any recommendations that practitioners use single or multiple donors. Instead it highlights the need for more research into appropriate strategies for re-introduction accounting for ecological similarity of donor and re-introduction sites, reproductive biology of the species of concern, and isolation and subsequent genetic impoverishment of existing wild populations.

Subgroups based on whether the cause of decline remains present and propagule provenance in terms of wild- or *ex situ*- sourced do not cause any significant splits in the data. Time has an impact with shorter timescales correlated with higher survival as might be expected (p<0.003, SE 0.0006, 132 observations, 10000 permutations, figure 7). The long term efficacy of introducing juvenile plants therefore requires further exploration.

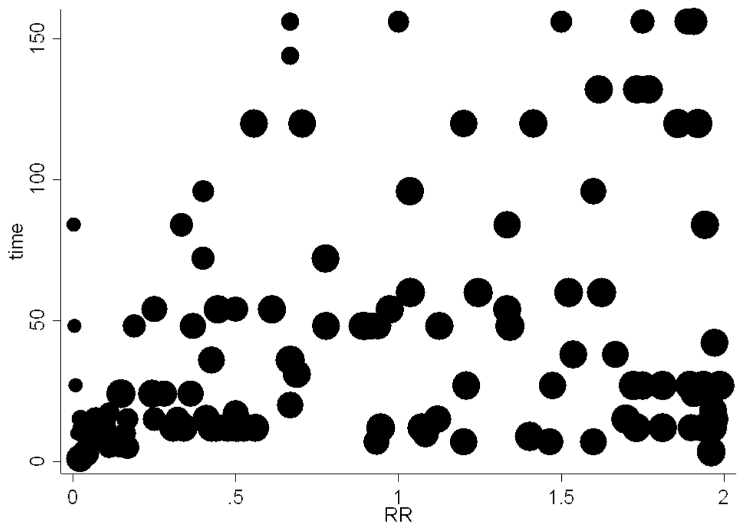

Figure 7 Risk ratio derived from proportion of juvenile propagules surviving to last survey against time between outplanting and last survey. Size of circle indicates analytical weight in the random effects analysis.

The funnel plot shown in Figure 8 indicates that there is no publication bias towards small studies reporting high survival rates, as evidenced by the presence of studies that are also small but report high mortality. However, it does show unusual asymmetry of points with clustering in the bottom left corner meaning that the risk ratios generated for juvenile re-introductions my be overly influenced by small studies reporting excellent survival rates.

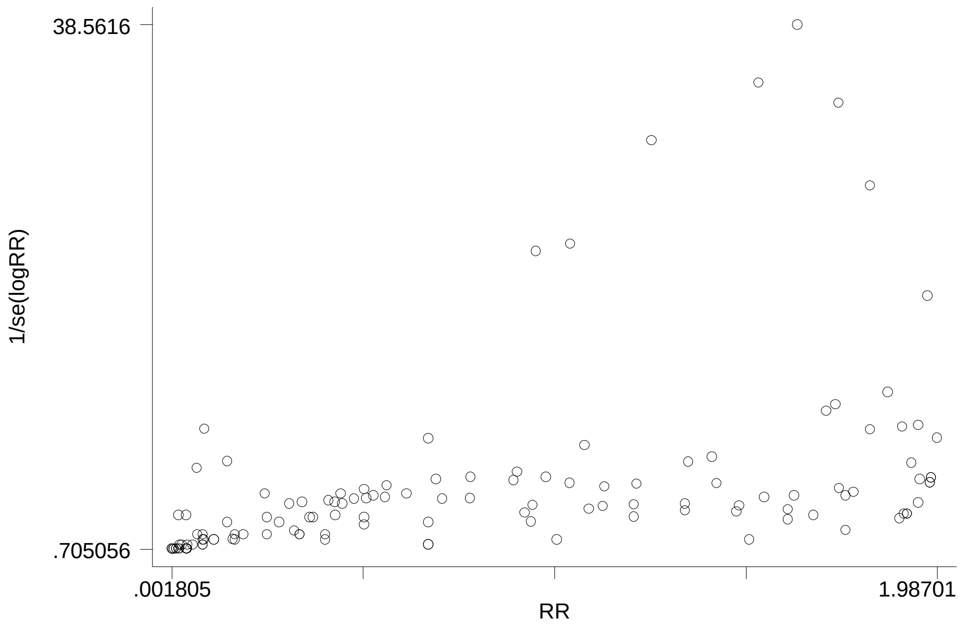

Figure 8 Funnel plot to assess the evidence of publication bias in studies using juvenile plants as a propagule for re-introduction attempts. Large studies (with small variance) have high *Y*-axis values, while small studies have low *Y*-axis values. Publication bias may be suspected if small studies reporting low mortality are present but small studies reporting high mortality are absent.

4.4.3 Meta-analysis of re-introductions using adult plants as propagules

The pooled risk ratio of re-introductions using adults is 0.747 (95% CI = 0.651 to 0.856, test of RR= 1: z = 4.19 p =0.001) and between study heterogeneity is high (heterogeneity chi-squared = 2621.24, d.f. = 121, p <0.001, I^2 95.4%). The risk ratio and confidence intervals suggest that propagule survival is slightly worse than that of juvenile-based attempts but not significantly different. This result demonstrates the advantage of using this analysis in addition to arithmetic means of survival shown in Table 3. The generation of pooled risk ratios more accurately summarises expected propagule mortality by downweighting the extremely high vegetative recruitment reported in the *Aldrovanda vesiculosa* re-introduction programme (Ademec and Lev, 1999; Ademec, 2005). Overall, whole plant re-introductions have much higher propagule survival than seed-based attempts as is expected for reasons discussed above.

The heterogeneity between studies is not significantly related to removal of cause of original decline prior to attempted re-introduction, re-introduction site status relative to species' range (although, as with juveniles there is a trend suggesting that re-introductions outwith historic range are associated with lowest mortality), single *vs.* multiple donor populations or provenance of propagules (direct translocations *vs.* cultivated *ex situ*).

Subgroups based on levels of endemicity do not show a clear pattern as that of juvenile propagules. The risk ratios and confidence intervals overlap for all groups and although those attempts using species which have continental-scale ranges are associated with higher mortality than all other levels of endemicity, this is not conclusive enough to state that cosmopolitan species are associated with significantly higher mortality. Similarly, IUCN threat assessment status appears to explain some of the variation within the dataset but only 10 attempted re-introductions using adult propagules have been assigned threat status so it would be misguided to conclude anything from that output.

Time may matter as shorter timescales are correlated with higher survival as might be expected (p<0.001, Sep 0.0001, 122 observations, 10000 permutations, Figure 9). However, the plot suggest that this relationship is not strong, and may be sensitive to outliers. The funnel plot in Figure 10 indicates that there is no publication bias towards studies that report positive results although a similar pattern of asymmetry can be seen for juvenile plant-based re-introductions.

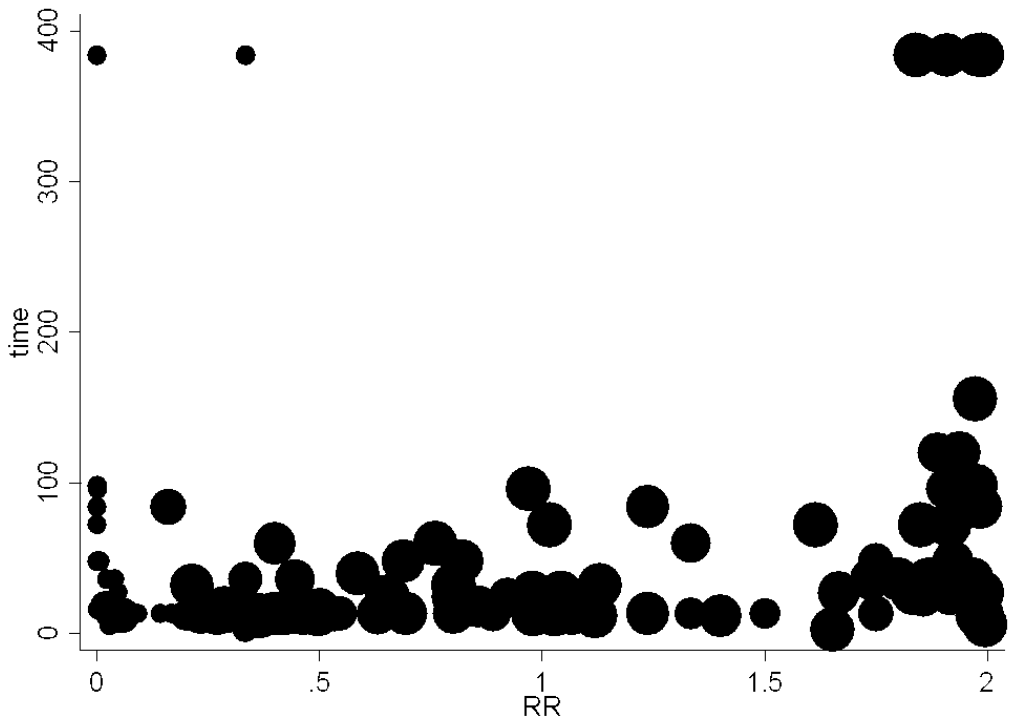

Figure 9 Risk ratio derived from proportion of adult propagules surviving to last survey against time between outplanting and last survey. Size of circle indicates analytical weight in the random effects analysis.

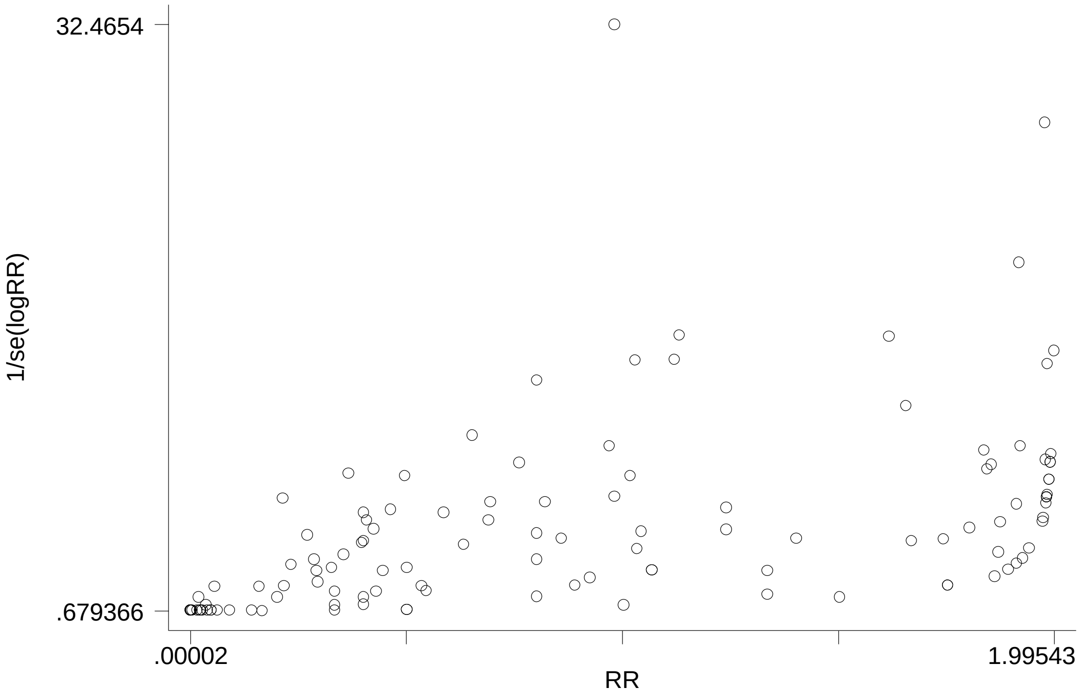

Figure 10 Funnel plot to assess the evidence of publication bias in studies using adult plants as a propagule for re-introduction attempts. Large studies (with small variance) have high *Y*-axis values, while small studies have low *Y*-axis values. Publication bias may be suspected if small positive studies are present but small negative studies are absent.

4.5 Outcome of the review

The comparative effectiveness of re-introductions can be judged in terms of propagule survival, population survival and recruitment of *in situ* progeny to the population. Propagule survival in each attempted re-introduction is the simplest way of comparing across the range of taxa, habitats and threat that this review has presented. It indicates that seed survival is generally extremely low and practitioners should be aware of typical germination rates of species in wild populations. This knowledge would allow practitioners to introduce enough seed to overcome expected mortality loss. Whole plant re-introductions have a much higher survival than seeds according to both arithmetic means of survival and risk ratios generated by meta-analysis. This is encouraging and might be taken as a recommendation to use whole plants rather than seed in re-introductions. However, this masks the wide range of variation in the success of whole plant re-introductions. In particular, the translocation of vegetatively propagating species has positively affected the average population size expressed as a percentage of propagule input when using whole adult plants. However, recorded population survival over longer timescales (10-25 years) indicates that re-introductions are as likely to fail as still be extant (Figure 2). This apparent discrepancy reflects the fact that pooled measures of propagule survival are combined at the point of last reported survey but this timescale is very variable. It also indicates that these measures are often reported before a full generation of a species might have elapsed; even for annuals, the mean reporting monitoring period will only have allowed three generations of plants to emerge and for those species where a long-lived seed bank is an important demographic feature of a typical population, the full demographic dynamics will not be apparent for some years to come. Recruitment of offspring from the introduced propagules is the final crucial outcome to ensure population longevity. Again, the short time periods reported in the reviewed studies mean that we cannot confidently judge the effectiveness of re-introductions using this metric other than to say that in the majority of attempts, recruitment has not been attained. Assuming that whole plant re-introductions maintain modest levels of mortality we might expect the proportion of re-introduction attempts resulting in recruitment to increase. This will occur as more individuals are able to reach reproductive maturity and progeny will be incorporated into propagule survival measures if further surveys are undertaken. The regeneration niche may be different to those supporting adult plants so the future success of re-introductions will depend on whether recipient sites meet the conditions for survival of all stages of the species' life cycle.

5. Discussion

5.1 Evidence of effectiveness

The evidence-base identified through this review is unfortunately, inadequate to properly judge the effectiveness of re-introductions of threatened plants as a tool for mitigating declines. The reasons for this inadequacy include the limited monitoring period reported in the reviewed literature and heterogeneity in the levels of site management associated with the preparation of and after each attempted re-introduction. This aspect of potential heterogeneity was identified in the protocol but was not extracted because we could not have included measures of relevant variables without breaking the dataset into very many small divisions. Consequently, there can be no allowance made for the exclusion of herbivores, application of water, continuation or cessation of fire regimes, the deliberate testing of marginal habitat conditions or any of the other types of intervention-related management variables. We have little doubt that in many cases these activities make a massive difference to the success of the project and in hindsight, we realise that the inclusion of some measure of management related to the intervention would have been valuable. However, even with the benefit of hindsight, it would have been very difficult to include such activities in the subgroup analyses due to the enormous variation of types of management and differences in time over which re-introduction-related management was maintained.

As this review cannot conclusively comment on the effectiveness of re-introductions, it is intended instead that it might provide guidance on improving the design and reporting of re-introduction projects so that a much needed evidence-base can be established. Recommendations are made in sections 6.1 and 6.2.

5.2 Review limitations

The main limitations of this review are the short timeframe over which published studies report the results of their re-introduction, the lack of our ability to assess the feasibility studies and preparatory stages of each attempt, the absence of a kappa statistic to indicate lack of bias when choosing to include studies, and our decision not to incorporate different site management into the meta-analysis.

A particularly powerful analytical tool which was not possible to use with the available data is *time to event* analysis, sometimes also called *survival* analysis. In this case the 'event' in question is death of individual propagules and can be used to investigate whether covariates (e.g. intervention-related management or properties of the species or system) can prolong the survival of each propagule. This requires that individual propagule survival over time is recorded; however, it is very unusual for re-introduction practitioners to record anything other than aggregated propagule survival, i.e. the proportion of individuals surviving at a given timepoint. To make this analysis possible practitioners must be able to identify certain individuals and follow their progress at regular time intervals. Whilst acknowledging that it is practically very difficult to follow seed-based re-introductions at an individual level, we recommend that new re-introduction projects identify a representative sample of propagules that can be tagged and monitored individually and that a sample is identified in each treatment and site incorporated in the project. This would allow multilevel modelling techniques to be applied to the data and identify whether factors at treatment-, site- and species-level conferred an advantage in terms of propagule survival.

As with any literature search which relies on a body of evidence which is, at least partly, classed as grey literature, this review has not been able to acquire data suitable for meta-analysis for the majority of the 708 taxa thought to be the subject of re-introduction attempts. It is assumed therefore, that data pertaining to an unknown number of those taxa exists as unpublished and possibly, unwritten reports. In addition there is likely to be much more information on the taxa that have been included in the meta-analysis and our classification of unknown status in terms of population survival overtime should be interpreted as being unknown to the reviewers.

The systematic review protocol included a description of study quality assessment with the intention that it be applied to evaluate the relative contribution of each study to the review findings. The scoring system placed high value on generating meaningful comparisons between and within the re-introduction projects reviewed. For example, a study which describes simultaneous outplantings of the same number of propagules sourced from the same location at five different sites and monitored for over ten years would score very highly. However, a feature of re-introductions is that they are often responsive to biological and socio-economic conditions. For this reason it is quite typical that a study might report re-introductions over subsequent years, at different sites of varying potential suitability and that projects may be interrupted by uncooperative landowners or discontinued funding. So few of the studies found had any sort of meaningful replication that we decided to use the simplest measure of effectiveness in order to maximise inclusivity whilst acknowledging that this diminished analytical power. Due to reducing all studies down to the lowest common denominator, i.e. proportional propagule survival over time, we can place little confidence in the capacity of the subgroup analyses to provide causal inferences. However, due to the large number of attempted re-introductions that can be included, we believe that the findings of the subgroup analyses make for useful foci for future debate and research.

We did not complete a 'kappa' analysis of agreement between reviewers as to the inclusion of studies within the review. Two persons were involved in this process and worked to an agreed protocol as described above. The inability to quote a kappa statistic weakens the review as the selection of studies for inclusion is a potential source of unquantifiable bias.

6. REVIEWERS' CONCLUSIONS

6.1 Implications for threatened plant species management

Re-introductions can be judged using a number of outcomes and in this review we have focused on proportional survival of propagules used in each re-introduction attempt. This has provided a comparable measure of effectiveness and allowed us to attempt to link relative re-introduction success to traits associated with the target species and intervention. However, in terms of priorities for practitioners, including those who have a strategic responsibility for preventing biodiversity loss, the key outcome of interest is whether re-introductions can deliver viable populations which will persist over the long-term without intensive management. Unfortunately, the available evidence is insufficient to enable conclusive statements to be made on the effectiveness of creating self-sustaining populations of threatened plants. This is due to the recent initiation of many re-introduction projects and the short timescales over which they have been monitored; in combination these factors have resulted in premature reporting of outcomes which may be misleading.

In terms of future management of threatened plants, we strongly recommend that existing re-introduction projects are monitored at regular intervals over long time periods and these results are made available to others in the conservation community. Indeed, some of the practitioners who contributed to the protocol development hoped that this review might be used to support calls to improve recording and reporting of plant translocations.

Subgroup analyses were used to investigate key questions associated with the use of, and debate surrounding, the efficacy of threatened plant re-introductions. Due to the problems associated with multiple testing it would be misguided to use these results to explain reasons for re-introduction success or failure. However, the lack of any convincing subgroup separation for many of the selected parameters suggests that they might not be as crucial to re-introduction success as published guidelines might suggest. Of course, we are not recommending that re-introduction guidelines should be ignored in future projects, but our results suggest that empirical evidence on key aspects of re-introduction attempts is lacking. Consequently, we recommend that re-introductions are recorded in a standardised format which would more effectively document the different intervention-related factors which introduce heterogeneity into the dataset which cannot be extracted using the methods presented in this review. In addition, this would address the need for formalised recording of plant translocations which is currently absent in many countries where re-introductions are taking place.

Attempts have already been made to encourage the adoption of stringent standards in using re-introductions. One example is a set of guidelines prepared by the North Carolina Plant Conservation Program Scientific Committee. These are the most detailed and usable guidelines we discovered and the level of detail of knowledge which is required by this document should be adopted as a minimum in every re-introduction attempt. Of particular merit in light of our findings are the requirements for information on reproductive biology, taxon site requirements, donor site information and re-introduction/translocation site information. However, the guidelines still do not specifically ask practitioners to elucidate the exact mechanism of decline and rely instead on broad reasons for rarity such as habitat loss or degradation. Therefore, there is still potential for practitioners moving plant propagules to overlook unidentified threats. If the donor site is no longer supporting a viable population but due to demographic lags, the decline is not discernable, it would be misguided to attempt a re-introduction to a recipient site that was selected simply because it was similar to the donor site. It seems likely that the evidence base to inform re-introduction decisions will improve if more stringent guidelines are adopted by relevant bodies such as the IUCN. The IUCN does of course, already have guidelines for the translocation of threatened species (IUCN, 1998) but these are not specific to plant-based programmes.

6.2 Implications for research

The evidence presented in this review is correlative not causal; there may be alternative explanations for the low success rate of re-introduction attempts. In an area of ecological investigation which is so closely tied to a particular problem, in this case threatened plant management, the distinction between implications for management and research is arguably an artificial one. With this in mind, we recommend that the correlative relationships in the above section be the focus of future research on re-introductions. We have divided our recommendations into two sections recognising that firstly, the use of re-introductions needs to be better recorded and evaluated, and secondly, that the debate on finding suitable habitat space for our threatened plant species will increasingly revolve around ideas of assisted colonisation.

The first area of research is aimed at improving the quality of re-introduction attempts in order to both improve the implementation of the technique, and our ability to synthesise and evaluate its use. Primarily, we would like to be able to better judge the quality of preparatory stages of re-introductions in order to use this as an assessment of data quality. The reporting of feasibility studies is one of the most variable areas in terms of detail given. We have no doubt that practitioners are conducting habitat surveys and that the reproductive biology of their target species is well described. However, we rarely know whether the recipient sites have been simply ‘donor-matched’ or if full surveys have been undertaken. These surveys need to identify population growth and decline across an appropriate proportion of the species’ range and link those to current occupancy and recent environmental change in order to determine the actual mechanism behind decline. Dalrymple and Broome (2010) applied part of this process retrospectively and found that supposedly ‘core’ habitat was marginal in terms of the species’ climatic tolerances possibly explaining relative success of several re-introduction attempts. Post-intervention monitoring must also be improved so that hierarchical analyses can be used. These multi-level frameworks would ideally follow individual plant survival and allow mortality curves to be constructed and compared within site-, study- and species-related parameters. This is in part being attempted by US researchers using data from the Center for Plant Conservation Reintroductions Database to infer re-introduction success but identification of reliable site comparisons has been limited despite rigorous data extraction (Matthew Albrecht pers. comm.).

Assisted migration or assisted colonisation is currently the focus of a largely speculative debate in the conservation literature. It refers to the deliberate movement of species to sites known to be outside the species' historic range and has been identified as a possible management option for allowing threatened species to cope with shifting climatic 'envelopes' whilst overcoming dispersal limitations and barriers. The experience generated by practitioners using re-introductions has obvious applicability to the assisted colonisation debate. In particular, the low success rate of re-introductions should be taken as a strong warning that translocating propagules to currently unoccupied sites is a high risk conservation strategy for reasons we do not fully understand. The addition of further levels of uncertainty associated with novel habitats and predicted climate scenarios suggests that assisted colonisation should not be used to mitigate for habitat loss. Assisted colonisation does however, require the level of detailed monitoring and analysis that is recommended for adoption in future re-introduction projects; it seems likely that many re-introductions have failed because the habitat selection or demographic assessment of wild populations has not been sufficiently detailed. We suggest that the distinction between re-introductions and assisted colonisation is becoming increasingly artificial given that we are already experiencing environmental changes making part of the historic range unsuitable as re-introduction recipient sites. We consequently urge conservation practitioners and researchers to adopt stringent scientific protocols that incorporate detailed and recent habitat evaluations of donor populations and recipient sites, demographic monitoring of wild populations, and monitoring of translocations that allows time to event analysis to discern causes for propagule death linked to species-, site- and intervention-related covariates.

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8. Potential Conflicts of Interest and Sources of Support

No potential sources of conflict declared. The project is supported by the British Ecological Society through an 'Ecology Into Policy' grant, number 921/1146.

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10. Appendices

Appendix 1. Reference list of studies used in meta-analysis

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APPENDIX 2. SUPPLEMENTARY SOURCES OF INFORMATION USED TO IDENTIFY STUDIES AND PROVIDE DATA FOR SUBGROUPINGS WITHIN META-ANALYSIS

Australian Government Department of the Environment, Water, Heritage and the Arts, Species Profile and Threats Database; available at: http://www.environment.gov.au/cgi-bin/sprat/public/sprat.pl

The Botanical Society of the British Isles Introductions Database (unpublished), compiled by David Pearman, Kevin Walker and Alex Lockton

Center for Plant Conservation, National Collection of Endangered Plants profile search available at: http://www.centerforplantconservation.org/collection/NationalCollection.asp

Encyclopaedia of Life available at: http://www.eol.org

Integrated Taxonomic Information System available at: http://www.itis.gov

IUCN Re-introduction Practitioners Directory

National Red Lists hosted by the Zoological Society of London available at: http://www.nationalredlist.org/site.aspx

UK Biodiversity Action Plan, Species Action Plans available at: http://www.ukbap.org.uk

United States Department of Agriculture, Agricultural Research Service, Germplasm Resources Information Network, GRIN Taxonomy for Plants available at: http://www.ars-grin.gov/

United States Department of Agriculture, Plants Database, plant profile search used to identify taxonomic, growth form and distribution data; available at: http://plants.usda.gov

APPENDIX 3. DATA EXTRACTION SUMMARY FOR ALL STUDIES INCLUDED IN META-ANALYSIS

Descriptor categories	Variables included in analysis	Number of attempts for which data attained	Data categories and frequency of attempts in each category (n in parentheses)
Species descriptors	Species name	301	123 taxa included, many associated with multiple attempts or sites. Taxonomic designations of authors were used; errors have not been corrected even when suspected.
	Life cycle	301	Annual (31), biennial (3), perennial (267); some species show mixed strategies, most typical strategy recorded here.
	Biome (data taken from WWF Atlas, site of re-introduction or region of concern used to determine biome where species occurs across several regions).	301	Tundra (0), temperate needleleaf forest (4), temperate broadleaf forest (155), temperate grasslands (1), cold winter deserts (0), evergreen sclerophyllous forest, scrub or woodland (40), tropical grasslands and savanna (0), warm deserts and semi-deserts (4), tropical dry or deciduous forest or woodland including monsoon forests (2), sub-tropical and temperate rainforests or woodlands (45), tropical humid forests (14), mixed mountain and highland systems with complex zonation (15), mixed island systems (7), river and lake systems (14).
	Endemism	298	Global (36), continental (46), regional (84), national (36), local (128), site (4).
	Cause of decline (many species have > 1 reason behind declines).	527	Urban and industrial development (39), agriculture when not specifically grazing (96), competition from other plants including invasives, alien or native (50), grazing including stock, goats, rabbits and native herbivores (38), fire (5), climate change (9), over-exploitation or collection (37), habitat loss when not specifically any other reason (101), flooding for reservoir construction or other water course engineering including draining (67), succession including disturbance suppression (67), disease (15), pollution (1), trampling or erosion (2).
	Cause of decline present at reintro site?	235	No, primary threats causing species' decline have been prevented from operating at this site (190) or, yes, primary threats still present (45).
	IUCN threat level assessed using the IUCN Red List 2009.1.	301	EW = extinct in the wild (0), CR = critically endangered (5), EN = endangered (0), VU = vulnerable (3), LR = lower risk (9), LC = least concern (5), NE = not evaluated (279).
Intervention descriptors	Site designation	155	No site designation (14), designated site (141).
	Country	301	14 countries
	Status of site within distribution	278	Previously extant site (39), within historic range (228), outwith historic range (11).
	Provenance	292	Ex situ (167), direct translocation from wild population (102).
	Single or multiple donor populations	140	Single donor (100), multiple donor populations (40).
	Life stage of propagules	299	Seed (47), juvenile (132), adult (115), mixed (5).
	Number of propagules	301	
	Monitoring period (months)	301	Number of months between re-intro and last reported survival date.
Outcome: abundance	Surviving individuals	301	Taken from last reported survey and includes progeny.
	Number of time points	301	Does not include time 0 when re-introduction undertaken, i.e. time points = number of surveys since translocation. Mean = 1.69 ± 0.09 (1 s.e.).
Outcome: recruitment	Reproductive potential	116	Yes, individuals with reproductive structures (89)
	Recruitment	66	Recruitment reported (55), no recruitment (11).
	If recruitment evident, vegetative or sexual?	55	Vegetative (12), sexual (42), mixed 1).

APPENDIX 4. SUMMARY OF INTERVENTIONS FOR ALL STUDIES INCLUDED IN META-ANALYSIS

Species	**Family**	**Country**	**Life history**	**Intervention**	**Reintroduction type**	**Propagule type**	**Year**	**Reference**
Abronia umbellata ssp. *breviflora*	Nyctaginaceae	USA	ann	Seeds from wild populations sown on suitable habitat within extant range.	intro. within	seed	1995	McGlaughlin et al., 2002
Acacia aprica	Mimosaceae	Australia	per	Attempted seed reintroduction followed in subsequent year by seedlings to less degraded site.	intro. within	seed	1998	Monks, 2002
Acacia cretacea	Mimosaceae	Australia	per	Two augmentations of habitat fragment containing mature individuals. Seedlings re-introduced into combination of exclosures to omit stock and rabbits.	aug.	juvenile	1992	Jusaitis, 2005
Acacia whibleyana	Mimosaceae	Australia	per	Seeds and seedlings sown into replicated weedy and weed-free plots.	intro. within	juvenile	1996	Jusaitis, 2005
Acanthomintha duttonii	Lamiaceae	USA	ann	Nutlets sown over 4 subsequent years and demographical variables of re-introduced population compared to natural population.	intro. within	seed	1991	Pavlik and Espeland, 1998
Acianthera saundersiana	Orchidaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Acianthera sonderana	Orchidaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Aechmea calyculata	Acanthaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Aechmea recurvata	Bromeliaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005

Agrimonia incisa	Bromeliaceae	USA	per	Seedlings translocated into fire research plots.	trans. intro within	juvenile	1998	Glitzenstein et al., 2001
Aldrovanda vesiculosa	Droseraceae	Czech republic	per	Vegetatively propagating perennial translocated to fishpools within historic range.	intro. within	adult	1995	Adamec and Lev, 1999; Ademec, 2005
Amorpha herbacea	Fabaceae	USA	per	Two attempts to translocate species: first site is a protected area within historic range, second site is a restored endemic pine rockland community, outside historic range by 25km.	intro. within, intro. out.	adult	1995	Wendelberger et al., 2007
Amsinckia grandiflora	Boraginaceae	USA	ann	Nutlets sown into sites within historic range and subjected to treatments to investigate effect of removing grass cover.	intro. within	seed	1989	Pavlik, 1991; Pavlik et al., 1993; Pavlik, 1996
Antennaria flagellaris	Asteraceae	USA	per	U.S. Bureau of Land Management investigation into the potential of transplantation as a future mitigation action.	trans.	adult	1983	Fiedler and Laven, 1996
Apium repens	Apiaceae	UK	per	*Ex situ* stock introduced to two sites.	intro. within	adult	1996	McDonald and Lambrick, 2006
Aquilegia canadensis	Ranunculaceae	USA	per	Experimental comparison of propagule type (seeds, seedlings, adults) and for seeds only, the effect of sowing into dug or undisturbed ground.	intro. within	seed	1994	Drayton and Primack, 2000
Aralia racemosa	Araliaceae	USA	per	Experimental comparison of propagule type (seeds, seedlings, adults) and for seeds only, the effect of sowing into dug or undisturbed ground.	intro. within	seed	1994	Drayton and Primack, 2000
Argusia argentea	Boraginaceae	Australia	per	Seedling establishment trial on West Island, Ashmore Reef	reintro.	juvenile	1999	McDonald, 2005
Aristida beyrichiana	Poaceae	USA	per	Seedlings outplanted into fire treatment plots at three sites, dry, mesic and wet.	intro. within	juvenile	1993	Glitzenstein et al., 2001
Aster linosyris	Asteraceae	UK	per	Mixture of seeds and plants used for translocations to 3 sites.		adult		BSBI Introductions Database

Barbosella cogniauxiana	Orchidaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Billbergia nutans	Bromeliaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Bletia urbana	Orchidaceae	Mexico	per	in vitro propagation and outplanting	intro. within	juvenile	1986	Rubluo et al., 1989
Brachycome muelleri	Asteraceae	Australia	ann	Seeds and seedlings translocated to two sites, one within existing wild population (augmentation), the other nearby but not known to have previously supported the species.	aug., intro. out.	seed	1996	Jusaitis et al., 2004
Caladenia arenicola	Orchidaceae	Australia	per	Seedlings and tubers propagated from *ex-situ*collections were translocated to field sites.	intro. out	juvenile	1996	Batty et al., 2006
Caltha palustris	Ranunculaceae	USA	per	Experimental comparison of propagule type (seeds, seedlings, adults) and for seeds only, the effect of sowing into dug or undisturbed ground.	intro. within, trans.	seed	1994	Drayton and Primack, 2000
Campylocentrum burchellii	Orchidaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Capanemia micromera	Orchidaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Capanemia superflua	Orchidaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Carex vulpina	Cyperaceae	UK	per	Supplementation of single individual using *ex situ* propagated seed from that plant.	aug.	juvenile		Porley, 2005
Cerastium nigrescens	Caryophyllaceae	UK	per	In 1995 some 600 seeds scattered over an area of bare, gravelly serpentine debris in about the place where F.J. Hanbury may have seen it in 1894.	reintro.	seed	1995	BSBI Introductions Database

Chamaesyce skottsbergii var. *Skottsbergii*	Euphorbiaceae	USA	per	Translocation of nursery-grown plants as mitigation for impending destruction of extant population due to industrial development.	intro. within	adult	1979	Mehroff, 1996
Cirsium pitcheri	Asteraceae	USA	per	Re-introduction using detailed autecological surveys to determine suitable habitat within Illinois Beach Nature Preserve.	reintro.	juvenile	1991	Bowles et al., 1993
Cirsium tuberosum	Asteraceae	UK	per	Four vegetatively propagated and two plants grown from seed translocated to site matching last known Cams. localit.	reintro.	adult	1989	Pigott, 1988
Conradina glabra	Lamiaceae	USA	per			adult	1991	Gordon, 1996
Ctenium aromaticum	Poaceae	USA	per	Seedlings outplanted into fire treatment plots in mesic and wet habitat conditions.	intro. within	juvenile	1993	Glitzenstein et al., 2001
Damasonium alisma	Alismataceae	UK	ann	Introduction to newly created ponds within historic range by seed and seedlings.	intro. within	mixed		Plantlife data; Wheeler, 2001
Daviesia bursarioides	Fabaceae	Australia	per	Experimental trial part of 10 species translocation programme. Augmentation to extant site.	aug.	adult	1997	Cochrane et al., 2000
Decalepis arayalpathra	Periplocaceae	India	per	Propagated shoots re-introduced to protected former habitat of the species.	reintro.	juvenile	1998	Gangaprasad et al., 2005
Diuris magnifica	Orchidaceae	Australia	per	Tubers propagated from *ex-situ* collections were translocated to field sites.	intro, within	adult		Batty et al., 2006
Diuris micrantha	Orchidaceae	Australia	per	Tubers propagated from *ex-situ* collections were translocated to field sites.	intro. within	adult		Batty et al., 2006
Echinacea laevigata	Asteraceae	USA	per	Experimental comparison of reintroduction planting methods, adults were planted singly or seedlings were planted in clumps of varying spacings. Grazing prevented in first year.	intro. within	juvenile	2000	Alley and Affolter, 2004
Erigeron parishii	Asteraceae	USA	per	Mitigation translocation from limestone quarry.	intro. within	juvenile	1991	Mistretta and White, 2001
Erigonum ovalifolium var *vineum*	Polygonaceae	USA		mitigation translocation from limestone quarry	intro. within	juvenile	1991	Mistretta and White, 2001

Eurystyles cotyledon	Orchidaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Filago gallica	Asteraceae	UK	ann	Material taken from cultivation and transplanted into last English native locality.	reintro.	adult	1994	Rich et al., 1999
Gomesa crispa	Orchidaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Grevillea calliantha	Proteaceae	Australia	per	Experimental trial part of 10 species translocation programme. Site within historic range of species.	intro. within	adult	1997	Cochrane et al., 2000
Grevillea scapigera	Proteaceae	Australia	per	Clones of 10 genetically representative plants planted into a 'secure 'natural' site'.	intro. out	juvenile	1996	Dixon and Krauss, 2001; Dixon, 2004
Hedyotis caerulea	Rubiaceae	USA	per	Experimental comparison of propagule type (seeds, seedlings, adults) and for seeds only, the effect of sowing into dug or undisturbed ground.	intro. within, trans.	mixed	1994	Drayton and Primack, 2000
Helenium virginicum	Asteraceae	USA	per	Seedlings transplanted to two sites within 16km of only known extant population.	intro. within	juvenile	2003	Rimer and McCue, 2005
Helianthemum apenninum	Cistaceae	UK		Mixture of seeds and plants translocated to 4 sites by J.F. Hope Simpson..		seed	1955	BSBI Introductions Database
Hieracium attenuatifolium	Asteraceae	UK		Augmentation of natural population.	aug.	adult	1999	BSBI Introductions Database
Humboldtia smithiana	Orchidaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Hutera rupestris		Spain	biennial	15 mature plants planted into limestone fissures		adult	1976	Sainz-Ollero and Hernandez-Bermejo, 1979
Ipsea malabarica	Orchidaceae	India	per	Micropropagated bulbs and plantlets placed into site within species range.	intro. within	juvenile	1995	Gangaprasad et al., 1999; Martin, 2003
Jacquemontia reclinata	Convolvulaceae	USA	per	Nursery grown plants and seeds translocated to sites within extant range but not to formerly extant sites as these were deemed to be unsuitable.	intro. within	seed	2001	Maschinski and Wright, 2006

Lambertia echinata ssp *echinata*	Proteaceae	Australia	per	Experimental trial part of 10 species translocation programme. Augmentation to extant site.	aug.	adult	1997	Cochrane et al., 2000
Lambertia orbifolia	Proteaceae	Australia	per	Experimental trial part of 10 species translocation programme. Site within historic range of species.	intro. within	adult	1997	Cochrane et al., 2000
Lepismium cruciforme	Cactaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Lepismium houlletianum	Cactaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Lepismium lumbricoides	Cactaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Lepismium warmingianum	Cactaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Lindera melissifolia	Lauraceae	USA	per	Seeds from wild population were propagated *ex situ* before translocating to site within historic range.	intro. within	juvenile	1990	Smith, 2003
Linnaea borealis	Caprifoliaceae	UK	per	Shoots taken from wild population or *ex situ* cultivated plants, placed in equal proportions in two sites.	intro. within	adult	1999	Kohn and Lusby, 2004
Liparis loeselii	Orchidaceae	UK	per	20 plants translocated over 2 years from wild popualtion to site within historic range but not known if exact location once supported this species.	intro. within, trans.	adult	2005	Land pers. comm.
Lobelia cardinalis	Campanulaceae	USA	per	Experimental comparison of propagule type (seeds, seedlings, adults) and for seeds only, the effect of sowing into dug or undisturbed ground.	intro. within	mixed	1994	Drayton and Primack, 2000
Lobelia urens	Campanulaceae	UK	per	Seedlings from the six plants salved from the Trewether site were planted at Ventongimps Moor.		juvenile	1968	BSBI Introductions Database

Luronium natans	Alismataceae	UK	per	Three 'clumps' transplanted from original site into ditches at Potter Heigham.		adult	1983	BSBI Introductions Database
Maxillaria ferdinandiana	Orchidaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Maxillaria juergensii	Orchidaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Maxillaria picta	Orchidaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Melampyrum pratense	Scrophulariaceae	UK	ann	Turf and seed translocation to coppiced woodland.	trans., intro within	adult	1985	Walter, 2005
Melampyrum sylvaticum	Scrophulariaceae	UK	ann	Seeds collected from three natural populations, mixed in equal proportions and transplanted into five sites within species presumed historic range.	intro. within	seed	2005	Dalrymple et al., 2008
Myricaria laxifolia	Myricaceae	China	per	Mitigation to avoid flood area of Three Gorges Dam. Seedlings taken from various sites in flood zone and transplanted in suitable habitat.	trans., intro. out	juvenile	2002	Chen et al., 2005
Nepeta rtanjensis	Lamiaceae	Serbia	per	*In vitro* propagation and outplanting within the historic range of species.	intro. within	juvenile	2004	Misic et al., 2005
Oncidium flexuosum	Orchidaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Oncidium macronix	Orchidaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Oncidium riograndense	Orchidaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Opuntia corallicola	Cactaceae	USA	per	Fallen 'pads' propagated and bulked up at Fairchild. Planted out at extant site.	aug.	juvenile	1996	Stiling et al., 2000

Oryza rufipogon	Poaceae	China	per	Plants raised *ex situ* used to recreate population destroyed when wetland habitat flooded during dam building.	reintro.	adult	1993	Liu et al., 2004
Osmorhiza claytonii	Apiaceae	USA	per	Experimental comparison of propagule type (seeds, seedlings, adults) and for seeds only, the effect of sowing into dug or undisturbed ground.	intro. within, trans.	mixed	1994	Drayton and Primack, 2000
Parnassia caroliniana	Saxifragaceae	USA	per	Seedlings outplanted into fire treatment plots in wet habitat conditions.	intro. out	juvenile	1995	Glitzenstein et al., 2001
Phymatidium delicatulum	Orchidaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Pinus torreyana	Pinaceae	USA	per	Seeds from trees killed by beetle outbreak were used to grow seedlings *ex situ* before transplanting back to sites in which parents had grown.	reintro.	juvenile	1994	Ledig, 1996
Plantago sparsiflora	Plantaginaceae	USA	per	Seedlings outplanted into fire treatment plots in wet habitat conditions.	intro. within	juvenile	1998	Glitzenstein et al., 2001
Pleurothallis aveniformis	Orchidaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Polystachya estrellensis	Orchidaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Posidonia australis	Posidonaceae	Australia	per	Rhizomes from large source population transplanted to five sites having undergone significant declines .	aug.	adult	1999	Meehan and West, 2002
Primula vulgaris	Primulaceae	Poland	per	Plants translocated from botanic garden to previously extant site.	reintro.	adult	1993	Kucharczyk and Teske, 1996
Prostanthera eurybiodes	Lamiaceae	Australia	per	10 seedlings translocated to each of three microsites to investigate microsite differences and effect on survival.	intro. within	juvenile	1996	Jusaitis, 2005
Pseudophoenix sargentii	Arecaceae	USA	per	Nursery grown plants translocated to three islands of the Florida Keys, two known to have once supported the species, the third supporting an extant population.	aug., intro. within	juvenile	1991	Maschinski and Dusquenel, 2007
Rhipsalis cereuscula	Cactaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005

Rhipsalis floccosa	Cactaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Rhipsalis teres	Cactaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Salix lapponum	Salicaceae	UK	per	Augmentation of remnant willow scrub using seedlings propagated from local seed sources into fenced exclosures.	aug.	juvenile	1991	Mardon, 2003
Salix myrsinifolia	Salicaceae	UK	per	Augmentation of remnant willow scrub using seedlings propagated from local seed sources into fenced exclosures.	aug.	juvenile	1991	Mardon, 2003
Salvia pratensis	Lamiaceae	UK	per	Augmentation of dwindling population.	aug.	adult	1999	BSBI Introductions Database
Sanguinaria canadensis	Papaveraceae	USA	per	Experimental comparison of propagule type (seeds, seedlings, adults) and for seeds only, the effect of sowing into dug or undisturbed ground.	intro. within, trans.	mixed	1994	Drayton and Primack, 2000
Sarracenia flava	Sarraceniaceae	USA	per	Mitigation of general decline in habitat extent using highway rights-of-way to recreate bog habitat.	intro. within	adult	1998	Sheridan and Penick, 2002
Saxifraga virginiensis	Saxifragaceae	USA	per	Experimental comparison of propagule type (seeds, seedlings, adults) and for seeds only, the effect of sowing into dug or undisturbed ground.	intro. within, trans.	mixed	1994	Drayton and Primack, 2000
Scleranthus perennis ssp. *prostrates*	Caryophyllaceae	UK	per	Adult plants raised in cultivation outplanted into formerly extant site, West Stow, in two years, 1995 and 1997. Seedlings outplanted at Icklingham in 1999.	reintro.	adult	1995	Leonard, 2006a, b
Senecio paludosus	Asteraceae	UK	per	Plants introduced.		adult	1996	BSBI Introductions Database
Silene douglasii var. *oraria*	Caryophyllaceae	USA	per	Experimental re-introduction to discern effects of selfing *vs.* Outcrossed plants and high and low planting densities.	intro. within	adult	1998	Kephart, 2004; Lofflin and Kephart, 2005

Sorghastrum nutans	Poaceae	USA	per	Seedlings outplanted into fire treatment plots in mesic and wet habitat conditions.	intro. within	juvenile	1997	Glitzenstein et al., 2001
Specklinia malmeana	Orchidaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Specklinia pabstii	Orchidaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Syzygium travancoricum		India	per	Augmentation of 3 exisitng populations in two forest areas.	aug.	juvenile	1999	Anand et al., 2004
Taraxacum palustre	Asteraceae	UK	per	Re-introduced using plants raised from seed originating from same locality.	reintro.	adult	1998	BSBI Introductions Database
Thlaspi perfoliatum	Brassicaceae	UK	ann	Introduction by seed sourced from wild population.		seed		Rich et al., 1998; Plantlife data
Tillandsia geminiflora	Bromeliaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Tillandsia stricta	Bromeliaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Tillandsia tenuifolia	Bromeliaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Tillandsia usneoides	Bromeliaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Trichocentrum pumilum	Orchidaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Trinia glauca	Apiaceae	UK	per	Mixed propagules translocated to two sites in 1955.		mixed	1955	BSBI Introductions Database
Veronica spicata	Plantaginaceae	UK	per	Mixed propagules translocated to two sites in 1955.		mixed	1955	BSBI Introductions Database

Vitis vinifera	Vitaceae	France	per	Two stage translocation of cuttings; survival ordinated against environmental variables to determine optimum habitat.		juvenile	1992	Arnold et al., 2005
Vriesea friburgensis	Bromeliaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Vriesea platynema	Bromeliaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005
Widdringtonia cederbergensis	Cupressaceae	South Africa	per	Seedlings planted in reserve area set aside for extant cedars.	reintro.	juvenile	1987	Mustart et al., 1995
Woodsia ilvensis	Dryopteridaceae	Estonia	per	Experimental introduction to test feasibility given potential habitat change in time lapsed since extinction.	reintro.	juvenile	1996	Aguraiuja pers. comm.
		UK	per	Combination of augmentation of extant populations and re-introduction to extinct sites.	reintro., aug.	adult	2000	McHaffie, 2005, 2006; Lusby et al., 2002
Zygostates alleniana	Orchidaceae	Brazil	per	38 species identified for protection and translocated from flood zone to permanent preservation area.	trans., intro within	adult	2001	Jasper et al., 2005

APPENDIX 5. REFERENCES FOR STUDIES EXCLUDED FROM META-ANALYSIS

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