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Identifying factors associated with the success and failure of terrestrial insect translocations

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Abstract

Translocation is increasingly used as a management strategy to mitigate the effects of human activity on biodiversity. Based on the current literature, we summarised trends in terrestrial insect translocations and identified factors associated with success and failure. As the authors' definitions of success and failure varied according to the individual sets of goals and objectives in each project, we adopted a standardised species-specific definition of success. We applied generalised linear models and information-theoretic model selection to identify the most important factors associated with translocation success. We found literature documenting the translocation of 74 terrestrial insect species to 134 release sites. Of the translocations motivated by conservation, 52% were considered successful, 31% were considered to have failed and 17% were undetermined. Our results indicate that the number of individuals released at a translocation site was the most important factor associated with translocation success, despite this being a relatively infrequent perceived cause of failure as reported by authors. Factors relating to weather and climate and habitat quality were the most commonly perceived causes of translocation failure by authors. Consideration of these factors by managers during the planning process may increase the chance of success in future translocation attempts of terrestrial insects.

Introduction

Translocation represents a valuable tool for wildlife conservation (Fischer and Lindenmayer, 2000; Germano and Bishop, 2009). There has been substantial growth in translocation practice during the past three decades (Seddon *et al.,* 2007; Taylor *et al.,* 2017), resulting in a taxonomically diverse assemblage of translocation case studies. In response to the growing use of translocation as a management tool, the International Union for the Conservation of Nature (IUCN) published a set of broad guidelines in 2013 for conservation-based translocations (IUCN, 2013). These guidelines offer a detailed framework for all phases of a translocation, generalised for all organisms and have likely contributed to the successful recovery of threatened species. In addition to the IUCN guidelines, there have been a number of global reviews, covering amphibians and reptiles (e.g. Dodd and Seigel, 1991; Germano and Bishop, 2009), birds and mammals (Griffith *et al.,* 1989; Wolf *et al.,* 1996), plants (Dalrymple *et al.,* 2012), freshwater fish (Cochran-Biederman *et al.,* 2015) and freshwater macroinvertebrates (Jourdan *et al.,* 2018). The majority of these reviews also aim to improve the success rate of translocations for their focal taxa, by identifying specific factors associated with success. Terrestrial insects represent one of the major taxonomic classes that is yet to be the focus of a global review. Terrestrial insects are defined as insect species with lifecycles that are partly or fully dependent on habitats existing in the terrestrial environment.

The Class Insecta has the highest abundance, biomass and diversity in the animal kingdom (Wilson, 1987; Kim, 1993). Insects occupy almost every type of terrestrial habitat and they provide numerous ecosystem services (Losey and Vaughan, 2006). The value of their ecosystem services has been conservatively estimated at US$57 billion per year in the United States alone (Losey and Vaughan, 2006). Despite their enormous contribution, insects are often neglected in conservation strategies, which typically focus on more iconic vertebrate species (Seddon *et al.,* 2005). The lack of attention given to insects is reflected by the paucity of policies that protect them, for example, legislation in Europe protects only 0.12% of the region's insect species (Leandro *et al.,* 2017). This figure is concerning, particularly given recent research revealing a dramatic global decline in insect populations that could lead to the extinction of over 40% of the world's insect species during the next few decades (Sánchez-Bayo and Wyckhuys, 2019). The growing recognition of the global decline in insect populations (e.g. Hallmann *et al.,* 2017; Vogel 2017; Taylor *et al.,* 2018) is likely to increase the demand for methods and approaches, such as translocation, to restore lost species and functions.

Despite having not featured as frequently in translocation projects as vertebrate groups such as birds and mammals (Seddon *et al.,* 2005), the life-history attributes of insects would suggest they are potentially ideal candidates for translocation. The small body size and short generation time of insects makes them comparatively low cost and quick to propagate in preparation for a translocation (Balmford *et al.,* 1996). They also require smaller habitat patches to support viable populations compared to most vertebrate species (e.g. Baur *et al.,* 2017), meaning pre- and post-release habitat management costs are more economical. Indeed, many managers already recognise the candidacy of insects for translocation, which has led to the instigation of insect translocation projects for a variety of motivations including conservation (e.g. Baur *et al.,* 2017), mitigation (e.g. Simon *et al.,* 2016), research (e.g. Forsman *et al.,* 2012) and biological control (e.g. Kapranas *et al.,* 2014).

In this paper, we begin by exploring the global trends in terrestrial insect translocations. This includes regional trends, taxonomic trends and their respective biases. We will then focus more specifically on conservation translocations with the objective of identifying the general mechanisms that explain past successes and failures. Knowledge of such mechanisms has the potential to inform future management decisions, and encourage further investigation into how these and other factors influence translocation outcome for terrestrial insects.

Methodology

Data Collection

We performed a literature search to find examples of terrestrial insect translocations from across the globe. We used the search engines 'Thomson Reuters Web of Science' and 'Directory of Open Access Journals', and the 'Conservation Evidence Individual Studies repository' to retrieve relevant papers published at the earliest possible date up until 08/10/2018 (for further detail on the search methodology and search terms used on each platform, see Supplementary Material 1). Once we had performed the search, we imported all of the resulting papers into EndNote referencing software and manually screened each record to verify its relevance to insect translocation. Articles were not included in the study if they were irrelevant to insect translocation based on their title and abstract or upon further scrutiny of the paper. We also screened the bibliographies of each relevant publication identified during our search to find additional studies of relevance. Using the methods outlined above, we found two national cross-taxonomic translocation reviews, one for the United Kingdom (Carter *et al.,* 2017) and one for New Zealand (Sherley *et al.,* 2010), which led to the addition of eighteen translocation projects that were not found individually through our search methodology. In every case, this was because these translocations were restricted to the grey literature or unpublished reports and accounts.

Once our literature search was complete, we categorised each translocation project based on its primary motivation. We identified five types of translocation motive from the dataset: conservation, mitigation, research, functional restoration and biological control. We could often infer the motivation of the translocation based on the article's stated aims or objectives and these were recorded accordingly. However, this was not possible for every article, in which case authors were contacted to corroborate. We categorised translocations as research-motivated if they aimed to further the field of conservation translocations through the release of insects in more experimental circumstances. For example, Willis *et al.* (2009) translocated two common butterfly species ~35 and ~65 km beyond their current ranges in the United Kingdom to test the use of species distribution models for identifying potential assisted colonisation release sites. In this study, the aim was to test the principle of the approach, rather than to establish populations of the two species for conservation purposes. We made the decision to remove biological control-related articles from the dataset, as this is an extensive discipline with core objectives that diverge significantly from the ones typical of the other motives. As one of the primary goals of our study is to identify the key determinants of success in insect translocations, we split the dataset based on motivation. Every translocation, irrespective of motivation (except biological control), was used to identify general trends in insect translocations, such as regional and taxonomic biases, i.e. descriptive statistics. However, in order to identify the key determinants of success using statistical analyses, we incorporated only translocations where the primary motivation was conservation. This decision was made because conservation translocations principally aim to establish a viable population (IUCN, 2013), whereas translocations motivated by other factors often do not (e.g. Willis *et al.,* 2009; Pratt and Emmel, 2010; Forsman *et al.,* 2012).

Data Extraction and Refinement

For every translocation, we collected data on the Order of species translocated, continent and country of translocation, type of translocation, motivation of translocation and year of release. For conservation translocations, we also collected data on most recent year of monitoring, population status at most recent year of monitoring, origin of source population, number of release years, life stage of released individuals, total number of each life stage released across all years, distance between release site and source population (if translocation was from wild to wild) and perceived cause of project failure (if applicable). We identified this set of variables based on their potential importance for terrestrial insect translocations and their inclusion and relative importance in previous translocation reviews (e.g. Germano and Bishop, 2009; Rummel *et al.,* 2016). The one exception being distance between release site and source population, which to our knowledge has not been considered in previous reviews, but is potentially important given the general assumption that populations that are physically closer to the release site will be better adapted to the environmental conditions present (e.g. IUCN, 2013). If the source individuals originated from both wild and captive-bred populations (*n*=4), we treated the source population as 'captive-bred'. Translocations that used headstarted individuals (*n*=2) were also grouped with 'captive-bred', as they had spent at least part of their lifecycle in captive conditions. In order to maximise the amount of data available for statistical analyses, we grouped translocation projects that released larvae, pupae or nymphs into one variable state labelled 'immatures'. Variable states with a small sample size (<4) were not included in the statistical analyses (e.g. release of 'colonies', *n*=2). In cases where we could not obtain all the required information by examining relevant articles we contacted authors directly to acquire missing information.

Defining Translocation Success

The authors' definitions of success varied according to the individual set of goals or objectives in each study. There is still no general and broadly accepted definition of translocation success (Robert *et al.,* 2015), therefore, in order to conduct a more objective analysis, we adopted a species-specific approach to defining translocation success. We considered a translocation successful if it met two criteria: i) the time elapsed between the most recent release and most recent post-release monitoring exceeded the lifecycle duration of the species and ii) the most recent monitoring results indicated population persistence at the release site. If a translocation did not meet these criteria, we did not necessarily consider the translocation to be unsuccessful, as a failure to meet this definition was often due to a lack of post-release monitoring; in this case the outcome was classified as undetermined. If the length of the lifecycle of a species was unknown, then we placed a minimum threshold of five years between date of latest release and date of latest monitoring. This covers most insects except in exceptional cases e.g. cicadas and certain wood boring beetles, e.g. Cerambycidae and Buprestidae.

Statistical Analyses

We used a generalised linear model (GLM) with a logit link and binomial random component that can be used with mixed data categories to identify variables associated with successful translocations (see Table 1 for list of predictor variables). The binary response variable was success or failure. We refer to this statistical approach herein as logistic regression. As our statistical analyses were of a more exploratory than confirmatory nature, we included all single-variable models and models with two-way interactions that represent potentially meaningful ecological relationships between variables and are not in breach of the assumptions of logistic regression analysis.

We used the information-theoretic approach to compare the different models by methods based on the Kullback-Leibler distance (Burnham and Anderson, 2003). Models were ranked using Akaike's information criterion corrected for small sample size (AICc). This method encourages parsimony by applying a penalty for the number of parameters in a model (Burnham and Anderson, 2003). AICc differences (Δ_i) representing the distance between the selected (best) model and *i*th model were also calculated. AICc differences were then used to estimate Akaike weights (w_i), indicating the probability that a particular model performed best for the sampling situation under consideration. All analyses were performed in R (Version 3.5.1) using the AICcmodavg package (Mazerolle and Mazerolle, 2017).

Values for the distance between source population and release site variable (SourceRelDist) could only be calculated for translocation projects that sourced wild individuals. As this caused SourceRelDist to be correlated with Origin, a separate analysis was conducted to test for differences in translocation outcome based on SourceRelDist. Shapiro-Wilk normality tests suggested that neither the original nor the log-transformed data followed a normal distribution. Therefore, the non-parametric Mann-Whitney U test (Mann and Whitney 1947) was adopted to compare the distributions of success and failure.

Results

We found literature documenting the translocation of 74 terrestrial insect species to 134 release sites. A total of seven different taxonomic orders received translocations (Figure 1). Lepidoptera was the most frequently translocated Order with 52 translocations (39%) involving this group, while Orthoptera was second with 39 translocations (29%) (see the Supplementary Material 2 for a list of species translocated). Translocations of insect species were most commonly conducted on the European continent (*n*=74), with the Oceania (*n*=35) and North America (*n*=19) carrying out the second and third most translocations respectively (Figure 2). There were a very limited number of terrestrial insect translocations in Africa, Asia and South America.

There were some notable regional biases in the orders targeted for translocation projects (Supplementary Material 2). For example, Orthoptera, the second most frequently translocated order globally, were not the subjects of any translocation projects in North America, but comprised the majority of projects in Oceania (71%). In Europe and North America, the taxonomic bias was skewed more towards Lepidoptera species, with 54% and 58% of translocation projects comprising this group, respectively. Just one project focused on the translocation of a Lepidoptera species in Oceania.

Conservation was the most commonly identified motivation behind terrestrial insect translocation projects, with a total of 107 translocations being conducted for this purpose. Research was a relatively frequent motivation (*n*=20), whereas translocations for mitigation (*n*=4) or functional restoration (*n*=3) were uncommon.

Based on our success criteria, 56 conservation translocation projects were successful (52%), 33 failed (31%) and 18 were undetermined (17%). Based on a subset of these translocations that were eligible for statistical analysis, the information-theoretic model selection resulted in the highest ranked logistic regression model consisting of the number of individuals released (NumRel) as a single predictor variable (Table 2). The second and third highest ranked models also featured the NumRel variable, with Origin and LifeHistory as additive terms, respectively. When Origin and LifeHistory were taken individually the models had considerably less support, suggesting that NumRel was more influential than these two variables. A proportion of support was given to every model considered in the analysis, with the three highest performing models accounting for 40% of the Akaike weights, which we acknowledge as being relatively low. However, the consistent presence of NumRel amongst the top performing models suggests that this variable was the most important determinant of success for terrestrial insect translocations.

Successful translocation projects released more individuals than failed projects - successful projects released a mean average of 2030 ± 706 individuals, while failed projects released a mean average of 667 ± 166 individuals. Most terrestrial insect translocation projects sourced their stock from wild populations, with 66% of translocation projects opting to release wild-caught individuals. Success rate was 67% when using wild stock, which was marginally higher than the 59% success rate achieved by translocation projects that used captive-bred stock. The average distance between source population and release site was 110.9 ± 28.9 km. However, there was no statistically significant difference in the distance separating source population and release site between successful and failed translocation projects (p=0.714).

Habitat quality, as well as weather and climate, were the most frequently cited causes of translocation failure according to those involved with terrestrial insect translocation projects (Figure 3). Of the 33 insect translocations that resulted in failure, over a third were believed to have failed due to poor habitat quality or the effects of weather and climate at the release site. After these two factors, the main reported causes of translocation failure were predation pressure and pollution.

Factors relating to the technique of a translocation were rarely considered as potential causes of failure. Similarly, an insufficient number of individuals released was rarely considered as a potential cause of failure (*n*=2), despite successful translocation projects releasing an average of around three times as many individuals compared to those that failed.

Discussion

<u>The state of terrestrial insect translocations</u>

The terrestrial insect translocation literature is regionally and taxonomically diverse, and contains a wealth of case studies possessing the potential to inform future translocation management decisions. Of the translocation projects summarised here, around half were defined as successful. This figure is slightly higher than the success rates reported for other animal groups (e.g. Griffith *et al.,* 1989; Germano and Bishop, 2009), suggesting that insects respond comparatively well to translocation. Although more translocations were defined as successful (52%), the proportion of undetermined (17%) and failed translocations (31%) suggests that there is room for improvement in terms of planning and conducting terrestrial insect translocations, as well as post-release monitoring and the reporting of results.

Unlike for other animal taxa (Fischer and Lindenmayer, 2000; Seddon *et al.,* 2014), the majority of insect translocation projects originated from Europe, rather than Oceania or North America. This places Europe as a global leader in insect translocations, a position that has generally been filled by Oceania with respect to vertebrate translocations due to the large number of translocations that have been undertaken there (Fischer and Lindenmayer, 2000; Seddon *et al.,* 2014). It is possible that some regional biases were introduced to the dataset through our decision to include national translocation reviews (e.g. Sherley *et al.,* 2010; Carter *et al.,* 2017). However, the omission of these reviews would have had little effect on the regional trends that were detected via our search methodology (Figure 1 and Figure 2) and their inclusion provided valuable additional case studies for analysis.

Taxonomic biases in reintroduction projects have been noted in the past towards different vertebrate groups (Seddon *et al.,* 2005), and our findings indicate similar biases in insect translocations. These biases may be partly explained by the composition of regional and national conservation lists of species-of-concern (e.g. Walsh *et al.,* 2013). In the United States, Lepidoptera, Coleoptera and Odonata dominate conservation priorities, representing a combined total of 89% of insect species listed, a proportion far greater than the relative species diversity in these orders (Bossart and Carlton 2002). In the present study, Lepidoptera formed the majority of insect translocations in the United States (58%), despite this group accounting for just 12.6% of insect species in the country (Bossart and Carlton, 2002). Conversely, we did not find any translocation projects targeting Diptera or Hemiptera species in the United States (or globally), despite these two orders accounting for a combined total of 34.1% of the named insect species in the country. Bossart and Carlton (2002) suggest that these taxonomic biases are likely as a result of both the iconic appeal of taxa such as Lepidoptera, and the availability of taxonomic specialists. These factors appear to be driving insect translocations globally, and they threaten the viability of countless other species by potentially misdirecting conservation priorities and limited resources towards species perceived as iconic or interesting (e.g. Sitas *et al.,* 2009; Di Marco *et al.,* 2017).

There are many motivations behind animal translocations (Seddon *et al.,* 2012) with conservation the most frequently identified motivation in the present study due to our search focus. However, translocations motivated by biological control, which were beyond the scope of this study, are frequently conducted with insects as the control agent species. Biological control has been used extensively around the world: 6,158 documented insect introductions were conducted prior to 2010 for this purpose (Cock *et al.,* 2016), of which 32.6% resulted in the establishment of the control agent species. This level of establishment is high given that such a large proportion of biological control releases are far outside the species indigenous range (e.g. Dahlsten *et al.,* 1998; Chauzat *et al.,* 2002; Quacchia *et al.,* 2007). Although the field of biological control is ecologically, economically and socially divergent from that of conservation translocations, there remains scope for practical skill exchange. Biological control programmes often involve highly skilled entomologists that use increasingly sophisticated technologies and protocols to maximise the population viability and chances of establishment for their captive-bred stock (e.g. Duan *et al.,* 2013; van Lenteren *et al.,* 2018). Conservation translocation programmes with a captive-breeding component, which remain less common than wild to wild translocations for insects, can incorporate many of the pathogen screening, animal husbandry and genetic management procedures used in successful biological control programmes to develop their own existing and future programmes.

<u>Characteristics of translocation success</u>

Ratios of translocation success based on academic literature reviews should be approached with a degree of caution, due to the decreased likelihood of authors publishing failed translocations. Successful translocation projects are more likely to be published than failures because authors do not wish to portray themselves or other involved parties unfavourably and publication bias favours articles with positive outcomes (Forstmeier *et al.,* 2017). A review of amphibian and reptile translocation projects in New Zealand found that the published success rate was considerably higher than the rate of success found across all translocations, and successful translocations were more likely to be published than those that failed (Miller *et al.,* 2014). Based on these findings, the proportion of failures found during our research may not be representative of all failed terrestrial insect translocations, but instead represent the available literature.

The definition of translocation success adopted for this research is similar to that for reviews of other animal taxa (e.g. Germano and Bishop, 2009; White *et al.,* 2012; Cochran-Biederman *et al.,* 2015). This definition ensures that the focal species has completed all phases of its lifecycle at the release site, which is widely regarded as a fundamental indicator of translocation success (McCoy *et al.,* 2014; Robert *et al.,* 2015). The potential drawback of defining success in this way is that it may allow for more translocations that only achieved short-term success to be defined as successful (e.g. translocated population still present after one lifecycle duration of a univoltine species). However, the conservation translocations analysed during this study generally established long-term populations, with 80% reporting the persistence of the translocated population for >5 years after the most recent release and 46% for >10 years (see Supplementary Material 2).

Our results indicate that terrestrial insect translocation success is influenced most by the number of individuals released – translocations are more likely to be successful when releasing more individuals. Our findings are unsurprising – with a greater number of founder individuals, a translocated population is less vulnerable to the effects of demographic stochasticity, loss of genetic diversity by drift, and inbreeding depression, which are more prevalent in smaller populations. Therefore, we suggest that managers should aim to maximise the number of individuals released. Population models can be a useful tool for predicting the optimal number of individuals for release (e.g. Wagner *et al.,* 2005; Unger *et al.,* 2013; Heikkinen *et al.,* 2015), but their outputs are less valuable for species with inadequate population and life-history data. The optimal number of individuals for release will vary depending on their life stage due to fluctuating mortality rates between adult, juvenile and egg phases (Price *et al.,* 2011). With a large enough sample size, we would have split the number of individuals released variable based on the life stage released variable and compared differences in translocation outcome for each life stage category, but this was impractical with the number of cases that were available.

Reviews of vertebrate translocations suggest that wild source populations are generally associated with greater translocation success than captive-bred source populations (e.g. Griffith *et al.,* 1989; Rummel *et al.,* 2016), and concerns have been raised over the behavioural, morphological, demographic and genetic changes resulting from captive-breeding programmes (Lewis and Thomas, 2001; Williams and Hoffman, 2009). Our results suggest that insect translocations are also more successful when individuals are sourced from wild populations, though the magnitude of this difference is marginal (<10%), and is much less than that found for vertebrate taxa (e.g. 37% for birds and mammals, Griffith *et al.,* 1989). It may not always be feasible to acquire large numbers of wild individuals for translocation as remaining wild populations may have declined in abundance and extent-of-occurrence to the point where they are too fragile to withstand the loss of a sufficiently large number of source individuals (Dimond and Armstrong, 2007). Under these circumstances, captive-breeding programmes provide a possible alternative for the acquisition of large numbers of individuals whilst minimising loss of viability of wild populations.

Insects are particularly suitable for captive-breeding due to their life-history attributes, such as small body size and rapid reproductive potential, meaning that viable populations can be managed more cost-effectively than most vertebrate species (Balmford *et al.,* 1996). In North America, zoological institutions are increasingly involved in captive-breeding programmes aiming to release animals into the wild (Brichieri-Colombi *et al.,* 2018). A specially designated breeding facility at Roger Williams Park Zoo has been responsible for the propagation and release of over 2,800 Critically Endangered American Burying Beetle (*Nicrophorus americanus* Olivier, 1790) to Nantucket Island, Massachusetts (Mckenna-Foster *et al.,* 2016). In addition to their contribution of valuable source stock, involving zoos in translocation projects has the additional benefits of promoting the conservation of the focal species, raising public awareness, educating the public and raising extra funds (Miller *et al.,* 2004).

The IUCN Guidelines for Reintroductions and Other Conservation Translocations (2013) recommend the selection of source populations that are physically closer to release sites, however, we found no statistical difference in the outcome of terrestrial insect translocations based on the distance between source population and release site. The international translocations of three butterfly species in Europe achieved long-term success (≥10 years) when sourcing individuals from populations more than 1,000 km away (Wynhoff 1998; Wynhoff *et al.,* 2008; Thomas *et al.,* 2009). Due to the perceived increase in risk (e.g. Scottish Natural Heritage, 2014), long-distance translocations are likely to be approached with extra caution, meaning more time and attention is paid to researching the ecological requirements of the focal species and optimising and maintaining release site habitat suitability; as was the case with the three long-distance European butterfly translocations.

<u>Examining translocation failure</u>

The effects of weather and climate were one of the most frequently reported causes of translocation failure. Insect life-cycles and abundance are influenced strongly by temperature (Danks, 1987) and precipitation (Roy *et al.,* 2008; Liberal *et al.,* 2011). Mismatches in climate conditions between source populations and release sites, and extreme weather (e.g. drought or high rainfall) can be detrimental to translocated insect populations (e.g. Dempster and Hall, 1980; Daniels, 2009) and difficult to avoid or manage. However, there are preventative steps prior to translocation that can be taken. For example, estimating the climate suitability of potential release sites under current and future environmental conditions can minimise the risk of selecting sub-optimal release sites or sites that will become unsuitable under future climate change (Guisan *et al.,* 2013). This is possible with the use of species distribution models (SDMs), which in their most widely used form, correlatively identify suitable environmental conditions for a species based on the conditions present at sites supporting extant populations.

The use of SDMs during the translocation planning process is highly advised when contemplating the movement of a species beyond its indigenous range (i.e. assisted colonisation) (Chauvenet *et al.,* 2013). However, SDMs are also useful for reintroduction planning (see Osborne and Seddon, 2012 for potential applications and issues of using SDMs for reintroductions), especially if the focal species became extinct at the proposed reintroduction site some time ago. It is risky to use historic site occupancy as a prerequisite for site suitability; climate change during the intervening period between the initial extinction and time of release could have rendered the site unsuitable. For example, the Apollo Butterfly (*Parnassius apollo* Linnaeus, 1758) went extinct in southern Finland in the 1950s and reintroductions were attempted to a number of islands between 2009 and 2011 (Fred and Brommer, 2015; J. Brommer pers. comm.). The reintroduction failed, and the authors hypothesise that climatic factors, such as unfavourable winter conditions and the timing of spring, may have played a role in the failure of the species to persist on the islands.

To our knowledge, no attempt has been made within the peer-reviewed literature to assess the extent to which climate conditions at release sites may have influenced the outcome of past translocation atempts. The frequent attribution of translocation failure to unsuitable weather and climate conditions by those involved with insect translocations suggests there is a necessity to investigate this factor further. A statistical modelling approach similar to the one applied in Csergő *et al.* (2017), in which predicted climate suitability values generated from SDMs were related to the demographic performance of plant populations, could be applied to detect potential correlations between release site climate suitability and the outcome of insect translocations.

The quality of release site habitat has been identified as an important factor for translocation success in previous animal translocation reviews (e.g. Dodd and Seigel, 1991; White *et al.,* 2012). We were unable to assess habitat quality for the projects that we reviewed, but habitat quality was one of the most frequently reported causes of translocation failure by authors. The importance of habitat quality for population viability has repeatedly been shown across a diverse range of insect taxa (Baur *et al.,* 2002; Franzén and Nilsson, 2010; Pasinelli *et al.,* 2013) and consequently, defining the crucial habitat requirements prior to reintroduction is required. Habitat descriptions of the focal species at sites supporting healthy populations, preferably including the candidate source population(s), should be conducted to ensure the proposed translocation site is suitable prior to release (IUCN, 2013). Furthermore, assurances of long-term active management should be obtained prior to translocation to safeguard habitat quality under future pressures. Changes to land tenure and discontinuation of habitat management activities have been responsible for the failure of insect translocations in the past (e.g. *Deinacrida mahoenui* Gibbs, 1999 C. Watts pers. comm; *Cicindela dorsalis* Say, 1817 M. Brust pers. comm.).

Based on our method of data collection, we were unable to obtain data on the habitat quality of release sites for insects. This type of data would be obtainable through the circulation of a survey to translocation practitioners, as demonstrated in a review of mammal and bird translocations in which respondents ranked habitat quality as “excellent”, “good” or “fair or poor” (Griffith *et al.,* 1989). However, it can be particularly challenging to gauge habitat quality for insects, as highlighted in Williams *et al.,* (2014), in which conservation professionals often ranked habitat quality for carabid beetles as both “good” and “bad” in areas where there was maximal diversity. The subjectivity of habitat quality assessment suggests that, although this variable is of importance, the method by which this data is collected requires careful consideration of how to maximise objectivity.

Recommendations for improving standardisation and dissemination

Many of the translocations reviewed during this research were poorly documented either methodologically and/or in terms of long-term results. This presents a challenge to managers who wish to learn from the successful and the unsuccessful aspects of previous translocations in order to make evidence-based decisions regarding their own projects. For vertebrates, there is a growing body of literature encouraging the standardisation of documenting and monitoring the methods and outcomes associated with translocations (e.g. Fischer and Lindenmayer, 2000; Sutherland *et al.,* 2010; Ewen *et al.,* 2012). Recently, similar standardisation-based recommendations have also been published for lepidopteran translocations (Daniels *et al.,* 2018). Complementary to improved standardisation, we also advise the dissemination of information, ideally through a centralised international database that facilitates the dispersion of information to an audience beyond academic circles (e.g. TRANSLOC, a translocation database for the Western Palearctic region, link: http://translocations.in2p3.fr/). In comparison to translocation reviews of other taxonomic groups (e.g. Griffith *et al.,* 1989; Cochran-Biederman *et al.,* 2015) the body of literature surrounding terrestrial insect translocations is limited; thus it is all the more important that platforms exist on which successful and unsuccessful projects can be shared and accessed effectively.

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Table 1. Predictor variables used in generalised linear models to identify factors relating to terrestrial insect translocation success.

Variable abbreviation	**Variable description (states)**
LifeHistory	Life History (Hemimetabolous or Holometabolous)
LifeStageRel	Life stage released (Adults, Immatures, Eggs or Mixed)
NRelYears	Total number of release years
NumRel	Total number of individuals released
Origin	Origin of source population (Wild or Captive-bred)

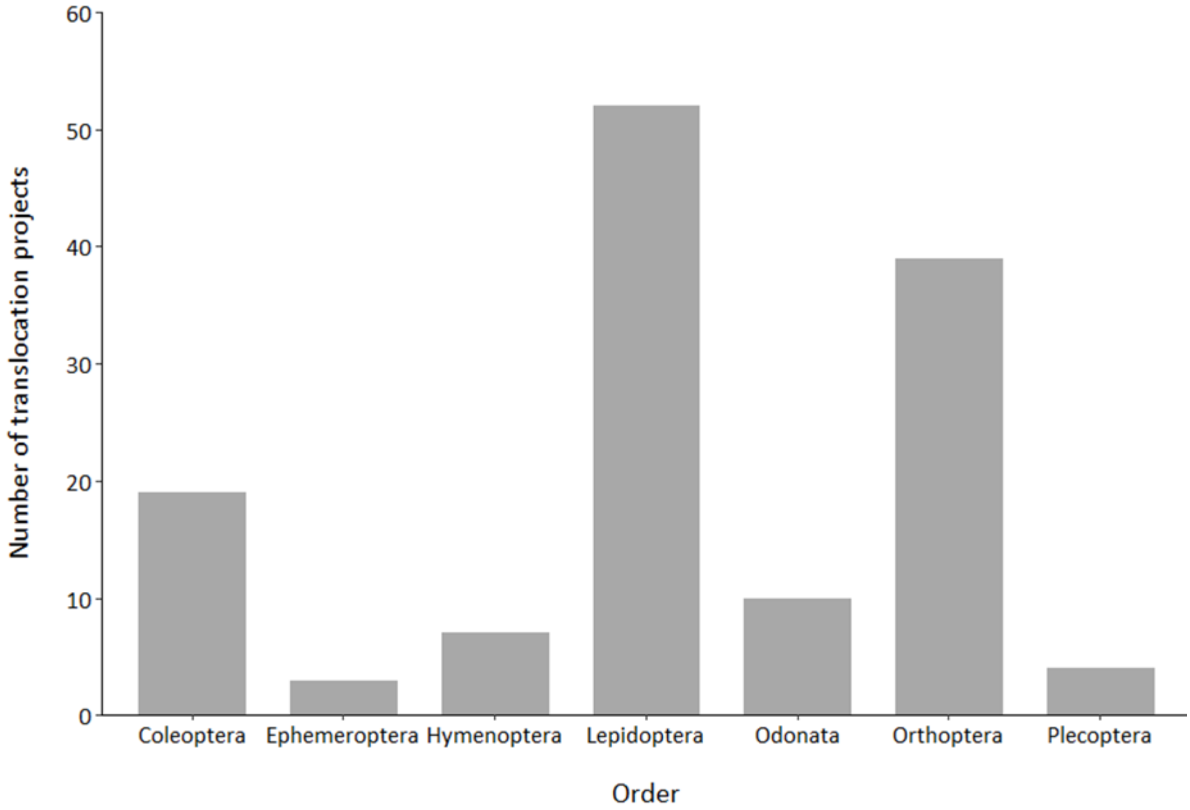

Figure 1. Number of terrestrial insect translocations reviewed for each insect Order (*n*=134).

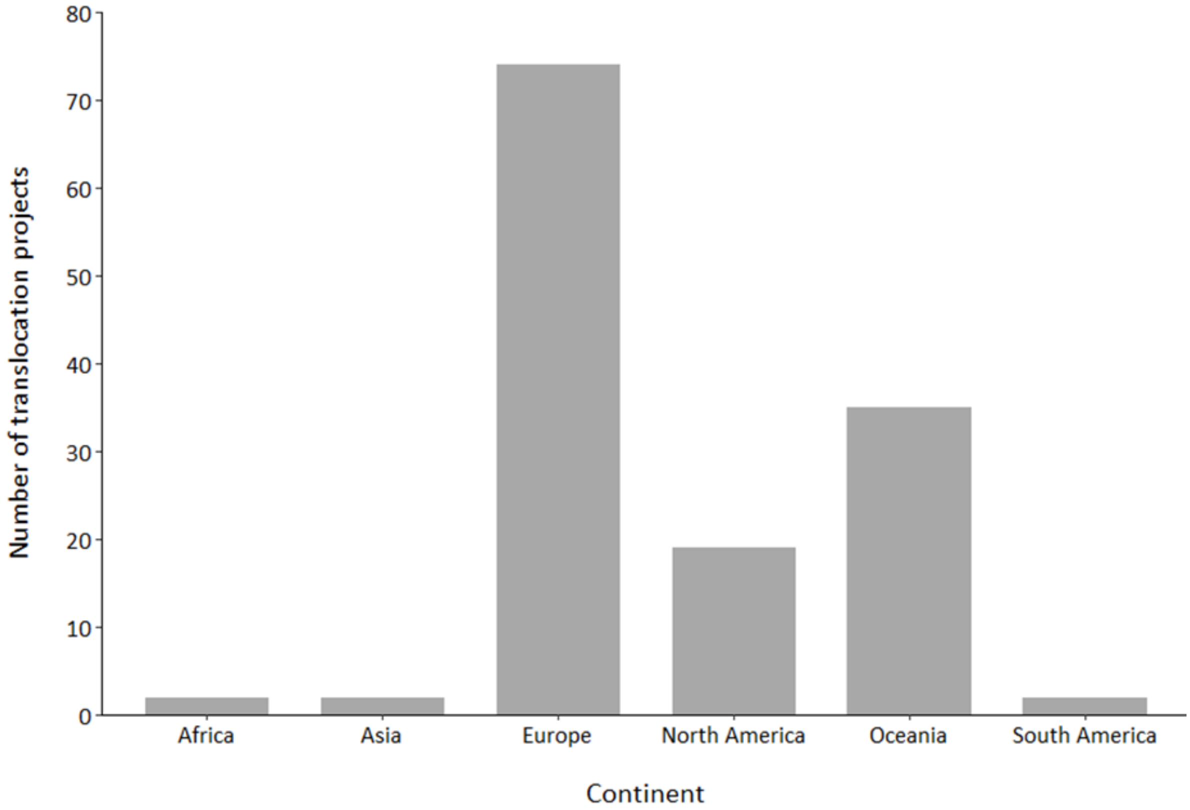

Figure 2. Number of terrestrial insect translocations reviewed by continent (*n*=134).

Table 2. Information-theoretic model selection results for models relating predictor variables with the probability of successful translocation of terrestrial insect species. Number of estimable parameters (k), the second order Akaike Information Criterion (AICc), the Akaike differences (Δ_i) and the Akaike weights (w_i) are presented.

Model description	**K**	**AICc**	Δ_i	w_i
NumRel	2	104.27	0	0.19
NumRel + Origin	3	104.96	0.69	0.13
NumRel + LifeHistory	3	105.87	1.6	0.08
Origin	2	106.38	2.12	0.06
LifeHistory	2	106.40	2.14	0.06
NumRel + NRelYears	3	106.42	2.15	0.06
NRelYears	2	106.78	2.52	0.05
LifeStageRel	4	106.86	2.59	0.05
NumRel * Origin	4	106.97	2.71	0.05
NumRel * LifeHistory	4	108.09	3.82	0.03
NumRel * NRelYears	4	108.15	3.88	0.03
Origin + LifeHistory	3	108.16	3.89	0.03
NRelYears + Origin	3	108.34	4.07	0.02
NRelYears + LifeHistory	3	108.43	4.16	0.02
LifeStageRel * LifeHistory	8	108.46	4.2	0.02
LifeStageRel + LifeHistory	5	108.46	4.2	0.02
NRelYears + LifeStageRel	5	109.14	4.87	0.02
Origin + LifeStageRel	5	109.14	4.87	0.02
Origin * LifeHistory	4	109.33	5.06	0.01
NRelYears * Origin	4	109.75	5.49	0.01
NRelYears * LifeHistory	4	110.41	6.14	0.01
NRelYears * LifeStageRel	7	110.99	6.73	0.01
Origin * LifeStageRel	8	111.02	6.76	0.01

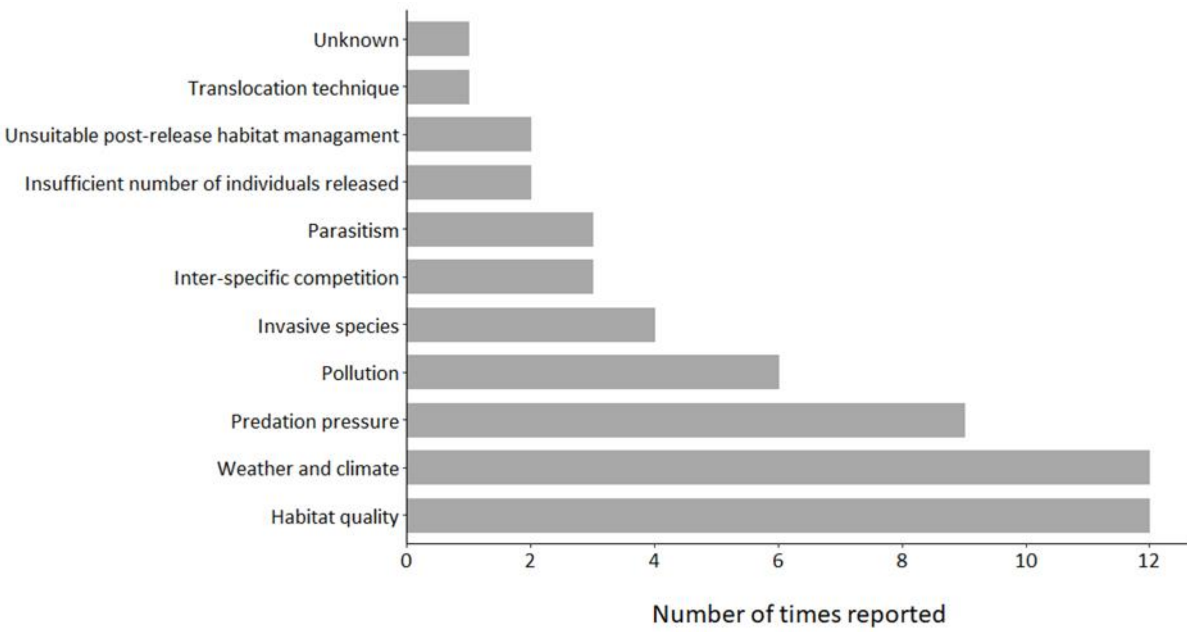

Figure 3. Factors reported as influencing the failure of terrestrial insect translocations (*n*=33). Several influential factors may have been reported for a single translocation project.