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Determinants of rapid response success for alien invasive species in aquatic ecosystems

By

Boris Beric

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2014

Determinants of rapid response success for alien invasive species in aquatic ecosystems

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Abstract

Alien invasive species (AIS) have received much attention for their harmful effects on health, ecology and the economy. Although the best approach is prevention of introductions, it is imperative that rapid response (RR) countermeasures be available, should prevention fail. I analyzed 127 cases involving RR to AIS in aquatic systems. Results indicated the rate of eradication success was greater, and slightly higher, for plant versus animal AIS, and when chemical versus mechanical methods were used, respectively, but was unaffected by habitat size. Suppression of AIS was most successful in small habitats and with chemical versus mechanical methods, but was unaffected by taxonomy (plant or animal). Outcome was not affected by the population size, project duration, ecosystem (marine or freshwater), or number (single or multiple) of methods used. Managers should expect that different factors will affect success depending on whether intervention aims for complete elimination or population reduction of AIS.

Dedication

To:

My family and friends, for their moral and professional support throughout my academic career.

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Introduction

The volume of invasion ecology literature has increased dramatically as the impacts of alien invasive species (AIS) introductions have garnered greater academic and government attention (Richardson & Pysek, 2008). It is important to acknowledge that many alien species are beneficial to mankind by providing food, ecosystem restoration, pest control or other benefits (Pimentel *et al*., 2005). However, AIS are defined as non-native species whose introduction and/or spread harms or threatens to harm biological diversity, economies, or human health (CEC, 2003). Most attention on AIS focuses on their negative impacts on ecosystem nutrient cycling, crop losses, or reduced abundances or diversity of native species owing to predation, competition, disease or parasitism (Mack *et al*., 2000). Moreover, in an analysis of species from the IUCN Red List database, Clavero and Garcia (2005) found that, out of 680 cases, AIS were the primary cause of species extinctions in 34 cases and a contributor to extinctions in 170 others.

The cost of damage to the global economy from biological invasions has been estimated by the Global Invasive Species Programme to be $1.4 trillion annually (UNEP, 1993). In the USA, economic losses due to AIS damage costs approximately $120 billion per year (Pimentel *et al*., 2005), while in Canada, only 18 AIS cost the economy between $13.2 and $34.8 billion in actual and potential economic losses (Colautti *et al*., 2006).

Some authors attribute recent increases in the rate of biological invasions, as well as the severity of impacts, to increasing rates of global trade (Hulme, 2009), with regions of high economic development (Lin *et al*., 2007) and large landmass (Tatum *et al*., 2006) being the most susceptible. Countries meeting these characteristics, including Canada, are especially vulnerable to the establishment of AIS. It has also been suggested that the current rate of biological invasion and damages associated with them are unprecedented in Earth's history (Ricciardi, 2006), and that only a handful of aquatic and terrestrial systems still remain immune to the effects of AIS (Mack *et al*., 2000).

Although a substantial portion of Canada's government funding on AIS is allocated for damage control (Colautti *et al*., 2006), impending threats pose a particular problem due to lack of information for management and/or from changes in global environmental conditions associated with climate change. In the Canadian Arctic, for instance, a continual rise in temperature has resulted in accelerated ice sheet retreat (IPCC, 2013), which may facilitate future invasions through increased surface currents passively introducing new AIS, or by human-mediated introductions associated with enhanced ship traffic and consequent ballast or hull fouling introductions. Enhanced food supply and more suitable environmental conditions for AIS that arrive could also increase establishment success (Vermeij & Roopnarine, 2008). Thus, the Arctic is a region of the country especially at risk of new invasions and plans are needed both to prevent invasions and to eradicate AIS that do establish.

In response to the environmental and economic threat posed by AIS, the Convention on Biological Diversity (CBD) was approved on December 1992 following the Rio summit, and requires countries to prevent invasions and develop countermeasures to address AIS established within their borders (UNEP, 1993; Government of Canada, 2004a). In 1995, Canada’s Biodiversity Strategy was released, which recognized that AIS are a threat to ecosystems, and that procedures were required to manage their impact on biodiversity. A 2002 audit by the Office of the Auditor General of Canada revealed that federal programs were lacking in preparedness for addressing the threat of biological invasions, in contrast to requirements of the CBD Convention (UNEP 1993; Office of the Auditor General, 2002). In response, the Canadian government released An Invasive Alien Species Strategy for Canada (Government of Canada, 2004a) which noted four key areas of concern: i) prevention of new invasions; ii) early detection of new invaders; iii) rapid response to new invaders; and iv) management of established and spreading invaders. Canada then adopted the Canadian Action Plan to Address the Threat of Aquatic Invasive Species, which cited ‘risk assessment’ of AIS invasions as a priority area (Government of Canada, 2004b). However, by 2008 there was still an evident gap in addressing priorities ii) and iii), indicating that an urgency existed with respect to research required in these areas (Office of the Auditor General of Canada, 2008).

Many countries, including Canada and the USA, recognize early detection and rapid response (EDRR) as top priority areas in their AIS management plans (Waugh, 2009). Early detection (ED) provides immediate warning signs of the presence of AIS and includes a combination of surveys, species verification, and archiving methods (Waugh, 2009; NISC, 2013). Rapid response (RR) is the capacity to respond to detected AIS and prevent or manage their establishment in a new location in a timely manner (McEnnulty *et al*., 2001). RR is considered the second line of defence against AIS if prevention has failed, with the ultimate goal being eradication (Locke & Hanson, 2009; Dimond 2010). In February 2014, Ontario released a strategic plan for addressing AIS in the province, including a commitment to comprehensive RR programming (LAO, 2014). Eradication is the "removal of every potentially reproducing individual of a species or the reduction of their population density below sustainable levels" (Myers *et al*., 2000). However, although complete removal of AIS populations is ideal, it is not always achieved.

Blackburn et al. (2011) developed a framework that depicts different stages of biological invasion by AIS, as well as corresponding management options for stakeholders. AIS begin in the Transport stage, and progress to Introduction, Establishment, and finally Spread. During each of these stages, there exist complementary management goals. This thesis considers only the Stage and Management sections of the framework. This model recognizes that prevention is the first management priority in dealing with AIS, which may be detected during the Transport or Introduction stages. The next option, eradication, is exercised only if prevention measures have failed, and if AIS are detected in later stages. It is during the early stages of invasion - associated with the period of early population growth - that RR measures are critical, as they determine whether AIS progress into successive stages. Eradication is considered economical and environmentally-friendly compared to control-the-spread or population suppression measures, which seek to constrain species' distributions or reduce species abundances, respectively (Peay, 2006). If eradication is not possible, a control-the-spread strategy may limit population growth and spread, and hence the damage associated with the AIS. In the case of the gypsy moth (*Lymantria dispar*) in the Great Lakes region, over $25 million was spent across 25 years in attempted eradication, to no avail (Tobin & Liebhold, 2011). Eventually, management programs focused instead on slowing the spread of the moth via pheromone traps and aerial spraying along the population's invasion front. Thus, although countries typically prioritize pre-incursion strategies, there are many instances where such measures fail or are impossible to implement (Hein *et al*., 2007), especially in aquatic ecosystems (Dimond, 2010). It is vital for countries to develop a suite of RR countermeasures for all scenarios should prevention fail. New Zealand, for instance, has RR protocols for eradication, control-the-spread, and suppression of AIS for use in both freshwater and marine habitats (Forrest *et al*., 2009).

In recent years, Canada has sustained multiple AIS introductions and lacked protocols to deal with them. For example, when the European green crab (*Carcinus maenas*) was detected in Newfoundland in 2007, the Department of Fisheries and Oceans did not immediately know what to do, though it eventually settled on a massive 'fishing' effort to dramatically suppress population abundance (DFO, 2011). Currently there is no universal reference guide for managers in Canada, thus AIS interventions are typically undertaken based on very limited information (Drolet *et al*., 2013). In addition to having the necessary tools available, it is also important that assessment tools be timely and user-friendly (DFO, 2009), as detailed species-based risk assessments commonly take considerable time to prepare, leading to loss in RR capacity owing to time delays. Development of RR strategies requires that key factors governing AIS management outcome be understood and made readily available for end-users. Since AIS identities and their impacts vary considerably, developing robust support models for selecting different management countermeasures is a challenging problem. In this thesis, I aim to provide a quantitative foundation for the development of a general RR decision support model that managers may utilize for implementing intervention programs to address aquatic AIS globally.

What factors affect success of rapid response?

Several studies have attributed different factors to the success or failure of their AIS intervention campaigns, but there appears to be minimal agreement with regard to universal determinants of management outcome. It may especially difficult to assign a key factor to all management campaigns as each project typically carries their own set of obstacles. Thus, in some situations, public support may be critical before a removal project may commence (ADFG, 2011), while in others, logistics or budget availability play a more dominant role (Woodfield & Merkel, 2006; Twohey *et al*., 2003). Moreover, analyses into which factors significantly contribute to management success greatly depend on observations made by authors of management studies, as well as the level of detail with which observations were recorded. Thus, if certain factors were important, but unrecognized, they will certainly not be not be included in reports and, in result, will go unnoticed by other authors. Alternatively, data may be catalogued by researchers using a unique standard, making it difficult to extend and compare findings to other studies, leading to loss of accuracy due to attenuation.

In searching for key factors that apply to all AIS RR projects, it may be reasonable therefore, to focus on variables that are both intuitively connected to project outcome, as well as those that are typically reported by researchers. Thus, although many different factors have been suggested to affect the success or failure of RR in aquatic environments, I catalogued those which I suspected to be logically connected to management outcome, while also being readily accessible in the literature. Locke and Hanson (2009) noted that the type of ecosystem that AIS were introduced to, marine or freshwater, could affect RR success. Cases of successful eradication in marine ecosystems have been recorded, such as the killer algae (*Caulerpa taxifolia*) near San Diego, California (Anderson, 2005), and black striped mussel (*Mytilopsis sallei*) in Darwin, Australia (Ferguson, 2000), though eradication appears to be overall less common in marine ecosystems as compared to terrestrial or freshwater ones (Locke & Hanson, 2009). Managers typically resort to control-the-spread, or suppression strategies in these systems instead (Locke *et al*., 2009). One possible explanation for the difference in success within these environments is that the rate of AIS introduction is much higher in marine ecosystems, due to operation of major pathways like ballast water release and hull fouling, pathways that are most potent in marine environments (Gollasch, 2005). Another pathway that is more potent in marine habitats is the aquarium trade (Padilla & Williams, 2004). In Prince Edward Island, containment of solitary tunicates (*Styela clava* and *Ciona intestinalis*) and colonial tunicates (*Botryllus schlosseri* and *Botrylloides violaceus*) was the only feasible management option in the open marine environment. In this case, regulation of aquaculture transfer was used to minimize the spread of solitary tunicates but was unsuccessful for colonial tunicates. Thus, there may be a discrepancy in success of eradication based simply on ecosystem type.

In every AIS management project, managers must choose amongst various methods of control, including mechanical removal, biological agents and/or chemicals. The choice of method may be pivotal to project success. In Crystal Lake, Wisconsin, workers employed induced thermal mixing, which took advantage of rainbow smelt's (*Osmerus mordax*) intolerance of warm environments (University of Wisconsin-Madison, 2013). Triploid grass carp (*Ctenophmyngodon idella*) was used as a biological control method against hydrilla (*Hydrilla verticillata*) in Imperial Country, California, as a more reliable, cheaper, and environmentally friendly alternative to herbicides (CDFA, 2014). In the attempted eradication of the European fan worm (*Sabella spallanzanii*) from Lyttleton and Waitemata Harbours, New Zealand, manual removal efforts were initially considered the most feasible means of management (Read *et al*., 2011). However, fan worm populations grew quickly and eradication was no longer feasible, nor were other methods. Another example where the choice of method was important, was in the removal of topmouth gudgeon (*Pseudorasbora parva*) from Goldings Hill Pond, London, England (Copp *et al*., 2007). Electrofishing was initially used upon discovery of the AIS, but managers then decided to dewater the pond when reoccurring gudgeon were found. Following the drawdown, the species quickly disappeared.

Although managers do not have the luxury of trial and error with AIS interventions, a combination of management methods may increase success as compared to a single method approach. For example, the addition of biological control methods to augment mechanical ones contributed to the management of rusty crayfish (*Orconectes rusticus*) in Sparkling Lake, Wisconsin (University of Wisconsin, 2013), and common carp (*Cyprinus carpio*) in Centennial Park, Sydney, Australia (Centennial Parklands, 2013). In both examples, biological control was added after initial mechanical methods were insufficient to eradicate AIS. In an extreme example of multiple methods, the eradication of hydrilla from Yuba County, California employed a total of 19 separate methods before signs of successful eradication were achieved (CDFA, 2014). Thus, there is some uncertainty in eradication success in regards to when managers should use single or combined methods.

Another less-studied factor that may influence RR success is the taxonomy of the AIS. For example, when considering removal of animal AIS, managers must consider methods that account for targets being able to hide and evade capture. For instance, during the removal of signal crayfish (*Pacifastacus leniusculus*) in Scotland, trapping efforts were rendered more difficult by crayfish burrowing in muddy pits, and from smaller size classes being more evasive than larger ones (Peay *et al*., 2006). This scenario is also important in management of alien invasive fish, such as in the attempted eradication of round goby (*Neogobius melanostomus*) in Pefferlaw Brook, Ontario (Dimond *et al*., 2010), where fish size and mobility made them very difficult to detect and capture. Alternatively, eradication of plant AIS often involves manual removal before employing biological or chemical methods, unless otherwise suggested by previous experience. For example, during the eradication of hydrilla from Tulare, Shasta, Calaveras and Imperial County, California, chemical treatments were employed only after it was discovered that manual removal was incapable of removing populations (CDFA, 2014). In some cases, the dispersal capability of plants was underestimated, leading to infestations in areas that were originally AIS free. The removal of giant salvinia (*Salvinia molesta*) from Caddo Lake, Louisiana/Texas, was rendered difficult because of the plant's high reproductive capacity and difficulty in detecting remaining fragments (TWRI, 2013).

Many authors have noted that the initial population size of AIS has a large impact on management actions employed, and the resulting outcome. The successful removal of topmouth gudgeon from Goldings Hill Pond, London, England, was attributed the small initial population abundance (Copp *et al*., 2007). Similarly, population abundance was a key variable for managers in combating the sabellid polychaete (*Terebrasabella heterouncinata*), near Cayucos, California (Culver & Kuris, 2000). In this case, the success of eradication heavily depended on lowering the AIS population below the minimum viable population size. In the campaign against gypsy moth in Wisconsin and North Carolina, a patch size threshold existed below which populations could not persist due to

Allee effects (Vercken *et al*., 2011). Sharov and Liebhold (1998) developed a model illustrating that eradication success was optimal when the extent of AIS infestation was low, and that alternative means of management were necessary when population size was larger. Miller et al. (2005) also considered the limited patch size of brown alga (*Ascophyllum nodosum*) to be a determinant factor in its successful removal from San Francisco Bay, California.

The eradication of AIS can also be affected by the surface area that agencies are forced to manage. McEnnulty et al. (2001) suggested that eradication should not even be attempted unless AIS are in very isolated areas. Larger surface areas require more manpower as compared to smaller ones, especially for manual removal projects. Managers quickly realized that spread of sea Lamprey (*Petromyzon marinus*) in Lake Superior, for instance, was inevitable due to the difficulty of detecting and capturing the entire AIS population spread across a 8,000,000 ha habitat (Twohey *et al*., 2003). On the other hand, some small-scale eradications were successful simply because AIS were in very isolated habitats. Hydrilla was found in small ornamental ponds in Yuba County, Tulare County, and Los Angeles, California, and was quickly eradicated by manual removal (CDFA, 2014). Similarly, pond burials were extremely effective in eliminating the same AIS in Shasta County, California because surface areas of ponds were less than 10 ha each (CDFA, 2014). Even in cases where AIS are mobile and difficult to capture, a relatively small isolated habitat can lead to successful AIS eradication. This was the outcome for northern pike (*Esox lucius*), which were eradicated from Lake Davis, California (~1500 ha) using a combination of chemical application (rotenone) and manual removal (Borucki, 2007). It seems plausible, then, that surface area of managed habitats may influence the outcome of AIS eradication.

Finally, the management project’s duration may contribute to eradication success. Many authors have suggested that their campaigns were successful due to quick detection and timely action against AIS. For example, Culver and Kuris (2000) noted that quick management initiative, in response to the invasion of the sabellid polychaete near Cayucos, California, was one of the factors that contributed to their success. McEnnulty (2001) proposed that one of the factors important to success against the black striped mussel in Darwin, Australia was the short time frame between detection and action by managers. In other situations, such as the control of water lettuce (*Pistia stratiotes*) in Kruger National Park, South Africa, short-term management was unsuccessful and managers then focused on a long-term strategy (Cilliers *et al*., 1996). An underlying view of the role of project duration in management success is discussed by Bender *et al*. (1984) in terms of ‘pulse’ versus ‘press’ perturbations. In a pulse perturbation, stress is applied to species populations only once, resulting in typically drastic reductions in population abundance, while press perturbations involve a continually applied long-term stress (e.g. management effort). It is possible that some species are more effectively managed using pulse-type intervention, such as the case near Cayucos, whereas others are more successfully managed by press-type intervention, such as in Kruger National Park.

In this study, I test eight hypotheses about key factors potentially important to management success: i) RR success is equally effective in marine and freshwater ecosystems; ii) chemical methods are equally effective in RR as mechanical ones; iii) single-method management approaches are equally effective as those undertaken with multiple-method strategies; iv) RR applied to plants has an equal success rate as that applied to animals; v) population abundance has no bearing on success of RR programs; vi) infestation extent has no bearing on success of RR programs; vii) habitat area treated by management agencies has no bearing on RR success; and viii) the duration of management projects has no bearing on RR success. Each hypothesis was investigated with respect to AIS eradication and suppression projects, as the success rate of interventions could differ based on the goal of managers (Locke *et al.*, 2009).

This project employed both null hypothesis significance testing (NHST) and a meta-analytical approach to test the above-mentioned hypotheses (Harrison, 2011). I followed the procedure for conducting a meta-analysis discussed by Harrison (2011), which ultimately allowed me to compare RR program results via a rigorous quantitative scale. Meta-analysis allows for the discovery of new findings based on combinations of published data on a specific hypothesis, in larger, synthetic analyses (Harrison, 2011). One of the strengths of meta-analysis is that it increases confidence of results, which may otherwise lack statistical power due to sample size limitations. Harrison (2011) suggested meta-analysis be conducted using the following six steps: i) a literature search where defined keywords and a reproducible method of search is undertaken, including searching for grey literature through personal communication; ii) development of inclusion criteria, including a record of discarded papers, with supporting reasons; iii) choosing an effect size appropriate to the type of data collected (mean difference, correlation coefficient or odds ratio, as appropriate); iv) cataloguing all data, including independent variables, dependent variables, effect size calculations, and references; v) implementation of the meta-analysis and interpretation of conclusions; and vi) assessment of the robustness of the study by considering the likelihood of type 1 and type 2 error rates. However, step vi) was instead accounted for by the use of confidence intervals, rather than a post-hoc power analyses, as this was suggested as being a more reliable measure of the error rate, especially for nonsignificant findings (Colegrave & Ruxton, 2002).

Methods

Data collection

I assessed RR successes and failures via vote-counting and meta-analysis of published and unpublished, grey literature. In order to increase access to published, as well as 'grey', literature, I performed a combined literature search using Google, Google Scholar, Thomson Reuters Web of Science v5.11, acknowledgment sections of publications, and personal communications. I utilized Google and Google Scholar between May 1, 2011 and August 31, 2013, to locate peer-reviewed publications or public reports on specific case studies, which were referred to me by authors or peers. This search yielded a total of 157 and 34 studies from Google and Google Scholar, respectively. Additionally, I searched Thomson Reuters Web of Science for papers published between 1965 and 2013, with the following keywords in the ‘title’ section: alien, invasive, exotic, nonnative, nonindigenous, introduced, pest; and combined this search with manage*, campaign, program, eradicat*, exterminat*, eliminat*, suppress*, mitigat*, remov*, reduc*, or restor*. This search produced 1,669,667 results. In order to refine the number of potential papers for review, I conducted a second search using the same keywords but including only the following Web of Science Research Areas: agriculture, engineering, plant sciences, environmental sciences ecology, marine freshwater biology, public environmental occupational health, science technology other topics, operations research management science, life sciences biomedicine other topics, forestry, rehabilitation, water resources, and fisheries. This second search yielded 467,275 publications, of which I deemed the first 202 to be of sufficient sample size for review. Some of these papers, however, were not readily accessible online. Therefore, I contacted authors directly and obtained five such papers and reports. In total, I reviewed 393 published papers and reports during this literature search, of which I incorporated 89 (127 case studies) into my final dataset, and discarded the remaining 304.

I considered treatments at separate study sites as independent case studies. In cases where multiple AIS were present during treatment, or where study sites were physically connected, I considered cases to only be truly independent if authors declared that populations were isolated from one another. In cases where study sites were physically connected and separate chemical or biological methods were employed at each site, I considered both sites affected unless authors claimed that treatment effects had not overlapped.

Missing data

In many cases, reports had not disclosed either dependent or independent variables that I sought to collect. In these situations, I conducted an additional Google search for specific data, attempted contacting authors directly, or, in cases of missing continuous variables, estimated them using Image J v1.47$^{(R)}$ software. I utilized Image J in instances where papers provided graphical images of data without accompanying text or numerical tables. Image J allows end-users to upload a digital image file and measure area and/or distance within plots by calibrating the software's internal pixel scale with that of a known measurement unit. I used Image J to estimate surface areas and stream lengths from maps of study sites, and population abundance and infestation extent from diagrams. When estimating mean river width, I made a total of five measurements along separate river sections and calculated an average value.

Statistical analyses

I performed the following univariate statistical analyses using IBM SPSS v.20, where I observed general relationships between different predictor variables and the outcome. In order to test hypotheses i), ii) and iv) with respect to eradication success, I performed a chi-square test using 108 of 127 available cases, and tested whether the proportion of successful eradications varied for different independent variables. Each test contained a binary response variable of failed or successful eradication, which I recorded as votes based on authors' observations for each case study. For hypothesis iii), I used Fisher's exact test instead of the chi-square test because cells of the contingency table contained expected values that were below five, thus violating the chi-square assumption (Field, 2009). For hypotheses i) through iv), the binary independent variables were freshwater or marine, chemical or mechanical, single method or multiple methods, and animal or plant, respectively. I employed binary logistic regression for the same 108 cases to test hypotheses v) through viii) with regard to eradication success by assessing the goodness-of-fit of data using the log-likelihood statistic. The statistic is a χ^2 value in SPSS, and is the difference between the log-likelihood of the model when the independent variable is absent and when it is included (Field, 2009). The outcome variable was a binary 'success' or 'failure', but independent variables were all continuous. I used the following independent variables to test hypotheses v) through viii), respectively: population abundance, in number of organisms; infestation extent, in hectares; study site surface area, in hectares; and project duration, in months.

In order to investigate hypotheses i) through viii), where the goal of projects was suppression of AIS populations rather than their eradication, I used parametric tests for the remaining 19 of 127 case studies. I recorded a continuous outcome variable for the suppression studies, used for hypotheses i) through vi), which was the log response ratio (R), as a measure of 'effect size' (Paolucci *et al.*, 2013). This value is: $R = \log([X_{final} / X_{initial}] + 1)$, where X_{final} and

$X_{initial}$ represent the population size (in units of abundance or surface area, depending on the case study) after and before suppression program implementation, respectively. Thus, larger values R indicate that AIS populations are larger after intervention than before, and that suppression was relatively unsuccessful compared to smaller values. For hypotheses i) through iv), I conducted an independent t-test to determine whether or not the R means differed between groups. The predictor groups for hypotheses i) through vi) were freshwater or marine ecosystem, chemical or mechanical method, single or multiple approach, and animal or plant taxonomy, respectively. For hypotheses i) and ii), a one-sample t-test was computed, because each predictor variable contained one group which consisted of only a single case study. Specifically, for the ecosystem type predictor, there was only a single marine study versus 18 freshwater studies. Similarly, for the method type variable, there was only one chemical methods study, compared to 13 cases of mechanical methods. For hypotheses iii) and iv), a two-sample t-test was used because both predictor groups were of sufficient sample size. For hypotheses vi) through viii), I used linear regression to assess whether there was a relationship between R and each independent continuous variable. Independent variables included: population abundance (in number of organisms), infestation extent (in hectares), surface area (in hectares) and project duration (in months).

Variable definitions

I utilized the following criteria during the data cataloguing process. I defined project duration as the length of time between the reported launch date of a management program and the end of final survey or project termination date, in months (whichever was later). In cases where projects were ongoing at the time of data retrieval, I used the most recent date of project activity (surveying or removal efforts) as the end date. Furthermore, for any dates reported by authors in months, I rounded the start date to the nearest first day of the month, and the project end date to the nearest last day of the month before. For example, a project described as lasting from May 2003 to August 2003, was rounded to May 1, 2003 to July 31, 2003. I did not subtract periods of project inactivity from the total project duration because projects were considered ongoing in all cases by authors. The mechanical methods that I catalogued consisted of dredging, drawdowns, screen installations, electrofishing, manual removals, raking, pond/canal lining, and/or trapping. Chemical methods included application of herbicides, pesticides, piscicides, or other toxic substances used to eliminate AIS. Among the cases I reviewed, I found no cases where only biological methods were employed, and therefore chose to exclude biological methods from the independent variables used in this study. I grouped methods that fell under the same category (mechanical or chemical) together for each case study when testing hypothesis ii). Therefore, a case that involved manual removal and electrofishing was considered a mechanical method approach, which did not discriminate among the number of mechanical methods used. However, I developed a separate category, in hypothesis iii), to differentiate whether one or multiple methods were used.

Assumptions of statistical tests used

I performed several additional tests to explore assumptions of both parametric and nonparametric tests prior to each analysis. If I discovered that any assumptions were violated, I transformed variables accordingly. Specific transformations are mentioned in the description of each analysis described below.

The chi-square test has two assumptions: i) independence of data and ii) expected cell counts greater than five for more than 25% of cells (Field, 2009). In order to meet assumption i), I treated study sites as separate case studies in any situations where I believed that the effects of treatment were not truly independent of one another. In some instances, authors mentioned that populations were isolated from one another or that barriers were installed to physically separate study sites. I made exceptions in such cases and considered study sites as independent of one another. For assumption ii), I utilized Fisher's exact test in situations where expected cell counts were less than five for any cells in the contingency table. This situation arose when evaluating the relationship between ecosystem type and eradication success.

Binary logistic regression has two assumptions: i) linearity between the independent variable and log independent variable, and ii) independent errors (Field, 2009). To test assumption i) I performed a binary logistic regression using the response variable ('success' or 'failure') and the interaction between each continuous variable (population abundance, infestation extent, habitat surface area, or project duration) and its log transformation as the independent variable (Hosmer & Lemeshow, 1989). I observed using the Wald statistic (Z) whether this interaction term contributed significantly to the regression model, in which case non-linearity was evident (Field, 2009). Z is a measure of the contribution of a predictor variable to the response, which if significant illustrates that a predictor variable significantly contributes to the model's predictive power. I evaluated assumption ii) by looking for overdispersion in the data using the dispersion parameter (Φ), which is the ratio of the model's chi-square statistic to its degrees of freedom (Field, 2009). Overdispersion is a cause for concern when Φ is outside the range of 1 to 2. I performed data transformations of population abundance and surface area in order to meet the above assumptions. For population abundance, I used the square root-transformation, and a $^4\sqrt{}$log (log [surface area + {1/surface area} + 200) transformation for surface area.

Assumptions of t-tests include: i) homogenous variance between groups; ii) normality of group data; iii) independent data; and iv) using a continuous outcome variable (Field, 2009). In order to assess assumption i), I used Levene's test for homogeneity of variance, which, if significant, indicates a violation (Field, 2009). I did not perform this test when evaluating hypotheses i) and ii) due to only having a single case study for the marine ecosystems and chemical methods groups. However, I considered this assumption met in these cases because groups with comprehensive studies had small variances (0.119

for freshwater ecosystems, 0.117 for mechanical methods), compared to variances of zero for both point estimates. I observed the data skewness and kurtosis statistics, from SPSS, to determine whether groups were normally distributed, for assumption ii) (Kim, 2013). Samples are considered normally distributed, at $P<0.050$, when the standardized skewness ($z_{skewness}$ = skewness statistic/standard error) and kurtosis ($z_{kurtosis}$ = kurtosis statistic/standard error) statistics are within the range ±1.96. I addressed assumption iii) by ensuring that data was retrieved from completely separate case studies, and by combining cases when treatment effects were not independent of one another. Finally, I met assumption iv) by using the log response ratio, R, as the continuous outcome variable. I transformed R, the dependent variable, during each test in order to meet the above assumptions. I transformed R for all of freshwater ecosystems, mechanical methods, plant taxonomy and animal taxonomy groups, using a fourth root-transformation. The single methods and multiple methods groups were transformed using the formula $\mathrm{Sin}(e^{\sqrt{R}})$. I did not perform transformations of the marine ecosystems nor chemical methods groups because each consisted of only a single case study.

Linear regression has eight assumptions (Berry, 1993): i) continuous dependent and independent variables; ii) non-zero variance within predictors; iii) no correlations between predictors and external variables; iv) homoscedastic variance; v) linearity between response and predictor; vi) normality of residuals; vii) independent data; viii) independent errors. I realized assumption i) by using R as the continuous response variable, and using all continuous independent variables (population abundance, infestation extent, surface area, project duration). For assumption ii), I collected a wide range of data for each predictor variable to ensure non-zero variance. I met assumption iii) by collecting data for different factors which I believed to contribute to suppression success, and tested them separately in order to observe their 'main effects'. I tested for correlations only if more than one variable contributed significantly to suppression success for any given statistical test. Next, I plotted the residual z-scores versus predicted z-scores to evaluate assumptions iv) and v) as per Field (2009). The resulting scatterplot is expected to display a random arrangement of data points, if both assumptions are met. If the data points are highly scattered on one end of the plot, but very clustered on the other, referred to as 'funneling', then heteroscedasticity is present. If data points display a trend across the plot, the relationship is non-linear. Next, I assessed the z-skewness and z-kurtosis of the standardized residuals to test assumption vi). As above, I observed whether each statistic was within the range ±1.96, in which case the residuals were normal (Kim, 2013). I met assumption vii) by ensuring that case studies where treatment effects impacted more than one suppression campaign, were treated as a single case study, unless otherwise recommended by authors. I evaluated assumption viii) by using the Durbin-Watson statistic (d) (Field, 2009). Errors are considered independent when d is within the range 1.5-2.5 (Garson, 2012). In order to satisfy all of the above assumptions, I transformed both the independent and dependent variables during each linear regression analysis. In the case of population abundance versus R, I transformed the former via a log-transformation, and the latter using a tenth root-transformation. In the case of infestation extent and R, I used a log-transformation and Sin(e^[^{3}√R]) transformation, respectively. I used a log-transformation for surface area and a ^{10}√(R+0.01) transformation for R, for surface area versus suppression success. Lastly, I transformed project duration and R using log and Sin(e^[^{4}√{R/1.9}]), respectively.

Results

I found no relationship between ecosystem type (marine vs. freshwater) and eradication success using Fisher's exact test (N=108, P=0.999, 95% CI=± 0.145; Table 1; Appendix 3). I observed a marginally significant relationship between method type (chemical vs. mechanical) and eradication success, with chemical methods being more effective than mechanical ones (N=71, χ^2_1=3.504, P=0.061, 95% CI=± 0.088). Next, the number of methods (multiple vs. single) had no effect on eradication success (N=108, χ^2_1=1.181, P=0.277, 95% CI=± 0.0.081). In contrast, I found that species taxonomy was significant, with plants successfully eradicated more often than animals (N=108, χ^2_1=9.366, P=0.002, 95% CI=± 0.081; Figure 1). I discovered nonsignificant relationships, in all cases, between population abundance, infestation extent, surface area or project duration, and eradication success, using binary logistic regression analysis (N=23, β=0.001, χ^2_1=1.236, P=0.266, 95% CI=± 0.001; N=85, β=-0.001, χ^2_1=1.939, P=0.175, 95% CI=± 0.002; N=108, β=-12.696, χ^2_1=0.671, P=0.398, 95% CI=± 29.473; N=108, β=-0.004, χ^2_1=1.523, P=0.217, 95% CI=± 0.006, respectively; Table 2; Appendix 4).

There was no relationship between ecosystem type and suppression success, using the t-test (N=19, $\overline{x}_{\mathbf{R}\text{freshwater}}$=0.508, $\overline{x}_{\mathbf{R}\text{marine}}$=0.506, t_{17}=0.019, P=0.985, 95% CI=± 0.172; Table 3; Appendix 5). However, case studies that used chemical intervention methods, had greater suppression success than those in which mechanical methods were used (N=14, $\overline{x}_{\mathbf{R}\text{chemical}}$=0.000, $\overline{x}_{\mathbf{R}\text{mechanical}}$=0.462, t_{17}=4.877, P=0.001, 95% CI=± 0.206; Figure 3). I also found that the number of methods used had no significant effect on suppression success (N=19, $\overline{x}_{\mathbf{R}\text{multiple}}$=0.943, $\overline{x}_{\mathbf{R}\text{single}}$=0.886, t_{16}=1.728, P=0.102, 95% CI=± 0.102). Next, I found that taxonomy had no effect, as plant and animal AIS were equally affected by suppression (N=19, $\overline{x}_{\mathbf{R}\text{animal}}$=0.507, $\overline{x}_{\mathbf{R}\text{plant}}$=0.511, t_{16}=-0.020, P=0.984, 95% CI=± 0.381). I observed no significant relationship between population abundance and suppression success, using linear regression (N=14, R^2=0.077, $F_{1,12}$=1.006, P=0.336, 95% CI=± 0.912; Table 4; Appendix 6). The relationship between infestation extent and suppression success was also nonsignificant (N=5, R^2=0.342, $F_{1,3}$=1.557, P=0.301, 95% CI=± 0.269). However, I discovered a significant negative relationship between habitat surface area and suppression success (N=19, R^2=0. 243, $F_{1,17}$=5.449, P=0.032, 95% CI=± 0.169; Figure 4). Lastly, I found that project duration of suppression campaigns had no influence on the degree of suppression success (N=19, R^2=0.002, $F_{1,17}$=0.036, P=0.851, 95% CI=± 0.169).

Discussion

Prevention of new introductions is the top priority in all national and provincial action plans designed to manage the threat of AIS. In many circumstances, prevention measures fail, leading in some cases to severe and irreparable damage to fisheries, eutrophication of lakes, blockage of waterways, and even spread of fatal diseases (Pysek & Richardson, 2010). When agencies are faced with the task of responding to newly introduced AIS, in most cases time, money, or other key resources mean the difference between a short-term, successful cleanup effort and billions of taxpayer dollars spent on long-term management.

In this study, I discovered that consideration of species taxonomy was significant to eradication success, with plant success rate surpassing that of animals (Figure 1). Sample sizes for plants (61) and animals (47) were fairly large, yet 89% of plants were successfully eradicated as compared to only 64% of animals. The underlying reason for this difference could involve the mobility of the AIS, where plants are 'sitting ducks' compared to animals in terms of being captured or affected by an herbicide. Alternatively, the eradication of plants may take longer to confirm as compared to animals, leading a higher false positive rate for plant interventions. For instance, the eradication of hydrilla in California took more than 20 years to achieve in several regions including Yuba, Calaveras, and Imperial counties (CDFA, 2014). In all situations, the plant had appeared on at least one occasion after it was thought to be completely eliminated.

I additionally observed that case studies employing chemical methods had a slightly higher rate of eradication success rate, and a significantly greater suppression success rate, compared to those using mechanical ones (Figure 2; Figure 3, respectively). Chemical methods are intuitively expected to have some advantages over mechanical methods in aquatic ecosystems. Toxicants applied to aquatic systems will naturally diffuse throughout the system, and potentially expose and affect all individuals within, including those organisms in early growth stages or those which are hiding and otherwise difficult to detect manually. As a result, toxicants can potentially eliminate all AIS individuals without prior detection by managers. Anderson (2005) and Cilliers (1996) noted that chemical methods were more effective than manual methods in the attempted removal of hydrilla and water lettuce, respectively, because of such obstacles. Moreover, it is expected that chemicals would be more effective, than mechanical methods in eliminating plant AIS from aquatic ecosystems. This is due to the potential for some plants (ie: hydrilla) to reproduce through seeds or detached fragments, both of which are less likely to be impacted by manual removal methods compared to herbicides. Of the reviewed eradication cases involving the use of chemical methods, 29% included eradication of aquatic plants, while 48% were found amongst cases using mechanical methods. Similarly, when evaluating suppression success, there was only a single case of chemical intervention involving a plant AIS (hydrilla), while, of the remaining 13 cases of mechanical removal, three involved plant AIS. Therefore, the lower eradication and suppression success rates experienced when using mechanical methods might be due in part to the larger proportion of plants being present in this dataset for which manual removal was attempted, as oppose to chemical treatment.

Lastly, I discovered a negative linear relationship between habitat surface area and the suppression success rate (Figure 4). This outcome is somewhat to be expected as it suggests that managers succeed more often when suppressing AIS populations in smaller study sites as compared to larger ones. Potential drivers of this phenomenon include the lower budget requirement, and thus greater ease of funding acquisition, for smaller versus larger scale projects. In addition, when AIS occupy isolated regions of a habitat, and especially when AIS are also immobile, less effort, and thus less funding, is required for both pre- and post-treatment surveying, as well as removal. Moreover, it is expected that AIS have a relatively more restricted freedom of movement in smaller versus larger habitats, thus their options for evasion or spread are also limited. Detection of newly established AIS, which typically occupy isolated and small spaces, is in turn more likely when AIS are introduced into smaller habitats. This is because smaller areas need be examined before AIS are noticed by personnel, whereas the same population would take more time to detect in a larger habitat.

Of the seven multiple method approach suppression cases investigated, all cases involved simultaneous treatment, rather than a sequential application of different methods. In such cases, suppression is typically a long-term goal, which is achieved by applying a significant, and relatively instant, stress to AIS, year after year (Cilliers, 1996). In contrast, when methods are applied one after another, methods are either being investigated for relative effectiveness by managers, or certain methods are found more suitable for specific stages of intervention than others. For instance, the suppression of northern pike in Box

Canyon Dam reservoir, Washington, involves regular intervention by means of fishing and electrofishing, simultaneously, on a seasonal basis, and drastic population reduction becomes achievable as this stress is maintained (WDFW, 2014). In other cases, the addition of methods to supplement initial treatment is an essential part of adaptive management. In the suppression of sea lamprey, authors found that the species population was rapidly growing, requiring the addition of bottom release pesticide, supplementing the use of sterile males, in attempt to restrict rapid population expansion (Twohey *et al*., 2003). Had these methods been employed sequentially, rather than simultaneously, sea lamprey populations would have had more time to rebound. Thus, although the management approach is highly dependent on AIS under study, as well as the availability of methods, there may exist a general discrepancy between approaches, with sequential methods providing a longer AIS rebound window than simultaneous approaches.

There potentially exist other key factors that may be vital to AIS eradication and/or suppression success. In many cases, aquatic AIS management sites provide limited access to AIS and difficulty in capturing and/or detecting all members of a population. In the case of northern pike eradication from Stormy Lake, Alaska (ADFG, 2011), this obstacle was overcome by the use of the chemical rotenone, which does not require the capturing of target AIS. Additionally, a workshop on signal crayfish management in the U.K. identified several contributors to successful suppression (EA, 2000). These included contractor preparation time, communication between stakeholders, and having a mission statement. Another important factor could be public awareness (McMillin, 2007), specifically public willingness to cooperate with the mission statement of managers involved in AIS removal. Public cooperation was key in the eradication of northern pike from Lake Davis, California (CDFG, 2007). However, some of these factors are fairly difficult to quantify, without utilizing proxy values and thus lose power due to attenuation (Garson, 2012). Furthermore, it is more likely that various factors interact and govern management success in combination rather than acting independently (Anderson, 2005). For example, knowledge of the killer alga's invasion history in the Mediterranean Sea, combined with quick detection and budget availability, led to an efficient and effective eradication campaign. An obstacle in meta-analytic research however, is the difficulty in quantifying the overall inter-case study effect size of certain factors, due to factors being unreported in some cases, or not standardized in others. This was indeed an obstacle in this study, as much data had to be estimated or acquired through personal communication. Unfortunately this, as well as limited sample size, also made it impractical to perform a multivariate analyses to assess the combined effects of predictor variables, as well as their degree of influence in the absence of other variables. In some cases, proper quantification is simply impractical, such as for instance attempting to accurately count the number of plants in a 100 ha system. In order to improve the reliability of meta-analytic findings, and thus in the magnitude of trends extending to various situations, it is essential that variables of AIS interventions be quantified accurately whenever possible.

In all cases involving attempted eradication of AIS, many authors made the assumption that populations were completely eliminated following eradication, and lack of detection. In some projects, the survey period, following eradication, was longer than in others. For example, in the eradication of hydrilla from California, staff required at least a three year hydrilla-free period, before declaring eradication (CDFA, 2014). However, the species was still found to reappear in some areas. In other scenarios, eradication was sooner declared due to lack of detection, such as in the removal of topmouth gudgeon from Clawford Lakes Fishery in the U.K. (EAUK, 2012). In both studies, a lack of detection was taken to imply complete elimination. This assumption is especially problematic when the source of AIS input is unknown, as populations have an opportunity to rebound due to the source remaining unmanaged. Unfortunately, in some cases managers must rely on this assumption, as other means of confirming eradication do not exist. However, this assumption can also be welcomed, such as when the goal of a project is simply the removal of all observable AIS members. Thus, although the assumption of ‘no detection’ does not necessarily imply ‘no AIS’, the result may nonetheless be acceptable to managers, depending on their interests, as well as those of stakeholders. Some of the more recognized obstacles to success of both eradication and suppression failure, are also worthy of mention. With respect to eradication projects, I noted that a lack of knowledge of AIS treatment, invasion pathway, and high false positive rate due to lack of detection, were prominent. Suppression cases seemed less successful when manual methods were used in aquatic systems, such as fishing, or when only one-time applications of methods were used.

An additional consideration for managers exists with respect to the style of suppression approach and source of AIS introduction. If regarded in terms of pulse versus press perturbations (Bender *et al.*, 1984), AIS can be steadily released into systems (ie: aquarium dumping, live bait use, between-system transit) or be released in ‘waves’ (ie: one-time accidents). In comparison, suppression could be carried out in a press-type fashion (long-term population reduction) or pulse-type fashion (seasonal removal). Taken altogether, suppression success is intuitively expected to be highest for situations where there is an infrequent input of AIS, and where removal is carried out continually. Such a phenomenon was illustrated, for instance, by the removal of northern pike, from Lake Davis, California (DFG, 2007), where authors suspected introduction to have occurred only once in the past, and where application of rotenone was used in a continuous fashion. Suppression success should be lowest in contrasting cases, where AIS input is continual but where management is not.

Currently, there exist various guidelines for the application of meta-analysis in ecological research (e.g., Gurevitch *et al*., 2001). A common obstacle in all of these is the occurrence of publication bias, the intentional publication of results only when they are favourable (Begg, 1994). In this study, I acknowledge that my dataset may suffer from publication bias, due to reports being potentially published by countries having the resources available to conduct RR (ISC, 2014). Although not entirely treatable, publication bias can be exploited using two approaches, as suggested by Harrison (2011). One method is to construct a funnel plot of effect size versus sample size. If data points show random scattering about the plot, publication bias is likely. However, I did not use the funnel plot method for evaluating publication bias because many authors, including Harrison (2011), believe it to be highly subjective. An additional method, to quantitatively assess publication bias is the calculation of the 'failsafe sample size' (Rosenberg, 2005). The failsafe sample size aims to predict the sample size which must be obtained in order to alter the significance value of the current dataset. So long as the failsafe number exceeds the current sample size, publication bias is less likely (Harrison, 2011). However, the failsafe sample size is also subject to criticism as it does not account for weighting of data. Because the reliability of my results differs by the robustness of statistical tests conducted, the failsafe sample size would also be highly subjective if applied to the entire dataset.

In conclusion, I discovered certain factors may be responsible for determining the outcome of AIS management campaigns. In regard to eradication RR, AIS taxonomy is key for determining success, and plant eradications are expected to succeed more often than animal ones. Chemical methods were also slightly more successful than mechanical methods. In AIS suppression, success was greatest when conducted in small habitats and by using chemical methods. Although many other variables were investigated, they proved unimportant to management outcome. The results of this project aim to inform management and other stakeholders on methods most likely to succeed in eradication or suppression of AIS prior to an attempted intervention, which ultimately leads to cost efficiency and effectiveness. Managers should also expect that, depending on whether AIS populations are eliminated or simply reduced, different factors, including the frequency in which intervention is applied, and knowledge of invasion pathways, will be important. Lastly, this study demonstrates the importance of quantitative reporting by managers, especially when studies are combined in a meta-analysis or when data are used to construct an overall prediction model.

Table 1. Chi-square and Fisher's exact tests comparing the eradication success rate between groups of varying ecosystem type, methods used, number of methods used, and taxonomy of AIS, with number of cases (N), 95% confidence interval (CI), chi-square statistic (χ^2), degrees of freedom (df), and probability (P). Values of P<0.050 are considered significant.

Predictor	**Group**	**N (failure)**	**N (success)**	**CI (±)**	χ^2	**df**	**P**
Ecosystem type	Freshwater	22	77	0.145	-	1	0.999
	Marine	2	7				
Method type	Chemical	3	28	0.088	3.504	1	0.061
	Mechanical	11	29				
Number of methods	Multiple	11	49	0.081	1.181	1	0.277
	Single	13	35				
Taxonomy	Animal	17	30	0.081	9.366	1	0.002
	Plant	7	54				

Table 2. Binary logistic regression analysis of the relationship between the eradication success rate and population abundance, infestation extent, surface area, and project duration, with number of cases (N), slope (β), 95% confidence interval (CI), chi-square statistic (χ^2), degrees of freedom (df), and probability (P). Values of P<0.050 are considered significant.

Predictor	**N (failure)**	**N (success)**	**β**	**CI (±)**	**χ^2**	**df**	**P**
Abundance	7	16	0.001	0.001	1.236	1	0.266
Infestation extent	17	68	-0.001	0.002	1.939	1	0.175
Habitat area	24	84	-12.696	29.473	0.671	1	0.398
Project duration	24	84	-0.004	0.006	1.523	1	0.217

Table 3. t-test and group mean comparisons of the suppression success rate between groups of varying ecosystem type, methods used, number of methods used, and taxonomy of AIS, with number of cases (N), mean transformed log response ratio ($\bar{x}_R$), 95% confidence interval (CI), t-statistic (t), degrees of freedom (df), and probability (P). Values of P<0.050 are considered significant. Log response ratio values (R) were transformed separately for each predictor variable in order to meet the statistical assumptions of the t-test, and should not be directly compared among predictors.

Predictor	**Group**	**N**	$\bar{x}_R$	**CI (±)**	**t**	**df**	**P**
Ecosystem type	Freshwater	18	0.508	0.172	0.019	17	0.985
	Marine	1	0.506				
Method type	Chemical	1	0.000	0.206	4.877	12	0.001
	Mechanical	13	0.462				
Number of methods	Multiple	7	0.943	0.065	1.728	17	0.102
	Single	12	0.886				
Taxonomy	Animal	15	0.507	0.381	-0.020	17	0.984
	Plant	4	0.511				

Table 4. Linear regression analysis of the relationship between the suppression success rate and different predictor variables, including population abundance, infestation extent, surface area, and project duration, with number of cases (N), correlation coefficient (R^2), 95% confidence interval (CI), F-statistic (F), degrees of freedom (df), and probability (P). Values of $P<0.050$ are considered significant.

Predictor	**N**	**R^2**	**CI (±)**	**F**	**df**	**P**
Abundance	14	0.077	0.912	1.006	1,12	0.336
Infestation extent	5	0.342	0.269	1.557	1,3	0.301
Habitat area	19	0.243	0.005	5.449	1,17	0.032
Project duration	19	0.002	0.169	0.036	1,17	0.851

Figure 1. Histogram of the number of successful and failed eradication case studies for animal and plant taxonomy groups.

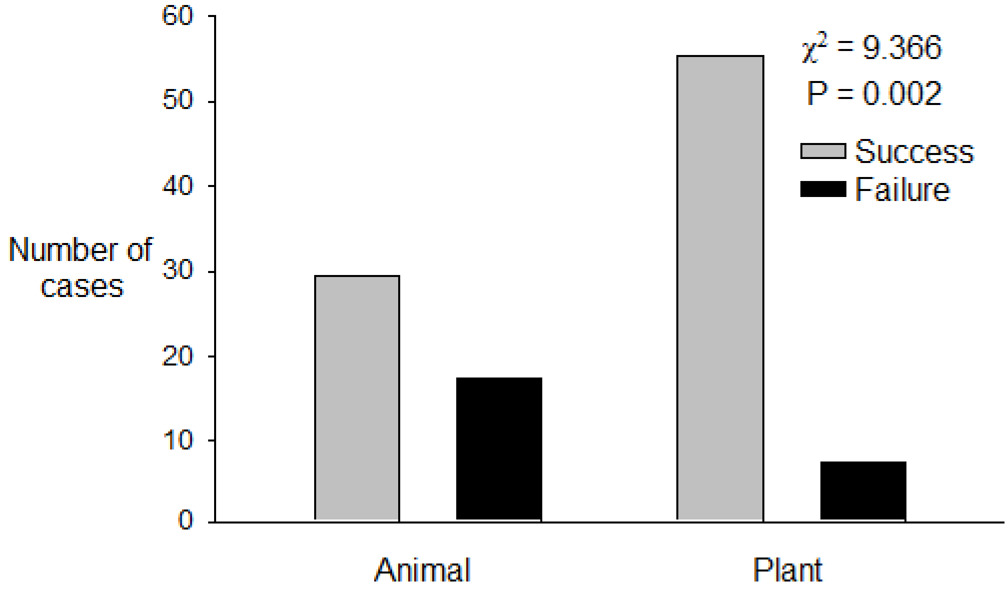


Figure 2. Histogram of the number of successful and failed eradication case studies for chemical and mechanical methods groups.

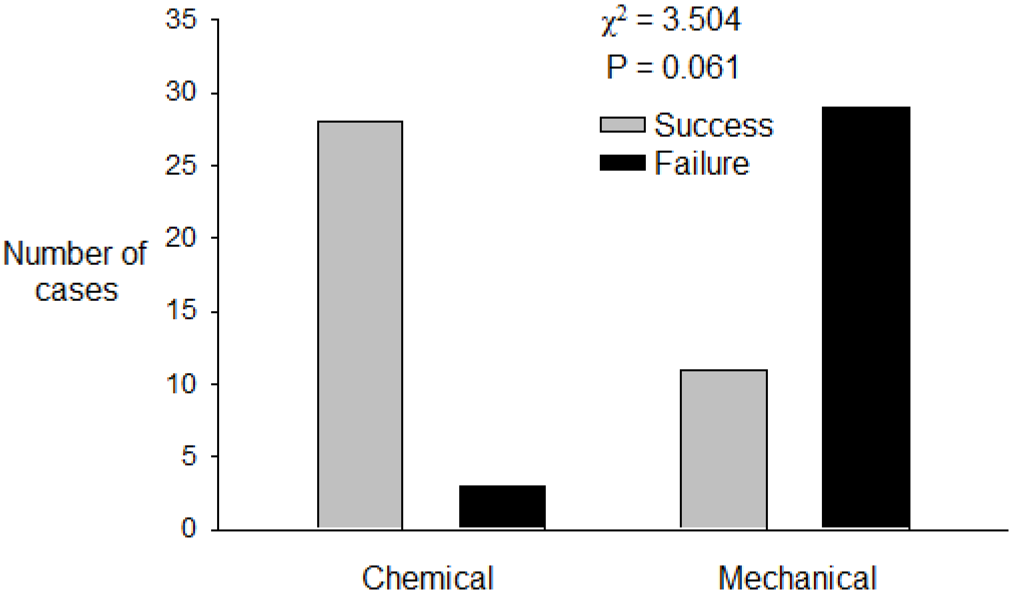


Figure 3. Box plot comparing the mean suppression success rate between case studies using chemical and mechanical methods. Black diamond indicates outlier value. Lower values of the log response ratio represent higher success.

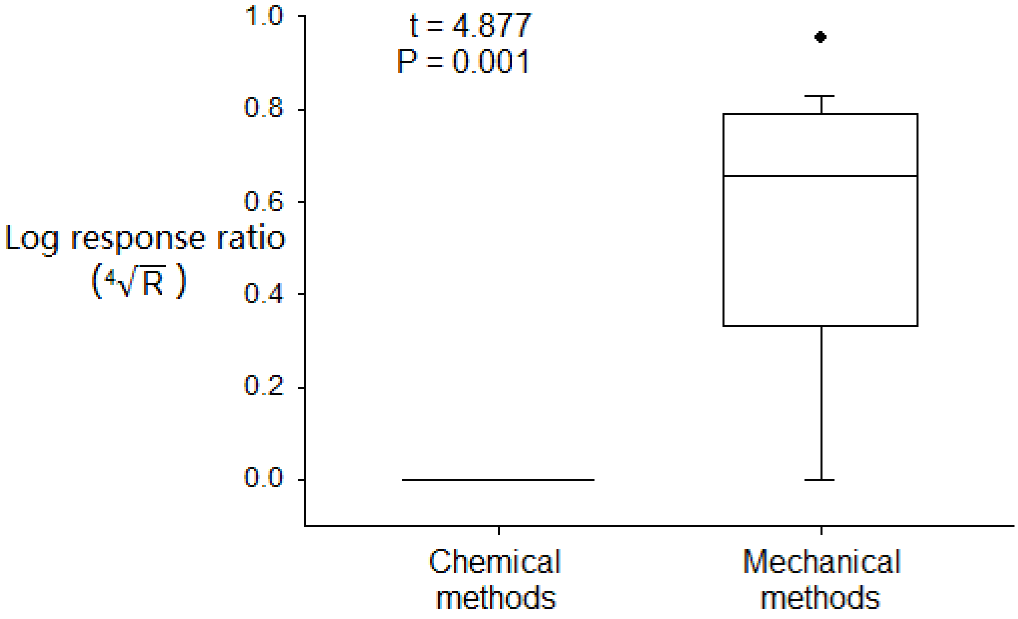
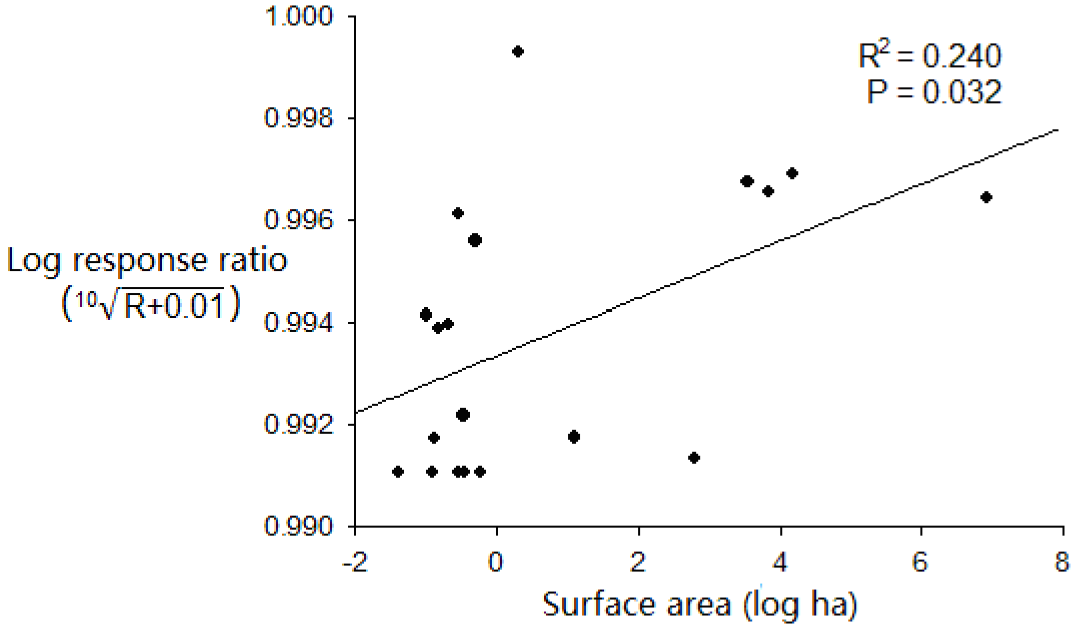


Figure 4. Linear regression plot depicting a negative relationship between the suppression success rate and the habitat surface area. Lower values of the log response ratio represent higher success.



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Appendix 1. Data catalogue for eradication case studies. An=animal species; Ch=chemical method; Fr=freshwater ecosystem; Ma=marine ecosystem; Me=mechanical method; Mu=multiple methods; Pl=plant species; Si=single methods.

Species ID	Ecosystem (Fr/Ma)	Method (Ch/Me)	#/Methods (Mu/Si)	Taxonomy (An/Pl)	Duration (months)	Area (ha)	Infestation (ha)	Abundance (#/organisms)	Eradication (Yes/No)
Killer alga	Ma	Ch	Mu	Pl	67	101	0.105	-	Yes
Killer alga	Ma	Ch	Si	Pl	63	153	1.06	-	Yes
Sabellid polychaete	Ma	Me	Mu	An	28	0.170	-	2.20x10^{6}*	Yes
Black striped mussel	Ma	-	Mu	An	1	12.5	-	1.00x10^{9}*	Yes
Black striped mussel	Ma	Ch	Mu	An	1	3.00	-	2.00x10^{5}*	Yes
Black striped mussel	Ma	-	Mu	An	1	5.00	0.0100*	-	Yes
Signal crayfish	Fr	Ch	Mu	An	11	0.630	-	4.00x10^{4}*	Yes
Signal crayfish	Fr	-	Mu	An	12	0.458	-	3.00x10^{4}*	No
Signal crayfish	Fr	Ch	Si	An	9	0.545	-	4.00x10^{4}*	Yes
Brook trout	Fr	Me	Si	An	57	1.60	-	97	Yes
Rainbow trout	Fr	Me	Si	An	36	1.60	-	477	Yes
Signal crayfish	Fr	Me	Si	An	17	3.00	3.00*	-	No
Signal crayfish	Fr	Me	Si	An	19	1.21	0.504*	-	No
Signal crayfish	Fr	Me	Si	An	25	0.500	0.100*	-	No
Signal crayfish	Fr	Me	Si	An	32	0.200	0.100*	-	No
Signal crayfish	Fr	Me	Si	An	13	0.0500	-	4.70x10^{3}*	No
Signal crayfish	Fr	Me	Mu	An	9	4.90	-	2.00x10^{5}*	No
Signal crayfish	Fr	Me	Si	An	4	4.00	4.00*	-	No
Brook trout	Fr	Me	Si	An	36	2.20	-	1.40x10^{3}*	No
Rainbow trout	Fr	Me	Si	An	29	270	-	3.94x10^{5}*	Yes
Brook trout	Fr	Me	Mu	An	96	0.200	-	300*	Yes
Brook trout	Fr	Me	Mu	An	96	0.200	0.200*	-	Yes
Brook trout	Fr	Me	Mu	An	96	0.200	-	100*	Yes
Hydrilla	Fr	-	Mu	Pl	72	12.5	6.25*	-	Yes
Hydrilla	Fr	-	Mu	Pl	276	2.40	2.40*	-	No
Hydrilla	Fr	-	Mu	Pl	185	9.20*	4.7*	-	No
Hydrilla	Fr	-	Mu	Pl	36	0.0800	0.0400*	-	Yes

Species ID	Ecosystem (Fr/Ma)	Method (Ch/Me)	#/Methods (Mu/Si)	Taxonomy (An/Pl)	Duration (months)	Area (ha)	Infestation (ha)	Abundance (#/organisms)	Eradication (Yes/No)
Hydrilla	Fr	-	Mu	Pl	36	0.0800	0.0400*	-	Yes
Hydrilla	Fr	-	Mu	Pl	36	0.0800	0.0400*	-	Yes
Hydrilla	Fr	-	Mu	Pl	72	0.00730	0.00240*	-	Yes
Hydrilla	Fr	-	Mu	Pl	193	3.20	2.84*	-	Yes
Hydrilla	Fr	-	Mu	Pl	98	4.38	4.38*	-	Yes
Hydrilla	Fr	-	Mu	Pl	72*	0.558	0.186*	-	Yes
Hydrilla	Fr	-	Mu	Pl	36	0.102	0.0680*	-	Yes
Hydrilla	Fr	-	Mu	Pl	72	0.0801	0.0534*	-	Yes
Hydrilla	Fr	-	Mu	Pl	221	1.74×10^4	465.4	-	No
Hydrilla	Fr	-	Mu	Pl	296	570	9.35*	-	Yes
Hydrilla	Fr	-	Mu	Pl	93	0.200	0.200*	-	Yes
Hydrilla	Fr	-	Mu	Pl	93	2.80	2.80*	-	Yes
Hydrilla	Fr	-	Mu	Pl	93	0.0400	0.0400*	-	Yes
Hydrilla	Fr	-	Mu	Pl	262	0.180	0.180*	-	Yes
Hydrilla	Fr	-	Mu	Pl	262	0.0610	0.061*	-	Yes
Hydrilla	Fr	Me	Mu	Pl	28	0.00340	-	100*	Yes
Hydrilla	Fr	Me	Mu	Pl	36	0.100	0.100*	-	Yes
Hydrilla	Fr	Me	Mu	Pl	36	0.100	0.100*	-	Yes
Hydrilla	Fr	Me	Mu	Pl	36	0.100	0.100*	-	Yes
Hydrilla	Fr	Me	Mu	Pl	36	0.100	0.100*	-	Yes
Hydrilla	Fr	Me	Mu	Pl	36	0.100	0.100*	-	Yes
Hydrilla	Fr	Me	Mu	Pl	36	0.100	0.100*	-	Yes
Hydrilla	Fr	Me	Mu	Pl	36	0.100	0.100*	-	Yes
Hydrilla	Fr	Me	Mu	Pl	36	0.100	0.100*	-	Yes
Hydrilla	Fr	Me	Mu	Pl	36	0.0400	0.0400*	-	Yes
Hydrilla	Fr	Me	Mu	Pl	36	0.0400	0.0400*	-	Yes
Hydrilla	Fr	Me	Si	Pl	36	0.400	0.400*	-	Yes

Species ID	Ecosystem (Fr/Ma)	Method (Ch/Me)	#/Methods (Mu/Si)	Taxonomy (An/Pl)	Duration (months)	Area (ha)	Infestation (ha)	Abundance (#/organisms)	Eradication (Yes/No)
Hydrilla	Fr	Me	Si	Pl	36	0.400	0.400*	-	Yes
Hydrilla	Fr	Me	Si	Pl	36	0.400	0.400*	-	Yes
Hydrilla	Fr	Me	Si	Pl	36	0.00400	0.00400*	-	Yes
Hydrilla	Fr	-	Mu	Pl	36	0.800	0.800*	-	Yes
Hydrilla	Fr	Me	Si	Pl	36	0.0490	0.0490*	-	Yes
Hydrilla	Fr	Me	Si	Pl	36	0.00400	0.00400*	-	Yes
Hydrilla	Fr	Ch	Si	Pl	180	27.0	27.0*	-	Yes
Hydrilla	Fr	Ch	Si	Pl	108	0.976	0.976*	-	Yes
Hydrilla	Fr	Ch	Si	Pl	108	0.976	0.976*	-	Yes
Hydrilla	Fr	Ch	Si	Pl	108	1.35	1.35*	-	Yes
Hydrilla	Fr	Ch	Si	Pl	108	0.720	0.720*	-	Yes
Hydrilla	Fr	-	Mu	Pl	108	9.00	9.00*	-	Yes
Hydrilla	Fr	-	Mu	Pl	108	9.00	9.00*	-	Yes
Hydrilla	Fr	-	Mu	Pl	108	9.00	9.00*	-	Yes
Hydrilla	Fr	-	Mu	Pl	108	9.00	9.00*	-	Yes
Hydrilla	Fr	-	Mu	Pl	204	16.0	9.00*	-	No
Hydrilla	Fr	-	Mu	Pl	228	5.30	5.30*	-	No
Hydrilla	Fr	Me	Si	Pl	36	29.0	29.0*	-	Yes
Hydrilla	Fr	Me	Si	Pl	36	0.00400	0.00400*	-	Yes
Hydrilla	Fr	-	Mu	Pl	300	2.22×10^3	2.22×10^3*	-	No
Hydrilla	Fr	-	Mu	Pl	241	731	43	-	Yes
Hydrilla	Fr	Ch	Mu	Pl	100	0.250	-	136	Yes
Hydrilla	Fr	Ch	Si	Pl	93	1.11	0.770*	-	Yes
Hydrilla	Fr	-	Mu	Pl	204	69.2	32.0	-	Yes
Hydrilla	Fr	Me	Mu	Pl	36	0.00400	0.00400*	-	Yes
Brown alga	Ma	Me	Si	An	1	0.0300	-	174	Yes

Species ID	Ecosystem (Fr/Ma)	Method (Ch/Me)	#/Methods (Mu/Si)	Taxonomy (An/Pl)	Duration (months)	Area (ha)	Infestation (ha)	Abundance (#/organisms)	Eradication (Yes/No)
Brook trout	Fr	Ch	Si	An	51	5.44×10^3	-	4000*	Yes
Convict cichlid	Fr	Ch	Si	An	7	0.300	-	654	Yes
Alien fanworm	Ma	Me	Si	An	4	1.80×10^4	-	724	No
Alien fanworm	Ma	Me	Si	An	19	50.0	-	8	No
Round goby	Fr	Ch	Si	An	29	28.0	20.0*	-	No
Water lettuce	Fr	-	Mu	Pl	100	0.150	0.150*	-	No
Northern pike	Fr	-	Mu	An	120	1.63×10^4	1.63×10^4*	-	Yes
Giant salvinia	Fr	-	Mu	Pl	35	1.33×10^4	405	-	Yes
Topmouth gudgeon	Fr	Ch	Si	An	9	0.270	0.270*	-	Yes
Topmouth gudgeon	Fr	Ch	Si	An	9	0.280	0.280*	-	Yes
Topmouth gudgeon	Fr	Ch	Si	An	9	1.73	1.73*	-	Yes
Topmouth gudgeon	Fr	Ch	Si	An	9	0.400	0.400*	-	Yes
Topmouth gudgeon	Fr	Ch	Si	An	9	0.475	0.475*	-	Yes
Topmouth gudgeon	Fr	Ch	Si	An	9	0.983	0.983*	-	Yes
Topmouth gudgeon	Fr	Ch	Si	An	9	0.904	0.904*	-	Yes
Topmouth gudgeon	Fr	Ch	Si	An	1	0.127	0.127*	-	No
Topmouth gudgeon	Fr	Ch	Si	An	1	0.127	0.127*	-	No
Rainbow smelt	Fr	Me	Si	An	27	36.7	-	600	No
Rusty crayfish	Fr	-	Mu	An	120	64.0	64.0*	-	No
Hornwort	Fr	Ch	Mu	Pl	62	13.2	13.2*	-	Yes
Topmouth gudgeon	Fr	Me	Mu	An	151	0.380	-	1	Yes
Common carp	Fr	-	Mu	An	168	26.0	26.0*	-	No
Northern pike	Fr	Ch	Si	An	8	34.0	34.0*	-	Yes
Northern pike	Fr	Ch	Si	An	3	8.00	8.00*	-	Yes
Northern pike	Fr	Ch	Si	An	1	163	163*	-	Yes
Northern pike	Fr	Ch	Si	An	5	9.70	9.70*	-	Yes
Northern pike	Fr	Ch	Si	An	8	30.0	30.0*	-	Yes
Northern pike	Fr	Ch	Si	An	8	7.30	7.30*	-	Yes

Appendix 2. Data catalogue for suppression case studies. An=animal species; Ch=chemical method; Fr=freshwater ecosystem; Ma=marine ecosystem; Me=mechanical method; Mu=multiple methods; Pl=plant species; Si=single methods.

Species ID	Ecosystem (Fr/Ma)	Method (Ch/Me)	#/Methods (Mu/Si)	Taxonomy (An/Pl)	Abundance (# organisms)	Infestation (ha)	Area (ha)	Duration (months)	R
Brown trout	Fr	Me	Si	An	1.66×10^3*	-	0.500	48	0.094
Sea lamprey	Fr	-	Mu	An	-	8.21×10^6*	8.21×10^6	60	0.149
Sea lamprey	Fr	-	Mu	An	3.56×10^4*	-	1.54×10^5	96	0.193
Brook trout	Fr	Me	Si	An	71	-	0.210	48	0.035
Brook trout	Fr	Me	Si	An	380	-	0.340	48	0.008
Brook trout	Fr	Me	Si	An	160	-	0.580	48	0.000
Brown mussel	Ma	Me	Si	An	1.26×10^4*	-	12.6	14	0.004
Brook trout	Fr	Me	Mu	An	1.07×10^3*	-	0.280	168	0.125
Water lettuce	Fr	-	Mu	Pl	2.90×10^6*	-	632	78	0.002
American bullfrog	Fr	Me	Si	An	9.16×10^3	-	0.150	17	0.033
American bullfrog	Fr	Me	Si	An	349	-	0.100	17	0.039
American bullfrog	Fr	Me	Si	An	2.85×10^3	-	0.360	5	0.000
American bullfrog	Fr	Me	Si	An	613	-	0.120	5	0.000
American bullfrog	Fr	Me	Si	An	1.76×10^3	-	0.280	5	0.000
American bullfrog	Fr	Me	Si	An	5.34×10^3	-	0.130	5	0.004
Giant salvinia	Fr	-	Mu	Pl	-	121	3.48×10^3	23	0.177
Giant salvinia	Fr	-	Mu	Pl	-	1.81×10^3	6.97×10^3	79	0.160
Northern pike	Fr	Me	Mu	An	-	2.00*	2.00	19	0.682
Hydrilla	Fr	Ch	Si	Pl	83	0.0400	0.0380*	-	0.000

Appendix 3. Chi-square test and Fisher's exact test outputs from SPSS v.20 statistics software.

Fisher's exact test output comparing proportions of eradication success between freshwater (Fr) and marine (Ma) ecosystem predictor groups.

Ecosystem * Eradication Crosstabulation

Count

		Eradication		Total
		No	Yes	
Ecosystem	Fr	22	77	99
	Ma	2	7	9
Total		24	84	108

Chi-Square Tests

	Value	df	Asymp. Sig. (2-sided)	Exact Sig. (2-sided)	Exact Sig. (1-sided)
Pearson Chi-Square	.000[a]	1	1.000		
Continuity Correction[b]	.000	1	1.000		
Likelihood Ratio	.000	1	1.000		
Fisher's Exact Test				1.000	.638
N of Valid Cases	108				

a. 1 cells (25.0%) have expected count less than 5. The minimum expected count is 2.00.

b. Computed only for a 2x2 table

Chi-square test output comparing proportions of eradication success between chemical (Ch) and mechanical (Me) methods predictor groups.

Method * Eradication Crosstabulation

Count

		Eradication		Total
		No	Yes	
Method	Ch	3	28	31
	Me	11	29	40
Total		14	57	71

Chi-Square Tests

	Value	df	Asymp. Sig. (2-sided)	Exact Sig. (2-sided)	Exact Sig. (1-sided)
Pearson Chi-Square	3.504[a]	1	.061		
Continuity Correction[b]	2.469	1	.116		
Likelihood Ratio	3.734	1	.053		
Fisher's Exact Test				.076	.056
N of Valid Cases	71				

a. 0 cells (0.0%) have expected count less than 5. The minimum expected count is 6.11.

b. Computed only for a 2x2 table

Chi-square exact test output comparing proportions of eradication success between multiple (Mu) and single (Si) method approach predictor groups.

Method# * Eradication Crosstabulation

Count

		Eradication		Total
		No	Yes	
Method#	Mu	11	49	60
	Si	13	35	48
Total		24	84	108

Chi-Square Tests

	Value	df	Asymp. Sig. (2-sided)	Exact Sig. (2-sided)	Exact Sig. (1-sided)
Pearson Chi-Square	1.181[a]	1	.277		
Continuity Correction[b]	.729	1	.393		
Likelihood Ratio	1.175	1	.278		
Fisher's Exact Test				.353	.196
N of Valid Cases	108				

a. 0 cells (0.0%) have expected count less than 5. The minimum expected count is 10.67.

b. Computed only for a 2x2 table

Chi-square exact test output comparing proportions of eradication success between animal (An) and plant (Pl) taxonomy predictor groups.

Taxonomy * Eradication Crosstabulation

Count

		Eradication		Total
		No	Yes	
Taxonomy	An	17	30	47
	Pl	7	54	61
Total		24	84	108

Chi-Square Tests

	Value	df	Asymp. Sig. (2-sided)	Exact Sig. (2-sided)	Exact Sig. (1-sided)
Pearson Chi-Square	9.366[a]	1	.002		
Continuity Correction[b]	7.992	1	.005		
Likelihood Ratio	9.430	1	.002		
Fisher's Exact Test				.004	.002
N of Valid Cases	108				

a. 0 cells (0.0%) have expected count less than 5. The minimum expected count is 10.44.

b. Computed only for a 2x2 table

Appendix 4. Binary logistic regression output from SPSS v.20 statistics software.

Binary logistic regression output assessing goodness of fit of population abundance predictor to the logistic model for eradication success.

Omnibus Tests of Model Coefficients

		Chi-square	df	Sig.
Step 1	Step	1.236	1	.266
	Block	1.236	1	.266
	Model	1.236	1	.266

Classification Table[a]

	Observed		Predicted		
			Eradication		Percentage Correct
			No	Yes	
Step 1	Eradication	No	0	7	.0
		Yes	0	16	100.0
	Overall Percentage				69.6

a. The cut value is .500

Variables in the Equation

		B	S.E.	Wald	df	Sig.	Exp(B)
Step 1[a]	Abundance_tr	.001	.002	.375	1	.540	1.001
	Constant	.583	.522	1.245	1	.264	1.791

a. Variable(s) entered on step 1: Abundance_tr.

Binary logistic regression output assessing goodness of fit of infestation extent predictor to the logistic model for eradication success.

Omnibus Tests of Model Coefficients

		Chi-square	df	Sig.
Step 1	Step	1.939	1	.164
	Block	1.939	1	.164
	Model	1.939	1	.164

Classification Table[a]

	Observed		Predicted		
			Eradication		Percentage Correct
			No	Yes	
Step 1	Eradication	No	1	16	5.9
		Yes	1	67	98.5
	Overall Percentage				80.0

a. The cut value is .500

Variables in the Equation

		B	S.E.	Wald	df	Sig.	Exp(B)
Step 1[a]	Infestation	-.001	.001	1.843	1	.175	.999
	Constant	1.473	.283	27.054	1	.000	4.362

a. Variable(s) entered on step 1: Infestation.

Binary logistic regression output assessing goodness of fit of habitat area predictor to the logistic model for eradication success.

Omnibus Tests of Model Coefficients

		Chi-square	df	Sig.
Step 1	Step	.671	1	.413
	Block	.671	1	.413
	Model	.671	1	.413

Classification Table[a]

	Observed		Predicted		
			Eradication		Percentage Correct
			No	Yes	
Step 1	Eradication	No	0	24	.0
		Yes	0	84	100.0
	Overall Percentage				77.8

a. The cut value is .500

Variables in the Equation

		B	S.E.	Wald	df	Sig.	Exp(B)
Step 1[a]	Area_tr	-12.696	15.037	.713	1	.398	.000
	Constant	12.523	13.359	.879	1	.349	274573.829

a. Variable(s) entered on step 1: Area_tr.

Binary logistic regression output assessing goodness of fit of project duration predictor to the logistic model for eradication success.

Omnibus Tests of Model Coefficients

		Chi-square	df	Sig.
Step 1	Step	1.523	1	.217
	Block	1.523	1	.217
	Model	1.523	1	.217

Classification Table[a]

	Observed		Predicted		
			Eradication		Percentage Correct
			No	Yes	
Step 1	Eradication	No	0	24	.0
		Yes	0	84	100.0
	Overall Percentage				77.8

a. The cut value is .500

Variables in the Equation

		B	S.E.	Wald	df	Sig.	Exp(B)
Step 1[a]	Duration	-.004	.003	1.583	1	.208	.996
	Constant	1.526	.329	21.546	1	.000	4.599

a. Variable(s) entered on step 1: Duration.

Appendix 5. Independent t-test output from SPSS v.20 statistics software.

One-sample t-test output comparing mean log response ratio (R) value for freshwater (Fr) ecosystem predictor group, and marine (Ma) ecosystem point estimate value.

One-Sample Statistics

	N	Mean	Std. Deviation	Std. Error Mean
R	18	.507943	.3456712	.0814755

One-Sample Test

	Test Value = 0.5064			
	t	df	Sig. (2-tailed)	Mean Difference
R	.019	17	.985	.00154

One-sample t-test output comparing mean log response ratio (R) value for chemical (Ch) methods point estimate value, and mechanical (Me) methods predictor group.

One-Sample Statistics

	N	Mean	Std. Deviation	Std. Error Mean
R	13	.4618	.34137	.09468

One-Sample Test

	Test Value = 0.0000			
	t	df	Sig. (2-tailed)	Mean Difference
R	4.877	12	.001	.46176

Two-sample t-test output comparing mean log response ratio (R) values between multiple (Mu) and single (Si) methods predictor groups.

Group Statistics

	Method#	N	Mean	Std. Deviation	Std. Error Mean
R	Mu	7	.942622	.0961654	.0363471
	Si	12	.885810	.0484013	.0139722

Independent Samples Test

		t-test for Equality of Means				
		t	df	Sig. (2-tailed)	Mean Difference	Std. Error Difference
R	Equal variances assumed	1.728	17	.102	.0568121	.0328807
	Equal variances not assumed	1.459	7.811	.184	.0568121	.0389401

Two-sample t-test output comparing mean log response ratio (R) values between animal (An) and plant (Pl) taxonomy predictor groups.

Group Statistics

	Taxonomy	N	Mean	Std. Deviation	Std. Error Mean
R	An	15	.5070	.33789	.08724
	Pl	4	.5110	.37986	.18993

Independent Samples Test

		t-test for Equality of Means				
		t	df	Sig. (2-tailed)	Mean Difference	Std. Error Difference
R	Equal variances assumed	-.020	17	.984	-.00395	.19452
	Equal variances not assumed	-.019	4.358	.986	-.00395	.20901

Appendix 6. Linear regression output from SPSS v.20 statistics software.

Linear regression output assessing goodness of fit of population abundance predictor to the linear model for suppression success.

Model Summary

Model	R	R Square	Adjusted R Square	Std. Error of the Estimate
1	.278[a]	.077	.000	.46521

a. Predictors: (Constant), Abundance_tr

ANOVA[a]

Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	.218	1	.218	1.006	.336[b]
	Residual	2.597	12	.216		
	Total	2.815	13			

a. Dependent Variable: R_tr

b. Predictors: (Constant), Abundance_tr

Coefficients[a]

Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.
		B	Std. Error	Beta		
1	(Constant)	.264	.461		.573	.577
	Abundance_tr	.133	.133	.278	1.003	.336

a. Dependent Variable: R_tr

Linear regression output assessing goodness of fit of infestation extent predictor to the linear model for suppression success.

Model Summary

Model	R	R Square	Adjusted R Square	Std. Error of the Estimate
1	.584[a]	.342	.122	.13740

a. Predictors: (Constant), Infestation_tr

ANOVA[a]

Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	.029	1	.029	1.557	.301[b]
	Residual	.057	3	.019		
	Total	.086	4			

a. Dependent Variable: R_tr

b. Predictors: (Constant), Infestation_tr

Coefficients[a]

Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.
		B	Std. Error	Beta		
1	(Constant)	.801	.078		10.246	.002
	Infestation_tr	.027	.022	.584	1.248	.301

a. Dependent Variable: R_tr

Linear regression output assessing goodness of fit of habitat area predictor to the linear model for suppression success.

Model Summary

Model	R	R Square	Adjusted R Square	Std. Error of the Estimate
1	.493[a]	.243	.198	.0023736

a. Predictors: (Constant), Area_tr

ANOVA[a]

Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	.000	1	.000	5.449	.032[b]
	Residual	.000	17	.000		
	Total	.000	18			

a. Dependent Variable: R_tr

b. Predictors: (Constant), Area_tr

Coefficients[a]

Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.
		B	Std. Error	Beta		
1	(Constant)	.993	.001		1729.816	.000
	Area_tr	.001	.000	.493	2.334	.032

a. Dependent Variable: R_tr

Linear regression output assessing goodness of fit of project duration predictor to the linear model for suppression success.

Model Summary

Model	R	R Square	Adjusted R Square	Std. Error of the Estimate
1	.046[a]	.002	-.057	.08643

a. Predictors: (Constant), Duration_tr

ANOVA[a]

Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	.000	1	.000	.036	.851[b]
	Residual	.127	17	.007		
	Total	.127	18			

a. Dependent Variable: R_tr

b. Predictors: (Constant), Duration_tr

Coefficients[a]

Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.
		B	Std. Error	Beta		
1	(Constant)	.885	.063		13.960	.000
	Duration_tr	.008	.042	.046	.190	.851

a. Dependent Variable: R_tr

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