

Research Article

Eradication economics for invasive alien aquatic plantsCarla Muller^{1,2}, Deborah Hofstra^{1,*} and Paul Champion¹¹National Institute of Water and Atmospheric Research, PO Box 11115, Hillcrest, Hamilton 3251, New Zealand²Present address: Perrin Ag Consultants Ltd, PO Box 596, Rotorua 3040, New ZealandAuthor e-mails: Carla@perrinag.net.nz (CM), Deborah.Hofstra@niwa.co.nz (DH), Paul.Champion@niwa.co.nz (PC)

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Abstract

Although there is a sizeable budget for biosecurity management in New Zealand, there will never be sufficient resources to intervene in all incursions of harmful species, including invasive freshwater plants. Given the inherent complexity in making decisions on interventions, responses are often decided in relation to specific incursions as they happen within their particular context. This paper uses New Zealand case studies to test if there are general economic principles related to intervention in invasive freshwater plants which can be used to improve decision-making. The general principles were; the sooner eradication is started and/or the smaller the incursion size, the lower the cost will be; eradication is more expensive than alternative responses in the short term but is cheaper than long term impacts of no intervention and/or ongoing control programs; and preventing new incursions is cheaper in the long run than eradication. The three principles were considered to be logical in an economic sense and can contribute to supporting intervention decision-making at a macro-level (i.e. across the majority of incursions) and provide more holistic outcomes for society in relation to management programs.

Key words: biosecurity; incursion management; intervention options; invasive alien aquatic species (IAAPs), non-market value

Introduction

Invasive species are a major cause of ecological change, with negative impacts on biodiversity and economic values (Jardine and Sanchirico 2018). It has been estimated in Europe alone that more than ten thousand non-native species have become invasive, causing damage estimated at 12 billion euros per year (Courtois et al. 2018). That said, economic estimates of impacts, damages and control costs are considered difficult to calculate, and when referred to in the literature, are often recognised as underestimates of the true costs (e.g., Myers et al. 2000; Pimentel et al. 2005; Courchamp et al. 2017; Jardine and Sanchirico 2018).

Eradication programs are inherently complex and each program is unique. Despite this, there is a growing body of literature trying to identify what makes eradication programs successful. This includes research that has looked at the general principles common to successful eradications

(such as Simberloff 2003, 2014; Panetta 2009; Mack and Foster 2009; Larson et al. 2011), those that empirically analysed past eradication programs (Rejmánek and Pitcairn 2002) and those that used models to predict outcomes (Cacho et al. 2006).

New Zealand has had considerable success with the national eradication of aquatic invasive species that have established naturalized populations (Champion 2018). However, there are also eradication attempts that have failed and there are still many populations of invasive freshwater plants present across the country. Additionally, there is a constant threat of new invasive freshwater plant incursions. The requirements for management of invasive aquatic macrophytes are outlined by the National Policy Direction for Pest Management (NPDPM) (New Zealand Government 2015) for species managed under National or Regional Pest Management Plans under the Biosecurity Act 1993 (New Zealand Government 1993). As part of these, central or regional government are required to undertake cost benefit analyses (CBA) for specific incursion responses as part of their decision-making process. There has been considerable work done to help management agencies with decision-making in relation to invasive species in New Zealand. However, most often the focus has been on terrestrial plants and animals (AgResearch 2019) and marine species (e.g. Wotton and Hewitt 2004).

The economic impacts of interventions to biosecurity incursions not only relates to the direct (and often ongoing) intervention cost of various intervention options, but also to the potential and ongoing impacts on stakeholders “values” (including social, economic, environmental and cultural values) (e.g. Larson et al. 2011; Courtois et al. 2018). For example, if a decision is made to ‘live with’ the invasive freshwater plant, then the total cost of this action includes the cost of making that decision as additional to the costs of ongoing impact on recreational users, commercial users (e.g. hydroelectric power generation), indigenous resources and native and desired non-native flora and fauna. While it is not appropriate to monetise all values (e.g. cultural values; Robb 2014) the ability to incorporate other non-market values into decision-making is crucial to ensure the best possible outcomes. This approach is supported by Pejchar and Mooney (2009) who believe that CBAs must include the cultural, social, economic and environmental impacts of each decision in the same framework as traditional monetary metrics in order to achieve the best outcomes. Where possible, quantifying non-market values will ensure that a more robust decision is made and will lead to more optimal outcomes for society (Bell et al. 2009). While non-market valuation is complex, it is necessary to incorporate this into the analysis of the general principles examined in this paper.

This paper builds on existing literature and uses New Zealand-specific case studies to test if there are general principles related to interventions against invasive freshwater plants, which can be used to improve decision-

making. It tests whether those principles commonly discussed by biophysical scientists are logical in terms of economics. The general principles identified for analysis were:

1. The sooner eradication is started and/or the smaller the incursion size, the lower the cost.
2. Eradication is more expensive than alternatives, such as not intervening in the short-term, but is cheaper than long term impacts of no intervention and/or ongoing control programs.
3. Preventing incursions will be cheaper in the long-term than eradication.

The overall aim is to better understand the economic implications of these general principles to improve decision-making at a macro-level (i.e. across the majority of incursions) and provide more holistic outcomes for society in relation to management programs. Through this analysis implications for incursion specific CBAs are also discussed (micro-level).

Materials and methods

This study first considered commonalities across a wide range of invasive alien aquatic plant (IAAP) incursions. Common intervention points and intervention options were identified (1). It was then possible to transfer principles to other incursion scenarios, and a common framework (2) was developed onto which examples were mapped. Following this, key case studies were selected to test the general principles. Management costs were estimated from the authors' (DH and PC) experience of past responses, augmented by publicly available reports detailing costs and benefits for a range of actual and possible incursion pathways for the case studies (3).

(1) Intervention Options and Intervention Points

Intervention options are described by numerous authors (e.g. Simberloff et al. 2013; Goka 2014; Genovesi et al. 2015; Hussner et al. 2017). In the present paper, six intervention options are defined based on the NPDPM (New Zealand Government 2015); exclusion, eradication, progressive containment, sustained control, site-led pest program and pathway program (Table 1). Different intervention options could be selected throughout different stages of an incursion lifecycle. To understand the commonalities of intervention across a range of potential incursions, key intervention points (Table 1) were defined to sit alongside the intervention options. Key intervention points were primarily related to the size of the incursion at a particular site. In a similar way, Cacho et al. (2006) identified “switching points” whereby the area of the weed infestation indicated a change in the optimal intervention option.

The present paper mapped the pathways of a range of actual incursion examples from New Zealand. These were used to test if the predefined set

Table 1. Description of Intervention options (IOs) and key Intervention points (IPs) for invasive freshwater plants. IO descriptors are taken from the National Policy Direction for Pest Management (New Zealand Government 2015).

Intervention Options (IOs)
<ul style="list-style-type: none"> • <u>Eradication</u> – The intermediate outcome for the program is to reduce the infestation level of the subject, or an organism being spread by the subject, to zero levels in an area in the short to medium term. • <u>Exclusion</u> – The intermediate outcome for the program is to prevent the establishment of the subject, or an organism being spread by the subject, that is present in New Zealand but not yet established in an area. • <u>Progressive containment</u> – The intermediate outcome for the program is to contain or reduce the geographic distribution of the subject, or an organism being spread by the subject, to an area over time. • <u>Sustained control</u> – The intermediate outcome for the program is to provide for ongoing control of the subject, or an organism being spread by the subject, to reduce its impacts on values and spread to other properties. • <u>Site led pest program</u> – The intermediate outcome for the program is that the subject, or an organism being spread by the subject, that is capable of causing damage to a place is excluded or eradicated from that place, or is contained, reduced, or controlled within the place to an extent that protects the values of that place. • <u>Pathway program</u> – The intermediate outcome for the program is to reduce the spread of harmful organisms.
Intervention Points (IPs)
<ul style="list-style-type: none"> • <u>IP 1. “New incursion- Not naturalised”</u> – An invasive plant, that is either new to an area or country or has previously been eradicated, is found in a non-natural waterbody, e.g. aquaria, ornamental ponds or farm dams. • <u>IP 2. “New incursion- Early naturalisation”</u> – An invasive plant, that is either new to an area or country or has previously been eradicated, is found in a natural waterbody or in small quantities within several water bodies (total area less than 1 hectare). • <u>IP 3. “Further naturalisation”</u> – An invasive plant naturalises further, within multiple water bodies within a similar geographical area and/or is in a large waterbody (greater than 1 hectare). • <u>IP 4. “Regionally common”</u> – The invasive freshwater plant is common within a geographical region. • <u>IP 5. “Nationally common”</u> – The invasive freshwater plant is widespread throughout the majority of the country.

of intervention options and intervention points were applicable when considering intervention at a strategic level (such as supporting the general principles tested in this paper). The species that were used (across a range of incursions for these plants) were *Butomus umbellatus* L. (flowering rush), *Alternanthera philoxeroides* (Mart.) Griseb. (alligator weed), *Eichhornia crassipes* (Mart.) Solms. (water hyacinth), *Ceratophyllum demersum* L. (hornwort, coontail), and *Hydrilla verticillata* (L.f.) Royle (hydrilla).

While IAAPs all have context-specific incursion pathways, costs, impacts and responses, there are similarities between the intervention points and intervention options for different species at a macro level. Mapping species and potential responses helps to understand these costs and impacts, thereby improving the decision making that leads to optimal outcomes. The final intervention options and intervention points that were defined for use in the following case study analysis are summarised in Table 1.

(2) Case Studies

Case studies were used to investigate whether there were economically robust general principles related to IAAP management, that could be utilised to improve outcomes. This enabled the research to be grounded in realistic scenarios and estimates of costs and benefits. While the purpose here was not to go into the level of detail that is required by mandated agencies (e.g., regional and central government) in conducting site specific CBAs, case studies were considered most appropriate for analysing the general principles. Two invasive plants were used for this section of analysis; *E. crassipes* and *C. demersum*. Three incursion examples were used

Table 2. Case study summaries.

No.	Species	Intervention point	Intervention option	Actual/ hypothetical	Location
1a	<i>C. demersum</i>	IP.2: New incursion -early naturalisation	Eradication	Actual	South Island
Successful eradication of <i>C. demersum</i> from two small waterbodies (4 and 1.4 ha). Detected in a 4ha waterbody in 2002, <i>C. demersum</i> was controlled by manual removal and herbicide (diquat dibromide). <i>C. demersum</i> was found in 1.4ha pond in 2006 and treated with herbicide (endothall dipotassium). In 2013, eradication was confirmed (MPI 2013).					
1b	<i>C. demersum</i>	IP.4: Regionally common	Sustained control	Hypothetical	South Island
Scenario based on expansion of <i>C. demersum</i> from single site incursions to the projected range of another submersed IAAP (<i>Lagarosiphon major</i> (Ridley) Moss ex Wager) in the same region (ca 96,900 ha of waterways). Of this, approximately 1% is treated annually, focusing on the littoral zone in high use areas.					
2a	<i>C. demersum</i>	IP.4: Regionally common	Sustained control	Actual	Lake Karāpiro
<i>C. demersum</i> is abundant in the lake, the primary inflow and the region. It is subject to sustained control using diquat dibromide and mechanical harvesting (cutting and removal). It is recognised that eradication is not feasible due to the extent of the weed, the potential for re-infestation, the available budget and tools. For this scenario it is assumed that 50 ha (out of 770 ha) is nuisance weed and is treated (Hofstra and de Winton 2016; LINZ 2016; 2017, 2018).					
2b	<i>C. demersum</i>	IP.4: Regionally common	No management	Hypothetical	Lake Karāpiro
<i>C. demersum</i> is present in Lake Karāpiro and is not managed (Hofstra and de Winton 2016).					
3a	<i>C. demersum</i>	IP.4: Regionally common	Exclusion	Actual	Rotorua Lakes (Rotomā, Rotokākahi)
<i>C. demersum</i> is not present in two lakes (Lake Rotomā 1,110 ha, Lake Rotokākahi 450 ha), with active exclusion controls in place. Lake Rotomā remains under serious threat from invasion due to the proximity of other infested sites (Burton 2017). Containment cordons for weeds are established at access points (Edwards and Clayton 2015). Lake Rotokākahi is free of <i>C. demersum</i> , primarily due to its sacred status to Te Arawa and restricted public access (Edwards and Clayton 2013).					
3b	<i>C. demersum</i>	IP.4: Regionally	Eradication, (and Progressive containment)	Actual	Rotorua Lakes (Ōkātina, Ōkāreka)
<i>C. demersum</i> is targeted for eradication in one lake (Ōkātina 1,080 ha) and Progressive containment (towards eradication) in another (Ōkāreka 340 ha). <i>C. demersum</i> was first found in Lake Ōkātina in 2007, eradicated and subsequently found in 2009 at a different site (Edwards and Clayton 2013). Incursion management includes a weed cordon and diquat use. <i>C. demersum</i> was found in Lake Ōkāreka in April 2012 (Bathgate 2015). In 2013, it was in ca 17.5 ha, of 102 ha of suitable habitat. Progressive containment focuses on high amenity and recreational value areas, annual herbicide use, monitoring and a public awareness campaign.					
3c	<i>C. demersum</i>	IP.4: Regionally common	Sustained control	Actual	Rotorua Lakes (Rotorua)
<i>C. demersum</i> is managed through Sustained control in a large lake (Lake Rotorua 8,060 ha). First recorded in 1975 (Edwards and Clayton 2013) but did not dominate the lake due to waterbody characteristics. In the early 1990s, the regular application of diquat was initiated to control <i>C. demersum</i> and IAAPs (LINZ 2016, 2017, 2018).					
4a	<i>E. crassipes</i>	IP.1 and 2: New incursion	Eradication	Actual	National
Each new incursion of <i>E. crassipes</i> is eradicated when found. Approximately 10 new sites each year.					
4b	<i>E. crassipes</i>	Pre-arrival	Prevention	Hypothetical	National
There is an increase in a targeted program to prevent new incursions of <i>E. crassipes</i> .					

for *C. demersum*, eradication (South Island incursions), sustained control (Lake Karāpiro, Waikato Region) and a regional management programme with a range of intervention options for different lakes (Rotorua Te Arawa Lakes). For *E. crassipes*, two scenarios were compared, one where every new field incursion is eradicated as soon as it is notified (current strategy), with one hypothetical example of prioritising exclusion. The case studies that were selected for analysis are described in Table 2.

(3) Management costs and benefits

For each scenario, the most appropriate estimates from literature and experts were elicited to value the costs and benefits. Although there are a range of methods that can be implemented for IAAP interventions (e.g. Gettys et al. 2014; Hussner et al. 2017; Champion et al. 2019), only the costs of those methods relevant for the case studies are summarised (Table 3).

Table 3. Generalised IAAP management costs associated with intervention options relevant to the case studies.

<u>Verification.</u> IAAP identification ca.\$5K delimitation ca.\$10K, pathway mapping ca. \$30K (up to \$60K for more intensive tracing)
<u>Management Plan.</u> ca. \$10K–40K depending on complexity e.g., size of site, amenity, utility and cultural values, number stakeholders, communications (e.g. Bathgate 2015).
<u>Containment.</u> ca. \$40 K legal requirements (such as restricted place notices). ca. cost of \$500K in year one to construct weed cordons (Lass 2012), ongoing cordon maintenance estimate of \$250 pa.
<u>Control</u> Herbicide ca. \$2K per ha (plus extras e.g. permissions) (Hofstra and de Winton 2016) Weed cutting ca \$10K per ha (Hofstra and de Winton 2016) Benthic barrier \$30K (de Winton et al. 2013) to ca. \$60Kper ha for a biodegradable benthic barrier. Manual removal > \$10K per ha
<u>Monitoring</u> Broad range in costs depending on the complexity and size of the site and the IAAP (e.g., submersed compared with floating), i.e., \$7K to \$20K pa in the Rotorua lakes with Eradication goals for <i>C. demersum</i> (Bathgate 2015) compared with estimates of \$10K p.a. for the first 5 years and \$5K p.a. for the next 15 years at <i>E. crassipes</i> sites.
<u>Legal action</u> While the cost of legal action against people found to be trading or growing IAAPs is difficult to approximate and therefore not included, financial penalties could provide a possibility for revenue to offset some of these costs. People found in breach of the Biosecurity Act 1993 can be fined up to \$500K (New Zealand Government 1993)
NB: Dollars are NZ\$, and K denotes numbers in the thousands, p.a is per annum.

As well as direct intervention costs, there are costs and benefits associated with a waterbody either with or without the IAAP. In this context, the economic impact not only relates to the interventions, but also the potential impacts (costs) on stakeholders' values, including social, economic, environmental and cultural values. These can be both market and non-market values.

Non-market valuation is a well-established technical field and includes current non-market costs and benefits as well as into the future (e.g. option value) (Hanley and Barbier 2009). There are several non-market valuation studies related to freshwater in New Zealand, which use various stated or revealed preference methods. While some studies relate to specific recreation benefits such as angling (e.g. Beville et al. 2012) or water quality as a value described by attributes such as "swimmability" and clarity (e.g. Marsh and Baskaran 2009; Bell and Yap 2004), others deal with ecosystem services valuation such as Mueller et al. (2016) and Patterson and Cole (2013). Few of these studies examine the change in value as a result of an IAAP. The exceptions are Bell et al. (2009) who assessed the potential impact of *H. verticillata* on an individual lake (Lake Rotorua in Hamilton City) and Beville et al. (2012) who examined the impact of the invasive diatom *Didymosphenia geminata* (Lyngbe) M. Schmidt on recreational angling (Canterbury region).

In general, Patterson and Cole (2013) and Mueller et al. (2016) provide the best, or most complete, estimate of the benefits. Patterson and Cole (2013) document a New Zealand-wide study of the "total economic value" of land-based ecosystems and their services, including lakes, rivers and wetlands. This is a widely used study for estimating changes in total economic value (e.g. Hutchison et al. 2017; Northland Regional Council 2017; Auckland

Council 2018), including traditional ecosystem services (provisioning, cultural, regulating, supporting) and, for some ecosystems, non-use values (option, existence, bequest). Patterson and Cole (2013) considered rivers and lakes to have a use value of approximately \$18,450 per hectare (inflation adjusted to 2018 New Zealand dollars). This included a recreation value of approximately \$665 per hectare, and food production of approximately \$70 per hectare. In the present paper, Patterson and Cole (2013) was used as the basis for estimates of value in the *C. demersum* scenarios of Eradication (1a South Island), Sustained control (1b South Island, 2a Lake Karāpiro) and No management (2b Lake Karāpiro). For the Lake Karāpiro scenario, an additional value was added to represent the use of Lake Karāpiro for significant sporting events (e.g. rowing regattas; Waipa District Council 2016) that are highly unlikely to take place without the control of *C. demersum* on the lake (Clayton et al. 2006).

Mueller et al. (2016) was used as the basis for estimating the value of interventions for Exclusion, Eradication with progressive containment and Sustained control 3a-c (Rotorua lakes, Table 2). Building from Patterson and Cole (2013), Mueller et al. (2016) calculate the ecosystem service value provided by Lake Rotorua, including some benefit transfer and primary data collection. Mueller et al. (2016) used selected indicators and direct market pricing, indirect pricing (hedonic pricing, replacement cost) and existence value pricing. This included social damage costs (from loss of income due to impaired recreation and reduced property values), as well as ecological damage costs (caused by algal blooms and decline in habitat quality for aquatic fauna). The inclusion of habitat decline and existence values, mean the results are likely to be a more accurate representation. However, Mueller et al. (2016) still had a primary focus on water quality as a result of eutrophication, rather than IAAPs. Mueller et al. (2016) had an estimate of ecosystem services provided by Lake Rotorua of \$12,625–\$18,500 per hectare per year, with the upper value similar to the estimate of Patterson and Cole (2013). As such, it was considered reasonable to use this value as the best estimate for the other lakes within the same region. This value comprised \$2,030 per hectare for biodiversity, between \$847–1,263 per hectare for food, \$551–1,788 per hectare for nutrient processing, \$2,150–2,567 per hectare for amenity and aesthetics and \$6,987–10,925 for recreation (Mueller et al. 2016).

Results

The specific costs, benefits and assumptions used in each scenario are described in Figures 1–4.

Where *C. demersum* was eradicated from two small sites (scenario 1a), the initial cost of eradication per hectare was high (Figure 1). However, this eradication cost is significantly smaller than the costs of sustained control if *C. demersum* had not been eradicated and were to spread more extensively

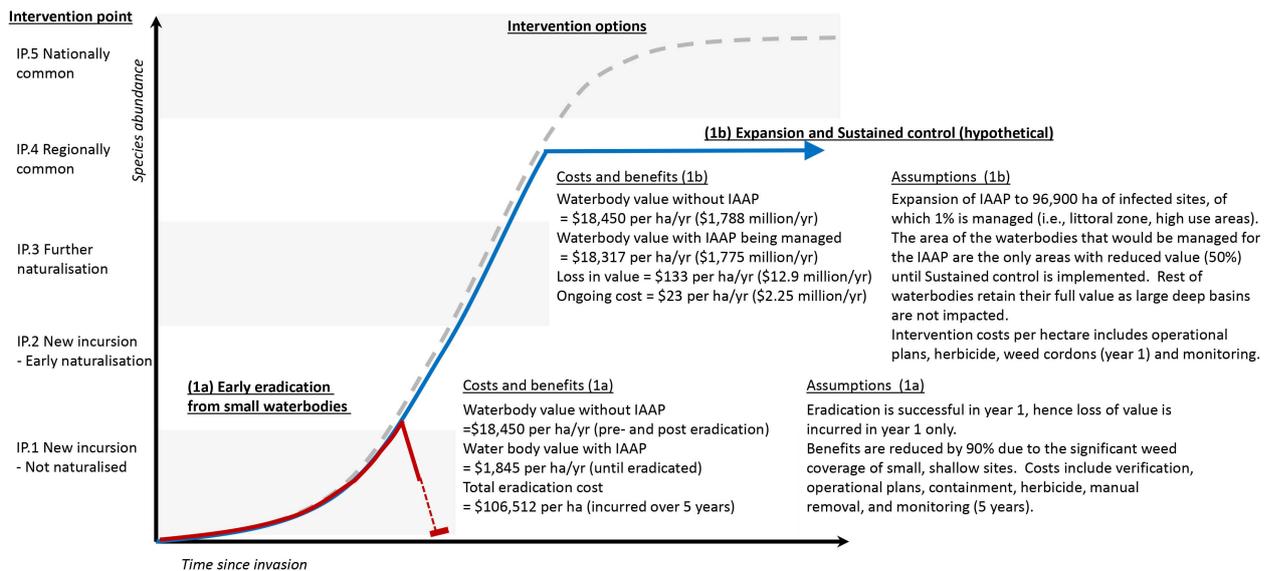


Figure 1. Diagram illustrating the relative costs and benefits of early eradication (1a) compared with sustained control (1b) of a submersed IAAP. Early eradication (red line) is an example from two small waterbodies, and Sustained control (blue line) represents a hypothetical scenario based on actual current distribution of a different species of submersed IAAP in the region. The grey (dashed) line represents the IAAP unmanaged and saturating all available habitat.

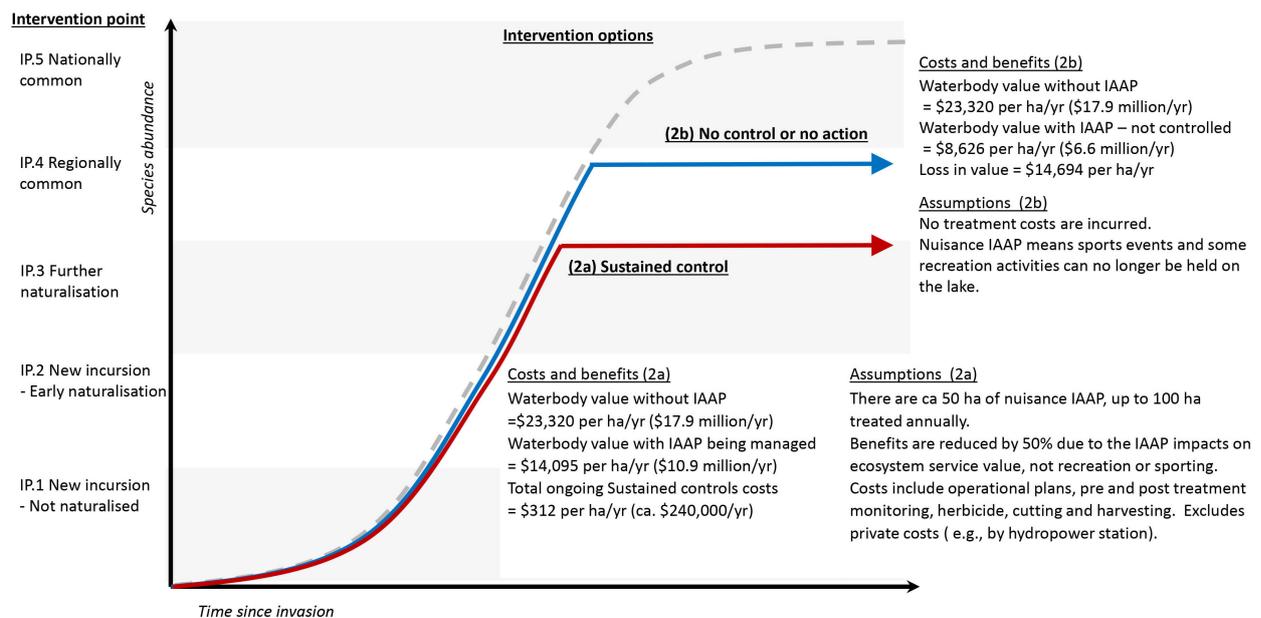


Figure 2. Diagram illustrating the relative costs and benefits of Sustained control (2a) of a submersed IAAP compared with No action (2b). Sustained control (red line), and No action (blue line) are scenarios from the same lake with high recreational and sporting uses and values. The grey (dashed) line represents the IAAP unmanaged and saturating all available habitat (national level).

(scenario 1b). In addition, Figure 1 shows that while the impact on value is high on a per hectare basis in a small waterbody, where the weed smothers the entire area, this is much smaller than the potential loss in value across other waterbodies if eradication were not undertaken. In this case, once *C. demersum* begins to spread more extensively, eradication becomes more costly (both in terms of impacts on values and in terms of intervention costs) and the probability of success decreases.

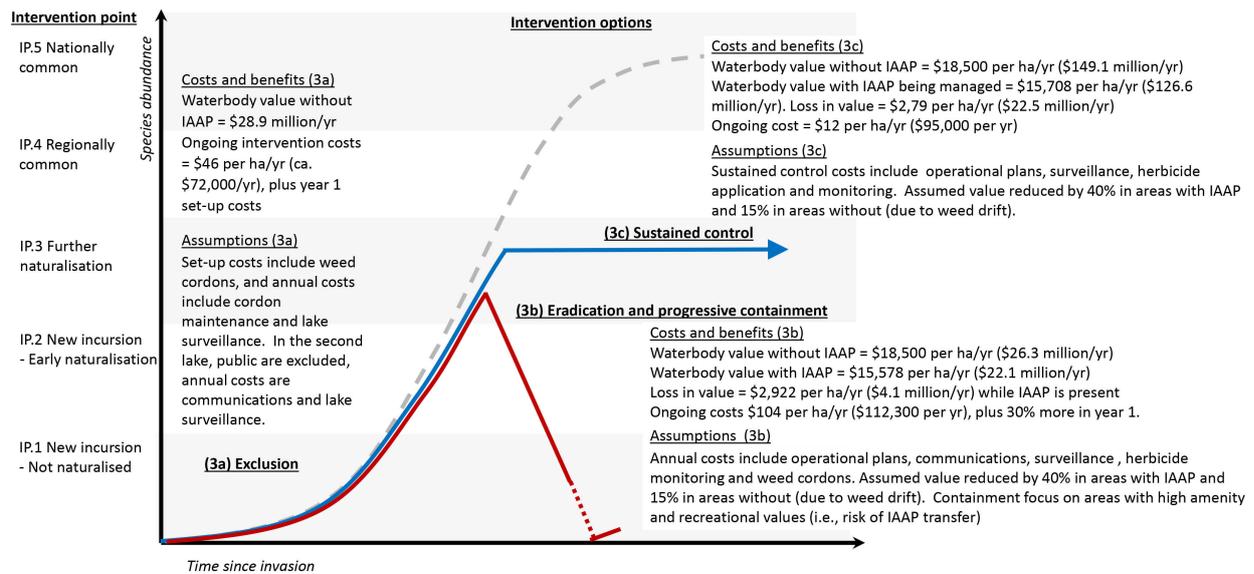


Figure 3. Diagram illustrating the relative costs and benefits of Exclusion (3a), Eradication and progressive containment (3b) and Sustained control (3c) of a submersed IAAP. Exclusion (no line, at IP.1), Eradication and progressive containment (red line), and Sustained control (blue line) are scenarios from different lakes within the same region. The grey (dashed) line represents the IAAP unmanaged and saturating all available habitat (national level).

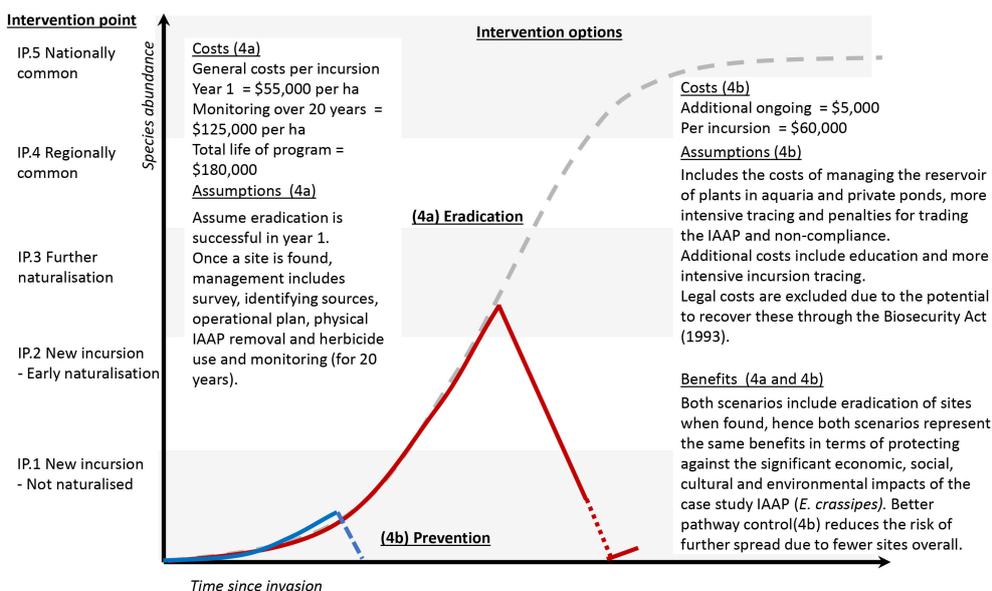


Figure 4. Diagram illustrating the relative costs and benefits of Eradication (4a) compared with Prevention (4b) of a free-floating IAAP. The grey (dashed) line represents the IAAP unmanaged and saturating all available habitat (national level). The red line represents the current recurring incursions and responses (Eradication 4a). The blue line (Prevention 4b) represents greater effort on tracking at source i.e., not naturalised.

The costs and benefits of sustained control versus leaving the submersed IAAP (*C. demersum*) unmanaged are illustrated in Figure 2 (scenarios 2a and 2b). In this case the annual cost per hectare is estimated at \$312 and this creates an annual benefit of \$5,470 per hectare. This significant benefit is largely because control works protect the ability of the lake to hold economic generating events (Figure 2).

The relative costs and impacts of three alternative intervention options for *C. demersum* across six lakes in a single geographic region are

illustrated in Figure 3 (scenarios 3a, 3b and 3c). In this example, exclusion has an ongoing cost, but preserves the value of the waterbody in perpetuity. The eradication option has a higher cost and a larger reduction in value than the sustained control option. This is because the initial costs are higher as there is a larger proportion of the lake area being treated to achieve eradication. However, these are short term costs that will cease once eradication is achieved and the value of the lake is likely to restore itself over time. Conversely, the costs and impacts of sustained control will continue in perpetuity.

Figure 4 illustrates the benefit from investing more in preventing new incursions rather than continuing to eradicate each new incursion once it is detected (scenarios 4a and 4b). This option reduces the costs estimated for eradication by reducing the number of new sites. For *E. crassipes* between 2009 and 2013 there were 31 new incursions (Yamoah et al. 2013), so assuming that 10 incursions of 1 hectare each were found per year, the total costs incurred each year would be \$550,000, with a further investment of \$1,250,000 required for the next 20 years to manage regrowth from the seed bank (Champion and Hofstra 2006) to deal with the additional 10 incursions each year. That also assumes that removal of all plant biomass was achieved in the first year. While there is no information as to how successful investing more in a prevention program would be, this scenario does illustrate that as long as the cost of the preventative pathway control program is less than the total expected costs of eradication for the sites it prevents, it will provide a positive return. This will also reduce the risk of further spread due to the presence of fewer sites. Although the sites that are found will still incur the same costs as in the eradication option (4a), the aim of improved pathway control is to minimise the multiplier, (i.e., the number of new incursions), thereby reducing the total cost of eradication by spending more upfront.

Discussion

This paper has used case studies to test principles related to intervening in IAAP incursions. The analysis has looked at three key principles:

1. The sooner eradication is started and/or the smaller the incursion site, the lower the cost.
2. Eradication is more expensive than alternatives such as not intervening in the short term but is cheaper than long term impacts from no action and/or sustained control methods.
3. Preventing incursions will be cheaper in the long run than eradication.

It is a widely accepted view that eradication has a higher probability of success and lower cost the sooner it is initiated (e.g., Cardoso and Free 2008; Larson et al. 2011; Simberloff et al. 2013; Brunel et al. 2013;

Carboneras et al. 2013; Hussner et al. 2017). This principle has been clearly demonstrated in the present study by the case studies of *C. demersum* incursions that were successfully eradicated, compared with scenarios where *C. demersum* is widely established.

The benefits of early eradication are discussed in other literature, including Panetta (2009) who note that eradication is desirable relative to other alternatives due to the ability to remove the requirement for permanent, ongoing investment of resources. Rejmánek and Pitcairn (2002) determined the limits of eradication feasibility from an analysis of actual eradication attempts. They used data from 18 species in California (53 separate invasions) over a 28 year period. While the paper excluded the biological and ecological attributes of the invasive species, they found that the costs associated with eradication increased as the infested area increased (Rejmánek and Pitcairn 2002). They report that the mean number of work hours per eradicated infestation increased from 63 for areas less than 1 hectare, to 1,845 for infestations between 101 and 1,000 hectares, while very few infestations over 1,000 hectares had been eradicated. This was relative to a mean eradication effort of 42,751 work hours per infestation for ongoing programs (Rejmánek and Pitcairn 2002). Harris and Timmins (2009) report that the costs associated with eradication and management increase as the incursion area increases, indicating that if control is postponed until a stage when the infestation is widespread or dense (i.e., \geq IP3), it is on average, 40 times more expensive than early removal, largely due to finding and removing all plants. Similarly, Hussner et al. (2017) note that while costs vary amongst species and intervention options, costs generally increase as the plant biomass increases.

Not only does the cost of intervening increase as a species moves through the successive intervention points, but often there are less intervention options available. For example, while physical intervention such as a benthic barrier can be used to eradicate *C. demersum* on a small scale (e.g. new incursions, IP1 and 2), it is unlikely to be an effective eradication tool over large areas (Champion et al. 2019; Bellaud 2014). However, the eradication goal may become attainable as a result of the development of future technologies and control options for IAAPs.

Simberloff (2014) noted that the likelihood of reinvasion must be identified and Panetta (2009) suggested that reinvasion must be prevented. In addition, we suggest that the likelihood of reinvasion, should not only be considered, but the economic costs of this should be included. For example, the additional costs of eradication from waterbodies that pose a high degree of risk for re-infestation of the site in question, or the costs of prevention (such as surveillance of high risk sites, weed cordons (Lass 2012) or restricted movement controls) should also be included. If these costs are not factored in, it is likely the cost incurred in the original eradication program will produce limited returns. This was illustrated in the scenario

(2, Lake Karāpiro) where eradication is not considered a possibility due to the connectivity to other infestation sites further upstream of this waterbody, compared to the regional approach (3a–c, Rotorua) where eradication is being considered in some lakes.

In addition, it is imperative that clear counterfactual and desired states are defined. The counterfactual is the situation that would exist if the decision is not made and/or action is not taken, sometimes defined as the “do nothing” or as the “do minimum” scenario (Treasury 2015). For example, as in the case studies 1a and 1b, the decision to eradicate should consider what would be the cost had no action been taken, and *C. demersum* spread extensively as another submersed IAAP has. This can be particularly challenging in invasive species analysis, as often the incursion pathway is not straight forward and depends on many unpredictable factors, not the least of which is human interaction with freshwater. In addition, defining the desired state is important. For example, if an IAAP is eradicated this does not necessarily mean that a water body will return to the pre-incursion state, or even an ideal state. While eradicating a weed may enable the return of a waterbody to an improved state, it is also possible that irreversible damage resulting from freshwater plant invasion may have occurred (for example destroying indigenous flora and/or fauna) (Champion 2002).

Hussner et al. (2017), amongst other authors, suggested that the most efficient way to reduce current and future negative impacts (including intervention costs) is to prevent the introduction of aquatic invasive species. This has been illustrated by the numerous sites of *E. crassipes* in New Zealand (case studies 4a and 4b). While incursion sites are typically small and are eradicated at a relatively low cost, seed longevity means that the costs continue for 20 years after all plants are destroyed, which makes the overall program costs high. In addition, new sites are reported each year, indicating that the current controls, prohibiting the propagation, distribution and sale of this plant, are not preventing its cultivation and subsequent release into field sites. Strictly-enforced pathway controls or prohibition of re-entry is one of the key lessons from other terrestrial eradication campaigns (e.g., *Berberis vulgaris*, Mack and Foster 2009). In the present study, the benefits of stricter pathway plans would be recognised if the cost of undertaking this IO reduced the number of incursions. Provided the costs saved from reduced incursions is less than the cost of implementing the stricter pathway controls, then this would represent a net benefit.

One of the primary challenges in identifying the economics of intervention options at different intervention points is estimating non-market valuation. While Patterson and Cole (2013) provided a useful starting point for estimating non-market values, this does have some challenges. This paper was founded on international literature with local adjustments where possible, but there are some specific values where this

was not feasible (such as cultural services). In addition, while Patterson and Cole (2013) calculated non-use values for some ecosystem services, they note that this was an area in need of further extensive work. There is also a considerable gap in relation to the habitat provided by lakes and rivers for a range of species. Passive use values (option, bequest and existence) for rivers and lakes are primarily based on a limited number of international studies not considered suitable for use in the present study. In addition, emphasis was placed on the importance of articulating what values have been counted in contributing studies so that values were not double counted (Patterson and Cole 2013).

An alternative method to valuing ecosystem services (e.g. Patterson and Cole 2013) would be to use willingness to pay studies. Studies which use willingness to pay or accept methodologies tend to reflect values as a financial cost that households or users (such as anglers) would pay to preserve something of value to them. For example, Kerr (2004) estimated recreation benefits in the order of \$36 per angler-day for freshwater sport fishing and \$21 per recreator-day for other activities in New Zealand. This is not readily translated to a per hectare calculation, nor is it directly comparable to Patterson and Cole (2013), given the differences in the values that were included or excluded. In addition, there are very few studies that directly link willingness to pay with IAAPs.

One study that does specifically evaluate what households would be willing to pay for protecting indigenous biodiversity is Bell et al. (2009). Bell et al. (2009) examined the potential impact of the IAAP, *H. verticillata*, on indigenous biodiversity using choice modelling for an urban lake (Lake Rotoroa) in a hypothetical example. Their study assessed the attributes of charophytes (native submerged plants perceived as beneficial to freshwater environments) and the percentage of success in preserving them, birds (number visiting the lake), fish and mussels (number of fish species and mussels retained) and water quality (four levels of deterioration relative to the current state). The probability of successfully controlling *H. verticillata*, and the cost to each household for five years were also included. The results were aggregated using 2006 census data for the areas surveyed, the net present value of the compensating surplus was \$348 million for the region and \$3 billion for the country (five years and 8% discount rate) (2006\$). While Bell et al. (2009) focused on *H. verticillata*, the way that choice modelling studies work is that they focus on specific impacts on the attributes measured, e.g. a change in abundance of native charophytes. Therefore, these benefit transfer values can also be attributed to *C. demersum* control scenarios. However, when undertaking benefit transfers, there needs to be careful consideration of the regional attributes and the context attributes. Namely, lakes in the same region with different dimensions, floral and faunal assemblages and amenity and utility values are not directly comparable for benefit transfer.

An alternative approach in the scenario of sustained control versus no action (2a and 2b, Lake Karāpiro) would be to use the examples of Marsh and Baskaran (2009) and Marsh et al. (2011) who valued water quality improvements through a choice modelling approach. Marsh and Baskaran (2009) looked at the values for those living in the lake catchment while Marsh et al. (2011) looked at recreational users of the lake. An issue with extracting benefits from Marsh and Baskaran (2009) and Marsh et al. (2011) is that they value water quality based on; suitability for swimming and recreation (probability of health warnings); water clarity (visibility under water in metres), and ecological health (percentage of excellent readings). They also exclude the returns from activities such as sporting events and power generation and non-use values such as bequest. Therefore, it is likely to undervalue some of the specific impacts of *C. demersum* (e.g. option value), while overvaluing some of the benefits provided by treating *C. demersum* (such as its contribution to swimming without drowning or “swimmers-itch” risk). Despite this, the approach is site specific and therefore provides a useful perspective.

Although there are challenges with non-market valuation, decision makers must still consider non-market values in any incursion. Analysis must include the cultural, social, economic and environmental impacts of each decision in the same framework as traditional monetary metrics in order to achieve the best outcomes (Pejchar and Mooney 2009). Where possible quantifying non-market values will ensure that a more robust decision is made and will lead to more optimal outcomes for society (Bell et al. 2009). In addition, legislation is increasingly requiring that where practicable impacts and costs are considered (e.g. European Union 2014; NPDP, (New Zealand Government 2015)). Where these values cannot be quantified in monetary forms, the relative magnitude of impact for various options should still be incorporated into a decision-making framework. Some values are not necessarily suitable for consideration in monetary terms, especially cultural values such as mauri, which would likely be culturally insensitive, if not a prohibitively controversial exercise (Robb 2014). However, there is also research to better understand how to incorporate cultural values into freshwater decision making and Miller et al. (2015) argue that removing cultural values from decision making can lead to sub-optimal outcomes which are often irreversible.

Currently, there is a shortage of studies which attempt to understand and quantify where possible the impact of IAAPs on non-market values. It is suggested that there is scope to build on this limited literature and that which examines the willingness to pay for, or the ecosystem services provided by, improved water quality, and refine this to focus on the specific impacts from IAAPs. This could focus on key species traits to ensure applicability for future incursions of new species. This would in turn enhance the ability of decision makers to consider non-market values

in CBAs in relation to specific incursions which are often too time and resource pressured to undertake suitable primary analysis in each case. This has been echoed by management agencies (e.g. Auckland Council 2018) who state that many of the most important benefits, such as impact on native biodiversity, currently cannot be meaningfully quantified due to a lack of data. This work should ensure that studies are designed in such a way to minimise the risks associated with benefit transfer of applying research results to specific incursions and CBAs.

While non-market valuation is challenging, estimating market costs can also be difficult. In particular, there is often a lack of transparency of estimation of intervention costs with many not being recorded and/or publicly available. Other challenges with estimating costs include estimating the costs carried by private or commercial groups (e.g. those who operate hydropower stations or private boat users) and where costs have multiple benefits (e.g., treating multiple IAAPs at one time). As mentioned, CBAs are often the main tool used to inform decisions around intervening in IAAP incursions and in many cases they are a useful tool. While this paper did not seek to create CBAs or delve into specific incursion scenarios with a view to make an intervention decision, many of the challenges are similar, including defining counterfactual scenarios, non-market valuation and discount rates. Discount rates adjust future values to today's dollars and are an essential part of CBAs. Discounting is one of the most controversial aspects of CBAs and often has a bigger impact on the outcome than any other factor (Treasury 2015). In addition, the length of time included in a CBA tends to extend to 10 years, after which the benefits and costs have a relatively small impact on the final result due to discounting. There is a growing discussion on what the appropriate discount rates and time periods are, that should be used in environmental decision making (e.g. Almansa and Martinez-Paz 2011; OECD 2018; Parker 2011), including the use of very low discount rates (1%), using time-declining discount rates (in part to better include uncertainty) and how intergenerational impacts (such as biodiversity decline) are accounted for. This is an area which warrants further investigation in relation to the costs and benefits of specific incursion.

Conclusions

IAAPs present a constant threat to freshwater resources. Managing existing and future incursions is incredibly complex and usually site and species specific. This paper shows that if the management goal is eradication, the sooner eradication is initiated the cheaper it will likely be, with a lower risk of the negative impacts extending to other waterbodies. There is no universal point where eradication becomes uneconomic, due to the varying values and costs associated with each incursion. However, to provide a positive return for society, it is economically logical to progress an

eradication program until the costs outweigh the benefits. This argument is not simple, as the benefit must consider the potential loss in value resulting from the likely spread of an IAAP to further waterbodies should eradication not be achieved, as well as the potential ongoing control costs. In addition, the ability to eradicate decreases with incursion size and therefore, decisions must also consider the potential feasibility of current preferred options, at future points in time.

This paper builds on existing literature and uses specific case studies to test general principles related to intervention in IAAPS which can be used to improve decision-making. Based on the analysis presented here the three principles proposed at the start of the paper can be broadly considered to be logical in an economic sense and contribute to supporting intervention decision-making. Further, some of the discussion provided here will contribute to incursion specific CBAs as well, in particular the debate concerning non-market valuation and costs.

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