RESEARCH ARTICLE

The economic cost of managing invasive species in Australia

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Abstract

Like most jurisdictions, Australia is managing a broad range of invasive alien species. Here, we provide the first holistic quantification of how much invasive species impact Australia's economy, and how much Australia spends on their management. In the 01–02 financial year (June to July), the combined estimated cost (economic losses and control) of invasive species was $9.8 billion, rising to $13.6 billion in the 11–12 financial year. Approximately $726 million of grants funded through the Commonwealth of Australia (i.e. federal funding) was spent on invasive species management and research between 1996 to 2013. In 01–02, total national expenditure on invasive species was $2.31 billion, rising to $3.77 billion in 11–12. Agriculture accounted for more than 90% of the total cost. For 01–02 and 11–12, these expenditure figures equate to $123 and $197 per person per year respectively, as well as 0.32 and 0.29% of GDP respectively. All values provided here are most likely to be underestimates of the real values due to the significant constraints of the data obtainable. Invasive species are clearly a significant economic burden in Australia. Given the extent of the issue of invasive species globally, there is a clear need for better quantifications of both economic loss and expenditure in more jurisdictions, as well as in Australia.

Keywords

cost, exotic, impact, invasion, economy

Introduction

Australia is a world leader in biosecurity policy and management, having some of the world's most stringent biosecurity. These controls are necessary to assist protecting Australia's biodiversity, agriculture, and aesthetic values. But like most jurisdictions, Australia is managing a broad range of invasive alien species due to a legacy of both accidental and deliberate introductions. The impacts of many invasive species in Australia are some of the most dramatic and well known globally and include extreme seasonal plagues of rabbits (Hall et al. 1964) and mice (Mutze 1989), dominance within, and turbation of, freshwater systems by European carp (Harris et al. 1998) to the blanketing of southern Australian agricultural landscapes by Paterson's curse, *Echium plantagineum* (Parsons and Cuthbertson 2001).

Of the approximately 2700 exotic plants species now established within Australia, 429 have been declared noxious or are under some form of legislative control (NRM-MC 2006) with considerably more subject to eradication and control measures such as plant species listed on the National Environmental Alert List (http://www.environment.gov.au/biodiversity/invasive/weeds/weeds/lists/alert.html). The economic cost of weeds to the Australian economy within agricultural areas alone is estimated to be approximately $4 billion annually (Sinden et al. 2004; BRS 2007). There are more than 80 species of exotic vertebrates that have established wild populations (Bomford and Hart 2002; BRS 2007), with the eleven most problematic species alone having negative impacts estimated a decade ago to cost $720 million annually (McLeod 2004). Invasive invertebrates are estimated to create annual agricultural production losses of $4.7 billion annually (BRS 2007) and cost up to $8 billion annually considering all impacts and expenses (Canyon et al. 2002), with the red imported fire ant, *Solenopsis invicta*, being the target of Australia's most expensive eradication campaign, costing approximately $300 million to date (Keith and Spring 2013).

Surprisingly, given the extent of Australia's issues with invasive species, and the global need to increase public awareness of the issue of invasive species, data of expenditure on invasive species management is difficult to obtain. However, this issue of poor data availability is not just restricted to Australia, and arises from both the difficult nature of costing the expense of invasive species as well as the lack of good data collection by agencies. As an example of a costing difficulty, most herbicides are broad spectrum and they are used to control both native and exotic weeds within the same crop making it difficult to cost the financial implications of exotic species alone. Where data exist they are largely estimates, predominantly associated with agriculture expenditure or loss, focused on individual taxa (e.g. weeds, vertebrates, invertebrates: McLeod 2004; Sinden et al. 2004; Gong et al. 2009), and reported in incompatible formats (e.g. project-level vs aggregated information) making holistic costs extremely difficult to calculate. Regarding data collation, the lack of financial transparency is largely the result of funding being provided by a multiplicity of agencies through a range of funding programs at various levels of jurisdictional responsibilities (i.e. local, state and federal government), or private enterprises with little to no requirement to

report such information in a co-ordinated manner. No level of government or any private enterprise details invasive species management as a distinct expenditure, and unlike the European Union (Scalera 2010), Australia has no publically available database that allows holistic quantification of expenditure on invasive species management for any jurisdictional level, for any purpose (e.g. agriculture versus conservation) or for any funding program. As such, it remains unclear just how much Australia actually spends on managing invasive species.

The largest single source of environmental funding within Australia is provided by the federal government and for the last 20 years has been primarily allocated through three programs: National Heritage Trust (NHT, 1997–2008), Caring for Our Country (CfOC, 2008–2013) and the Biodiversity Fund (2011 to current). Additionally there have been two more programs specifically targeting invasive species: Defeating the Weeds Menace (DtWM, 2004–2009) and the National Weeds and Productivity Research Program (NWPRP, 2010–2012). Although some analyses have been conducted to quantify expenditure on invasive species within these programs, for example against weeds for NHT (Martin and van Klinken 2006), the holistic figure of expenditure for all invasive species by these programs is not clear. Here, we provide the first holistic quantification of Australia's economic loss and expenditure on invasive species in terrestrial and freshwater systems by examining data available from annual reports for these programs as well as that from other sources that calculate the economic loss imposed by, and expenditure on, invasive species. We envisage that these data will be useful globally to assist with raising general awareness of the importance of invasive species and biosecurity. Importantly our data do not include diseases or pathogens because management expenditure on these taxa largely do not come from competitive federal grants, and these taxa also cross into the human health arena which is outside of the focus of this study. Where possible we have excluded data for these taxa from cited publications. We also intentionally only conduct analyses at the national level to provide a broad overview of national expenditure for an international audience.

Methods

Holistic economic loss and expenditure

To provide a holistic picture of the economic loss imposed by, and expenditure on, invasive species within Australia we obtained financial data from accessible sources with a key focus on invasive species management or research within the 01–02 and 11–12 financial years (July to June). These reporting periods were used as they were the only years where documents reported some of these data. Data of estimates of economic loss imposed by invertebrates, vertebrates and weeds for 01–02 were sourced from Canyon et al. (2002), McLeod (2004) and Sinden et al. (2004) respectively. Because for invertebrates and weeds there were no documents that superseded those used for 01–02,

and because the vertebrate data presented in Gong et al. (2009) was not as comprehensive as those presented in McLeod (2004), the 01–02 data for these three taxa were used again for 11–12 but were adjusted for inflation to 2012. Data of expenditure was sourced from federal grants (detailed below), federal agencies (e.g. Australian National Parks and Wildlife Service), state-level agencies (e.g. Departments of primary industries and conservation), Cooperative Research Centres (CRC's), and reports of farm expenditure. Expenditure by federal and state-level agencies were sourced from Sinden et al. (2004) for 01–02 and Gong et al. (2009) for 11–12, although the data from Gong et al. (2009) was for 07–08. CRCs are funding hubs for research and were chosen because they are the subject of significant funding initiatives and have relatively transparent reporting. Budgets of four CRCs with a primary focus on invasive species (i.e. Invasive Animals CRC, Plant Biosecurity CRC, Weed Management CRC, and Biological Control of Pest Animals CRC) were sourced from annual summary documents accessed from the CRC website: www.crc.gov.au (accessed 29 April 2014). Because only total budget data over the lifespan on CRCs (5–7.5 years) were provided, we divided the total budget for each CRC by its lifespan to estimate the budget for each financial year. Additionally we divided the CRC funding data by whether the focus of each CRC was on animals or plants. Data of economic loss and expenditure on farms for 01–02 were sourced from Sinden et al. (2004), and for 11–12 from the 2013 National Landcare Survey of the impact of pests and weeds on farming enterprises and the costs associated with their control (www.landcareonline.com.au/wp-content/uploads/2013/10/Final-2013-NLF-Survey-Results-Summary.pdf).

Data of economic loss and expenditure were also obtained and summarised for as many other global jurisdictions that we could find. Data were obtained for the USA in 2003 (Pimentel et al. 2005), the European Union in 2006 (Kettunen et al. 2009; Scalera 2010), Australia in approximately 1998 (Pimentel et al. 2001), and for the USA, UK, India, South Africa, Australia and Brazil combined in approximately 1998 (Pimentel et al. 2001). Data of loss and expenditure combined were obtained for Canada in approximately 2000 (Colautti et al. 2006) and Southeast Asia in 2011 (Nghiem et al. 2013). Data of loss and expenditure were also obtained for New Zealand in 2008 (Giera and Bell 2009), for a few species in Germany in approximately 2002 (Reinhardt et al. 2003), and for 12 species in Sweden in 2006 (Gren et al. 2009). Data of economic loss only were obtained for Great Britain in 2010 (Williams et al. 2010), and two estimates were obtained for China in 2000, being of indirect economic loss of forest insects and pathogens (Li and Xu 2005) and total economic loss (Xu et al. 2006). It should be noted that although all of these studies had the same broad goal to quantify the financial cost of invasive species, they differ greatly in methods, assumptions, data availability, and effort expended, and therefore comparisons can only be made very broadly. In all cases, the data sourced relied heavily on surveys and conservative calculations (see details within respective publications), and often did not report on all exotic species within the target group (e.g. vertebrates). Where data ranges were provided, we used the lowest value which further makes our calculations conservative and likely to underestimate real expenditure.

Federal grants

Annual reports and listings of approved projects within the five Australian federal government funding programs were sourced from each program's respective website [NHT (http://nrmonline.nrm.gov.au/catalog), CfOC (www.nrm.gov.au/funding/approved/index.html), Biodiversity Fund (Round 1, 2011–12, www.environment.gov.au/cleanenergyfuture/biodiversity-fund/round-1/index.html#lists)], or reports obtained elsewhere [DtWM (Oliver et al 2008), NWPRP (RIRDC 2010)]. For the NHT, specific investment through the National Feral Animal Control and Weeds Programs was ascertained for each financial year. Because invasive species management also occurs as part of other natural resource management (NRM) activities, project titles and objectives (where available) were assessed for key words including invasive, control, eradication, pest, weed, rehabilitation, restoration and the common or species names of known invasive and pest species to determine the intent of each project. Funding details for project titles that specifically addressed an invasive species issue were extracted and the number and total value of these were calculated for each financial year. In 2002, NHT changed its reporting methodology and no longer produced lists of approved projects. Hence, we were only able to ascertain total values for funding allocated to their Activity Areas (i.e. project-based themes) of "Pests and weeds" and "Significant invasive species". For CfOC and the Biodiversity Fund, project titles and descriptions were similarly assessed to determine whether projects directly addressed an invasive species issue or intended to undertake management actions as part of broader objectives. Although this approach is likely to inflate total investment by including projects if only a proportion of the grant was directed towards invasive species management, we consider that this is at least partially offset by the exclusion of projects that included invasive species management but this action could not be ascertained from the project title or objectives. For example, projects that involved restoration would have certainly included weed management component but these were not included in our calculations. Any artificial inflation would also be offset by our data not including both direct and indirect financial support by partner organisations in each project.

Additional inconsistencies within and among funding sources were encountered that limit the accuracy of the data. First, CfOC funding was allocated through multiple programs, but the allocation and reporting of funding in each program varied making it difficult to maintain a consistent approach to calculating expenditure. Second, was that CfOC adopted a new model with respect to how Commonwealth funds were devolved. In recognition of the regional and local roles that Australia's 56 regional natural resource management organisations play, these were able to secure 3–5 years of funding (base-level funding) to work with local communities to identify and set local priorities for investment. Consequently, comparisons of pre- and post-2008 for base-level funding cannot be made. In subsequent years where base-level funding is included, these data are likely to be inflated as only a proportion would have been used directly for invasive species management. As such, and because base-level funding was often the greatest proportion of the overall funding, we report data with and without base-level funding.

Analyses

Although we preferably would have analysed data down to numerous taxonomic levels (e.g. lifeform: plants/vertebrates/invertebrates or species-level: snakeweed/rabbits/fire ants), we were not able to provide accurate divisions for most data. For example, for the federal grants, there was no way to determine what the focal invasive species was if the name was not in the project title, and such level of discrimination was only possible in the oldest data (NHT). Additionally, funding allocation was problematic in the numerous instances where multiple invasive species were targeted simultaneously (e.g. crop spraying, woodland restoration). Therefore all invasive species are considered together. In addition to basic data summation, Australian data of economic cost and national expenditure were also expressed in some instances per person (Australian citizens) and as a percentage of Gross Domestic Product (GDP: national financial turnover). Australia's population and GDP data were accessed from the Australian Bureau of Statistics website www.abs.gov.au (accessed 29 April 2014). Because population data were obtained only every five years, we calculated data for the other years by averaging data between census years. We present financial data along timeframes both as raw data, and adjusted for inflation to 2012 for total economic loss and expenditure, and to 2013 for federal expenditure. Data were adjusted for inflation using the Reserve Bank of Australia's Inflation Calculator at: http://www.rba.gov.au/calculator/annualDecimal.html.

Results

Economic loss and expenditure

In the 01–02 financial year, the combined estimated cost (economic loss and expenditure combined) of invasive species was $9.79 billion ($12.88 billion adjusted to 2012 values) (Table 1). For the expenditure that we could obtain data for both 01–02 and 11–12 (federal grants, CRCs, state and federal agencies, farms), after adjusting the 01–02 data to 2012 values, there was an increase in expenditure by $639.55 million from 01–02 to 11–12. When we added this figure to the 01–02 cost data with no 11–12 equivalent, after adjusting the 01–02 data to 11–12 values, the total estimated cost (economic loss and expenditure combined) of invasive species in 11–12 was $13.6 billion (Table 1). However, because this 11–12 figure is predominantly reliant on inflation adjusted 01–02 data we expect it to be a great underestimate because we consider it highly unlikely that costs have not stayed constant or decreased. These estimates of economic cost represented 1.37 and 0.92% of GDP in 01–02 and 11–12 respectively.

Expenditure by federal grants

Approximately $726 million was spent by the Australian federal government on invasive species between 1996 to 2013 in the five funding sources, but with base-level
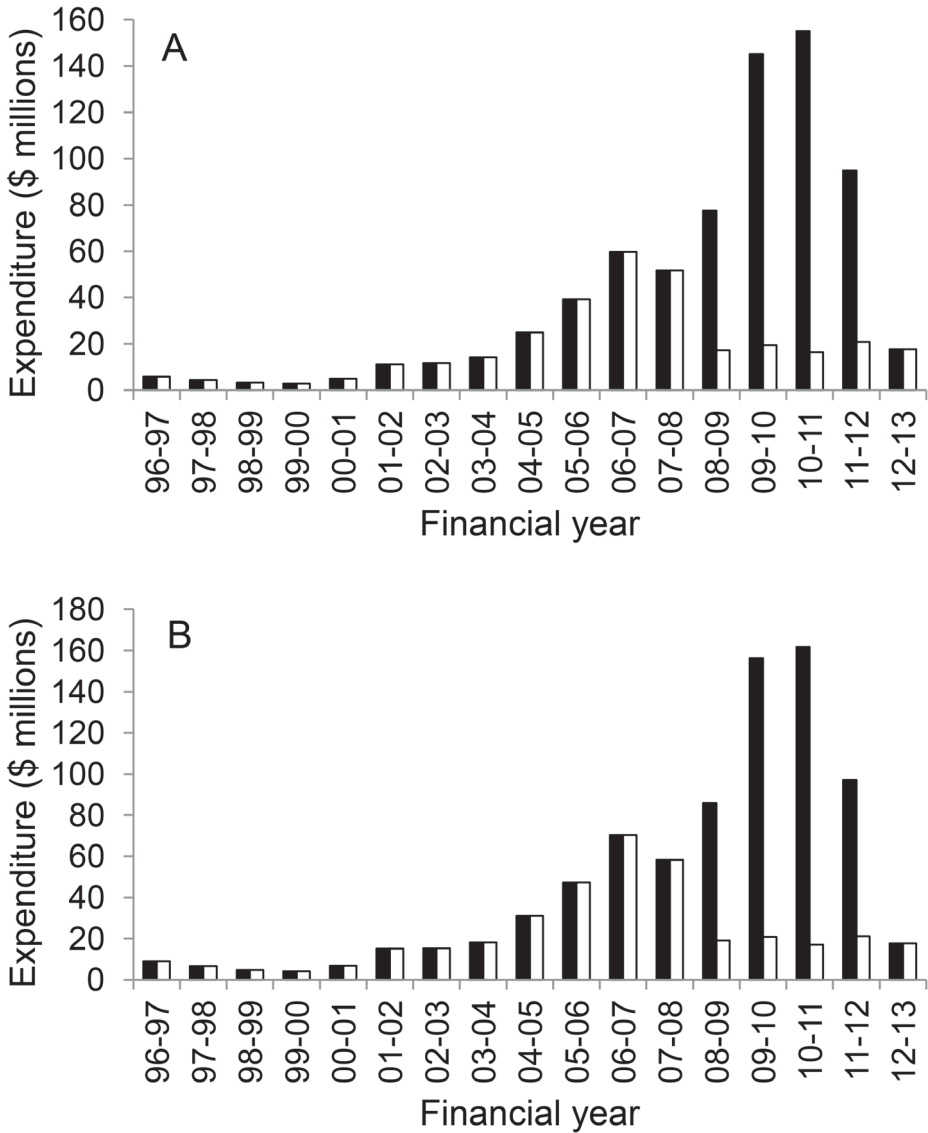
Figure 1. Total expenditure of federal grants per financial year for invasive species management with base-level funding included (black) and excluded (white) using **A** actual data, and **B** data adjusted for inflation to 2013 values.

funding excluded this figure was reduced to $282 million (Figure 1A). The lowest figure spent was $2.9 million in 98–00, and the greatest was $155.08 million in 10–11 with base-level funding included, and $45.6 million in 06–07 with base-level funding excluded. Patterns of expenditure among years were almost identical for raw data and data adjusted for inflation to 2013 values (Figure 1B).

Table 1. Estimated economic loss and management expenditure for exotic species for numerous countries and timeframes. Data of different timeframes are not adjusted for inflation. For reports of individual countries except Australia only the most recent report was used. Reports are listed in order of the year that the data relate to.

Location	Data sources	Description of data used here and applicable year	Economic loss	Management expenditure	Management expenditure as % of economic loss	Economic loss and management expenditure as % of GDP
USA, UK, India, South Africa, Australia, Brazil combined	Pimentel et al. 2001	All data^, for approx. 1998	USD$306 billion	USD$30 billion	8.9	–
Germany	Reinhardt et al. 2003	All data for 20 species, for approx. 2002	EUR€113.4 million	EUR€53.7 million	Not calculated as data are only for a few species	Not calculated as data are only for a few species
China	Li and Xu 2005	Indirect costs only of forest insects and pathogens^ for 2000	¥15.44 billion	–		0.01
China	Xu et al. 2006	All data, for approx. 2000	USD$14.45 billion**	–	–	0.01
Canada	Colautti et al. 2006	16 species^, for approx. 2000	CDN$34.5 billion**	–	–	Not calculated as data are only for a few species
USA	Pimentel et al. 2005	Species with data of both impact and control, excluding microbes and disease for approx. 2003	USD$40.31 billion	$USD9.01 billion	22.4	0.96
USA	Pimentel et al. 2005	All data excluding microbes and disease for approximately 2003	USD$88.64 billion	USD$12.03 billion	13.6	1.96
Sweden	Gren et al. 2009	All data for 12 species, excluding HIV, for 2006	SEK1911.5 million	SEK852.5 million	Not calculated as data are only for a few species	Not calculated as data are only for a few species
EU	Impact cost from Kettunen et al. 2009, management expenditure from Scalera 2010	25 species, for 2006	EUR€20 billion	EUR€18.3 million*	Not calculated as data are only for a few species	Not calculated as data are only for a few species

Location	Data sources	Description of data used here and applicable year	Economic loss	Management expenditure	Management expenditure as % of economic loss	Economic loss and management expenditure as % of GDP
New Zealand	Giera and Bell 2009	All data, for 2008	NZ$2454 million	NZ$836 million	34.1	1.86
Great Britain	Williams et al. 2010	All data^, for approx. 2010	£1.68 billion**	–	–	0.07
Southeast Asia	Nghiem et al. 2013	All data^, for 2011	$USD33.5 billion**		–	2.61##
Australia	Canyon et al. 2002, McLeod 2004, Sinden et al. 2004	All data, for 2001-2002 financial year	AUD$7.48 billion	AUD$2.31 billion	30.9	1.37
Australia	Canyon et al. 2002, McLeod 2004, Sinden et al. 2004, Gong et al. 2009, 2013 National Landcare Survey & our calculated difference between expenditure in 01-02 and 11-12 for federal grants and CRCs.	All data for 2011-2012 financial year	AUD$9.83 billion	AUD$3.77 billion	38.4	0.92

^ data includes diseases and/or pathogens

* data are for a single funding source, and therefore will be a gross underestimate of the actual figure.

data are older than that for management expenditure but are the most recent obtainable.

** data are both economic loss and management expenditure
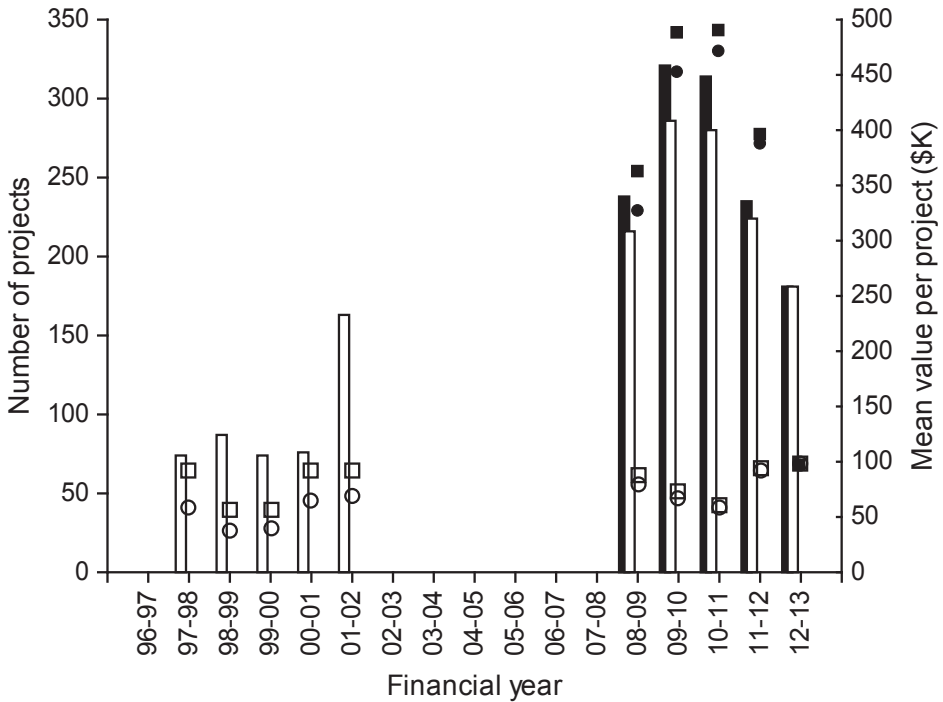
average, excluding Myanmar and Brunei

Figure 2. Number (columns) and mean value (points) of projects funded per financial year including (black) and excluding (white) base-level allocations of funding using actual data (circles), and data adjusted for inflation to 2013 values (squares). Missing data could not be obtained.

Overall there was a trend of an increase in the number of projects and the average value of each project over time, with the patterns of value being consistent for raw data and data adjusted for inflation to 2013 values (Figure 2). The pattern of the number of projects funded was similar with and without base-funding, with years up to 2001 having an almost consistent average of 78 projects, spiking to 163 projects in 01–02, and from 08–09 to 12–13 having an average of 258 projects (Figure 2). In the years up to 2002 the average funding per project was $54K ($77.5 adjusted to 2013 values), whereas from 09–10 to 12–13 the average project cost approximately $359K and $136K with and without base-level funding respectively ($366.3K and $83.1K respectively adjusted to 2013 values) (Figure 2).

Of the three funding sources that were not specific for invasive species (NHT, CFoC and Biodiversity Fund), the proportion of total expenditure on invasive species had a clear gradual increase with time (Figure 3). Lowest expenditure was only 1.4% in 99–00, and greatest expenditure (without base-level funding only) was 40.6% in 12–13. Research funding for the four CRCs with a primary focus on invasive species was split almost evenly for animals (42.4 %) and plants (57.6%) across all years (Figure 4), with expenditure on animals being almost all for vertebrates.
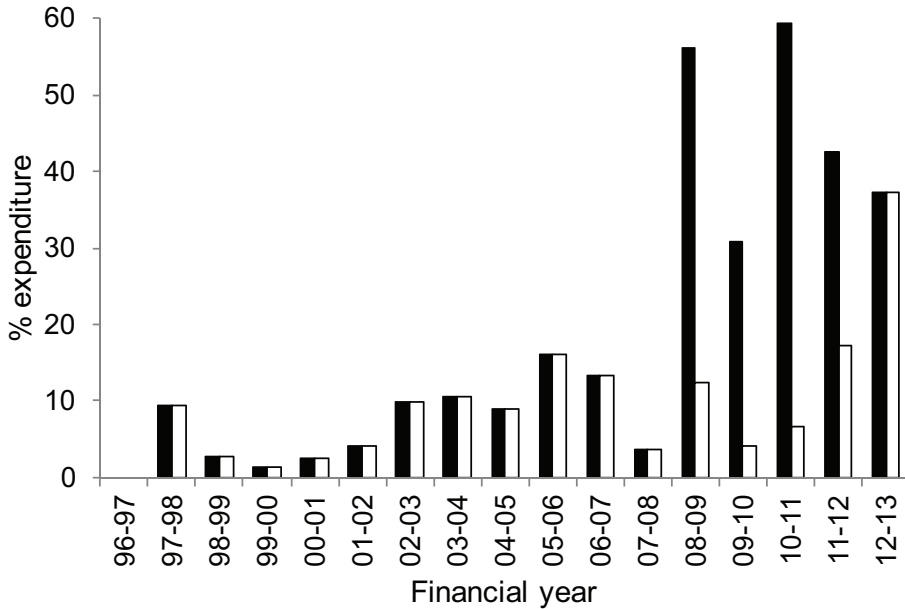
Figure 3. Percentage of total expenditure of federal grants per financial year used for invasive species management with base-level funding included (black) and excluded (white).
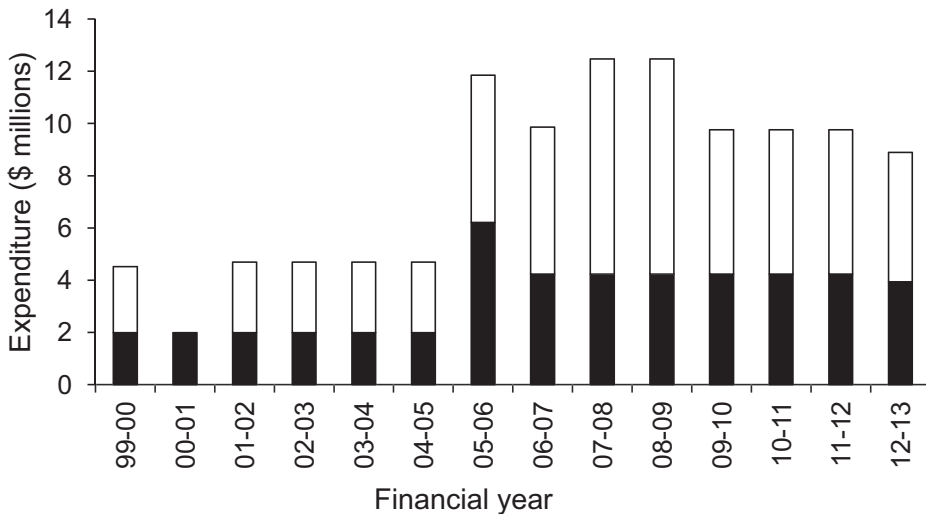

Figure 4. Funding (unadjusted for inflation) supplied to Cooperative Research Centres (CRC) for scientific research focused on invasive animals (black) and plants (white) per financial year. Note that much of the data are uniform across years because yearly data were calculated by averaging the total funding figure of each CRC over its lifespan.

Total national expenditure

In 01–02, total national expenditure on invasive species was $2.31 billion ($3.03 billion adjusted to 2012 values), rising to $3.77 billion in 11–12 (Table 1). Notably, the 11–12 figure is predominantly reliant on inflation adjusted 01–02 data and therefore is expected to be greatly underestimated because we consider it highly unlikely that costs have not stayed constant or decreased. For 01–02 and 11–12, these total expenditure figures equate to $123 and $197 per person per year respectively, as well as 0.32 and 0.29% of GDP respectively. Although exact relative contributions could not be determined, agriculture accounted for more than 90% of the total economic loss and expenditure in both years (data not presented). Total expenditure in 01–02 relative to the economic cost was 30.9% and was among the highest relative expenditure calculated in the world (Table 1). This figure rose to 38.4% in 11–12.

Discussion

Dealing with invasive species is clearly a significant expense to the Australian economy and environmental budgets. As far as we are aware, the only prior attempt to calculate the holistic cost of economic loss and management expenses for invasive species in Australia is Pimentel et al. (2001). They calculated that in 1998, excluding diseases, invasive species cost Australia $USD 12.33 billion (Pimentel et al. 2001). Here we calculated that in 01–02 the combined cost of economic loss and control was $AUD 9.8 billion. Our figure is lower most likely because Pimentel et al. (2001) made independent calculations for all invasive species within Australia based on many assumptions of the costs of the impacts, rather than relying on the reports that we have here that predominantly detail actual expenditure, but do not include calculated impacts for all species, and also either do not include, or do not provide holistic expenditure for other financial sources such as state-level and local governments, as well as private expenditure. For both Pimentel et al. (2001) and here, many of the data used, are based on conservative minimums, and the calculations made are based on conservative assumptions especially for environmental costs, and therefore the actual costs are likely to be greater. Additionally, our data do not include the multiple hundreds of millions of dollars per year spent on border protection preventing incursions by new species and pathogens. Border protection was not included here because we were unable to obtain the data, and if we had it there would be no clear way to split the relative contribution of the costs for preventing incursions of pathogens which were not included in our analyses. Moreover, our analyses have focused on the cost of exotic species after they have established in Australia rather than preventing further incursions.

Clearly both federal government and total expenditure on invasive species increased over time, with the exception of a more recent decline, both in real terms and as a proportion against all measures (i.e. GDP, per capita, calculated impacts), rising to $3.77 billion in 11–12. But it remains unclear what drove the pattern of increasing ex-

penditure, especially the notable increase post 2008 and then the subsequent decline. Was the increase a response to increasing numbers of invasive species and/or their impacts, new legislation such as the *Environment Protection and Biodiversity Conservation Act 1999* which is the Australian Government's central piece of environmental legislation to protect biodiversity of national and international significance, multiple government biosecurity reviews (Nairn et al. 1996; ECITARC 2004; Beale et al. 2008), or other drivers such as a better informed public calling for more government support, increasing levels of land secured for conservation requiring management action, or that the efficacy of management is improving thereby making expenditure on invasive species management more appealing? We suggest that the answer is likely to be the interplay of all of these factors. Notably, the increase in 2008 funding possibly reflects the election of a new Commonwealth Government in November 2007 and the subsequent shift to the Caring for Country Program in 2008. We are unsure of why funding decreased in 2011 but possibly this reflects Commonwealth Government environmental expenditure priorities changing towards clean energy, coupled with the decline in the Australian economy, rather than money being transferred to sources unaccounted for in this analysis.

An aspect of the expenditure that remains unquantified is the outcome. Are management efforts reducing the influence and extent of invasive species, or are they merely slowing an inevitable spread and rise of impacts? This question has been queried for weed management funded by NHT grants, but it was found that it was not possible to assess program effectiveness due to inadequate reporting requirements, as well as the timing of management programs usually extending far beyond short-term funding arrangements (Martin and van Klinken 2006). Likewise we were unable to perform any such analyses here that quantify management success, value for money, or even progress towards mitigating invasive species impacts in Australia. There is no doubt that there are many localised successes that have mitigated an environmental issue by preventing incursions, successfully controlling an invasive population (Reid and Morin 2008; Palmer et al. 2010), eradicating an invasive species (Oppel et al. 2010; Hoffmann 2011; Tobin et al. 2014), and restoring environments following invasive species removal (Hoffmann 2010; Holsman et al. 2010; Jones 2010), but what about at the regional and national levels? If the latest Australian State of the Environment report is a good indicator, then it is likely that for most species, and hence for invasions in general, that the presence and influence of invasive species is increasing. In its summary, the report states that "pressures, such as those from invasive species, are generally increasing." (SOEC 2011). Unfortunately the rise in the influence of invasive species in not just restricted to Australia, and appears to be the consistent pattern globally (SCBD 2014).

At the global scale, Australia was the jurisdiction with the highest expenditure relative to the estimated economic losses in 11–12 and among the highest is 01–02, but these data should be interpreted with caution because there is such data paucity that very few comparisons could be made. Importantly, most data available for comparison are approximately a decade older than the most recent data presented here, and all

data have limited accuracies. In the only estimate of the cost of invasive species to the global economy, Pimentel et al. (2001) estimated that invasive species cost the world USD$1.4 trillion in 1998, which equated to 5% of the global economy. However, this figure included losses due to diseases and pathogens, as well as management costs. Excluding diseases and pathogens, this re-calculates to a global cost of USD$974 billion, equal to 3.1% of global GDP. Interestingly this global percentage figure is much higher than that for Australia in 01–02 (1.37%) and 11–12 (0.92%). It is not clear if this discrepancy is due to Australia having a relatively less problem with invasive species (a very doubtful reality), a relatively stronger economy that is based on industries with less invasive species issues (e.g. mining), fewer invasive species due to stronger biosecurity laws and enforcement, inadequate calculations of the true cost of invasive species to Australia, or the estimates of Pimentel et al. (2001) are too high.

In collating the data for Table 1, it became particularly noticeable that the methods used by different studies to calculate or estimate monetary values varied considerably, often resulting in extremely different outcomes. For example, the most recent attempt to quantify the cost of invasive species to the UK determined the figure to be £1.68 billion (Williams et al. 2010), whereas a report approximately a decade earlier estimated that exotic insects and plant pathogens alone cost the UK £3.08 billion annually (Pimentel 2002). This methodological issue is not easily resolved, because many real values, such as expenditure on pest controllers at a national level divided by taxa, are not able to be sourced, and other non-physical values, such as the value of environmental services or aesthetics, are completely dependent upon opinionated assumptions. Additionally multiple invasive species usually co-occur to produce an impact, or are managed simultaneously, and often the influence of an invasive species is completely unknown. Such high levels of uncertainty result in researchers depending upon "rough guesses" (Pimentel 2002) to estimate costs. Given that there is no simple solution to the problem of how best to determine the cost of invasive species, there can be no doubt that this issue will continue to plague this research arena for quite some time.

Our attempts to collate data for our analyses also highlighted some serious shortcomings globally of the recording and availability of data relating to invasive species, which then greatly hinders analyses that can be conducted. Inadequate data recording was particularly noticeable for Australia's federal funding whereby in most instances we were not even able to obtain information on the focal species of individual projects, thereby preventing even the most basic species-level analysis. These data could be obtained for the oldest data (NHT: 1997–2002), but only if the species name was in the project titles, so any analysis would likely be under-reporting the reality. As such, we cannot even provide a basic analysis showing current relative expenditure on individual species to determine a hierarchy of focus. In turn, this prevents analyses that assess whether funding priorities reflect priorities of need (ie whether species estimated to have the greatest economic cost receive the greatest management and funding).

Given the extent of the issue of invasive species globally (Mack et al. 2000; Pimentel et al. 2001), and that the issue continues to increase, there is a clear need for better quantifications of both cost and expenditure throughout much more of the world

to improve awareness and action, as well as to accurately understand the extent of the problem. The ability to provide such information, at least for expenditure, would be greatly enhanced by the use of publicly available databases, such as the LIFE and CORDIS databases recently analysed by Scalera (2010). Within Australia, reporting of project outcomes for the latest federal funding program (Biodiversity fund) has recently been changed to an online system (MERIT) that aims to provide the first long-term holistic reporting system that can be analysed to quantify project outcomes and national goals for invasive species management. The system is housed by the Atlas of Living Australia (http://spatial.ala.org.au). This development represents a great step towards better quantification of management costs and outcomes against invasive species, and is required for two reasons: 1. to determine the effectiveness of programs given the investment that has been made, and, 2. to guide investment in future programs to ensure these are effective. A major criticism of many invasive species management programs is that they are constantly underfunded, however, without a critical evaluation of cost and effectiveness such claims cannot be substantiated.

Overall, invasive species are a significant economic burden in Australia. The cost of managing invasive species is likely to increase due to more species arriving each year, more species already here becoming problematic and therefore requiring management, and because of problematic species continuing to enlarge their distributions. Better quantification of the cost of invasive species is required to help improve public and political awareness of the issue of invasive species, and to assist with decisions of how to respond appropriately to them, particularly regarding cost-effectiveness of management expenditure. Ultimately, the data support Australia's use of stringent biosecurity measures to help reduce the arrival and subsequent establishment of new species (and pathogens).

Acknowledgements

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RESEARCH ARTICLE

A cost-benefit analysis of controlling giant hogweed (*Heracleum mantegazzianum*) in Germany using a choice experiment approach

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Abstract

Since first of January 2015, the EU-regulation 1143/2014 obligates all member states to conduct cost-benefit analyses in preparation of control programs for invasive alien species to minimize and mitigate their impacts. In addition, with ratification of the Rio Declaration and the amended Federal Nature Conservation Act, Germany is committed to control any further spread of invasive species. This is the first cost-benefit analysis estimating positive welfare effects and societal importance of *H. mantagezzianum* invasion control in Germany. The paper analyses possible control options limiting stands of giant hogweeds (*H. mantegazzianum*) based on survey data of n = 287 German districts. We differentiate between several control options (e.g. root destruction, mechanical cutting or mowing, chemical treatment and grazing) depending on infested area size and protection status. The calculation of benefits is based on stated preference results (choice experiment; n = 282). For the cost side, we calculate two different invasion scenarios (i) no re-infestation after successfully conducted control measures (optimistic) and (ii) re-infestation twice after conducting control measures occurring within ten years (pessimistic). Minimum costs of eradication measures including a time span of ten years and a social discount rate of 1% result in a total of 3,467,640 € for optimistic scenario and 6,254,932 € for pessimistic invasion scenario, where no success of the first eradication attempt is assumed. Benefits of invasion control in Germany result in a total of 238,063,641 € per year and overassessment-factor corrected in 59,515,910 € per year.

Introduction

Invasive species are considered to be a primary driver of biodiversity loss across the globe (UNEP 2015). Results of invasion experiments indicate that the loss of species may have profound effects on the integrity and functioning of ecosystems (see e.g. Mwangi et al. 2007, van Ruijven et al. 2003, Pfisterer et al. 2004). In addition, invasive species cause public health concerns (EEA 2012, EPPO 2009, Bundesamt für Naturschutz 2015). Currently, about 10 million € are spent annually in Germany for control measures of invasive plant species and about one million € only for health-treatment expenses (Bundesamt für Naturschutz 2015, Reinhardt et al. 2003). *H. mantegazzianum* is originally an endemic species of the sub-alpine zone in the Western Greater Caucasus. It was introduced to Central Europe as an ornamental plant in the 19th century (Pysek 1991, Starfinger and Kowarik 2003, Pergl et al. 2012). Beekeepers established giant hogweed as fodder plant (Westhus et al. 2006). Currently, *H. mantegazzianum* is spread all over Europe (Nehring et al. 2013a, Pergl et al. 2012, Kowarik 2010); and in Germany, *H. mantegazzianum* currently occupies 68 % of grid cells of the national floristic map (NetPhyD and BfN 2013). Field studies in Germany[1] revealed a high variability of cover-abundances; about one third of surveyed stands were dominant[2] with cover-abundances exceeding 50% (Thiele and Otte 2008). *H. mantegazzianum* occurs in a variety of different habitat types, such as roadsides, grasslands, riparian habitats and woodland margins (Thiele and Otte 2006). The highest invasion percentage (18.5%) was found for abandoned grasslands, field and grassland margins and tall-forb stands (Thiele and Otte 2008). Open stands generally prevailed over dominant ones and single stands with sizes between 100 and 1,000 m² occurred most frequently (145 of 233 stands) while stands larger than 1,000 m² were found as minority (32 of 233 stands; Thiele and Otte 2008).

H. mantegazzianum has impacts on biodiversity through competitive displacement of native plant species, particularly at abandoned sites (Thiele et al. 2010), although this seems not a serious threat to protected habitats or regional diversity (Bundesamt für Naturschutz 2015). More attention has to be drawn on the health risk to humans (Bundesamt für Naturschutz 2015, Maguire et al. 2008, Hipkin 1991, Camm et al. 1976, Drever and Hunter 1970). The species is dangerous to humans because it exudes a clear watery sap, containing several chemical agents (e.g. furocoumarins) which sensitise human skin and lead to severe blistering when exposed to sunlight (Drever

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[1]   The quoted field studies were conducted in 2001 at 16 German sites at the Western Low Mountain Ranges (Thiele and Otte 2008).

[2]   The observed limitations indicate only partly dominant stands in the future, namely those representing early habitat invasion and disturbances or land-use change (Thiele and Otte 2008).

and Hunter 1970, Hipkin 1991). Blisters can take up to twenty four hours to appear and the entire reaction can recur for many years (Maguire et al. 2008). In addition to the health hazard, occurrence of *H. mantegazzianum* can limit public accessibility of sites, trails and amenity areas (Tiley and Philp 1994). Besides, it bears the risk to cause ecological damages, e.g. erosion at riverbanks (Pyšek 1991).

With entering into force, the EU-regulation 1143/2014 obligates member states to develop concrete action plans (including timetables for action, description of the measures to be adopted, voluntary actions, codes of good practice) to limit (further) spread of invasive alien species into or within the European Union. After establishing a national list of invasive alien species of concern, member states have eighteen month for comprehensive cost-benefit analysis of pathways and spread and three years for the implementation of one single action plan (European Commission 2014). Appropriate monitoring needs to be planned to reduce density and abundance of invasive species and to keep its impact to an acceptable level (Emerton and Howard 2008, Genovesi and Shine 2003).

However, life-cycle variation between stand types makes it difficult to infer simple management rules (Hüls et al. 2007). Small and open stands of *H. mantegazzianum* may eventually serve as initiators for further spread after land-use changes, whereas dense stands might be stable (Hüls et al. 2007). Westhus et al. (2006) suggest eradication of single plants or initial populations to prevent invasion of the whole area or district. Mowing or grazing are suitable for the management of grasslands and grassland-like fringe habitats (Nielsen et al. 2005, Buttenschøn and Nielsen 2007) to prevent growth and any further development stages. Since the threat of giant hogweed spread towards biodiversity and the health risk for humans are recognized in Germany, there are despite some attempts of local eradication not enough efforts for spatially inclusive and comprehensive management (Nehring et al. 2013b). Increasing habitat fragmentation and climate change will be forcing the spread of giant hogweed if no eradication action will be undertaken (Nehring et al. 2013b).

The aim of this paper is to identify costs of efficient eradication measures and their benefits to society and oppose them within a cost-benefit analysis at national level. The cost dimension covers a wide range of eradication measures with varying sizes depending on infestation share and site status in infested German districts (e.g. grazing for large area types). The benefit dimension is focused on the recreational value in terms of willingness to pay (WTP) for an environment being free of giant hogweed and its risks for humans.

Current control and management of *H. mantegazzianum*

Currently used control methods comprise a variety of manual or mechanical methods, grazing and chemical control (Nielsen et al. 2005). The probability of eradication success increases if control measures are conducted exhaustively and repeatedly within seven to ten years (Holzmann et al. 2014, Nicholas et al. 2005). The prevalence of *H. mantegazzianum* populations along steep embankments, in deep ravines and other

inaccessible places makes manual treatments difficult (Nielsen et al. 2005, Nicholas et al. 2005). The current *preventive* control options are:

- *Avoidance of vegetation gaps,* dense vegetation cover, respectively (Pergl et al. 2007) or presence of the same functional group as invader in the endangered plant community (Longo et al. 2013, Wang et al. 2013, Mwangi et al. 2007, Kahmen et al. 2005, Pokorny et al. 2005),
- *General increase of diversity* in endangered plant communities (Henry et al. 2009, Pfisterer et al. 2004, van Ruijven et al. 2003, Kennedy et al. 2002, Moore et al. 2001, Naeem et al. 2000),
- *Low grade of human disturbance* of the 'natural' ecosystem 'degree of hemeroby' (Mwangi et al. 2007, Machado 2004, Steinhardt et al. 1999, Jalas 1955).

Recent active control options according to the literature are:

- *Manual or mechanical control* such as pulling out the whole plant by hand (EPPO 2009), root cutting or umbel removal by hand (Pyšek et al. 2007a, Pyšek et al. 2007b, Nielsen et al. 2005), cutting the plant from above surface with scythe, mowing or milling machines (Westhus et al. 2006, Nielsen et al. 2005),
- *Grazing by sheep* whereas a time frame of at least 10 years was most effective (Westhus et al. 2006, Nielsen et al. 2005, Andersen and Calov 1996, Williamson and Forbes 1982) Grazing seems meaningful for control of large stands and areas inaccessible for machinery,
- *Chemical control* whereas glyphosate[3] was the most successful herbicide (Nielsen et al. 2007, Nicholas et al. 2005, EPPO 2009). To reduce damage to surrounding vegetation, applications are recommended as spot spraying early in the growing season. However, in the final calculation we suggested chemical treatment with hand-held equipment due to the lowest cost of the alternatives for medium and unprotected areas and its suitability for areas difficult to access.

Revegetation programs after giant hogweed eradication are required to restore the dense vegetation layer and prevent further re-infestations successfully (Noxious Weed Control Program 2015).

Basic assumptions for cost calculation of control measures

Since there are only few data available for giant hogweed management in Germany, cost estimations are based on a nationwide survey of n = 287 German districts (Thie-

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[3] Application of glyphosate beyond agricultural fields, in critical areas or their buffers as well as in areas used for forestry has to be permitted by the nature conservation agency in charge (Paragraph 13 and 17 of German Plant Protection Act).

Table 1. Suggested measures for control of *H. mantegazzianum* depending on area size and protection status.

Area size	Root destruction with shovel	Mechanical cutting with scythe	Mechanical cutting with flail mower	Chemical treatment with hand-held equipment	Chemical treatment with machines	Grazing
Unprotected areas						
Small (up to 100 m²)	X	-	-	X	-	-
Medium (>100–1000 m²)	-	X	-	X	-	X
Large (>1000 m²)	-	-	X	-	X	X
Protected areas (nature reserves)						
Small	X	-	-	-	-	-
Medium	-	X	-	-	-	-
Large	-	X	-	-	-	-

le and Otte 2008) and benefit estimations are based on a choice experiment survey of n = 282 German households. The data from the survey of Thiele and Otte (2008) contain also information on population density. In detail, the questionnaire included questions about the *maximum* spatial extent of single *H. mantegazzianum* stands in three proposed categories (up to 100 m², >100–1,000 m², >1,000 m²) for different habitat types (e.g. roadside or forest margin) and, about occurrences in nature reserves per district or city. Because no conclusion of the total frequency of single stands per district or city was possible, our calculations are based on the assumption of a *minimum* occurrence of the evaluated stands per district or city. This means, the available data indicate, if *at least* one small, medium or large area is infested and if at least one of these areas is protected. A range of possible control measures (manual, mechanical, chemical and grazing) is identified and shown in Table 1. The crosses (X`s) indicate meaningful applications of infestation control measures. Suggested measures are root destruction with shovel (small areas), mechanical cutting with a scythe (medium areas) or flail mower (large areas). Regarding chemical control, hand-held equipment is suitable for small and medium areas, tractors with spraying machines for large areas. Chemical treatment includes the cost of restoration such as seeds, sowing and working hours. For nature reserves, where chemical control is prohibited by law, we suggest root destruction with shovel (small areas) and mechanical cutting with scythe (medium and large areas).

Workload, frequency and effectiveness of treatments are shown in Table 2 (based on Nielsen et al. 2005). It must be considered that chemical control has several restrictions. Grazing is a 'continuous treatment' and includes the workload for fencing and maintenance. Grazing is supposed for medium and large areas, where suitable conditions for livestock farming are given with regard to soil and climatic conditions.

Table 2. Estimated workload and effectiveness of different control methods.

Control methods	Workload	Number of treatments per year	Effectiveness
Root cutting	Estimated time for control: 100 plants/hour	One	high
Mechanical control by scythe	Estimated time for control: 500 plants/hour	Three	low
Mechanical cutting by flail mower	0.5 ha/1,000 plants/hour	Three	low
Chemical control by hand held equipment	Estimated time effort: 100 plants/hour	Two	high
Chemical control by machinery	Estimated time effort: 0.5 ha/1,000 plants/hour	Two	high
Grazing	Fencing: 4-wire electric pasture fencing Maintenance: yearly inspection of the fence, other inspections	'Continuous' treatment	high if conducted regularly

Source: Nielsen et al. (2005) for information on control methods, workload and number of treatments; own estimations on effectiveness of control methods.

Methods

Cost-benefit analysis

Costs and benefits arise because invasive species interfere with the functioning of natural or human-modified ecosystems which yields flows of economically valuable goods and (ecosystem) services (Emerton and Howard 2008). Cost-benefit analysis (CBA) aims to quantify the value of all positive and negative consequences of a project or measure to all members of society in monetary terms. Usually, benefits and costs accrue over extended periods (years). From today's point of view all resources available in the future are less valuable than those available today. Therefore, in CBA future benefits (costs) are discounted relative to present benefits (costs) to obtain their present values. A benefit (cost) that occurs in year t is converted to its present value by dividing it by $(1+d)^t$ where d is the social discount rate.

So if a project has a duration of n years with yearly benefits (B_t) and costs (C_t), the present value of the benefits ($PV(B)$) is

$$PV(B) = \sum_{i=0}^{n} B_t /(1+d)^t$$

and the present value of the costs ($PV(C)$) is

$$PV(C) = \sum_{i=0}^{n} C_t /(1+d)^t$$

If the present value of the benefits exceeds the present value of the costs, the project is valued positively because it leads to a more efficient allocation of society's resources (Boardman et al. 2011). The purpose of this CBA application is to compare different control measures in order to select the most efficient eradication strategy. As they can crucially influence CBA outcome, we also vary social discount rates between 1%

and 3% (Florio and Sirtori 2013, Drupp et al. 2015).Within sensitivity analysis, we consider potential overestimation of empirically investigated benefits (scenario 1). In the second part of sensitivity analysis, we assume a worst case scenario (scenario 2), in which every German district is infested. Benefits of control measures are based on results of a choice experiment (n = 282 respondents) investigated as WTP per person and year (for further details see chapter *Calculation of benefits*). WTP can be regarded as indicator showing if responents are in favor or disfavor for a change from the status quo situation when comparing different alternatives (see Suppl. material 2). WTP results are particularly important where no market proxies or prices are available, as this is usually the case for public goods. In order to meet potential criticism on WTP results in terms of possible overassessment, we suggest several approaches to calibrate our WTP results. In the sensitivity analysis, we recalculate WTP results based on Arrow et al. (1993) proposing a calibration factor of 0.5. It seems impossible to develop a unique calibration factor but we can at least compare our WTP results with other empirical studies which is done in the discussion part of this paper.

Bräuer and Suhr (2005) evaluated 43 empirical studies comparing hypothetical and real WTP for various environmental conservation programs. The term 'hypothetical WTP' describes the fact that WTP is 'just' stated as answer in a survey situation, whereas the term 'real WTP' means the truly payment of the amount stated by respondents. The authors suggest calculating 'switching values' which equal WTP necessary for benefit-cost relations >1. WTP divided by 'switching values' identify the maximum allowed overestimation (Bräuer 2002:264). The maximum allowed overestimation in the study of Bräuer results in a factor of 9.38 and the switching value is 0.08 € (recreation tax as average one-time payment), meaning that if respondents where only willing to pay 8 cent per day during their vacation or stated WTP was overestimated by a factor of 9, the benefits of the described program would still exceed the costs.

Calculation of benefits

For some public goods, such as recreation in uninfested landscapes, there are simply no market proxies for preferences. Many analysts have concluded that in this case, there is no alternative to *asking* a sample of people directly about their preferences (Boardman et al. 2011, Bateman et al. 2002, Hanemann 1994). Questionnaires to elicit such preferences have to be prepared carefully, e.g. the formulation of valuation scenario, including sampling, and data analysis. In some countries, economically relevant benefits from eradication of invasives on direct production may arise (e.g. benefits from commercial crops and livestock), as well as secondary effects on other sectors and times in terms of markets and nutrition (Emerton and Howard 2008). This seems not to be the case for Germany in an economically relevant dimension (Bundesamt für Naturschutz 2015). In our calculations, we focus only on *one* benefit of eradication control: the recreational value in terms of WTP for an environment being free of giant hogweed and its risks for humans (for further benefits see Pergl et al. 2007, Nielsen et al. 2005,

Table 3. Overview of main steps undertaken in the CBA application.

Steps in CBA	Eradication of *H. mantegazzianum* in Germany
Definition of purpose	Compare different control measures for *H. mantegazzianum*; select the most effective strategy for eradication
Definition of perspective	Perspective of benefits: direct use value for population from uninfested landscapes in terms of recreation value; perspective of costs: costs for implementation of eradication measures
Identification of scope and scale	National level based on regional data of districts; costs: based on survey data of n = 287 districts and own calculations; benefits: survey data of n = 282 German households and own calculations
Assumptions for time frame	Costs were calculated over a time period of 10 years; benefits were calculated as one single payment as result of a choice experiment survey for change of the status quo situation ('willingness to pay' for defined eradication measure per household and year)
Assumptions for discount rate	We assume 1-3% discounting (material costs) per year, 1% increase of labor costs and 1% inflation rate per year; additionally we added an excess burden of taxation at the rate of 15%
Definition of baseline scenario	No official intervention (due to unknown/uncertain data); (uncertain) national cost estimations of average 10 million per year (Reinhardt et al. 2003) in discussion section
List and select control options	Root destruction, mechanical cutting, chemical treatment and grazing (for further details see Table 1 and 2)
Select appropriate scenarios	We calculate optimistic and pessimistic scenarios for small, medium, large, non-protected and protected areas. In the pessimistic scenario, we assume twice re-infestation within ten years; in the optimistic scenario, we assume no re-infestation after successfully conducted control measures. Chemical eradication includes costs of renaturation. Because we do not consider all measures to be successfull at once, we calculate 30% additonal costs for monitoring (ten years) and 50% additonal costs for after-treatment (each measure).
Estimate direct costs and benefits	Cost of labor and cost of materials (see Table 5), net present values for suggested control options within the two scenarios (see Table 6); benefits: willingness to pay of 9 Euro for measure per year and person, received from n = 282 German households
Estimate indirect costs and benefits	Due to lacking reliable data base, no precise cost of indirect effects or side effects have been calculated. However, we address this issue. Indirect benefits are the avoided indirect cost of the baseline scenario (which we do not include here).
Compare benefits and costs	B/C ratio was determined by comparing the costs incurred by eradication control with the benefits resulting from eradication as direct use value. The resulting ratio expresses the efficiency of the policy scenario.
Perform sensitivity analysis	We calculate switching values and overestimation factors to address the reliability of WTP results (compare Bräuer and Suhr 2005).

Source: Summary of main CBA steps inspired by Kehlenbeck et al. (2012), Boardman et al. (2011) and Pearce et al. (2006).

Pfisterer et al. 2004, Williamson and Forbes 1982). The calculation of benefits is based upon an empirical face-to-face survey using results of a choice experiment (stated preference method see e.g. Adamowicz et al. 1998, Bateman et al. 2002). The main survey was preceded by qualitative preliminary studies (face-to-face; n = 16), pre-test interviews (as mail survey and face-to-face; n = 57) and pilot study (face-to-face; n = 106). Our qualitative pre-study showed that respondents were aware of non-native plants,

Table 4. Attributes and levels presented to respondents in the main survey.

Attribute	Measure	Level of change (Coding in parenthesis if not directly given; *: *Status Quo*)		
Climate change	Afforestation	Sequestration/emission equivalents of 540 persons* (540)	Sequestration/emission equivalent of 630 persons (630)	Sequestration/emission equivalent of 720 persons (720)
Invasive plants	Removal of invasive plants	Only if harmful and in particular cases (1) *	Large scale removal if harmful or not (2)	
Insect pests and storms	Planting site-adapted trees	Low resistance and resilience (1)	Medium resistance and resilience (2)*	High resistance and resilience (3)
General ecosystem resilience	Changes in the diversity of mycorrhizal fungi	Low resistance and resilience (1)	Medium resistance and resilience (2)*	High resistance and resilience (3)
Price	Income change per year/person	0 €*, 5 €, 10 €, 20 €, 35 €, 50 €, 60 €, 80 € (=coding)		

particularly the giant hogweed was mentioned in several independent interviews (open question format). Thus, we decided to use *H. mantegazzianum* as an indicator for invasive plants. To prepare respondents to the choice experiment task, there was a section in the questionnaire, were some details of the attributes and levels were explained. In the explanation we focused on the risks of giant hogweed to humans in order to justify different types of potential eradication measures. The proposed eradication measures of the attribute 'Invasive plants' are shown in Table 4.

An examplary choice set used in the main survey is shown in Suppl. material 2. Choice cards included a picture of the *H. mantegazzianum*. Within the choice experiment, the following two options were offered to respondents:

- Option 1: removal of invasive plants in particular cases for which negative effects are known, or
- Option 2: large-scale removal of invasive plants even when unclear if they have negative effects or not.

Respondents were asked to state their choice regarding the preferred option. Including the 'price' (mandatory tax payment) of the hypothetical measure each choice option indicates benefits of respondents obtained by the choices. The 'price' for implementation of the proposed measures ranged from 0 to 80 Euro per programmed year. For the Status Quo situation, the cost was always zero. Statistically significant attribute coefficients allow for the calculation of WTP for attribute level changes. In the econometric analysis, WTP can not only be identified for a program or scenario but also for single attributes (details in Suppl. material 2). Average WTP of respondents for control measures is calculated as follows. For attributes linear in parameters, the marginal WTP equals the negative ratio of the respective attribute coefficient c_z and the coefficient of the monetary attribute c_y:

$$\text{WTP} = -\frac{c_z}{c_y}$$

WTP values refer to *one*-level change in the attributes. For respondents protesting the choice experiment, '0' WTP is assumed in order to avoid a bias in favor of higher WTP than stated in the sample. Benefits are opposed as single payment to the costs of a ten year eradication program limiting stands of *H. mantegazzianum*. In the following analysis WTP results for the single attribute 'Invasive plants' are multiplied with the number of households per infested district accounting for nationwide control measures.

Calculation of costs

For the cost side, we calculate two different invasion scenarios for each area size, type and measure: (i) no re-infestation after successfully conducted control measures (optimistic) and (ii) re-infestation twice after conducting control measures within ten years (pessimistic). Both scenarios include the suggested number of treatments per yer (up to three treatments) and measure such as displayed in Table 2. For the cost-benefit analysis, we chose the measures with lowest costs for each area type (protected or not) and size. We calculate with yearly discount rates of 1%, 2% and 3% (Florio and Sirtori 2013, Drupp et al. 2015) and a yearly inflation rate of 1% (e.g. national bonds with expiry date of 2026, corresponding to a 10 year program starting this year, comprising a value of 0,96%, Deutsche Bundesbank 2016). Additionally, 1% increase in labor costs per year is assumed. Both scenarios include 50% additional costs for after-treatment and 30% additional costs for monitoring (30% of labor costs) for each year.

In the following, the procedure of cost calculation is briefly described (see Table 5): As hourly rate of labor costs, 33 € are calculated for all measures. For root destruction measures of *H. mantegazzianum*, additional job training of 5 hours for instruction are considered. One worker is suggested for every small area (up to 100 m²; average 50 m²), ten workers for every medium area (>100–1,000 m²; average 550 m²), and five workers for every large area using machines (>1,000 m²; average 5,500 m²). We considered establishment costs for protective clothing, shovel, scythe and flail mower. Running costs include monitoring (30% of labor costs) and two additional treatments, plus repair costs for machines (e.g. flail mower). Costs for chemical control include two treatments per area, protective clothing (safety glasses, (mouth-) mask, cap, coat and trousers, shoes and gloves), herbicide sprayer for small and medium areas and tractor with spraying machine for large areas, diesel and machine oil, technical inspection and machine check, glyphosate concentrate, restoration (seed mixture, e.g. 70% grass, 30% herbs, 4,000 seeds or 20 g per m²; planting costs, two cuttings per year), plough and seeder. Besides working hours for the described measures, we add five hours for job training for each area. Establishment costs for chemical control include protective clothing, shovel, scythe, machines (tractor with spraying machine, plough and seeder), herbicide sprayer, glyphosate and seeds for restoration. Running costs for chemical control are for diesel and machine oil, tech-

Table 5. Basic assumptions for labor and material cost calculations of giant hogweed eradication measures.

Description of measure	Cost of labor	Cost of materials
Root destruction and mechanical cutting	33 € per hour; additional job training of 5 hours, one treatment and one after-treatment	Protective clothing, shovel, scythe, flail mower, repair cost
Chemical treatment	33 € per hour; additional job training of 5 hours, two treatments, restoration (plough and seeder, planting costs and two cuttings per year)	Protective clothing, machines, herbicide sprayer, diesel and machine oil, technical inspection agency and machine check, machine repair, glyphosate
Grazing	33 € per hour; maintenance of fencing, periodic inspection, daily inspection of animals, moving of animals between fenced area, scrub removal, branch pruning, building of stiles, supplementary cutting outside the fencing with 1,000 hours per year and administration with 15 hours per site and year	Fencing, purchase of animals, shelter, water supply, additional fodder, veterinary inspection and treatment

Source: Based on suggestions from Nielsen et al. (2005) and adjusted to the concrete case of eradication in the infested German districts.

nical inspection and machine check. For calculations of technical agricultural cost (e.g. agricultural machines), we used KTBL software (2015). Grazing is suggested for medium (>100–1,000 m^2) and large (>1,000 m^2) infested areas. Considering sheep having to get used to *H. mantegazzianum*, we included an additional 5 % of total costs for initiation of the measure. We consider establishment costs as those associated with the purchase of animals, fencing in a lifespan of 10 years and shelter. Running costs include maintenance of fencing, periodic inspection, and moving of animals between fenced areas as well as supplementary cutting of *H. mantegazzianum* outside the fenced area, in total, 1,000 hours (33 € per hour) workload per year. Additionally, we calculate 15 hours in administrative costs per area and year for planning, organisation and coordination of the grazing measures. Furthermore, additional fodder for the winter time as well as veterinary inspection with treatment in the case of diseases and water supply are considered. Thirty percent of total costs are suggested for yearly monitoring. Costs of labor are calculated with three people for medium areas (average 550 m^2; by maximum 1,000 m^2) and 5 people for large areas (average 5,500 m^2; by maximum 10,000 m^2). Assuming that the costs are financed by the public authority, we include an excess burden of taxation at the rate of 15% (see Boardman et al. 2011). The excess burden or efficiency cost of taxation recognizes that transfers between consumers, producers and the government are not costless to implement (Boardman et al. 2011). Finally, cost-effectiveness of eradication strategies depend on the length of the period over which they are implemented and observed.

Results

Benefits from control measures

The choice experiment was conducted as a household survey using face-to-face interviews in central Germany. Of the successfully contacted 302 households, 282 respondents completed the choice task (6.6% protest answers). An average interview took 35 minutes. Respondents preferred on average the first control option offered in the choice experiment, the 'removal of invasive plants in particular cases for which negative effects are known'. Interviewees were willing to pay 9 € (p < 0.05) as annual contribution when compared to the more abrasive eradication program. For the 20 respondents (6.6% of interview respondents) protesting the choice experiment, '0' WTP was assumed. Accepting a minimum advantage of invasion control for the German population living in infested districts, in terms of recreation in an environment free of giant hogweed plants, benefits amount to 238,063,641 € per year, average 829,490 € per district. To avoid overestimation, we calculated direct use values as only one single payment *per household*, despite WTP was investiagted as annual payment *per person*. The control of *H. mantegazzianum*, offered in two options, was identified as significant predictor of choice within the econometric model (p < 0.05; Chi^2 < 0.001; R^2 - values 0.19–0.22[4]). For more details on the conducted choice experiment and further results see Rajmis et al. (2009).

Costs of control measures

Table 6 shows the costs in terms of net present values for a ten year eradication program with varying discount rates (1%, 2% and 3%) for each proposed measure. Costs of control measures result in a total of 3.3 milion € for the optimistic invasion scenario and 5.8 million € for the pessimistic invasion scenario[5] at a discount rate of 3%. Calculating a discount rate of 2% costs result in a total of 3.4 milion € for the optimistic invasion scenario and 6 million € for the pessimistic invasion scenario. The 1% discount rate leads to a total cost of 3.5 million € for the optimistic invasion scenario and 6.3 million € for the pessimistic invasion scenario. The costs for the single area types are as follows: for an optimistic scenario in non-protected areas, the lowest cost identified for small areas are root destruction with shovel and result in min. 810 € (max. 855 €), for medium areas the lowest cost resulted in chemical treatment with hand-held equipment including; which amount to min. 5,180 € (max. 5,385 €) and for large areas mechanical cutting with flail mower resulting in min. 44,631 € (max. 45,406 €). For a pessimistic scenario in non-protected and small areas lowest costs were also identified for root destruction with shovel resulting in min. 1,511 €

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[4]   In the calculated model we received R^2 - values between 0.06–0.07 which corresponds to R^2 - values of 0.19–0.22 of linear models (see for details Hensher et al. 2005:338).

[5]   Costs are calculated for available data of n=287 districts (see Thiele and Otte 2008).

Table 6. Costs (net present values) for suggested control measures of infested areas for a time-period of ten years.

Area size	Scenario	Root destruction with shovel			Mechanical cutting with scythe			Mechanical cutting with flail mower			Chemical treatment with hand-held equipment			Chemical treatment with machines			Grazing		
DR[a]		1%	2%	3%	1%	2%	3%	1%	2%	3%	1%	2%	3%	1%	2%	3%	1%	2%	3%
								Unprotected areas											
S[b]	Opt.[c]	**855**	**831**	**810e**	-	-	-	-	-	-	3,023	2,938	2,860	-	-	-	-	-	-
	Pess.[d]	**1,628**	**1,567**	**1,511**	-	-	-	-	-	-	5,636	5,426	5,230	-	-	-	-	-	-
M	Opt.	-	-	-	7,727	7,569	7,424	-	-	-	**5,385**	**5,279**	**5,180**	-	-	-	-	-	-
	Pess.	-	-	-	16,834	16,240	15,685	-	-	-	**11,832**	**11,416**	**11,028**	-	-	-	12,670	12,483	12,310
L	Opt.	-	-	-	-	-	-	**45,406**	**45,003**	**44,631**	-	-	-	129,552	129,462	129,379	-	-	-
	Pess.	-	-	-	-	-	-	119,988	116,011	112,295	-	-	-	385,014	372,692	361,161	**52,850**	**50,981**	**49,251**
								Protected areas (nature reserves)											
S	Opt.	**855**	**831**	**810**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	Pess.	**1,628**	**1,567**	**1,511**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
M	Opt.	-	-	-	**7,727**	**7,569**	**7,424**	-	-	-	-	-	-	-	-	-	-	-	-
	Pess.	-	-	-	**16,834**	**16,240**	**15,685**	-	-	-	-	-	-	-	-	-	-	-	-
L	Opt.	-	-	-	**24,071**	**23,362**	**22,707**	-	-	-	-	-	-	-	-	-	-	-	-
	Pess.	-	-	-	**43,310**	**41,677**	**40,157**	-	-	-	-	-	-	-	-	-	-	-	-

a Discount rate

b S = small areas, M = medium areas, L = large areas

c optimistic scenario: no re-infestation after successfully conducted control measures

d pessimistic scenario: twice re-infestation within ten years

e cheapest option for each area size in bold

(max. 1,628 €), for medium areas chemical treatment with hand-held equipment including restoration at a min. price of 11,028 € (max. 11,832 €), for large areas the lowest cost were reached by grazing with a min. of 49,251 € (max. 52,850 €). For an optimistic scenario in small protected areas, lowest costs are also reached by root destruction with shovel and amount to min. 810 € (max. 855 €), for medium areas the lowest cost result for mechanical cutting with scythe amounting to min. 7,424 € (max. 7,727 €). For large areas mechanical cutting with scythe is suggested which amounts to min. cost of 22,707 € (max. 24,071 €). For the pessimistic scenario in protected and small areas the lowest cost result in root destruction with shovel and amount to min. costs of 1,511 € (max. 1,628 €) for medium areas mechanical cutting with scythe resulting in min. cost of 15,658 € (max. 16,834 €), and for large areas (mechanical cutting with scythe as well) in min. cost of 40,157 € (max. 43,310 €). Some details of scenario calculations are shown in the supplementary material (Suppl. material 1).

Benefit-cost relation of control measures and sensitivity analysis

We chose the measures with lowest costs for each area type (protected or not) and size (small, medium and large) for the final calculation. The lowest cost measures are root destruction with shovel in small areas (optimistic and pessimistic scenario), chemical treatment with hand-held equipment in medium areas (optimistic and pessimistic scenario), mechanical cutting with flail mower in large areas (optimistic scenario), and grazing for large areas in the pessimistic scenario. Root destruction with shovel and mechanical cutting with scythe are due to legal constraints the only options for protected areas. The benefit-cost relation of German districts for control measures of *H. mantegazzianum* lies between 69:1 (discount rate of 1%) and 72:1 (discount rate of 3%) for optimistic scenario and 38:1 (discount rate of 1%) and 41:1 (discount rate of 3%) for pessimistic scenario calculations for each area size. Results indicate that every euro of calculated costs can be opposed to averagely 55 € of benefits (discount rates between 1% and 3%). To give consideration to the earlier mentioned concerns of potential overestimations, we calculate the maximum allowed overestimation (Bräuer 2002, Bräuer and Suhr 2005).

Switching values range between 0,02 and 0,03 € (average 2,5 cent) in Scenario 1 and between 0,24 and 0,30 € (average 26 cent) in Scenario 2. This is the amount necessary to result in a benefit-cost relation >1. Calculating the net-benefit of measure implementation (WTP/switching value), a factor of 448 results for optimistic and 299 for pessimistic scenario calculations. This means, if our empirically investigated results would be overestimated by factors between 299 (pessimistic scenario) and 448 (optimistic scenario), 'necessary' real WTP would be still the amount of the switching values (0.03 € and 0.02 €), hence high enough to keep the benefit-cost relation positive.

Since the utilized source of data (Thiele and Otte 2008) may not represent the current state of invasion status in Germany, we also provide a sensitivity analysis in terms of infestation assumptions (Scenario 2). We assume that every German district is

Table 7. Scenario 1. Benefit-cost relation of infested German districts (N= 287) based on data from Thiele and Otte (2008) and overestimation factor of WTP results.

Results	Optimistic scenario			Pessimistic scenario		
Discount rate (DR)	1%	2%	3%	1%	2%	3%
Average benefit-cost relation of German districts	68.65	70.27	71.83	38.06	39.5	40.96
Switching value (in €)	0.02			0.03		
Maximum allowed overestimation (WTP/switching value)	448			299		

Table 8. Scenario 2. Benefit-cost relation of worst case scenario: every German district (N= 440) infested and overestimation factor of WTP results.

Results	Pessimistic scenario		
Discount rate (DR)	1%	2%	3%
Average benefit-cost relation of German districts	3.9	4.0	4.2
Switching value (in €)	0.30	0.25	0.24
Maximum allowed overestimation (WTP/switching value)	29.9	35.8	37.3

infested with at least one small, one medium and one large area and calculate the pessimistic infestation scenario without chemical treatment (due to possible infested nature reserves or other sensitive landscape areas), control cost amount to 57 and 61 million euro for ten years of treatment. As this is a worst case scenario, we assume the most expensive cost for each measure and area size here. This cost estimate is well within the range of similar calculations for other countries (Reinhardt et al. 2003, Sampson 1994, van Wilgen et al. 2004). The cost estimates of Gren et al. (2009) are somewhat higher than our results (see below). Opposed to the benefits of our survey with one single payment per German household in the infested districts, this results in a benefit cost-relation of 4:1. The maximum allowed overestimation ranges between 30 and 37 and is thus lower as in scenario 1 (between 38 and 72). This result seems reasoned due to average benefit-cost relation in scenario 2 is ten times lower (e.g. 4 versus 40 using DR of 2%) comparing pessimistic scenario calculations. Switching values range between 0,24 and 0,30 €, meaning that even if WTP would have been be overestimated 37 times, we would still have a benefit-cost relation >1. As mentioned earlier, the NOAA Panel suggests calibrating empirical WTP results by a factor of 0.5 (Arrow et al. 1993). If we recalculate scenario 1 with halved WTP, the allowed overestimation is 50% reduced and results in a factor of 150 considering pessimistic assumptions and a factor of 225 for optimistic assumptions. For scenario 2, the allowed overestimation with halved WTP has still a factor of 17, meaning that even if WTP would have been stated 17 times higher then conceived by respondents, we would still have a benefit-cost relation >1.

Discussion

Our cost-benefit-analysis clearly shows that control measures limiting *H. mantagez-zianum* in infested German districts are efficient from an economic point of view. The most promising measures from the control perspective are root cutting and chemical treatment by hand-held equipment or machinery, although chemical control includes two treatments and revegetation. Root cutting is an important control measure for protected areas. These findings are in line with experiences from Latvia (Olukalns 2007) and United Kingdom (Sampson 1994; see below). If the suggested measures are implemented successfully including after-treatment, the probability of re-infestation is low and the measures may have a very positive benefit-cost ratio in the long term as well. Reducing monitoring frequency increases yearly costs up to 162% (Breukers et al. 2008). *Net present values* of control measures range between 810 € for root destruction with shovel (DR of 3%) and 385 thousand € (DR of 1%) for chemical treatment with machines for a time period of ten years depending on area size and type of treatment or 4 and 8 cent per capita in Germany for all necessary control measures. The cost have to be recognized as lower limit of minimum necessary eradication cost. The identified benefits of our survey are approximately 9 Euro per capita in Germany, resulting in a benefit cost-relation of 225:1 (lowest cost within 1% DR) and 113:1 (highest cost within 3% DR). If we consider just one person per German household willing to pay, the benefit-cost relation lies between 113:1 and 56:1. By the way, this is again equivalent to a 50% reduction of our WTP results, which is suggested by the NOAA-panel as factor of calibration (Arrow et al. 1993). Especially the cost estimates are somewhat lower in comparison to the calculations for other countries. The benefit cost-results are in the dimension of van Wilgen et al. (2004).

Unfortunately, there are very few studies about costs and benefits of invasion control.

In the following, available *cost estimations* on invasion control scenarios are presented and – if possible – compared to the findings of our cost-benefit analysis. The only economic assessment of giant hogweed eradication cost especially for Germany can be found in Reinhardt et al. (2003). The authors estimate annual control cost of giant hogweed in Germany amounting to average 12 million euro, including 1 million euro for medical treatment of injured humans, 1.2 million euro for measures in nature reserves, 2.1 million for measures in road management, 2.4 million for measures in municipal management and 5.6 million euro for district management measures (no further differentiation of costs). If we assume a minimum infestation of each area size and type in the surveyed districts (n = 287) and add the (uncertain) current cost of about 12 million spent in Germany for yearly giant hogweed eradication (Reinhardt et al. 2003) for a ten year eradication program in our analysis and compare the benefits for only one year to the cost results, the benefit-cost relation ist still 2:1. The resulting values still demonstrate an environmental improvement and welfare improvement for the society even if we look at more costly invasion scenarios.

Gren et al. (2009) estimated the total costs of 13 invasive species in Sweden. All species are subject to control by Swedish public authorities, and estimates for most

invasive species include either damage cost or actual control cost. The results indicate a total annual cost between approximately 153 million € and 479 million €, which correspond to 17 euro and 53 € per capita in Sweden. The total annual cost for giant hogweed control range from 38 thousand € to 47 thousand € (0.004 to 0.005 € per capita in Sweden). In our study, annual control cost per measure type and size range between 81 € and 39 thousand € (up to 0.0005 € per capita in Germany), which is in the lower limit of Gren et al. (2009).

Sampson (1994) estimated control cost of giant hogweed in UK for 150 infested sites identified by a postal survey conducted in 1990. The three main adopted control strategies were: cutting plus glyphosate, cutting alone or glyphosate alone. Overall expenditure of control costs for 1989 range between approximately 148 € and 42.630 € (historical exchange values from 2000; 1989 not available). These results are in the same dimension of our calculations for control measures of n = 396 infested sites ranging between 810 € and 385 thousand €, that is 1 € to 284 € per site in UK versus 2 € to 972 € per site in Germany.

In the study of van Wilgen et al. (2004), costs and benefits from biological control of six invasive alien weed species (e.g. red sesbania and jointed cactus) in South Africa are compared. Red sesbania replaces indigenous riverine and wetland species, especially the seeds are poisonous and lethal to mammals, birds and reptiles. The jointed cactus competes with indigenous species as well. Dense infestations reduce the grazing potential (up to 90%) and hence the value of the agricultural land. The authors calculate benefits as economic losses in water use, biodiversity, and preservation of the value of agricultural land. Benefit-cost ratios range from 8:1 for the red sesbania (*Sesbania punicea*) to 709:1 for the jointed cactus (*Opuntia aurantiaca Lindley*). The sensitivity analysis showes that the returns on investment in biological control generally remain positive with some variations between species (van Wilgen et al. 2004). In our study, we did neither include benefits as economic losses from values of agricultural land nor biodiversity deducting that our benefit estimates are rather underestimated than overestimated.

Conclusions

The studies mentioned above result in positive benefit-cost outcomes indicating that invasion control is sensefull from an economic point of view: the control activities are economically efficient and they have in large part positive effects on biodiversity, water use, human and animal health. This might be a more convincing argument for policymakers than nature conservation as good achievement. Since the EU regulation 1143/2014 entered into force, member states are anyway obligated to conduct cost-benefit analysis to identify cost efficient control measures. However, we quantified just *one* benefit of giant hogweed control in terms of direct use value for recreation; there might be much more benefits which we did not include. The true benefits of giant hogweed control to society might be much higher. Compared to the studies in this

discussion section, we conclude that our results might reveal only the lower limit of control costs. Based on our findings and the review literature, we suggest for future control programs:

- to support research on prevention methods in different ecosystems e.g. biodiversity conservation at landscape level as invasion insurance
- incorporate non-market values such as loss in aesthetic values, recreation or other ecosystem services as benefit of control programs;
- to plan control measures at an adequate spatial scale taking into account potential re-infestations.

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Supplementary material 1

Details of scenario cost calculations
Authors: Sandra Rajmis, Jan Thiele, Rainer Marggraf
Data type: Adobe PDF file
Explanation note: In Suppl. material 1 we present some cost calculation details of different infested area types and sizes.

Supplementary material 2

Exemplary choice set and details of scenario benefit calculations
Authors: Sandra Rajmis, Jan Thiele, Rainer Marggraf
Data type: Adobe PDF file
Explanation note: In Suppl. material 2 we present an exemplary choice set used in the choice experiment and some details of econometric analysis.

Is it worth the effort? Spread and management success of invasive alien plant species in a Central European National Park

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Abstract

The management of invasive alien species (IAS) in protected areas has become increasingly important in recent years. In this study, we analyse IAS management in the bilateral National Park Thayatal-Podyjí at the Austrian-Czech border. Based on two surveys from the years 2001 and 2010 and on annual management data from 2001-2010 we analyse changes in distribution and the efficiency of IAS management of three invasive alien plants (*Fallopia × bohemica*, *Impatiens glandulifera*, *Robinia pseudoacacia*).

In 2010, the three study species had invaded 161 ha (2%) of the study area. Despite a decade of management, *F. × bohemica* has become widespread, whereas *I. glandulifera* distribution has decreased strongly. The most widespread species, *R. pseudoacacia*, has declined substantially in cover, but the area invaded has increased.

From 2001 to 2010, annual management effort declined by about half. Management effort per hectare and decade was highest for *F. × bohemica* (2,657 hours), followed by *R. pseudoacacia* (1,473 hours) and *I. glandulifera* (270 hours). Management effort for achieving the same amount of reduction in population size and cover was highest for *R. pseudoacacia*, followed by *F. × bohemica* and *I. glandulifera*.

We conclude that substantial effort and resources are necessary to successfully manage the study species and have to be provided over prolonged time periods, and thus continued management of these species is recommended. We highly recommend a systematic approach for monitoring the efficiency of IAS management projects in protected areas.

Keywords

Introduction

The number of alien species are rapidly increasing worldwide, causing large and increasingly detrimental impacts on biodiversity and human well-being (Vilà et al. 2010, 2011). Protected areas play a pivotal role for nature conservation, and this is particularly the case in Europe as this continent is characterized by a long history of human impact on ecosystems. Recent studies have shown that protected areas are vulnerable to the spread of alien species (Foxcroft et al. 2013). Although only a small fraction of these become invasive, i.e. cause negative impacts on the environment (Blackburn et al. 2014) by outcompeting native species, changing ecosystem functioning and processes, or modifying species' interactions (Hulme et al. 2012, Pyšek et al. 2012), their impacts on the environment may be substantial (Foxcroft et al. 2013). Thus, the need for managing IAS in protected areas is high (Pyšek et al. 2013, Schmiedel et al. 2015, 2015, Sitzia et al. 2016). To date, however, analyses of IAS management effort and success in protected areas are scarce (see Foxcroft et al. 2013 and references therein), but urgently needed for informing evidence-based conservation.

In this study, we analyse management success of three invasive alien plant species (*Fallopia × bohemica*, *Impatiens glandulifera*, *Robinia pseudoacacia*) in the National Park Thayatal-Podyjí, over the period of a decade. These species had previously been identified by the management authority of being likely the most detrimental alien species in this protected area (C. Übl pers. comm.). We use spatially explicit distribution data from the years 2001 and 2010, and data of management effort and allocation from 2001 to 2010, to ask the following questions: (1) What is the change in distribution (extent and cover) of the study species? (2) What are the differences in trajectories of distribution change of managed and unmanaged populations, and between species? (3) How successful and efficient is the management? Finally, we discuss the implications of our study for IAS management in protected areas.

Material and methods

Study area

The National Park Thayatal-Podyjí is located in the Bohemian Massif on both sides of the Austrian-Czech Border (Fig. 1). It covers an area of 7,630 ha (Austria: 1,330 ha, Czech Republic: 6,300 ha) and it was established in 2000 (Austria) and 1991 (Czech Republic) to protect a heavily forested steep river valley. The Thaya River is the main water course in the National Park, whose narrow, meandering valley forms the border between Austria and the Czech Republic. The climate of the region is temperate, with cool winters and warm summers, average annual temperature ranging between c. 9.0°C in the lowest parts to 7.5°C in the highest parts, and annual precipitation of c. 500–600 mm (1961-90) (Chytry and Grulich 1993, ZAMG 2001). The bedrock consists of granite and gneiss which are partly covered by loess (Chytrý et al. 1999, Nagl 2002).
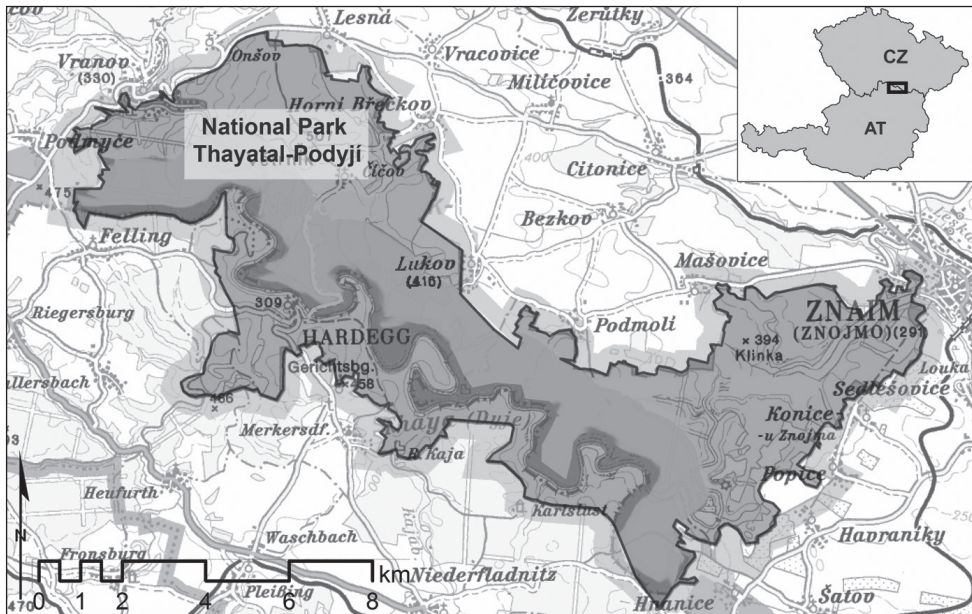
Figure 1. The National Park Thayatal-Podyjí (in grey) located at the Austrian-Czech border. The river Thaya/Dyje forms the border between the two countries. The insert shows the location of the National Park in Austria/Czech Republic.

Besides widespread zonal broadleaved-forests (mostly dominated by *Carpinus betulus*, *Fagus sylvatica*, *Quercus petraea*, *Q. robur*), rare and threatened habitats such as dry forests on steep slopes, floodplain-forests along the Thaya River, nutrient poor mesic and dry grasslands, contribute to the high value for nature conservation. After World War II, the Iron Curtain prevented land use intensification and ensured the conservation of this highly valuable landscape.

Study species

This study focuses on three study species which exert substantial negative impacts on native biota in Central Europe (DAISIE 2013).

Fallopia × bohemica (Polygonaceae) is a hybrid of *F. japonica* and *F. sachalinensis*, both native to East-Asia. The parental species were introduced into Europe in the 19th century for ornamental purposes, with the hybrid most probably originating in Europe (Bailey 1988). In the last decades, *F. × bohemica* spread vigorously (Mandák et al. 2004, Pauková 2013). It can establish dense populations, particularly in riparian habitats (Hejda et al. 2009, Pauková 2013), which change the vegetation structure and outcompete native species (Aguilera et al. 2010, Gerber et al. 2008, Stoll et al. 2012). Rhizome fragments of *Fallopia*-taxa are easily dispersed by running water during floods

(Beerling et al. 1994, Sheppard et al. 2006, Bímová et al. 2004) or contaminated soil. Due to the high regenerative ability of *Fallopia*-taxa (Bímová et al. 2004, Pauková 2013), management presents a substantial challenge (Rudenko and Hulting 2010, Delbart et al. 2012).

Impatiens glandulifera (Balsaminaceae) is native in the Himalayas. It was introduced as an ornamental plant to England in 1839. Despite its early introduction, only in the last few decades has it become one of the most widespread alien species in Europe (Beerling and Perrins 1993, Essl and Rabitsch 2006). It favours moist and nutrient-rich riparian habitats (Pyšek and Prach 1995). As an annual plant species it is reliant on seed dispersal, again often by water and contaminated soils (e.g. Dawson and Holland 1999). The impacts of *I. glandulifera* on native flora and fauna are contested (Drescher and Prots 1996, Tickner et al. 2001, Hejda and Pyšek 2006, Tanner et al. 2014). For instance, it has been shown that it may monopolize pollinators, thus likely lowering seed production of native plants (Chittka and Schürkens 2001), and that it exerts negative impacts on invertebrate communities (e.g. bugs, beetles) (Tanner et al. 2013). On the other hand, pollinators use its flowers as important nectar resource (Bartomeus et al. 2010).

Robinia pseudoacacia (Fabaceae) is a pioneer tree native to south-eastern North America. It has been extensively planted in Central Europe since the end of the 18th century, and has become widely naturalized in warm lowlands. Once established, *R. pseudoacacia* spreads efficiently via root suckers (e.g. Kowarik 2010, Cierjacks et al. 2013). As a consequence, the further local spread of *R. pseudoacacia* from invasion foci is difficult to control. *Robinia pseudoacacia* prefers nutrient poor dry and semi-dry habitats (e.g. dry grasslands and dry forests; Kleinbauer et al. 2010). Due to its symbiotic nitrogen fixing nodule bacteria, its encroachment into nutrient poor habitats severely increases productivity and modifies nutrient cycling (Rice et al. 2004, Kowarik 2010, Cierjacks et al. 2013).

Distribution mapping and data on management effort

The distribution of the study species within the Austrian part of the National Park Thayatal-Podyjí was first surveyed in 2000 (Essl and Hauser 2002). Distribution maps of the study species and aerial photos of the Natura 2000-site "Thayatal bei Hardegg", which includes the National Park Thayatal-Podyjí, were used for the survey. All populations of the study species were assigned to size classes (0–10 m^2; 10–100 m^2; 100–1,000 m^2; 1,000–10,000 m^2; >10,000 m^2) and population density was assessed according to the cover-abundance-scale of Braun-Blanquet (1964). For each population, the management options were assessed according to the local situation (i.e. population size, accessibility, the likelihood of further spread), and taking into account the nature conservation value of the invaded habitat; this assessment served as basis for subsequent management. In the second survey, all populations were resurveyed in summer 2010. Supplementary distribution data of the study species for the Czech part of the National Park were provided (R. Stejskal pers. comm.), which date from 2007–2013 (*F.* × *bohemica*, *I. glandulifera*), and 2003 (*R. pseudoacacia*).

We obtained data on management effort for the study species from the National Park Thayatal-Podyjí management authority for the years 2001, 2008, 2009 and 2010. Information provided included monthly working hours (differentiated into project-coordination and field-work) for each of the study species. Equipment and the amount of time of its usage were also recorded. For the period 2002–2007, data on management effort were incomplete. Thus, we used the available data from the National Park Authority, which showed that overall management effort changed gradually from year to year. We then interpolated data by assuming a linear change in management effort per study species from 2002–2007 based on data for the years for which complete information was available. We do acknowledge however, that this interpolation of management effort may introduce some uncertainties in the overall calculation of species specific management efforts.

Data analyses

Distribution data of the study species were entered into a database and a Geographical Information system (ESRI ArcView). For the Austrian part of the National Park, information on cover and size of the populations were available for both surveys. For each species, we analysed the changes in the spatial extent of polygons (Fig. 2) and in average cover values. For the latter, the cover-abundance data (according to Braun-Blanquet 1932, 1964) were converted into numerical cover values (Van der Maarel 1979) (r = 0.005%; + = 0.1%; 1 = 5%; 2 = 17.5%; 3 = 37.5%; 4 = 67.5%; 5 = 87.5%). We note that this approach is limited in its analytical power (as values are transformed into a standard numerical value for each Braun-Blanquet class), but nevertheless, it is the most appropriate method for analysing plot data (Van der Maarel 2007).

In addition, we calculated a combined index which gives equal weight to population size and cover. This "Area-Density-Index" ("ADI") was calculated by multiplying the extent of the polygons (in m^2) by the average plant cover (in %) for each population. We then calculated the ADI for each species by summing up all population-level ADIs; this was done separately for both surveys.

Management effort for the three study species between the two surveys (2001–2010) was calculated based on data from the National Park Authority. Again, these data were only available for the Austrian part of the National Park, and analyses thus excludes the Czech National Park-section. Management data provided include the number of working hours spent and the allocation of these hours to the study species per year. We calculated overall management effort for each study species by summing the annual working hours from 2001–2010. As management was done mostly manually and the necessary machinery was already available in the National Park Management Authority, we discard additional costs for machinery and equipment. Therefore, our calculation of management effort is conservative.

To analyse the efficiency of management, we used as a common metric the reduction of the ADI between both surveys. Only managed populations were considered,

Figure 2. Changes in distribution of *Fallopia* × *bohemica*, *Impatiens glandulifera* and *Robinia pseudoacacia* between the first (2001) (**A**) and the second survey (2010) (**B**) in the surroundings of the village Hardegg. This section of the National Park Thayatal-Podyjí is the most heavily invaded.

whereas populations which were not managed were excluded. Finally, we calculated the number of working hours necessary for a reduction of the ADI by 10 points between both surveys; again, this was done separately for each study species.

Results

Current distribution of the study species

The study species differ strongly in their abundance and distribution in the study area (Suppl. material 3, 4). *Fallopia* × *bohemica* was by far the rarest species, with a total of 21 populations in 2010. Of these, eight are located in Austria (total area: 2,700 m²) and the remainder (total area: 300 m²) in the Czech part of the National Park. Thirteen populations invade ruderal habitats and tall herb vegetation near settlements, whereas eight populations are situated in riparian habitats (mostly in tall herb vegetation). Five populations are dense with cover values >50%. However, as the spatial extent of the populations is generally low, the ADI is also low.

In 2010, *I. glandulifera* was uncommon as well. In total, 90 populations with an extent of 1.2 ha were recorded whereby most of these (78) were located in the Czech

part of the National Park. All populations were situated in near-natural habitats (tall herb vegetation, riparian forests) adjacent to the Thaya River. Populations in Austria are larger (up to 2,500 m^2), whereas the largest population in the Czech part covers only 300 m^2. Population cover values were mostly low (0.1–5%), and therefore the ADI is low.

In 2010, *R. pseudoacacia* was the most widespread study species. A total of 229 populations were recorded, of which 204 were situated in the Czech Republic. *Robinia pseudoacacia* invades 146 ha, whereby the Austrian part contributes only 13.8 ha. Most populations are found in dry oak-forests, a few populations have been recorded in dry grasslands. Population sizes vary substantially, with 41 populations being larger than one ha. Plant cover in the Austrian part of the National Park varied widely between low (5%) and high (three populations with >50%) values. For the Czech Republic, no information on plant cover was available. Managed populations were girdled, but due to re-sprouting, *R. pseudoacacia* still remains present in the herb and shrub layers.

Changes in abundance between 2001 and 2010

Numbers and extent of *F. × bohemica* populations increased between both surveys. Seven populations were found in 2001 and eight in 2010; four of these were newly established ones, while three populations were eradicated between both surveys (Suppl. material 1, 2). The ADI of the only population being present in both surveys decreased due to a major decline in plant cover, while the ADI of the other populations combined remained almost unchanged (Fig. 3). In total, *F. × bohemica* populations increased by c. 2,000 m^2 in size, and populations being present in both surveys increased by c. 300 m^2 or 54% (Suppl. material 3).

In contrast, numbers and extent of *I. glandulifera* population decreased (Suppl. material 1, 2). Ten populations were eradicated, whereas five new populations were found in 2010. Consequently, the ADI declined strongly as well (Fig. 3). This is mostly due to a large decrease in size, as the total area invaded by *I. glandulifera* (4.5 ha in 2001) decreased by 77% (-3.5 ha) in 2010. Densities of populations recorded in both surveys remained largely unchanged at a low level (Suppl. material 3, Suppl. material 1, 2).

Finally, *R. pseudoacacia* populations showed opposing trends. Whereas the number of populations and their extent increased, densities declined moderately. Of the 21 populations recorded in 2001, three were eradicated and six were newly recorded in 2010 (Suppl. material 3). The trend in ADI differs between managed and unmanaged populations with strongest declines for managed populations (Fig. 4).

Management measures, effort and efficiency

In total 6.4 ha (~0.1% of the National Park area) invaded by one of the three study species were managed between 2001 and 2010 (Suppl. material 3). Of these, 0.06 ha
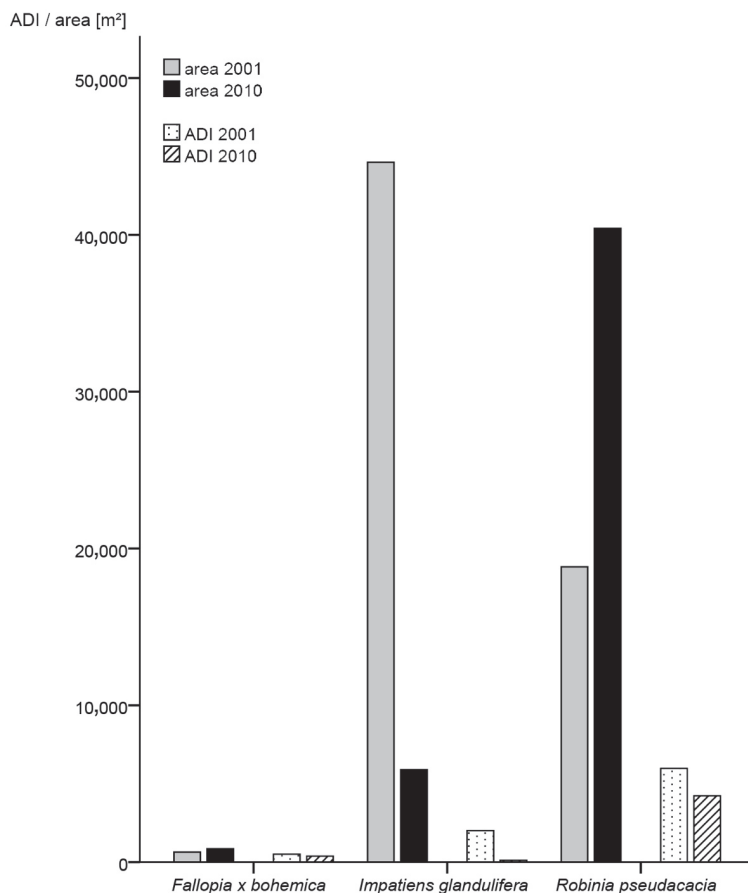
Figure 3. Total invaded area [m²] and Area-Density-Index (ADI) of *Fallopia* × *bohemica*, *Impatiens glandulifera* and *Robinia pseudoacacia* in the first (2001) and second survey (2010) in the Austrian part of the National Park Thayatal-Podyjí.

(1%) were invaded by *F.* × *bohemica*, 4.5 ha (77%) by *I. glandulifera* and 1.8 ha (28%) by *R. pseudoacacia*. Management of the study species varied in terms of methods and effort applied. *Fallopia* × *bohemica* populations were managed in varying ways and from different parties (road maintenance department, commune, National Park Management Authority). One population was not managed, one was mown once a year, and two were mown and had herbicides (Roundup, with the active ingredient Glyphosate) applied several times via stem-injections. Of the three eradicated populations, one was mown several times a year, the second was dug up, and one small population was removed manually by continuous hand-pulling. All populations of *I. glandulifera* were managed since 2001. Management measures include mowing once before the onset of flowering, small populations were managed by hand-pulling. *Robinia pseudoacacia* was managed by girdling at breast-height, leaving a small section
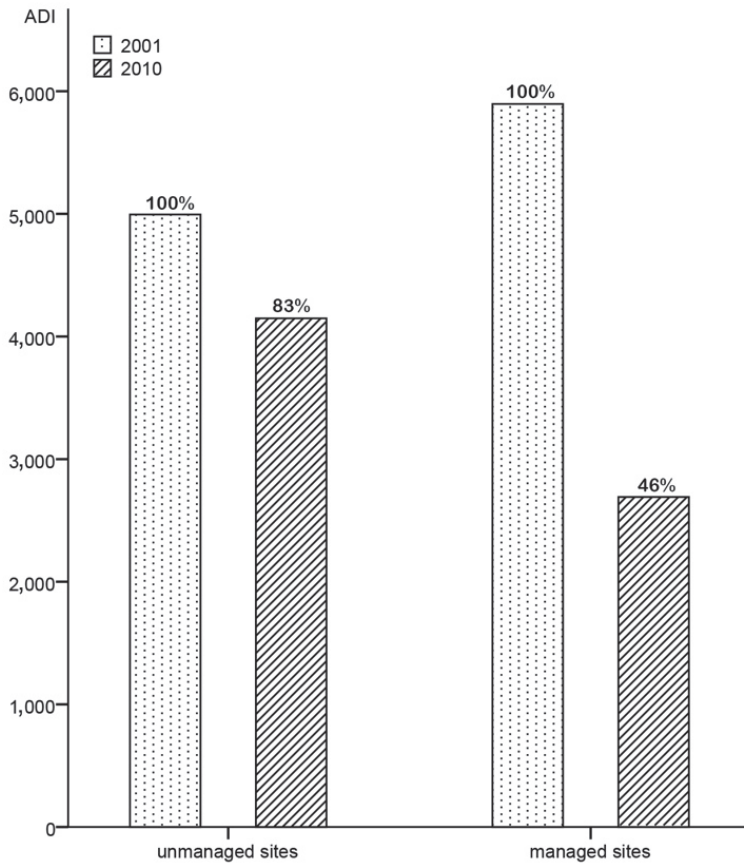
Figure 4. Change of the Area-Density-Index (ADI) of unmanaged and managed populations of *Robinia pseudoacacia* in the Austrian part of the National Park Thayatal-Podyjí between the first (2001) and second survey (2010).

of the bark, which was removed in the following year. Root suckers were cut several times in subsequent years.

A total of 4,150 working hours were spent for the management of the study species during the ten-year period. On average, 16 hours were spent annually for managing *F. × bohemica*, 122 hours for *I. glandulifera* and 277 hours for *R. pseudoacacia*. Annual management effort declined by about half between the two surveys. Accordingly, in the first five-year period (2001–2005) 503 working hours were spent on average annually for managing the study species, whereas in the second five-year period (2006–2010) this value declined to 328 hours (Table 1). Management effort per hectare and decade was highest for *F. × bohemica* (2,657 hours), followed by *R. pseudoacacia* (1,473 hours) and *I. glandulifera* (270 hours). Management effort required for achieving the same reduction in plant cover and extent (calculated as reduction by 10 ADI-points), was highest for *R. pseudoacacia* (16 hours), followed by *F. × bohemica* (12 hours) and *I. glandulifera* (6 hours) (Table 2).

Table 1. Estimated annual working hours used for managing *Fallopia × bohemica*, *Impatiens glandulifera* and *Robinia pseudoacacia* in the Austrian part of the National Park Thayatal-Podyjí. Note that incomplete data on management effort for the years 2002-07 were interpolated (see main text for details).

	F. × bohemica hours	*I. glandulifera* hours	*R. pseudoacacia* hours	**TOTAL**
2001	21	163	372	**557**
2002	20	155	354	**530**
2003	19	147	336	**503**
2004	18	140	318	**476**
2005	17	132	300	**449**
2006	16	124	282	**422**
2007	15	116	264	**395**
2008	12	117	239	**368**
2009	11	70	170	**250**
2010	9	55	139	**203**
TOTAL	**159**	**1217**	**2774**	**4151**

Table 2. Estimated management effort and management efficiency for *Fallopia × bohemica*, *Impatiens glandulifera* and *Robinia pseudoacacia* in the Austrian part of the National Park Thayatal-Podyjí. Given are the total change in ADI (=Area-Density-Index, calculated by multiplying the extent of populations (in m^2) by the average plant cover (in %) for each species) of managed populations between the first (2001) and second survey (2010), the total number of working hours spent for management per study species, the total extent of managed populations at the start of management (in 2001), the working hours per decade and ha, and the numbers of working hours necessary to achieve a decrease of 10 ADI-points between both surveys. Note that incomplete data on management effort for the years 2002-07 were interpolated (see main text for details).

	F. × bohemica	*I. glandulifera*	*R. pseudoacacia*
ADI - change 2001/2010	-129	-1889	-1750
Working hours/10y	159	1217	2774
Area managed since 2001 (ha)	0.06	4.50	1.88
Hours/ha/10y	**2657**	**270**	**1473**
Hours/10 pts ADI-decrease/10y	**12**	**6**	**16**

Discussion

Changes of distribution and cover

Although the study species are amongst the most abundant IAS in Central Europe (Drescher and Prots 1996, Mandák et al. 2004, Chytry et al. 2008, Cierjacks et al. 2013), they are still rather uncommon in the study area. This moderate level of invasion can likely be attributed to a low level of anthropogenic disturbance, low regional human population density and low propagule pressure (Essl et al. 2012) due to the location of the study region along the former Iron Curtain.

The changes in population sizes and densities of the three study species showed opposing trends between both surveys. Most populations of *I. glandulifera* were eradicated by 2010 and the ones still existing had strongly declined in ADI. As an annual herb *I. glandulifera* is dependent on reproduction by seeds, thus management measures can result in rapid reduction in population size (e.g. Dawson and Holland 1999).

Fallopia × *bohemica* mainly regenerates vegetatively via rhizomes. As a result, management of *F.* × *bohemica* is difficult, and the species increased its distribution in the national park. Even though Bímová et al. (2004) report that *F.* × *bohemica* is mostly found in river bank vegetation in the Czech Republic, the majority of records in the study area have been made in anthropogenic habitats away from watercourses. However, populations growing adjacent to watercourses likely function as source for downstream spread, as has already occurred in two cases between both surveys. Several populations recorded in 2010 were located downstream from the populations found in 2001, suggesting that propagule dispersal by running water played a role (Dawson and Holland 1999).

Similar to *F.* × *bohemica*, *R. pseudoacacia* extended its distribution, although populations show a moderate decrease of ADI, because of strong declines in population density. After disturbance (i.e. logging) it may regenerate by root suckers, and may therefore even increase the size of the invaded area. Hence, complete eradication of managed populations was not feasible during the ten-year period. In the study area, *R. pseudoacacia* occurs in deciduous dry forests and semi-dry grasslands. Populations in forests changed little in size between both surveys, whereas populations in grasslands did. Terwei et al. (2013) and Motta et al. (2009) identified light availability and disturbances as crucial factors facilitating *R. pseudoacacia* regeneration. This is consistent with the slower spread of populations in forests (compared to open grasslands) in the National Park Thayatal-Podyjí.

The role of management on distribution changes

Management of *F.* × *bohemica* reduced population size, but only chemical management and mowing several times a year was effective. In contrast, populations mown once a year increased in size and cover remained high. Increases in population extent were even higher than in unmanaged populations, indicating that mowing once a year is not a suitable management measure (Catford et al. 2012, Delbart et al. 2012).

The management of *I. glandulifera* was highly successful as nearly all populations in the National Park Thayatal-Podyjí were eradicated within a decade. Only few populations in the eastern part of the study area persisted, possibly due to propagule pressure from populations further upstream the Thaya River. Such newly established populations were detected soon while still small (often consisting only of few individuals) and were rapidly included in the management. The existence and scale of impacts of *I. glandulifera* invasions are, however, discussed controversially (e.g. Prowse 2001, Hulme and Bremner 2005, Bartomeus et al. 2010), and some studies found no negative impacts on

species richness and composition (e.g. Bartomeus et al. 2010, Hejda and Pyšek 2006). In addition, Hejda and Pyšek (2006) and Hulme and Bremner (2005) indicate that control measures may pave the way for subsequent invasions of other alien species. In this context, we found in the second survey that several locations formerly occupied by *I. glandulifera* were invaded by *Solidago gigantea*, which had been largely absent in the first survey (Essl and Hauser 2002). This species may cause severe effects on soil properties and species assemblages in Central European habitats (Güsewell et al. 2006, Kowarik 2010, Koutika et al. 2011).

Managed populations of *R. pseudoacacia* decreased more strongly in ADI than unmanaged ones, but complete eradication of managed populations has not been achieved to date, as sprouting individuals are still frequent.

Implications for management of alien species in protected areas

Managing *Fallopia* spp. is difficult and costly, and thus early response is crucial for management success. For small populations eradication is achievable, whereas for large populations halting further spread is often the only option. *Fallopia × bohemica* exhibits a greater tolerance to clipping than its parental species (Rouifed et al. 2011), so mechanical treatment is less effective than application of herbicides (Delbart et al. 2012). The non-selective herbicides Imazapyr and Glyphosate are the most effective and most commonly used (Rudenko and Hulting 2010). Herbicide application is also the least laborious management measure (Delbart et al. 2012). However, under the current legislation in Central European countries, mechanical control often represents the only option, especially in protected areas. To conclude, we suggest that stem injection of herbicides be allowed in protected areas. However, future options for chemical control might become more limited because of increasing concerns of Glyphosate application. Potentially, the release of biocontrol agents for *Fallopia* spp. in Central Europe, as has already been done in Great Britain (Shaw et al. 2011), might be added to the portfolio of future management strategies. However, this option should only be used after rigorous host-specificity testing to avoid unwanted side-effects.

We found that management of *I. glandulifera* in large protected areas is feasible when the species is relatively rare. To achieve complete eradication continued monitoring of suitable habitats along river stretches close to the national park boundaries is important, to avoid re-colonization from populations outside the National Park Thayatal-Podyjí (Malíková and Prach 2010, Kowarik 2010). The management procedure used, i.e. one early clipping before flowering (June/early July) and manually removing overlooked plants in August, proved to be appropriate. As *I. glandulifera* is widespread in Central Europe, regional eradication will need long-term monitoring (Hejda and Pyšek 2006).

We found that even using the most effective management measures (girdling), complete eradication of managed populations of *R. pseudoacacia* is difficult to achieve within a decade. Our findings therefore emphasize that managing *R. pseudoacacia*

needs to be done over prolonged time periods (Cierjacks et al. 2013), although we found that management effort decreases sharply within a few years. Experience from the Czech part of the National Park Thayatal-Podyjí suggests that a combined approach of girdling and grazing with goats is another successful management option in habitats where grazing is possible (e.g. dry grasslands). Other suitable methods are girdling and planting native tree species to provide shade in the future, and the injection of herbicides into the stem. As all management measures are costly and time-consuming, *R. pseudoacacia* populations situated in forests may best be controlled by minimising anthropogenic disturbances (Terwei et al. 2013). This approach, which is compatible with conservation goals in protected areas, is consistent with observations in the Czech part of the National Park, where old-growth *R. pseudoacacia*-populations are starting to collapse. Similarly, the ADI of managed as well as of unmanaged populations in forests decreased between the two surveys (Fig. 4).

Implications for managing IAS in protected areas

In this study, we analysed the study species management by using an Area-Density-Index (ADI). This metric has the advantage that it allows comparison between management efficiency using a standardized measure which considers changes in population size and cover values. We found substantial differences in management effort that are necessary for the same reduction in ADI between species, with *R. pseudoacacia* requiring the most, and *I. glandulifera* the least effort. Although context-specificity (e.g. difficult accessibility of some *R. pseudoacacia* populations on steep slopes, which increases management effort per area) affects these results, some general conclusions can be drawn. Managing perennial plant species which spread vigorously vegetatively is particularly difficult and after a decade of management, only few populations are eradicated. Both *F. × bohemica* and *R. pseudoacacia* are known to be particularly difficult to manage (Delbart et al. 2012, Cierjacks et al. 2013, Schmiedel et al. 2016), but also are IAS with the highest environmental impacts in Europe (DAISIE 2013).

Conclusions

We found that substantial resources provided over prolonged time periods are needed for effectively managing invasive alien plant species in protected areas. The paucity of quantitative data on management effort is a severe constraint for assessing the efficiency of IAS management (Delbart et al. 2012). This is unfortunate, as currently we lack a profound understanding on the efficiency of alien plant species management, in particular in protected areas (Pyšek et al. 2013) and over the long-term (Sitzia et al. 2016). To improve monitoring of the efficiency of IAS management, we highly recommend a systematic approach for data collection on management effort in IAS management projects in protected areas.

Acknowledgements

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Supplementary material 1

Figure S1. Distribution of the three study species (*F. × bohemica*, *I. glandulifera*, *R. pseudoacacia*) in the National Park Thayatal-Podyjí in 2010

Authors: Verena Schiffleithner, Franz Essl

Data type: images

Explanation note: *Robinia pseudoacacia* predominantly invades forests near settlements, *Impatiens glandulifera* the Thaya river valley, and *Fallopia × bohemica* occurs mostly near settlements close to streams.

Supplementary material 2

Figure S2. Distribution of the three study species, *F. × bohemica*, *I. glandulifera*, and *R. pseudoacacia* in the Austrian part of the National Park Thayatal-Podyjí in 2001

Authors: Verena Schiffleithner, Franz Essl

Data type: images

Supplementary material 3

Table S1. Populations of *Fallopia × bohemica*, *Impatiens glandulifera* and *Robinia pseudoacacia* in the Austrian part of the National Park Thayatal-Podyjí

Authors: Verena Schiffleithner, Franz Essl

Data type: species data

Explanation note: Populations of *Fallopia × bohemica*, *Impatiens glandulifera* and *Robinia pseudoacacia* in the Austrian part of the National Park Thayatal-Podyjí, indicating population size, changes in population size between both surveys, and if management was applied.

Supplementary material 4

Table S2. Percentage and numbers of populations of the three study species

Authors: Verena Schiffleithner, Franz Essl

Data type: species data

Explanation note: Percentage and numbers of populations of the three study species in the five size classes (1 = 0–10m², 2 = 10–100m², 3 = 100–1,000m², 4 = 1,000–10,000m², 5 = >10,000m²) in the National Park Thayatal-Podyjí in 2010. Density classes (according to Braun-Blanquet 1964) are provided for *F. × bohemica* and *I. glandulifera*.

RESEARCH ARTICLE

Confronting the wicked problem of managing biological invasions

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Abstract

The Anthropocene Epoch is characterized by novel and increasingly complex dependencies between the environment and human civilization, with many challenges of biodiversity management emerging as wicked problems. Problems arising from the management of biological invasions can be either tame (with simple or obvious solutions) or wicked, where difficulty in appropriately defining the problem can make complete solutions impossible to find. We review four case studies that reflect the main goals in the management of biological invasions – prevention, eradication, and impact reduction – assessing the drivers and extent of wickedness in each. We find that a disconnect between the perception and reality of how wicked a problem is can profoundly influence the likelihood of successful management. For example, managing species introductions can be wicked, but shifting from species-focused to vector-focused risk management can greatly reduce the complexity, making it a tame problem. The scope and scale of the overall management goal will also dictate the wickedness of the problem and the achievability of management solutions (cf. eradication and ecosystem restoration). Finally, managing species that have both positive and negative impacts requires engagement with all stakeholders and scenario-based planning. Effective management of invasions requires either recognizing unavoidable wickedness, or circumventing it by seeking alternative management perspectives.

Introduction

The Anthropocene Epoch represents an era of unprecedented environmental change driven by human activities, a key component of which is the widespread transportation, spread, and resulting homogenization of fauna and flora (Williams et al. 2015). In a world fundamentally altered by anthropogenic processes, problems encountered in ecosystem management, and in particular in conservation biology and resource management, are becoming increasingly complex, where problems may not have a single, technical solution (Haubold 2012). More specifically, decisions regarding conservation in the Anthropocene need to consider the social and economic context (Ban et al. 2013), including the differing values stakeholders use when assessing risk (Liu et al. 2011, Kumschick et al. 2012). Conservation goals are set more often by the social-political perspectives of different stakeholders than by the empirical evidence (Geist and Galatowitsch 1999, Sagoff 2009). The consequent multitude of conflicting perspectives, objectives, and management goals can make the problem almost impossible to characterize, let alone solve, to the satisfaction of all stakeholders.

Such problems were first recognized in the policy and planning field by Rittel and Webber (1973), who coined the term "wicked problem". They defined a wicked problem according to 10 interrelated criteria, later condensed to six criteria by Conklin (2005; see Box 1). Wicked problems can also be viewed in the context of complexity theory as management problems where the cause-and-effect relationships between components, whether they be logistical components or stakeholders involved in management, are unordered and thus have solutions that are not obvious and require collaboration among stakeholders to determine appropriate actions (Kurtz and Snowdon 2003, Van Beurden et al. 2011). Such problems are contrasted against "tame" problems where the cause-and-effect relationships between components are ordered and the solutions obvious or discernible after careful investigation (Box 1).

Problems in the management of biological invasions have previously been referred to as wicked problems. The term was used by Evans et al. (2008), citing difficulties encountered when managing aquatic pests in the Crystal River, Florida; by McNeely (2013) when describing the management of plant introductions in conservation areas; and by Seastedt (2014) when describing the socio-political and ethical issues surrounding biocontrol. The management of biological invasions is particularly susceptible to wickedness in the form of conflicting social pressures. Differing values and risks ascribed to individual taxa by affected parties can lead to social conflicts around their management (Liu et al. 2011, Estévez et al. 2015). The wickedness of a problem will vary from case to case. Not all criteria might apply, some criteria may out-weigh others in making a particular problem more or less wicked, and the wickedness of a problem can vary by region or country according to the perspectives of the different stakehold-

Box 1. Criteria for a wicked problem and glossary of related terms.

A wicked problem is defined as one with the following properties:[1]

    1)   **You do not understand the problem until you have developed a solution.** Different stakeholders might disagree on some or all aspects of another stakeholder's definition to the problem, if they are personally invested in pursuing a particular solution.

    2)   **There is no stopping rule.** Because neither the problem nor its potential solutions are definitive, there is no obvious point or stage at which problem solving activities can be curtailed.

    3)   **Solutions to the problem are not right or wrong.** Rather, you can have solutions that are viewed as "better" or "worse" by consensus of the stakeholders.

    4)   **Every solution to the problem is a 'one-shot operation'.** An enacted solution causes new aspects of the problem to emerge, which must then be dealt with in turn, using follow-up solutions.

    5)   **Wicked problems have no given alternative solutions.** Many potential solutions could be thought of, but only some will be appropriate to pursue, depending on the problem's individual nature and social context.

    6)   **Each problem is essentially unique.** The source of wickedness lies in the social complexity of the stakeholders, and this will always vary from case to case.

Glossary of related terms

Complexity: In the context of project management, complexity is the number of components required to solve a problem, and the nature of the interactions between all components[2]. In complexity theory, the gradient of increasing complexity can be divided into ordered (where interactions between components are known or knowable), and unordered (where these relationships are unknown or disputed)[3,4]. Wicked problems thus represent problems with unordered complexity.

Tame: A problem which falls within the ordered domain of complexity theory. The components to the problem may vary in number, but their interactions are known or knowable[4].

Simple: A tame problem with few components, which share known interactions[4].

Complicated: A tame problem with many components, which share known or knowable interactions[4].

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[1] Conklin 2005; [2] Baccarini 1996; [3] Kurtz and Snowdon 2007; [4] Van Beurden et al. 2011

ers involved. In each of these cases, however, it is important to understand how the nature of the problem affects how it can be managed.

In this review, we assess how altering perceptions of managers and stakeholders to the nature and scope of problems presented by biological invasions can complicate or simplify the management solution. The options available to conservationists and environmental managers change with subsequent stages of invasion from initial incursion to spread to widespread establishment (Blackburn et al. 2011, McGeoch et al. 2016) and the complexity associated with solving the problem will intensify as invasions progress through these phases. We interrogate four examples of invasive species management problems across aquatic and terrestrial ecosystems, which focus on achieving prevention, eradication, or impact reduction. Our aim was to illustrate how wickedness in conservation management can arise and might be counteracted, realising that this is not always possible. We also identify situations where biological invasions can best be managed by shifting one's perspective and subsequent management approach to the problem.

Case study 1: Limiting wickedness in the prevention of invasions: managing ballast water in the Laurentian Great Lakes.

Much of the complexity in invasive species management stems from the complications of managing individual species once they have arrived in an environment. This can, however, be avoided by minimizing the chance of such species arriving in the first place. Indeed, many governments and policies worldwide (e.g. Convention on Biological Diversity) now focus on vector management, aiming to preclude non-indigenous species from being introduced (e.g. Environment Protection and Biodiversity Conservation Act 1999 (Australia); Environment Canada 2004; National Environmental Management: Biodiversity Act (South Africa) 2004; EU Regulation 1143/2014 (European Union) 2014; Genovesi et al. 2015). A substantial literature recognizes the importance and addresses the issue of vector (or pathway) prioritization (e.g. see Ruiz and Carlton 2003, Hulme 2009, Essl et al. 2015).

Ballast water and hull fouling are potent vectors responsible for transmitting alien species internationally. Both vectors represent major threats to ecosystems for two reasons: they carry from tens to hundreds of species simultaneously, and the number of individuals of each species may range from low to very high (Briski et al. 2014). The task of preventing the arrival of these species may initially appear to be a wicked problem, but can be approached as a straightforward, tame problem, provided it is addressed appropriately (Box 2).

Managers seek to reduce the risk of introducing a new species either by targeting the species itself or by focusing on pathways that allow the target species, and others, to arrive in a new environment. Species-specific risk assessment uses information on the number of individuals introduced and other demographic data. This approach may allow researchers to prioritize areas at highest risk of an invasion by a single species, although estimating the probability of successful establishment in any one ecosystem remains problematic (Herborg et al. 2007). It is, however, extremely challenging to develop single-species risk assessment models for species that use a vector capable of transporting multiple taxa. The wickedness of this problem lies in the fact that each newly introduced species will have its own propagule pressure, physiological tolerance to ambient conditions, and demographic constraints (see Seebens et al. 2013, Chan et al. 2014). This combination of factors results in tremendous variation in the probability of individual species successfully establishing in a new community and renders it virtually impossible to calculate the overall probability of a successful invasion. Drake and Lodge (2004) attempted to identify areas of greatest risk of future invasions from ballast water releases by analysing global shipping networks. Seebens et al. (2013) took a similar approach but also considered environmental matching and biogeography.

By switching the approach from species management to vector management, the risk management proposition becomes far simpler, as does the number of possible solutions (Box 2 - Figure 2). The framing of the problem around introduction events rather than focusing on species, removes nearly all wickedness from the problem according

Box 2. Ballast water management in the Laurentian Great Lakes.

Background

Water was first utilized as a form of shipping ballast in English coal vessels during the 1850s[1]. Ballast water largely supplanted soil ballast by the early twentieth century, after which invasions to the Great Lakes became increasingly dominated by this vector[2]. Following the opening in the late 1950s of the modern St. Lawrence Seaway – which provided access to all five lakes by transoceanic commercial ships – ballast water dominated all other vectors of introduction, accounting for between 55 and 70% of the 56 known aquatic invasive species that were recognized during this period[3]. Formal ballast-water regulation began in 1993 for international vessels with tanks filled with fresh water. In 2006 (Canada) and 2008 (USA), these regulations were extended to vessels with only residual water in tanks. In both cases, vessels were required to conduct open-ocean exchange or flush salt water through their tanks, respectively, to reduce invasion risk. No new ballast-mediated invasions have occurred since 2006.

Mediators of wickedness

Species-specific risk assessments consider the likelihood of a species interfacing with, and being transported by, a transport vector, survival during transit, and likelihood of introduction to and survival in a new environment. Assessing overall risk is highly problematic when discharged ballast water contains multiple species, each with a different population abundance, life history, and physiological tolerance. The alternative approach of a pathway-level assessment treats each species and every propagule as equivalent, akin to neutral theory models used to predict species replacements in natural communities[4]. Managers can then assess total propagule pressure combined across all species, as well as colonization pressure (number of species introduced), released into the new environment to determine relative invasion risks of different introduction events[5]. This approach allows a wicked problem to be analysed at the pathway level, transforming it into a resolvable or tame problem. It should be noted that, within this conceptual framework, increasing numbers of vectors can make a simple problem become complicated in terms of the number of pathways and variation in associated regulations that can be brought to bear to maintain biosecurity[6].
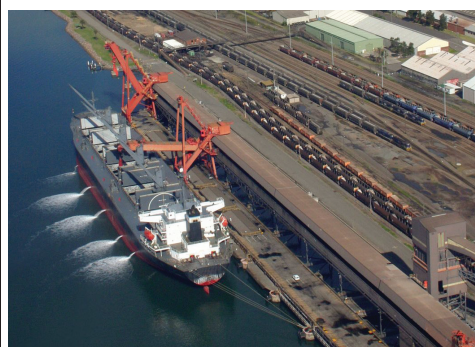

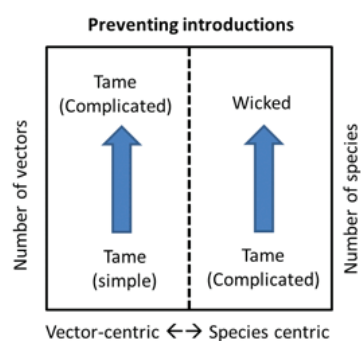**Figure B2-1.** Ballast water being emptied into the St Lawrence River

Figure B2-2. Conceptual diagram for Case 1*.

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[1] Carlton 1985; [2] Mills et al. 1993; [3] see Bailey et al. 2011; [4] Hubbell 2001; [5] Drake et al. 2014; [6] e.g. Padilla and Williams 2004.

* In this conceptual diagram, the dichotomous x-axis reflects the two management approaches that can be brought to bear on biosecurity management. The left and right y-axes reflect the dominant driver of complexity for each approach, although both drivers (number of species and number of vectors) can affect overall complexity of a particular management problem whether a species-centric or vector-centric approach is taken.

to Conklin's criteria (Table 1). Ultimately, the solution to the problem of ballast-water introductions lies in the effective regulation of the use of ballast water in shipping. This has been partially achieved in the Great Lakes, as both USA and Canadian authorities enacted regulations (see Bailey et al. 2011) that have resulted in measurable declines in new introductions to the Laurentian Great Lakes (Box 2). These empirical findings are consistent with Drake and Lodge's (2004) theoretical model that predicted that reducing per-ship invasion risk would be more effective at preventing invasions than knocking out key ports in a shipping network.

Successful vector management in the case of the Great Lakes works because focusing on one stage — a choke point — in the invasion process simultaneously knocks out the vast majority (but not all; MacIsaac et al. 2015) of the possible invaders prior to introduction. Vector control may not always be as simple, however. Other trade vectors that allow hitchhiking by invasive species can be harder to treat effectively (e.g. wood dunnage in shipping), despite internationally mandated treatment standards (Haack et al. 2014). Moreover, some pathways for introduction (e.g. the aquarium pet trade) comprise multiple vectors and are largely unregulated at a global scale (Padilla and Williams 2004). In such cases, biosecurity risk management becomes far more complicated, due to the diverse number of companies and organizations involved, and the fact most of the players are not subject to a uniform set of regulations that is enforceable in practice, unlike ballast water management in North American waterways. Thus, the geo-political scope of the vectors will determine the practicality of vector management and the availability of workable solutions (Box 2). Nonetheless, we advocate that vector-centric management solutions to problems of biosecurity should be explored given their potential to reduce wickedness.

Case study 2: Ecological scope can determine wickedness: the eradication of invasive species from islands

The case of multiple vectors enabling the transport of potential invaders highlights that, while changing problem formulation can often reduce the wickedness of a problem, the scope of the problem can be a fundamental driver of complexity in the management of biological invaders. This is illustrated by our second case study, which examines the challenge of eradicating invasive species (Box 3). At a superficial level, the tamest invasive species problem is that of an invader that has established on a small island with no human habitation, high conservation value, and where the chance of reinvasion is negligible (e.g. Donlan et al. 2014). There is often, though not always, agreement among stakeholders (in this case the governmental custodians of the island) that, if budget allows, an attempt should be made to eradicate the invader. The removal of such a species, however, is implicitly an attempt to remove its impacts on the receiving environment, which adds multiple permutations to the formulation of the goal (Box 3 - Figure 2). As one increases the scope of the problem to reflect broader conservation goals, the number of potential solutions, and the number of potential

Box 3. Eradicating invasive species from islands.

Background

Here, we consider eradication to be the elimination of a species from an area to which recolonization is unlikely to occur[1]. In this sense, invasive vertebrates have been eradicated from islands a number of times as part of conservation initiatives[2]. Eradication success generally depends on the biological traits of the target species, the ecology and environment of the island (especially whether it is remote enough for recolonization to be unlikely), and socio-economic factors involved in implementing the eradication attempt. While such eradication efforts might be pro-active (e.g. to remove a new incursion), they are often in response to documented evidence of substantial undesirable impacts. The goal of eradication in this case is essentially to contribute towards island restoration.

Mediators of wickedness

The eradication of invasive mammals from islands has led to substantial conservation benefits[3], but such actions can result in unintended consequences[4]. Thankfully, past experiences have provided a framework for planning that has worked in practice[5], so while the problem might be complicated, it is still tame. However, the problem becomes more challenging if all non-native species on a given island are considered. The eradication of plants, invertebrates, and micro-organisms pose additional practical and theoretical challenges (e.g. being able to detect and treat all individuals and to understand which taxa are actually non-native). This quickly leads to a management problem that is impractical to solve under any reasonable budget. Similarly, larger islands, and those with multiple stakeholders (in particular those that are inhabited), will typically be more difficult to manage[6].

Where the problem becomes wicked (as opposed to being complicated in terms of resource allocation) is if the management goal is not eradication per se, but island restoration. Often, after an agent of perturbation (the invader) has been removed, even if there is a clear baseline to which the island should be restored, the process will need to be on-going and adaptive. Instead of following set best-practice procedures for eradicating a particular species, or proscribed good practice for eradicating multiple taxa, there will need to be an emergent practice of restoration tailored for the local conditions.

Figure B3-1. Baited rat station in Gwaii Haanas National Park Reserve, British Columbia, Canada. Photo courtesy of Laurie Wein, Parks Canada.
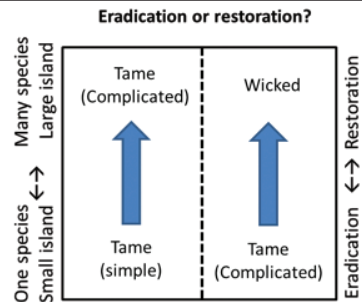

Figure B3-2. Conceptual diagram for Case 2*

[1] Myers et al. 1998; [2] DIISE 2015; [3] Jones et al. 2016; [4] Bergstrom et al. 2009; [5] Cout et al. 2009; [6] Glen et al. 2013.

* Note: In this conceptual diagram, the left-hand y-axis represents the drivers of complexity for eradication, while the right-hand y-axis represents the impact of a shift of strategy from eradication to restoration. Eradication tends to be more complicated as more species are targeted or the island is larger. But, shifting the overall goal from individual species to ecosystem processes can transform the problem from complicated to wicked. If multiple stakeholders are involved (e.g. inhabited islands), the problem can also become wicked (see case studies 3 and 4 below).

unintended consequences, increases rapidly to the point of posing a wicked problem in terms of most criteria (Table 1). The ecological context of the invasive species on the island might also add complexity to the problem that, if unaddressed, may lead to management solutions that exacerbate, rather than improve, the situation. A classic example is that of the feral cat *Felis catus* eradication on Macquarie Island. The successful eradication of cats led to an upsurge in the invasive rabbit *Oryctolagus cuniculus* population that worsened the ecological functioning and conservation status of the island (Bergstrom et al. 2009). This example clearly illustrates the implications of criteria 4-6 in Conklins' (2005) formulation (Table 1). Recognizing the interplay between different invasive and native species in the island ecosystem has prevented such unintended negative consequences on other islands (Caut et al. 2009), but avoiding such surprises requires a more comprehensive assessment of the ecosystem-level consequences of a management plan (e.g. incorporating food web and functional networks into ecological risk assessment) prior to its implementation (Zavaleta et al. 2001).

To provide a meaningful assessment of the ecological risk of a planned eradication, heuristic, qualitative modelling approaches such as community matrix loop analysis (to determine likely positive and negative trophic interactions) and fuzzy interaction webs (providing qualitative predictions of complex community responses to a particular perturbation) can broadly model the likely interactions within island food webs under different consumer control regimes (Dambacher et al. 2002, Ramsay and Veltman 2005). These approaches thus provide a tool for managers to recognize the hidden wickedness within a superficially tame problem. Through these heuristic approaches, managers can select individual management strategies (e.g. targeting high-impact predators with weak trophic links to invasive grazer species) that are less likely to result in novel and unintended consequences.

The eradication of individual species from islands is, thus, a management problem that can be worthwhile pursuing, provided that the likely implications of the chosen solution are adequately understood. In contrast, there will be invasive species which have little impact on ecological communities. In such cases, it might be a waste of limited resources to attempt eradication. A prioritization framework proposed by Kumschick et al. (2012) provides a structured procedure by which managers can focus limited budgets towards invasive species with high negative environmental impact. This framework is also applicable in the case of inhabited islands where humans are potentially impacted by the invasive species, or may object to an eradication program on ethical (in the case of animal eradications) or aesthetic (in the case of flowering plants) grounds (Estévez et al. 2015). Through such prioritization mechanisms, conservation managers can choose sufficiently tame goals that are specific, measurable, achievable, relevant, and time-bound, following the principles of management goal-setting advocated by Doran (1981).

The potential for conflict surround eradications on inhabited islands demonstrates a major diver of wickedness in invasive species management, namely the involvement of multiple stakeholders with different perspectives on the invasive species problem (Glen et al. 2013). Problems in invasive species management shift from complicated

to truly wicked when one has to deal with species that can be either harmful or useful depending on the socio-economic context within which they are assessed, so that eradication is no longer a viable option. At this point, management of the species generally shifts towards minimizing the known or perceived negative impacts of the species, which allows many new opportunities for the problem to become wicked. This is especially true in cases where the species in question was deliberately introduced to provide benefits. The final two case studies of this review explore "conflict species" in terrestrial and aquatic ecosystems respectively. Both case-studies focus on taxa that proved extremely difficult to manage for contrasting reasons. In the first of these (case study 3), the problem was initially formulated without all stakeholders engaged, and so the enacted solutions were incomplete and largely ineffective.

Case study 3: Changing circumstances heighten wickedness: Controlling invasive alien pine trees in the Cape Floristic Region of South Africa

Pine trees (*Pinus* spp.) were originally planted in the Cape Floristic Region of South Africa to provide timber in a region that had few natural forests. While that benefit still applies today, they are now also seen as a threat to water resources and biodiversity (Box 4). Pines are, therefore, conflict species—they are simultaneously seen as useful (by foresters) and harmful (by conservationists). Moreover, the funding for projects aimed at reducing the extent of invasive populations is secured on the basis that these control projects can generate employment (van Wilgen et al. 1998). This has meant that the primary focus of management has shifted from utilization to control to job creation, adding to the difficulty of achieving effective control in priority areas. Instituting partial solutions over time that address the problems of some, but not all, affected stakeholders, has given rise to new problems, and this cycle has led to a situation that meets every criterion of a wicked problem (Table 1). Here, a shortage of timber was addressed by planting alien trees (ignoring conservation), which led to invasions; this was addressed by retaining commercial forestry but combining control programs with job creation. The addition of job creation to the stated goals of the management solution has led to a loss of focus on control, making control ineffective, and further fuelling on-going, intractable conflict. Thus, as the invasion spread over time, the competing interests regarding their preferred management has resulted in a clearly wicked management problem (Box 4 - Figure 2).

In theory, there is a solution to the problem of pine management that would satisfy all stakeholders. Such a solution would see populations of invasive pines in vulnerable catchment areas reduced to levels where they can be sustainably controlled at these low levels and where plantations of the same species can simultaneously be maintained for their benefits in the landscape. The very large extent of invasions and the exorbitant costs of such a solution render it practically unattainable, and all alternative partial solutions are contentious (van Wilgen and Richardson 2012). For example, it may be advantageous to focus control efforts on priority areas while abandoning others, to

Box 4. Controlling invasive alien pine trees in the Cape Floristic Region of South Africa.

Background

Pine trees (*Pinus species*) have been extensively planted in South Africa since the 1930s to provide timber[1]. Pines began spreading beyond the borders of formal plantations, where they invaded the adjacent fynbos shrubland vegetation of the Cape Floristic Region. Invasion by alien pine trees was recognized as a problem as early as the 1940s, and coordinated attempts to clear these invasions began in the 1970s. Although clearing attempts have continued at often substantial levels since then, the extent of invasions continues to grow[2]. Because pine trees are simultaneously useful and harmful, depending on the perspective adopted, the situation becomes more and more polarized, exacerbated by the fact that perspectives change over time as value systems and economic circumstances change[3].

Mediators of wickedness

The problems associated with the management of pine invasions were initially complicated, but arguably manageable. Complexity initially arose from attempts to grow a crop species that was also highly invasive. The species spread into inaccessible areas where clearing was difficult, and wildfires promoted further spread, making control difficult. However, with time and increasing geographic extent of invasions, a number of new factors were added to this complexity. Both the need to prevent biodiversity loss and to stimulate economic growth are becoming more acute, leading to polarized views regarding the advantages (timber, shade and amenity values) and disadvantages (biodiversity and water losses, and increased fire hazard) of pines. Recent analysis predicts the net value of benefits minus impacts will become negative as invasive pines spread[3], but suggestions to phase out pine based plantation forestry[1] and introduce biological control agents[4] have been met with strong opposition from stakeholders with interests in the current benefits from forestry and downstream industries. A shift in the emphasis of control projects (from the restoration of ecosystems to employment creation and poverty relief associated with managing the invasive stands) has introduced the often competing needs of meeting dual goals. To date, suitable compromises to these problems have not been found, nor do they seem possible, signalling that this issue has become wicked.

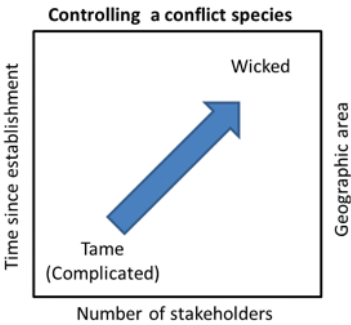**Figure B4-1.** Invasive pines spreading from a plantation in the Cape Floristic Region.

Figure B4-2. Conceptual diagram for Case 3*.

[1] van Wilgen and Richardson 2012; [2] van Wilgen et al. 2012; [3] van Wilgen and Richardson 2014; [4] Hoffmann et al. 2011.

* Note: In this conceptual diagram, the x-axis and both y-axes represent independent drivers that can impact complexity individually or in combination. Invasive pines were originally perceived by managers to be in the lower left of the concept space, though in reality the problem was more towards the upper right. Today, all three drivers continue to contribute to the wickedness of invasive pine management.

more effectively utilize scarce funds (Forsyth et al. 2012). There is, however, reluctance to phase out control projects in lower-priority areas to achieve this, because of the political implications of cutting jobs in areas where unemployment is high. Similarly, phasing out plantation forestry to reduce propagule pressure on vulnerable watersheds is an option (van Wilgen and Richardson 2012), but this proposal was met with stiff resistance from the forest industry (van Wilgen and Richardson 2014). Finally, it may be necessary to accept that the problem cannot be solved and that management may need to recognize the existence of a novel ecosystem (*sensu* Hobbs et al. 2014) in which pines constitute a permanent component.

As the pine management example demonstrates, acknowledgement of all relevant stakeholders to an invasive species management problem is a key requirement for generating sustainable solutions that can be supported by both government and civil society. Knowing all the players does not, however, mean a solution that satisfies all is easy or even possible. Our final case study deals with an invasion problem where key stakeholders hold diametrically opposed positions on the nature of the problem and its preferred solution.

Case study 4: Conflict species with polarized stakeholders maximize wickedness: Managing invasive rainbow trout around the world.

Invasive alien rainbow trout (*Oncorhynchus mykiss*) is a classic conflict species. It is both highly desirable as a resource and detrimental to the aquatic environments in which it establishes (Box 5). Where introduced, salmonids have had considerable ecological impacts on recipient ecosystems that span multiple biological domains (e.g., Dunham et al. 2004, Garcia De Leaniz et al. 2010, Ellender and Weyl 2014). They nonetheless represent significant recreational and economic value for the regions into which they were introduced, with the result that management goals can be polarized among conservationists, anglers, and fish farmers.

This has resulted in direct opposition by some stakeholders to the management goals of others. In New Zealand, proposed efforts to control invasive trout by the Department of Conservation were vociferously opposed by angling bodies, seeing the proposals as the "thin edge of the wedge" to begin removing their preferred sport fish from popular fishing waters (Chadderton 2003). In South Africa, trout are held in such esteem by some recreational anglers that they have prompted the formation of sporting associations such as the Federation of South African Flyfishers, whose mandate is to protect trout angling from the threat of conservation authorities (Ellender et al. 2014). This organized reaction to conservation authorities in government became more active in response to draft regulations in 2013 that classified trout as an alien species requiring control (Ellender et al. 2014). The result was a coordinated lobbying effort that managed to prevent the inclusion of trout on the promulgated list of regulated alien species, despite scientific evidence that demonstrated the invasiveness and impact of trout within South Africa (e.g. Ellender and Weyl 2014, Shelton et al. 2014).

Box 5. Managing invasive rainbow trout around the world.

Background

The rainbow trout (*Oncorhynchus mykiss*), included in a list of 100 of the world's worst invaders[1], has been introduced to 99 countries[2]. Like most invasive fishes, it is among a few groups of organisms that have been deliberately introduced into the environment with the express purpose of creating self-sustaining populations in the wild or to maintain wild population abundance, regardless of wild reproduction[2]. Trout introductions often achieved the desired objective of developing sport and commercial fisheries that contribute significantly to local and regional economies[3]. For example, one estimate places the economic benefit of alien sport fishes to the USA at US$69 billion annually[4]. These intentional introductions continue to occur despite changing views on the stocking of alien species due to their potential ecological impacts[5]. Negative impacts of the species include hybridization with congeneric species, parasite transfers between cultured and wild individuals, extirpations of native fishes and amphibians due to competition and predation, and cascading food web impacts at community and ecosystem levels.

Mediators of wickedness

Management of alien salmonids is complicated by differences in value systems and the risk perceptions of stakeholders and decision makers. For example, illegal introductions of invasive fishes are also a source of conservation concern and the effective long-term management of invasive fishes relies on stakeholder support[6]. This is complicated by the predominantly positive angling values associated with invasive salmonids, which are a source for conflicts when attempting to control invasions and typically resolved in favour of alien sport fisheries[6]. A major problem with managing invasive fishes is that, once established, control is extremely difficult. In many regions, implementing management interventions is also complicated by the economic contributions of angling and aquaculture to local economies[7] and by resistance by some anglers and managers, whom actively support stocking and argue in favour of considering alien salmonids part of the native biodiversity[6] and often use the term "naturalized" to distance themselves from the term "invasive".

Figure B5-1. A rainbow trout caught and about to be released back into the Broken River, New Zealand.
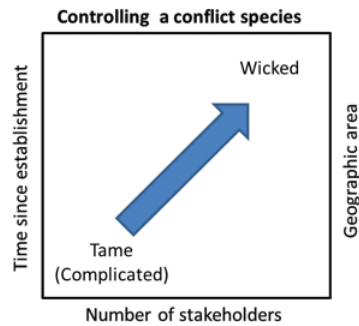

Figure B5-2. Conceptual diagram for Case 4*.

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[1] www.issg.org; [2] Crawford and Muir 2007; [3] Cambray 2003; [4] Gozlan et al. 2010; [5] Helfman 2007; [6] Ellender et al. 2014; [7] Quist and Hubert 2004.

* Note: In this conceptual diagram, the x-axis and both y-axes represent independent drivers that can impact complexity individually or in combination. The problem of managing introduced trout tends to fall in the upper right of the concept space in regions where the species is established. Unlike with pines, time since establishment has not been a major driver of complexity in trout management, as the underlying problems were apparent shortly after initial establishment in most countries.

The situation is less polarized but more spatially complex in North America, where invasive rainbow trout is highly valued as a sport fish by anglers, except when it is perceived to impact other sport fishes, often congeners, of higher value. In the past, rainbow trout, brown trout (*Salmo trutta*), and brook trout (*Salvelinus fontinalis*) had been stocked over native cutthroat trout (*O. clarkii*) populations in many Rocky Mountain streams to enhance angling opportunities. This has resulted in competition from all three invasive salmonids and, more alarmingly, introgression with rainbow trout, threatening the persistence of pure strains of cutthroat trout (COSEWIC 2006). As cutthroat trout is preferred by anglers, particularly fly fishers, angling organizations like Trout Unlimited support the eradication of rainbow trout from waters where the cutthroat trout is present. This organization aims, "to conserve, protect and restore North America's coldwater fisheries and their watersheds" and to "ensure that robust populations of native and wild coldwater fish once again thrive within their North American range…" and is against stocking non-native hatchery trout on top of native wild trout populations (Trout Unlimited 2015).

As a result of the apparent conflicts between establishment and eradication, and associated economic and ecological impacts, the management of introduced salmonids provides a thoroughly wicked set of problem formulations and potential solutions, further influenced by spatial and political variation globally (Table 1). The likelihood of achieving practical solutions for managing conflict species such as pines or trout will depend on managers understanding the different players, their perspectives, and directly engaging with them to identify equitable management goals.

Conclusion: Recognizing and effectively dealing with wickedness in invasive species management

The four case studies represent the types of problems that conservation managers regularly face when managing the incursion, establishment, and impact of invasive species. A consistent theme throughout these examples is the frequent disconnect between the perception of the problem by managers and the reality they face. Indeed, the first, and possibly most important, of Conklin's criteria is that of problem formulation. In many ways, wickedness begins when the scope of the problem is misinterpreted or, worse, underestimated. This disconnect can lead to a succession of inappropriate or incomplete solutions being offered that, in the case of pines in South Africa, have historically led to ineffective management policy. Our four case studies represent a matrix of management problems in which the perception and the reality of wickedness vary (Figure 1). By recognizing when such disconnects exist, managers may be able to devise management solutions to biological invasions that are more effective, more sustainable and less prone to unexpected negative consequences, whether it be unwanted ecological interactions or push-back from negatively affected stakeholders.

In the case of ballast-water management, shifting the problem formulation from species-oriented to vector-oriented actually revealed a perceived wicked problem to be a
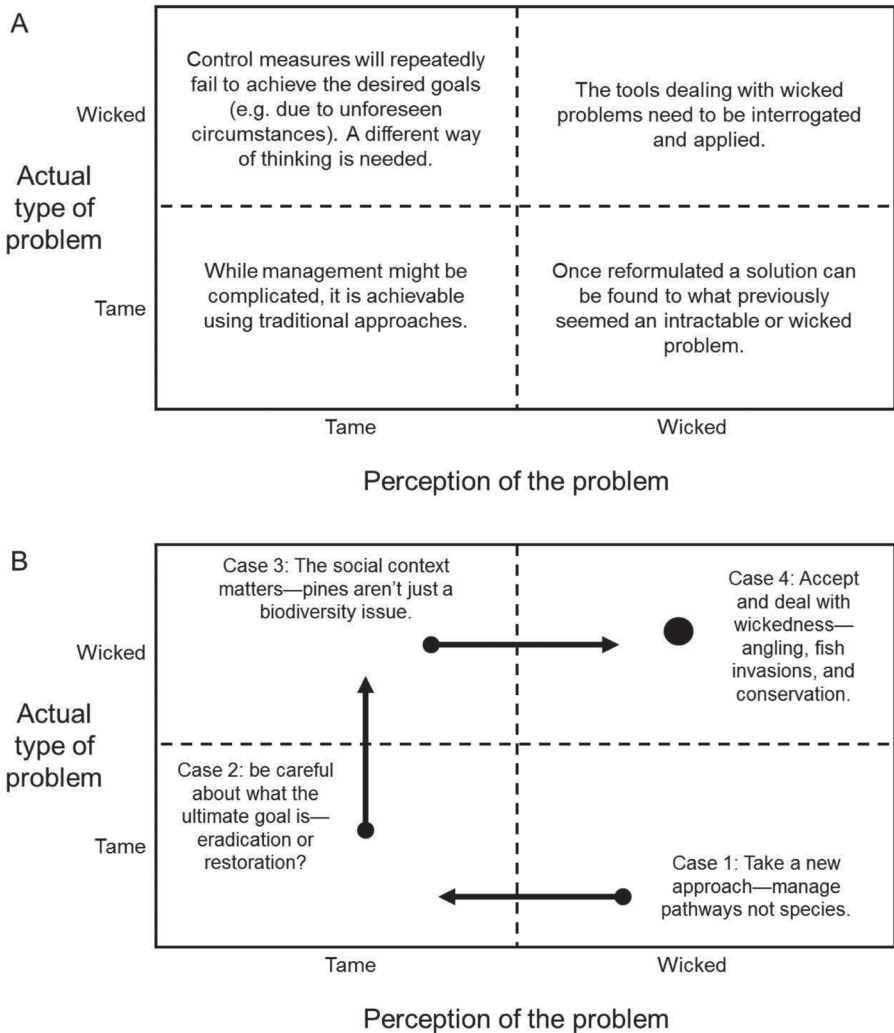
Figure 1. Conceptual diagram of perceived and real wicked problems in managing biological invasions. **Panel A** represents a matrix of how perceived and actual wickedness can influence the outcome of management; **Panel B** illustrates emergent lessons from the four case studies of invasive species management discussed here. Vectors represent shifts in problem perception and management paradigms necessary for improving the manageability of each case study.

relatively tame, if complicated and potentially expensive, problem to tackle. The key to the ultimate success of ballast-water control in the Great Lakes was to realize that the risk posed by the vector would apply to any species that used it for dispersal. Thus, a shift in perspective was the key to limiting the scope of problem formulation and its solutions.

Once an unwanted invasion has occurred, the management problem shifts from one of biosecurity to one of ecosystem management, where conservation managers seek first to eradicate, then to control the invader. In the case of mammal eradications from islands, most operations have been highly successful, with the few examples of documented negative impacts usually temporary in nature (Jones et al. 2016). However, eradication programs do need to explore the potential consequences of individual species eradications to ecosystem restoration before settling on a management direction. Our assessment of the complexities of island eradications revealed them to ultimately conform to 4 of the 6 criteria for wicked problems (Table 1), highlighting how managers will need to recognize the wickedness hidden within an apparently tame problem if they are to achieve success (Figure 1). Nonetheless, it is important for managers to recognize when limited funds mean that complete solutions, such as the removal of all invasive species from the island, are unachievable. It is in these situations that prioritization of invasive species and their likely impact is critical for pragmatic management solutions (Kumschick et al. 2012, McGeoch et al. 2016). The only criteria not met by case study 2 (Criteria 2 and 3; Table 1) are implicitly linked to variation in stakeholder perspectives, which can rapidly increase the complexity of invasive species management.

Conflict species represent the most widespread kind of wicked problem in invasive species management, because there is inherent disagreement on the formulation of the problem and its potential solutions. Invasive pines and trout do, however, differ in the divergence between the perception and reality of wickedness. In the case of the pines, it was the sequence of historical management solutions, put in place reactively as perceptions and the socio-economic context of pines changed over time, which led to a build-up of unintended consequences reflected in the present-day situation (an inherently wicked problem was, at first, incorrectly perceived as tame; Figure 1). A greater acknowledgement of contrasting stakeholder groups may have enabled a more balanced set of solutions to be implemented earlier, if the wickedness of the problem created by multiple stakeholders with divergent perspectives and priorities had been recognised from the start (Figure 1). The trout example, in contrast, represents an invasive species problem perceived as wicked from the outset of it being considered a problem at all (Figure 1). By the time conservation managers began to recognize the species' negative impacts, a strong lobby of anglers opposed proposed control in principle. Here, all the relevant stakeholders were recognized since the start of the conflict, but their opposing views on the nature of the problem have, in some cases, prevented any solutions from being developed.

An emerging field of structured stakeholder engagement, including scenario-based planning (SBP) can enable the development of solutions for wicked problems in invasive species management. The fundamental strength of SBP is that it enables stakeholders to bridge the gaps in their relative perceptions of a problem, by creating plausible future scenarios based on a limited set of proposed management actions, and then deciding which scenario is likely to have the most agreeable outcome to all parties (Peterson et al. 2003). This technique offers solutions that unify the problem

Table 1. Fitting Conklin's (2005) criteria of wickedness to four case studies of invasive species management.

Criterion	Case 1: Tame problems that may appear wicked – managing ballast water as a vector	Case 2: Problems that may be tame or wicked, depending on management goals – eradications on islands	Case 3: Wicked problems incorrectly perceived as tame become more wicked – invasive forestry species	Case 4: Disagreement over the nature of the problem ensures wickedness – invasive sport fishes
1) You don't understand the problem until you have developed a solution	**No.** Although management plans aimed at every potential invasive species are impractical, a management approach that deals with all potential invaders simultaneously (e.g. vector control) becomes simple to define.	**Yes.** Although the problem of eradicating a single species is easy to define, and has a clear solution, this would not guarantee ecosystem restoration. If the problem is more appropriately formulated as "Restore Island A to pre-invasion state" both the problems and potential solutions arguably become difficult to define *a priori*.	**Yes.** The solutions proposed to address this problem have dealt with a particular aspect of the problem (e.g. provide timber, protect water resources or biodiversity, or create employment) which has led to unsatisfactory outcomes for stakeholders who were ignored initially.	**Yes.** Many countries recognize invasive salmonids as a both problem and an asset, and hence have not developed a broadly accepted solution. In most countries, stakeholders have a diversity of views based on varying perspectives, values, politics, and financial resources. As a result, if deemed a problem, solutions may vary widely.
2) Wicked problems have no stopping rule	**No.** A comprehensive risk assessment and management plan for all species transported in ballast water is impossible, as the potential species pool is unbounded. It is however possible to successfully manage the vector itself.	**No.** The problem might be declared solved if a single species is eradicated, and new introductions can be prevented.	**Yes.** Pines can never be eradicated, so their management can never be stopped. The question becomes one of whether the invasions can be brought to a level where they can be contained sustainably. This should be possible but, despite considerable control efforts, pines continue to spread.	**Yes.** When management for any of the conflicting goals is the solution, there is never a point of ultimate success. Decision makers are often reluctant to identify a stopping rule given the diversity of stakeholder views.
3) Solutions to wicked problems are not right or wrong	**No.** One could argue that the vector-management approach to ballast water invasions is appropriate, as it nullifies other drivers of wickedness in this case.	**No.** A method that completely eradicates a single species can be called "correct", although methods used to restore ecosystems may be subjectively assessed on their overall success.	**Yes.** Pines are "conflict" species (simultaneously bringing benefits and doing harm), so it is necessary to make trade-offs, because it is both "right" to encourage benefits and "wrong" to tolerate harm.	**Yes.** Managing against the spread of invasive trout and its impacts may be viewed as "right" by conservationists but are likely to be viewed simultaneously as "wrong" by anglers who utilize the resource.

Criterion	Case 1: Tame problems that may appear wicked – managing ballast water as a vector	Case 2: Problems that may be tame or wicked, depending on management goals – eradications on islands	Case 3: Wicked problems incorrectly perceived as tame become more wicked – invasive forestry species	Case 4: Disagreement over the nature of the problem ensures wickedness – invasive sport fishes
4) Every solution to a wicked problem is a 'one-shot operation' that leads to new problems.	**No.** The management of invasion risk by controlling the vector through effective regulations means that each potential species invasion is prevented by the same, repeatable method.	**Yes.** Eradicating a species from an island will always depend on environmental context (geographic extent, logistical feasibility) for its success. Context dependency increases significantly with island size and ecosystem diversity. Removal of one species can lead to new problems.	**Yes.** Pines were introduced to provide timber, but became invasive, leading to reduced water supplies and biodiversity. The solution was to initiate control operations, but these could not be sustained. This was "solved" by combining control with poverty-relief to create employment leading to a shift in emphasis to job creation at the expense of effective control.	**Yes.** The historical, social and environmental context of each invasive trout population makes each solution have a wide range of potential unintended consequences.
5) Wicked problems have no given alternative solutions	**No.** Whether attempting to prevent a single species or all species from successfully using the ballast water pathway to enter North American waters, the treatment of ballast water is the clear solution to minimize the risk of introduction.	**Yes.** Some species can be eradicated from a defined geographic area using a small number of known methods. Ecosystem restoration has innumerable potential solutions based on the definition of restoration.	**Yes.** We seek to maintain forestry production in conjunction with control, but this appears to be unattainable, and all alternative partial solutions remain contentious.	**Yes.** There are at least three solutions – accept the invasion, eradicate, control. The latter two have many options, though many would be considered unacceptable by anglers.
6) Every wicked problem is essentially unique and novel	**No.** Ballast water as a vector has several key traits that make standardized treatment solutions viable across many different shipping routes.	**Yes.** The solution for eradicating one species on an island is likely to work on another island with the same species, but the implications of the eradication for ecosystem rehabilitation will be case-specific.	**Yes.** The problem of invasive pines in the Cape Floristic Region is embedded in a dynamic social-ecological context, where numerous factors interact, resulting in a unique situation for each stand of invasive pines.	**Yes.** Each salmonid population will have unique logistical constraints surrounding its management, as well as an associated group of stakeholders, who add individuality to the nature of the problem and its potential solutions.

formulation among stakeholders, thus, leading to negotiated solution sets that can limit wickedness. Building such scenarios can also alert managers to the potential unintended consequences of a proposed management action (Game et al. 2014). There will be cases where the perceived risk of an invasive species to different stakeholders is extremely variable, and the values attributed to impacts of a management action may fundamentally differ among them (e.g. for pine management: the risk to conservation vs. forestry revenue vs. poverty alleviation by contracting conservation work to rural communities). In such situations, a structured risk evaluation such as the Deliberative Multi-Criteria Evaluation approach (DMCE; Liu et al. 2011) could offer a potential way forward in the negotiation process. This approach compels each stakeholder to rank perceived risks of a proposed management strategy in terms of importance, thus, potentially highlighting cases where projected negative outcomes of management are likely to be less severe than initially perceived. For example, a potentially contentious action, such as controlling an economically important invasive species within a vulnerable conservation area, may be less prone to protest from stakeholders if it can be demonstrated that the management action will not pose a significant risk to their continued utilization of nearby invasive populations (Weyl et al. 2014).

To illustrate how SBP might enable solution development for trout management, we can examine a specific conflict currently underway in South Africa. Rainbow trout is fished for, and grown in a hatchery, within a sub-catchment of the Breede River system, which is also a conservation area that contains a threatened native fish species (Weyl et al. 2015). It is clear that removing the trout from some reaches also used by anglers would improve the conservation status of the native species, though local angling organizations have opposed this proposed intervention. To negotiate a solution, SBP could be used, involving conservation authorities, fish biologists with expert knowledge on the species involved, local NGOs, the angling society responsible for the trout fishery and the trout hatchery owners. Scenarios for different management options (e.g. the removal of trout from different river sections) could be proposed, mapped out and debated for their likely impacts on the various stakeholders present at the negotiating table. A key logistical consideration of these scenarios would be the construction of artificial barriers to upstream movement, to ensure reclaimed river reaches are not re-invaded (Weyl et al. 2014). In this particular example, the positions and risk-perceptions of the players involved are likely to be well enough understood that a DMCE process is unnecessary, although engaging the stakeholders in this process may nonetheless facilitate the softening of positions on trout control, thus facilitating negotiation towards and equitable solution.

In any country where invasive species have become established, there can be no hope for all-encompassing, "silver bullet" solutions to the problem. Rather, management practices should be focused on mitigating the long-term negative impacts of the species, at whatever spatial scale consensus can be reached among stakeholders on the nature of the problem, with the consensus being found through structured engagements such as SBP or DMCE. But, as the invasive pines case study shows, identifying and including all the stakeholders in the negotiation and planning will be critical to

ensure that even pragmatic, partial solutions are less likely to create new problems for conservation management. Similarly, even if stakeholders can be brought to a negotiated consensus, the chosen solution set must be within the capacity of the management authority to act upon, lest budget or technical constraints render the preferred solution unachievable (as in the island eradications case study).

As the human-mediated biogeographic processes that characterize the Anthropocene continue to intensify, there is a growing recognition of wicked problems in conservation management around the world (Game et al. 2014, Seastedt 2014). As anthropogenic dispersal of organisms continues to grow and conservation budgets remain constrained in a volatile global economy, the management of invasive species will increasingly require novel approaches, including heuristic assessments of the ecological risk associated with proposed interventions, and adaptive, stakeholder-conscious management through structured engagement initiatives, to enable positive outcomes for ecosystem integrity. By correctly identifying the complexity of interactions between these species, their environment, and the people that benefit or suffer from their presence, managers may better frame their response to the threat of new invasions and, thus, produce more pragmatic and effective solutions.

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DISCUSSION PAPER

Biological invasions and natural colonisations are different – the need for invasion science

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Abstract

In a recent Discussion Paper, Hoffmann and Courchamp (2016) posed the question: are biological invasions and natural colonisations that different? This apparently simple question resonates at the core of the biological study of human-induced global change, and we strongly believe that the answer is yes: biological invasions and natural colonisations differ in processes and mechanisms in ways that are crucial for science, management, and policy. Invasion biology has, over time, developed into the broader transdisciplinary field of invasion science. At the heart of invasion science is the realisation that biological invasions are not just a biological phenomenon: the human dimension of invasions is a fundamental component in the social-ecological systems in which invasions need to be understood and managed.

Keywords

Invasion science, invasion biology, definitions

Something in the way they move

Hoffmann and Courchamp (2016) argue that human-mediated extra-range dispersal does not differ in terms of processes or mechanisms from natural colonisation, but that all dispersal events sit on a broad but continuous spectrum of species movements. Their rationale is that in both human-mediated extra-range dispersal and natural colonisations, populations have to overcome the same barriers (survival, reproduction, dispersal and further range expansion), and differ only in the "inconsequential" way in which they move from the original to the novel recipient locations (using their own means versus human transportation). Hoffmann and Courchamp (2016) suggest that there are four main reasons why scientists traditionally consider human-mediated extra-range dispersal and natural colonisations separately: (i) propagule pressure is greater for human-mediated extra-range dispersal; (ii) colonisation pressure is greater for human-mediated extra-range dispersal; (iii) genetic diversity is different; and (iv) human-mediated extra-range dispersal is more likely to result in invasions which lead to mass extinctions. They then argue that these differences are not clear-cut, and that even if such differences exist, they are differences of degree (e.g. rate or magnitude) rather than of kind. They conclude that human-mediated extra-range dispersal events "do not represent a distinctly different or change in process, just an acceleration of the colonisation process through multiple mechanisms".

We agree with Hoffmann and Courchamp (2016) that there is much to be learnt by invasion scientists from studying processes of natural colonisation (and vice versa; ecologists researching colonisation processes may learn from developments in invasion science). The same mathematical and theoretical models of dispersal and establishment can sometimes apply. For example, the concept of hierarchical filters for delineating pools of native species in studies of assemblages of natural communities resembles the concept and stages of the invasion pathway (Blackburn et al. 2011; Karger et al. 2016). In comparing when and where particular models are useful, and how parameter values differ, there can be useful insights for understanding and management.

Nevertheless, we disagree with the rest of their thesis—and note that the arguments made have already been well identified, characterised, and repeatedly rebutted (Cassey et al. 2005; Ricciardi 2007; Richardson and Ricciardi 2013). In particular, the dynamics and processes of dispersal leading to biological invasions are often quantitatively and qualitatively different from dispersal leading to natural colonisation (Wilson et al. 2009a; Wilson et al. 2009b). Wilson et al. (2009b) identified seven key properties of dispersal pathways: propagule pressure, genetic diversity, potential for simultaneous movement of coevolved species, selectivity of what is moved, the duration of the dispersal opportunities, evolutionary distance (time since divergence) between species in the original and new ranges, and the level of human assistance provided in spread and establishment. Hoffmann and Courchamp (2016) examined three of these, but all aspects are important (and there may be others). By focusing on the properties of different types of dispersal it becomes clear that human-mediated extra-range dispersal often varies very substantially in both kind and degree from natural colonisation. There is something in the way humans move species that moves them like no others.

Here, there and everywhere

In some cases, natural and human-mediated extra-range dispersal are qualitatively similar. Hoffmann and Courchamp (2016) provide three examples of this (tsunamis, range-shifts due to climate changes, and Lessepsian migration). Wilson et al. (2009b) categorised such dispersal events as extreme long-distance dispersal, leading-edge dispersal, and corridor respectively. However, there are other types of dispersal that simply never happened before humans evolved and started moving around the world, termed mass dispersal and cultivation (Fig. 1, Wilson et al. 2009b). These pathways are now major drivers of invasions. Such dispersal often results in the movement of massive numbers of individuals of species, that never would have dispersed naturally, to locations where they are provided substantial resources that facilitate establishment and invasion.

Hoffmann and Courchamp (2016) use examples of the colonisation of volcanic islands to argue that organisms move regardless of humans. This is true but disingenuous. Most individuals of most animal species move some distance in their lifetimes, but those distances are constrained at a range of scales. Thus, we can study activity schedules, home ranges, migration routes, and geographic ranges as more or less real entities. At the broadest of these scales, the presence of biogeographic regions shows that there are fundamental barriers to the spread of species that are not normally breached even over evolutionary timescales. If dispersal was not limiting why can we distinguish Gondwanan and Laurasian taxa? The existence of examples like the Great American Interchange (cited by Hoffmann and Courchamp (2016)) simply serve to highlight how rare are major faunal exchanges across such barriers. Biogeographic breaks are hugely important. Some groups might be less restricted by biogeographical features (particularly those groups that can form part of the aerial plankton), but other groups (in particular soil organisms) can be profoundly affected, with the resulting biogeographical breaks hard to distinguish. This is why species that cross such biogeographical breaks (either naturally or through human-mediated means) can occasionally have profound impacts.

The transportation of alien species by human agency across biogeographic barriers that have never historically been crossed before is essentially a daily occurrence now (Seebens et al. 2016). No passerine birds of European origin had colonised New Zealand over the tens of millions of years of the archipelago's independent existence, but now there is a thriving assemblage of such species that has developed in New Zealand over the last 150 years thanks to human intervention (Duncan, Blackburn & Cassey 2006). The pond slider (*Trachemys scripta*) is native to the southeast of the U.S.A. and Mexico. The maximum recorded dispersal distance of nesting females is 1.4 km (Steen et al. 2012; Garcia-Diaz et al. 2015). Yet, since the 1960s, the species has been introduced to 77 countries around the world, establishing self-sustaining populations in 36 countries (i.e. a dispersal distance of ~20,000km). Consider the alien fauna and flora of the region where you live. Which, if any, of these organisms could have arrived by natural colonisation? Are these simply differences in rates? Notwithstanding events like the Tohoku tsunami, to view human-mediated extra-range dispersal as simply a difference of degree from natural colonisation is to stretch the concept of degree beyond breaking point.

Figure 1. Human-mediated dispersal and natural colonisation: are they that different? **A** Stonehenge and **B** a rocky shore were both created by rolling stones, but they are quite different in origin and these differences are important. **A** is courtesy of Diego Delso, under the CC BY-SA 4.0 licence, https://commons. wikimedia.org/w/index.php?curid=35323153); **B** is courtesy of Tim Blackburn.

Even in instances where the differences between natural colonisation and human-mediated extra-range dispersal appear qualitatively similar, the degree can be important. Mass extinctions are "just" extinctions that occur at a higher rate; conservation biology is really only the population biology of species with small or declining populations; epidemiology is "just" the population biology of disease-causing organisms; medicine is "just" the diagnosis, treatment, and prevention of disease in one particular primate species. Are medical doctors basically specialised vets? That they are not is because differences in degree have important implications for the causes and consequences of the processes under investigation. For example, small populations are affected by stochastic events in ways that large populations are not, justifying the distinction between conservation and population biology. Differences of degree also matter because natural systems are frequently non-linear, such that increases in some parameters can lead to step changes in their responses. This is why we worry about humanity's contribution to atmospheric CO_2, even though this is a natural (and naturally varying) component of the atmosphere, and the concentrations of CO_2 in the atmosphere are well within the levels seen over geological time scales. As a further example, Gaston et al. (2003) compared natural and alien colonisations to Gough Island. Gough has accumulated 28

indigenous pterygote insect species over its 2–3 million year existence, but a further 71 pterygote species have been added to its insect fauna in the 325 years since humans first landed. Thus, the rate of accumulation on Gough Island has increased by 2–3 orders of magnitude as a result of human transport. This does not represent an acceleration, but rather a step change in species accumulation, akin to the difference between rates of background and mass extinctions (Pimm et al. 2014). Degree matters.

Finally, one of the main reasons such types of dispersal need to be distinguished from natural colonisation is what happens post-arrival. While conceptually the same barriers are present, the resources provided for establishment mean that some barriers are rendered inconsequential. How and where individuals arrive matters a great deal. For example, every year dozens of geese, ducks, raptors, rails, gulls, terns, pigeons, cuckoos, shorebirds, flycatchers, vireos, thrushes, warblers, sparrows, orioles, and other North American bird species arrive in the UK to the immense excitement of bird-watchers. Yet, since naturalists recognised the phenomenon in the early 19th century, none of these species has colonised and established permanent populations in the UK. In contrast, over the same period, the UK has gained well-established breeding populations of at least two North American species (Canada goose *Branta canadensis* and ruddy duck *Oxyura jamaicensis*) as a result of deliberate introductions. The largely stochastic and widely distributed arrival of small numbers of (probably exhausted) birds is extremely unlikely to have the same establishment outcome as concentrated and oftentimes intentional introductions of large numbers of well-provisioned individuals.

All down the line

An important emerging lesson in invasion science is that the manner by which species are introduced has long-lasting consequences on invasion trajectories (Donaldson et al. 2014). The invasion process (progression along the introduction-naturalisation-invasion continuum) is different for organisms introduced by humans to the processes associated with establishment and colonisation of organisms that arrive without human assistance (Hulme et al. 2016). Invasions differ from natural colonisation in biogeographical, ecological and anthropogenic dimensions (Rejmánek 2000), and historically too little research has focussed on how species are moved around (Puth and Post 2005). This is changing, and there has been a recent focus on introduction pathways (Essl et al. 2015a; Cope et al. 2016; Faulkner et al. 2016; Ricciardi 2016; Seebens et al. 2016). Moreover, species that have arrived in a new region through human-mediated extra-range dispersal or through natural colonisation can, of course, also co-opt the same dispersal pathways once in a region. This has been acknowledged many times before (e.g. Richardson and Pyšek 2006; Hulme et al. 2008) and is an important part of the reasoning behind the unified framework for biological invasions proposed by Blackburn et al. (2011). This is why there is an important distinction between transport and introduction in this framework, a distinction which is lost in Hoffmann and Courchamp's unhelpful edits to it.

The terminology of biological invasions that was proposed for plants in 2000 and generalised across taxa a decade later (Richardson et al. 2000; Blackburn et al. 2011) has been accepted by the majority of researchers because it is useful. The scheme has provided the basis for several recent large scale syntheses of the macroecology of invasions in a variety taxonomic groups (e.g. Essl et al. 2015b for bryophytes, Capinha et al. 2015 for gastropods, and van Kleunen et al. 2015 for plants). Such applications highlight strengths and weaknesses, and we welcome such tests of the scheme. As noted by Hoffmann and Courchamp (2016), there is more work to be done on the coding of the different stages proposed in the scheme by (Blackburn et al. 2011), and in particular we need recommendations on how to apply it in practice (Wilson et al. 2014). However, by failing to appreciate the importance of introduction dynamics, the revisions proposed by Hoffmann and Courchamp (2016) reduce the scheme's general applicability rather than increase it.

Hoffmann and Courchamp (2016) argue for more work on impacts, and we strongly support this call. Despite recent efforts to provide robust insights (Vilà et al. 2011; Pyšek et al. 2012), data on impacts are rare (be they by native or alien species). More information is urgently needed both observational and experimental (Kumschick et al. 2015). However, what data there are strongly suggest that natives are significantly less likely than aliens to be problematic for local ecosystems (Simberloff et al. 2012; Paolucci, MacIsaac & Ricciardi 2013; Buckley and Catford 2016), and that aliens can be extremely problematic. Hoffmann & Courchamp's suggestion that "with the (dramatic) exception of a few mammals, ants and pathogens,…there is little evidence that exotic species induce species extinctions" flies in the face of the abundant evidence that aliens are a major driver of native species extinction, including alien molluscs, fish and reptiles (Pyšek et al. 2016) – aliens have been the major cause of vertebrate extinctions over the last 500 years (Bellard, Cassey & Blackburn 2016). Even were that not the case, population-level declines (see Pyšek et al. 2016 for examples), introgression and losses of genetic diversity (Munoz-Fuentes et al. 2007), and the loss of community-level identity (i.e. homogenisation; Lockwood and McKinney 2001) are all crucial, and increasingly well documented, impacts of biological invasions. Impact should not be measured solely by species-level extinctions, but by a suite of measure of impacts on people, places and biodiversity. Standardised schemes for categorising environmental impact designed for invasive species (Blackburn et al. 2014) can potentially be adapted for native species, and proposed schemes to classify socio-economic impacts of alien taxa hold much promise for conservation more generally. But understanding where a taxon has come from, and in particular whether it is invasive or not, is often essential to understanding why these impacts occur and how they can be managed.

There are some excellent schemes that provided a basis for how to determine if invasive taxa are different from other taxa (van Kleunen et al. 2010), but in our view, efforts to partition off the "real biological" signal from the influence of humans is not only impractical, but at heart fails to recognise that invasions are intrinsically a human

product. Understanding the processes and mechanisms of biological invasions requires not just ecological studies, but an understanding of how humans move organisms to their new ranges, how they interact with them on arrival, and how they move them around their new ranges. This is why invasion biology has grown and developed into invasion science (Richardson 2011).

No expectations?

Hoffmann and Courchamp (2016) make an attempt at *reductio ad absurdum* with the question: are humans an "invasive" species or not. We agree that this discussion is mostly extraneous semantics, albeit one that can lead to some very disturbing conclusions, but it highlights the essential problem. We cannot extricate humans from invasions or invasions from humans. Invasions provide not only valuable test cases for ecologists, evolutionary biologists, and physiologists, but also important insights for our understanding of humans and their interactions with the environment. Hoffmann and Courchamp (2016) argue that if we want to understand the ecological process of dispersal, then invasion ecology should do more to productively engage with scientists in other fields. That is exactly what invasion ecologists do (in fact most of us learnt our trades in other fields before turning to invasion science). There are plenty of examples of transdisciplinary research on biological invasions produced by collaborations between invasions ecologists and social scientists, economists and evolutionary ecologists, decision scientists and mathematical biologists (Lockwood, Hoopes & Marchetti 2013). These collaborations only serve to highlight the pivotal role of humans.

Neither can we observe any evidence within the field of invasion science of the isolation of researchers working on different taxa. This may have been true twenty years ago, but the last decade has seen rapid development, as data from a broader range of taxa and standardised analytical and conceptual frameworks became available (Hulme et al. 2008; Walther et al. 2009; Blackburn et al. 2014; Essl et al. 2015a). The result has been a series of multi-author collaborations comparing invasion patterns in multiple taxa (Lockwood, Cassey & Blackburn 2005; Pyšek et al. 2010; Vilà et al. 2010; Essl et al. 2011; Aronson et al. 2014; Kumschick et al. 2015).

To conclude, biological invasions and natural colonisations are very often different; sometimes this matters, sometimes it does not. We should clearly focus more on processes and mechanisms, but the null expectation should be that biological invasions are qualitatively and quantitatively different from natural colonisation. Indeed, that is why we are moving from a Holocene period characterised by biogeographic regions with a rich global texture of unique and distinctive biotas, into an Anthropocene characterised by homogenisation, extinction and other massive global changes (Lewis and Maslin 2015). If future civilisations will be able to recognise this change in the geological record, we should be able to recognise it while it goes on around us each and every day of the year.

Acknowledgements

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COMMENTARY

When similarities matter more than differences: a reply to Wilson et al.

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Abstract

In our recent Discussion paper, we presented our view that the only real distinction between biological invasions and natural colonisations is the human element. We agree that invasion science is a very important science, not only to better understand the role that human mediation plays for colonisation, but also for many other science fields. We agree with all invasion researchers that the human influence can result in spectacular differences, including in rates of species movement, rates of successful colonisation, the particular species being moved, the biogeography of dispersal pathways and rates of any resulting ecological disturbance and biodiversity loss. Our deep point is that that species dispersed by human-mediation or natural colonisation are all subject to the same basic laws and rules of ecology, identical to many other phenomenon that occur naturally and can be greatly influenced by people. The human dimension is merely a mechanistic distinction, albeit important because it exposes insights about the colonisation process that cannot be seen by the study of natural colonisations alone. We provide 10 hypotheses that can be scientifically tested to determine whether biological invasions and natural colonisations are two separate processes or the same process being influenced by different mechanisms.

Keywords

Humans have a huge influence on many phenomena

In our recent paper (Hoffmann and Courchamp 2016) we point out at that despite many obvious differences between human-mediated biological invasions and natural colonisation, there are also many similarities. Ultimately, yes, they have differences, no one denies that, but are they THAT different? In a reply, Wilson et al (2016) have focused on the differences to argue the point that they are distinct, concluding biological invasions and natural colonisations are very often different; sometimes this matters, sometimes it does not. Here we respond to their arguments, ultimately concluding that biological invasions and natural colonisations are very often similar; sometimes this matters, sometimes it does not. In doing so, we further reinforce that the concepts in invasion ecology should not be artificially dissociated from general ecology.

The deep intent of our original paper was to argue that species dispersed by human-mediation or by natural colonisation are all subject to the same laws of ecology and that overlooking these similarities could slow down scientific progresses in invasion biology. As succinctly put by Lawton (1999) "The most useful scientific laws yield deep insights into the workings of nature: rules are less grand. Mechanisms are weaker still: a mechanism or mechanisms can generate unique phenomena: but general rules require common mechanisms". Undoubtedly the mechanism of human-mediated dispersal generates the unique phenomenon of modern biological invasions, but species dispersed by the two transport modes are subject to the same ecological laws and rules. To cite Jeschke (2014) "a look at the concepts and hypotheses in invasion ecology reveals that these are deeply connected with those in general ecology." The two fields have essentially the same hypotheses but with nuanced terminology, (eg invader vs coloniser). Their respective hypotheses, when boiled down to basics, are identical. Would anybody dare argue that hypotheses of propagule pressure, biotic resistance and enemy release only apply to species dispersed by human mediation but not natural colonisation? The fields and their hypotheses are not THAT different, because they focus on the same processes subject to the same biological laws and rules within a general theory of ecology (Scheiner and Willig 2008).

We cannot understate how important research about biological invasions have been to understanding colonisation, because it adds the other side of the continuum that has been studied by island biogeography (MacArthur and Wilson 1967). The commonalities between invasion science and general ecology have long been noted by many (Thompson et al.1995; Davis et al. 2001; Economo and Sarnat 2012). We agree that the human dimensions of biological invasions are pivotal to invasion biology, with clear implications for management and policy. But we are arguing that human-mediation is a mechanism, and thus essentially a biological invasion is a sub-class of the process of colonisation. Equally, natural colonisation is also sub-class of colonisation, and both are subject to the same natural laws.

We agree with Wilson et al. (2016) that scientific testing of hypotheses should determine whether colonisation and biological invasions are truly unique, or just the same overall process initiated by different mechanisms. Naturally there would be huge differences if the focus was at the level of mechanisms such as pathways (i.e. propagule

pressure for most species is undoubtedly statistically significantly different between human-mediated dispersal and self-dispersal). We argue that this is not the correct level for a fundamental scientific distinction, but instead the difference should lie much deeper at the level of laws and rules. As a suggestion of how to test this, we refer to the first 10 hypotheses of general ecology explored in Prins and Gordon (2014). Although specifically written for biological invasions, if the terms for biological invasions and natural colonisations were interchanged in each hypothesis, the hypotheses would be identical for both invasion biology and general ecology.

Hypothesis 1: A species will not be able to invade/colonise an area that has abiotic conditions that are outside its physiological tolerance levels.

Hypothesis 2: The extent of an invasion/colonisation is negatively correlated to species diversity of functional guild competitors in the invaded environment.

Hypothesis 3: An invasive/colonising species will not be able to replace a native species if they occupy the same niche and are in all other ways equal.

Hypothesis 4: A species will not be able to invade/colonise an area that harbours pathogens (that cause disease) or predators (that prey on the invading species) that it has not encountered before.

Hypothesis 5: A species will not be able to invade/colonise an area if its co-evolutionary species (necessary for parts of the invader's life cycle) is/are not present in the area.

Hypothesis 6: Species that occur at low population densities in their native range will not be invasive/prone to colonise.

Hypothesis 7: A species will not be able to invade/colonise an area if it has a lower use efficiency of its limiting resource than a native species that occupies the same location.

Hypothesis 8: Species can more easily invade/colonise highly disturbed areas; this disturbance can be either man-made or natural.

Hypothesis 9: Species from older lineages are more vulnerable to being replaced by invasive/colonising species that occupy a similar niche.

Hypothesis 10: A species will only be able to invade/colonise an area if it has a life-history strategy which is more r-selected (or 'weedy') that that of the species which already is occupying the niche.

The testing of the hypotheses for this purpose is not to determine if individual hypotheses should be rejected or not (for an extremely interesting insight into this refer to Prins and Gordon 2014), but instead to determine if there is a difference in the rejection of the null hypothesis for individual hypotheses between the two species movements. If the pattern of acceptance/rejection of these hypotheses is the same between species dispersed by human-mediation vs self-dispersal, then we would have to conclude that species dispersing by the two transport modes are subject to the same process with the same underlying ecological laws.

We agree with Wilson et al. (2016) on the differences of mechanisms (including the degree [rate] at which they occur), as well as their implications, especially for management and policy. But absolutely nowhere have Wilson et al. (2016) demonstrated that the two are subject to a different process. Specifically they give an example

about an apparent difference between the creation of the stones at Stonehenge and the smooth pebbles on a rocky shore. One is obviously human-mediated, the other natural. We agree that the mechanism(s) that gave rise to the rocks being modified into other shapes and configurations is different, but we argue that the underlying laws dictating the erosion process (e.g. laws of thermodynamics and the physics of friction) are identical for both. Indeed humans could create a beach of round pebbles, and there is absolutely nothing in the Laws of Thermodynamics preventing Stonehenge from naturally forming, but the probability (and hence rate) of it occurring is extremely low.

An important point that we want to make clear again is that we by no means are saying that invasion science is redundant. It is a vital science focused on species that are dispersed by people, and forms part of a family of sciences around biology and ecology with wide-ranging linkages with many other sciences including epidemiology and conservation sciences. This seems to be repeatedly lost in arguments by Wilson et al. (2016). For example we fully recognise the fields of conservation biology and population biology. But rather than seeing a black-and-white distinction based on differences of susceptibility of populations to stochastic events relative to population sizes, we instead see that this demonstrates that population sizes follow a continuum, which partly determines population's relative susceptibility to stochastic events. Our argument is one of a holistic view that whether a population is studied by either science, the process(es) underlying susceptibility to a stochastic event are the same, all that differs is the degree to which a population is susceptible.

Wilson et al. (2016) presented numerous arguments to demonstrate that the magnitude of differences of the two transport modes is an important science distinction, particularly noting that differences are often not just a linear change in rate, but a "step change". The influence of people on climate change was also provided as an example of such a "step change", implying some important science distinction. We find the climate change argument presented by Wilson et al. (2016) is instead very pertinent to our argument because of the identical nature of it with the holistic view of colonisation (Figure 1). Are the authors suggesting that the processes that influence global climates will change differently if the CO_2 is naturally released or human induced? Climate scientists agree that increasing CO_2 levels in the atmosphere is resulting in climate change. Climate scientists agree that there are human-mediated and natural contributions to global atmospheric CO_2. Much climate science also focuses solely on the human-mediated component. Despite there being big differences between the rate of CO_2 release, the mechanisms by which CO_2 is released, and the geography of CO_2 release, for the two modes of atmospheric CO_2 accumulation no climate scientist would argue that there is some fundamental difference in implications for projected climate changes due to the human-mediated or natural source of CO_2. Certainly, whilst there are marked implications for management of the different sources of emissions, none are advocating for a separate science or that there are different processes at play. Similarly, if a significant volcanic eruption somehow induced an atmospheric CO_2 step-change, are Wilson et al. (2016) arguing that the laws of chemistry and thermodynamics would influence the climate differently to an identical human-induced step-change? Ultimately, for all of their examples claiming that rates and degree differ, not a single difference in underlying process was given.
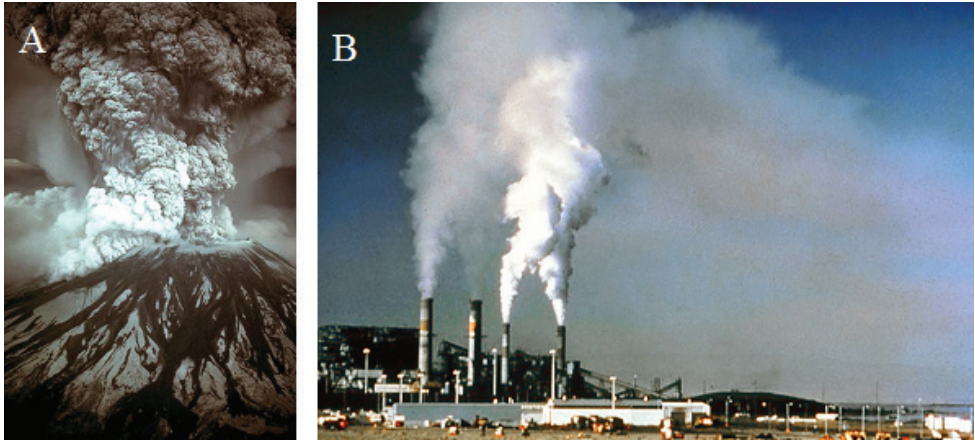
Figure 1. The process of climate change can be influenced by atmospheric composition change from natural sources such as volcanoes **A** or human-mediated sources such as electricity power stations **B** Clearly the two mechanisms of gas release have many significant differences, including the origins of the gases, nuanced sciences, rates, and policy implications, but their influence on the process of climate change is governed by identical laws of chemistry and thermodynamics. When considered holistically in the context of climate change, the two mechanisms are not THAT different, and are not two separate processes. **A** is courtesy of the U.S. Geological Survey, and **B** is courtesy of the U.S. National Parks Service.

We restrict comment about impacts here, just as we did previously, because impacts are not part of the colonisation process, but can be a consequence of it. For this reason, impacts were also not included in Blackburn's framework (Blackburn et al. 2011). We agree that some invasive species have created much extinction, and have done so spectacularly. The intent of our text was twofold: first to point out that most species moved by people have not been unambiguously shown by science to induce extinction. Indeed, only about 30 taxa are implicated with extinctions (Bellard et al. 2016), and therefore extinction is an exception, not a rule. Second, extinction as a result of colonisation is historically not confined just to species dispersed by people (Bellard et al. 2016), and as also stated by Wilson et al. (2016) nor is it likely "to be just a modern phenomenon".

We agree with Wilson et al. (2016) that, "at the heart of invasion science is the realisation that … the human dimension of invasions is a fundamental component in the social-ecological systems in which invasions need to be understood and managed." Naturally, without the human component there could be no human-mediated dispersal. But that is as deep as it gets. Without denying the pivotal role of humans in invasion biology, one must keep in mind that the biological, or ecological, component of it is at least, if not more, important. Even Wilson et al. (2016) wrote "While conceptually the same barriers are present, the resources provided for establishment mean that some barriers are rendered inconsequential." We couldn't agree more. Humans override some barriers by overcoming the limitation of the process that affect colonisation - species self-dispersing have to overcome the same barriers by themselves. Invasion science asks the focused question, how do people influence colonisation, and how do those influences change colonisation outcomes?

Ultimately, despite the statement by Wilson et al. (2016) "we strongly believe that the answer is yes: biological invasions and natural colonisations differ in processes and mechanisms" not a single process was presented, there or elsewhere. Let science be the judge. If there is indeed some deep and fundamental science difference between species undergoing range expansion by the two transport modes, then an underlying biological law needs to be found that applies to only one. To conclude along the same lines as Wilson et al. (2016), we believe that future civilisations will recognise from the geological record that at the turn of the Anthropocene, the major mechanisms driving the colonisation process for species globally changed from those that are "natural" and largely random and slow, to those that are driven by a dominant species (humans) and are largely non-random and fast.

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CORRIGENDA

Corrigendum:
A subcontinental view of forest plant invasions

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The authors inserted an incorrect figure in Oswalt et al. (2015) that was printed as Fig. 2. The mapped species represented in Oswalt et al. (2015) is *Triadica sebifera* or Chinese tallow. The correct Fig. 2, representing *Imperata cylindrica*, is reproduced below. The correction does not alter the conclusions of Oswalt et al. (2015).
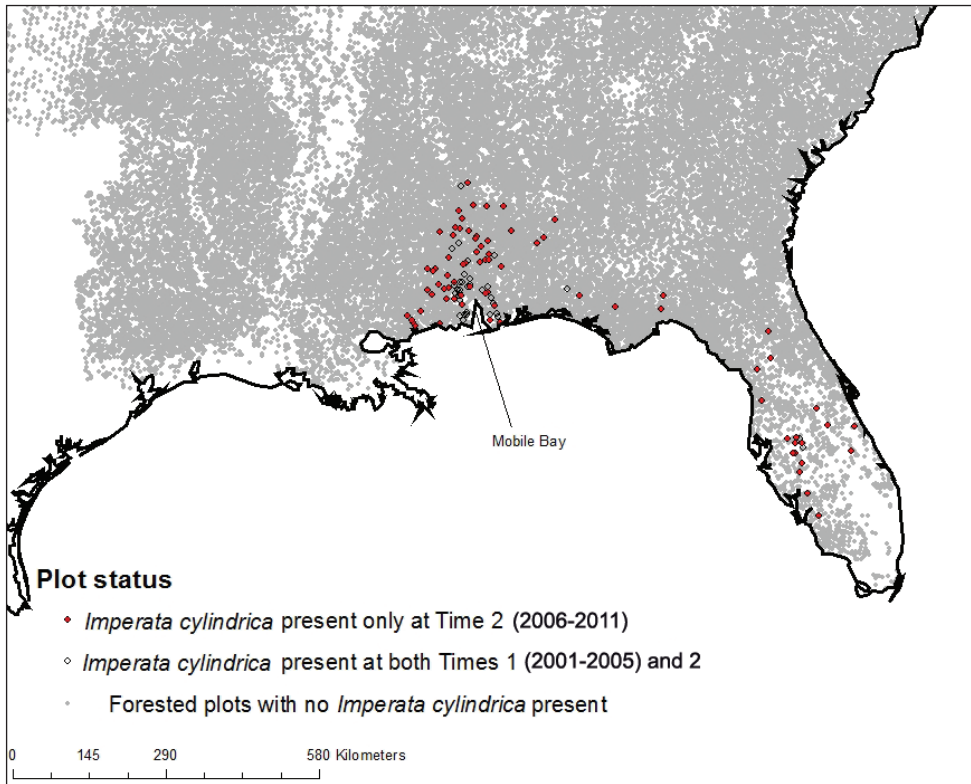
Figure 2. Spatial progression of *Imperata cylindrica* based on multiple measurements of forested plots from the FIA sampling grid. Time 1 represents data collected from 2001 to 2005 and Time 2 from 2006 to 2011.

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Oswalt CM, Fei S, Guo Q, Iannone III BV, Oswalt SN, Pijanowski BC, Potter KM (2015) A subcontinental view of forest plant invasions. NeoBiota 24: 49–54. doi: 10.3897/neobiota.24.8378