

Estimating the financial costs of freshwater invasive species in Great Britain: a standardized approach to invasive species costing

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Abstract Both ecological and economic impacts factor into invasive alien species (IAS) management considerations; however, economic impacts are often difficult to assess, much less quantify. Studies frequently aggregate identified financial costs as a proxy for IAS economic impacts, but these aggregate figures are often generated in an ad hoc fashion. Such estimates typically sum disparate costs, which might vary with respect to precision, accuracy, and scope. A standardized approach for IAS costing would better enable the comparison of cost estimates between taxa and across studies by controlling for surveying and scaling inconsistencies. This study develops a simple, survey-based approach to generate economic cost estimates for non-native freshwater invasive species (FIS) in Great Britain. The approach scales an average cost for each species by a ratio of management effort, thereby estimating the actual, annual expenditures incurred by a variety of stakeholders. The Great Britain-wide cost of controlling FIS is

estimated to be approximately £26.5 million year⁻¹; however, the costs of control could total £43.5 million year⁻¹ if management efforts were undertaken at all FIS infested locations. Cost estimates are highest for Canadian pondweed (*Elodea canadensis*), a particularly widespread species, and for the zebra mussel (*Dreissena polymorpha*), which adversely impacts both industrial water users and boaters. This assessment of the relative economic impacts between species provides policy-makers with a monetary basis for rank-ordering species' economic impacts and prioritizing management efforts. In addition, the cost assessment approach developed in this study could serve as a model for IAS economic impact assessments elsewhere.

Keywords Invasive alien species · Nonindigenous species · Economic impacts · Environmental management · Zebra mussel

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Introduction

Invasive alien species economic costing

Invasive alien species (IAS) disrupt native ecosystems and adversely impact economic interests the world over, necessitating effective management policies (Keller et al. 2008). Although much work remains to be done (Vilà et al. 2009), ecologists have

made significant progress detailing IAS threats to native ecosystems (e.g. Cohen et al. 1995; Manchester and Bullock 2000; Miller and Crowl 2006). IAS economic impacts are often harder to discern but no less critical for environmental managers. Monetizing IAS economic impacts provides a standard metric for cost-benefit analyses, and studies which provide IAS monetary figures (e.g. Pimentel et al. 2000) are routinely cited by managers and ecologists alike seeking to justify management efforts (Cusack et al. 2009). Nevertheless, the body of research on IAS economic impacts remains small and varied, without a standardized approach to IAS costing (Gren 2008).

IAS rarely affect established economic markets directly, making the accurate monetization of lost total economic value difficult, if not impossible. Therefore, studies often present documented financial costs as proxies for underlying IAS economic impacts (Vilà et al. 2009), especially where species-related damages are difficult to quantify (e.g. O'Neill 1997; Park and Hushak 1999; Pimentel et al. 2000, 2005). Alternative IAS costing approaches exist but typically require specific information which might be difficult to ascertain, such as expected losses to resource markets (Colautti et al. 2006) or a separate valuation of affected ecosystem services (Vilà et al. 2009). In comparison, financial costs are easily identified (e.g. McConnachie et al. 2003) and provide immediate insight even when other valuation methods are feasible (e.g. Xu et al. 2006).

As related to IAS, financial costs refer to identified monetary expenditures or losses, including both management costs and monetary damages. Management costs are often directly observable and relate to IAS exclusion, containment, mitigation, and adaptation actions undertaken by affected stakeholders (Perrings 2002). Damages are here defined as lost yield and/or infrastructure repair costs. Such costs are often identified (e.g. Pimentel 2005) but often cannot be quantified directly, as is often the case attributing a percentage of lost yield to IAS. Both types of financial costs feature prominently in the IAS economic impact literature (Lovell et al. 2006), either as individually reported costs (e.g. Britton et al. 2009) or, increasingly, as constituents of aggregate cost figures. Prominent examples of the latter include the \$137 billion year⁻¹ cost figure for non-indigenous species in the United States (Pimentel et al. 2000) and a cost of €12 billion year⁻¹ for IAS in

Europe (Kettunen et al. 2008). These figures underscore the severity of IAS issues (Andreu et al. 2009), but concerns about cost aggregation cast doubt on the validity of using compiled estimates in IAS impact assessments (EPA 2005).

Three specific concerns exist regarding the use of financial costs as IAS impact proxies. First, costs are often scattered and of variable quality, frustrating compilation efforts (Kettunen et al. 2008; USOTA 1993). Some studies present multiple, precise costs (e.g. O'Neill 1997), whereas others present single, round figures, such as the \$1 billion year⁻¹ cost attributed to *Corbicula fluminea* in the United States (Isom 1986; USOTA 1993). Second, a tendency exists to aggregate damages and financial control costs in an ad hoc fashion, potentially confounding cost types (e.g. Pimentel 2005; Pimentel et al. 2000, 2005). Pimentel et al. (2000) combine fish damage cost estimates with human disease control costs in generating their overall cost figure. Fish control costs and disease damage costs are listed as unavailable but may exist, complicating comparisons between these groups. Other costs incorporated in the Pimentel et al. (2000) figure, like the aquatic weeds cost, include losses, damages, and control costs. Management expenditures and damage levels potentially vary inversely with respect to one another (Leung et al. 2002); therefore, combining the two for a given taxon in a single time period could double count overall costs. The third concern with the use of financial costs as IAS impact proxies stems from a difficulty accounting for lost non-use values (i.e. bequest, existence values), which are often quantifiable only through contingent valuation (Carson 2000).

This study develops a standardized methodology for cost assessing multiple IAS over a given geographic area. The application of this survey-based approach to estimate the financial costs of non-native freshwater invasive species (FIS) in Great Britain attempts to avoid the first two concerns outlined above. Although the generated FIS costs underestimate total economic impacts by excluding potentially lost non-use values, the cost estimates are directly comparable between taxa, owing to the uniformity of the cost estimation process. The estimation concept employed herein, scaling a species' unit cost by an affected area, is not new (Parker et al. 1999; Pimentel 2002); however the addition of a scaling factor based on observed management effort allows one to better

estimate actual IAS economic expenditures, accounting for the uneven distribution of IAS impacts to stakeholders.

Non-native freshwater invasive species in Great Britain

IAS have a long history of establishment in Great Britain, where they exact ecological and economic harm (Britton et al. 2009; Manchester and Bullock 2000). Of particular concern are FIS, non-native species which spend at least part of their lifecycle in inland waters (Keller et al. 2009). New freshwater species continue to establish in Great Britain (Britton and Brazier 2006; Holdich 1993) at an increasing rate (Keller et al. 2009). In addition to profoundly impacting indigenous freshwater biodiversity (Vilà et al. 2009), FIS cause unintended economic impacts to stakeholders ranging from anglers (Hickley and Chare 2004) to industrial surface water users (Aldridge et al. 2004). Most of these impacts concern the degradation of natural capital stocks or impediments to economic activities dependent on freshwater environments (EPPO 2007; Sheppard et al. 2005). Such economic impacts are presumed to be substantial in Great Britain, as they are elsewhere (Lovell et al. 2006; Pimentel 2005).

Although studies like Manchester and Bullock (2000) document British IAS ecological effects, FIS economic impacts in Great Britain remain poorly understood. Only one study attempts to document financial costs for a group of British invaders (Pimentel et al. 2001), and these costs reveal little about the extent of FIS impacts. Lacking a comprehensive study, British policy-makers occasionally speculate on IAS/FIS costs. The non-native, riparian weed *Fallopia japonica* has received considerable attention for blocking water access (Defra 2003, 2007; JNCC 2007; POST 2008; Wade et al. 2007), and a £1.56 billion *ex ante* control cost estimate for the species is heavily cited (e.g. Sheppard et al. 2005; Warren 2007). Cost estimates for actual FIS appear less certain. Defra (2008) estimates that managing the aquatic weeds *Crassula helmsii*, *Hydrocotyle ranunculoides*, and *Myriophyllum aquaticum* collectively costs about £3 million year⁻¹ (2006 GBP). *Ex ante* estimates for eradicating the species range slightly higher. Defra (2003) suggests that eradicating *C. helmsii* would cost between £1.8 million and £4

million, whereas eradicating *M. aquaticum* would cost between £900,000 and £3 million. Eradicating *Azolla filiculoides* is expected to cost £200,000 to £2 million (Defra 2003). Most FIS control projects are undertaken by individual localities (POST 2008), but such projects go largely unrecorded in the public domain (Shaw and Tanner 2008).

This study seeks to determine the annual financial costs associated with FIS in Great Britain, including both public and private management expenditures. In recognition that overall costs will vary with respect to management effort, this study differentiates between a taxon's total potential costs and actual incurred costs. The relationship between reported damages and stakeholder management costs is also investigated to determine which measure better represents FIS economic impacts.

Methods

Identification of prominent FIS in Great Britain

At least 117 non-indigenous freshwater species have established in Great Britain, but not all of them cause sufficient impacts to warrant designation as FIS (Keller et al. 2009). Seventeen FIS (Table 1) were identified owing to their inclusion on the “high impact aquatic alien species” list maintained by the United Kingdom Technical Advisory Group (UKTAG) on the Water Framework Directive (UKTAG 2004, 2008). The species were all freshwater taxa, as defined by Keller et al. (2009). Fourteen of the 31 “high impact” aquatic species listed by UKTAG (2008) were excluded from this study. Eight were either coastal or estuarine taxa. An additional four species, *F. japonica*, *Impatiens glandulifera*, *Heracleum mantegazzianum*, and *Rhododendron ponticum*, classified as riverine by the UKTAG (2008), were also excluded on account of their predominantly terrestrial life habit (Keller et al. 2009). Two additional species, *Dikerogammarus villosus* and *Dreissena bugensis*, have not yet been documented in Great Britain. A literature survey subsequently identified potential ecological and economic impacts associated with the FIS, including likely costs and possible benefits (Online Resource 1). Likely control or eradication methods were noted (Online Resource 2), which informed the stakeholder survey process (Online Resource 3).

Table 1 Identified freshwater invasive species in Great Britain

Name	Common name	Native range
<i>Azolla filiculoides</i>	Water fern	South America ^a
<i>Crassula helmsii</i>	Australian swamp stonecrop	Australia, New Zealand ^{b,c}
<i>Elodea canadensis</i>	Canadian pondweed	North America ^b
<i>Elodea nuttallii</i>	Nuttall's pondweed	North America ^b
<i>Hydrocotyle ranunculoides</i>	Floating pennywort	Cosmopolitan ^b ; North America ^d
<i>Lagarosiphon major</i>	Curly waterweed	South Africa ^e
<i>Ludwigia grandiflora</i>	Water primrose	Americas ^b
<i>Myriophyllum aquaticum</i>	Parrot's feather	South America ^b
<i>Dreissena polymorpha</i>	Zebra mussel	Ponto-Caspian ^f
<i>Crangonyx pseudogracilis</i>	(amphipod)	North America ^g
<i>Eriocheir sinensis</i>	Chinese mitten crab	East Asia ^h
<i>Hemimysis anomala</i>	Bloody red mysid	Ponto-Caspian ⁱ
<i>Pacifastacus leniusculus</i>	American signal crayfish	Washington State ^j
<i>Procambarus clarkii</i>	Red swamp crayfish	North America ⁱ
<i>Carassius auratus</i>	Goldfish	East Asia ^k
<i>Cyprinus carpio</i>	Common carp	China ^l
<i>Pseudorasbora parva</i>	Topmouth gudgeon	East Asia ^m

^a McConnachie et al. 2003; ^b Sheppard et al. 2005; ^c EPPO 2007; ^d EPPO 2006; ^e Gassman et al. 2006; ^f USGAO 2002; ^g Johanna et al. 2003; ^h Herborg et al. 2005; ⁱ Minchin and Rosenthal 2002; ^j Holdich 1993; ^k Gozlan 2008; ^l Kelly 2003; ^m Britton and Brazier 2006

Identification and distribution of British FIS stakeholders

British stakeholder groups were identified by their connection to marketed outputs from freshwater ecosystems and environments. Managed lakes, inland boat yards/marinas, aquaculturalists, freshwater fisheries, municipal water suppliers, hydroelectric power facilities, nuclear power facilities, industrial water users, public agencies (e.g. the Environment Agency (EA) and the Broads Authority), non-profit managers (e.g. Natural England and British Waterways), and public docks provided information on specific sites they manage. Other stakeholder groups without primary responsibility for individual sites provided supplemental information used to corroborate site-level feedback. These included environmental consultancies, academics, and pest removal contractors. 'Managed lakes' referred to recreational lakes not designated as fisheries. 'Aquaculture' referred to fish farms, some of which permitted angling, whereas 'fisheries' were those angling water bodies not involved in fish stock distribution. Municipal water sites included water treatment works, reservoirs, and pipelines. Industrial water users included a cement works, construction and development firms, and

industrial equipment suppliers. Environmental consultancies and pest removal contractors were asked about localities they serviced.

Stakeholders were identified from internet facility listings, directory searches, and conference proceedings (i.e. the Great Britain Non-native Species Secretariat Sixth Non-native Species Stakeholder Forum and the Norfolk Non-native Species Stakeholders' Forum). Identified stakeholders were stratified by stakeholder group. Individual stakeholders were randomly selected for surveying within each stratum. Surveyed localities, defined as discrete sites managed by an individual stakeholder, were distributed throughout England, Wales, and Scotland (Fig. 1).

Stakeholders were surveyed using a combination of mail questionnaires and follow-up telephone interviews. The stakeholders were asked about the presence of FIS and about ecological (i.e. changes in organism abundance, algal growth) and economic problems associated with the taxa. Stakeholders were asked to describe incurred expenditures if they monitor, control, or eradicate any FIS. In addition, stakeholders were asked to describe species' damages to their output, operations, or facilities and to estimate damage costs arising from lost yield or damage repair. Stakeholders were asked to provide

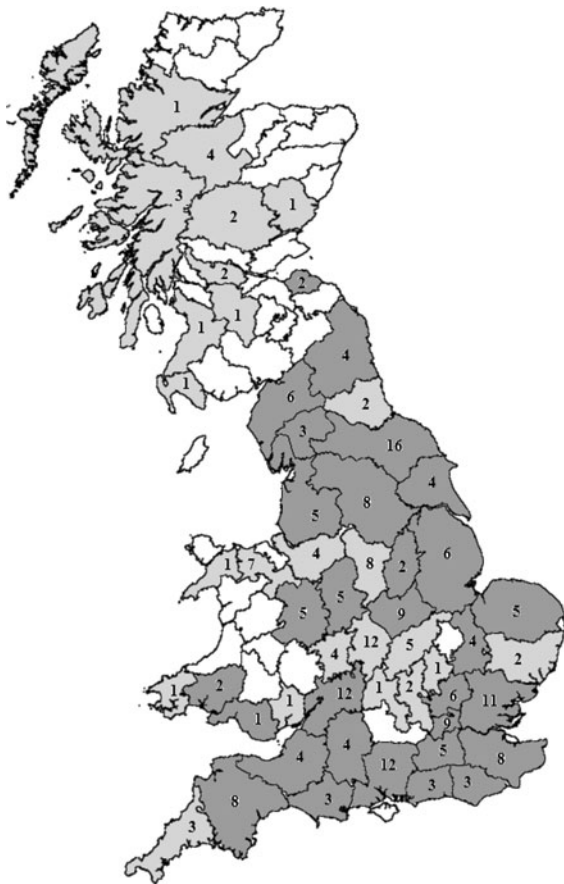


Fig. 1 The number of responding localities in Great Britain by county; *dark grey* counties include at least one locality reporting specific financial costs; *light grey* localities report the presence of FIS but do not report costs

information on when costs were incurred and the nature of steps taken to address FIS.

Estimation of FIS financial costs in Great Britain

The total financial cost of controlling each FIS taxon was estimated for Great Britain using a modification of the Parker et al. (1999) approach,

$$I = R \times A \times E \tag{1}$$

where *I* represented overall impact, *R* represented range size, *A* represented abundance, and *E* pertained to an effect/unit. This study substituted a species’ number of occurrences for *R*, and used an observed ratio of management effort at infested localities in place of *A* to scale the unit cost, *E*. Restated, this study employed the following two steps:

$$C = U \times N \tag{2}$$

C represented the estimated total, annual financial costs for a species. *U* represented an annual, unit cost per locality, which was determined by averaging the reported, non-zero stakeholder costs for a given species in a given year, and *N* represented a species’ number of occurrences.

Occurrence information for each species, excluding *Eriocheir sinensis* and *Pseudorasbora parva*, came from the largest single database available for each species on the National Biodiversity Network (NBN) Gateway (NBN 2009a, b, c, d, e, f, g, h). A compilation of four NBN datasets was necessary to obtain the 8 occurrences cited for *E. sinensis*. Occurrences for *P. parva* were obtained from the EA (Renals, pers. comm.). Scaling the two separate effects of *Dreissena polymorpha* required an exception to the outlined methodology. Boat occurrences (British Waterways 2008) and the number of affected water treatment works (Elliott 2005) were substituted for biogeographic occurrences. Equation 2 estimated control costs assuming simultaneous control of a taxon in all present localities in Great Britain. In reality, FIS are not managed everywhere they occur, so the Eq. 2 costs were scaled down to estimate actual, annual British control costs.

$$C = U \times N \times M \tag{3}$$

Equation 3 added a ratio of management effort, *M*, derived from the stakeholder survey response data. *M* represented the proportion of localities reporting a taxon which actually spent money controlling the taxon.

Results

Results of the British FIS stakeholder survey

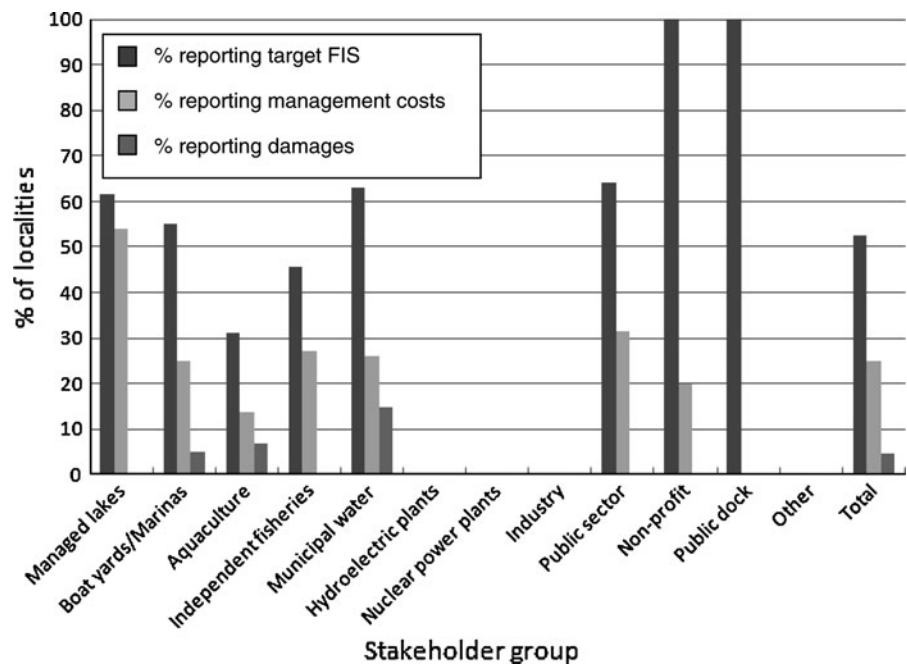
This study incorporates results from 245 different localities (Table 2; Fig. 1), representing a total response rate of 47.1% after follow-up telephone interviews with non-respondents augmented the sample size. Slightly more than half the responding localities report at least one FIS, and a quarter report FIS financial costs (Fig. 2, Online Resource 4). Of those, 61 localities report management costs (i.e. control, monitoring, or eradication expenditures), and

Table 2 Composition of survey respondents and breakdown of reported costs

	Response rate (%)	Final no. of localities	No. of localities reporting target AIS	No. of localities reporting costs	No. of localities reporting management costs	No. of localities reporting damages
Managed lakes	88.8	13	8	7	7	0
Boat yards/Marinas	25.9	20	11	5	5	1
Aquaculture	72.5	29	9	5	4	2
Separate fisheries	61.1	11	5	3	3	0
Municipal water	57.1	54	34	16	14	8
Hydroelectric plants	100	17	0	0	0	0
Nuclear facilities	100	9	0	0	0	0
Industry	20	NA*	0	0	0	0
Public sector	43.7	86	55	27	27	0
Non-profit	44.4	5	5	1	1	0
Public docks	100	1	1	0	0	0
Other	33.3	NA	1	0	0	0
Total	47.1	245	129	64	61	11

* No problems reported at the company level, so respondents did not discuss specific sites

Fig. 2 The percentage of responding localities reporting target FIS and associated costs by stakeholder group



11 report damages (i.e. lost yield or repair costs) (Table 2). Relatively few respondents report specific FIS ecological impacts. One boat yard reports macrophyte growth attributable to *D. polymorpha* water filtration, and one fish farm reports suspected *Pacifastacus leniusculus* impacts on native crayfish populations. Five public sector stakeholders and one

academic stakeholder also note changes in the relative abundance of taxa over time.

FIS occurrence varies according to stakeholder group (Fig. 2) and species (Fig. 3). Almost all of the publically managed localities report FIS, whereas no FIS are reported by hydroelectric plants, nuclear plants, or industry stakeholders (Fig. 2). With respect

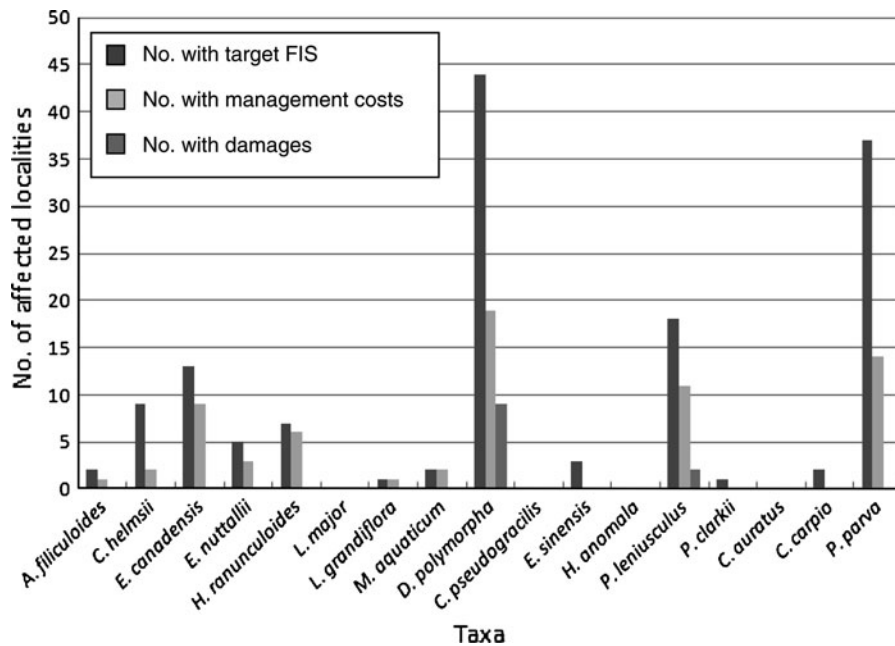


Fig. 3 The number of surveyed localities reporting FIS and associated costs by target species. Management effort ratios are found by comparing the number of facilities reporting a species

with the number of facilities reporting management costs for that species. Exceptions to this method are described for the cases of *L. major* and *D. polymorpha* in Tables 3 and 4

to individual taxa, *D. polymorpha* occurs at more localities (44) than any other species (Fig. 3). The majority of these occurrences (34) pertain to municipal water facilities. Public sector responses from monitored localities constitute all 37 *P. parva* reports, the entirety of the species’ known British distribution, and 11 of the 18 reported *P. leniusculus* occurrences. *Lagarosiphon major*, *Crangonyx pseudogracilis*, *Hemimysis anomala*, and *Carassius auratus* are not reported at any of the localities, although *L. major* might be misidentified as either *Elodea canadensis* or *Elodea nuttallii* (Pickup, Aquaclear Water Management, pers. comm.).

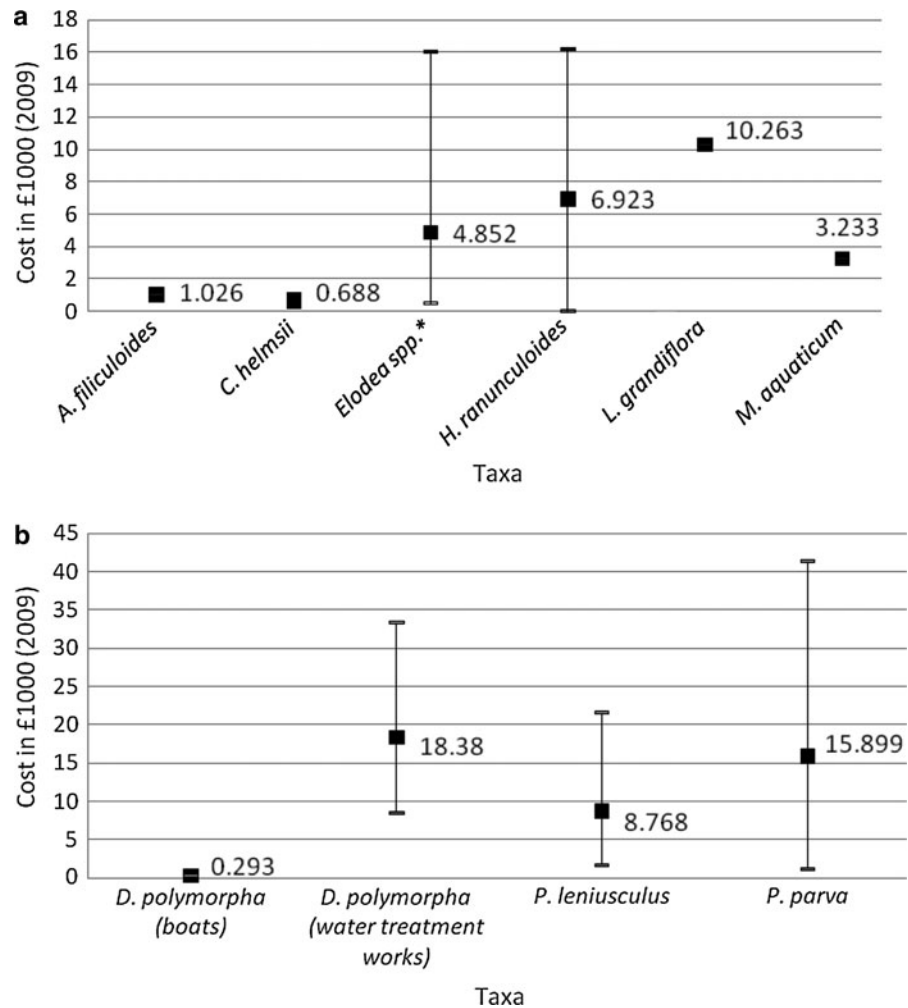
Where present, certain species are more likely to be managed than others (Fig. 3). Two taxa, *Ludwigia grandiflora* and *M. aquaticum*, are managed everywhere they are reported, yielding management effort ratios of 1. *Eriocheir sinensis*, *Procambarus clarkii*, and *Cyprinus carpio* occur at several localities, but no associated costs are reported for the taxa, yielding management effort ratios of zero (Fig. 3).

The range of reported costs also varies by species (Fig. 4a and b) and relates to control method (Online Resources 2 and 4), affecting the average cost localities spend on a given taxon. The cost for

A. filiculoides corresponds with weevil biocontrol. Both reported costs for *C. helmsii* pertain to chemical control. *Myriophyllum aquaticum* and *H. ranunculoides* are often removed manually. *Elodea canadensis* and *E. nuttallii* frequently occur together, and managers generally do not differentiate them during mechanical control, which is why the *Elodea* spp. expenditures are combined (Fig. 4a). Two sets of costs are included for *D. polymorpha* (Fig. 4b) because control techniques and expenditures are drastically different between boat yards/marinas and municipal water suppliers. The first set represents the range of reported annual costs for cleaning and sealing the hull of a mussel fouled boat. The second set represents the range of annual control expenditures at six water treatment works. The *P. leniusculus* and *P. parva* figures derive from EA managed localities, and costs include site consultation, monitoring, and control (Renals, EA, pers. comm.). No costs are reported for the remaining animals or for *L. major* (Fig. 3).

Reported damages are less common than management expenditures (Figs. 2 and 3). Damages are reported at seven municipal water facilities where *D. polymorpha* fouling in pipes increased energy costs, and at one water facility where mussels

Fig. 4 Reported average (labeled), minimum, and maximum annual financial costs for plant (a) and animal FIS (b) by localities; costs adjusted for inflation and reported as 2009 GBP (* *E. canadensis* and *E. nuttallii* costs combined)



damaged tanks and water pumps. One boat yard reports *D. polymorpha*-related boat engine repairs and lost boat fuel efficiency from hull fouling. Two fish farms report lost revenue attributable to *P. leniusculus*; the EA prohibits some fish stock transfers to prevent the spread of the invasive crayfish (Bull, Dunnington Trout Farm, pers. comm.).

Estimating total FIS financial costs of control in Great Britain

Employing Eqs. 2 and 3 yields a range of annual, estimated control costs for FIS in Great Britain. The financial cost of controlling the eight FIS plants in Great Britain amounts to £19.3 million year⁻¹ (Eq. 3) but could range as high as £29.2 million year⁻¹ (Eq. 2) (Table 3). Controlling *D. polymorpha*, *P. leniusculus*,

and *P. parva* is estimated to cost Great Britain £7.2 million year⁻¹ (Eq. 3) but could reach £14.3 million year⁻¹ (Eq. 2) (Table 4). *Elodea canadensis* is found to have the highest estimated control costs, followed by *D. polymorpha*. No financial costs appear to exist for the remaining animal FIS in Great Britain.

Discussion

The utility of the outlined approach for IAS management

The FIS cost extrapolation methodology developed and employed by this study represents a broadly applicable approach capable of providing a minimum estimate of economic impacts for a wide array of

Table 3 Determination of Great Britain-wide control costs for FIS plant taxa (costs in 2009 GBP)

	Control type	Avg. cost/ locality/year (Fig. 4a, £)	No. of occurrence records	Estimated max. control costs (£)	Management effort	Actual control cost estimate (£/year)	FIS Rank order by actual control estimate
<i>A. filiculoides</i>	Biocontrol ^a	1,026	867 ^h	889,542	0.5	444,771	7
	Mechanical control ^b	–	–	–	–	–	–
<i>C. helmsii</i>	Diquat control ^c	688	875 ^h	602,000	0.22	132,440	10
	Mechanical control	4,852	3477 ^h	16,870,404	0.69	11,640,579	1
<i>E. canadensis</i>	Mechanical control	4,852	1,501 ^h	7,282,852	0.6	4,369,711	3
<i>E. nuttallii</i>	Manual/chemical control ^d	6,923	63 ^h	436,149	0.86	375,088	8
	Mechanical control ^e	–	–	–	–	–	–
<i>L. major</i>	Mechanical control	4,852 ^g	403 ^h	1,955,356	0.6 ^g	1,173,214	5
	Dichlobenil ^f	–	–	–	–	–	–
<i>L. grandiflora</i>	Manual/chemical control ^d	10,263	“South Coast” ⁱ	10,263	1	10,263	11
<i>M. aquaticum</i>	Manual/chemical control ^d	3,233	350 ^h	1,131,550	1	1,131,550	6
	Total			29,178,116		19,277,616	

^a Weevil biocontrol unit cost provided by CABI (2009): £325/50 m²

^b Biocontrol has replaced mechanical control (Ackerman, Kingcombe Aquacare, pers. comm.); not reported by respondents

^c Use is now banned in the United Kingdom (Ackerman, Kingcombe Aquacare, pers. comm.); costs refer to recent reported cases

^d Control requires multiple treatment events (Ackerman, Kingcombe Aquacare, pers. comm.)

^e Recommended for invasions >1 ha (Ackerman, Kingcombe Aquacare, pers. comm.); not reported by respondents

^f Not reported by respondents

^g Often controlled with *Elodea* spp. (Pickup, Aquaclear Water Management, pers. comm.); assumes same avg. costs and management effort

^h Occurrence records obtained from the Vascular Plants Database (2009)

ⁱ Occurrence information courtesy of the Great Britain Non-native Species Secretariat (Booy, pers. comm.)

Table 4 Determination of Great Britain-wide control costs for FIS animal taxa (costs in 2009 GBP)

	Control type	Avg. cost/locality/year (Fig. 4b, £)	No. of occurrence records	Estimated max. control costs (£)	Management effort	Actual control cost estimate (£/year)	FIS rank order by actual control estimate
<i>D. polymorpha</i>	Boat washing, hull sealing	293 ^b	428 ^d	9,541,838 ^j	0.44 ^l	4,198,409	2
	Mechanical control	18,380 ^c		551,400 ^k	1	551,400	
<i>C. pseudogracilis</i>	Not controlled	NA	No data		0		
<i>E. sinensis</i>	Not controlled	NA	8 ^e		0		
<i>H. anomala</i>	Not controlled	NA	No data		0		
<i>P. leniusculus</i>	Pyblast	8,768	413 ^f	3,621,184	0.61	2,208,922	4
<i>P. clarkii</i>	Pyblast ^a	8768 ^a	"A few sites" ^g		0		
<i>C. auratus</i>	Not controlled	NA	259 ^h		0		
<i>C. carpio</i>	Not controlled	NA	444 ^h		0		
<i>P. parva</i>	Rotenone	15,899	37 ⁱ	588,263	0.38	223,540	9
Total				14,302,685		7,182,271	

^a Not reported but assumed to follow *P. leniusculus* control

^b Unit cost corresponds to one boat

^c Unit cost corresponds to a water treatment works

^d Occurrence records obtained from Mollusc (non-marine) data for Great Britain and Ireland (2009)

^e Occurrence records obtained from Invertebrate Site Register—England (2009), Seasearch Marine Surveys (2009), Volunteer sightings data held by DASSH Data Archive Centre (2009), Crayfish data update for Environment Agency Thames Region March 2009 (2009)

^f Occurrence records obtained from Crayfish (Crustacea: Astacura) data for Britain and Ireland to 2003 (2009)

^g *P. clarkii* shows a restricted British range (Environment Agency 2009)

^h Occurrence records obtained from the Database for the Atlas of Freshwater Fishes (2009)

ⁱ Occurrence records courtesy of the Environment Agency (Renals, pers. comm.)

^j Figure represents the unit cost multiplied by 32,566, the number of long term boat licenses in England and Wales (British Waterways 2008)

^k Unit cost (this study) multiplied by 30 affected facilities (Elliott 2005)

^l Management effort equals the percentage of affected boat yards/marinas surveyed which mention control

taxa. Application of this approach to British FIS indicates that 11 species each cost stakeholders in Great Britain several thousand to several million pounds annually. These financial costs are not necessarily a net economic cost to Great Britain, because the majority of control expenditures constitute revenue for individuals charged with species management. Nevertheless, the costs do highlight the severity of adverse FIS impacts. *D. polymorpha*, in particular, constitutes a huge nuisance to boat owners and water treatment works, and *Elodea* spp. constitutes a huge nuisance to private water body managers.

The results of this study can assist FIS managers, particularly regarding management focus. The EA currently includes seven freshwater species on its 10 “most wanted foreign species” list (EA 2006), *P. leniusculus*, *Mustela vison*, *H. ranunculoides*, *C. helmsii*, *E. sinensis*, *M. aquaticum*, and *P. parva*. This study concludes that *E. canadensis*, *D. polymorpha*, and *E. nuttallii* are the three most costly FIS in Great Britain, but they do not appear on the list. Conversely, *M. vison* is not listed as a FIS by UKTAG (2008), and only four localities mention the animal’s presence. It should be noted that widespread species, with potentially high estimated costs (i.e. *E. canadensis*), might not be as susceptible to control as localized invaders (i.e. *L. grandiflora*), possibly explaining omissions in the “most wanted” list. However, this study suggests that such widespread invaders are of greatest concern to private stakeholders. In contrast, six FIS are not considered nuisances by British stakeholders, despite their designation as such by the UKTAG (2008). *Carassius auratus* and *C. carpio* are widespread but largely ignored. *C. pseudogracilis* or *H. anomala* are not reported at all. These species are not necessarily benign (Online Resource 1), but an absence of financial costs for these “high impact” species suggests “low impact” non-indigenous species (e.g. *Lepomis gibbosus*, *Ctenopharyngodon idella*) are even less likely to cause economic impacts in Great Britain. Stakeholders do not view such species as pests.

In addition to directing management focus, survey-based cost estimates can justify control expenditures. Britain’s recently adopted Invasive Non-Native Species Framework Strategy (Defra 2008) signals a commitment to dealing with IAS/FIS, but cost-benefit analyses are still necessary to justify specific public control or eradication programs. Such analyses should

weigh anticipated control costs against existing private financial costs. Publically directed efforts to eradicate IAS, including the proposed *A. filiculoides* eradication (Defra 2003), would likely cost less and achieve more than the cumulative efforts of individual locality managers, who often cannot prevent IAS reestablishment. The cost scaling process employed in this study also has a potential use at the microeconomic level. Producing average, unit costs for individual IAS could benefit locality managers seeking perspective for their own management efforts.

The cost estimation process outlined in this study has broad possible applications outside Great Britain. Very little is known about IAS costs, public or private, in areas outside Europe, North America, and Australia. A stakeholder survey would be especially beneficial in these regions as a first attempt to document the magnitude of IAS impacts. In areas where significant work already exists, the process could be used to independently assess existing cost estimates, such as those produced for the Great Lakes and Hudson River system (Pimentel 2005).

Accounting for uncertainty and discrepancies in cost estimation

Despite its general applicability, the cost estimation approach employed herein highlights certain difficulties inherent to IAS economic impact estimation. Slight adjustments to the approach are necessary to generate cost estimates for *L. major* and *D. polymorpha*. Many of the *E. canadensis* reports might actually be *L. major* or *E. nuttallii*. *Dreissena polymorpha* causes very different impacts to two different stakeholder groups. Had the costs not been separated, the high municipal water costs would have been extrapolated over a wide range of localities with no relation to the water treatment industry. The approach outlined in this study is dependent on knowledgeable stakeholders and is most applicable to species with uniform impacts across all localities.

It is also important to reiterate that IAS financial costs underestimate lost total economic value and cannot account for those ecological impacts unrelated to monetary expenditures. The use of management costs as impact proxies admittedly excludes certain damage costs, lost option values, and lost non-use values. However, these values are often non-uniformly distributed and difficult for stakeholders to

estimate. Non-uniform costs cannot necessarily be extrapolated. The results yield few reported damage costs, and consideration of the reported damages alone would suggest that only two species cause economic impacts in Great Britain. These damages are excluded from the cost extrapolation in recognition that they might also account for a percentage of the reported management expenditures. For example, the lost fuel efficiencies attributed to *D. polymorpha* factor into boat hull cleaning decisions. In a given time period, stakeholders will either sustain the lost efficiencies or the cleaning costs but not both simultaneously.

Some ecological impacts are represented in the generated cost estimates, such as *P. parva* competition with native fish as a factor in removal costs, but many more ecological effects cannot be monetized. To this end, the emerging use of ecosystem services might better estimate overall IAS impacts. However, it is worth noting that the survey approach captures perceived costs, which factor heavily in public sentiment toward IAS. These cost perceptions cannot include ecological considerations if the stakeholders are unaware of IAS ecosystem alterations. Incomplete information might explain why so few respondents reference FIS ecological impacts or why *C. auratus* and *C. carpio* are effectively ignored by water body managers. Furthermore, if cyprinids are viewed by anglers as substitute goods for imperiled native fish, the loss of the latter might not register as an economic impact, despite the clear ecological effect.

Four assumptions related to methodological uncertainty should also be considered before the generated FIS cost estimates can inform policy. First, scaling the unit costs of control linearly by a number of occurrences assumes that economies of scale do not apply as management area increases. This assumption appears reasonable considering most occurrences relate to discrete localities (NBN 2009a, b, c, d, e, f, g, h). Second, scaling unit costs by a number of occurrences assumes control might be undertaken everywhere a species occurs. All localities could be publically managed if necessary. Anthropogenic dispersal also represents a primary vector for the distribution of these species, suggesting some level of human contact at all of the occurrence sites. A third key assumption involves the representativeness of the average costs generated by this study. Control costs for certain species (e.g. *A. filiculoides*) appear

uniform; however, control costs and methods vary for species such as *H. ranunculoides*. Finally, there are a wide variety of confounding variables related to differences in locality size, the site level abundance of FIS, the seasonal susceptibility of FIS to control, and subtle differences in operations between localities that are difficult to control but likely affect the data. In extrapolating average costs over a large geographic area, one assumes that these discrepancies balance out. Uncertainty cannot be completely eliminated from IAS costing, but Pimentel (2002) defends the outlined cost estimation approach, noting that IS impact data is rarely good enough to account for multivariate effects.

Despite these considerations, a correspondence between estimates generated by this study and accepted costs for certain taxa in Great Britain seems to support the employed methodology. This study estimates a combined cost of control between £1.6 and £2.2 million for *C. helmsii*, *H. ranunculoides*, and *M. aquaticum*, which corresponds with the £3 million year⁻¹ (2006 GBP) cost of control generated by Defra (2008) for those same species. This study also estimates that controlling *A. filiculoides* throughout Great Britain using biocontrol would cost approximately £890,000 year⁻¹, which is similar to the Defra (2003) £200,000 to £2 million estimate for eradicating the species.

Conclusions

Considering the ease with which financial costs can be identified, and their direct relation to underlying economic impacts, financial costs will likely remain a facet of IAS impact assessments. The cost scaling approach developed in this study provides a conservative and widely applicable model for future IAS economic impact studies by controlling for inconsistencies common to financial cost estimation. The uniform acquisition of costs and the addition of a management effort ratio enable the estimation of the actual IAS costs incurred within a given geographic area. IAS cost compilation concerns are addressed in large part by direct stakeholder surveying, and double counting concerns can be eliminated by focusing solely on management expenditures. The marginal contribution of damage costs to this study suggests that management expenditures alone can approximate

perceived IAS costs. Using the cost scaling method employed herein, managers can better justify both direct comparisons between generated costs and the rank-ordering of IAS economic impacts.

Financial costs provide perspective for IAS management, but other IAS impact assessments should be considered as well, specifically those pertaining to IAS ecological effects. Financial costs likely underestimate the lost total economic value attributable to IAS, although observed costs avoid problems with cost perception inherent to contingent valuation. An independent assessment of economic impacts, perhaps from an ecosystem services perspective, would help verify the accuracy of the financial cost estimates reported here for British FIS. Nevertheless, the conservative cost estimates generated by this study still suggest a large magnitude for the problems associated with FIS in Great Britain, particularly for the aquatic plants and for *D. polymorpha*. These results provide managers in Great Britain with a preliminary basis for prioritizing the management of these harmful species.

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