El coste económico de las especies exóticas invasoras y la base de datos INVACOST

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En este documento vamos a hacer una breve introducción a la base de datos InvaCost, la cual compila los costes económicos de especies invasoras a nivel global. Además, este documento viene acompañado por una serie de apéndices que incluyen publicaciones científicas que explican, utilizan o completan la base de datos.

1. **¿Qué es InvaCost y por qué surge?**

La base de datos InvaCost surge por la necesidad de sensibilizar y convencer al público en general y a los políticos y otros actores de la problemática causada por las invasiones biológicas. Hace unos años, Franck Courchamp, de la Universidad de París-Saclay, decidió compilar, en una base de datos, los costes económicos producidos por especies invasoras que se hubiesen documentado hasta el momento. Su objetivo era el de poder usar el dinero, una métrica común y fácil de entender, para comprender mejor las invasiones biológicas y sensibilizar a diferentes actores y sectores de la sociedad sobre los impactos causados por las especies invasoras y la necesidad de gestionarlas.

→ Apéndice 1. Diagne C, Leroy B, Gozlan RE, Vaissière A-C, Assailly C, Nuninger L, Roiz D, Jourdain F, Jarić I, Courchamp F (2020) InvaCost, a public database of the economic costs of biological invasions worldwide. Scientific Data 7: e277. https://doi.org/10.1038/s41597- 020-00586-z

2. **Necesidad de una revisión global de los costes económicos: la importancia de los costes en lenguas no inglesas**

En el documento del Apéndice 1 (Diagne et al. 2020) se explica cómo se creó la primera versión de InvaCost. Para comenzar, se realizó una búsqueda estandarizada usando palabras clave en inglés (por ejemplo, especie invasora o costes económicos) en webs como Google Scholar o Web of Science. Los documentos resultantes se filtraron para obtener aquellos que contenían costes. Además, una pequeña proporción de los datos de costes se obtuvieron a través del contacto directo con expertos y otros actores clave de las invasiones biológicas.

La primera versión de la base de datos, InvaCost v1.0, contenía 2 419 datos de costes económicos, de los cuales la mayor parte procedían de documentos escritos en inglés y sólo una pequeña parte procedía de documentos en otros idiomas. Esta situación indicaba que claramente había lagunas en la base de datos, ya que los costes documentados en otros idiomas podrían estar siendo ignorados. Por ello, surgió la idea de actualizar InvaCost para que también contuviera costes documentados en lenguas no inglesas. Los detalles de cómo se llevó a cabo esta actualización se encuentran detallados en el Apéndice 2 (Angulo et al. 2021a).

→ Apéndice 2. Angulo E, Diagne C, Ballesteros-Mejia L et al (2021a) Non- English languages enrich scientific knowledge: the example of economic costs of biological invasions. Sci Total Environ 775:144441

Extendiendo la búsqueda a lenguas no inglesas se obtuvieron 5 212 datos adicionales de costes económicos, que llenaron parte de las lagunas que tenía la primera versión de la base de datos documentados en inglés. Esta nueva búsqueda añadió a InvaCost datos sobre costes causados por 240 especies invasoras adicionales, costes más locales, que afectaban más a las autoridades que gestionan las especies invasoras, y costes en 15 países para los que antes no había información disponible (ver Cuadro 1). Estos datos (procedentes de 10 lenguas distintas) se incorporaron en la versión v3.0 de InvaCost.

3. **Los costes económicos en España**

La búsqueda de datos de costes económicos en lenguas no inglesas supuso el marco ideal para buscar datos de costes económicos en España. Por ello, decidimos hacer un estudio detallado de los impactos económicos de las invasiones en España (el cual se encuentra en el apéndice 3). El objetivo era ofrecer una aproximación de los costes de las especies invasoras en España, así como proponer medidas para hacer una gestión óptima a nivel nacional. Se recopilaron más de 3000 datos. Esto fue posible gracias a las contribuciones de un gran número de gestores de las administraciones públicas que trabajan con estas especies en su día a día. Los datos recopilados mostraron que las especies invasoras están costando, como mínimo, unos 13 millones de euros al año en España. Pero sabemos que hay muchos más costes de los que hemos encontrado: la mayor parte de los costes de investigación sobre estas especies, lo que cuesta en las aduanas detectar las especies invasoras (p.e. en puertos y aeropuertos), los gastos que sufren los parques nacionales, los gastos generados por los daños a la agricultura, o la gestión realizada por algunas confederaciones hidrográficas, son algunos de los campos que no aun no hemos podido explorar. La mayor parte de los 230 millones de euros que hemos cuantificado desde 1997 se han gastado en control y erradicación de las poblaciones que ya están establecidas (Ver Cuadro 2). Nuestros resultados sugieren que se debería invertir más en detección temprana y en un servicio que a nivel nacional coordine los esfuerzos que hacen todas las administraciones de manera independiente.

→ Apéndice 3. Angulo E, Ballesteros-Mejia E, Novoa A, Duboscq-Carra VG, Diagne C, Courchamp F. Economic costs of invasive alien species in Spain. NeoBiota, 67:267-297.

4. **Los costes económicos en áreas protegidas del mundo**

Las áreas protegidas ofrecen un refugio para la biodiversidad global. Sin embargo, los esfuerzos de conservación realizados por las áreas protegidas se encuentran amenazados por las invasiones biológicas. Es por ello, que decidimos realizar un estudio sobre los costes económicos de las especies invasoras en áreas protegidas. El documento resultante de este estudio se encuentra en el Apéndice 4 (Moodley et al. 2022).

Los resultados obtenidos en este estudio nos mostraron que los costes de las especies invasoras en áreas protegidas a nivel global han ascendido, como mínimo, a unos US$ 22,24 mil millones entre 1975 y 2020. La mayoría de estos costes corresponden a gastos en gestión de especies invasoras en ecosistemas terrestres europeos.

→ Apéndice 4. Moodley D, Angulo E, Cuthbert RN, Leung B, Turbelin A, Novoa A, Kourantidou M, Heringer G, Haubrock PJ, Renault D, Robuchon M, Fantle-Lepcyk J, Courchamp F, Diagne C. (2022). Economic costs of biological invasions in protected areas worldwide-where do we stand?. Biological Invasions 24:1995-2016. DOI Preprint: https://doi.org/10.1007/s10530-022-02732-7

CUADRO 1. Nota de prensa. **Las lenguas no inglesas enriquecen los datos científicos. Por ejemplo, cuando los científicos estimaron los costes económicos de las especies exóticas invasoras, al incluir 10 lenguas diferentes añadieron más de 200 mil millones de dólares con respecto a los datos en inglés.**

Los desplazamientos internacionales del Hombre y sus mercancías tienen como efecto secundario el desplazamiento de miles de especies en los nuevos ecosistemas. Entre ellas algunas se instalan bien, prosperan y causan daños. La ciencia las estudia y comunica sobre ellas, pero siempre en inglés. Entonces la ciencia, ¿está teniendo en cuenta todas las informaciones disponibles?

Para responder a esta pregunta, los ecólogos Elena Angulo y Franck Courchamp de la Universidad Paris Saclay en Francia, han dirigido a un grupo de jóvenes investigadores de más de 18 países diferentes a la búsqueda de datos de los costes económicos de las especies exóticas invasoras. Cada experto se ha dedicado a buscar los costes existentes en todos los países donde se habla su lengua nativa.

En un artículo publicado en la revista *Science of the Total Environment*, este equipo internacional ha puesto en evidencia la sorprendente riqueza de las informaciones en lenguas no inglesas. Partiendo de una base de datos que ya era rica, con 2500 registros de costes, el equipo ha multiplicado por cuatro esos registros gracias a esas 10 lenguas. También han encontrado costes para más de 240 especies nuevas, y para 15 países para los cuales no había costes reportados en inglés. Entonces ¿en que lengua habla la ciencia?

«*Cuando lanzamos este estudio esperábamos enriquecer nuestra base de datos en un bajo porcentaje, no más del 10%» explica Franck Courchamp; «aumentarla de 300% ¡ha sido una gran sorpresa !: incluso nosotros nos hemos quedado sorprendidos de la amplitud de los datos científicos que existen en otras lenguas, diferentes del inglés, la que ha servido tradicionalmente para comunicar la ciencia*».

Los autores insisten sobre la necesidad de no subestimar las lenguas no inglesas. Del español al francés y al portugués, pasando por el alemán, el holandés o el griego, pero también el ruso, el árabe, el chino o el japonés, los investigadores han obtenido costes en la mayoría de estos idiomas. Además de una ciencia sin fronteras lingüísticas, los autores han mostrado la importancia de la lengua para una comunicación eficaz entre todos los actores implicados con las especies exóticas invasoras: expertos científicos y expertos profesionales de la gestión de las especies en el campo.

Elena Angulo, primera autora de este estudio concluye: «*una información global fiable necesita por lo tanto una información multilingüe y multi-actores. En lo que concierne a nuestros trabajos, es la base para una mejor comprensión de los costes de las especies exóticas invasoras, lo que finalmente repercutirá en una información más correcta y fiable que optimizará los planes de gestión de estas especies a nivel mundial*».

CUADRO 2. Nota de prensa **¿Cuánto le cuestan las invasiones biológicas a España?**

Las invasiones biológicas causan grandes pérdidas en las economías de muchos países. Un equipo de científicos internacional acaba de presentar la estimación más completa hasta el momento de los costes de las especies exóticas invasoras en España: 232 millones de euros entre 1997 y 2022, con un coste anual en los últimos años de 13 millones de euros. Dicha estimación, publicada en un número especial de la revista científica Neobiota, también pone de manifiesto que los costes de las especies invasoras no afectan por igual a todo el territorio Español.

El transporte internacional del hombre y sus mercancías provoca el desplazamiento intencionado o involuntario de miles de especies de animales, plantas y microorganismos a zonas lejos de su origen. Entre ellas, algunas especies pueden sobrevivir, establecerse, y causar daños. Son conocidas como especies exóticas invasoras, o EEI.

En los últimos años, un grupo de científicos internacional, liderado por Franck Courchamp, director de investigación en el CNRS en París, Francia, ha puesto a punto la primera base de datos mundial sobre los daños económicos de estas especies y los gastos destinados a gestionarlas. Esta base de datos de libre acceso, llamada *InvaCost*(https://doi.org/10.6084/m9.figshare.12668570.v4), agrupa más de 13 000 costes de unas 1000 EEI en 176 países. Franck Courchamp explica "*hemos buscado la ayuda de expertos de todo el mundo para analizar estos costes".* Como resultado de esta gran colaboración, un número especial de la revista científica Neobiota acaba de publicar 19 estudios firmados por 63 investigadores, que analizan el coste de las EEI en los 5 continentes y, en particular, en 13 países.

Entre estos investigadores, se encuentran las españolas Elena Angulo y Ana Novoa. En uno de estos estudios, Elena y Ana han buscado y analizado los datos disponibles de los costes de las especies exóticas invasoras en España. "*Los datos disponibles para España en la primera versión de InvaCost no representaban de forma fiable el coste de las EEI en el país. Por ello, contactamos con los servicios de medio ambiente de las CCAA y algunas de las confederaciones hidrográficas, los cuales muy amablemente nos facilitaron datos más exactos sobre los costes de estas especies en España"* explica Elena Angulo, primera autora del estudio, que colabora con el equipo de la Universidad de París Saclay. Ana Novoa, actualmente investigadora en la Academia de Ciencias de la República Checa, y coautora de este estudio, añade: "los datos obtenidos muestran *que el coste de EEI en España asciende al menos a unos 232 millones de euros*".

"*Es muy interesante que para España, gracias a la participación de los gestores, tenemos datos de los costes de gestión de muchas más especies invasoras que para otros países europeos. Sin embargo, aunque sabemos que existen, casi no encontramos datos sobre sus daños económicos, excepto para dos especies: el mejillón cebra y el cangrejo rojo"*

La especie más costosa según los datos obtenidos resultó ser el jacinto de agua, *Eichhornia crassipes*, afectando principalmente a las confederaciones hidrográficas del Guadiana y del Tajo o a CCAA como Valencia, Extremadura o Castilla-la-Mancha.

El jacinto de agua, *Eichhornia crassipes:*

El mejillón cebra, *Dreissena polymorpha*, también presenta elevados costes, sobre todo en las confederaciones hidrográficas del Ebro, el Guadiana y el Tajo, y en las CCAA de sus recorridos, como el País Vasco, Aragón, Cataluña, Castilla-la-mancha o Extremadura.En el medio acuático también destaca el caracol manzana, *Pomacea maculata*, que afecta sobre todo a las CCAA de Cataluña y Valencia.

En cuanto a los insectos, la avispa asiática, *Vespa velutina*, extendida por la zona norte peninsular, está causando importantes daños económicos en Galicia, Asturias, País Vasco, La Rioja, Aragón, Cataluña, y las dos Castillas, y se ha extendido hasta las Islas Baleares. Otro insecto invasor de amplia distribución en el territorio Español y altos costes es el picudo rojo, *Rhynchophorus ferrugineus*, que afecta a las palmeras, aunque sólo se han reportado costes en Cataluña y Canarias.

Las plantas terrestres son responsables del 60% de los costes recopilados para España. Entre ellas cabe destacar el Eucalipto (*Eucaliptus* sp.), la caña (*Arundo donax*), y la uña de gato (*Carpobrotus* sp.). Esta última "*se encuentra en toda la costa española, principalmente en dunas, donde modifica el suelo, altera las comunidades de herbívoros y microorganismos y desplaza la flora nativa local*" explica Ana Novoa, que lidera actualmente un proyecto sobre esta especie.

Carpobrotus sp. y *Arundo donax*:

El único vertebrado que aparece entre las 10 especies invasoras más costosas es el visón americano, *Neovison vison*, con actuaciones para su control en prácticamente toda la península.

Los costes de las EEI en España varían entre regiones. Hay regiones con una gestión muy organizada, que tienen buenos registros anuales, como Valencia y Cataluña, y otras que destacan por sus altos costes, como las islas Canarias, lo que es lógico dado que tienen muchas zonas protegidas y que los ecosistemas insulares son muy frágiles ante las especies invasoras.

A pesar del esfuerzo realizado para sacar a la luz los costes de las EEI en España, los autores ponen de manifiesto que las cantidades obtenidas se quedan muy cortas. Por ejemplo, aún no han podido recoger los costes de la gestión de las EEI en Parques Nacionales, de los proyectos de investigación nacionales e internacionales que estudian las EEI en España, los daños económicos de las EEI a los sectores agrícola, forestal o sanitario, o el coste de los controles fronterizos de las EEI.

Los autores proponen la creación en el futuro de un servicio interministerial, que recoja la información de los servicios de las diferentes administraciones que trabajan con las EEI en España, para poder optimizar los recursos y gestionar mejor las EEI a nivel nacional. "*Ha sido muy muy fructífero el poder trabajar con gestores repartidos por el territorio nacional, y esperamos que tanto la base de datos, de libre acceso, como el análisis que hemos publicado, sirva para mejorar la gestión de EEI y sobre todo, coordinar los esfuerzos a nivel nacional entre los diferentes organismos que están implicados en gestionar las EEI*" dice Elena Angulo.

Listado de Apéndices:

→ Apéndice 1. Diagne C, Leroy B, Gozlan RE, Vaissière A-C, Assailly C, Nuninger L, Roiz D, Jourdain F, Jarić I, Courchamp F (2020) InvaCost, a public database of the economic costs of biological invasions worldwide. Scientific Data 7: e277. https://doi.org/10.1038/s41597- 020-00586-z

→ Apéndice 2. Angulo E, Diagne C, Ballesteros-Mejia L et al (2021a) Non- English languages enrich scientific knowledge: the example of economic costs of biological invasions. Sci Total Environ 775:144441

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SCIENTIFIC DATA

OPEN

DATA DESCRIPTOR

InvaCost, a public database of the economic costs of biological invasions worldwide

C. Diagne[1]✉, B. Leroy[2], R. E. Gozlan[3], A.-C. Vaissière[1], C. Assailly[1,8], L. Nuninger[1,8], D. Roiz[4], F. Jourdain[4,5], I. Jarić[6,7] & F. Courchamp[1]✉

Biological invasions are responsible for tremendous impacts globally, including huge economic losses and management expenditures. Efficiently mitigating this major driver of global change requires the improvement of public awareness and policy regarding its substantial impacts on our socio-ecosystems. One option to contribute to this overall objective is to inform people on the economic costs linked to these impacts; however, until now, a reliable synthesis of invasion costs has never been produced at a global scale. Here, we introduce InvaCost as the most up-to-date, comprehensive, harmonised and robust compilation and description of economic cost estimates associated with biological invasions worldwide. We have developed a systematic, standardised methodology to collect information from peer-reviewed articles and grey literature, while ensuring data validity and method repeatability for further transparent inputs. Our manuscript presents the methodology and tools used to build and populate this living and publicly available database. InvaCost provides an essential basis (2419 cost estimates currently compiled) for worldwide research, management efforts and, ultimately, for data-driven and evidence-based policymaking.

Background & Summary

A biological invasion is the successful introduction, establishment and spread of a species outside its native range, mostly driven by human activity[1]. Invasive species are pervasive drivers of global change, responsible for substantial ecological (*e.g.* biodiversity loss[2], disturbance of ecosystem functioning[3]), health (*e.g.* spread of diseases[4,5]) and social (*e.g.* declining quality of life[6]) damages almost everywhere in the world. Another important dimension of these impacts is the massive economic losses suffered by our societies (*e.g.* consumption of crops[7], degradation of infrastructures[8], decreasing business activities[9], loss of income[10]). Management efforts aimed at prevention, control and eradication of invaders represent additional, often substantial expenditures for human societies[11–13]. As such, a recent synthesis has shown that invasions of insects alone cost a minimum of US$76.0 billion per year globally[14].

Despite these enormous impacts, a lack of relevant data and clear public understanding of outcomes associated with invasions provide important barriers to their effective management and mitigation[15]. The need for knowledge and awareness becomes even more crucial in the changing global context in which many more species invasions are expected in the near future[16]. However, despite increasing interest and progress in considering invasions as a crucial problem over the last few decades, the necessity to mobilize more efforts on invasion issues is still urgent[17]. Developing efficient solutions necessarily requires involvement of non-scientists (*i.e.* general public, decision makers, policymakers) worldwide. One option understandable by a wide and varied audience is to describe these impacts in terms of economic costs. Informing people on the potential expenditures and losses

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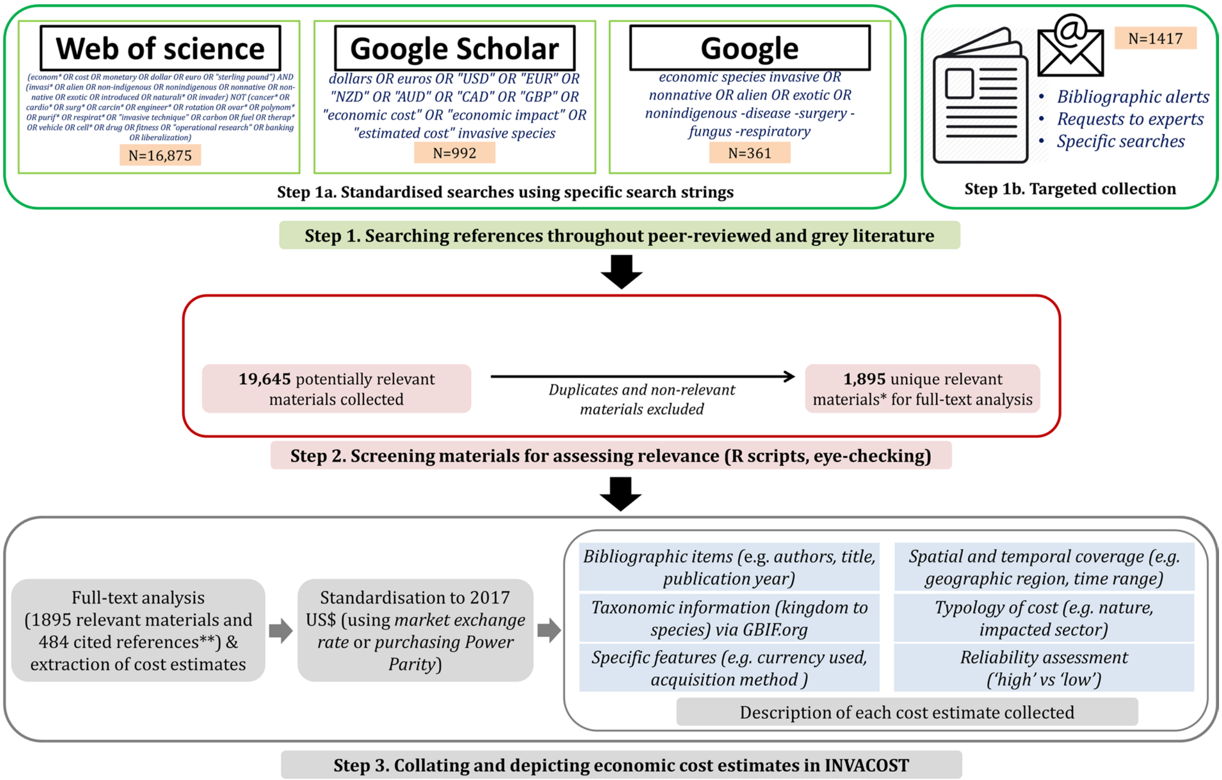

Fig. 1 General outline of the different construction steps of InvaCost. * Relevant materials were those *(i)* readable by the review team (*i.e.* written in English or French) for ensuring reliable assessment, *(ii)* containing at least one cost estimate *(iii)* exclusively associated with (at least one) invasive species. This assessment was based on the progressive analysis of titles, abstracts and keywords. Materials whose abstracts were not accessible were conservatively considered as relevant for further full-text analysis. ** Cited references were materials not gathered with our literature search process, but mentioned as original references in the relevant materials (initially collected) we analysed when seeking for cost estimates. Currently, 484 cited references are referenced in InvaCost.

dues to impacts of the invaders appears as a fundamental step to *(i)* raise public awareness and compel policy-makers to focus a more appropriate attention on invasions, *(ii)* estimate the costs of invasions for specific taxa, geographic regions or activity sectors as well as their drivers, *(iii)* improve assessment of proactive surveillance and control actions as well as prioritising management efforts at relevant scales[18], and *(iv)* support efficient and cost-effective decision-making[19].

Consistent, broad-scale approaches and synthetic analyses are increasingly recommended in invasion science to harmonize information, help set priorities of actions and improve coordination of efficient responses at different scales[20,21]. Although significant progress was recently made collating and analysing information on economic data associated with invasions, most of the studies attempting to broadly quantify impacts have so far individually focused on either few model taxa (often at species level)[14], specific economic sectors[22] and/or restricted spatial scales (local or regional)[23]. Unfortunately, the even rarer, still widely cited and considered as the only available global estimates[24,25] are outdated and suffer from methodological flaws that were already highlighted[26,27]. Moreover, whilst being essential for the purpose of policy, management and reporting, open access to full information reported on such type of data remains challenging[28]. To date, such an accessible, current and broad inventory of economic costs associated with invasions exists only for invasive insects[14].

In this paper, we introduce InvaCost as the most up-to-date, comprehensive, harmonised and robust global-scale data compilation and description of economic cost estimates associated with invasive species. InvaCost has been constructed to provide a contemporary and freely available repository of monetary impacts that can be relevant for both research and evidence-based policy making issues. To achieve this goal, we have developed a defensible, systematic, transparent and repeatable collaborative work strategy (Fig. 1). A large pool of both scientific peer-reviewed articles and grey literature (*i.e.* the diverse and heterogeneous body of material available outside, and not subject to, traditional academic peer-review processes[29]) was collected and scrutinized (Table 1). We extracted therein explicit estimations of costs and expenditures associated with invasive species, and then coupled them with a range of descriptors presented in this paper (Online-only Table 1). Here, we provide a full description of the process of InvaCost development, as well as specific details of all materials analysed. This unique, globally representative database (n = 2419 cost estimates currently described) is freely accessible online[30] and will be regularly updated with contributions from both authors and future users.

Bibliographic repositories	N	n	Spreadsheets in *InvaCost_references**
Web of Science	16.875	1.333	*WoS-collected_refs; WoS-screened_refs*
Google Scholar	992	310	*GS-collected_refs; GS-screened_refs*
Google search engine	361	119	*Go-collected_refs; Go-screened_refs*
Targeted collection	1.917	634	*TC-collected_refs; TC-screened_refs; Cited references*

Table 1. Quantitative summary of the literature search process. Search was performed using four bibliographic repositories. N: the number of references initially collected after applying specific search strings (Web of Science, Google Scholar, Google search engine) or as the result of targeted collection. n: the number of relevant materials (*i.e.* materials expected to contain cost estimates based on the contents of titles, abstracts and keywords) derived from N. The figures for the targeted collection include 484 references gathered within the analysed relevant materials. *The spreadsheets in which all details of associated references were recorded[30] are also referenced.

Methods

General scheme. We reviewed the literature published until April 2018 on the economic impacts of invasive species. For reasons of feasibility (linguistic skills of the review team, restriction to a reasonable scale of the review), we conducted all searches in the English language assuming that a large body of knowledge (mostly from international peer-reviewed papers and reports) is written in English. The dates of each search process were systematically recorded. We used the following strategy for all repositories (Fig. 1), while also taking into consideration the specificity of their algorithms.

First, a literature search was performed using three online bibliographic sources successively to minimize the risk of omitting relevant materials (Fig. 1, step 1a): ISI Web of Science platform (https://webofknowledge.com/), Google Scholar database (https://scholar.google.com/) and the Google search engine (https://www.google.com/). We carefully composed appropriate search strings that were consensually retained as the most efficient among a set of potential candidates. A decision was taken following preliminary tests based on a handful of relevant articles provided by consulted subject experts on some taxonomic groups (amphibians, reptiles, fishes and ants). Final selection of search strings comprised those considered to have the largest potential to identify key references. Each search string was set to include a combination of two search terms, related to 'invasive' and 'economics'. For both terms, we used a range of synonyms or related words. For example, for 'invasive' we used *invasi**, *invader* or *exotic*; for 'economics', we used *econom**, *cost* or *monetary*. In addition, the search string included exclusion terms to omit mismatches, for example, with studies from the field of medicine that are focused on pathologies or procedures that can be 'invasive' for patients. We complemented this search with documents gathered opportunistically (Fig. 1, step 1b). The potentially relevant materials derived from all these sources were combined in a single file and screened for duplicates. Second, retrieved documents were individually assessed at progressive levels (titles, then abstracts, keywords, and finally full text when abstracts were missing; Fig. 1, step 2) based on three criteria. Hence, materials were deemed relevant if *(i)* they matched with the linguistic competencies of the review team (*i.e.* written in English, or French where English language was restricted only to the title and/or abstract) for allowing reliable assessment, *(ii)* they contained at least one cost estimate (studies exclusively providing benefit estimates from direct use or exploitation of invasive species were excluded), and *(iii)* that this cost estimate is exclusively associated with invasive species (estimates merging non-invasive and invasive species, without the possibility of distinguishing the respective contribution of each group to the overall cost, were excluded). To ensure transparency and validity, each document was checked by two reviewers and in case of a disagreement between assessors, a third reviewer was involved. However, it was often difficult to judge from the topic whether the content of an article was relevant and so consequently many more articles were conservatively kept when final agreement was lacking among assessors.

Finally, relevant materials were scrutinized for data on economic costs (Table 1; Fig. 1, step 3). During this step, additional relevant materials were found as cited by the analysed materials. Obtained cost data were collated in a database and the costs were converted to a common and up to date currency (2017 US$), and then depicted by different descriptors. Categories extracted from relevant materials allow search of the database and data pre-selection to facilitating analysis of costs based on taxonomic groups, geographical areas, impacted sectors, types of costs, or other categories. The reliability of cost estimates and all associated information recorded in the final InvaCost database was systematically checked at least twice, and every ambiguous element was discussed to reach a consensus. We also checked all entries in the database to ensure that there were no obvious duplicate reports (i.e. multiple documents reporting the same cost estimate) or mistakes.

Hereafter, we specifically describe each of the steps made to generate InvaCost.

Literature search. *Web of Science.* We used the Web of Science (hereafter called WoS) to conduct a search for potentially relevant materials on 7 December 2017 (Fig. 1, step 1a). We applied the following search string: ***(econom* OR cost OR monetary OR dollar OR euro OR "sterling pound") AND (invasi* OR alien OR non-indigenous OR nonindigenous OR nonnative OR non-native OR exotic OR introduced OR naturali* OR invader) NOT (cancer* OR cardio* OR surg* OR carcin* OR engineer* OR rotation OR ovar* OR polynom* OR purif* OR respirat* OR "invasive technique" OR carbon OR fuel OR therap* OR vehicle OR cell* OR drug OR fitness OR "operational research" OR banking OR liberalization)***. The terms were searched in the field code "Topic" which includes title, abstract and keywords, and which also comprises 'Keywords Plus' that are generated by WoS through an automatic computer algorithm, based on words and phrases that appear frequently in the titles of article's bibliographic references and not necessarily in the main text of the article itself. To limit the search to relevant fields of research, we used the function 'refine' to exclude subject areas not related to economics and/or invasion biology.

We exported all records (n = 16,875) into an Excel worksheet[30] (Table 1) to identify the relevant materials by a two-step procedure. First, we excluded the references identified only based on 'Keywords Plus', which were shown to be poor specific descriptors of the content of articles[31]. We also excluded references identified based on the presence of only a single search term in the topic, as we assumed that words related to both search terms ('invasive' and 'economics') should be mentioned at least once in the title, abstract and/or keywords of a relevant material. To identify these irrelevant materials within the references collected, we developed a script (see Code Availability) in the R programming language (R v.3.4.3)[32]. Subsequently, 10,592 references were kept for the next screening step based on the described criteria.

In the second step, the topic of every reference selected was checked manually to ensure potential relevance of its contents. This allowed the elimination of documents incorrectly identified as relevant, such as studies without a true monetary assessment, or those focusing on economic estimates not directly attributable to invasive species only. Finally, 1,333 documents were judged as relevant materials (Table 1) and moved to the final data collation step.

Google scholar. The Google Scholar database is a large source of grey as well as peer-reviewed literature. Nevertheless, we had to modify our approach in order to address inherent limitations of this database as a search tool (see Haddaway *et al.*[33] for a comprehensive analysis). Typically, Google Scholar allows limited Boolean operators (no nesting using parentheses permitted) and search strings are limited to 256 characters. Additionally, only the first 1,000 search results can be viewed and the order in which results are returned is not disclosed. We also wanted to maximize novel information by avoiding too much overlap between the references collected with WoS and those gathered here.

In light of the above, we adapted our search string to generate the most efficient outcome, *i.e*. sufficiently pertinent to bring the most relevant items to the top of the result list while not unnecessarily large so as to limit the host of non-viewable results. Thus, the following search string was applied on 26 April 2018, using the advanced search facility to search for selected words anywhere in the article (see https://scholar.google.se/intl/en/scholar/help.html#searching for further details): ***dollars OR euros OR "USD" OR "EUR" OR "NZD" OR "AUD" OR "CAD" OR "GBP" OR "economic cost" OR "economic impact" OR "estimated cost" invasive species***. We specified currencies for prioritising materials with monetary data in the top of the resulting list. These currencies were chosen as they were the most often used to express economic costs in the literature collected from the WoS. Nevertheless, any reference evoking economic costs in other currencies was expected to be also captured by some specific combinations of '*economic*' terms in our search string that we would expect to be mentioned at least once in the full-text of relevant papers. In addition, we included the concomitant presence of 'invasive' and 'species' terms to restrict the outcomes to papers within the scope of our synthesis. Subsequently, we collected all viewable results (100 pages, n = 992 references of the 668,000 generated), thus going beyond the traditional and arbitrary sample size of first 50–100 results, which is frequently selected in many systematic reviews. We used a web-scraping programme (https://www.webscraper.io/) to extract all the titles' references returned by the search in an Excel spreadsheet. Because we could not efficiently export the abstract for every reference, we screened them online to assess their potential relevance.

As a result of a search and relevance assessment within Google Scholar, the references, abstracts and specific bibliographic details of 432 documents were added to the sample for further analysis. After excluding duplicates with WoS retrieved references, 310 additional documents were included in the sample as potentially relevant materials (Table 1).

Google. We used the Google search engine to complete the standardised literature search. As when searching with Google Scholar, we took into account specific constraints related to the use of this search engine. Moreover, browsing through Google search results can be overwhelming due to the vast amount of information of highly variable quality. We attempted to implement a search strategy that could allow overcoming these limitations as much as possible. We used the following search string: ***economic species invasive OR nonnative OR alien OR exotic OR nonindigenous -disease -surgery -fungus -respiratory***. We added four exclusion terms (*disease, surgery, fungus, respiratory*) identified during preliminary tests to restrict the number of irrelevant studies, associated with medical research. We did not use a range of economics-related terms, such as impact or cost, as they returned overly large numbers of mismatches.

The web search was conducted on 8 May 2018 by searching for specified terms within page titles of each document, in order to maximize the likelihood of identifying grey literature. We especially targeted grey literature because searches by the other two platforms mainly led to peer-reviewed publications. We assumed that documents published online by various governmental and non-governmental organisations (NGO), research centres and academic institutes are more likely to contain relevant data than other types of documents such as blogs and catalogues[29]. Therefore, we restricted our search to the documents located on governmental, academic and NGO webpages to ensure that explicit, traceable and expertise-based information was retrieved. We conducted independent searches for each type of webpage by specifying the type of web extension in the advanced search facility (*.gov* for governmental,*.edu* for academic, and*.org* for organisational webpages).

361 search hits were collected (document name, publishing year and URL of the main website homepage, if available) and stored in the database with the same host of dedicated information (Table 1). If the item analysed was a website homepage, we conducted on-line searches of potentially relevant materials within the website database(s), by filters if available, or by using the search bar with combinations of keywords. Websites that did not contain a database or search bar were searched manually. We then eliminated all duplicates resulting from references

being listed on multiple websites, or due to typographical mistakes and/or incomplete records when reporting a reference within different repositories. A total of 119 potentially relevant materials was finally obtained (Table 1).

Targeted collection. Finally, we sourced other potentially relevant materials that did not originate from the above-described processes (Fig. 1, step 1b). On one side, we dedicated specific efforts on gathering cost estimates for particular taxa or areas for which data previously obtained seemed scarce. First, we made sure that some key species were adequately covered; for example, costs associated with invasive mosquito species responsible for much of the burden of mosquito-borne viral diseases worldwide (*Aedes aegypti* that mainly invaded the intertropical zone from the 15th-17th centuries, and *Aedes albopictus* for which the global dissemination was more recent[34]) were searched in a specific way using WoS and PubMed (https://www.ncbi.nlm.nih.gov/pubmed/) repositories (see supplementary file 1 for details on search strings and matching with PRISMA statements). Second, materials were also retrieved following requests to specialists (*e.g.* Aliens mailing list, https://list.auckland.ac.nz/sympa/info/aliens-l) to bridge gaps identified for Russia and China, two of the five largest countries for which available on-line data were particularly scarce. A typical message first summarized the objectives of our research project and second, requested recipients to provide relevant material and/or suggest further contacts in this regard. On the other side, we also compiled additional materials when establishing the methodology for the project (*e.g.* when testing different search string combinations at initial stages of the work), from the bibliographic alerts set up by the review team. All 1417 documents obtained from this process were entered in the database, with information on the person providing the document (Table 1;[30]). Subsequently, 150 documents identified as not previously retrieved were considered relevant for further, full-text screening (Table 1).

Extraction of cost estimates. The Online-only Table 1 comprises all the information of InvaCost that we mention further in this article, using simple quotation marks for 'Columns' of the database and italic letters for the different categories within each column. The full-text of each relevant material was scrutinized for any cost estimate that could be incorporated into InvaCost[30]. The final stage of inclusion/exclusion took place during this data extraction. When the screened documents reported cost estimates by citing sources that were not retrieved by our literature search, whenever possible we assessed the original sources of data in order to better characterize the reported cost. These novel information sources not initially captured by our literature search were then added to the collection list (Table 1). In such cases, we provided information on all documents that were consulted to trace back the original source ('Previous materials'). In contrast, if no original cost data were found in the cited source, the document was discarded. For all reported costs where the original source was not available or accessible, we emphasized this in a dedicated column ('Availability').

Then, we first extracted raw cost data, *i.e.* how they appear in the material in local currency ('Raw cost estimate local currency'). When multiple cost estimates were provided for a single instance, we calculated median values (e.g. different cost estimates according to several management scenarios dedicated to the same invasive population) and collated the minimum and maximum estimates provided (columns 'Min/Max raw cost estimate local currency'). When costs were estimated at different time and/or spatial scales in the same material, we opted to choose – when possible – those estimate(s) that summarise(s) as effectively as possible the figure(s) shown in the study. If such an estimate was not obvious to identify throughout the full-text, we extracted every relevant cost estimate. In these latter cases where several cost estimates were provided in a single study, we also collated the minimum and maximum estimates provided.

Temporal information on the costs were also retrieved: the 'Period of estimation' as stated in the material and hence, when possible, the 'Probable starting/ending year' of the period of estimation and the 'Time range' (*year* if the estimate is given yearly or for a period up to one year, *period* if the estimate is given for a period exceeding a year). The 'Occurrence' column gives the status of the cost estimate as *potentially ongoing* (if the cost can be expected to continue beyond the period of estimation) or *one-time* (if the cost was deemed as unlikely to continue). For cost estimates provided without a clear indication on the timeframe considered, or covering periods shorter than a year, we considered them with a *year* 'Time Range' and a *one-time* 'Occurrence' to avoid the risk of overestimating the duration of collated costs. The 'Raw cost estimate'– with complementary information on the 'Time range', 'Period of estimation' and 'Occurrence' – can be used to estimate total costs over a given period of time. We then transformed the raw cost estimates to cost estimates per year ('Cost estimate per year') by dividing the raw costs with a *period* 'Time Range' by the duration of the 'Period of estimation' (obtained from the difference between the 'Probable ending year' and 'Probable starting year'). The raw costs with a year 'Time Range' were reported as they are, because they are already considered at the scale of a year.

Description of cost estimates in InvaCost. Each of the cost estimates recorded was characterized by a number of information, including ***(a)*** the reference from which the cost was extracted, ***(b)*** the taxonomy of the associated species, ***(c)*** the spatial and temporal coverage of the study, ***(d)*** the typology of each cost estimate and ***(e)*** the evaluation of the reliability of the estimation method(s). For most of the variables considered in InvaCost, a non-negligible part of the cost estimates was not attributable to a single existing category due to the lack of precise information provided by the authors or because they simultaneously belong to multiple categories. In such cases, we respectively reported them as either *Diverse/Unspecified* or as slash-separated lists of categories (*e.g. Artiodactyla/Carnivora* for the 'Order').

Details about the nature of the information retrieved as well as the choices made to characterize each cost are synthesized in Online-only Table 1:

(a) We provided bibliographic information on each reference (*e.g.* 'Reference title', 'Authors', 'Publication year'). Others specific details (*e.g.* abstract, journal, download link) are given in a dedicated file[30] with which the columns 'Repository' and 'Reference ID' of InvaCost allow correspondence of information.

Sector	Description
Agriculture	Considered at its broadest sense, food and other useful products produced by human activities through using natural and/plant resources from their ecosystems (*e.g.* crop growing, livestock breeding, beekeeping, land management)
Authorities-Stakeholders	Governmental services and/or official organisations (*e.g.* conservation agencies, forest services, associations) that allocate efforts for the management *sensu lato* of biological invasions (*e.g.* control programs, eradication campaigns, research funding)
Environment	Impacts on natural resources, ecological processes and/or ecosystem services that have been valued by authors such as disruption of native habitats or degradation of local habitats
Fishery	Fish-based activities and services such as fishing and aquaculture
Forestry	Forest-based activities and services such as timber production/industries and private forests
Health	Every item directly or indirectly related to the sanitary state of people such as vector control, medical care and other derived damage on human productivity
Public and social welfare	Activities, goods or services contributing - directly or indirectly - to the human well-being and safety in our societies, including local infrastructures (*e.g.* electric system), quality of life (*e.g.* income, recreational activities*)*, personal goods (*e.g.* private properties, lands), public services (*e.g.* transports, water regulation), and market activities (*e.g.* tourism, trade)

Table 2. The different market and/or activity sectors mentioned in InvaCost. Note that most of the cost estimates recorded in the database are associated with more than a single sector and are thus reported as slash-separated lists of sectors.

(b) We normalised and harmonised all taxonomic information on the invasive species ('Kingdom' to 'Species' level) using the GBIF.org Backbone Taxonomy[35]. At this stage, spelling and other taxonomic errors were corrected. While each cost extracted was generally associated with a single invasive alien species, in some cases the data was related to multiple species without the possibility of disentangling species-specific costs. In this case, we mentioned either all species concerned if explicitly indicated by the author(s), or *Diverse/Unspecified* if not.
(c) We dedicated seven columns to describing the impacted area according to its environment (terrestrial and/or aquatic habitats), the temporal extent as mentioned earlier (*e.g.* 'Period of estimation', 'Time range') and the spatial coverage from the 'Geographic region' (*e.g.*, Central America, South America, Oceania-Pacific Islands) - rather than the official continent for better accuracy - down to the exact site ('Location') when possible. Each area was related to its country of attachment, leading to some mismatches between the 'Geographic region' and 'Official country' columns due to the existence of countries with non-contiguous overseas territories. For instance, costs found from invaders in La Réunion (a French oversea department) were attributed to *Africa* as 'Geographic region' and France as 'Country', while France obviously belongs to European continent.
(d) We characterised the typology of each cost mainly based on the following descriptors. The 'Implementation' at the moment of the cost evaluation states whether the reported cost was *observed* (*i.e.* cost actually incurred by an invasive species within its invasive distribution area) *or potential* (*i.e.* not incurred but expected cost for an invasive species beyond its actual distribution area and/or predicted over time within or beyond its actual distribution area). The 'Acquisition method' provides information on how the cost data was obtained, *i.e. report/estimation* directly obtained or derived (using inference methods) from field-based information, or *extrapolation* relying on computational modelling. The 'Impacted sectors' indicates which activity, societal or market sectors were related to the cost estimate (see Table 2 for details). The 'Type of cost' ranges from the economic *damages* and *losses* incurred by an invasion (*e.g.* value of crop losses, damage repair) to different levels of means dedicated to the management of biological invaders (*e.g.* control, eradication, prevention).
(e) Lastly, we evaluated the level of 'Reliability' of the methodology reported by the authors to provide cost estimates (Fig. 2). Prejudging the relevance of each cost estimate is not straightforward and could suffer from a high level of subjectivity. Here, we rather aimed to evaluate in the most objective manner whether the approach used for cost estimation was documented and traceable. Hence, materials that could not be accessed for full-text investigation were conservatively considered as of *low* reliability. Alternatively, each cost estimate recorded from any accessible material was qualitatively assessed as of *high* or *low* reliability following a procedure depending on the 'Type of material' analysed (*peer-reviewed article* or *grey material*; Fig. 2). Peer-reviewed articles and official documents (*e.g.* institutional or governmental reports) are likely validated by experts before publication. We assumed therefore that all cost estimates collected from these materials may likely be of *high* reliability. Conversely, for grey materials other than official reports, the attribution to one or other of these categories (high vs low reliability) was based on specific analysis of each cost estimate. We checked whether the method estimation was fully described, independently of its comprehensiveness, *i.e.* if the original sources or potential assumptions were properly documented or justified, and/or the calculation methodology was explicitly demonstrated. Here, we opted for a conservative strategy that might be not optimal, as depending mostly on the nature of the publication.

Beyond the factual elements included in the descriptors from *(a)* to *(c)*, those presented in *(d)* and *(e)* (to which we can add the descriptor 'Spatial scale') are the result of a conceptual and analytical framework created based on our own experience. This experience was gained when collecting and getting acquainted with the diversity and complexity of situations one can find behind the "economic costs" linked to biological invasions, as well as the strategies used for estimating them. We think that the different subcategories identified therein (*e.g. observed* vs *potential* costs within the descriptor 'Implementation') should not be aggregated to limit potential confusions in future analysis. Also, we acknowledge that the possible sub-categories of these descriptors might be improved and adapted according to the scope of future analyses made using InvaCost. We are convinced that the descriptors thus defined and categorised may strongly help in this perspective.

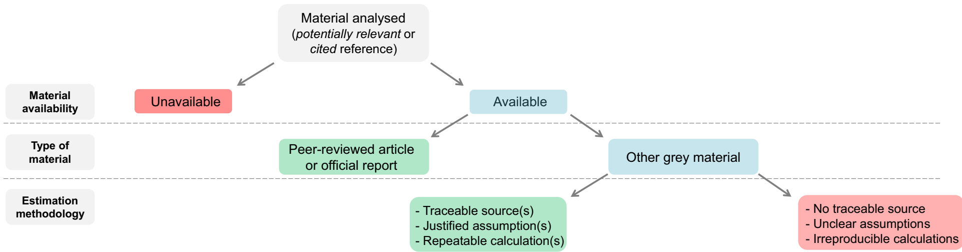

Fig. 2 Decision tree approach for assessing the reliability of the method used for estimating each cost. The colour of the boxes indicates which decision was taken: *green* when material was deemed as of *high* reliability, *red* when material was deemed as of *low* reliability, *blue* when taking any decision needs further investigation. The intended purpose of this process was not to evaluate the quality, relevance or realism of the studies performed for providing cost estimates, but rather to assess if the methodology *(i)* has been reviewed and validated by peers or experts prior any publication, or *(ii)* if not, whether this methodology was clearly stated and demonstrated.

Standardisation of cost data. Using definitions, data and indicators provided by the World Bank Open Data and the Organisation for Economic Cooperation and Development (OECD), we expressed all retrieved costs (raw costs and costs per year) in US dollars (US$) for the year 2017[30] using a multi-step procedure. We provided here two ways for standardising cost estimates according to the conversion factor: one based on the market exchange rate (local currency unit per US$, calculated as an annual average), and another based on the Purchasing Power Parity (PPP, local currency unit per US$, calculated as an annual average) that is the rate of currency conversion that standardises the purchasing power of different currencies by eliminating the differences in price levels between countries. Opting for one strategy or the other for further investigation or discussion is beyond the scope of this paper and will befall on the author(s) of future analyses made using InvaCost.

We first converted the cost estimates from local currencies to US$, by dividing the cost estimate with the official market exchange rate (https://data.worldbank.org/indicator/PA.NUS.FCRF?end=2017&start=1960) corresponding to the year of the cost estimation ('Applicable year', that is the year of the 'Currency' value, but not necessarily the year of the cost occurrence). The cost obtained in US$ of that year was then converted in 2017 US$ using an inflation factor that takes into account the evolution of the value of the US$ since the year of cost estimation. The inflation factor was computed by dividing the Consumer Price Index (CPI, which is a measure of the average change over time in the prices paid by consumers for a market basket of consumer goods and services; https://data.worldbank.org/indicator/FP.CPI.TOTL?end=2017&start=1960) of 2017 by the CPI of the year of the cost estimation.

As an alternative, we also converted costs to 2017 US$ value based on PPP instead of the classical market exchange rates in the initial conversion step. PPP values were primarily collected from data provided by the World Bank (https://data.worldbank.org/indicator/PA.NUS.PPP?end=2017&start=1990), or by the OECD (https://data.oecd.org/conversion/purchasing-power-parities-ppp.htm) when information was not retrievable through the World Bank database. For this purpose, we had to deal with published costs that were expressed in currency that was different from the country where the costs were estimated (*e.g.* published cost in African countries expressed in US or Canadian $). Thus, prior to using PPP as a conversion index, we had to perform a retro-conversion by multiplying the original cost estimate by the official market exchange rate (local currency unit per currency unit used). For PPP-based standardisation, it was not possible to perform the process for all cost estimates as PPP data do not exist for all countries and/or specific periods (we mentioned NA in the database when such information was missing).

In summary, we used the following formula to convert and standardise each cost estimate:

$$C_e = (\boldsymbol{M_V}/\boldsymbol{C_F}) \times \boldsymbol{I_F}$$

with $\boldsymbol{Ce}$ = Converted cost estimate (to 2017 US dollars based on exchange rate or Purchase Power Parity), $\boldsymbol{M_V}$ = Cost estimate (either the 'Raw cost estimate local currency' extracted from analysed paper or the 'Cost per year local currency' transformed by us), $\boldsymbol{C_F}$ = Conversion factor (either the official market exchange rate or the purchasing power parity, in US dollars), $\boldsymbol{I_F}$ = Inflation factor since the year of cost estimation, calculated as $CPI_{2017}/CPIy$ with CPI corresponding to the Consumer Price Index and y corresponding to the year of the cost estimation ('Applicable year').

We thus provided four columns with the raw cost estimates or the cost estimates per year, expressed in 2017 USD based on the exchange rate or PPP.

Data summary. InvaCost currently contains 2419 cost estimates (1215 from peer-reviewed articles, 1204 from grey materials), collected from 849 references, of which 1769 estimates were deemed as of *high* reliability. In total, twenty currencies are reported in our database, the majority being US dollars, n = 1348 cost estimates. Not all cost estimates were successfully converted to 2017 US$ as *(i)* conversion data from official sources are available only since 1960 (cost estimates range from 1945 to 2017 in InvaCost) or simply not found for some years and countries, and/or *(ii)* cost data are sometimes simultaneously associated with several countries, constraining the PPP-based standardisations. Hence, respectively 2416 and 2126 estimates were successfully converted using market exchange rates and PPPs. Cost estimates are either direct reports/estimations (n = 2127) or values

Geographic regions	Number of materials	Number of cost estimates	Number of high reliability estimates	Number of taxonomic units
Africa	46	182	163	51
Asia	75	132	111	20
Central America	29	61	51	9
Europe	102	338	255	89
Mixed	33	38	30	19
North America	316	716	438	139
Oceania-Pacific islands	272	867	646	129
South America	45	85	75	16

Table 3. Quantitative summary of information recorded in InvaCost according to the 'Geographic region' of the cost estimates. Central America includes cost data from the Carribean area. 'Mixed' contains data concomitantly associated with two or more geographic regions. Taxonomic units refers either to a single species or, if any, to a unique group of species for which specific contribution to the whole cost is not possible to disentangle.

gathered from extrapolative computations (n = 292). At a taxonomic level, these estimates are associated with 343 species belonging to six kingdoms (Animalia, Bacteria, Chromista, Fungi, Plantae, Riboviria). InvaCost has global coverage (90 countries) and includes continental, insular and overseas territories. Data are associated with terrestrial as well as aquatic (freshwater, brackish and marine) environments. Costs were estimated at different spatial scales (*continental* (n = 35), *country* (n = 1111), *global* (n = 17), *intercontinental* (n = 9), *regional* (n = 67), *site* (n = 836), *unit* (n = 329)). The Table 3 summarises quantitative data and information reported in InvaCost for each geographic region considered (see also Supplementary file 2).

Possible applications. InvaCost is expected to help bridge the gap between a growing scientific understanding of invasion impacts and still inadequate management actions. This work is thus in line with the aims of a panel of decisions recently adopted by the Convention on Biological Diversity (Decision XIII/13, https://www.cbd.int/doc/decisions/cop-13/cop-13-dec-13-en.pdf) advocating the incorporation of invasion science knowledge into management planning. In addition to offer unique opportunities for future research, InvaCost will provide a strong quantitative and evidence-based support for impacts of invasive species reported in other databases such as the Global Register of Introduced and Invasive Species (GRIIS)[20], helping refine information in this database. Also, invasive populations recorded in InvaCost but data deficient in the GRIIS should be ultimately classified in that database.

Additionally, InvaCost could be considered as another data-based component, adding novel and significant information on invader impacts categorised by the Socio-Economic Impact Classification of Alien Taxa (SEICAT)[36]. The latter is a classification system, applicable across a broad range of taxa and spatial scales, providing a consistent procedure for translating the broad range of measures and types of impacts into ranked levels of socio-economic impacts, assigning alien taxa on the basis of the best available evidence of their documented deleterious impacts. Quantitative support provided by InvaCost will strongly contribute to impact classification. Ultimately, integrating data from these diverse sources could allow a complete description of the overall impacts of biological invasions at regional and global scales.

Caveats and directions for further database improvement. Rather than claiming exhaustiveness of data collated, we highlight that InvaCost should be considered as the most current, standardised, accurate and globally representative repository of various economic losses and expenditures documented for the largest possible set of invaders. We are aware that our database can be improved in at least three ways.

First, InvaCost mostly does not include publications and reports not yet available in electronic format and/or using non-English language, leaving open the possibility of increasing data comprehensiveness and limiting potential biases. Indeed, local reports as well as research results from some countries (e.g., China, Russia) are likely to be published in non-English language[37]. Again, accessing grey literature is challenging as it is not systematically digitalised and/or included in well-curated bibliographic databases[29]. We strongly encourage future users of InvaCost to help gathering this currently unreachable information when possible. Furthermore, some mistakes might have occurred despite our best efforts when constructing InvaCost. In this regard, we advocate for regular public updates of InvaCost in order to improve it both quantitatively (by adding currently inaccessible or missed information) and qualitatively (if errors are identified).

Second, as the distribution and impacts of invaders are inherently dynamic for a number of reasons[38], InvaCost should further consider the status of the species recorded for their economic impacts in order to improve both the relevance and the usefulness of the database. As an illustration, InvaCost likely includes invasive populations currently extirpated from particular areas after successful eradication campaign(s) as well as those still established but for which impacts are locally reduced as a result of management efforts. Attempting to obtain and integrate such information into InvaCost was beyond the scope of this work. Nonetheless, it should be reciprocally beneficial to establish connections between InvaCost and other databases such as the GRIIS that provides a harmonised, open source, multi-taxon database including verified information on the continued presence of introduced and invasive species for most countries[20]. In light of such additional information, the value of InvaCost will be its application for policy purposes, such as identification of exotic invaders that are currently associated with economic losses in particular areas. Also, crossing information between databases may allow the refinement of the descriptor 'Spatial scale' we propose here.

Third, we would recommend, for a future updated version of InvaCost that would require screening back all the materials, to improve the 'Acquisition method', 'Implementation' and 'Reliability' descriptors, to pay attention to the specificity of "avoided costs" and to create a new descriptor for 'non-market values'. We detail these possibilities below.

Improving descriptors. An improved version of the 'Acquisition method' could lead to a subdivision of the *extrapolation* category into *spatial*, *temporal* and *spatio-temporal extrapolation*. This would allow simultaneous refinement from the currently binary 'Implementation' descriptor (*observed* vs *potential*) into several levels of certainty regarding the incurred cost (*e.g.* taking into consideration the temporality (past/current or predicted) of the onset of the cost and of the status of the invasive species in the study area). The next step for deeming the 'Reliability' of the cost estimates recorded in InvaCost would consist of assessing the repeatability of the methodology used, by adapting the approach previously developed by Bradshaw *et al.*[14]. The latter evidenced that assuming the reproducibility of published methods should not rely only on the nature of the materials and recognized the qualitative nature of the procedure, although applying this approach to InvaCost was constrained by the large sample size and high diversity in our database (Bradshaw *et al.*'s study focused on a single taxonomic class). Also, because InvaCost involves several collaborators and potential future contributors, consistent and objective criteria should be further defined to cope with the large array of materials, methods and situations encountered.

Avoided costs. Introducing certain actions against biological invasions leads to avoided costs. Such avoided costs are sometimes evaluated, for instance to examine the relevance of different potential actions or to assess the effectiveness of an action that was taken. However, avoided costs cover a great variety of situations and require a careful consideration for future analysis, even if they do not have to be analysed separately from the other economic costs gathered in InvaCost. For instance, in the case of hypothetical actions, avoided costs can be considered as minimum estimates of the "real" costs (if they are unknown). However, in the case of completed or planned actions, the reported data should be the original costs (if known) minus the avoided costs, because the latter do no longer exist. Some avoided costs are probably already included in InvaCost but they are likely underestimated because keywords such as "savings" or "benefits" were not included in the search strings. Also, even if they are sometimes mentioned as "benefits" in the literature, care should be taken not to confuse these avoided-costs with the benefits incurred by direct use or exploitation of invasive species. The latter have been ignored in InvaCost since they were relatively few (and beyond of the scope of this database), but might constitute a twin project.

A new 'Non-market values' descriptor. The means dedicated to preventing or managing an invasion (*e.g.* manual removal of invasive plants) and certain economic *losses and damage* due to an invasion (*e.g.* the value of crop losses or the repair costs of damaged infrastructures due to an invasive insect) are observable on markets. However, some costs are not observable on markets but can be translated in monetary terms using several valuation methods – for instance, the willingness to pay for the conservation of a native species that is impacted by an invasive species is considered as the value given by a group of people to preserving the native species (*i.e.* the value that would be lost if this native species was impacted). We recognize the importance of informing the public about "non-market values", as giving an economic value to ecosystems or biodiversity can be a way of recognising and taking them into account in public decision-making processes[39], but attention should be paid to the issues linked to their assessment[40,41]. Among others, the different methods for assessing non-market values do not necessarily capture the same aspects of the values, so the resulting estimates might be different. Moreover, the very principle of giving a value to "benefit from nature" through economic valuation is not necessarily acknowledged by the entirety of scientific and civil communities[39,42]. For future analysis, the 'non-market values' should not be systematically aggregated with the other economic costs gathered in InvaCost. It is to note that while some non-market values are probably already included in InvaCost within the *losses* and *damage* 'Type of cost', the loss of non-market values is probably largely underestimated in the database because they were not the primary focus of InvaCost and therefore the related keywords were not included in the search strings.

These possible ways of improvement call for completion and/or refinement of existing entries as well as integration of newly published or acquired data by future contributors in InvaCost, with the aim to consolidate its long-term relevance (cf. Usage Note paragraph).

Data Records

All collected and examined references along with their bibliographic details and/or links for on-line access are recorded in an Excel workbook called *InvaCost_references* uploaded on Figshare[30]. This workbook consists of nine spreadsheets so that each bibliographic source considered (namely WoS, Google Scholar (GS), Google search engine (Go), and Targeted collection (TC)) is represented by two spreadsheets. The first four spreadsheets contain the complete list of potentially relevant materials obtained after applying specific search strings (*WoS-collected_refs*, *GS-collected_refs*, *Go-collected_refs*) or targeted searches (*TC-collected_refs*). The next four spreadsheets (*WoS-selected_refs*, *GS-selected_refs*, *Go-selected_refs*, and *TC-selected_refs*) comprise the list of relevant materials selected for final full-text screening. When a relevant material was found in more than one bibliographic repository, we considered it once only, as coming only from the first repository where it was chronologically recorded (respectively *WoS*, *GS*, *Go* and *TC*). The last spreadsheet (*Cited references*) contains the cited references collected during the full-text analysis of relevant materials. In each spreadsheet, each reference recorded is associated with a 'Reference_ID' that allows correspondence with InvaCost[30]. Cells with missing or unavailable information were marked as 'NA'.

All cost estimates gathered in the abovementioned references were compiled in a second Excel workbook called *InvaCost_database* uploaded on Figshare[30]. It contains a dataset with a spreadsheet recording all the cost estimates arranged in such a way that each line refers to information retrieved from one bibliographic reference and related to a single cost estimate associated with one taxonomic group (generally at specific level) in one determined location (regardless of the geographic scale), and related to one specific period of time. Each entry is associated with a host of information provided for detailed characterisation of each cost recorded (Online-only Table 1, Table 2). Missing information was marked as 'NA'. A second spreadsheet contains all the information used to standardise the cost estimates to 2017US$.

Technical Validation

Considerable effort was made to ensure the highest possible degree of reliability in our database development. Each step was undertaken by at least two researchers to mitigate individual subjectivity when dealing with the documents. The whole process carefully followed the recommendations on rule-based search operations for collecting and synthesising relevant evidence[43], as those reported in PRISMA (Preferred Reporting Items for Systematic Reviews and Meta-Analyses)[44]. Furthermore, extensive literature searches within several major bibliographic resources ensured that the collected sample was representative to the greatest extent possible. We also accounted for possible publication bias (*i.e*., the propensity for journals to publish studies with positive, hypothesis-affirming, or significant results rather than negative, contentious, or non-significant findings[45]) by searching both published and grey literature.

We also circumvented inherent limitations that appear when working with web search engines (e.g., the threshold for repetitive activity that triggers an automated block to a user's IP address) by using IP-mirroring software. We also made sure when systematically accessing online systems that their terms of use were not violated. Furthermore, we followed the recommendation made by Haddaway *et al*.[32] to check all viewable search results from all search engines used (contrary to the common practice of considering only the first 50–100 search results), leading to an improvement in both the transparency and coverage of the search process, especially with respect to grey literature. To validate our reference selection procedures, we ensured that no potentially relevant material would remain in the host of collected references that were considered as not relevant. For this purpose, we checked five random sub-samples (each sub-sample comprised at least 100 documents) of the list of non-relevant references to ensure that no potentially relevant material was excluded. As previously mentioned, each step of the entire process was systematically double-checked by at least two colleagues

Usage Notes

InvaCost is expected to be of interest for a broad range of actors directly or indirectly interested in biological invasions (governments, researchers, conservation agencies, etc.). We stored the full dataset in a public repository to facilitate global access. We wish to highlight that InvaCost is open to corrections and updates from authors as well as any interested contributor through a systematic, standardised process. We are willing to receive any feedback that could improve our database. Any reader or user can therefore add new information and/or correct existing ones within InvaCost. The contributors are expected to send either a message, or data as a spreadsheet containing specific information corresponding to each column (Online-only Table 1) of InvaCost to our e-mail address (updates@invacost.fr). We encourage the future contributors to give as many narrative elements as possible in the 'Details' column that can contribute to better understanding of the cost estimates or to support choices made for completing the database, in order to allow backtracking investigations and facilitate the review process.

Contributors should provide access to the document from which information was taken. We strive to establish a collaborative community (including experts as well as non-expert contributors) and an online platform in order to sustain regular reviews of InvaCost. The intent is to ensure transparency of the process as well as relevance of this database as it is expanded and updated. Regularly-updated versions will be dated and permanently stored in the same repository as the original version, with a unique stable DOI and unique version numbering for each release. Each contributor bringing relevant additional information will be acknowledged in the updated version that will be released in the online repository (Column 'Contributors'). Contributors are also encouraged to contact the authors in order to obtain supplementary information on how to use or update the database in case that information is not available here.

Users interested in working on the data described here are also asked to cite this manuscript as well as the specific version release of the database used, along with its DOI if necessary. The latter information should be systematically provided with the updated version downloaded from the public repository. We emphasise that users who aim to perform statistical analyses should take care to extract (from the database) and prepare relevant information based on the research question. One should keep in mind that duplicate or overlapping cost entries (*e.g*. multiple cost estimates for a single invasive species in a same location over a similar period) may exist in the database, and that these should be identified before any analysis. Importantly, neither the raw cost estimates ('Raw_cost_estimate_2017_USD_exchange_rate', 'Raw_cost_estimate_2017_USD_PPP') nor the cost estimates per years ('Cost_estimate_per_year_2017_USD_exchange_rate', 'Cost_estimate_per_year_2017_USD_PPP') should be simply summed since they do not necessarily cover the same time range or spatial scale, and they were not necessarily categorised within a working framework directly implementable for all types of studies. These columns should be used with complementary information on the 'Period of estimation', 'Time_range' and 'Occurrence' columns to take into account the temporal extent of the costs. Additionally, the *diverse/unspecified* nature of information retrieved (see previous sections) requires filtering out the data at hand before robustly quantifying the respective part of each descriptive category in the total cost. Furthermore, the spatial scale considered in InvaCost provides only an order of magnitude of the geographic extent considered for each cost estimate. Further exploring the economic costs of invasions based on data compiled in InvaCost should therefore ideally integrate specific information on the area dimensions actually impacted by the invasive species.

Code availability

The R script used in this manuscript to deal with the references collected from the WoS is provided as additional information (Supplementary file 3).

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Author contributions

F.C. conceived the initial project. C.D. designed the study strategy. C.D. and F.C. managed the extensive teamwork. C.D. and B.L. designed the database. All the authors performed the literature searches, material screening and data collation. C.D., C.A., L.N. and I.J. developed the scripts for analyses. C.D. and A.-C.V. carried out calculation and standardisation of cost estimates. C.D. took the lead in writing the paper with inputs and reviews from all contributing authors. All authors read and approved the final manuscript.

Competing interests

The authors declare no competing interests.

Additional information

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Non-English languages enrich scientific knowledge: The example of economic costs of biological invasions

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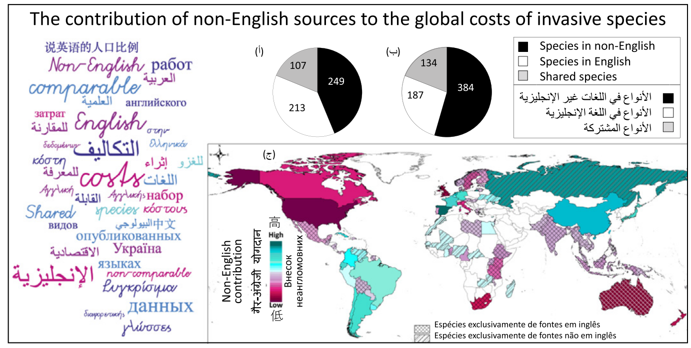
HIGHLIGHTS

- We compiled global economic cost data of invasive species from non-English sources.
- A large number of costs was added for new invasive species and new countries.
- As a result, global cost estimates of invasions increased by 16.6% (US$ 214 billion).
- Multi-language collaborations are necessary to enrich scientific knowledge.
- The use of non-English sources enhances data completeness and reduces knowledge gaps.

GRAPHICAL ABSTRACT



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ABSTRACT

We contend that the exclusive focus on the English language in scientific research might hinder effective communication between scientists and practitioners or policy makers whose mother tongue is non-English. This barrier in scientific knowledge and data transfer likely leads to significant knowledge gaps and may create biases when providing global patterns in many fields of science. To demonstrate this, we compiled data on the global economic costs of invasive alien species reported in 15 non-English languages. We compared it with equivalent data from English documents (i.e., the InvaCost database, the most up-to-date repository of invasion costs globally). The comparison of both databases (~7500 entries in total) revealed that non-English sources: (i) capture a greater amount of data than English sources alone (2500 vs. 2396 cost entries respectively); (ii) add 249 invasive species and 15 countries to those reported by English literature, and (iii) increase the global cost estimate of invasions by 16.6% (i.e., US$ 214 billion added to 1.288 trillion estimated from the English database). Additionally, 2712 cost entries — not directly comparable to the English database — were directly obtained from practitioners, revealing the value of communication between scientists and practitioners. Moreover, we demonstrated how gaps caused by overlooking non-English data resulted in significant biases in the distribution of costs across space, taxonomic groups, types of cost, and impacted sectors. Specifically, costs from Europe, at the local scale, and particularly pertaining to management, were largely under-represented in the English database. Thus, combining scientific data from English and non-English sources proves fundamental and enhances data completeness. Considering non-English sources helps alleviate biases in understanding invasion costs at a global scale. Finally, it also holds strong potential for improving management performance, coordination among experts (scientists and practitioners), and collaborative actions across countries. Note: non-English versions of the abstract and figures are provided in Appendix S5 in 12 languages.

1. Introduction

English is the language that dominates scientific publications in peer-reviewed journals in all research fields (O'Neil, 2018). However, in recent years there has been an increasing recognition of the importance of non-English literature for filling knowledge gaps, expanding the scientific knowledge base and successfully complete global pictures in multiple facets of science (Salager-Meyer, 2008; Amano et al., 2016; Hartling et al., 2017). Despite its importance, non-English literature remains largely underutilized by most researchers due to the language barrier that impedes understanding of the published materials, in addition to the lower accessibility to these sources (Ito and Wiesel, 2006; Lazarev and Nazarovets, 2018; Tao et al., 2018).

Knowledge gaps due to neglecting non-English literature are particularly severe for studies covering topics in ecology and biodiversity. Indeed, many geographic regions still remain highly underrepresented in the English ecological literature, simply because they lie in areas where mother tongues are not English (Di Marco et al., 2017; Hickisch et al., 2019; Nuñez et al., 2019). For example, it is known that directionality in transboundary research is extremely unbalanced, with English-speaking countries (e.g., USA, UK, Australia) dominating over non-English speaking regions, such as francophone Africa or Latin America (Verde Arregoitia and González-Suárez, 2019). Additionally, non-English knowledge from countries where English is not an official language is largely under-utilized, since it is not always accessible to the international scientific community, which undervalues the relevance of local expertise (Fazey et al., 2005; Zenni et al., 2017). Thus, researchers are geographically biased, which limits our understanding of global ecological patterns (Amano et al., 2016; Bellard and Jeschke, 2016). Researchers that are non-native English speakers might prefer to publish part of their work in their native language or in local journals (Verde Arregoitia and González-Suárez, 2019; but see Nuñez and Pauchard, 2010). While this maximizes local or national impact, it restricts the scope of their results to the scientific community and popular press globally, and thereby decreases opportunities for sharing experiences, novel ideas, observations or methodological advances (Nuñez et al., 2019). The value of accounting for data and results beyond just those made available in English has also been recently recognized for global meta-analyses (Konno et al., 2020).

In applied sciences, such as conservation biology or applied ecology, language is an essential factor for the transfer of knowledge and practices at different spatial scales, from global to local and vice versa. Language barriers are among the top obstacles to the use of science in policy, also negatively affecting the interaction between scientists and practitioners (Rose et al., 2018). On the one hand, scientific information is not always correctly transferred to practitioners, local managers, and policy makers, and this may be exacerbated if the relevant English publications cannot be accessed or if their formats are unusable (Rose et al., 2018). As an example, the prevalence of English as a primary publication language limited the use of scientific information by directors of protected areas in Spain (Amano et al., 2016). On the other hand, knowledge produced locally, beyond academic institutions, is not fully transferred to the international scientific community. For example, Verde Arregoitia and González-Suárez (2019) showed that one quarter of presenters from non-academic institutions (i.e., government entities, private foundations, NGOs, or civilian groups) at the 25th International Congress of Conservation Biology, published their work slower and less often than presenters from academia. Even if in that case presenters interacted in English, the knowledge produced outside academia adds to the language gap. This observation is reinforced by the low priority of non-academic stakeholders in having their findings published in the scientific literature. While English literature is characterized by a higher number of citations (Di Bitetti and Ferreras, 2017), a significant amount of data is compiled in reports that are not further published. For instance, local authorities may collect and report biodiversity related information in order to meet their environment and biodiversity management targets. As the information is intended for local stakeholders, most often non-researchers, the country's language is often used in these reports. These issues highlight the need to find ways that foster increased communication and collaboration among stakeholders and across regions, in order to favor the extrapolation of applied management strategies from one region to others (Nuñez et al., 2019).

In invasion biology, a global synthesis in the field has acknowledged the gaps of using only English literature (Lowry et al., 2013). Moreover, it is well-known that there is a strong geographical bias, partially caused by omitting non-English literature (Nuñez and Pauchard, 2010; Bellard and Jeschke, 2016). There is a misleading view of how non-English speaking countries are currently dealing with invasions: Zenni et al., (2017) showed how non-English literature reporting world leading efforts was internationally largely ignored, most likely due to well established expert scientific communities of biological invasions pertaining to English speaking countries. Hence, our objective here was to assess the potential gaps and biases in data compiled exclusively from sources written in English. To this end, we used InvaCost, a recently published database that synthesizes the reported economic costs of biological invasions worldwide ($N = 2419$ cost entries; Diagne et al., 2020a). Diagne et al. (in press) explored the distribution of these costs across space, taxa, and types of expenditure over time, and found that invasions cost a minimum of US$1.288 trillion (2017 US dollars) from 1970 to 2017 globally. Beyond these results, the authors also found large geographical data gaps, with few data outside North America, Europe, and Australia/New Zealand, and the majority of source documents being scientific peer-reviewed articles. In this sense, InvaCost (hereafter English database) for now consists of English sources exclusively. It is very likely that the studies on economic costs are not as rare as usually admitted, and that this preconception comes from a focus on English sources. In addition, not considering non-English sources can bias economic assessments, and hinder analyses that inform prioritization and expenditure on the management of invasive species.

We performed a data search in non-English languages, to compare it with the English database. We focused mainly on the most widely spoken languages, or the ones where we assumed that reports of economic costs of biological invasions could be found, such as Bengali, Chinese, French, or Spanish. By comparing the non-English and English data, we aimed: (i) to show how much more cost data we were able to capture when considering non-English languages (i.e., the gaps of considering only English documents), and (ii) to detect the magnitude and type of costs that were missing from the English literature (i.e., the bias produced when only considering English documents).

2. Methods

2.1. Data searching methods

We searched costs associated with biological invasions in 15 non-English languages by native speakers (Table 1). Following the methodology used to compile the English database (Diagne et al., 2020a), we used two complementary approaches for collating cost information. First, we performed a standardized literature search using three online bibliographic sources successively: ISI Web of Science platform (WoS hereafter; https://webofknowledge.com/), Google Scholar database (https://scholar.google.com/) and the Google search engine (https://www.google.com/). In the WoS, we used the same search string as those used for the English database, and used the "language" option to retrieve results for each non-English language (Appendix S1). This standardized search method was the only one that was exactly comparable to the methodology used in the English database (Diagne et al., 2020a). Search strings used in Google and Google Scholar were unavoidably slightly different in each language, which was due to inherent linguistic differences and methodological constraints in Google engines (Haddaway et al., 2015; Appendix S1). Second, similar to the English database, albeit more targeted, an opportunistic search was carried out in each language (Appendix S1). This included (i) searching web pages of national institutions, NGOs, and other organizations, (ii) seeking specific literature databases of the countries/languages considered, and (iii) contacting official national managers or researchers that could provide cost data.

Data were retrieved until May 2020 (Angulo et al., 2020; doi:https://doi.org/10.6084/m9.figshare.12928136). All data were compiled using the same structure as the English database (Diagne et al., 2020a; Appendix S2). Briefly, the database consisted of about 40 columns with four types of information: raw and standardized cost estimates; characteristics of data source documents (e.g., type of document, authorship, title, year); taxonomic classification of the invasive alien species for which costs were given; and cost characteristics (e.g., impacted sector, type of cost, spatial and temporal coverage, type of environment in which the cost occurred). We followed the procedures described in Diagne et al. (2020a) to screen for duplicates within the non-English database entries and against the English entries, as costs reported in non-English could have been the source of costs reported in English; in which case, exact cost entries were removed. Whenever possible and to ensure validity, each document was checked independently by two co-authors (i.e., all languages except Ukrainian and Greek). Cost standardization to 2017 US Dollars ($) also followed Diagne et al. (2020a).

2.2. The non-English database and comparability to the English database

Given that the non-English search was performed more recently than the English one (data for the English database - original version

Table 1
Number of cost entries (Entries) and documents (Docs) for each language in the non-English database. The four Indian languages are Hindi, Tamil, Telugu and Bengali.

Languages	Entries	Docs
Arabic	0	0
Chinese	117	33
French	1148	55
German	47	5
Greek	10	6
Indian languages (4)	0	0
Japanese	328	22
Portuguese	34	21
Russian	89	4
Spanish	3289	97
Dutch	50	15
Ukrainian	100	98
Total	5212	356

of the InvaCost database; Diagne et al., 2020a - was retrieved up until December 2017) and the search methods were slightly different, we consider that the two databases could not be fully compared. Thus, we divided the non-English database into two datasets (Fig. 1): one containing exclusively costs gathered from documents published before 2018, which could be quantitatively compared with the English database (hereafter called "comparable dataset"); and another one containing data from documents published after 2017 as well as unpublished data obtained from expert requests (i.e., that was not quantitatively comparable to the English database; hereafter called "non-comparable dataset").

Although most documents from the English database were published before 2018, we also extracted an "English comparable dataset" in which the few cost entries from unpublished documents or materials published after 2017 were removed (Fig. 1).

2.3. The effect of the proportion of English speakers on the number of costs

We analyzed the correlation between the numbers of cost entries of each non-English language per country and the proportion of English speakers per country. To do so, we used the complete non-English database. The number of entries was log10 transformed. We obtained data for the proportion of English speakers for 26 countries from Amano and Sutherland (2013) and Eberhard et al. (2020). Amano and Sutherland (2013) obtained the total number of speakers of English as the first or second language from four different sources - including a previous version of Eberhard et al. (2020) -, related it to the national population, and used the maximum value obtained for each country. When no data was available in Amano and Sutherland (2013), we referred to Eberhard et al. (2020). We related the number of entries (log transformed) to the proportion of English speakers in each country.

2.4. Differences between non-English and English data in cost descriptors

Using only the comparable datasets of both non-English and English databases, we evaluated the differences between them in three ways. First, we tested whether the number of entries was different for each of the following cost descriptors: geographic region and type of environment where the cost occurred, spatial scale and impacted sector of the cost, as well as the type of cost. The original categories of the "Spatial_scale" column of the English database (Appendix S2) were reassigned to three categories as follows: 'supranational' costs (regrouping the original categories of *global*, *intercontinental*, *continental*, and *regional*, i.e., costs estimated for more than one country), 'country-level' costs (estimated for a whole country) and 'local-level' costs (regrouping the original categories of *site* and *unit*, i.e., costs estimated within a country). The original categories of the type of cost of the English database were re-grouped in three categories: cost related to 'damage or loss', cost related to 'management', and 'mixed' costs when both costs categories are reported together or when the type of cost was unspecified meaning that it could not be easily classified under one or the other category (Appendix S3). For the type of economic sector, we used the categories: 'authorities/stakeholders', 'agriculture', 'health', 'environment', 'forestry', 'public and social welfare', 'fishery'; and we merged mixed categories with 'diverse/unspecified'. To assess differences in the taxonomic composition of invasive species between English and non-English entries, we only used the most represented, broad categories: the kingdom Plantae for plants and the phyla Arthropoda, Chordata, and Mollusca for animals. For the purposes of this analysis, we excluded data assigned to more than one of these categories.

To perform all of these comparisons, we fitted generalized linear mixed models with a binomial distribution and a logit link (SAS Institute Inc., 2018). For this purpose, we added dummy variables for each category within each of the above cost descriptors, with '0' (when the cost entry was not assigned to a specific category) or '1' (when the cost entry was assigned to a specific category). We considered each dummy variable as the dependent variable, and whether they come from the non-English or the English datasets as the independent variable. Because there could be more than one cost estimate within a given document (e.g., reporting five cost estimates for a given species in different years, or reporting costs for the control of five different aquatic species), entries coming from the same document were not statistically independent. Thus, we included the "Reference_ID" (the identification code for each document) as a random effect to explicitly model the covariance structure due to cost entries extracted from the same document ("repeated_subject" in Proc Genmod).

We also calculated, for each category of the cost descriptors, the percentage that the monetary costs of the comparable non-English dataset represented to the total costs obtained once combining the English and non-English (comparable datasets) (in 2017 US dollars).

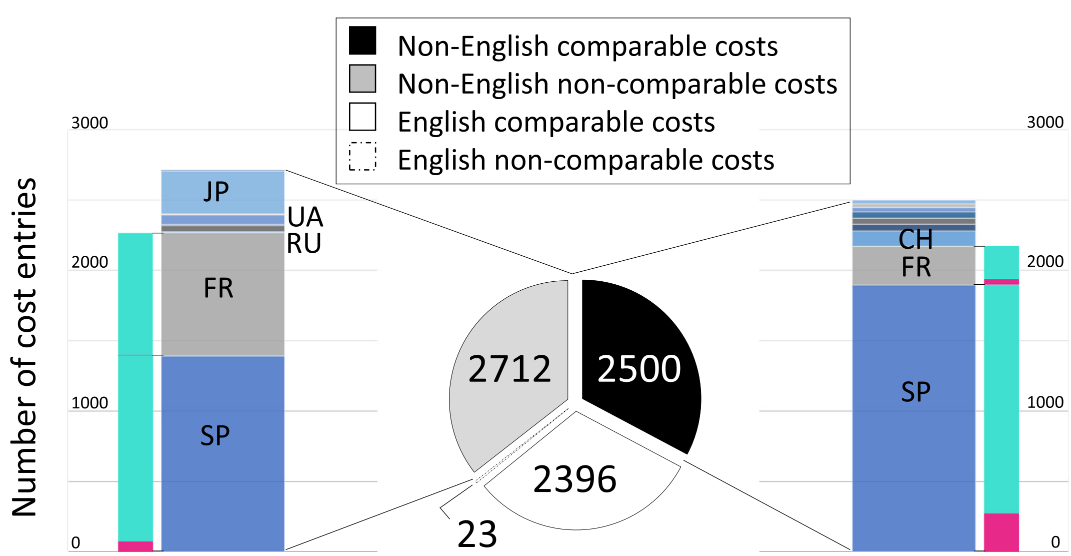

Fig. 1. Number of cost entries of invasive species in non-English languages and in English (InvaCost database), distinguishing comparable and non-comparable datasets. For each of the non-English datasets, the number of cost entries by language are represented in the bar diagrams. Languages shown: SP, Spanish; FR, French; JP, Japanese; UA, Ukranian; RU, Russian; CH, Chinese. For SP and FR, turquoise and magenta bars distinguish entries from Spain and France (turquoise) and entries from Spanish-speaking South American countries and francophone African countries (magenta).

2.5. Differences in invasive species recorded in both databases

First, we compared invasive species reported in the non-English and English databases, using both the comparable and the complete datasets. Specifically, we examined whether species with costs in the non-English datasets were already included in the English dataset (shared species), or whether they were only included in the non-English datasets; and similarly for species with costs in the English dataset. Since some cost entries were assigned to multiple species simultaneously, we obtained the complete list of species having reported costs, as follows: we expanded all species contained in these cost entries (in the column "Species"), so that each species was individually considered. In order to avoid over-estimations, we also removed subspecies or genus when the corresponding species was present in the same dataset (e.g., we removed *Canis lupus dingo* when *C. lupus* was already present; we removed *Ludwigia* sp. if any *Ludwigia* species, such as L. *grandiflora*, was present). When comparing both species lists (non-English and English), if a genus was present in a list (but the species name was missing) while in the other list there were one or more than one species, we considered that only one species was shared between the two databases (e.g., *Rubus* sp. appears in the English dataset, while both *Rubus glaucus* and *R. constrictus* do in the non-English dataset; so only one shared species was counted).

Second, we quantified the contribution of costs reported from species in the non-English relative to the English dataset, and graphically mapped the results. Using only the comparable datasets, we developed an index that reflects the difference between the number of species by country in the non-English and English datasets (that is, for each country, we subtracted the number of species in the English dataset from the number of species in the non-English dataset). This index is positive when the number of species in the non-English dataset is higher than those in the English dataset for that particular country; or negative otherwise. In this analysis, species costs reported for Great Britain, England and Scotland were considered as belonging to a single country: the United Kingdom. Additionally, overseas territories are represented in their main country territory (e.g., Martinique or French Guiana are represented in France).

3. Results

3.1. The relevance of non-English documents reporting costs

The non-English database includes 5212 cost entries from 356 documents, which covered 10 out of the 15 non-English languages examined in this study (Fig. 1, doi:https://doi.org/10.6084/m9.figshare.12928136). Despite our extensive search efforts, we could not find cost reports in five of the languages we considered. These languages are Arabic and four languages used in India: Hindi, Telugu, Tamil, and Bengali. Some documents obtained directly from Spanish official managers were written in two co-official languages: Catalan and Galician. From the 356 documents collected, 30 were unpublished materials ($N = 1635$ cost entries), and 149 documents were published after 2017 ($N = 1850$). This resulted in a total of 2500 entries that were comparable to the English entries (i.e., the comparable non-English dataset) and 2712 entries were not comparable (Fig. 1). In general, Spanish and French dominated over the other languages, and mostly Spanish from Spain (>85%) rather than from Latin American countries, and French from France (>80%) rather than from francophone African countries.

From the English database, the non-comparable dataset consisted of 15 cost entries from six unpublished documents (i.e., "Type_of_material" column: "Unpublished material") and eight entries from five documents published in 2018. The English comparable dataset had therefore 2396 entries from 838 documents.

In relation to the total economic cost, the non-English comparable dataset resulted in US$ 214 billion (sum of the annual estimated costs), and when including the non-comparable dataset, the contribution from the non-English database resulted in US$ 234 billion. In comparison, a refined version of the English database led to about US $ 1.288 trillion, considering either the comparable English dataset or both comparable and non-comparable English datasets. Thus, considering non-English data increased the English-based global cost estimates of invasions by 16.6% (only the comparable dataset) or by 18.1% (the full non-English database).

3.2. Relationship between the number of cost entries and the proportion of English speakers

We found a negative relationship between the number of cost entries (log transformed) and the proportion of English speakers per country (correlation coefficient $r = -0.216$, $N = 26$; Fig. 2), suggesting that countries with a low proportion of English speakers published more in their native languages. This pattern is highly driven by the Spanish and French-speaking countries, from where many of our cost entries originated. European countries followed this trend, with countries with a higher proportion of English speakers, such as the Netherlands (68.3%), Germany (44.1%), and Belgium (48.6%) having fewer documents published in their own language compared with countries with a lower proportion of English speakers, such as France (24.3%) or Spain (20.7%). The rest of the countries were grouped as follows: African countries with a variable range of English speakers, but very few non-English cost entries, South American countries with an average number of cost entries and low proportion of English speakers (<10%), and Asian countries (i.e., China and Japan) with a high number of entries and a low proportion of English speakers (<0.05%) (Fig. 2).

3.3. Differences in cost descriptors

Compared to the English dataset, the number of entries in the non-English dataset was significantly higher for European countries, and significantly lower for countries from Africa, North and Central America, and Oceania and Pacific Islands (Fig. 3a, Appendix S4). The number of entries in the non-English dataset was significantly higher at the local scale, but significantly lower at the country and global scales compared to the English dataset (Fig. 3b, Appendix S4). With respect to the environment where the cost occurred, the number of cost entries was not significantly different between the non-English and English datasets (Fig. 3c, Appendix S4). The number of entries in the non-English dataset was significantly higher for the authorities and stakeholders, but significantly lower for agriculture, forestry, and public and social welfare sectors than in the English dataset (Fig. 3d, Appendix S4). The number of entries in the non-English dataset was significantly higher for management costs, but significantly lower for damage costs than in the English dataset (Fig. 3e, Appendix S4). Finally, we obtained a significantly higher number of entries for invasive alien plants in the non-English dataset while significantly lower entries for Chordata and Arthropoda, and no difference for Mollusca (Fig. 3f, Appendix S4).

Regarding the differences in the spatial scale of cost entries between non-English and English comparable datasets, we observed that only African countries had entries (in French) at the supranational scale (Fig. S1a). Those costs had a higher proportion than those in the English database (12% vs. 5.3% respectively). In the English database, the proportion of cost entries at the local scale or at the country level were very similar (48.5 and 46.2% respectively) (Fig. S1b). Besides African countries, there were many countries with most entries at the local scale (e.g., 100% for Spain, Ecuador, and Cuba; >90% for Ukraine, France, and Belgium); while few countries had costs mostly at the country level (e.g., >85% for Russia, the Netherlands, and Colombia), or with a proportion of costs more equally distributed between the country and the local scale (e.g., Chile: 60 vs. 40%; Argentina: 72 vs. 28%; or Germany: 76 vs. 23% respectively) (Fig. S1a).

Concerning the cost figures, we observed that non-English economic costs were very important at the geographic level for South America,

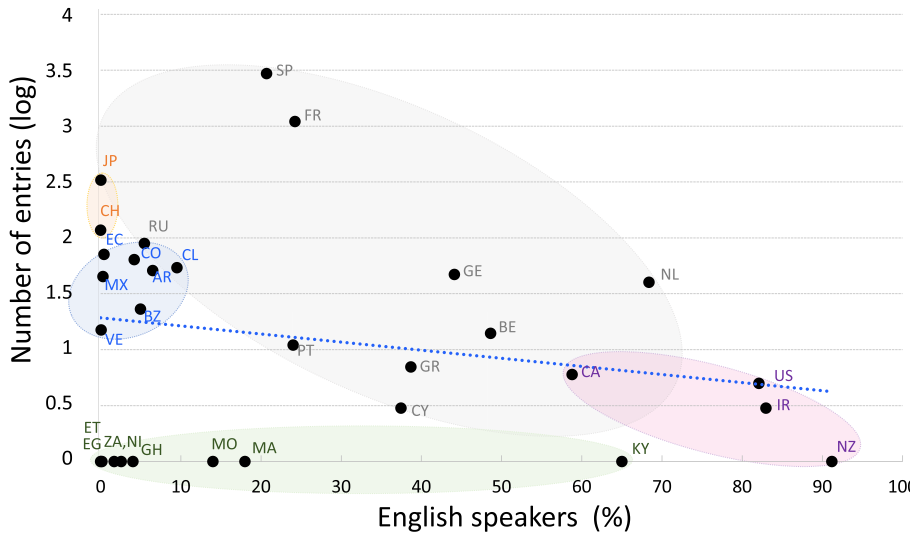

Fig. 2. Relationship between the number of entries of economic costs of invasive species in non-English languages and the percentage of English speakers in each country. Regression line is marked in blue. Countries are grouped according to their occupied convex hull area and encompassed with standard ellipses (considering confidence intervals = 95% of their respective data); European countries in grey, African countries in green, South American countries in blue, Asian countries in yellow, and English speaking countries in pink. Country abbreviations: AR, Argentina; BE, Belgium; BZ, Brazil; CA, Canada; CL, Chile; CH, China; CO, Colombia; CY, Cyprus; EC, Ecuador; EG, Egypt; ET, Ethiopia; FR, France; GE, Germany; GH, Ghana; GR, Greece; IR, Ireland; JP, Japan; KY, Kenya; MA, Madagascar; MX, Mexico; MO, Morocco; NL, Netherlands; NI, Nigeria; NZ, New Zealand; PT, Portugal; RU, Russia; SP, Spain; US, United States; VE, Venezuela; ZA, Zambia.

and at the taxonomic level for invasive alien plants (Fig. 3). Costs for South America constituted 53.7% of the total non-English cost and 56% when comparable non-English and English costs were combined (Fig. 3g). Non-English costs were also relatively higher at the local scale (US$ 24 billion, Fig. 3h), for Chordata (US$ 28 billion, Fig. 3l), when occurring in semi-aquatic environments (US$ 1.5 billion, Fig. 3i), and when spent by authorities and stakeholders (US$ 11 billion, Fig. 3j). Costs for invasive plants in non-English amounted to US$ 120 billion, which constituted 67.2% of the total non-English costs, and 31% when non-English and English costs were combined (Fig. 3l).

3.4. Differences in invasive alien species recorded in both databases

The comparable and non-comparable datasets of the English database had the same species lists. In the non-English database, the non-comparable dataset had a higher number of species than the comparable dataset, resulting overall in more species being listed in the non-English than in the English database. The species lists of the two comparable (English and non-English) datasets shared only 19% of species, and species brought up by the search in non-English languages represented 44% of the total (249 out of 569 species; Fig. 4a). When considering the full non-English database (comparable and non-comparable datasets), the percentage of shared species remained 19%, but amounted to 54% for species reporting cost only in non-English languages (384 out of 705 species; Fig. 4b).

The difference in species per country between non-English and English datasets varied from −102 to 132 species. Positive values represent more species in the non-English dataset, which was found in 18 countries, with the highest value in Spain (Fig. 4c). Negative values represent more species in the English dataset, which was found in 5 countries, with the highest (negative) value in the USA (Fig. 4c). Additionally, for countries with species in one dataset only, positive values were found in 15 countries (i.e., reporting costs for species only in non-English languages) and negative values occurred in 59 countries (i.e., reporting costs for species only in English). In both cases, the extreme values were lower: a total of 43 species was the maximum number of species with reported costs only in the non-English dataset, and was found from Russia; and − 56 species was the minimum number of species with reported costs only in English and was found from Australia (Fig. 4c).

4. Discussion

The relevance of considering non-English languages was substantiated as non-English data: (i) increased the content of the published English database by more than 100% (2500 non-English vs. 2396 English entries), (ii) increased the global cost estimate of invasions by 16% (by ~US$ 214 billion), and (iii) provided costs for 249 new species and 15 new countries. In addition, 135 other species were found by considering 2712 cost entries from non-published sources, directly obtained from practitioners or managers and/or from documents produced after 2017. Moreover, these gaps resulted in an underrepresentation of cost entries (i) associated with European countries, (ii) measured at the local scale, (iii) impacting primarily authorities and stakeholders, (iv) corresponding to management, and/or (v) reported for plants. In summary, relying on data exclusively published in English has some important implications, particularly when the concerned discipline has a strong applied component, for e.g., through informing policy on invasions.

4.1. Knowledge gaps when considering only English in the costs of invasive species

The large number of costs of invasive species reported exclusively in non-English languages highlights the importance of increasing efforts to capture all available literature beyond English only. This is in agreement with previous findings that provide evidence for gaps in global assessment and ecological patterns, e.g., the assessments of IUCN population status of endangered taxa (Amano et al., 2016) or the use of interviews in conservation biology (Young et al., 2018). Here, we also demonstrated that relying on only English sources results in a distorted picture of lower invasion costs. For example, management expenses were under-represented in English versus non-English datasets. This could be explained by the fact that a third of the cost entries in the non-English database were obtained from local managers and/or practitioners. Also, it could depend on how local funds are distributed, with priority on management rather than on damage evaluation, which would require additional resources and scientific skills (and would likely be reported in English). The gaps reported are in line with those of Zenni et al. (2017), whose work supports the notion that invasion biologists should work more intensively with managers and practitioners,

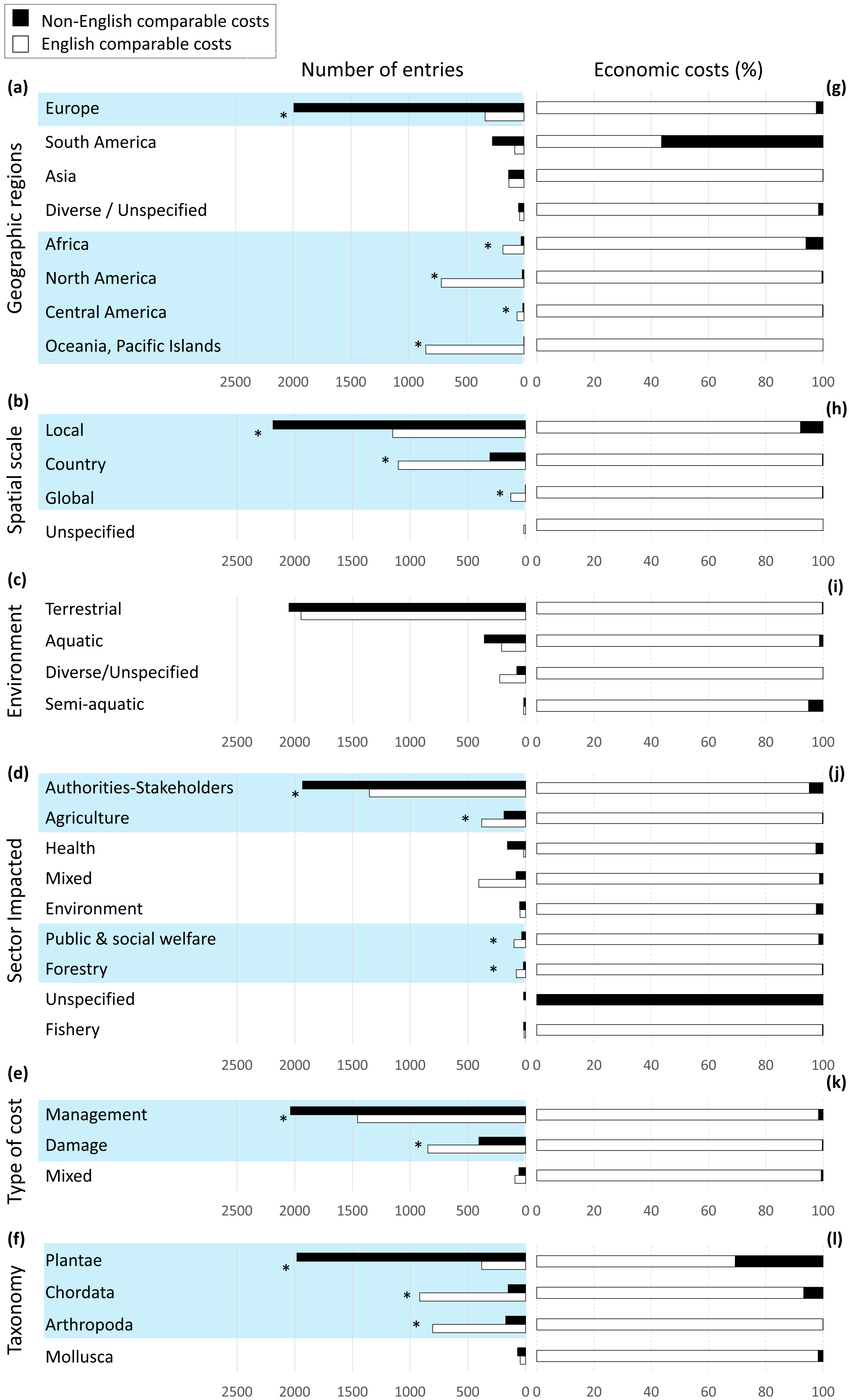

Fig. 3. Number of entries (a, b, c, d, e, f) and relative amount (g, h, i, j, k, l) of economic costs of invasive alien species in non-English languages and in English (from InvaCost database), by (a, g) geographic regions where the cost occurred, (b, h) spatial scale of the cost, (c, i) environment where the cost occurred, (d, j) impacted sector of the cost, (e, k) type of cost, and (f, l) main taxonomic groups. Significant differences in the number of entries between non-English and English are marked with asterisks and highlighted in blue.

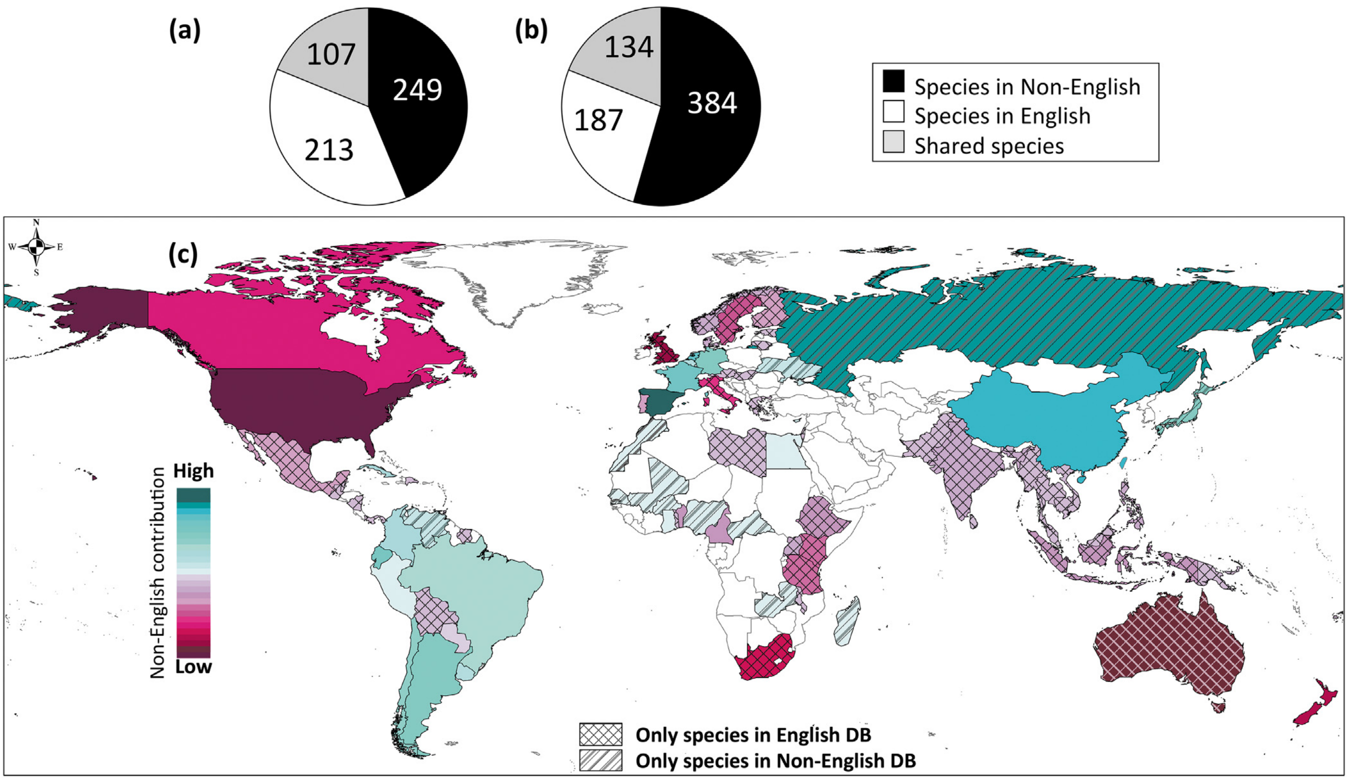

Fig. 4. Number of shared and unshared species between the non-English database and the English database (InvaCost), considering (a) only comparable data ($N = 569$ species) and (b) all data in non-English languages ($N = 705$ species). (c) The map shows the number of species that the non-English comparable dataset contributed minus the total number of species of the English comparable dataset by country (turquoise-magenta scale). Countries having only species in the non-English comparable dataset are marked with stripes and countries having only species in the English comparable dataset are marked with a grid. The borders illustrated in the map may not represent the current political reality.

and more broadly, with society as a whole. Similar gaps were also found in other applied ecological global databases, such as the Forest Global Earth Observatory (ForestGEO: https://forestgeo.si.edu/) and the Nutrient Network (Nutnet: http://www.nutnet.umn.edu/) (Nuñez et al., 2019).

We also found marked differences in the number of cost entries among languages. This uneven geographic distribution is similar to what Amano et al. (2016) reported in the context of biodiversity and conservation, when comparing 16 major languages. These researchers found that 64.4% of the documents were published in English, followed by 12.6% in Spanish, 10.3% in Portuguese, 6% in Chinese, and 3% in French. In our case, and considering together the English and non-English databases, we obtained 43% of cost entries in Spanish, 31.7% in English, 15% in French, 4.3% in Japanese, and 1.5% in Chinese. We observed that Spanish and French represented a large proportion of the cost entries that were not reported in English. Not surprisingly, countries with a high proportion of English speakers were more represented in the English database compared to the non-English database. In multilingual countries, several of them located in Africa or Asia, publishing in the native tongue(s) may not be the most practical or efficient. Indeed, there may be several native tongues within a single country, making it complicated to opt for consensual non-English language(s) to report information. For example, while Kenya and the Netherlands have a similar proportion of English speakers, the non-English speakers in Kenya are linguistically more diverse, where about 70 languages are spoken, whereas for the Netherlands the remaining almost entirely speak Dutch (Eberhard et al., 2020). In addition, other implications, such as political or historical ones, may explain low reported costs in some languages/countries. For example, the long colonial history and a large middle class that is fluent in English in India could explain the predominant use of this language in publications (Fazey et al., 2005).

Some languages have been targeted to attempt increasing the visibility of papers written in that language. For example, Tao et al. (2018) claimed that 79 million papers have been published in Chinese since 1979, some of them describing important advances that remain unseen by Western researchers. Acknowledging these omissions, along with the fact that 1.39 billion people speak some dialects of Chinese, the journal *Conservation Biology* announced that their papers will include abstracts in Chinese from 2017 onwards (Conservation Biology, 2017). Other journals in the field are following suit, such as *Biological Invasions*, or the *Journal of Applied Ecology* which translated the 'Guide to Getting Published' in Chinese and is promoting abstracts in local languages (Nuñez et al., 2019).

4.2. Ignoring non-English data biases cost patterns for invasive species

We identified the biases from considering exclusively English sources when reporting global trends in costs. First, we identified a geographic bias, both in the number of entries and in the magnitude of costs, in agreement with a previous hypothesis (Zenni et al., 2017). The non-English search provided substantially more entries for Europe, especially Spain and France. Concerning the amount of money they represented, costs reported in non-English from South America and, to a lesser extent, from Africa, were highly relevant. This could be the result of the increasing development of national strategies and research budgets for the control of invasive alien species (Zenni et al., 2017). In fact, the recent release of InvaCost_3.0 (Diagne et al., 2020b), which included English as well as non-English data, permitted to show that for some continents and countries economic assessments of invasive species mostly rely on non-English data. For instance, in Central and South America over 40% of cost estimates have been published in non-English languages (Heringer et al., in press); among those, in

Ecuador 51% of all costs have been published in Spanish (Ballesteros-Mejia et al., in press). A similar situation is observed in Asia (reviewed in Liu et al., in press), where all cost estimates from Japan have been reported in Japanese (Watari et al., in press), and cost entries from Russia have predominantly originated from Russian-language documents (Kirichenko et al., in press).

Costs reported at larger spatial scales were more frequent in the English database, whilst the non-English search added significantly more cost entries at the local scale (~8% of the total money spent on combining English and non-English databases). This is likely due to local researchers and practitioners being more informed on a local level, but maybe not speaking English, or not being encouraged to publish their data in traditional scientific outlets (Nuñez et al., 2019). Some journals have launched specific spaces for practitioners to publish their opinions and examples of best practice (Hulme, 2011). Improved connections with other scientists or practitioners can help promote good practices between localities with similar applied problems (Nuñez et al., 2019). In fact, we detected costs for similar concepts in different regions or sites, showing that although local discoveries of efficient control interventions for invasive species can be relevant for successful control elsewhere, the language barrier may have applied consequences. It is apparent that a stronger link is required between researchers and stakeholders to increase the international visibility of local knowledge (Sutherland et al., 2019). For example, BiodivERsA attempts to facilitate this by forming a network of funding organizations to support biodiversity research (Durham et al., 2014). The non-English database can constitute an essential tool for practitioners (e.g., searching for cost information associated with specific management types actions or specific species), policy makers (e.g., searching for damage-related costs in order to motivate, guide and/or prioritize prevention or response actions towards invasive species), and scientists (e.g., macroecological analyses, data syntheses, or meta-analyses).

Our results also show that an English-only search missed a large number of cost entries impacting authorities and stakeholders. Species invasions are context-dependent, with developing countries typically facing challenges different to those by more developed countries. Therefore, the way invasive species are perceived by local populations, stakeholders and leaders, as well as funders, including the nature of their costs, might differ between countries (Nuñez et al., 2019). For example, the predominant number of costs from Spain and France seem to be primarily related to management costs, whereas a higher amount of costs reported in Spanish corresponded to South America and seemed to be related to damage costs. Nuñez and Pauchard (2010) found that the scarcity of scientific reports on invasive species in developing countries was associated with low funding for ecological research in comparison to other disciplines closely related to medicine, water shortage and food supply. This may explain the high proportion of reported costs related to agriculture in South American countries.

Finally, the number of cost entries coming from invasive plant species reported in non-English languages also contributed significantly, and amounted to ~30% of the total money associated with plants when considering both English and non-English datasets. Local knowledge on plants could be higher than for other taxa, as plants are resources for medicine, food, or animal breeding, and plant invasions dominate the English literature in invasion science (Lowry et al., 2013; Carboneras et al., 2018).

5. Conclusions and perspectives

The aim of this study was not to exhaustively search for information on the economic costs of biological invasions in all possible languages. Rather, we aimed at showing that sources beyond English literature are available and rich in primary data. In fact, the amount of retrieved data was dependent on multiple factors such as country or language specificities; for example, some countries have policies to make data publicly available, or have specific budgets for invasive species, while others do not. In some cases, we also observed a kind of domino effect, e.g., in France, experts increasingly sent us new cost data as they heard about the project. Also, our research was limited to the languages spoken by the authors, and many languages have not been searched at all and could provide much additional data. Non-English sources on invasive species that are often overlooked mostly include the grey literature and unpublished reports from practitioners, resource managers, and researchers. Therefore, we demonstrated the importance of multi-language collaborations in biological invasions, which are in essence an international issue. The non-English database now complements the original English database in an updated version of InvaCost (InvaCost_3.0, Diagne et al., 2020b), and we hope that this study will encourage others that aim to bridge linguistic barriers. The benefits of these collaborations are clear: improving management efficiency, decreasing research effort, and adequately guiding policy. In that way, we have provided the Appendix S5, with abstracts and figure legends in several languages, as a proof of concept for promoting the overall message of this study. We hope that our results and our suggestions will encourage future proposals to alleviate language barriers as a means to enrich scientific knowledge, and in particular, lead to a reduction of economic costs with improved management strategies of invasive alien species.

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2020.144441.

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Authors contributions

FC, CD and EA conceived the idea. CL and WX compiled the Chinese data; DRe, CAKM, LBM, GD and TA compiled the French data; EA, LBM, VGD, MN and DRo compiled the Spanish data; NK and EAk compiled the Russian data; GH and CC compiled the Portuguese data; PH and MG compiled the German data; LV compiled the Dutch data; MG compiled the Ukrainian data; MK compiled the Greek data; YW compiled the Japanese data; AKB performed the Indian languages search; and AT performed the Arabic search. CD, LBM and EA refined and standardized the data. EA took the lead in writing the original draft of the article with inputs from all co-authors. All authors read and approved the final version of the manuscript.

CRediT authorship contribution statement

Elena Angulo: Conceptualization, Resources, Methodology, Writing – original draft, Writing – review & editing. **Christophe Diagne:** Conceptualization, Resources, Methodology, Writing – review & editing. **Liliana**

Ballesteros-Mejia: Resources, Methodology, Writing – review & editing. **Tasnime Adamjy:** Resources. **Danish A. Ahmed:** Resources, Writing – review & editing. **Evgeny Akulov:** Resources. **Achyut K. Banerjee:** Resources, Writing – review & editing. **César Capinha:** Resources, Writing – review & editing. **Cheikh A.K.M. Dia:** Resources, Writing – review & editing. **Gauthier Dobigny:** Resources, Writing – review & editing. **Virginia G. Duboscq-Carra:** Resources. **Marina Golivets:** Resources, Writing – review & editing. **Phillip J. Haubrock:** Resources, Writing – review & editing. **Gustavo Heringer:** Resources, Writing – review & editing. **Natalia Kirichenko:** Resources, Writing – review & editing. **Melina Kourantidou:** Resources, Writing – review & editing. **Chunlong Liu:** Resources, Writing – review & editing. **Martin A. Nuñez:** Resources, Writing – review & editing. **David Renault:** Resources, Writing – review & editing. **David Roiz:** Resources, Writing – review & editing. **Ahmed Taheri:** Resources, Writing – review & editing. **Laura N.H. Verbrugge:** Resources, Writing – review & editing. **Yuya Watari:** Resources, Writing – review & editing. **Wen Xiong:** Resources. **Franck Courchamp:** Conceptualization, Resources, Methodology, Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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SPANISH Title / Título

- EN: **Non-English languages enrich scientific knowledge: the example of economic costs of biological invasions**
- SP: **Las lenguas no inglesas enriquecen el conocimiento científico: ejemplo de los costes económicos de las invasiones biológicas.**

Authors / Autores

Elena Angulo, Christophe Diagne, Liliana Ballesteros-Mejia, Tasnime Adamjy, Danish Ali Ahmed, Evgeny N. Akulov, Achyut Kumar Banerjee, César Capinha, Cheikh A. K. M. Dia, Gauthier Dobigny, Virginia G. Duboscq-Carra, Marina Golivets, Phillip Joschka Haubrock, Gustavo Heringer, Natalia Kirichenko, Melina Kourantidou, Chunlong Liu, Martin A. Nuñez, David Renault, David Roiz, Ahmed Taheri, Laura Verbrugge, Yuya Watari, Wen Xiong, Franck Courchamp.

Translated by the authors / Traducido por los autores:
Liliana Ballesteros-Mejia / Elena Angulo

Abstract / Resumen en español:

- El uso exclusivo del inglés para divulgación de resultados de investigación puede afectar la comunicación entre investigadores y gestores, o políticos cuya lengua materna es diferente al inglés. Esta barrera en el conocimiento científico y en la transferencia de los datos puede potencialmente causar lagunas en el conocimiento y sesgos a la hora de estimar patrones globales en todos los campos de la ciencia.
- Para demostrar estos hechos compilamos datos sobre el costo económico de las invasiones biológicas a nivel global, a partir de documentación en quince idiomas diferentes al inglés, y los comparamos con datos recopilados en la base de datos InvaCost (i.e. el repositorio más actualizado sobre los costos de las invasiones biológicas a nivel mundial, en inglés).
- La comparación de las dos bases de datos (~7 500 entradas en total) reveló que las fuentes no inglesas: (i) capturan un mayor número de datos que si se usaran sólo fuentes inglesas (2 500 vs. 2 396 entradas); (ii) ofrecen datos para 249 nuevas especies y 15 nuevos países que no son reportados en inglés; (iii) incrementan las estimaciones globales en inglés de los costos de las invasiones biológicas en un 16.6% (US$ 214 mil millones de los 1.288 billones de la base de datos en inglés). La base de datos no inglesa incluye además de 2 712 entradas complementarias, que no son directamente comparables con los datos ingleses, ya que la mayoría fueron obtenidos a partir del contacto directo con los gestores, revelando así el valor de la comunicación entre investigadores y gestores.
- Demostramos además que las lagunas causadas por la omisión de los datos no ingleses, causaron sesgos significativos en la distribución de los costos, tanto geográficamente como taxonómicamente, al igual que en cuanto al tipo de costo y al sector que atañe. Los costos para Europa, a escala local, y particularmente relativos al manejo de especies invasoras, estuvieron sub-representados en la base de datos en inglés.
- Por lo tanto, combinar datos científicos en inglés con datos que proceden de otros idiomas es fundamental, y mejora la integridad de los datos. Además, mejora potencialmente la eficacia de las estrategias de manejo, la coordinación entre expertos (investigadores y gestores) y la colaboración entre países.

Figures / Figuras

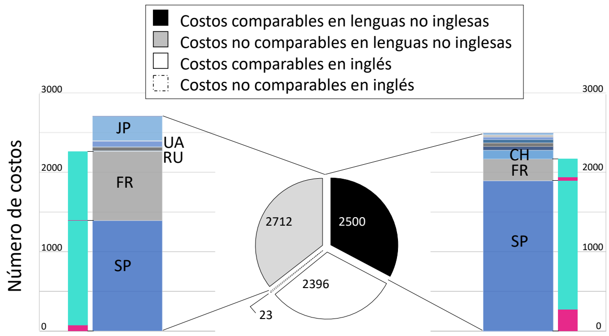


Fig. 1. **Número de entradas de costos económicos de especies invasoras en lenguas no inglesas, y en inglés (base de datos InvaCost), distinguiendo los conjuntos de datos comparables y no comparables.** Para cada conjunto de datos en lenguas no inglesas, el número de entradas por idioma están representados en los diagramas de barras. Idiomas mostrados: SP, Español, FR, Francés; JP, Japonés; UA, Ucraniano; RU, Ruso; CH, Chino. Para SP y FR, las barras turquesa y magenta distinguen las entradas de España y Francia (turquesa) de las de Sudamérica y los países francoparlantes de África (magenta).

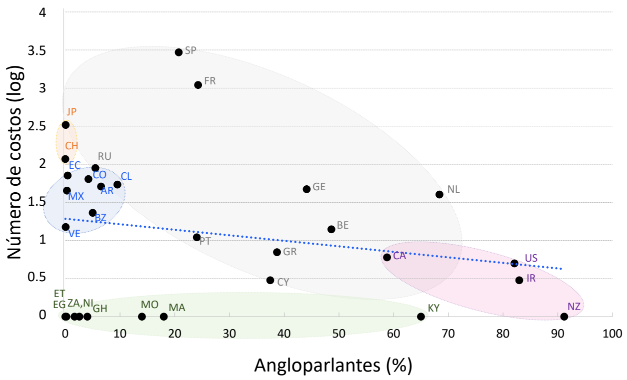


Fig. 2. **Relación entre el número de entradas de costos económicos de especies invasoras en lenguas no inglesas y el porcentaje de angloparlantes en cada país.** La línea de regresión está marcada en azul. Los países están agrupados según el área convexa de hull, teniendo en cuenta las elipses estándar (considerando los intervalos de confianza al 95% de sus datos): los países Europeos en gris, los Africanos en verde, los Sudamericanos en azul, los Asiáticos en amarillo y los de habla inglesa en rosa. Abreviaturas de los países: AR, Argentina; BE, Bélgica; BZ, Brasil; CA, Canadá; CL, Chile; CH, China; CO, Colombia; CY, Chipre; EC, Ecuador; EG, Egipto; ET, Etiopía; FR, Francia; GE, Alemania; GH, Gana; GR, Grecia; IR, Irlanda; JP, Japón; KY, Kenia; MA, Madagascar; MX, México; MO, Marruecos; NL, Holanda; NI, Nigeria; NZ, Nueva Zelanda; PT, Portugal; RU, Rusia; SP, España; US, Estados Unidos; VE, Venezuela; ZA, Zambia.

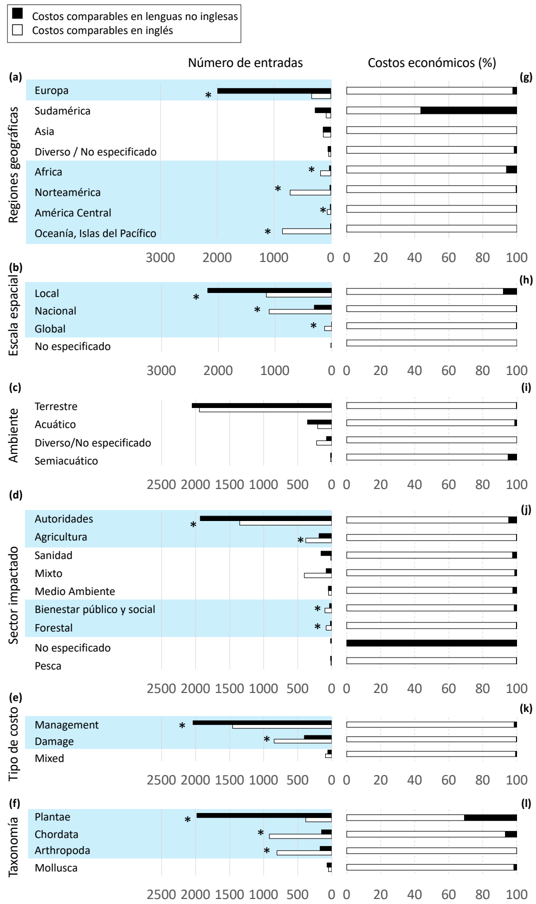
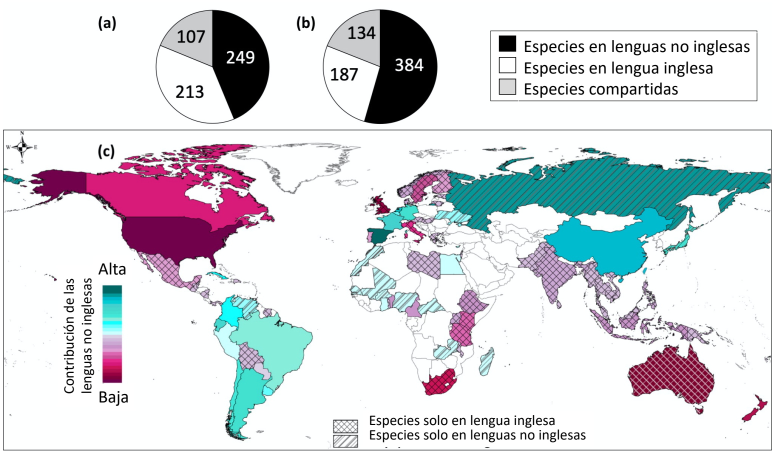


Fig. 3. **Número de entradas de costos (a,b,c,d,e,f) y costo relativo (g,h,i,j, k, l) de las especies invasoras en lenguas no inglesas y en inglés** (a partir de la base de datos InvaCost), distribuidas por (a,g) regiones geográficas donde el costo ocurre, (b,h) escala espacial del costo, (c,i) ambiente donde ocurre el costo, (d,j) sector impactado por el costo, (e,k) tipo de costo, y (f,l) grupos taxonómicos principales. Las diferencias significativas en el número de entradas entre la lengua no inglesa y el inglés están marcadas con asteriscos y destacadas en azul.

Fig. 4. **Número de especies compartidas o no entre los conjuntos de datos en lenguas no inglesas y en inglés** (InvaCost), considerando (a) sólo los datos comparables (n = 569 especies) and (b) todos los datos en lenguas no inglesas (n = 705 especies). (c) El mapa muestra el número de especies con las que el conjunto de datos en lenguas no inglesas contribuye al número total de especies del conjunto de datos en inglés por país (escala turquesa-magenta). Los países que tienen sólo especies en los conjuntos de datos en lenguas no inglesas están marcados con rayas diagonales y los países que tienen especies sólo en el conjunto de datos inglés están marcados con rombos. Las fronteras ilustradas en el mapa pueden no representar la realidad política actual.



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RESEARCH ARTICLE

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Economic costs of invasive alien species in Spain

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Abstract

Economic assessments for invasive alien species (IAS) are an urgent requirement for informed decision-making, coordinating and motivating the allocation of economic and human resources for the management of IAS. We searched for economic costs of IAS occurring in Spain, by using the InvaCost database and requesting data to regional governments and national authorities, which resulted in over 3,000 cost entries. Considering only robust data (i.e. excluding extrapolated, potential (not-incurred or expected) and low reliability costs), economic costs in Spain were estimated at US$ 261 million (€ 232 million) from 1997 to 2022. There was an increase from US$ 4 million per year before 2000 to US$ 15 million per year in the last years (from € 4 to 13 million). Robust data showed that most reported costs of IAS in Spain (> 90%) corresponded to management costs, while damage costs were only found for 2 out of the 174 species with reported costs. Economic costs relied mostly on regional and inter-regional administrations that spent 66% of costs in post-invasion management actions, contrary to all international guidelines, which recommend investing more in prevention. Regional administrations unequally reported costs. Moreover, 36% of the invasive species, reported to incur management costs, were not included in national or European regulations (i.e. Black Lists), suggesting the need to review these policies; besides, neighbouring regions seem to manage different groups of species. We suggest the need of a national lead agency to effectively coordinate actions, facilitate communication and collaboration amongst regional governments, national agencies and neighbouring countries. This will motivate the continuity of long-lasting management actions and the increase in efforts to report IAS costs by regional and inter-regional managers which will adequately provide information for future budgets gaining management effectiveness.

Abstract in Spanish

Costos económicos de las especies exóticas invasoras en España. Las evaluaciones de los costos de las especies exóticas invasoras (EEI) son un requisito urgente para informar en la toma de decisiones, coordinar y motivar la asignación de recursos económicos y humanos para la gestión de las EEI. En este estudio, buscamos información sobre los costos económicos de las EEI en España, usando la base de datos InvaCost, y solicitando datos a las administraciones regionales y nacionales, lo que resultó en más de 3000 entradas de costos. Considerando solamente los costos robustos (es decir, excluyendo los costos extrapolados, potenciales (no observados o esperados) o de baja fiabilidad), los costos económicos de EEI en España fueron estimados en 261 millones de dólares americanos (US$, 232 millones de €) entre 1997 y 2022. Observamos un incremento desde 4 millones de US$ por año antes del año 2000 hasta 15 millones de US$ por año en los últimos años (de 4 a 13 millones de €). Los datos robustos indicaron que la mayoría de los costos reportados en España (>90%) correspondieron a costos de gestión, mientras que los daños económicos sólo fueron observados para 2 de las 174 especies con costos reportados. Los costos económicos correspondieron principalmente a las administraciones regionales o inter-regionales que gastaron 66% de los costos en acciones de manejo después de la invasión, al contrario de lo recomendado en las guías internacionales, que es invertir más en prevención. Las administraciones regionales reportaron de manera desigual los costos. En este sentido, el 36% de las especies invasoras reportadas con costos de gestión no estaban incluidas en las leyes nacionales o Europeas (listas negras), lo que sugiere la necesidad de revisar esas leyes; además, las regiones vecinas parecen gestionar diferentes grupos de especies. Sugerimos la necesidad de una agencia que coordine las acciones de manera efectiva a nivel nacional, y facilite la comunicación y la colaboración entre gobiernos regionales, agencias nacionales y países vecinos. Esto motivará la continuidad de las acciones de gestión a largo plazo, que proveerán de información adecuada a los futuros presupuestos, ganando en efectividad en la gestión.

Abstract in French

Coûts économiques des espèces exotiques envahissantes en Espagne. Les évaluations économiques des espèces exotiques envahissantes (EEE) sont une nécessité urgente pour motiver et orienter les actions des autorités et décideurs en matière de gestion des EEE. Nous avons recherché les coûts économiques des EEE en Espagne via (i) la base de données InvaCost et (ii) des sollicitations adressées aux gouvernements régionaux et autorités nationales. Ce travail a abouti à l’obtention de plus de 3000 données individuelles de coûts. Si l’on ne tient compte que des données considérées comme les plus robustes (c’est-à-dire lorsqu’on exclut les coûts extrapolés, potentiels (i.e. prédits ou non-observés) et/ou peu fiables d’un point de vue méthodologique), les coûts économiques en Espagne ont été estimés à 261 millions de dollars américain (232 millions d’euros) entre 1997 et 2022. Il y a eu une augmentation annuelle de 4 millions de dollars avant 2000, puis de 15 millions de dollars par an ces dernières années. Ces données robustes ont montré que la plupart des coûts déclarés des EEE en Espagne (> 90%) correspondaient aux coûts de gestion, tandis que les coûts des dommages n’ont été constatés que pour 2 des 174 espèces dont les coûts étaient reportés. Nous avons montré que les coûts économiques reposaient principalement sur les administrations régionales et interrégionales; celles-ci ont consacré 66% des coûts enregistrés aux actions de gestion post-invasion, contrairement aux directives internationales qui recommandent d’investir davantage dans la prévention. Les administrations régionales ont déclaré les coûts de manière inégale. De plus, 36% des espèces envahissantes, déclarées comme entraînant des coûts de gestion, n’étaient pas incluses dans les réglementations nationales ou européennes (c’est-à-dire les listes noires). Ceci suggère la nécessité de revoir ces politiques; en outre, les régions voisines semblent gérer différents groupes d’espèces. Nous suggérons la nécessité d’une agence nationale ‘chef de file’ pour coordonner efficacement les actions, faciliter la communication et la collaboration entre les gouvernements régionaux, les agences nationales et les pays voisins. Cela motivera la continuité des actions de gestion à long terme et l’intensification des efforts pour rendre compte des coûts des EEE par les gestionnaires régionaux et interrégionaux. Tout ceci permettra de fournir des informations adéquates pour les budgets futurs, avec un bénéfice certain pour l’efficacité des mesures de gestion.

Abstract in Italian

Costi economici delle specie aliene invasive in Spagna. Le valutazioni economiche delle specie aliene invasive (SAI) sono un requisito urgente per processi decisionali informati, e per coordinare e motivare l'allocazione di risorse economiche e umane per la gestione delle SAI. Usando la banca dati InvaCost e richiedendo i dati ai governi regionali e alle autorità nazionali, abbiamo cercato i costi economici delle SAI presenti in Spagna, ottenendo come risultato 3000 voci di costi. Considerando solo i dati robusti (i.e. escludendo i costi estrapolati, potenziali (non sostenuti od attesi) e con bassa attendibilità), i costi economici in Spagna dal 1997 al 2022 stati stimati a 261 milioni di $ americani (232 milioni di €). C'è stato un aumento da 4 milioni di $ americani all'anno prima del 2000 a 15 milioni di $ americani all'anno negli ultimi anni (da 4 a 13 milioni di €). I dati robusti hanno mostrato che la maggior parte (> 90%) dei costi riportati per le SAI in Spagna corrispondeva a costi di gestione, mentre i costi riferiti ai danni sono stati trovati solo per 2 delle 174 specie con costi riportati. I costi economici si basano soprattutto sulle amministrazioni regionali e interregionali, che hanno speso il 66% dei costi in azioni di gestione post invasione, contrariamente a tutte le linee guida internazionali, che raccomandano di investire di più nella prevenzione. Le amministrazioni regionali hanno riportato i costi in modo diseguale. Inoltre, il 36% delle specie invasive per cui sono riportati costi di gestione non era incluso nei regolamenti nazionali o europei (i.e. Liste Nere), il che suggerisce il bisogno di rivedere queste politiche; inoltre, regioni limitrofe sembrano gestire gruppi diversi di specie. Suggeriamo la necessità di un'agenzia principale nazionale per coordinare efficacemente le azioni, facilitare la comunicazione e la collaborazione tra i governi regionali, le agenzie nazionali e i Paesi vicini. Questo motiverà la continuità di azioni di gestione a lungo termine e l'aumento degli sforzi per riportare i costi delle SAI da parte dei gestori regionali e interregionali, che forniranno informazioni adeguatamente per far sì che i futuri bilanci acquisiscano efficacia gestionale.

Abstract in Portuguese

Custos econômicos das espécies invasoras na Espanha. Avaliações econômicas para espécies exóticas invasoras (EEI) são uma necessidade urgente para informar, coordenar e motivar tomadores de decisão na alocação de recursos econômicos e humanos para a gestão das EEI. Nós buscamos por custos econômicos de EEI na Espanha utilizando o banco de dados InvaCost e solicitamos dados para governos regionais e autoridades nacionais, o que resultou em mais de 3.000 registros de entrada. Considerando apenas dados robustos (ou seja, excluindo custos extrapolados, potenciais (não observados ou esperados) e custos de baixa confiabilidade), os custos econômicos na Espanha foram estimados em 261 milhões de dólares (232 milhões de euros) de 1997 até 2022. Houve um aumento de 4 milhões de dólares por ano, antes do ano 2000, para 15 milhões anuais nos anos mais recentes (de 4 para 13 milhões de euros). Com base nos dados robustos, os custos com manejo foram os mais reportados na Espanha (> 90%), enquanto custos com danos foram encontrados apenas para 2 das 171 espécies com custos reportados. Os custos econômicos dependem principalmente de administrações regionais e inter-regionais que gastaram 66% do recurso com ações de manejo pós-invasão, ao contrário de todas as diretrizes internacionais que recomendam investir mais em prevenção. Administrações regionais reportaram os custos de forma desigual. Além disso, 36% das espécies invasoras, que foram responsáveis por custos com manejo, não foram incluídas em regulamentações nacionais ou europeias (tal como, listas de espécies indesejadas), sugerindo a necessidade de revisão dessas políticas. Ainda, regiões vizinhas parecem gerir diferentes grupos de espécies. Nós sugerimos a necessidade de uma agência nacional central para coordenar ações de forma efetiva, facilitar a comunicação e a colaboração entre os governos regionais, agências nacionais e países vizinhos. Isso irá motivar a continuidade de ações de gestão a longo prazo e o aumento dos esforços para reportar custos com EEI por gestores regionais e inter-regionais, que fornecerão informações adequadas para orçamentos futuros ganhando eficácia na gestão.

Abstract in Arabic

التكاليف الاقتصادية للأنواع الغريبة الغازية في إسبانيا. تعتبر التقييمات الاقتصادية للأنواع الغريبة الغازية حاجة ملحة لتحفيز وتوجيه إجراءات السلطات وصناع القرار في إدارة وتسيير الأنواع الغريبة الغازية. لقد بحثنا، من خلال الدراسة التي بين أيدينا، في التكاليف الاقتصادية للأنواع الغريبة الغازية في إسبانيا، وذلك باستثمار المعلومات المتاحة في قاعدة بيانات أنفاكوست وبتوجيه طلبات إلى الحكومات الإقليمية والسلطات الوطنية الإسبانية. وقد نتج عن هذا العمل الحصول على أكثر من 3000 بيانات تكلفة فردية مختلفة للأنواع الغازية. وإذا ما اعتبرنا فقط البيانات الأقوى (أي عند استبعاد التكاليف المستقرأة (أي المتوقعة أو غير الملاحظة) و/أو غير الموثوق بها من وجهة نظر منهجية)، فقد بلغت التكاليف الاقتصادية في إسبانيا نحو 261 مليون دولار أمريكي (232 مليون يورو) بين عامي 1997 و2022. وكانت هناك زيادة سنوية قدرها 4 ملايين دولار قبل عام 2000، ثم 15 مليون دولار سنويًا في السنوات الأخيرة. وقد أظهرت هذه البيانات القوية أن معظم التكاليف المبلغ عنها للأنواع الغريبة الغازية في إسبانيا (أكثر من 90%) ترتبط بتكاليف التسيير، في حين تم العثور على تكاليف الضرر لنوعين فقط من أصل 175 نوعا تم تحديد تكلفتها. كما أظهرت هذه الدراسة أن التكاليف الاقتصادية تقع بشكل رئيسي على عاتق الإدارات الإقليمية وبين-الإقليمية. هذا وقد خصصت هذه الأخيرة 66% من التكاليف المسجلة لإجراءات التسيير ما بعد الغزو البيولوجي، خلافًا للتوجيهات الدولية التي توصي بزيادة الاستثمار في مجالات الوقاية القبلية. كما أبلغت هذه الإدارات الإقليمية عن التكاليف بشكل متفاوت. وبالإضافة إلى ذلك، فإن 36% من الأنواع الغازية، التي أبلغ عن أنها تسبب في تكاليف التسيير، لم تكن مدرجة في التشريعات الوطنية أو الأوروبية (أي القوائم السوداء). وهذا يشير إلى الحاجة الماسة إلى مراجعة هذه السياسات؛ وعلاوة على ذلك، يبدو أن المناطق المجاورة تدير مجموعات مختلفة من الأنواع. وعطفا على ما سبق، فإننا نحث، كتوصية، على خلق وكالة وطنية رائدة لتنسيق الإجراءات بشكل فعال وتسهيل الاتصال والتعاون بين الحكومات الإقليمية والوكالات الوطنية والبلدان المجاورة. وسيحفز ذلك، لا محالة، على استمرارية إجراءات الإدارة على المدى الطويل وتكثيف الجهود لتحديد دقيق لتكاليف الأنواع الغريبة الغازية من قبل المسيرين الإقليميين وبين-الإقاليميين

Abstract in Galician

Custos económicos das especies exóticas invasoras en España. As avaliacións económicas para especies exóticas invasoras (EEI) son un requisito urxente para a toma de decisións informadas e a coordinación e motivación da asignación de recursos económicos e humanos para a súa xestión. Neste estudo buscamos información dos custos económicos das EEI en España mediante a utilización da base de datos InvaCost e solicitude de datos aos gobernos rexionais e autoridades nacionais, o que deu lugar a máis de 3.000 entradas de custos. Considerando só datos sólidos (é dicir, excluíndo os custos extrapolables, potenciais (non ocasionados ou esperados) e de baixa fiabilidade), os custos económicos en España estimáronse en US$ 261 millóns (232 millóns de euros) entre 1997 e 2022. Houbo un aumento de US$ 4 millóns ao ano antes do 2000 a US$ 15 millóns ao ano nos últimos anos (de 4 a 13 millóns de euros). Os datos sólidos mostraron que a maioría dos custos reportados das EEI en España (> 90%) corresponden a custos de xestión, mentres que os custos dos danos só se atoparon en 2 das 174 especies con custos notificados. Os custos económicos dependen principalmente das administracións rexionais e interrexionais que gastaron o 66% dos custos en accións de xestión posterior á invasión, en contra de todas as directrices internacionais, que recomendan investir máis en prevención. As administracións rexionais informaron desigualmente dos custos. Ademais, o 36% das especies invasoras con custos de xestión reportados, non foron incluídas na normativa nacional ou europea (é dicir, as listas negras), o que suxire a necesidade de revisar estas políticas; ademais, as rexións veciñas parecen xestionar diferentes grupos de especies. Suxerimos a necesidade dunha axencia líder nacional para coordinar de xeito eficaz as accións de xestión, e facilitar a comunicación e a colaboración entre gobernos rexionais, axencias nacionais e países veciños. Isto motivará a continuidade das accións de xestión de longa duración e o aumento dos esforzos para reportar os custos das EEI por parte dos xestores rexionais e interrexionais, o cal proporcionará información para os futuros orzamentos que mellorarán a eficacia da xestión de EEI.

Abstract in Catalan

Custos Costos econòmics de les espècies exòtiques invasores a Espanya. L'avaluació econòmica del impacte d'espècies exòtiques invasores (EEI) és un requisit urgent per a la presa de decisions informades, promovent i coordinant l'assignació de recursos humans i econòmics per a una gestió adequada de les EEI. Hem cercat informació sobre els costos econòmics de les EEI a Espanya, mitjançant la base de dades InvaCost i consultes als governs regionals i les autoritats nacionals, amb un resultat de més de

3.000 entrades sobre costos. Tenint en compte només dades sòlides (és a dir, excloent els costos extrapolats, potencials (no incorreguts o esperats) i costos de baixa fiabilitat), els costos econòmics a Espanya es van estimar en 261 milions de dòlars (US$, 232 milions d'euros) des del 1997 fins al 2022. Hi va haver un augment de 4 milions de dòlars per any abans del 2000 a 15 milions de dòlars en als darrers anys (de 4 a 13 milions d'euros). Les dades sòlides van mostrar que la majoria dels costos reportats de les EEI a Espanya (> 90%) corresponien als costos de gestió, mentre que els costos de danys només es van trobar en 2 de les 174 espècies amb els costos reportats. Els costos econòmics es basaven principalment en administracions regionals i interregionals que gastaven el 66% dels recursos en accions de gestió postinvasió, contràriament a totes les directrius internacionals, que recomanen invertir més en prevenció. Les administracions regionals van informar desigualment de costos. D'altra banda, el 36% de les espècies invasores, que suposaven un cost de gestió, no estaven incloses en les regulacions nacionals o europees (és a dir, les llistes negres), cosa que suggereix la necessitat de revisar aquestes polítiques; a més, les regions veïnes semblen gestionar diferents grups d'espècies. Suggerim la necessitat d'una agència líder nacional per coordinar eficaçment les accions, facilitar la comunicació i la col·laboració entre governs regionals, agències nacionals i països veïns. Això motivarà la continuïtat de les accions de gestió de llarga durada i una millora de la informació sobre els costos derivats de les EEI per part dels gestors regionals i interregionals, proporcionant la informació adequada per tal de maximitzar una eficaç gestió en futurs pressupostos.

Abstract in Basque

Espezie exotiko inbaditzaileen kostu ekonomikoak Espainian. Espezie exotiko inbaditzaileen (EEI) kudeaketarako kostuen ebaluazioa ezinbestekoa da, bai erabakiak hartzeko, informazioa emateko zein baliabide ekonomikoen eta giza baliabideen esleipena koordinatu eta motibatzeko. Ikerketa honetarako Espainiako EEIen kostu ekonomikoei buruzko informazioa bilatu genuen. Horretarako InvaCost datubasea erabiliz gain, eskualdeko eta nazioko administrazioei datuak eskatu genizkien. Guztira, bilaketak 3.000 kostu-sarrera baino gehiago ekarri zituen. Kostu sendoak bakarrik kontuan hartuta (hau da, espero ziren kostuak, aurreikusiak edo potentzialak alde batera utzita), 1997 eta 2022 bitartean Espainian EEIren kostu ekonomikoak 261 milioi dolar (232 milioi €) izan zirela kalkulatu zen. 2000. urtea baino lehen urteko kostua 4 milioi US$-koa bazen, azken urteetan 15 milioira igo da (hau da, 4 milioi eurotik 13 milioi eurora). Datu sendoen arabera, Espainian jakinarazitako kostu gehienak (>90%) kudeaketa-kostuei zegozkien. Kalte ekonomikoak, berriz, 174 espezieetatik 2rekin bakarrik erlazionatu ziren. Kostu ekonomikoak eskualdeko edo eskualde arteko administrazioenak izan ziren batez ere. Nazioarteko gidetan gomendatzen den moduan prebentzioan gehiago inbertitu beharrean, kostuen %66 inbasioaren ondorengo erabilera-ekintzetan gastatu zuten. Eskualdeetako administrazioek ez zituzten kostuak modu berean aurkeztu. Kudeaketarako kostuak ezarritako espezie inbaditzaileen artean, %36a ez zen lege nazionaletan edo Europako legeetan agertzen (zerrenda beltzak). Gertaera honek, legeak berrikusteko beharra adierazten du. Horrez gain, aldameneko eskualdeek espezie-talde desberdinak kudeatzen dituztela dirudi. Hori dela eta, estatu mailan ekintzak eraginkortasunez koordinatuko dituen agentzia baten beharra iradokitzen dugu. Agentziak gainera eskualdeetako gobernuen, agentzia nazionalen eta auzoko herrialdeen arteko komunikazioa eta lankidetza erraztu beharko luke. Kudeaketa eraginkorragoa izan dadin, agentziaren sorrerak epe luzeko kudeaketa-ekintzak aurrera jarraituko dutela eta etorkizuneko aurrekontuei buruzko informazio egokia emango dela ziurtatuko luke.

Keywords

Iberian Peninsula, InvaCost, management costs, monetary impacts, non-native species, prevention costs, socioecology, stakeholders

Introduction

Invasive alien species (IAS) can cause significant negative environmental and socioeconomic impacts (Blackburn et al. 2019). These include loss of biodiversity (Simberloff et al. 2013; Bellard et al. 2016), changes to ecosystem functioning (Ehrenfeld 2011), impacts on human health and well-being (Jeschke et al. 2014) and large economic losses. Knowledge about the economic impact of IAS is, however, generally limited geographically, taxonomically or to some socioeconomic sectors. In the 2000s, Pimentel et al. (2005) provided the first estimations of the economic costs of IAS at large spatial scales. Since then, other studies have attempted to collect further data on these costs, such as in Europe (Kettunen et al. 2009), in invasion research and management (Scalera 2010) or for specific taxonomic groups (e.g. insects, Bradshaw et al. 2016). However, available data are scarce, scattered and not easily accessible and extrapolation-based approaches underlying most of these estimates are methodologically questionable (Cuthbert et al. 2020). These fragmented data and methodological flaws are reflected by critical knowledge gaps on the economic costs of IAS for most taxa, countries and regions of the world (Aukema et al. 2011). Such economic assessments are, therefore, an urgent requirement for informed decision-making by policy-makers and other stakeholders, for coordinating and motivating the allocation of economic and human resources for the management of IAS and for raising public awareness (Hulme 2006; Andreu et al. 2009; Diagne et al. 2020a, 2021a).

Europe represents a hub for alien species introductions (Turbelin et al. 2017), of which several thousands are already established (Dawson et al. 2017), inducing substantial economic impacts to the continent (Haubrock et al. 2021a). As a consequence, there is an increasing awareness to tackle IAS throughout the continent (García-de-Lomas and Vilà 2015; Turbelin et al. 2017). With an area of 505,992 km^2, Spain is one of the largest countries in Europe, presenting a considerable geographical, topographical, climatic, geological and species diversity. It also has a large diversity of IAS: the Spanish Government estimates that up to 190 alien species have already established invasive populations in the country (Spanish Catalogue of Invasive Alien Species, Royal Decree 630/2013). Spain has adopted legislation aiming at tackling biological invasions for the last 25 years. However, although the introduction of IAS was already considered as a criminal offence since 1995 (through an Organic Law, 10/1995) and the Spanish Strategy for the Conservation and Sustainable Use of Biodiversity (following the Convention of the Biological Diversity's recommendations to protect biodiversity from IAS) was developed in 1998, it was not until 2007 when policies for preventing and managing IAS were strengthened. The Law of Natural and Biodiversity Heritage (Law 42/2007) includes not only the need for prevention (through the Spanish Catalogue of Invasive Alien Species, Royal Decree 630/2013), but also the creation of strategic management plans for those IAS that threaten native species, natural habitats, agronomy and economic resources associated with environmental resources. The responsibility for implementing the Law falls into the "competent authorities", which are mainly the regional governments (i.e. the autonomous communities) and the national authorities (e.g. national authorities managing borders, continental waters or national parks that spatially correspond to more than one region).

Andreu et al. (2009) showed that environmental managers from regional authorities in Spain were generally aware of the risks posed by biological invasions. However, they claimed that there were limited economic funds to manage invasive alien species, and a lack of coordination amongst different regional and national administrations, scientific research on the performance of different strategies to manage invasive alien species and knowledge on the economic costs of IAS in the country (Andreu et al. 2009). The latter is known to be essential to help regional and national authorities to set up efficient budgets for IAS management. In this context, the InvaCost database (Diagne et al. 2020b), the most up-to-date repository of invasion costs worldwide, provides an excellent opportunity to tackle the current lack of data on the economic costs of IAS in Spain. Here, we extracted the data available in the InvaCost database regarding the economic costs of IAS in Spain. We expanded these data by requesting further information directly from Spanish regional and national environmental managers. Our aims were to (i) describe the distribution of reported economic costs of IAS across regions, environments, taxonomic groups, cost types and economic sectors; (ii) identify those IAS causing the highest costs; and (iii) examine the temporal trends of the economic costs reported over the last decades.

Methods

Data collection

We extracted data on the costs of IAS from the most updated version of the InvaCost database: InvaCost_v.3.0 (9,823 entries, Diagne et al. 2020b, https://doi.org/10.6084/m9.figshare.12668570) (Fig. 1a). This database consists of cost data extracted from documents obtained through standardised literature searches (i.e. using SI Web of Science platform, Google Scholar and the Google search engine) and opportunistic targeted searches (i.e. expert consultations for which data gaps were identified). One of these targeted searches addressed cost data in non-English languages (Angulo et al. 2020, https://doi.org/10.6084/m9.figshare.12928136). Cost values (including Spanish) recorded in InvaCost_v.3.0 were converted from local currencies to US$ by dividing the cost estimate by the official market exchange rate corresponding to the year of the cost estimation and then to 2017 US$ using inflation factors (Diagne et al. 2020b). From InvaCost_v.3.0, we extracted specific relevant data, resulting in a total of 3,260 entries of economic costs of IAS in Spain (Suppl. material 1; Fig. 1b).

Due to the importance of the non-English targeted search for the Spanish dataset (i.e. only 49 of the 3,260 entries in our dataset were extracted from documents written in English – 20 vs. 61 documents), we expand here the methods used by Angulo et al. (2021) to collect cost data in non-English languages. Spain is administratively divided into 17 autonomous regions (herein "regions"). Each of these regions manages IAS independently. We explored the web pages of regional government offices in charge of managing invasive species in each region and, when available, emailed environmental managers or sent administrative forms requesting economic data on the costs of IAS.

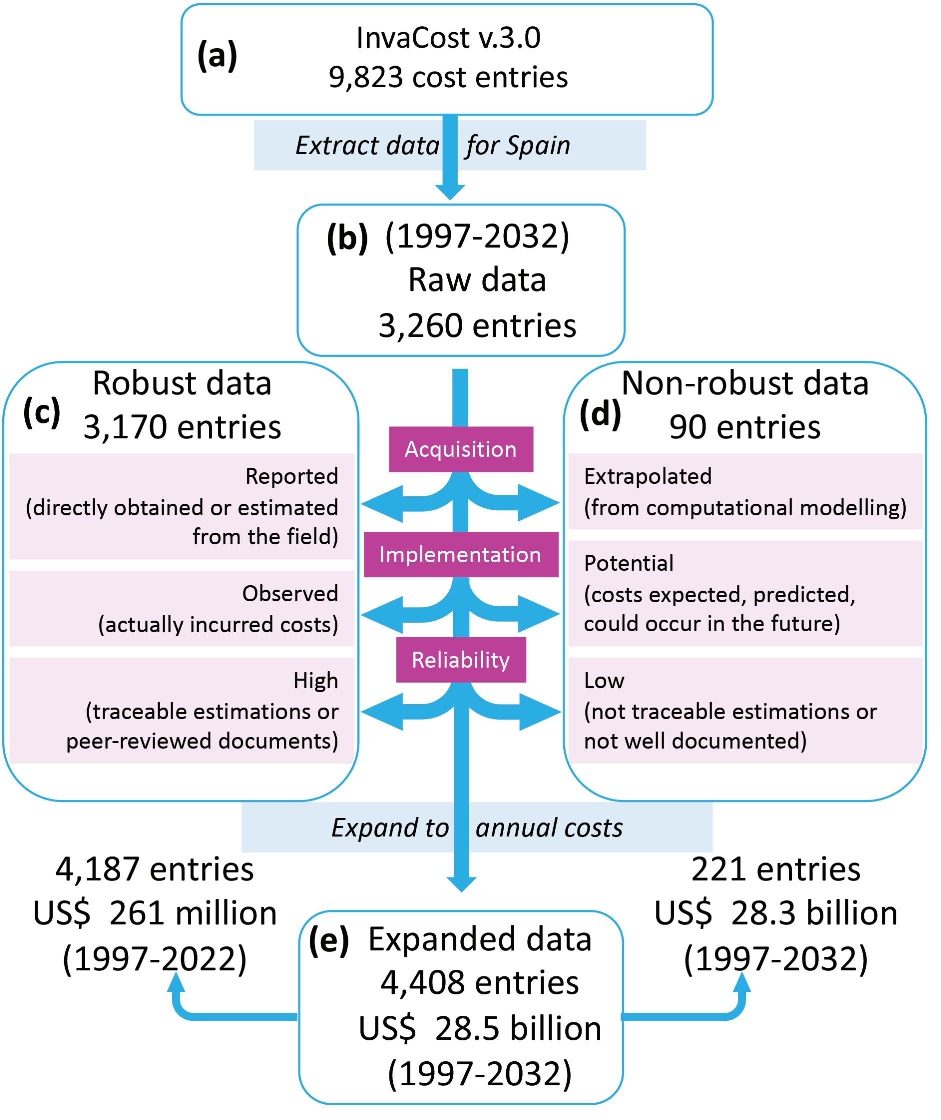

Figure 1. Data collection and filtering processes (**a**) data sources (**b**) raw data (timeframe and number of entries) obtained after extracting the data for Spain; raw data were segregated in two groups (**c**) robust data and (**d**) non-robust data using three variables, acquisition, implementation and reliability (**e**) expanded data to obtain comparable yearly costs.

Moreover, Spanish continental waters are managed in coordination with the Ministry for the Ecologic Transition and Demographic Challenge, through independent river basin authorities (hydrographic confederations). Therefore, we searched for available information in their web pages and contacted those river basin authorities from whom we could obtain the contact details of their environmental managers (i.e. Guadiana, Tajo, Segura, Basque Country, Cantábrico). In the region of Valencia, costs were reported as working days and we transformed them into economic costs by multiplying the reported number of working days by € 128 (i.e. cost per day, Vicente del Toro, Biodiversity Service, Generalitat Valenciana, pers. comm.). We obtained data for Spain up to December 2020, with costs being reported in Spanish and in two co-official languages: Catalan and Galician (Suppl. material 1: Tab InvaCost_3.0_Spain, column "language").

Data structure

Cost data extracted for Spain (herein raw data, Fig. 1b, Suppl. material 1: Tab InfoVariables) were described with a set of variables pertaining to: (i) information on the document reporting the cost, (ii) spatial information (e.g. location, spatial scale, environment – aquatic or terrestrial – and whether the location corresponds to a protected area or to an island), (iii) taxonomy of the invasive species incurring the cost, (iv) temporal information, (v) typology of costs reported (e.g. management actions or economic damages, impacted sector) and (vi) a set of variables reporting the raw cost estimates, currency used and the converted US$ values.

With respect to the type of cost, we first used the column "type_of_cost_merged" which included three categories: "damage" costs: economic losses due to direct and/or indirect impacts of invaders, such as yield loss, health injury, land alteration, infrastructure damage or income reduction; "management" costs: economic resources allocated to prevention, control, research, long-term management, or eradication; "mixed" costs: when costs include both damage and management expenditure. We also used the column "management_type" to divide further management costs in the following categories: "pre-invasion management": monetary investments for preventing successful invasions in an area (including quarantine or border inspection, risk analyses, biosecurity management, etc.); "post-invasion management": money spent for managing IAS in invaded areas (including control, eradication, containment); "knowledge/funding": money allocated to all actions and operations that could be of interest at all steps of management at pre- and post-invasion stages (including administration, communication, education, research etc.); "unspecified" for costs without detailed types; and a "mixed" category was assigned when costs included at least two of the above categories.

Categories for the economic sector included: "agriculture": considered at its broadest sense, such as crop growing, livestock breeding, beekeeping, land management; "authorities-stakeholders": governmental services and/or official organisations – such as conservation agencies and forest services – that allocate efforts for the management of biological invasions (e.g. control programmes, eradication campaigns, research funding); "environment": impacts on natural resources, ecological processes and/or ecosystem services; "forestry": forest-based activities and services, such as timber production/industries and private forests; "health": for every item directly or indirectly related to human health, such as control of disease vectors (e.g. mosquitoes transmitting pathogens to humans) or medical care and damage to work productivity due to impacts on health; "public and social welfare": activities, goods or services contributing to human well-being and safety in our societies, including local infrastructure, such as the electricity system, quality of life (e.g. income, recreational activities), personal goods (e.g. private properties, lands), public services (e.g. transport, water regulation) and market activities (e.g. tourism, trade).

Data processing

Three variables about the typology of the costs are important for the further selection of the data we used (Diagne et al. 2020b): (i) the acquisition method for the cost value ("reported" if the cost data were directly obtained or derived using inference methods from field-based information or "extrapolated" if the cost data were obtained using computational modelling), (ii) the implementation of the cost ("observed" if the cost was actually incurred or "potential" if the cost is predicted to occur over time) and (iii) the reliability of the cost value reported ("high" or "low", based on whether the approach used for cost estimation in the document was reported and traceable). We filtered our dataset to differentiate the most robust data, i.e. directly reported, observed and highly reliable costs (corresponding to 3,170 raw entries, Fig. 1c). Indeed, we considered as non-robust data 90 cost entries that were extrapolated, not yet actually incurred and/or of low reliability (Fig. 1d).

We considered the full dataset (raw data, 3,260 entries, Fig. 1b) to explore general differences in the number of cost entries for Spain amongst descriptors. The number of entries is a good indicator of how detailed reported costs are (e.g. costs obtained from a single report for one region covering all invasive species, invaded locations, years and types of management can be assumed to be less detailed than costs obtained from several reports, each of them covering different invasive species and their management). Moreover, since the period of estimation across reported costs varied from months to years, we homogenised the cost values for the full dataset (including both robust and non-robust data) as follows: we recalculated costs covering several years on an annual basis and repeated these annual values over the duration time (in number of years) of each cost occurrence. For example, a cost reporting US$ 500 occurring in the period 1996–2000 was transformed into five identical costs of US$ 100 for each of those years. Costs occurring in less than one year were assumed as having occurred during a single complete year in order to avoid overestimation. Hence, we obtained comparable annual costs for all cost entries. This was performed using the "expandYearlyCosts" function of the ‘invacost’ package version 0.3-4 (Leroy et al. 2020 in R version 3.6.3 (R Core Team 2020). The expanded full dataset resulted in 4,408 entries (Fig. 1e) from which 4,187 cost entries correspond to robust data and 221 to non-robust data. All the analyses presented in the main text were carried out with the robust data. Results including the non-robust data are briefly presented in the first sentences of the results and shown in Figure 1 and in Suppl. material 2: Fig. S1.

Data analysis

We first described the number of entries and the economic costs for each of the 17 autonomous regions and mapped the information across the country using the package "ggplot" in R version 4.0.2 (R Core Team 2020). We also described the costs across specific descriptors: main taxonomic groups, main environments in which the costs occurred, economic sectors impacted by the cost, the spatial scale at which the costs occurred and whether or not the costs occurred in protected areas.

We calculated the temporal trends of IAS economic impacts in Spain by using the function summarizeCosts of the "invacost package" version 0.3-4 (Leroy et al. 2020) in R version 3.6.3 (R Core Team 2020). This function allowed us to calculate average annual costs between 1997 and 2019, providing averages in 4-year periods throughout the study period using the extended entries calculated by the "expandYearlyCosts" function described above.

Finally, we identified the costliest IAS in Spain and assessed whether the species causing economic costs in Spain are those recorded as invasive in the country or included in European or national regulations. We collected information on the identity of those alien species (i) recorded as invasive in Spain (sensu the Global Invasive Species Database; http://www.issg.org/database); (ii) included in the Spanish Catalogue of Invasive Alien Species (Royal Decree 630/2013), (iii) included in the List of Invasive Species of Union Concern (EU, No 1143/2014 of the European Parliament); and (iv) proposed as potential candidates to be included in the List of Invasive Species of Union Concern (Carboneras et al. 2018). Besides European and National regulations, some Spanish regions also present regional invasive alien species regulations. For example, in the region of Aragon, it is not allowed to introduce, catch, keep, transport or sell any freshwater alien crayfish species (Decreto 127/2006 of the Aragon Government). However, most regions rely exclusively on national and European regulations and have no specific lists of invasive alien species (with the exception of Valencia; Decreto 14/2013 of the Consell). Therefore, we only considered national and European regulations in our analysis.

Results

Costs of invasive species in Spain amounted to US$ 28.52 billion (€ 25.38 billion, using the 0.89 conversion factor for 2017) from 1997 to 2032 (Fig. 1e). However, although only 90 out of 3,260 raw entries were extrapolated, potential and/or unreliable costs, these constituted 99.08% of the economic costs in our dataset (Fig. 1e). Most of these high costs were driven by one single entry: a cost derived from an extrapolation of the potential loss of forestry stock caused by *Bursaphelenchus mucronatus*, the pine wood nematode, over a period of 22 years (2008–2030, Suppl. material 2: Fig. S1). Without considering non-robust data, the reported, observed and reliable costs for invasive species in Spain constituted US$ 261.28 million (€ 232.54 million). These costs occurred from 1997 to 2020, except for two raw entries that went over this year: one corresponding to a LIFE+ project ranging from 2019 to 2022 aimed at controlling *Lampropeltis californiae* in the Canary Islands and the second corresponding to an annual management programme for invasive plants in Sierra Espuña Regional Park (Murcia) that included part of the year 2021. Thus, both reported budgets are considered already delivered costs.

Only using the robust dataset, we showed that the highest amount of costs was reported for plants (66%; especially from the orders Myrtales and Commelinales), followed by arthropods (12%; mainly insects) and mollusca (11%; mostly bivalves) (Fig. 2a–e). Most costs corresponded to IAS from terrestrial environments (53%), while

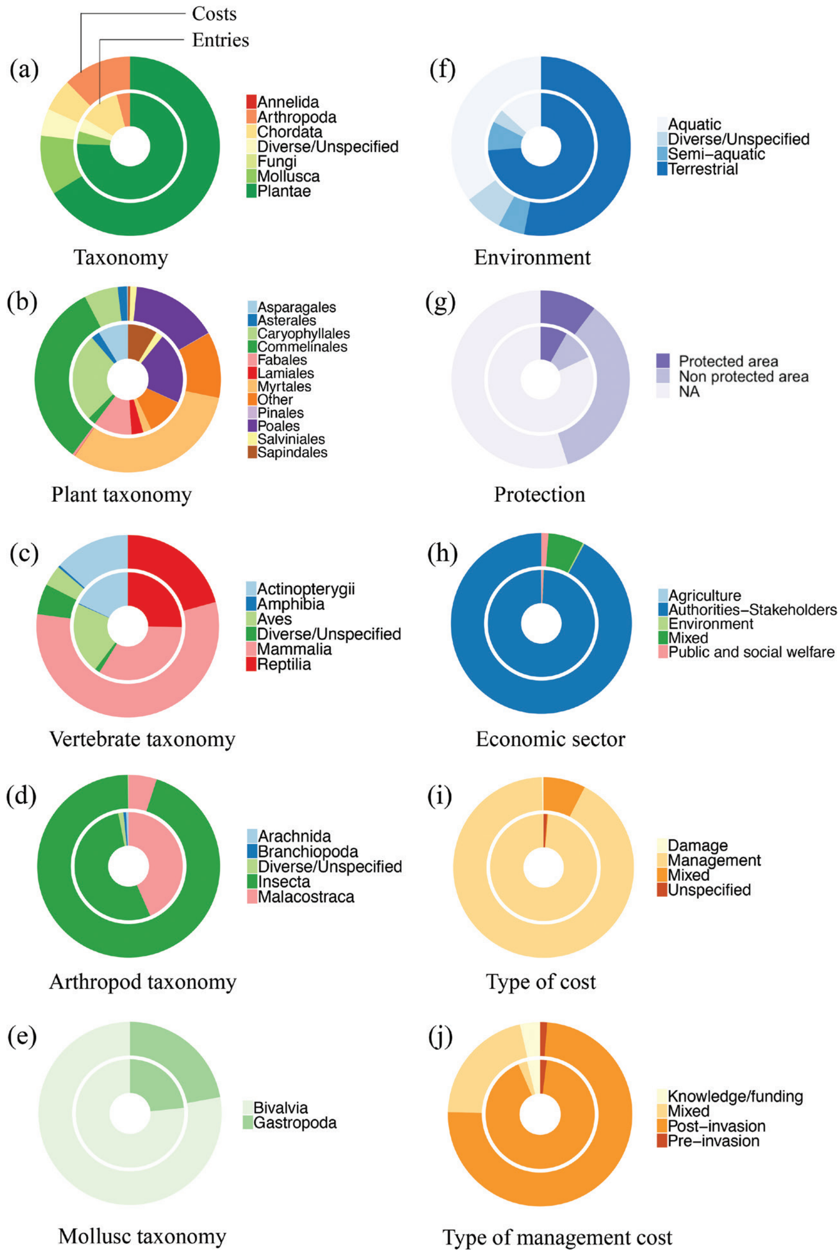

Figure 2. Total economic costs (outer circles) and number of entries (inner circles) for invasive species in Spain for each cost descriptor (**a**) taxonomy in general (**b**) plant taxonomy (**c**) vertebrate taxonomy (**d**) arthropod taxonomy (**e**) mollusc taxonomy (**f**) environment (**g**) protection (**h**) economic sector (**i**) type of cost and (**j**) type of management cost. See methods for description of categories.

aquatic and semi-aquatic environments contributed with 35% and 5% of the costs, respectively; the number of entries was much higher for terrestrial environments (Fig. 2f). Only 10% of the total costs were reported to occur specifically in protected areas

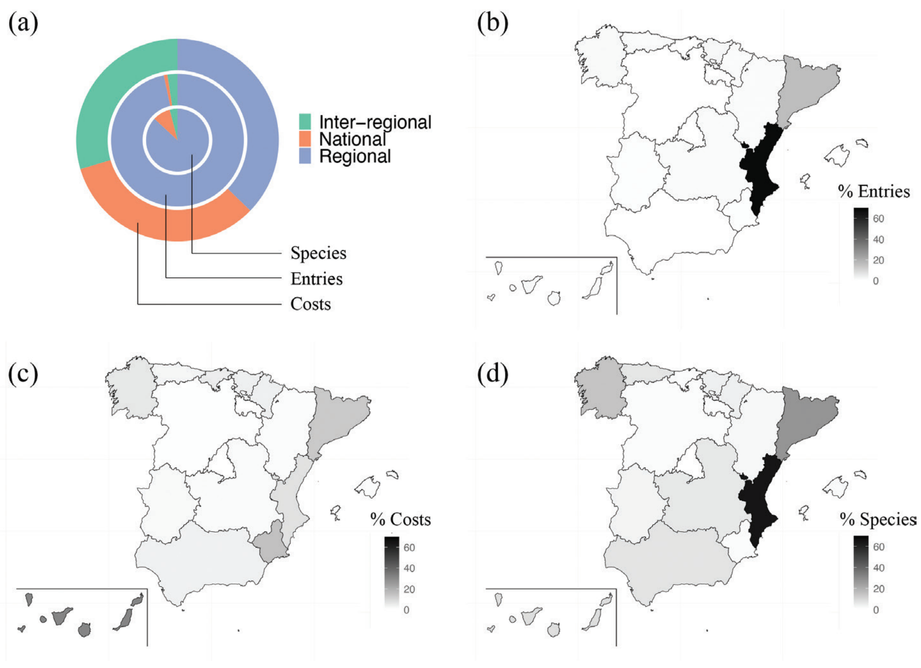

Figure 3. Distribution of the observed economic costs of biological invasions in Spain across the autonomous regions (**a**) relative importance of country, inter-regional and regional levels in costs and number of species with costs (**b**) proportion of entries (**c**) total economic costs (US$ million), and (**d**) number of species with costs. All values correspond to the robust data (reported, incurred and reliable costs); values in (**b**) and (**d**) correspond to the raw data and (**c**) to the expanded data.

(Fig. 2g). The most impacted sector was authorities and stakeholders (92%, Fig. 2h); i.e. governmental services and/or official organisations (e.g. conservation departments) that allocate efforts to the management of IAS (e.g. prevention, eradication campaigns, control or monitoring programmes, research funding). The forestry and health economic sectors had only one (for *B. mucronatus*) and two (for *Ambrosia artemisiifolia*) entries, respectively. These entries consisted of extrapolated amounts and, therefore, were not considered as robust data. Costs impacting agriculture came from both scientific papers (three entries that consisted of extrapolated costs and, thus, not included in the robust data) and information obtained directly from managers (four entries for *Pomacea* spp.). Less than 1% of the costs corresponded to economic damage while 92% corresponded to management costs (Fig. 2i). Taking into account only management costs, most costs reporting management actions consisted in post-invasion management (74%), while relatively low costs were spent for knowledge/funding (3%, including education, communication etc.) and pre-invasion management actions (1%, Fig. 2j).

Although a high number of entries corresponded to information obtained directly from the regional autonomous communities, economic costs were divided almost equally at the country (33%), inter-regional (30%, such as river basins situated across regions) and regional levels (37%, Fig. 3a). Within the autonomous regions, there were differences in the amount of costs and number of entries amongst them (Fig. 3b, c).

Both variables showed different patterns; for example, Valencia reported a high number of detailed entries (i.e. including information on time, location, type of management etc.), while their costs were not as high as those reported by other regions, such as Murcia and Canary Islands. In other cases, for example, Catalonia, a high number of entries corresponded to a high amount of costs. Valencia had the highest number of entries, expanding from 2009 to 2019, followed by Catalonia, from 2014 to 2018 (Fig. 3b). Canary Islands constituted the region with the highest reported costs, followed by Murcia, Catalonia, Valencia and Galicia (Fig. 3c). The rest of the regions accounted for less than US$ 5 million. Castilla y León and La Rioja showed the lowest costs (i.e. lower than US$ 1 million). With respect to the number of IAS managed by region, these largely differed amongst regions, ranging from 1 to 111 IAS (Fig. 3d). Mean number of IAS reported to incur costs amongst regions (16.5) was intermediate between the ones managed at the country level (18) and the ones managed at the inter-regional level (8).

The average annual costs of biological invasions in Spain, taking into account only the robust data, was US$ 10.85 million (€ 9.66 million) over a time period from 1997 to 2020 (Fig. 4). Most of the robust data were reported between 2017 and 2020. Annual costs increased from US$ 4.22 million per year (€ 376 million) before 2000 to US$ 14.60 million per year (€ 12.99 million) in the last four years (Fig. 4). Using the robust dataset, trends of costs in Spain showed an initial increase during the first decade of cost reporting (1997–2007) and seemed to stabilise afterwards (Fig. 4). The apparent decrease in reported costs between 2013 and 2016 is most likely an artefact arising from a lack of cost estimates, given the multi-year delay between occurrence and reporting in literature.

Robust data show that economic damage in Spain was only observed for two species (*Dreissena polymorpha* and *Procambarus clarkii*), while the rest of the costs corresponded to managing IAS (Fig. 5a). Of the 174 IAS incurring management costs in Spain (robust data), 63 (36%) were not recorded as invasive for the country (GISD; http://www.issg.org/database) nor included in the current European or national regulations or proposed to be assessed to potentially include them in European regulations (Fig. 5a, Suppl. material 3). The management costs corresponding to these 63 invasive species (US$ 48.24 million, € 42.93 million) were recorded in the regions of Asturias (1 species), Balearic Islands (1), Canary Islands (4), Cantabria (1), Castilla La Mancha (1), Catalonia (4), Galicia (4), Navarra (2) and Valencia (46). Most of the costs invested in managing IAS that are not included in national or European regulations corresponded to terrestrial and aquatic plants (Fig. 5b, Suppl. material 3).

The 10 IAS presenting the highest economic costs (considering only robust management costs) include five terrestrial plants, one aquatic plant, two terrestrial animals and two aquatic animals (Table 1). Of these, seven species are included in the national regulations (*Arundo donax*, *Carpobrotus* sp., *Cenchrus setaceus*, *Cylindropuntia rosea*, *Eichhornia crassipes*, *Rhynchophorus ferrugineus* and *Vespa velutina*) and four in the European regulations of IAS (*E. crassipes*, *C. setaceus*, *C. rosea* and *V. velutina*). Regarding the number of cost entries of IAS in Spain, 50% of the entries corresponded to 15

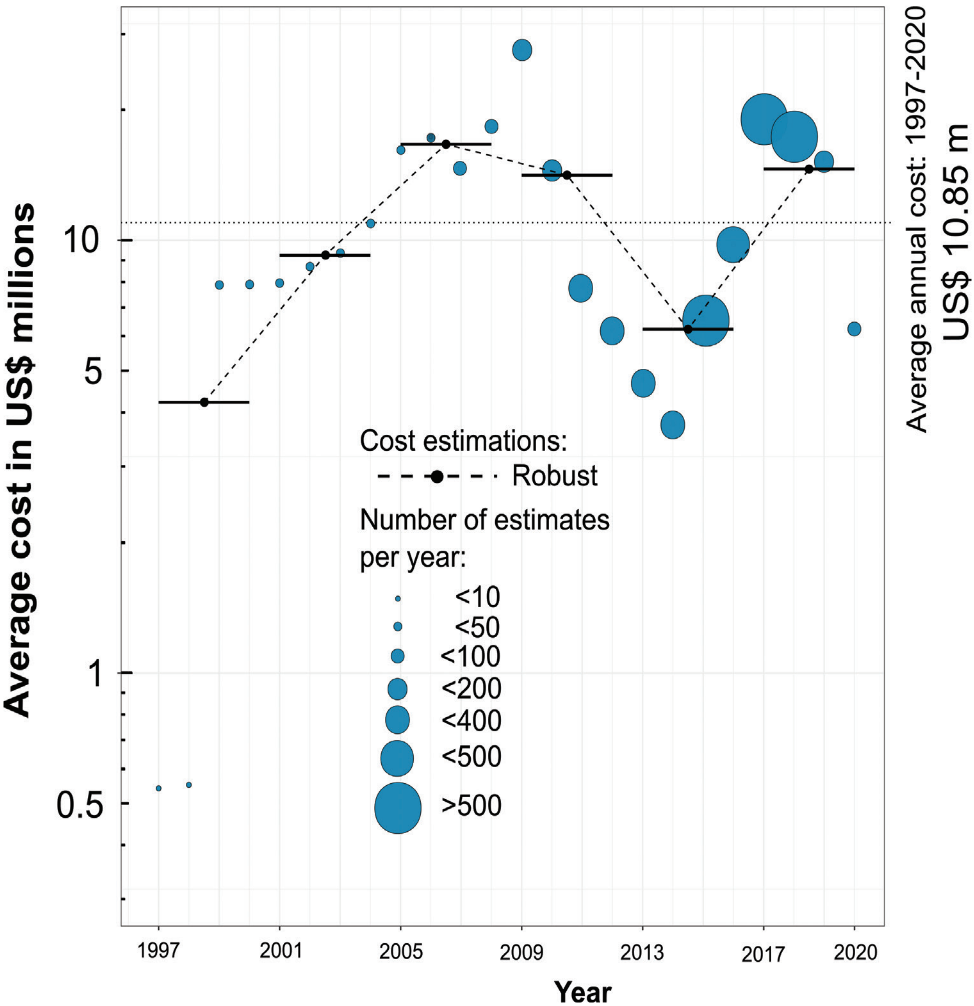

Figure 4. Temporal trends of the economic costs (US$, log) of invasive alien species in Spain using the robust data (reported, incurred and reliable costs). Each blue circle represents the cumulative cost for a given year, whereas its size is proportional to the number of estimates for that particular year. Average annual costs are calculated in 4-year periods and are represented by black dots and horizontal solid lines. Dashed lines connect the average annual costs for these 4-year periods.

species, all registering more than 50 cost entries each (Suppl. material 4: Fig. S2). The species with the highest number of entries was *Cylindropuntia pallida*, with a total of 203 records that represent 6.40% of the data, extracted from a total of six documents from Valencia. The 15 species with the highest number of cost entries did not vary when considering only management costs, while the 15 species with the highest economic costs slightly differed due to the damage reported for *D. polymorpha* (Suppl. material 4).

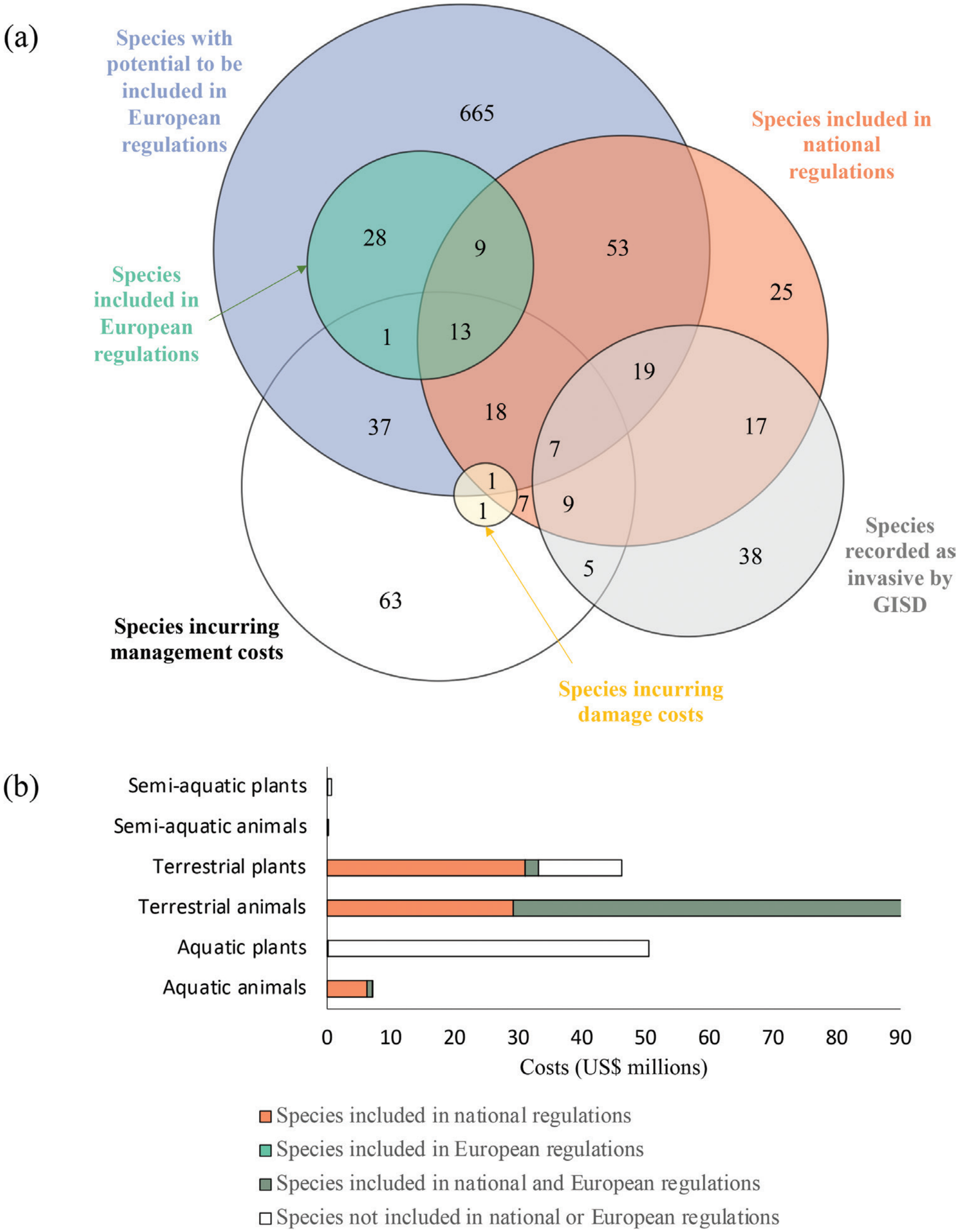

Figure 5. Invasive species with cost data in Spain with respect to national and international regulations (**a**) Venn Diagram illustrating the number of invasive species incurring management costs in Spain that are not recorded as invasive in the country (GISD; Pagad et al. 2018), listed in European or national regulations or proposed as potential candidates to be included in European regulations. Numbers indicate the number of invasive species (**b**) costs (in US$ millions) incurred by taxonomy and environment of those IAS incurring management costs in Spain, that are included in national regulations, in European regulations or in neither one.

Table 1. Lists of the ten costliest species in Spain considering only robust management costs. Costs are in US$ million; "Environ" corresponds to the environment where the cost occurred, "Taxon" refers to the taxonomic group of the species; "Regulation" indicates whether the species is listed in national (SP) and/or European (EU) regulations.

Species	Costs	Environ	Taxon	Regulation
Eichhornia crassipes	55.63	Aquatic	Plant	SP & EU
Eucalyptus sp.	50.25	Terrestrial	Plant	–
Rhynchophorus ferrugineus	24.12	Terrestrial	Animal	SP
Arundo donax	13.98	Terrestrial	Plant	SP
Cenchrus setaceus	10.13	Terrestrial	Plant	SP & EU
Neovison vison	7.91	Aquatic	Animal	–
*Pomacea maculata**	6.20	Aquatic	Animal	–
Vespa velutina	5.33	Terrestrial	Animal	SP & EU
Carpobrotus sp.	4.92	Terrestrial	Plant	SP
Cylindropuntia rosea	2.92	Terrestrial	Plant	SP & EU

* The taxonomy of *P. maculata* is not clear; however, it was reported with this name in the InvaCost database.

Discussion

General costs of IAS in Spain

We analysed economic costs of IAS occurring in Spain and explored more than 3,000 entries, using the InvaCost database and additional sources. Invasive species cost Spain at least US$ 261.28 million between 1997 and 2019. Contrary to what Haubrock et al. (2021a) found at the European continent scale, our estimations of expenditure were mostly incurred by governmental organisations (including regional administrations and river basin authorities) in managing IAS (92% of all costs). Damage costs were only found for two species (i.e. *D. polymorpha* and *Procambarus clarkii*). Since a large number of invasive species are known to cause high environmental and socioeconomic impacts in Spain (Andreu et al. 2009), these results highlight the need for future investments in research efforts to understand and quantify the economic damage of biological invasions in the country. Such knowledge on the economic damages of IAS in Spain could help increasing societal awareness, prioritising the management of IAS and motivating further investments in IAS management actions.

Compared with other countries of the Mediterranean basin, Spain has been reported as the fifth most impacted country regarding observed costs associated with IAS (Kourantidou et al. 2021), after France (US$781 billion, n = 1,036 cost entries), Italy (US$503 million, n = 94), Libya (US$340 million, n = 4) and Turkey (US$326 million, n = 11). From a continental perspective, Haubrock et al. (2021a) ranked Spain at a similar place than The Netherlands and Ireland, both countries being much smaller than Spain.

As for other countries and regions, our results show that not accounting for sources of information besides those written in English would have led to a significant knowledge gap and bias for this first assessment of global costs of invasive species in Spain (Angulo et al. 2021). The majority of costs and entries in our dataset came from non-English sources, mainly consisting of unpublished documents in Spanish, which resulted in a high percentage of cost entries reported in Spanish (98%), consistent with findings in some other European countries that reported costs in their native language (e.g. 97% for France, Renault et al. 2021; 69% for Germany, Haubrock et al. 2021b). For instance, in Central and South America over 40% of cost estimates came from Spanish and Portuguese sources (Heringer et al. 2021); and in Ecuador 51.8% of the costs were reported in Spanish (Ballesteros-Mejia et al. 2021). An extreme situation is observed in Japan, where all recorded costs were in Japanese (Watari et al. 2021), although this was a common trend in Asia (reviewed in Liu et al. 2021).

Management costs focused in aquatic and terrestrial environments, but mostly targeting invasive plants. The costliest invasive species in Spain was the aquatic plant *E. crassipes* (commonly known as water hyacinth), which was first recorded in the Guadiana River in 2004 and by 2005 it was already covering 75 km of the river surface. A large research effort has been invested in understanding the management options available to control this invasive plant. For example, in 2008, a workshop, arranged by European organisations, was attended by international experts, aiming to share experiences in the management of *E. crassipes* (e.g. the successes or failures resulting from applying different management actions) to facilitate the design of management actions in the Guadiana River (http://archives.eppo.int/meeTingS/2008_conferences/eic-chornia_workshop.htm). However, its management is still a challenge for the area (Téllez et al. 2008; Kriticos and Brunel 2016). This species, together with *D. polymorpha* or *Neovison vison*, which are amongst the ten costliest species in Spain, are also amongst the invasive aquatic species causing the most widespread economic impacts (Cuthbert et al. 2021a). *E. crassipes* also seems to be one of the costliest species in several African countries, in Asia and in North American countries, such as Mexico (Diagne et al. 2021b; Liu et al. 2021; Rico-Sánchez 2021); while *D. polymorpha* seems to be very costly in the USA and *N. vison* in other European countries such as Germany (Crystal-Ornelas et al. 2021; Haubrock et al. 2021b). Being in the list of the 100 of the worst invasive species, *D. polymorpha* was also ranked as the 8th costliest species of that list (Cuthbert et al. 2021b).

Regional management and the need for effective national coordination of actions

Regional administrations unequally reported costs, with regions, such as Catalonia or Valencia, reporting detailed annual economic costs from the last decade and others reporting relatively low amounts of costs. Many of the regions reporting high numbers of entries and large amounts of costs present high levels of development, trading and tourism activities, which are normally associated with biological invasions (Pyšek et al. 2010; Haubrock et al. 2021a). However, regional administrations reporting low numbers of entries and low costs are also largely invaded by IAS (e.g. Dana et al. 2009) and, therefore, might need further investments in reporting and managing IAS in the future.

The use of lists including IAS with known invasive potential is a widely used regulation tool at international and national levels (García-de-Lomas and Vila 2015). Most Spanish regions relied on the national catalogue of IAS rather than establishing their own regional listing (except, for example, Valencia). Based on the national list, managers can prioritise either IAS already present and expanded in their regions or the ones identified as potentially harmful in the future, in order to prevent their entrance. However, our results show that economic costs for pre-invasion management actions related to biosecurity issues, such as early detection, early warning, risk assessment or prioritisation analyses, constituted less than 1% of all costs; while most economic costs (74%) were spent in post-invasion management actions, such as monitoring, control or eradication. Although the importance of prevention rather than post-invasion management to efficiently manage IAS is known (Leung et al. 2012; Wilson et al. 2016), there could be an under-estimation of the costs of pre-invasion management actions in the data analysed for Spain. In many cases, managers communicating costs recognised that some prevention actions, such as risk analyses or monitoring for early detection, were not included in the reported costs, as no additional funding was required to implement such actions (e.g. managers use already existing resources, their time, computers or cars), while eradication or control campaigns need extra work (i.e. worker teams, machinery etc.).

A large number of the managed invasive species (63 IAS, 36% of all managed species) were not listed as invasive in Spain (sensu the GISD database; http://www.issg.org/database), included in European regulations or proposed to be assessed to potentially include them in European regulations. This suggests that Spanish environmental managers do not prioritise the management of invasive species according to current regulations or tools, such as the Global Invasive Species Database, or published expert assessments. The rationale for prioritising the management of IAS in the country, therefore, remains unknown. One possible explanation is that some managers are following the common approach of developing and implementing management actions for groups of species with similar management requirements, instead of doing this separately for individual species (van Wilgen et al. 2011). For example, in 2019, the Global Cactus Working Group (GCWG) identified a set of invasive and potentially-invasive cacti and key actions that can be taken to manage them worldwide (Novoa et al. 2019). In our dataset, six of the cactus species identified as invasive by the GCWG have reported management costs. However, only two of these are included in national regulations. Additionally, our data showed that, in aquatic environments, control of known invasive species, such as invasive turtles, fishes or crayfishes, lead to capture of other non-native species as a by-catch, such as other turtles of different genera (e.g. *Graptemys*, *Mauremys* or *Pelodiscus*), fishes (e.g. *Carassius auratus*) or crayfishes (e.g. *Callinectes sapidus*), not included in national regulations. Managing species that are not included in national lists is not uncommon; for example, Elvira and Almodóvar (2019) showed that only 2 out of 11 fish species introduced in Spain since 2000 are included in the national catalogue. Even if it is laudable and even encouraging that most managers are proactive and in advance of regulations, we suggest that the national catalogue should be revised to account for all species that are or should be managed.

A substantial amount of research has been recently focused on developing strategies to prioritise the management of IAS, including optimisation frameworks and decision processes (e.g. McGeoch et al. 2016; Curtois et al. 2018; Novoa et al. 2018), all in collaboration with different stakeholder groups (Novoa et al. 2020). Our results suggest that future efforts should focus on stakeholder engagement in Spain, in order to develop transparent and evidence-based management decisions. Moreover, inter-regional management costs, such as those incurred in river basins, were equal to the sum of the costs of all regions together. Such inter-regional management actions are generally more effective than single regional ones, since managing different species pools in neighbouring regions can hinder the effectiveness of the actions at larger geographic scales (Faulkner et al. 2020). Therefore, species prioritisation should ideally be done in collaboration with neighbouring regions in order to achieve effective management results (Sutcliffe et al. 2017).

Our results suggest that there is a need for a country-level organism responsible for the management of IAS that can effectively coordinate joint management strategies, facilitate communication and collaboration between regional governments, national and inter-regional agencies (such as river basin authorities), neighbouring countries and other stakeholders (Caffrey et al. 2014; Piria et al. 2017). This will motivate the continuity of long-lasting management actions and reporting of the costs of IAS that will adequately provide information for future budgets increasing management effectiveness (Pergl et al. 2020). The non-native species secretariat in the UK (http://www.nonnativespecies.org/) is a good reference for this, while a starting point in clarifying competencies across different administrations is suggested.

The good and the bad: high costs in aquatic environments and low costs in protected areas

Although terrestrial environments had more and higher reported costs than other environments(US$ 138.6 million), invasions were also relatively costly in aquatic (US$ 91.9 million) and semi-aquatic environments (US$ 12.4 million). There are generally few reports on the global economic impacts of invasive species in aquatic ecosystems (Lovell et al. 2006). However, compared with the whole InvaCost database (Diagne et al. 2020b), our estimates for these ecosystems are exceptionally high (but see the case of Mexico, Rico-Sánchez et al. 2021). A global assessment of all data included in InvaCost reported that the monetary costs of aquatic invasive species only constituted 5% of the total reported costs. This percentage increased to only 9% when considering management costs only (Cuthbert et al. 2021a). In contrast, we show that, in Spain, 35% of the funds allocated to the management of invasive species were spent in aquatic environments (plus 5% in semi-aquatic environments). Interestingly, some of the management costs reported by the river basin authorities along the Iberian Peninsula river basins were really high, such those reported in the Guadiana River related to the control of the water hyacinth (*E. crassipes*) since 2005 or that from the Ebro Basin related to the control of the Zebra mussel (*D. polymorpha*) in the 2000s (Table 1).

Protected areas in the Iberian Peninsula are known to be effective as natural biodiversity refugia (Araújo, Lobo and Moreno 2007; Gaston et al. 2008). In some Spanish regions, such as Andalusia, protected areas represent 30.5% of the total surface, which was reported as more than twice the European average (13.7%, Angulo et al. 2016). However, our results show that only 10.3% of the economic costs of IAS in the country (8.2% of cost entries) incurred specifically in protected areas in Spain. These low numbers suggest a lower reporting of costs or a lower investment in managing IAS in protected areas than in non-protected land, which is worrisome given the high ecological impacts of IAS in protected areas in the country. For example, Gallardo et al. (2017) showed that 38% of marine and 24% of inland protected areas in Europe were already affected by at least one of the 86 most threatening invasive species in Europe. Moreover, Capdevilla-Argüelles and Gallardo (2019) ranked a set of top-invaders by their menace to the Spanish national parks and some of those that constituted the highest menace, are amongst the ones we reported here with the highest costs, such as *E. crassipes*, *Cenchrus setaceus*, *N. vison*, *V. velutina* or *Cortaderia selloana*. Furthermore, Moodley et al. (2021) classified *Baccharis halimifolia* and *V. velutina* among the costliest species in European protected areas, while *N. vison* was among the costliest in semi-aquatic environments within protected areas (*B. halimifolia* was ranked 11th in Spain when looking only at management costs). However, it could be that our data are conservative regarding the real costs incurred in protected areas. For example, Saavedra and Medina (2020) showed that an eradication programme implemented in La Palma Island, Spain, prevented the expansion of the ring-necked parakeet (*Psittacula krameri*) into La Palma Island Biosphere. These costs were, however, recorded at the island level, not only in the protected lands.

Limitations of the study

Our study shows high economic costs of IAS in Spain, despite our conservative selection of data. Mainly, four potential sources of costs in Spain remained unexplored. On the one hand, while most protected areas are managed by the regional authorities, national parks, the most important figure of conservation for protected areas in Spain, are managed by a national authority, the Autonomous Organism of National Parks (OAPN). Although we also contacted environmental managers from the OAPN, they could not provide us with data on the economic costs of IAS, since this was not readily available. The main reason for this was that their management is shared by a number of private enterprises (mainly from the TRAGSA group) that work for the administration in broad public services, not only in the management of invasive species (Pep Amegual, Chief of Research Office in the OAPN, pers. comm.). Therefore, future engagement with these enterprises is needed to include these data in further analysis.

On the other hand, many research projects in Spain, commonly founded by national or international agencies, study biological invasions, despite few entries reporting research costs (n = 166). Scalera (2010) reported an increasing number of EU funded projects focusing on IAS from 1992 to 2006, with a budget for this period exceeding € 132 million; Spain, together with Italy and France, hosted 52% of these projects. Although we approached European Programmes' Advisors from the Spanish National Research Council (CSIC) and searched the web of the Ministry for Education and Professional Career, the information on these costs was difficult to obtain. We only consider costs of a few European projects that took place in Spain, for which cost information was available on the web or was reported by targeted researchers (i.e. Invasep, Ripisilva, Lampropeltis, ConHabit, Margal Ulla, La Rioja Life, Estuarios del Pais Vasco). New ways to obtain this information are needed in order to include such economic costs in future assessments.

Third, even if costs for invasive aquatic species were well reported in our database, costs for marine species were not reported in Spain; a possible explanation is that we did not specifically target national administrations with governance in marine species. However, this is a common problem for the global InvaCost project, since only 2% of all global aquatic invasion costs were related to marine-tolerant invasive species (Cuthbert et al. 2021a).

Finally, border controls, phytosanitary measures against invasive pests or private efforts to control invaders, have not been searched specifically. Border control measures exist in Spain. For example, in the Canary Islands, there are strict border controls, but control of invasive species is difficult to quantify separately from other border activities. Some private efforts have been recorded, such as those targeting the eradication of the first outbreak of the invasive termite *Reticulitermes flavipes* in Tenerife Island between 2010 and 2015 (Hernández-Teixidor et al. 2019) or the management of *D. polymorpha* in the Ebro Basin, which costs € 615,000/year to energy companies and € 321,450 in 2009 to the private companies using its water (Durán et al. 2012). However, we argue for a better reporting of these private costs. In relation to damage caused by invaders, it is likely that our targeted research did not succeed in obtaining such information from the public administrations that could hold such data. For example, our database does not include data on damage caused to agriculture or forestry by invasive pest species, such as apple snails or bark beetles (Golzanadera et al. 2012; Joshi and Parera 2017) or damage caused by disease vectors, such as health-associated costs by invasive *Aedes* sp. mosquitoes (Collantes et al. 2015). However, it could also depend on how local funds are distributed, prioritising management actions rather than damage evaluation, which would require additional resources and scientific skills.

Conclusion

This study is the first one attempting to economically evaluate the impact of IAS in Spain. We collected cost data mainly from the literature, regional governments and river basin authorities. Beside certain extrapolated costs on the economic impacts of IAS in the forestry sector, most of the reported costs consisted of funding used for managing established IAS (such as control or monitoring costs). Despite invasive species posing high environmental and economic impacts in Spain (Andreu et al. 2009), most of the collected costs corresponded to management actions, while damage costs were only found for two species. These results suggest the need for further investment in understanding the damage costs of IAS in the country and reporting them. Taxonomically, Spanish environmental managers expended more funds in managing invasive plants than animals and substantial efforts were directed to manage IAS in aquatic environments. From a geographic perspective, a country-level organism responsible for the management of IAS could promote long-lasting research-based management strategies and reporting of costs that expand political borders amongst regions and efficiently coordinate actions amongst all the implicated actors.

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Supplementary material 1

Dataset of the economic costs of invasive alien species in Spain and descriptive variables

Authors: Elena Angulo, Liliana Ballesteros-Mejia, Ana Novoa, Virginia G. Duboscq-Carra, Christophe Diagne, Franck Courchamp

Data type: excel file

Explanation note: Spreadsheets: "InvaCost_3.0_Spain" contains the 3,260 raw entries; "InfoVariables" contains information on each variable and their categories; "Corrections": report of corrections made with respect to the original source (Invacost_3.0).

Link: https://doi.org/10.3897/neobiota.67.59181.suppl1

Supplementary material 2

Figure S1

Authors: Elena Angulo, Liliana Ballesteros-Mejia, Ana Novoa, Virginia G. Duboscq-Carra, Christophe Diagne, Franck Courchamp

Data type: pdf file

Explanation note: (a) Descriptors of the economic costs of invasive alien species in Spain using the non-robust data (extrapolated cost, not occurring and/or unreliable costs) and (b) temporal trends of these economic costs (US$). In (b), each blue circle represents the cumulative cost for a given year, whereas its size is proportional to the number of estimates for that particular year. Average annual costs are calculated in 4-year periods and are represented by black points and horizontal solid lines. Dashed lines connect the average annual costs for these 4-year periods. Non-robust data started with low values in the 2000s increasing highly (mainly due to the predicted costs for the pine wood nematode) for the period between 2008 and 2030. See sample size in Fig. 1.

Link: https://doi.org/10.3897/neobiota.67.59181.suppl2

Supplementary material 3

Lists of species used in Figure 5

Authors: Elena Angulo, Liliana Ballesteros-Mejia, Ana Novoa, Virginia G. Duboscq-Carra, Christophe Diagne, Franck Courchamp

Data type: excel file

Explanation note: Spreadsheets: "Management and damage" contains species recorded as invasive in the country (GISD; Pagad et al. 2018); species in national regulations, European regulations or proposed as potential candidates to be included in European regulations (Carboneras et al. 2018); species having management costs and damage costs in InvaCost. "Management" contains only managed species with their presence in the lists reported in the previous spreadsheet and information about their environment and taxonomy (Env/Phyl), as well as their costs in US$. "Management_non_listed" contains the species not listed in any of the previous lists and the regions where each one has been reported as having management costs. Codes 1-0 mean presence or absence in the list, respectively.

Link: https://doi.org/10.3897/neobiota.67.59181.suppl3

Supplementary material 4

Figure S2

Authors: Elena Angulo, Liliana Ballesteros-Mejia, Ana Novoa, Virginia G. Duboscq-Carra, Christophe Diagne, Franck Courchamp

Data type: PDF file

Explanation note: Lists of the costliest species in Spain considering all the cost types or only management costs and separating robust and non-robust data. (A) Economic costs (US$ million) and (B) number of entries. Colours in the tenth costliest invasive species using the robust data facilitate comparison of species amongst different lists.

Link: https://doi.org/10.3897/neobiota.67.59181.suppl4

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ORIGINAL PAPER

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Surprisingly high economic costs of biological invasions in protected areas

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Abstract Biological invasions are one of the main threats to biodiversity within protected areas (PAs) worldwide. Meanwhile, the resilience of PAs to invasions remains largely unknown. Consequently, providing a better understanding of how they are impacted by invasions is critical for informing policy responses and optimally allocating resources to prevention and control strategies. Here we use the *InvaCost* database to address this gap from three perspectives: (i) characterizing the total reported costs of invasive alien species (IAS) in PAs; (ii) comparing mean observed costs of IAS in PAs and non-PAs; and (iii) evaluating factors affecting mean observed costs of IAS in PAs. Our results first show that, overall, the reported economic costs of IAS in PAs amounted to US$ 22.24 billion between 1975 and 2020, of which US$ 930.61 million were observed costs (already incurred) and US$ 21.31 billion were potential costs (extrapolated or predicted). Expectedly, most of the observed costs were reported for management (73%) but damages were still much higher than expected for PAs (24%); in addition, the vast majority of management costs were reported for reactive, post-invasion actions (84% of management costs, focused on eradication and control). Second, differences between

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costs in PAs and non-PAs varied among continents and environments. We found significantly higher IAS costs in terrestrial PA environments compared to non-PAs, while regionally, Europe incurred higher costs in PAs and Africa and Temperate Asia incurred higher costs in non-PAs. Third, characterization of drivers of IAS costs within PAs showed an effect of environments (higher costs in terrestrial environments), continents (higher in Africa and South America), taxa (higher in invertebrates and vertebrates than plants) and Human Development Index (higher in more developed countries). Globally, our findings indicate that, counterintuitively, PAs are subject to very high costs from biological invasions. This highlights the need for more resources to be invested in the management of IAS to achieve the role of PAs in ensuring the long term conservation of nature. Accordingly, more spatially-balanced and integrative studies involving both scientists and stakeholders are required.

Keywords Invasive alien species · InvaCost · Biodiversity conservation · Monetary impacts · Management actions · Protection status

Introduction

Biological invasions represent a global environmental problem and management challenge (Pyšek et al. 2020; Ricciardi et al. 2021). The plethora of environmental impacts posed by invasive alien species (IAS) range from declines in biodiversity (Ellstrand and Schierenbeck 2000; Vilà et al. 2000; Hejda et al. 2009; Butchart et al. 2010) to disruption of ecological processes and provisioning of ecosystem services (Vitousek 1990; Charles and Dukes 2008; Pejchar and Mooney 2009; Ehrenfeld 2010). IAS also negatively impact human health and well-being (Conn 2014; Hulme 2014; Mazza and Tricarico 2018; Schaffner et al. 2020), and cause losses to multiple sectors of the economy (Pimentel et al. 2005; Martins et al. 2006; Kettunen et al. 2009; Paini et al. 2016; Diagne et al. 2021). Alarmingly, with no signs of abatement in the numbers of established alien species in recent decades, their associated environmental, social and economic impacts will likely continue to substantially increase in the foreseeable future (Seebens et al. 2017, 2020; Bailey et al. 2020). As a result, there is an urgent need for establishing effective management responses. One way of achieving this is by effectively managing IAS in areas that protect a broad range of species and habitats, such as protected areas (PAs)—a pillar for global biodiversity conservation efforts.

With 15.7% of the global land surface and 7.9% of the ocean (www.protectedplanet.net/) currently covered in the network of PAs, the designation of PAs has been a critical means of mitigating biodiversity threats worldwide. In addition, the European Union plans to protect 30% of its land and sea territory by 2030 (https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:52020DC0380). PAs worldwide comprise a large range of designations with different management regimes, ranging from highly to minimally protected sites. When appropriately designed and successfully managed, PAs can be effective in conserving native biodiversity (including species of conservation concern), maintaining ecosystem function and keeping ecosystem services intact (Chape

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et al. 2005; Foxcroft et al. 2011; Geldmann et al. 2013; Daněk et al. 2017; Ziller et al. 2020). Effectiveness of PAs for biodiversity conservation can be measured in many different ways, depending on the conservation goals in place. For example, PA network design can be assessed to determine the diversity of species and habitats, and/or inclusion of the highest priority conservation areas to meet global biodiversity conservation goals (Rodrigues and Gaston 2001; Rodrigues et al. 2004; Le Saout et al. 2013; Heringer et al. 2020). However, regardless of their official designation, PAs tend to be more vulnerable and challenged by IAS than unprotected landscapes, since they often host a larger proportion of native, endemic and threatened species which are less adapted to anthropogenic disturbances (Foxcroft et al. 2013; Heringer et al. 2020).

Understanding the economic costs of IAS is critical to ensure adequate funding for conservation efforts and to design appropriate management actions that will help mitigate impacts and safeguard biodiversity (Dana et al. 2014; Diagne et al. 2020a). However, a detailed understanding of the costs incurred by IAS is still lacking for PAs. A preliminary analysis of the number of post-1970 English-language publications available in the Web of Science on costs of biological invasions (Supplementary Material 1), showed that despite the numerous IAS publications (n = 58,729), studies involving PAs have received relatively little attention (12.6%), and only a few of these evaluate the economic costs in PAs (1.6%). Many IAS studies in PAs have attempted to decipher the drivers of invasions (Gaertner et al. 2014; Gantchoff et al. 2018; Iacarella et al. 2020; Liu et al. 2020; Moodley et al. 2020), thereby improving our understanding of the role of designation type (i.e. nationally designated PAs, such as national parks, have fewer IAS), designation year (i.e. younger PAs have more IAS), PA size (i.e. larger PAs have more IAS) and/or human activities (i.e. IAS increases with accessibility and higher human footprint index) in driving the success and impacts of IAS in PAs (Gallardo et al. 2017; Liu et al. 2020). Yet, despite progress in our knowledge of these ecological and environmental drivers of invasions in PAs, the incurred economic impacts from IAS, and ability to mitigate them, remain unexplored. Moreover, while there is evidence that geographic bias (towards North America and the Pacific Islands) and taxonomic bias (towards plants and insects) largely drive our understanding of IAS success and impact (Pysek et al. 2008; Hulme et al. 2014), thus far, there has been little effort made on exploring the global patterns of IAS in PAs.

To fill the knowledge gaps on the cost of biological invasions in PAs worldwide, we structured our study around three broad aims, each employing a distinct subset of the *InvaCost* database (see Materials and Methods and Fig. 1). Specifically, we sought to: (i) characterize the overall costs of IAS within PAs, based on an all-inclusive PA dataset: we expected higher costs dedicated to management (which prevents the introduction or mitigates the impacts of IAS) than to damage; (ii) investigate whether observed IAS costs (i.e. excluding potential costs, see below) differ between PAs and non-PAs, using carefully matched criteria and after accounting for other factors (i.e. environment, taxonomy, geography, type of costs and economic sectors): we expected lower costs in PAs, which should be better protected from invasion and contain less economic assets; and (iii) further examine which economic and PA characteristics (e.g. year of designation, PA size) drive differences in IAS-related costs, using a subset of PAs with observed cost data: we expected that costs within PAs are driven by both protection and economic characteristics.

Materials and methods

Original data

We used information from the *InvaCost* database (version 4.0 containing 13,123 entries; openly available at https://doi.org/10.6084/m9.figshare.12668570), the most recent, comprehensive database on globally reported economic costs of IAS in English and 15 other languages (Diagne et al. 2020b; Angulo et al. 2021). Each database entry contains a cost value associated to a unique combination of cost descriptors, including: (i) bibliography of the documents reporting the costs; (ii) details on the impacted area (e.g. location, spatial scale, environment, and whether the location corresponded to a protected area); (iii) taxonomy of the IAS causing the cost, (iv) temporal extent over which the cost occurred, or was predicted to occur; (v) type of cost: whether the cost is a management action

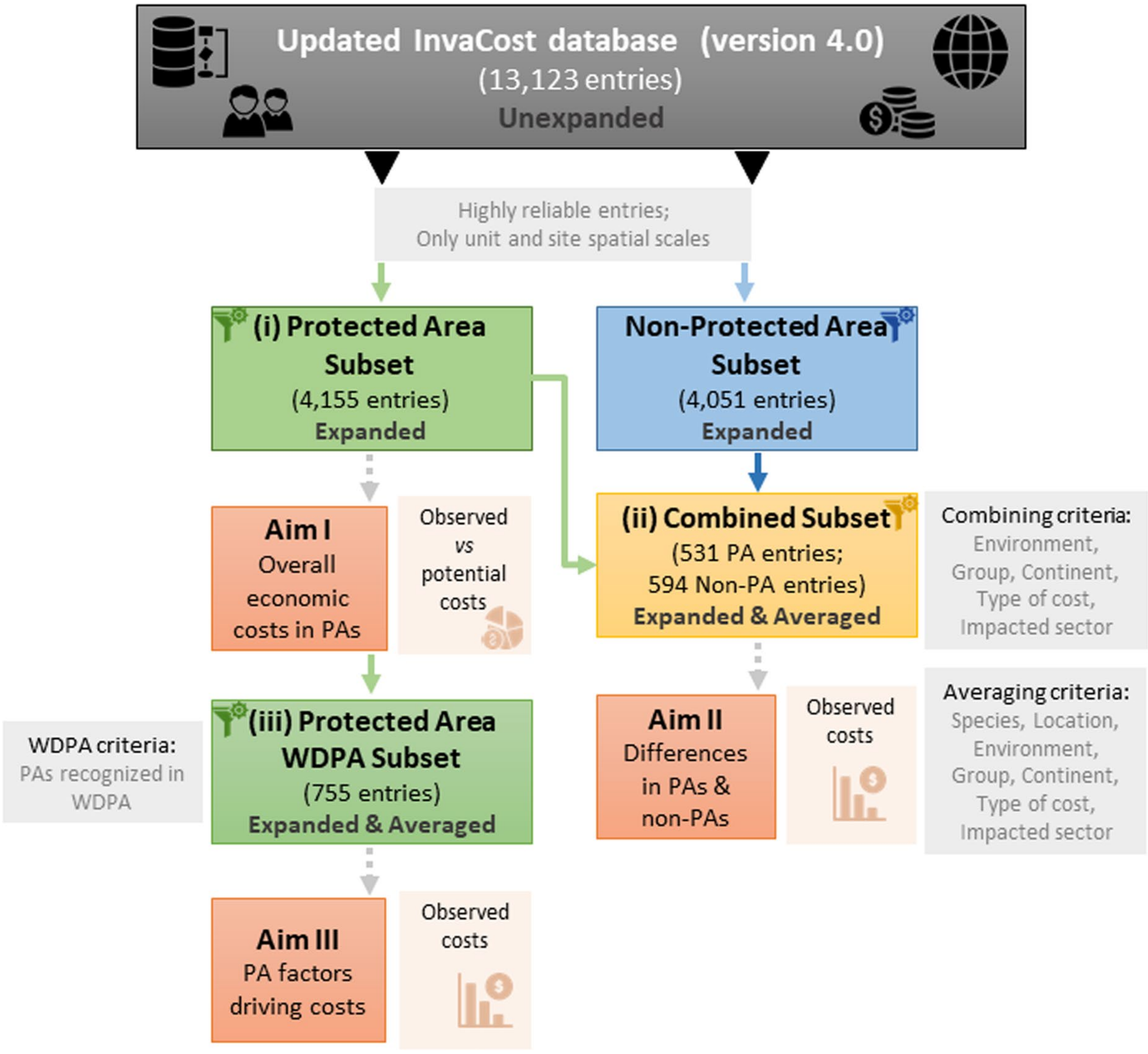


Fig. 1 Workflow chart detailing the filtering process and cost entry eligibility, carefully designed to address the three study aims. Opaque, gray boxes indicate the main criteria used to filter each subset while opaque, orange boxes indicate whether the subset contains observed (i.e. cost incurred by an IAS) and/or potential costs (i.e. extrapolation cost for an IAS beyond its current distribution and/or predicted in the future within or beyond its current invasive range)

or an economic damage; (vi) implementation of the cost: whether the cost was observed (actually incurred) or potential (extrapolated or predicted to occur); (vii) impacted sector (which activity, market or societal sector was related to the cost); and (viii) reliability of the source document providing the cost estimate (whether the cost source was peer-reviewed, official and/or reproducible, or not). To allow for comparable cost values, all cost estimates were standardized and converted to 2017 US$ (Diagne et al. 2020b). We re-classified some of the original columns of the *InvaCost* database to ensure that our study is comparable with other PA studies (Supplementary Material 2) and then filtered and extracted the data into different subsets as described below (all subsets are available in Supplementary Material 3).

Data processing

Data preparation

To obtain robust subsets from the original database, we only included highly reliable and appropriately-spatially scaled cost entries (Fig. 1). Specifically, we first filtered the *Reliability* cost descriptor column to only select those costs characterised as having "high" reliability. This distinction of the cost entries into having "high" or "low" reliability, indicates if the approach used for cost estimation in the original source is reported, reproducible and traceable (see Diagne et al. 2020b for details on criteria used). Second, we filtered the *Spatial scale* cost descriptor column to exclude those estimated at "country" and "regional" scales. Rather, we only considered those at the "site" and "unit" scales, since these are the spatial scales in which PA costs occur. Subsequently, we used the information provided in the *protectedArea* column to identify the status of land protection for each cost entry: "yes", only pertaining to protected areas; "no", only pertaining to non-protected areas; or "NA", when there was no information about the status or when costs were attached to both protected and non-protected areas. We excluded entries that were identified as "NA".

We further refined the resulting dataset to specifically address the aims of this study. Thus, we applied consecutive filtering procedures which resulted in the creation of three subsets (Fig. 1): (i) PA cost entries only (hereafter referred to as the *Protected Area Subset*; see subset details below); (ii) costs for protected and non-protected areas together, to examine the effect of protected areas on cost, after controlling for other variables (hereafter referred to as the *Combined Subset*; see subset details below); and (iii) costs for PAs listed in the World Database on Protected Areas (WDPA; UNEP-WCMC and IUCN 2020), which allowed the addition of WDPA descriptor variables (hereafter referred to as the *Protected Area WPDA Subset*; see subset details below).

All cost estimates were annualized in the original database (see *Cost estimate per year* columns; Diagne et al. 2020b). Here, we expanded all subsets (Subsets i-iii in Fig. 1) to account for the duration (in years) of each cost estimate by using the *expandYearlyCosts* function of the 'invacost' package version 0.3–4 (Leroy et al. 2020). This function relies on information contained in the *Probable starting year adjusted* and *Probable ending year adjusted* columns to repeat each annualized cost as many times as years of cost occurrence. This resulted in comparable annual costs for all cost entries (i.e. expanded format) which are unbiased with respect to time. As an illustration, cost estimates spanning multiple years (e.g. $10 million for the period 2006–2010) are divided according to their duration (e.g. $2 million for each year between 2006 and 2010).

Averaging annualized costs estimates

For the *Combined Subset* and the *Protected Area WDPA Subset* (Subsets ii and iii in Fig. 1), we averaged the annual cost values across descriptors so that individual entries associated with a single species from the same location and environment, which incur the same type of costs and affect the same activity sectors, were averaged into one single cost entry (Fig. 1, Supplementary Material 2 and 3 for the interpretation of these descriptive fields and the subsets). This allowed us to control for pseudoreplication and also partition the effect of these factors when estimating the effect of PAs. All statistical analyses presented in the main text were performed on these expanded and averaged subsets.

The protected area subset

Filtered and expanded entries classified as PAs (i.e. "yes" in the *protectedArea* column) resulted in the *Protected Area Subset* with a total of 4155 cost entries, which was used to describe economic trends associated with IAS in PAs (Subset i in Fig. 1). Regarding the temporal variation of the costs of IAS in PAs, we used the *summarizeCosts* function of the 'invacost' R package to quantify annual average costs at five-year intervals between 1975 and 2020 (Leroy et al. 2020). We considered both the magnitude of costs (in 2017 US$), as well as the number of cost entries (expanded) over time. We also investigated the spatial distribution of PA associated costs by continents, explored the taxonomic groups responsible for costs in PAs, and categorised the type of costs and the economic sectors impacted by the cost. Moreover, we examined costs separately for observed costs (i.e. if the cost was actually incurred) and potential costs (i.e. if the cost was extrapolated or predicted to occur) using the *Implementation* column. It is expected that potential costs will be higher than observed costs, because observed costs are restricted to actual, often hardly-quantifiable impacts that are monetized in a limited time frame and in areas of established invasions. Conversely, potential costs are extrapolations or predictions of costs that will occur in the future or in the probable IAS range. However, potential costs should not be ignored, since they provide information concerning costs that are difficult to quantify or may occur under different scenarios. Thus, we report potential costs in PAs using this complete subset, but clearly distinguish them.

The combined subset

A total of 4,051 cost entries were identified in non-PAs (i.e. "no" in the *protectedArea* column) and constituted the *Non-protected Area Subset* (Fig. 1). The *Combined Subset* comprises the *Non-protected Area Subset* and the *Protected Area Subset,* and further filters cost entries of both subsets by the *Implementation* column, in order to retain only observed costs and remove potential costs (Fig. 1). The resulting entries of both subsets were then selected using matched rows relating to their combined environment, taxonomic group, continent, type of cost, and impacted sector (Supplementary Material 2 and 3). Thus, we only retained cost entries containing the specified combination of these five descriptors in both the *Protected Area Subset* and in the *Non-protected Area Subset*, as our interest was to compare PA versus non-PA costs. This resulted in a total of 1,125 expanded and averaged combined entries (531 PA vs 594 non-PA entries), which constitutes the *Combined Subset* (Subset ii in Fig. 1) and was used to identify descriptors driving differences in costs between protected and non-protected areas.

The protected area WDPA subset

Observed costs (in the *Implementation* column) from the *Protected Area Subset* that were associated with PAs categorized within the WDPA (UNEP-WCMC and IUCN 2020) were considered as the *Protected Area WDPA Subset*. This resulted in 755 expanded and averaged entries, and this subset was used to understand which descriptors drives costs generated by IAS in PAs (Subset iii in Fig. 1). To do this, we extracted information on four descriptors related to characteristics of the PAs from the WDPA (i.e. PA designation, PA designation year, PA surface area (km^2), and human development index (HDI)) and four descriptors related to the characteristics of the costs from the *InvaCost* database (i.e. continent, environment, taxonomic group, and type of cost) (Supplementary Material 2 and 3).

Statistical analyses

The combined subset: differences in economic costs between protected and non-protected areas

In order to compare differences in the economic costs of IAS between PAs and non-PAs, and to understand which descriptors could affect these differences, we performed a multiple linear regression using the *Combined Subset* (see Fig. 1). The dependent variable was the *average yearly economic cost* ($\log_{10}$-transformed) and the independent variables included a binary *PA status* factor (i.e. whether the cost pertains to a PA or non-PA) and its two-way interactions with each of the following invasion descriptors: *continent*, *taxonomic group*, *environment*, *impacted sector* and *type of cost* (Table 1). This allowed us to assess the differences in the *average yearly economic cost* between PAs and non-PAs for these descriptors. We only focussed on the interaction effects of PA status with each of these descriptors to explain costs and not on the main effects of the descriptors individually, with the exception of *PA status. PA status* was evaluated as a main effect in order to quantify the overall difference between PAs and non-PAs. Otherwise, its inclusion as an interaction term allowed for disentangling the contribution of the descriptors driving differences between PAs and non-PAs. Prior to performing this analysis, we assessed that none of the predictors were highly intercorrelated, suggesting the absence of multicollinearity (Pearson's $r < 0.65$; Supplementary Material 4). Consequently, all predictors were retained in the analysis. We used the adjusted R^2 to assess the percentage of mean annualized economic cost variation that is explained by the models. Significant interactions were assessed using the *drop1* function to obtain Type III sum of squares ANOVA containing p-values from an F-test. Residuals were analyzed using the *simulateResiduals* function of the 'DHARMa' package version 0.3.3 (Hartig 2020) and they satisfied all classical regression assumptions. Additionally, when an interaction factor showed a significant effect, we carried out a post-hoc Wilcoxon Signed Rank test with Holm correction in order to determine which categories were significantly different between PAs and non-PAs.

The protected area WDPA subset: factors affecting costs of invasive alien species across protected areas

To assess which variables potentially affect the economic costs of IAS across PAs, we performed a multiple linear regression using the data corresponding to the *Protected Area WDPA Subset* (see Fig. 1). We used the $\log_{10}$-transformed *average yearly economic costs* as the dependent variable and added *PA designation*, *year of PA designation*, *PA surface area*, *human development index*, *continent*, *taxonomic group*, *environment*, and *type of cost* as independent variables. We first excluded incomplete cases (i.e. rows with missing values), and we assessed the correlation among predictors (all predictors were retained in the analysis; Pearson's $r < 0.65$; Supplementary Material 5). We then ran a multiple linear regression, and similar to the previous model, produced the output using the *drop1* function, assessed residuals (which satisfied all regression assumptions) and tested for differences among categories for the significant factors using the Wilcoxon signed-rank test with Holm corrected p-values.

For each test, when reporting the statistical results of average economic costs, we provide medians and standard deviation because these estimates fairly approximate the mean values of our $\log_{10}$-transformed data and avoid skewed distributions due to cost outliers. All figures were produced in R using ggplot2 (Wickham 2016).

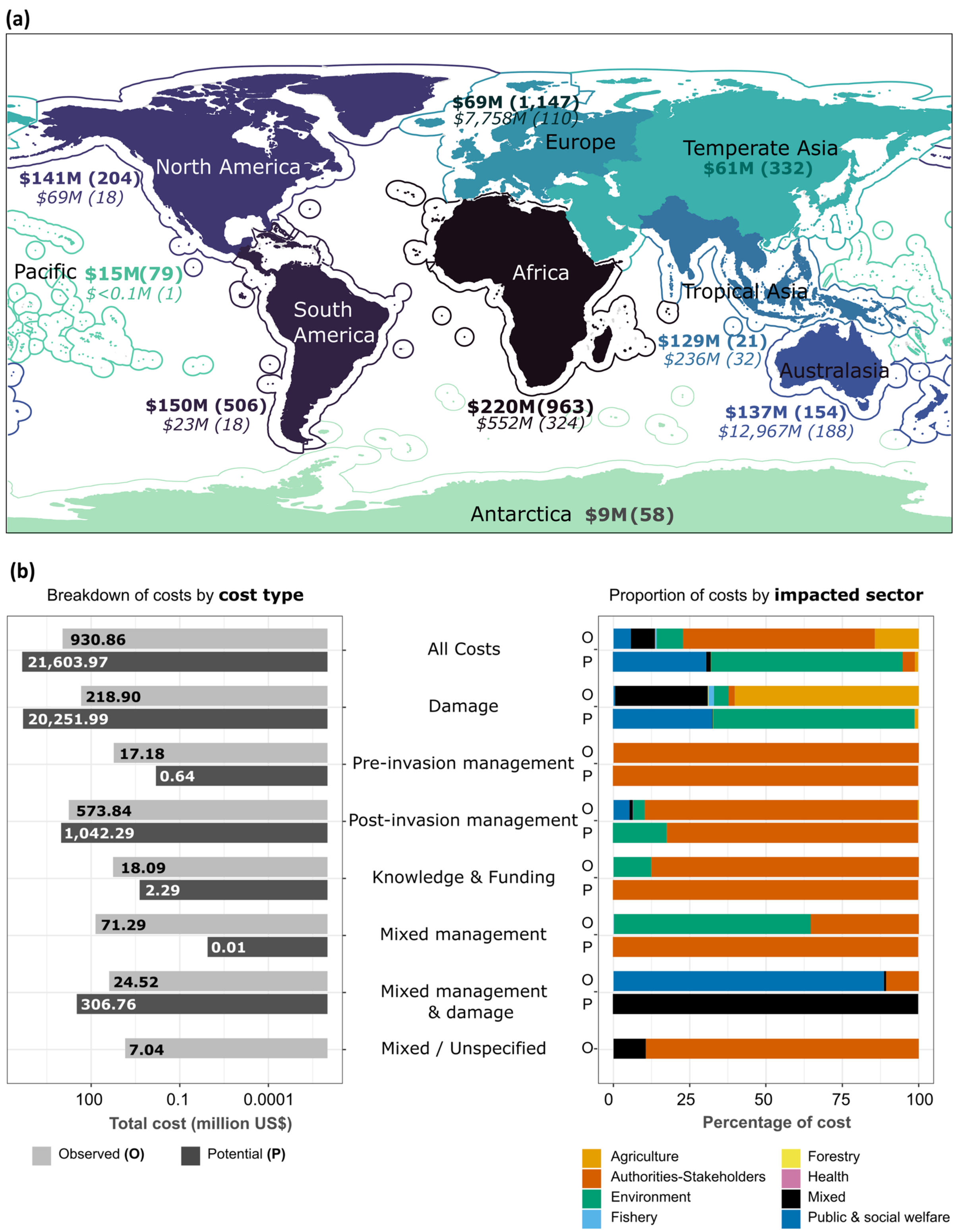
Results

What are the overall economic costs of invasive alien species in protected areas?

Using the complete dataset (*Protected Area Subset*), we show that the total reported economic costs in at least 55 PAs (excluding 40 unique references with unspecified PA names) amounted to $22.24 billion over the last 46 years (1975–2020). These high costs represent only a fraction of the designated PAs around the world (in total 266,561 terrestrial and marine PAs) suggesting that global PA costs could be several orders of magnitude higher. More specifically, observed costs amounted to $930.61 million between 1975–2020 and averaged $20.23 million annually, while potential costs, as expected, amounted to $21.31 billion and averaged $463.34 million over the same period (Supplementary Material 6). Both these cost types were generally characterised by an increase over time, with potential costs increasing markedly between 1995 and 2000. Furthermore, observed costs exhibited a gradual increase over time, with reductions in recent years likely due to time lags in cost reporting. The number of entries for both types of costs has been increasing over time, and especially those of observed costs.

PA costs were not distributed homogeneously across continents (Fig. 2a). In particular, Africa reported the greatest share of observed costs (24%), followed by South America (16%), North America (15%), Australasia (15%), Tropical Asia (14%), Europe (7%), Temperate Asia (7%), Pacific Islands (2%) and Antarctica (1%). However, this pattern is quite different when taking into account both observed and potential costs: more costs were reported by PAs located in Australasia (60%), followed by Europe (36%) and distantly followed by PAs located in Africa (3%), Tropical Asia (1%), North America (0.3%), and South America (0.1%). Potential costs were not reported for Antarctica or Temperate Asia. Moreover, most cost occurrences were reported for PAs in Africa (1,287 cost entries) and Europe (1,257 entries), followed by South America (524 entries), Australasia (342 entries), Temperate Asia (332 entries) and North America (222 entries), while the remaining three continents reported less than 200 entries.

In terms of the types of IAS cost incurred by PAs, the majority of observed costs were management focused (i.e. "pre-invasion", "post-invasion", "knowledge and funding" and "mixed management" costs), which totalled $680.40 million (73%), thereby, dominating over damage costs (i.e. $218.90 million) (Fig. 2b). Within management costs, "post-invasion" management (i.e. control, eradication, harvesting, management and monitoring) represented the highest proportion of observed costs caused by IAS (84% of management costs). In terms of potential costs,


(a)
$69M (1,147)
$7,758M (110)
Europe
Temperate Asia
$61M (332)
North America
$141M (204)
$69M (18)
Pacific
$15M(79)
$<0.1M (1)
Africa
South America
Tropical Asia
$129M (21)
$236M (32)
Australasia
$150M (506)
$23M (18)
$220M(963)
$552M (324)
$137M (154)
$12,967M (188)
Antarctica
$9M(58)
(b)
Breakdown of costs by cost type
Proportion of costs by impacted sector
930.86
21,603.97
All Costs
218.90
20,251.99
Damage
17.18
0.64
Pre-invasion management
573.84
1,042.29
Post-invasion management
18.09
2.29
Knowledge & Funding
71.29
0.01
Mixed management
24.52
306.76
Mixed management & damage
7.04
Mixed / Unspecified
O
P
100
0.1
0.0001
Total cost (million US$)
0
25
50
75
100
Percentage of cost
Observed (O)
Potential (P)
Agriculture
Authorities-Stakeholders
Environment
Fishery
Forestry
Health
Mixed
Public & social welfare

◂**Fig. 2** Total observed and potential costs of invasive alien species (US$ million) in protected areas, as well as reported cost entries, presented according to their: **a** spatial patterns across continents. Colours represent continents together with their associated marine territories. The total observed economic costs are displayed in bold, potential costs are in italics and their corresponding number of cost entries are shown in parenthesis; and **b** cost type and associated impacted sectors. For the impacted sector, upper bars correspond to observed costs (O) and lower bars to potential costs (P)

damage costs constituted the majority (94%), whilst management costs represented only 5%.

Governmental services and/or official organizations (e.g. conservation agencies, forest services, or associations) that allocate funding for the management of biological invasions ("authorities and stakeholders") reported the highest observed costs ($584.69 million; 63%) compared to other sectors (Fig. 2b). This sector accounted for more than 80% of all types of management costs, except for "mixed management", where it represented 35% of costs. The "agriculture" and "public and social welfare" sectors sustained the most observed "damage" and "mixed damage and management" costs respectively (60% and 89%, respectively). The "environment" and "public and social welfare" sectors accounted for 94% of all potential costs generated by IAS in PAs (63% and 31%, respectively) and collectively close to 100% of damage costs (66% and 33%, respectively). "Authorities and stakeholders" accounted for the majority of potential management costs (ranging from 82 to 100% according to the management type).

Plants dominated the *Protected Area Subset* with 64% of the observed cost entries and 79% of the potential cost entries (Fig. 3). However, the values of observed costs for animals were four times larger than for plants ($641 vs 172 million, respectively) and potential animal costs were almost 1.5 times larger than for plants ($13 and 9 billion, respectively). Magnoliopsida (61% of observed plant costs and 63% of observed plant entries), Mammalia (43% of observed animal costs and 50% of observed animal entries) and Insecta (42% of observed animal costs and; 15% of observed animal entries) substantially influenced both costs and number of entries in their respective plant and animal groups.

Specifically, the costliest IAS across PAs comprised mammals (e.g. rats and cats), aquatic plants (e.g. *Ludwigia* sp.), insects (e.g. mosquitoes and coleopterans: *Aedes albopictus*, *Sternochetus frigidus*), and one tree (*Prosopis juliflora*) (Fig. 4a). With the exception of *Ludwigia* sp. (the second costliest IAS in our dataset), terrestrial IAS presented higher costs than IAS invading semi-aquatic and aquatic PA environments. The costliest IAS in semi-aquatic and aquatic environments were mainly plants (e.g. *Eichhornia crassipes, Baccharis halimifolia, Myriophyllum spicatum*), one amphibian (*Rhinella marina*), one mammal (*Castor canadensis*), and mosquitoes (*Aedes albopictus*).

IAS with the highest reported costs differed across continents (Fig. 4b). Reported costs were mostly associated with trees and insects in Africa and Tropical Asia, animals (particularly reptiles, mammals and two invertebrate species) in Temperate Asia; mammals in Australasia; aquatic and semi-aquatic plants, and the insect *Vespa velutina* in Europe; mammals and plants in South America and the Pacific; and one mammal *Sus scrofa*, two aquatic plants, and two insects affecting forests in North America.

How do costs differ between non-protected and protected areas?

Using the *Combined Subset*, we first observed that the reported *average observed economic costs* caused by IAS did not differ significantly between non-PAs and PAs ($F = 2.72$, $p = 0.100$; Fig. 5a). This could be explained by the skewness of the cost data, reflected by the fact that PAs had a 3 × lower mean cost but a 3 × higher median compared to non-PAs (Supplementary Material 7b). Note that this outcome was a main effect and excluded interaction effects. When looking at the interactions of *PA status* with the descriptors, two interactions were significant: *PA status* interacting with the *environment* and with the *continent* ($F = 6.88$, $p < 0.001$; $F = 11.02$, $p < 0.001$, respectively; Supplementary Material 7a). The percentage of variance explained by the model was 28.39% (adjusted R^2).

In terms of the type of *environments* and *PA status*, only terrestrial ecosystems displayed a significant effect with higher costs incurred in PAs compared with non-PAs ($p < 0.001$; Fig. 5b; Supplementary Material 7b). Additionally, the total reported IAS cost entries in terrestrial ecosystems accounted for 72% across all environments. Examining the expenditure for IAS across *continents*

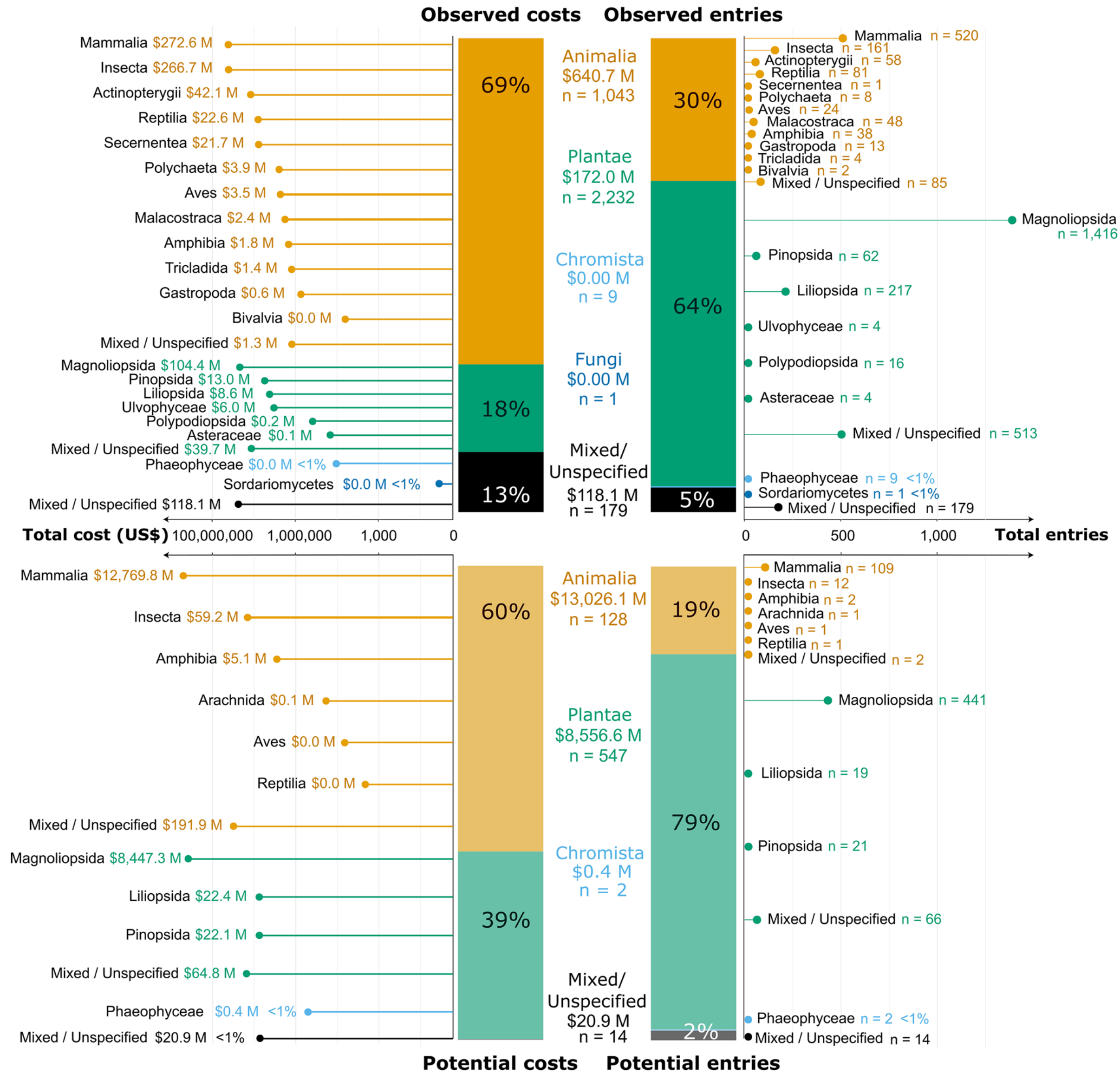


Fig. 3 Taxonomic distribution of cost estimates (US$ million, left panels) and cost entries (right panel). Coloured bars represent the percentage of costs by kingdoms and lollipops depict total economic costs of the associated classes

and *PA status*, we found that PA costs incurred in Africa and Temperate Asia were significantly lower compared to costs incurred in non-PAs ($p = 0.002$, $p < 0.001$, respectively; Fig. 5c; Supplementary Material 7b). Conversely, in Europe, IAS costs were significantly higher in PAs compared to costs incurred in non-PAs ($p < 0.001$).

Which factors influence invasive alien species costs across protected areas?

Using the *Protected Area WDPA Subset*, we found six descriptors that are important in driving the *average yearly observed costs* across PAs (three PA descriptors and three cost descriptors): *year of designation*,

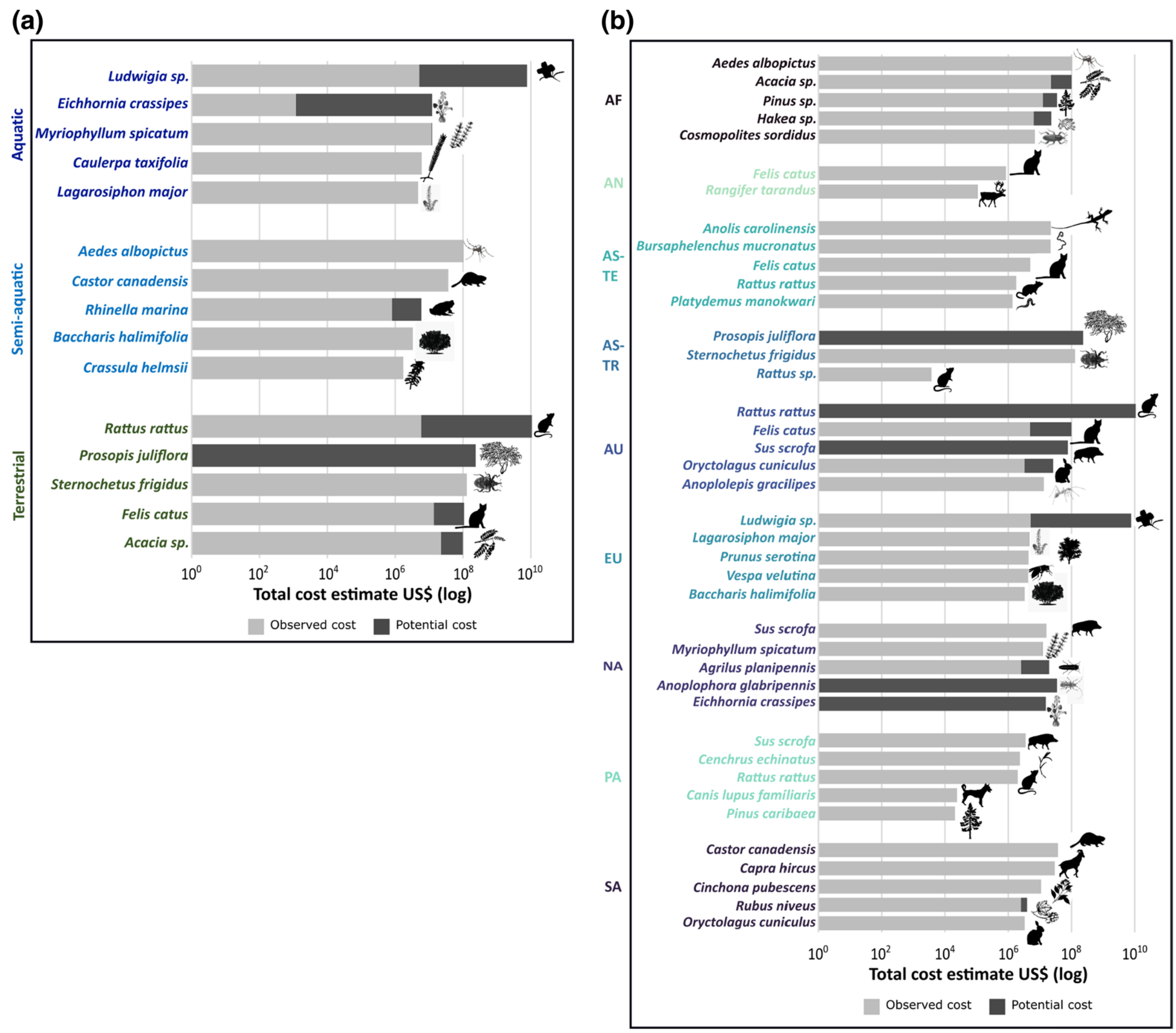


Fig. 4 Bar plot depicting the distinction of observed (light gray) and potential (dark gray) costs in protected areas for the costliest species across **a** environments and **b** continents. On the right panel: AF = Africa, AN = Antarctica, AS-TE = Asia-Temperate, AS-TR = Asia-Tropical, AU = Australasia, EU = Europe, NA = North America, PA = Pacific, SA = South America. Clip arts were obtained from phylopic.org

protected area size, *human development index*, *type of environment*, *continent* and *taxonomic group;* Supplementary Material 8a, b). These descriptors explained 20% (adjusted R^2) of the variance.

Costs of IAS in PAs were significantly related to the *year of designation* and the *protected area size*. Costs increased in recently designated PAs and with PA size ($F = 8.54$, $p = 0.004$; $F = 5.20$, $p = 0.023$, respectively; Fig. 6a, b). Further, we found a significant positive relationship between the average costs of IAS and the HDI, which represents the socio-economic wellbeing of a country ($F = 7.19$, $p = 0.008$; Fig. 6c).

Costs in PAs also differed according to the environment ($F = 3.84$, $p = 0.010$). Terrestrial environments sustained significantly higher costs compared to costs incurred in aquatic environments ($p = 0.001$), while costs in semi-aquatic environments were not significantly different to other environments (Fig. 6d). Moreover, PA costs in terrestrial environments comprised the majority of

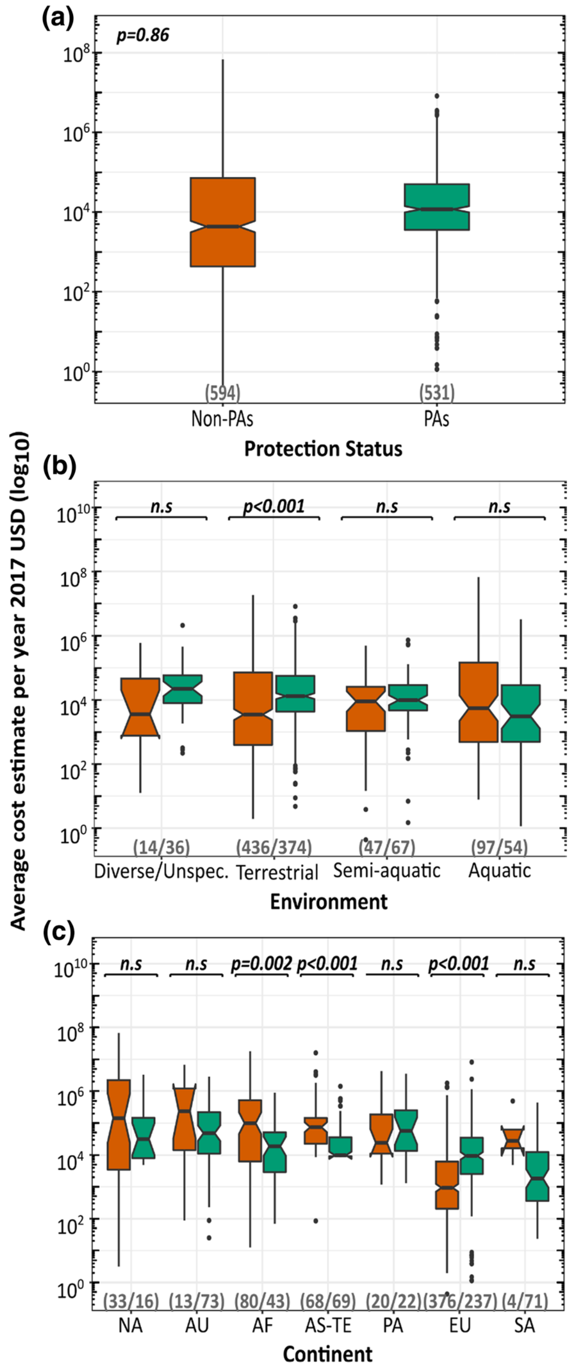


Fig. 5 Distribution of the average economic observed cost ▸ estimates ($\log_{10}$-scale) incurred by species across: **a** non-protected and protected areas; **b** environments with different land protection statuses; and **c** continents with different land protection statuses (NA = North America, AU = Australasia, AF = Africa, PA = Pacific, AS-TE = Asia-Temperate, SA = South America, EU = Europe). Boxplots display median (line), interquartile range (box) and range (whiskers), and solid circles display outliers. Significant differences between PAs and non-PAs for each category are marked using *p*-values above each boxplot pair, and were tested using a Kruskal Wallis rank sum test for (**a**) and Wilcoxon signed-rank post hoc comparisons for (**b, c**). Sample sizes are shown in brackets below each box

reported cost entries (69%) and incurred 40% of the total *average yearly economic costs*.

With respect to the differences in IAS costs regionally, continent was a significant factor ($F = 4.69$, $p < 0.001$; Fig. 6e). Specifically, costs in European PAs were only similar to South American PAs and differed significantly to all other groups. In turn, Africa and South America were significantly different to Australasia, Pacific Islands and Antarctica. Additionally, South America significantly exceeded Temperate Asia and North America. Further, Africa and South America incurred the majority of the total *average yearly economic costs* (40% and 28%, respectively), followed by Australasia (10%), Europe (8%), North America (7%), Pacific Islands (4%), Temperate Asia (2%) and Antarctica (1%) (Fig. 6e). Despite the highest reported costs incurred in African PAs, the reported number of cost entries are not congruent (12% of all cost entries). On the contrary, European PAs incurred much lower costs despite having the highest number of reported costs (41% of all cost entries). Lastly, since the effect of continents among covariates is context-dependent, this suggests that the overall effects of spatial distribution on the magnitude of costs is significantly affected by IAS (Supplementary Material 8a, b).

Average yearly economic costs in PAs also differed significantly with respect to different taxa ($F = 4.54$, $p = 0.004$; Fig. 6f). Costs were similar between invertebrates and vertebrates ($p = 0.368$) but these groups incurred significantly higher costs than plants ($p = 0.001$; $p < 0.001$, respectively). Surprisingly, even though more costs were reported for plants (49% of cost entries), they incurred the lowest total costs (4.6% of the *average yearly economic costs*).

Discussion

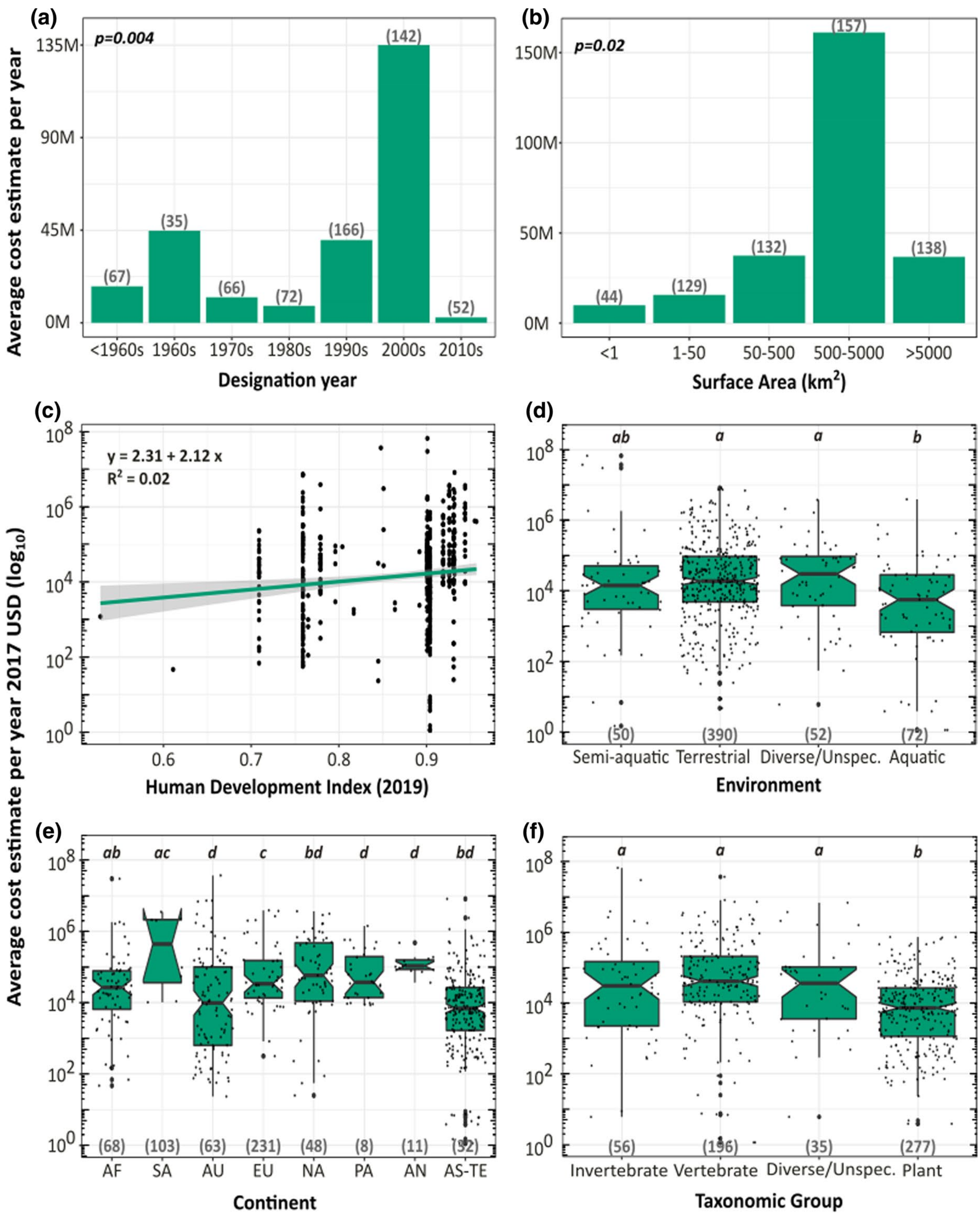
Where do we stand?

This study provides the first comprehensive compilation and analysis of reported costs of biological invasions in PAs around the world, which amounts to $22.24 billion over the last 46 years (1975–2020). Our results partially confirm our expectations that costs of IAS in PAs are (i) high, but predominantly comprise management costs (nonetheless, the proportion of damage costs was higher than expected); (ii) different between PAs and non-PAs, but only within certain continents and environments; and (iii) driven by factors linked to both the invasion costs and PA characteristics. Firstly, we showed that the highest number of reported entries was in Europe but the highest observed costs from invasions in PAs were found in Africa. These costs were caused primarily in terrestrial environments and caused mainly by mammals and insects (and surprisingly to a lesser extent by flowering plants, though more cost entries were reported for plants). Most observed costs were incurred post-invasion and largely affected PA authorities and stakeholders. Secondly, we found that, in general, mean invasion-related costs in PAs were globally similar compared to matched costs in non-PAs. However, per unit of area, costs are higher in PAs (costs in PAs represent 20% of the total cost, when the combined area for which we have PA costs is equivalent to ~0.3% of the total global PAs), even though we expected the opposite due to PAs consisting of less monetized assets and less disturbances (such as agriculture, infrastructure, human population density, etc.). Nevertheless, costs in PAs were significantly higher than non-PAs only in Europe and terrestrial environments. Thirdly, we found that the costs of IAS also increased in PAs that were recently designated, are larger in size and those located in countries with a higher HDI. These results highlight already burgeoning but growing economic impacts of biological invasions in PAs, despite pervasive knowledge gaps at geographic, taxonomic and sectoral scales, context-dependencies and challenges to cost estimation and collation.

Taxonomic bias

Our study, like many others in invasion science, reflects a strong taxonomic bias, in terms of economic costs, research effort and taxonomic awareness (Pysek et al. 2008; Rico-Sánchez et al., 2020). Invasive predatory mammals are largely known to have the most devastating effects on biodiversity worldwide (Doherty et al. 2016; Liu et al. 2020). This aligns with findings in this study that mammals are the costliest to manage in PAs, and is similar to patterns within PAs of other countries (see for example Rico-Sánchez et al. 2021 in Mexico or Ballesteros-Mejia et al. 2021 in Ecuador). Further, in line with the suggestions of Pyšek et al. (2008) indicating that invasive invertebrates were abundantly studied, we found that insects were among the costliest species in PAs. These costs were mostly driven by the mango pulp weevil (*Sternochetus frigidus*; primarily costs incurred in a Palawan game refuge and bird sanctuary), followed by the Asian tiger mosquito (*Aedes albopictus*). The high investment in controlling this weevil is likely due to mango, the third most important fruit crop in the Philippines. Mosquitoes pose a significant threat to humans, as they can serve as vectors of pathogens which can lead to the spread of diseases (Schaffner et al. 2013). In this study, the high expenses for the Asian tiger mosquito were incurred in the Réunion National Park, which is not surprising since this species has become a major human health concern across the island of Réunion (Latreille et al. 2019).

Although we found that plants are managed more often in PAs (as shown by the highest proportion of reported cost entries in the *Protected Area WDPA Subset*; 49%), their reported observed costs are substantially lower than those of either mammals or insects. Among plants, flowering taxa accrued the highest reported expenses, with most costs associated with aquatic plants (*Ludwigia* sp. and all incurred in France; Dandelot et al. 2008) and trees (*Acacia* sp.). Globally, invasive plants, specifically trees, have significant (and growing) impacts on the environment and the economy (Richardson and Rejmánek 2011; Hirsch et al. 2017). Plants are generally studied more than other taxonomic groups (Pysek et al. 2008; Warren II et al. 2017) and successfully dominate many ecosystems (Pyšek et al. 2017). However, plant invasion management is notoriously difficult and high management costs can be associated with plant control and/or eradication (Gardener et al. 2010), due to for example, persistent seed banks (Gioria et al. 2012; Strydom et al. 2017), or limited effective management of aquatic plants (e.g. submerged macrophytes; Hussner et al., 2017; Coughlan et al., 2020). Furthermore, failed control attempts may not be reported, potentially distorting the true magnitude of damage costs and management expenditure (Zenni and Nuñez



2013). This highlights the importance of long-term control efforts, as well as inclusive reporting of all invasion costs.

Looking at the type of costs associated with these species, it is apparent that the prevalent management focus on mammals (pigs and rats), insects (mango

◂**Fig. 6** The relationship between the average observed economic costs and **a** the year in which the PA was designated (y-axis in US$ million); **b** the size of the PA (y-axis in US$ million); **c** the invaded countries' HDI; **d** the environment in which the taxa causing impact in PAs are located; **e** the continent in which the impacted PA is located (AF = Africa, SA = South America, AU = Australasia, EU = Europe, NA = North America, PA = Pacific, AN = Antarctica, AS-TE = Asia-Temperate); and **f** the taxonomic group of the invasive alien species. In (a) and (b) the data were categorized for better visualization, but in the model these variables were fitted as continuous variables. Categories with different letters show significant differences among them, which were tested using the Wilcoxon signed-rank test. For the boxplots, the solid line shows the median, the lower and upper hinges of the box represent the lower and upper quartiles, the whiskers indicate the range of the data, solid circles are outliers, and solid squares depict the observed data points. Sample sizes are in brackets below each box

weevil) and plants (water primrose) in PAs is mostly a result of their associated damage costs (i.e. damage was the second highest type of observed cost reported after post-invasion management). In this regard, it is important to emphasize that our cost data predominantly relate to economic rather than ecological effects. However, the costliest species economically might not be the most ecologically harmful. Given ecological impacts of biological invasions are seldom monetized due to their indirect nature, it is important to bear in mind that our data are inherently taxonomically biased towards IAS that incur economic cost, either through management, or damages to primary sectors such as agriculture and fisheries. While evidence for economic impacts should not underpin investments to safeguard biodiversity from global change pressures, our results provide broad scale incentives for better management within PAs to contain and prevent both current and future IAS.

Geographical bias

IAS research in PAs has shown to be geographically biased towards the Americas and Pacific Islands while less frequently studied regions include Europe, Africa and Asia (Hulme et al. 2014). However, conversely, we show that observed costs are more frequently reported in Europe, Africa and South America, respectively, while cost reporting for PAs in the Pacific Islands is considerably lagging. Highest cost reporting in Europe may firstly be explained as an artefact of the available literature as well as greater opportunities to study invasions across different contexts (e.g. landforms, islands, peninsulas and climates), especially since Europe contains the highest number of PAs worldwide (n = 158,450; UNEP-WCMC and IUCN 2020). Secondly, the implementation of the European Union's Natura 2000 PA network, the largest coordinated network of PAs globally, may have contributed to shaping this pattern (European Environmental Agency 2012), which could be supported by a concomitance between an increase in the cost data recorded in our database and the establishment of the Natura 2000 PA network since the early 1990s. Overall, geographic unevenness in cost reporting might also reflect differences in economic output among countries, because GDP is predictive of economic damages by IAS as well as investments towards management (Haubrock et al., 2021). Accordingly, countries with a higher GDP may be more likely to invest in managing IAS in PAs, conduct research on IAS impacts, and/or designate and conserve PAs more broadly.

We identify marked gaps in the spatial distribution of our complete dataset (i.e. subset i). According to the number of PAs classified in the WDPA dataset, our complete dataset reports PA costs from only 0.3% of the PAs in Africa, 0.3% in South America, 0.1% in North America, 0.1% in Asia and the Pacific, and 0.03% in Europe. This substantial underrepresentation across space further highlights that the high costs presented here represent a very small fraction of the real economic burden of invasions in PAs. Despite these knowledge gaps and greater cost reporting in Europe, African and South American PAs incurred greatest total observed costs — probably an artefact of spatial representation in our data (e.g. Africa and South America had ten times more PA coverage in *InvaCost* than Europe). The majority of recorded costs in Africa were accrued in South Africa (i.e. Western Cape Province and Kruger National Park), a leader in invasion biology research with a long history of (i) conservation efforts (van Wilgen et al. 2020), as an example, through substantial investments in IAS management in the Western Cape by the South Africa's Natural Resource Management Programmes (van Wilgen et al. 2010) and (ii) research effort in invasion biology (Richardson et al. 2020), stimulated for example by the highest number of IAS in the Kruger National Park where control efforts have taken place since the 1950s (Foxcroft and Freitag-Ronaldson

2007; Foxcroft et al. 2017b). In South American PAs, most research investment, as reflected by the high observed total costs, largely occurred in the Galápagos Islands, Ecuador, where IAS are the biggest biodiversity threat (Trueman et al. 2010). Consequently, the high economic costs reported on these islands may be the result of intensive management activities (Gardener et al. 2010, 2013), as well as the importance of the Galápagos Islands to both biological research and tourism (Ballesteros-Mejia et al. 2021).

These trends may additionally be influenced by differential cost reporting practices among regions, with many countries evidently lacking in their cost reporting efforts. Therefore, our results must be interpreted in the context of the available data and may change as further costs become available. Given the differences in PA designations, further research should be conducted at the country-scale, particularly because PAs are not uniformly classified across the world. Here, we coarsened our analyses to the global scale to avoid (i) insufficient sample size for some regions which would have been otherwise removed from our analyses; (ii) making dubious assumptions on potential uniformness in PA designation within each continent; and (iii) counteracting the global-scale approach adopted throughout this study. Consequently, our study should be seen as a starting point for further investigation at finer scales, which will allow more specific conclusions to be drawn and provide context-based recommendations.

Methodological bias

Sporadic cost reporting for PAs, as evidenced both by our data and the literature at large, points to a lack of reporting structures, mechanisms and/or incentives for tracking invasion costs, and/or methodological expertise for monetary quantifications (Diagne et al. 2020a; Robertson et al. 2020). Nevertheless, costs in PAs have steadily increased over time, both in terms of magnitude and numbers of reported cost entries. A vast majority of costs in PAs emanate from predictions, models or simulations (i.e. "potential" costs in the database) and therefore further work is needed, wherever possible, to capture and report actual invasion costs on the ground.

One potential reason for the apparent methodological bias could be the result of language barriers that have so far precluded the capturing of cost data. While *InvaCost* has compiled cost information in both English and 15 non-English languages (Angulo et al., 2021), countries with language barriers for capturing data still remain, which most likely contributed to the unevenness presented in this study. Ultimately, data collection can result in biased information which can significantly impact outcomes. *InvaCost* is a dynamic database, limited by available and accessible electronic literature, and will grow and be updated over time. Our study only provides a snapshot of a 'living' database, focusing on PAs, and therefore the results need to be interpreted cautiously. We note that data were not available to account for different proportions of IAS with reported costs and their respective abundances among contexts, despite the importance of abundances in impact prediction (Parker et al., 1999; Dick et al., 2017). Nonetheless, we believe that given the most up-to-date information on the monetary cost in PAs, this study stresses that more should be invested in the management of IAS to realise the role of PAs and achieve long term conservation of nature. The collection of PA cost data through the use of tailored surveys (e.g. focused on IAS management costs and challenges) which targets PA managers will be instrumental in this context. This will facilitate the collection of scarce and/or inaccessible information, which ultimately helps to implement concerted and evidence-based recommendations.

Drivers of invasive alien species costs in protected areas

The present study found overall similarities in costs between PAs and non-PAs, with differences conserved within particular continents (i.e. Africa, Temperate Asia and Europe) or environments (i.e. terrestrial). This finding is surprising for several reasons. Firstly, PAs only cover up to 23.6% of the planet's surface (www.protectedplanet.net, October 2021). As such, we expected that much higher investments would be made for IAS outside PAs. Land use and other impacts outside PAs significantly influence species and ecosystems within PAs (Foxcroft et al. 2011; Liu et al. 2020), and therefore managing IAS outside PA networks is necessary to ensure effective conservation within PA networks. Nevertheless, although we matched PA and non-PA costs according to their environment, taxonomic group and continent, this does not mean that these non-PAs and PAs are ecologically connected. In addition, PAs are likely to be subject to reduced anthropogenic activity with less human-made infrastructure or economic activities that can be damaged or degraded by invasions compared to non-PAs. This is either because PAs have been established in relatively pristine areas or because their protection status inherently limits economic activities. Therefore, non-PAs are more likely to incur resource damage or loss costs, whereas PAs are more likely to incur management costs. Moreover, the unmatched costs (hence those not analysed) under our criteria are likely related to damage costs in non-PAs (Haubrock et al., 2021) and management costs within PAs (Rico-Sánchez et al., 2021). Therefore, we expect our trends to correspond with these additional data.

Our results show that the HDI influenced the economic costs of IAS in PAs. This positive effect between the degree of wealth and conservation decisions is likely due to developed countries having the ability to better document damage costs, and most importantly having more means to manage IAS (Nuñez and Pauchard 2010). In addition, for most developing countries, the primary goal of economic growth does not always go hand in hand with conservation goals. Consequently, major environmental problems, such as biological invasions, continue to be a challenge in countries with fragile economies (Early et al. 2016). Further, invaders from terrestrial and semi-aquatic environments have the highest reported costs while aquatic taxa have the lowest. The high costs associated with terrestrial species in this study are driven by both vertebrates (particularly mammals) and invertebrates (particularly insects). In contrast, high semi-aquatic costs are most likely driven by mosquitoes (which go through an aquatic life stage) and this can lead to high health costs. The low costs for aquatic species are likely due to the cryptic nature of submerged environments and the respective difficulty in managing their invasive populations (e.g. aquatic macrophytes), compounded by research bias towards terrestrial systems (Cuthbert et al. 2021).

Overcoming management challenges in protected areas

The control and eradication of IAS in PAs are time- and resource-consuming, and prioritization schemes are necessary in light of limited available budgets for conservation (Ziller et al. 2020). Information on the costs of IAS can serve as valuable input to priority-setting schemes aimed at managing biological invasions in PAs. This becomes even more important given that most countries have a limited capacity to effectively respond to invasions (McCarthy et al. 2012; Early et al. 2016; Faulkner et al. 2020), and decisions about resource allocation for biosecurity, control and post-invasion management are thus often made on an ad hoc basis (Epanchin-Niell 2017; Liebhold and Kean, 2019; Kourantidou and Kaiser 2021). Knowledge of IAS economic costs is key to help PA managers invest in efforts that optimize large scale positive results at the lowest possible cost (Gallardo and Aldridge 2013). We observed that 73% of incurred IAS costs in PAs were management costs allocated by governmental agencies for the management of IAS or environmental impact costs.

Lower expenditure on pre-invasion than post-invasion management suggests that management strategies are more reactive than proactive and indicates that management costs are much higher than prevention costs. This pattern may extend beyond PAs, as this type of reactive management has also been noted in non-PAs of Central and South America (Heringer et al. 2021). Further, the dominance of post-invasion management expenditure points to the need for more preventative measures, such as biosecurity, to curtail the increased expenses associated with post-invasion management (Leung et al. 2002; Ahmed et al. 2021). Given that preserving biodiversity is one of the main goals of PAs (although in some cases goals are combined with others depending on the management category of the PA; see https://www.iucn.org/theme/protected-areas/about/protected-area-categories), pre-invasion management is seen as essential to avoiding the myriad impacts of IAS on native species and ecosystems.

Perspectives on underestimated costs

While biological invasions continue to increase, the efficacy of PAs in conserving biodiversity remains limited (Rodrigues et al. 2004; Liu et al. 2020). Generally, the effectiveness of management can differ markedly across PAs, with just 22% of PAs recognized as having "sound management" (Leverington et al. 2010). Ineffective management of these so-called "paper parks" (i.e. parks in name only which provide little or no protection) mainly stems from a lack of investment. This is the case of many PAs from developing countries due to chronic financial deficits (James et al. 1999; Wilkie et al. 2001; Gill et al. 2017; Lindsey et al. 2018). Balmford et al. (2002) suggested yearly investments of approximately \$45 billion (over 30 years) to efficiently maintain an expanded network of tropical PAs covering 15% of terrestrial and 30% of marine ecosystems; a study by McCarthy et al. (2012) put forward that \$76 billion per year is needed to conserve terrestrial PAs globally; Balmford et al. (2004) proposed that it would cost \$5–19 billion per year to conserve 20–30% of marine ecosystems globally; and finally, Lindsey et al. (2018) suggested that \$1–2 billion per year are required to conserve African PAs with lions. PAs in developing countries generally receive significantly less funding than that required for basic conservation management (James et al. 1999; Bruner et al. 2004). Our results reveal that although PAs in continents such as Africa and South America reported higher costs, they do not equivalently invest in IAS management. However, since limited cost data was available for PAs (~0.3% of PAs), this suggests massive underestimation of economic costs in PAs globally. Further, the taxa highlighted in this paper, as well as the number of reported entries, indicate that current reporting of IAS costs in PAs is greatly underestimated.

PAs serve as the backbone of global conservation and biological invasions are a key driver of change in PAs (Foxcroft et al. 2017a; Shackleton et al. 2020a, b). Our study shows that many IAS have already caused significant management and damage costs across all environments and continents. If not managed resourcefully, their impacts can only be expected to intensify, thus lowering the biodiversity preservation goal of PAs. Globally, the number of IAS are expected to increase as more species are introduced via global trade and more invasions are discovered as a result of invasion debt (Essl et al. 2011; Seebens et al. 2017, 2020). As such, we strongly encourage comprehensive economic estimations and reporting of IAS costs across PAs in order to improve invasive species management. This will provide an opportunity to maximize return on conservation investments which will have a significant impact on biodiversity outcomes in PAs.

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Availability of data and material All data used are available in the Supplementary Material (ESM_3).

Code availability Not available.

Declarations

Conflict of interest The authors declare that there is no conflict of interest.

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Consent for publication All authors have read and approved the submitted version of the manuscript.

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Supplementary information of the article

Surprisingly high economic costs of biological invasions in protected areas

by Moodley, Angulo, et al. 2022

Biological Invasions

List of supplementary information material:

ESM_1. **Cumulative temporal trends** showing the number of papers retrieved from the **English literature search on invasions** registered in the Web of Science between 1970 and 2019. These papers address alien, naturalized and invasive species generally (i.e. red line; n=58,729); alien, naturalized and invasive species occurring in protected areas (i.e. blue bars; n=7,403); and alien, naturalized and invasive species occurring in protected areas that mention economic cost(s) (i.e. black bars; n=923).

ESM_2. **Descriptors used to characterize the costs of invasive alien species**: (a) cost descriptors and (b) protected areas descriptors

ESM_3. **Datasets**. Delineation of cost and protected area descriptors, as well as, all the three subsets used for this study. (XLSX 2031 kb, visit the article webpage https://link.springer.com/article/10.1007/s10530-022-02732-7#Sec25)

ESM_4. **Correlation matrix** of descriptors analyzed in the Combined Subset

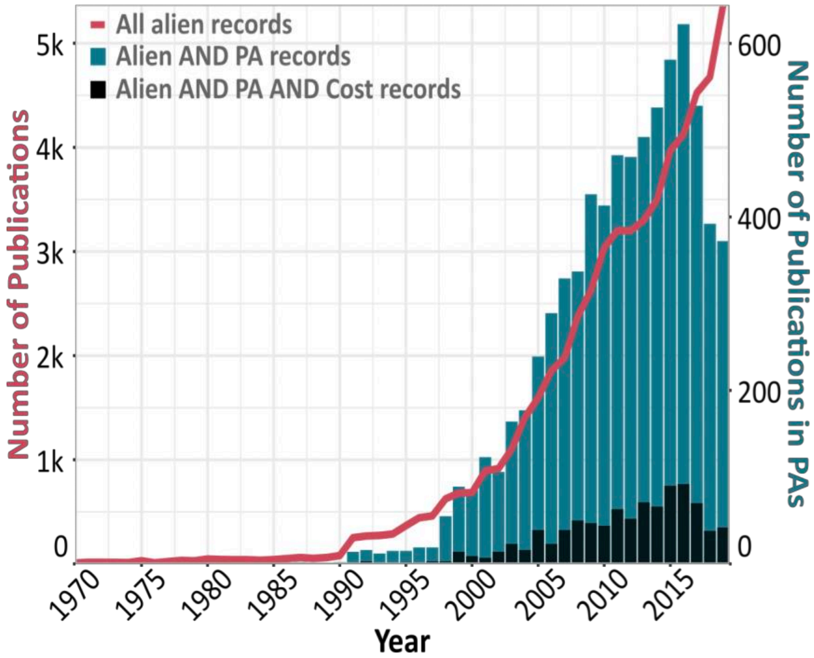
ESM_5. **Correlation matrix** of descriptors analyzed in the Protected Area WDPA Subset

ESM_6. **Temporal dynamics between 1975 and 2020.**

ESM_7. **(a) Effects of cost descriptors and protected area status on the mean economic costs** (log10-transformed) incurred by invasive alien species. Variables in bold indicate significant effects. Column legends: degrees of freedom (Df), sum of squares (SS), residual sum of squares (RSS), F-ratios (F-value) and Prob > F (p-value). **(b) Summary statistics** of each descriptor according to land protection status (0 = non-protected area; 1 = protected area). N = total number of entries considered for each level, SD = Standard deviation, SE = Standard error, and CI = confidence intervals.

ESM_8. **(a) Effects of cost and PA descriptors on the mean observed economic costs (log10-transformed) across protected areas.** Significant variables are highlighted in bold. Column legends: degrees of freedom (Df), sum of squares (SS), residual sum of squares (RSS), F-ratios (F-value) and Prob > F (p-value). **(b) Summary statistics** of each descriptor. Note that statistics for continuous variables are excluded (i.e. year of designation, reported area and the human development index). N = total number of entries considered for each level, SD=Standard deviation, SE = Standard error, and CI = confidence intervals

ESM_1. Cumulative temporal trends showing the number of papers retrieved from the English literature search on invasions registered in the Web of Science between 1970 and 2019. These papers address alien, naturalized and invasive species generally (i.e. red line; n=58,729); alien, naturalized and invasive species occurring in protected areas (i.e. blue bars; n=7,403); and alien, naturalized and invasive species occurring in protected areas that mention economic cost(s) (i.e. black bars; n=923).



Time span of search: 1900-2021

Refined to search the following **fields**: title, abstract, keywords

Search strings used to identify: alien, naturalized and invasive species literature (a); alien, naturalized and invasive species literature specifically in protected areas (b); and alien, naturalized and invasive species literature specifically in protected areas which report economic costs (c).
(a) exotic OR alien OR "non-native" OR naturalized OR naturalised OR invasive OR invader
(b) exotic OR alien OR non-native OR naturalized OR naturalised OR invasive OR invader AND "protected area" OR "national park" OR reserve
(c) exotic OR alien OR non-native OR naturalized OR naturalised OR invasive OR invader AND "protected area" OR "national park" OR Reserve AND economic OR economy OR cost OR monetary

Subject areas included in the search: Ecology; Plant Sciences; Biodiversity Conservation; Environmental sciences; Marine Freshwater Biology; Entomology; Zoology; Agronomy; Forestry; Fisheries, Biology; Evolutionary Biology; Water Resources; Horticulture; Agriculture Multidisciplinary; Parasitology; Geography Physical; Environmental Studies; Soil Science; Humanities Multidisciplinary; Economics; Remote Sensing; Urban Studies; Infectious Diseases; Public Environmental Occupational Health

ESM_2. **Descriptors used to characterize the costs of invasive alien species**: (a) cost descriptors and (b) protected areas descriptors (see Supplementary Material 3 for the interpretation of the descriptive fields).

	Categories	***Combined Subset***	***WPDA Subset***
(a) Cost descriptors			
Continent	Africa	×	×
	Antarctica		×
	Asia-Temperate	×	×
	Australasia	×	×
	Europe	×	×
	Northern America	×	×
	Southern America	×	×
	Pacific	×	×
Taxonomic group	Invertebrate	×	×
	Vertebrate	×	×
	Plant	×	×
	Algae	×	
	Diverse/unspecified		×
Environment	Aquatic	×	×
	Semi-aquatic	×	×
	Terrestrial	×	×
	Diverse/unspecified	×	×
Impacted sector	Agriculture	×	
	Authorities-Stakeholders	×	
	Environment	×	
	Health	×	
	Mixed/Unspecified	×	
Type of cost	Pre-invasion management	×	×
	Post-invasion management	×	×
	Knowledge & Funding	×	×
	Mixed management	×	×
	Mixed management & damage	×	×
	Damage	×	×
(b) Protected area descriptors			
Designation	National Park		×
	Park		×
	Terrestrial Reserve		×
	Marine Protected Area/Wetland		×
	Areas of Special Conservation		×
	Other Terrestrial Protected Areas		×
	Multiple		×
Designation year	Year reported in WDPA (1889-2019)		×
Reported area (Sq. km)	Area reported in WDPA (0.05-25,468)		×
Human Development Index (HDI)	HDI Data 1990-2019 (0.528-0.957) (http://www.hdr.undp.org/en/data, accessed January 2021)		×

Categories are added for each descriptor and a description or the range of continuous variables are given in parenthesis.

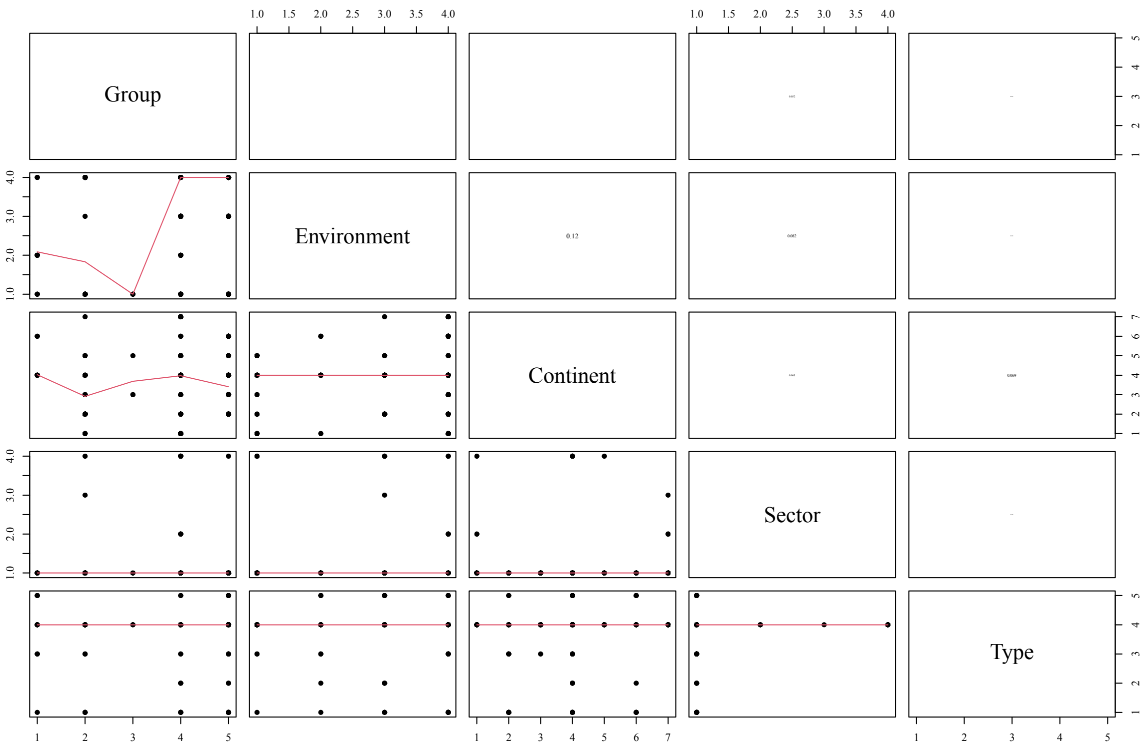
Descriptors used in the analyses of protected and non-protected areas (i.e. Combined Subset) and across protected areas (i.e. Protected Area WDPA Subset) are marked with crosses.

Note that two categories could not be matched in the Combined Subset (Antarctica and Diverse / unspecified taxa) and one in the Protected Area WDPA Subset (Algae), while the "economic sector" was not used in the WDPA Subset due to the unbalanced sample size among categories.

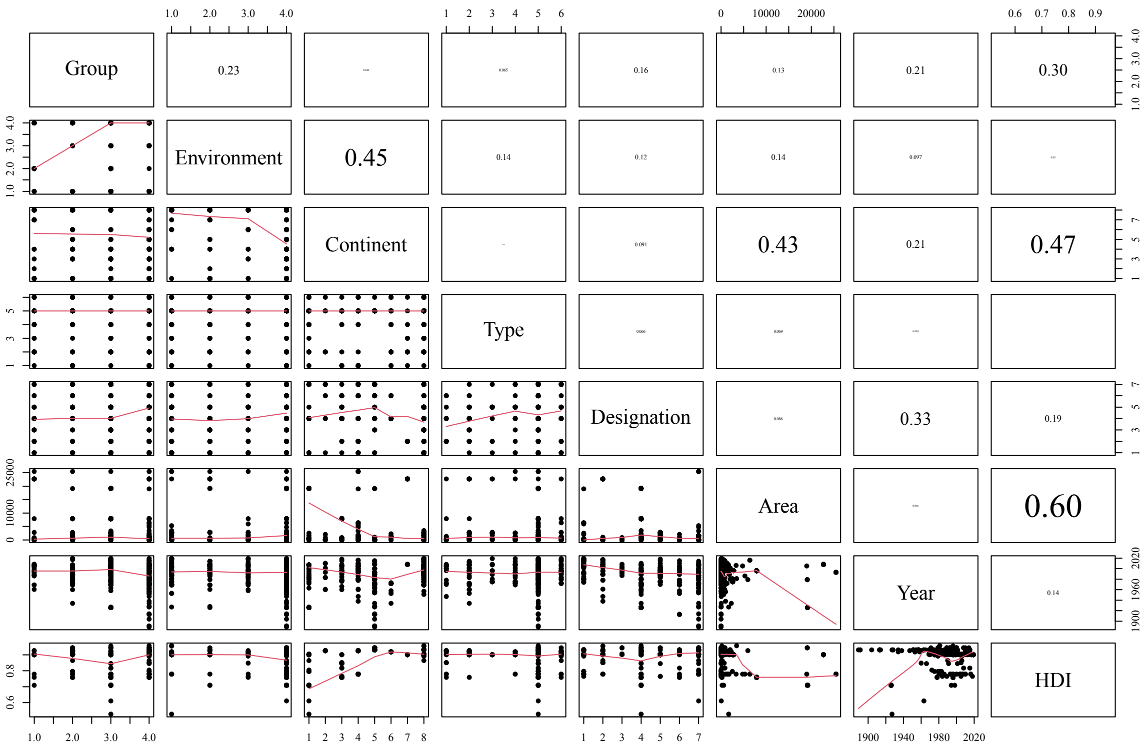
Sample sizes were 1,125 for the Combined Subset and 564 for the Protected Area WDPA Subset.

For designation year and reported area, the mean was used for multiple entries.

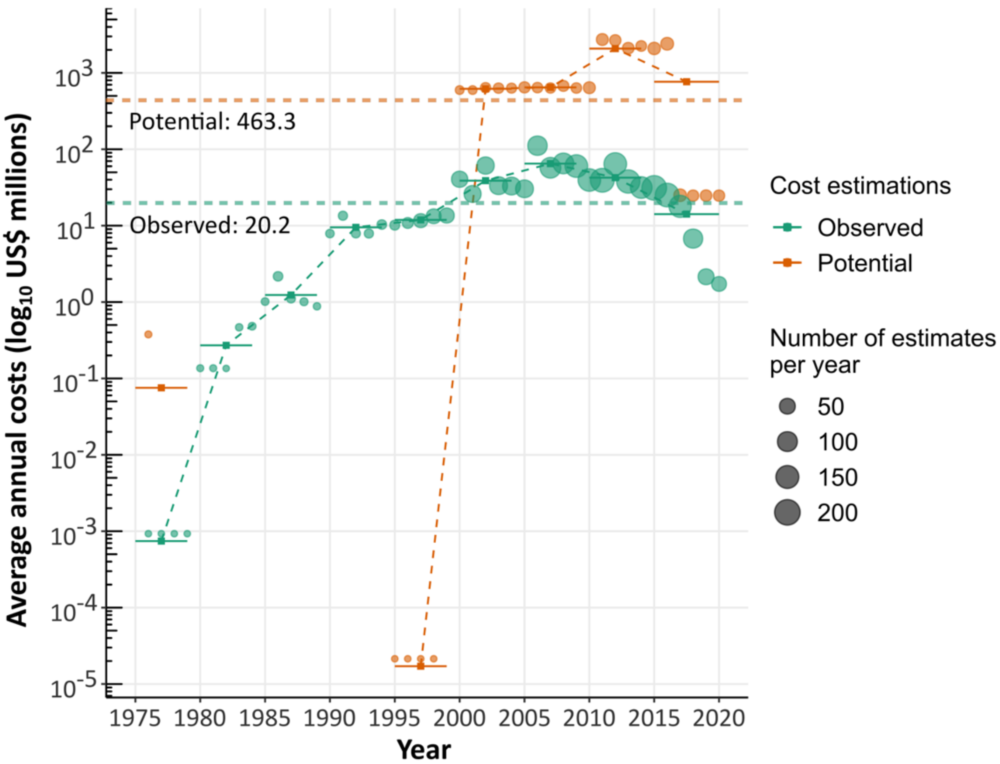
ESM_4. **Correlation matrix** of descriptors analyzed in the Combined Subset



ESM_5. **Correlation matrix** of descriptors analyzed in the Protected Area WDPA Subset



ESM_6. **Temporal dynamics between 1975 and 2020.**



ESM_6. Temporal dynamics between 1975 and 2020. The horizontal dotted lines represent annual averages for observed and potential costs over the entire period, solid bars represent 5-year means, and filled circles represent annual total costs which are scaled by size to match the number of entries.

ESM_7. **(a)** Effects of cost descriptors and protected area status on the mean economic costs (log_{10}-transformed) incurred by invasive alien species. Variables in bold indicate significant effects. Column legends: degrees of freedom (Df), sum of squares (SS), residual sum of squares (RSS), F-ratios (F-value) and Prob > F (p-value). **(b)** Summary statistics of each descriptor according to land protection status (0=non-protected area; 1=protected area). N=total number of entries considered for each level, SD=Standard deviation, SE= Standard error, and CI=confidence intervals.

(a)

Variable	Df	SS	RSS	F-value	*p-value*
PA status	1	3.350	1341.000	2.715	0.100
PA status×Taxonomic group	4	8.960	1346.700	1.815	0.124
PA status×Environment	**3**	**25.470**	**1363.200**	**6.875**	**<0.001**
PA status×Continent	**6**	**81.710**	**1419.400**	**11.025**	**<0.001**
PA status×Impacted sector	3	0.860	1338.500	0.231	0.875
PA status×Type of cost	4	2.350	1340.000	0.475	0.754

(b)

Taxonomic group							
Status	**Levels**	**N**	**Mean**	**SD**	**Median**	**SE**	**CI**
0	Diverse/unspecified	11	49323.69	116914.50	2743.34	35251.05	78544.23
0	Invertebrate	75	617091.98	2011849.00	80238.12	232308.31	462884.41
0	Other	3	961614.80	651660.80	1215921.21	376236.57	1618815.28
0	Plant	410	549808.47	3835570.80	1968.02	189425.34	372368.75
0	Vertebrate	95	221752.65	756381.20	9387.57	77603.08	154082.73
1	Diverse/unspecified	21	129750.01	455261.30	12711.07	99346.15	207232.44
1	Invertebrate	29	108876.59	184739.30	14211.85	34305.24	70271.09
1	Other	2	757619.44	1058618.00	757619.45	748555.93	9511304.96
1	Plant	298	39962.73	105620.40	9102.85	6118.43	12040.96
1	Vertebrate	181	307913.70	854791.00	28873.60	63536.12	125371.42

ESM_8. (a) Effects of cost and PA descriptors on the mean observed economic costs (log_{10}-transformed) across protected areas. Significant variables are highlighted in bold. Column legends: degrees of freedom (Df), sum of squares (SS), residual sum of squares (RSS), F-ratios (F-value) and Prob > F (p-value). **(b)** Summary statistics of each descriptor. Note that statistics for continuous variables are excluded (i.e. year of designation, reported area and the human development index). N=total number of entries considered for each level, SD=Standard deviation, SE= Standard error, and CI=confidence intervals.

(a)

Variable	Df	SS	RSS	F-value	*p-value*
Designation	6	3.612	663.500	0.489	0.817
Year of designation	**1**	**10.508**	**670.400**	**8.535**	**0.004**
Reported area	**1**	**6.406**	**666.300**	**5.203**	**0.023**
Human development index	**1**	**8.851**	**668.740**	**7.189**	**0.008**
Environment	**3**	**14.186**	**674.080**	**3.841**	**0.010**
Taxonomic group	**3**	**16.783**	**676.880**	**4.544**	**0.004**
Continent	**7**	**40.384**	**700.28**	**4.686**	**<0.001**
Type of cost	5	8.474	668.370	1.377	0.231

(b)

Designation						
Levels	**N**	**Mean**	**SD**	**Median**	**SE**	**CI**
Areas of Special Conservation	69	57454.13	156961.79	12008.51	18895.98	37706.34
Marine/Wetland	41	271926.60	760510.34	39603.60	118771.76	240046.68
Multiple	32	24827.16	90258.33	7677.38	15955.57	32541.60
National Park	212	901559.11	5646119.50	28154.22	387777.08	764413.57
Park	68	49102.14	125919.53	9580.62	15269.99	30479.02
Private/Other terrestrial PAs	33	412747.56	939234.37	9815.00	163499.72	333038.03
Terrestrial Reserve	109	332835.46	1008780.17	17079.97	96623.62	191524.77