Invasive alien species as simultaneous benefits and burdens: trends, stakeholder perceptions and management

Melina Kourantidou, Phillip Haubrock, Ross Cuthbert, Thomas Bodey, Bernd Lenzner, Rodolphe Gozlan, Martin Nuñez, Jean-Michel A Salles, Christophe Diagne, Franck Courchamp

Invasive alien species as simultaneous benefits and burdens: trends, stakeholder perceptions and management

Melina Kourantidou[1,2], mkour@dal.ca; Phillip J. Haubrock[3,4]; Ross N. Cuthbert[5,6]; Thomas W. Bodey[7,8]; Bernd Lenzner[9]; Rodolphe E. Gozlan[10]; Martin A. Nuñez[11,12]; Jean-Michel Salles[13]; Christophe Diagne[14]; Franck Courchamp[14]

1. Institute of Marine Biological Resources and Inland Waters Hellenic Center for Marine Research 164 52 Athens Greece
2. Department of Sociology, Environmental and Business Economics University of Southern Denmark Degnevej 14 6705 Esbjerg Ø Denmark
3. Department of River Ecology and Conservation Senckenberg Research Institute and Natural History Museum Frankfurt 63571 Gelnhausen Germany
4. South Bohemian Research Center of Aquaculture and Biodiversity of Hydrocenoses, Faculty of Fisheries and Protection of Waters University of South Bohemia in České Budějovice Zátiší 728/II 389 25 Vodňany Czech Republic
5. GEOMAR Helmholtz-Zentrum Für Ozeanforschung Kiel 24105 Kiel Germany
6. School of Biological Sciences Queen's University Belfast Belfast BT9 5DL Northern Ireland UK
7. School of Biological Sciences, King's College University of Aberdeen Aberdeen AB24 3FX UK
8. Penryn, Environment and Sustainability Institute University of Exeter Cornwall TR10 9FE UK
9. Bioinvasions, Global Change, Macroecology Group, Department of Botany and Biodiversity Research University of Vienna Vienna Austria
10. ISEM, CNRS, IRD, EPHE Université de Montpellier Montpellier France
11. Grupo de Ecología de Invasiones, INIBIOMA, CONICET Universidad Nacional del Comahue Quintral 1250 CP 8400 San Carlos de Bariloche Argentina
12. Department of Biology and Biochemistry, University of Houston Houston TX 77204 USA
13. CEE-M, Université de Montpellier, CNRS, INRAE, Institut Agro 34060 Montpellier France
14. CNRS, AgroParisTech, Université Paris-Saclay, Ecologie Systematique Evolution 91405 Orsay France

Received: 2 March 2021 / Accepted: 28 December 2021

Abstract

In addition to being a major threat to biodiversity and ecosystem functioning, biological invasions also have profound impacts on economies and human wellbeing. However, the threats posed by invasive species often do not receive adequate attention and lack targeted management. In part, this may result from different or even ambivalent perceptions of invasive species which have a dual effect for stakeholders -- being simultaneously a benefit and a burden. For these species, literature that synthesizes best practice is very limited, and analyses providing a comprehensive understanding of their economics are generally lacking. This has resulted in a critical gap in our understanding of the underlying trade-offs surrounding management efforts and approaches. Here, we explore qualitative trends in the literature for invasive species with dual effects, drawing from both the recently compiled InvaCost database and international case studies. The few invasive species with dual roles in InvaCost provide evidence for a temporal increase in reporting of costs, but with benefits relatively sporadically reported alongside costs. We discuss methods, management, assessment and policy frameworks dedicated to these species, along with lessons learned, complexities and persisting knowledge gaps. Our analysis points at the need to enhance scientific understanding of those species through inter- and cross-disciplinary efforts that can help advance their management.

Keywords

Double-edge invasive alien species; Benefits; Costs; InvaCost; Management; Trade-offs; Policies; Conflict

Supplementary Information

The online version contains supplementary material available at https://doi.org/10.1007/s10530-021-02727-w.

Introduction

Invasive alien species (IAS) are a well-recognized threat to biodiversity, ecosystem functioning, human well-being and livelihoods around the world (Paini et al. 2016; Walsh et al. 2016; IPBES 2019). In recent years, much effort has been levelled at investigating the negative impacts of IAS (Bellard et al. 2016; Bradshaw et al. 2016; Crystal-Ornelas and Lockwood 2020; Diagne et al. 2020b). This literature, both theoretical and empirical, spans multiple disciplines, from applied ecology to evolutionary biology, economics and human health, and has advanced our understanding of different dimensions of IAS impacts and management (Keller et al. 2009; Britton et al. 2011; Epanchin-Niell 2017; Mazza and Tricarico 2018). Despite this, public awareness, as well as policy and legislation efforts towards IAS are insufficiently developed compared to other biodiversity threats, such as climate change (Courchamp et al. 2017). Part of this inconsistency is likely due to a common confusion between non-native (or alien, i.e. not necessarily problematic) and invasive (i.e., problematic) species. Indeed, there is also a dichotomy between stage-based definitions (i.e. independent of impact; Blackburn et al. 2011) and those that require negative impact for IAS, with the latter more often employed by authorities or practitioners. There is also debate surrounding so-called "neo-native" species which expand their range naturally as a result of anthropogenic climate change, and could be considered differently by managers (Essl et al. 2019; Wilson 2020). Adding to the confusion, some IAS with a known ecological or economic impact may simultaneously bring ecological or economic benefits, although not necessarily on the same aspects, or to the same people, spatial or temporal scales (Braysher 2017; Beever et al. 2019). This ambivalence concerning species that can be viewed alternatively for their costs as burdens, or for their benefits as assets, is burdensome for management and communication. Relatively little research effort, however, has been dedicated to exploring cases of IAS with potential or realized benefits (but see for example Gozlan 2008; Gozlan and Newton 2009; Beever et al. 2019; Oficialdegui et al. 2020; Vimercati et al. 2020), at least compared to IAS with negative impacts only. As a result, much of the coverage on IAS with multiple, and potentially beneficial roles, remains incomplete, resulting in a limited literature base, especially within resource economics and management, that offers targeted tools and approaches to address their management.

IAS with potential or realized benefits are referred to with different names depending on their perceived impact or assessed risk, prior knowledge on the species and the type of conflict they create (Gozlan et al. 2013). These species have been referred to as "mixed-blessing" invaders, "multiple-use" resources, "conflict species", species with "ambivalent human preferences", and species that have a "dual-role" or that are both a "value and a nuisance" (Zivin et al. 2000; Vigliano and Alonso 2007; Courtois et al. 2012; Han 2016; Hui and Richardson 2017; van Wilgen and Wilson 2018; Kourantidou and Kaiser 2019a; Skonhoft and Kourantidou 2021). These terms imply that their impact, either economic, ecological or other (for example cultural), may be seen through both a positive and a negative lens. Gozlan (2015) describes this as the "Janus syndrome". Hereafter, we refer to these species collectively as Double-Edge IAS (DE-IAS), while recognizing the complex and diverse dimensions behind each case, including those for which impacts are intangible. However, this paper focuses on those IAS with documented economic costs that also carry ecological or economic benefits.

Biological invasions are also variously defined in the literature, with common terms used including "alien", "exotic", "non-native", "non-indigenous", "introduced" (Falk-Petersen et al. 2006). Here we refer to IAS as those alien species that have an undesirable or negative effect on biodiversity, ecosystems or human livelihood anywhere in their novel range and continue to expand. However, we note that stage-based definitions of IAS do not require impact to occur (Blackburn et al. 2011) and that invasiveness and impact are not related (Ricciardi and Cohen 2007). Nevertheless, given a focus on economic costs, our analysis considers impact at any stage along the invasion process. Additionally, although "invasive species" is the most commonly used term, ecologists tend to agree that "invasive population" more accurately describes individuals outside their native range that cause negative impacts. The undesirable and desirable effects of DE-IAS may arise in the same population, or in different populations of species that do not overlap spatially, so that benefits and costs may not overlap either (see also Beever et al. 2019 for a series of factors spatially differentiating positive and negative effects). The population density and invasion stage are also critical determinants. For example, at early stages of an invasion, or at low population densities, costs or benefits are likely limited or may not yet be apparent and the cost–benefit ratio may change significantly once spatial spread or density increases (e.g. Shackleton et al. 2007; Wise et al. 2012; van Wilgen and Richardson 2014; Fleming et al. 2017; Ahmed et al. 2021). This may also include, for example, cultural connections to the IAS which at an early stage of the invasion likely have yet to be realized (Gaertner et al. 2016). Market conditions affect perceptions on how the species is viewed as well (see for example in Braysher (2017), where feral pigs are viewed by banana growers as a nuisance when banana prices are high, but valued when prices are low since they clean up excess fallen fruit that can harbour disease to the crop). Other factors, particularly the nature and intensity of human settlement, along with property rights of an area under invasion, also influence species' classification and the consideration of costs and/or benefits. For example, private landowners may promote the benefits of an invasion appealing to tourists, while resource managers of a nearby protected area may view the species exclusively as a burden (Shackleton et al. 2019b). Similarly, departments within the same governing body might have opposing views to invasions, for example with forestry/fishery departments placing economic value on invasive trees / introduction of new fish species, but conservation departments voicing concern (Mack et al. 2000; Forseth and Innis 2004; Virtue et al. 2004; Raghu et al. 2006). The aesthetic appeal or sentience of an IAS is another strong determinant of classification as a DE-IAS. For example, people may see benefits in the presence of some invasive but charismatic animal, or can favour a colorful, yet invasive ornamental plant (Dickie et al. 2014; Jarić et al. 2020). Previous use and status of a landscape may also factor into considering an invasion as a DE-IAS, with an invasive tree species in a treeless grassland seen as a promising resource (Ngorima and Shackleton 2019).

Benefits of DE-IAS may include market and non-market uses, with the former being easier to identify. Species may become DE-IAS over time, such as when their direct benefits for the economy are realized, or economic activity patterns change. Similarly, the economic benefits of a species might degrade with time as societies change, reducing beneficial aspects and potentially driving a shift towards net negative effects. Indeed, often economic activities can mediate the spatio-temporal patterns of the invasion process and contemporary invasions can reflect legacies of economic activities (i.e. "invasion debts" following Essl et al. 2011). For alien plants in particular, economic use has been key to invasion success, which is more prominent in ornamental species and those used as animal feed (van Kleunen et al. 2020). In addition, because benefits can be "value-laden" (i.e. linked to ethical and societal values affecting human well-being), some negative ecological impacts can be seen by the public as benefits (Vimercati et al. 2020). Thus, it becomes clear that, depending on the relative economic importance of the anthropogenic activities affected (see for example Gozlan 2017), costs and/or benefits may be assigned with different weights, determining whether the population will be seen as an IAS or as a DE-IAS.

Non-market uses may overlap with marketable gains which has been illustrated in a socio-ecological framework of ecosystem services and disservices (Vaz et al. 2017). One example is the provisioning of recreational benefits as seen with exotic salmonids (for example Salmo trutta, Salmo salar, Oncorhynchus mykiss), species that like many others, were introduced widely for the development of sport fisheries, but also for aquaculture (Soto et al. 2001; Quist and Hubert 2004; Garcia De Leaniz et al. 2010; Woodford et al. 2016 and references therein). However, benefits can also be less tangible, and reflected for example through cultural values or the provision of hard-to-quantify ecosystem benefits. For example, in Colombia the invasive population of hippopotamuses (Hippopotamus amphibius), coming from the private zoo of famous drug cartel leader Pablo Escobar, is still growing and remains unregulated because of the charismatic nature of the species and its colorful historical context, which provide value in the public's eye (Jarić et al., 2020). Schlaepfer et al. (2011) and Sladonja et al. (2018) detail multiple examples of IAS that serve an important ecosystem role, including provision of habitat, shelter or food for native species, catalysts for restoration or substitutes for extinct taxa.

Figure 1 provides an illustration of the different categories of costs and benefits that DE-IAS may entail for some of the species discussed as examples in this paper.

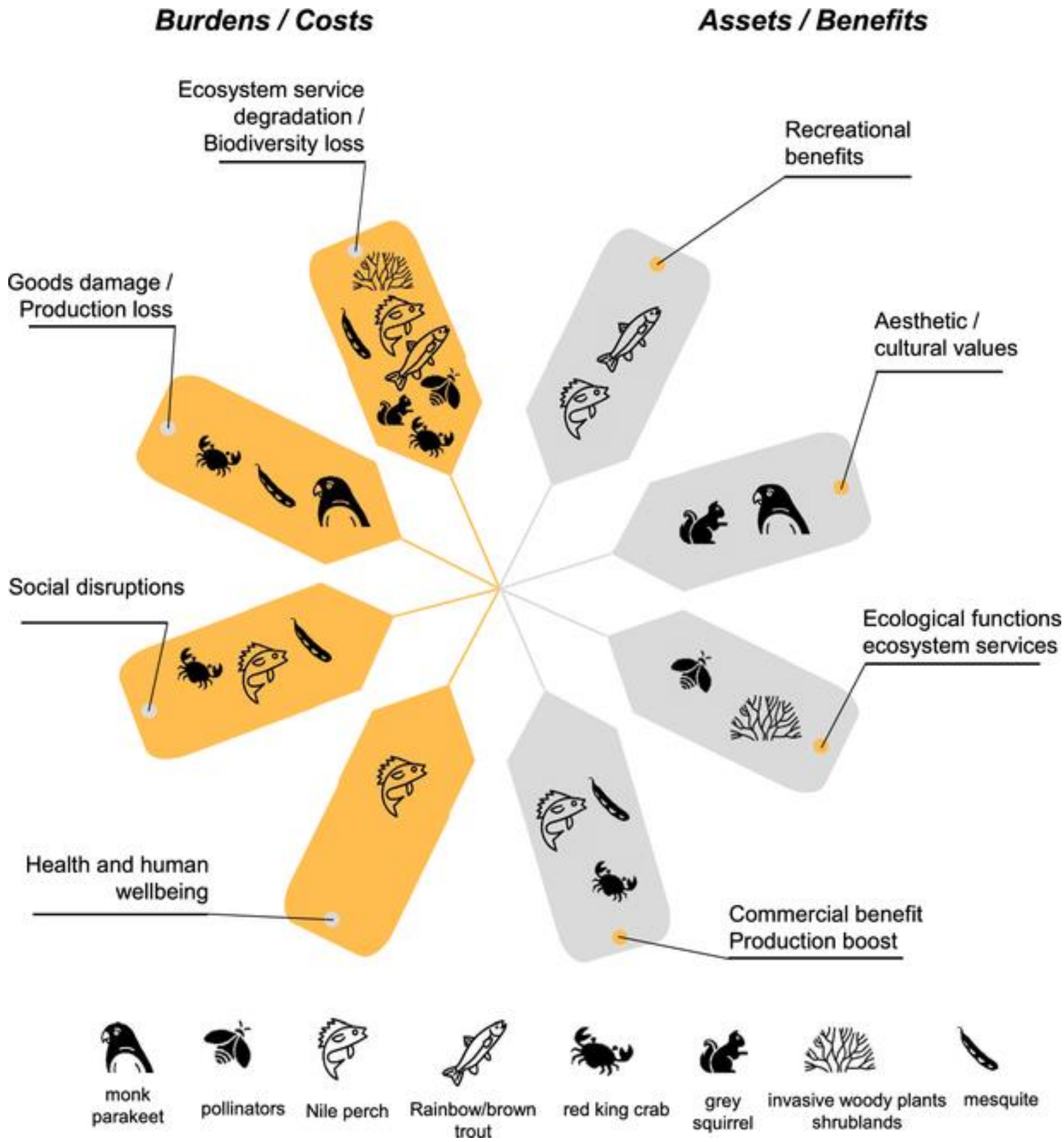

Fig. 1. Schematic representation of prominent examples of DE-IAS along with their costs and benefits. Orange labels represent types of costs while grey labels represent types of benefits and icons within represent examples of DE-IAS for those types. The connection between the labels illustrates the possibility for one single species to have simultaneously several types of costs and/or

> benefits, as is the case for the Nile perch in Lake Victoria. The Nile perch has boosted gains from fisheries (commercial and recreational) but has (among many other ecological impacts) led insectivorous cyclids to extinction, causing outburst of viruses vectored by mosquitoes and more broadly disrupting human livelihoods in different ways. The monk parakeet, for example, is appreciated by people for its charisma but affects the yield of orchards and vineyards. The same applies to the charismatic grey squirrel which is negatively affecting native red squirrels, woodland and birds. The mesquite (*prosopis*) produces fodder and shade for livestock, provides firewood and timber but at the same time negatively impacts native biodiversity, ecosystem services and human livelihoods. Invasive plants may generally be a threat to biodiversity and ecosystem services but some affect avian nesting success. Invasive pollinators displace native pollinators and yet can remain the only remaining pollinators for some native plants. Trout species may support recreational and commercial fishing but bring along several ecological impacts. Last, the red king crab has negatively impacted cod fishers but evolved into a profitable fishery changing local community dynamics and causing degradation of the local ecosystem

As a result of these dual impacts and perceptions within a socio-ecological context, the management of DE-IAS is often highly contested among stakeholder groups with conflicting incentives to either maintain long-term sustainable populations of DE-IAS, or to control and minimize their spread and impact (Novoa et al. 2018). Understanding the economic trade-offs behind DE-IAS is therefore key to their management, as it can help to better understand differences in value systems and socio-ecological mismatches, anticipate potential opposition and reconcile contrasting views, identify avenues to maximize social wellbeing and better manage stakeholder conflicts (Kumschick et al. 2012; Estévez et al. 2015; Beever et al. 2019; Latombe et al. 2020). Furthermore, in the context of limited budgets, an improved understanding of such trade-offs, magnitude and significance of ecosystem services and disservices, can help decision makers and conservation planners to prioritize and optimally allocate resources for management of different species as well as resources for research on different aspects of a single species (for example its potential as a commercial resource vs. its impacts as an invader) (Epanchin-Niell and Hastings 2010; Vaz et al. 2017; Kourantidou and Kaiser 2021).

Our goal here is to explore the challenges of managing DE-IAS through an economic lens and discuss lessons learned from prominent cases of such species. In recognition of the many facets of DE-IAS, we bring forward social and ecological dimensions which are necessary to convey the various forms of positive and negative impacts. We describe the potential of a recent and comprehensive database on the costs of IAS, which was developed under the InvaCost project (Diagne et al. 2020a), to offer novel insights on trends from the literature on these species. Because the InvaCost database does not include detailed information on benefits, we refrain from making quantitative comparisons between costs and benefits. We focus instead on the value of comparing the presence of benefits and costs of DE-IAS reported in InvaCost by (1) types of sources, (2) geographic regions, (3) impacted activity sectors, (4) cost types and (5) taxonomic groups. Using prominent examples of DE-IAS, we illustrate conflicts relating to their presence and management and then turn a critical eye on policies and perceptions on their management and highlight the effects of knowledge gaps and biases on these approaches. Our work responds to an ongoing gap in managing DE-IAS and, by extension, addressing stakeholder conflicts that arise from these species. Through our analysis, lessons learned from selected examples and relevant frameworks, we seek to inform policy recommendations for socially desirable outcomes and highlight critical gaps in assessment and management frameworks.

Double-edge invasive alien species: a qualitative description

Trends from InvaCost

The InvaCost database that was used to identify trends in the literature for DE-IAS through time (Diagne et al. 2020b), documents monetary costs of IAS reported globally during 1960–

2020. InvaCost was constructed using systematic searches of cost information from a wide variety of source documents (for example peer-reviewed articles, books, reports, fact sheets, conference materials) relying on both standardised search strings conducted in different online platforms (Web of Science, Google Scholar and Google search engine) and opportunistic targeted searches (for example expert consultations, contacts with regional national experts). The most up-to-date version of the complete living database (currently InvaCost_4.0) --along with details and information -- is fully accessible at https://doi.org/10.6084/m9.figshare.12668570). The range of descriptors used to describe costs included a binary column that identifies for each recorded reference whether there are associated benefit values (yes when benefit has been reported and no when benefit has not been reported) alongside information on costs. The term “Benefit” refers here to the presence of some profitable/beneficial to socioeconomic well-being activity derived from an IAS.

It is important to note that the InvaCost database search strings did not specifically target DE-IAS, so studies providing exclusively economic benefits were not collected. Despite the focus on costs, an assessment of the benefits reported in InvaCost (now and in the future) is valuable, as it enables us to identify if the dual effects of DE-IAS are generally acknowledged and assessed simultaneously or not, and how this trend has changed over time. What is presented here is the most up-to-date compilation of benefit estimates associated with DE-IAS, available thus far. Considering possible gaps in the compiled studies and errors in the compilation process, the trends and patterns discussed here should be cautiously considered and rather be viewed as an initial snapshot providing a first basis or an avenue for future research improvements given the ‘living’ nature of the InvaCost database (Diagne et al. 2020b; Leroy et al. 2021). These improvements may include, for example, updates based on focused searches targeting benefit estimates documented from IAS recorded in InvaCost or corrections to existing entries.

Considering the likely incompleteness of the benefits data, we only use this analysis to examine differences in trends between IAS and DE-IAS using the number of database entries instead of monetary values. It is important to note though, that, from version 4.0 of the InvaCost database, most of the entries for DE-IAS (89.4%) were from pre-assessed materials (peer-reviewed articles and official reports) or grey material but with documented, repeatable and traceable methods. Additionally, most entries were the result of observed costs (78.2%) rather than potential extrapolations, implying that those costs were actually incurred (Supplementary Material 1, Table 1). We selected only entries where specific single species were recorded ($n = 9226$), with only 191 recording benefits and costs, contrasting the vast majority ($n = 8918$) referring to costs only. In total, 87 unique species were recorded with both costs and benefits, and were thus classified as DE-IAS for the purposes of the analysis below. Taking together all database entries listed for these 87 species gave a total of 2499 entries. The analysis that follows compares this DE-IAS subset (representing approximately 27.1% of the specific species entries) to the IAS with only reported costs.

The number of entries for both DE-IAS and IAS increased over time (Fig. 2). However, the occurrence of benefit reporting for DE-IAS has not shown a clear increasing pattern over time, remaining relatively sporadic. The fewer entries in the last years likely result from lag times in the publication process. The number of publications on DE-IAS and IAS started increasing in the early 1990s and peaked in recent years (~200 and ~800 entries, respectively). The composition of DE-IAS and IAS entries differed significantly among source types (Chi-square = 361.96, $df = 5$, $p < 0.001$). The majority of the published material for DE-IAS was found in either peer-reviewed papers (36.4%) or official reports (37.9%). This trend differed for IAS, with more than half found in official reports (59.7%), with peer-reviewed papers significantly fewer (19.2%) (Fig. 3). This may indicate that studies on species that include benefits are more likely to arise from peer-reviewed research than official reports that have a greater focus on documenting costs alone. This might be the case if one considers that reports tend to be motivated by policy needs aimed at informing management options. In contrast, peer-reviewed research is more diverse, addressing questions related to trade-offs, without necessarily aiming to provide solutions but rather to inform and address knowledge gaps (see also Billé et al. 2012; Laurans et al. 2013; Laurans and Mermet 2014).

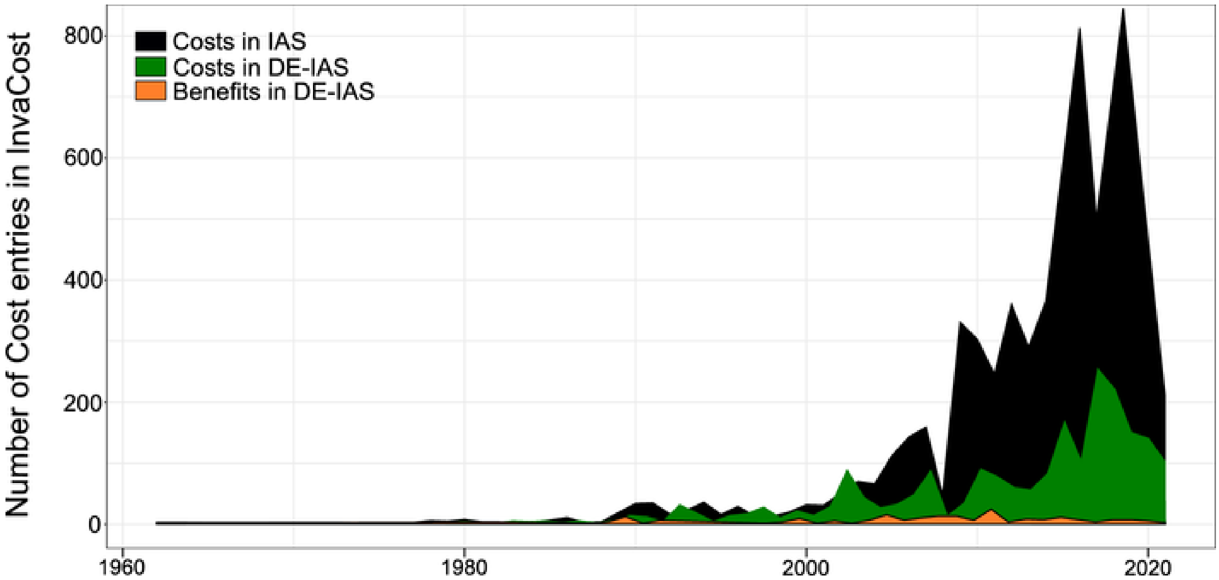

Fig. 2. Recorded database entries over time for DE-IAS and IAS. The green shading highlights cost entries linked to benefits. We note that the zones are not additive, i.e. entries for IAS costs do not include the other categories

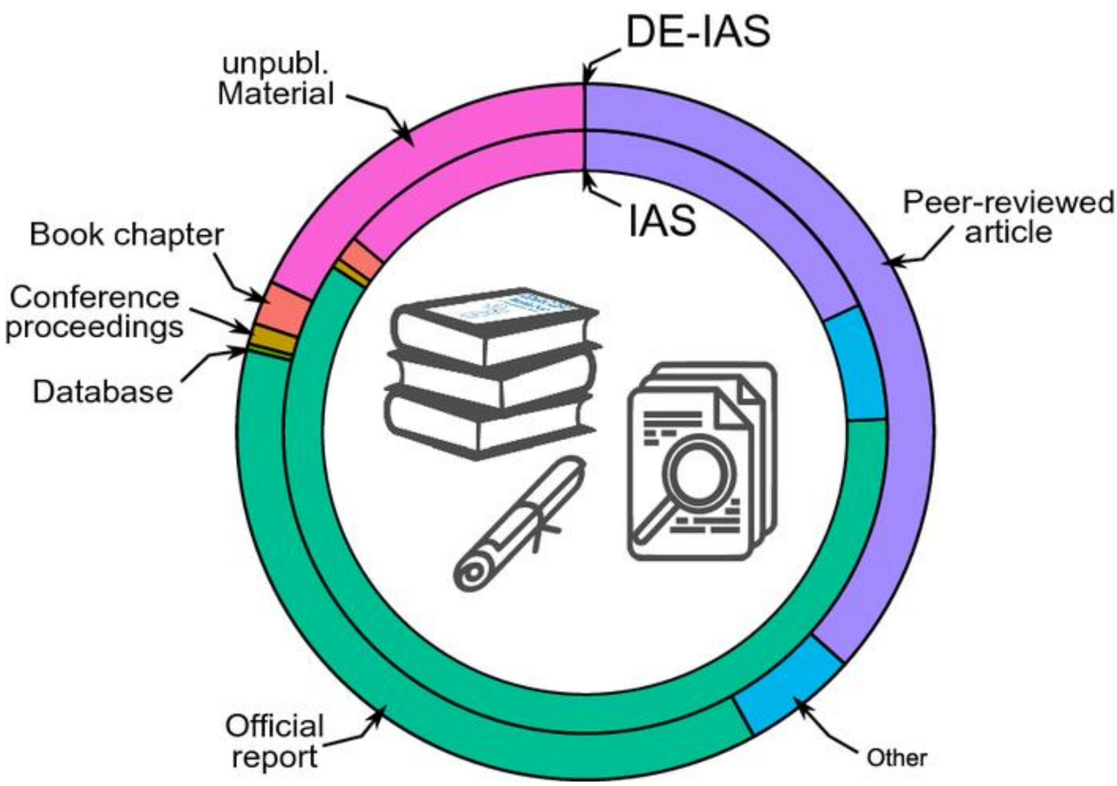

Fig. 3. Sources of publications for IAS (costs) and DE-IAS (benefits). The category "Other" includes: theses, web pages, slide show presentations and fact sheets. Proportions in the figure correspond to the numbers of database entries per group

The costs reported were classified into (a) "Damage" costs for example for damage repair, resource losses and medical care incurred by an invasion, (b) "Management" costs for example for control, monitoring, biosecurity, prevention, management and eradication expenditure and (c) "Mixed" costs including those that could not be easily classified under (a) or (b) (see Epanchin-Niell and Hastings (2010) for an overview of limitations and economic considerations in estimating these types of costs). The proportion of damage and management costs for IAS and DE-IAS is informative and may suggest the ways in which identification of benefits alongside costs alter (or not) approaches to determining an understanding of impacts of these species (see more details in Supplementary Material 1 and Fig. 1 in specific). Similarly, the share of impacts of IAS and DE-IAS on different sectors of the economy can be particularly informative. InvaCost defines sectors of the economy as all the activities, societal or market sectors impacted by the cost and offers a classification of impacts across the following sectors: Agriculture, Authorities -- Stakeholders, Environment, Fisheries, Forestry, Health, Public and social welfare, Unspecified, Mixed (see more details Supplementary Material 1, Fig. 1). Last, an improved understanding of how costs for these species are distributed across space can help inform regional-scale interventions, and be of particular value to cooperative management initiatives in regions that are interconnected biogeographically. Supplementary Material 1 Fig. 2 and 3 offers such a preliminary analysis based on the existing data for cost entries of IAS and DE-IAS across continental regions and habitat types (aquatic, terrestrial, semi-aquatic).

Double-edge invasive alien species as source of conflict

DE-IAS can be a significant source of conflict, especially in cases where the interests of different stakeholders may be at odds. In what follows, we bring forward a few prominent examples of DE-IAS that have caused deep-seated conflicts between user-groups and whose management remains contested.

The Nile perch (*Lates niloticus*) establishment in Lake Victoria (East Africa) provides an infamous example of how an IAS introduction can bring fundamental ecosystem changes and substantially shift regional socio-economic conditions, bringing in a series of different costs and benefits. Nile perch was introduced purposefully, yet unofficially, in the 1950s and 1960s to support recreational and industrial fishing (Hamblyn 1961; Downing et al. 2013). Following its establishment, it transformed local small-scale fisheries into an export-oriented industrial fishery that increased export volumes from \$0.4 million in 1980 to \$66 million in 2003 (Yongo et al. 2005), boosting at the same time processing, marketing and tourism. Notably, the number of livelihoods with a direct or indirect dependence on the lake's fisheries increased from 1.2 to 35 million today (Yongo et al. 2005; Aloo et al. 2017). At the same time, however, Nile perch had dramatic ecological impacts by altering the local fish community composition and the trophic network of the lake (Witte et al. 2013). Of particular concern has been the ecological impact on haplochromine species (Witte et al. 1992). During the second half of the twentieth century, it became the dominant species in fishery catches, replacing many native species and is responsible for the extinction of around 200 native species (via predation), of which several were endemic to the lake. This loss of biodiversity is considered among the largest documented human mediated changes to an ecosystem (Ligtvoet et al. 1991) and the largest modern mass-extinction. As several key functional roles became deficient, biodiversity loss and community compositional changes resulted in cascading effects on habitats in the lake (i.e. water quality) (Kaufman 1992; Mugidde et al. 2005) or outside the lake resulting, for example, in mosquito outbreaks from lack of insectivorous fishes and declines of piscivorous birds from lack of prey. In parallel, an increase in firewood demand to process the fish (Njiru et al. 2005) or deforestation to build more fishing vessels led to further habitat alterations (Yongo et al. 2005). Last, its commercialization brought several other socio-economic disruptions, such as regional increase in HIV/AIDS due to greater fisher mobility (Harris et al. 1995), border conflicts between

Uganda and Kenya over commercial exploitation rights (Harris et al. 1995; Shaka 2013) and declining food security due to commercialization of the fisheries for export markets (Abila 2000; Odongkara et al. 2005). The rainbow trout (Oncorhynchus mykiss) deliberate introduction in South Africa is another example of an invasion for which management is controversial and hindered by industry interests. The trout supports primarily recreational fisheries but at the same time brings severe ecological impacts. With significant economic stakes of angling and aquaculture actors, attempts to recognize the population as an invasion and implement regulation have created controversy and ultimately failed (van Wilgen and Wilson 2018).

The introduction of plant species for the forestry and agricultural sectors can similarly lead to contrasting views considering gains to the industry and employment benefits, but at the same time biodiversity loss and impacts to other industries such as recreation and tourism (Castro-Díez et al. 2019; Brundu et al. 2020). For example, the Pinus genus, an almost exclusively Northern Hemisphere group of species, has been widely planted in the Southern Hemisphere as part of forestry operations for timber and wood pulp, as well as erosion control and aesthetics. With wind dispersal, seeds often 'escape' from deliberate plantings and these wilding conifers have turned into invasive populations across many parts of Australasia, South America and southern Africa (Nuñez et al. 2017). Such trees then create a range of negative impacts, including depletion of soil nutrients and water reserves, increasing fire risk, reduction in landscape values and impacts on native ecosystems and species (Le Maitre et al. 1996; Richardson and Rejmánek 2011; Castro-Díez et al. 2019). Thus, there are substantial negative costs to be mitigated from industries that simultaneously generate significant proportions of nations' GDP. For example, in New Zealand, forest products are the third largest export and in 2018 generated almost NZ$6.4 billion (MPI 2019), dwarfing the NZ$166 million estimated costs for effective control of wilding pines, but not the estimated $5.3 billion loss from taking no action (2018 present value estimated over 50 years) (Wyatt 2018). Given their benefits, control and regulation has been very controversial in places such as in South Africa (van Wilgen and Wilson 2018). Pasture plants also produce an obvious conflict of interest since an ideal species from the point of view of the agricultural sector is one that naturalizes and is self-sustaining in its new environment, growing sufficiently to provide abundant fodder for livestock. Such a description would also likely identify a successful IAS and, for example, the vast majority of useful pasture introductions in Australia are also significant weeds (Lonsdale 1994). Paterson's curse (*Echium plantagineum*) for example, that is poisonous to grazing animals, has triggered a conflict in Australia between the agricultural and the apicultural sectors, with the latter benefitting from nectar and having significantly slowed down attempts of control (Messing 2000).

The economic benefits of such species may often be clear and readily measurable, whereas the negative impacts on biodiversity and indigenous values can be far harder to quantify given the potentially longer timescales involved and the less tangible economic effects often associated with these impacts (Ens et al. 2015; Woodford et al. 2017; Shackleton et al. 2019c; Palmer et al. 2020). In recognition of the limited understanding of biotic interactions triggered by alien species (Goodenough 2010) and the lack of comprehensive and systematic efforts to document benefits across environments and sectors (see for example Gozlan 2015 for aquatic species) as well as across all ecosystem services including cultural ones (Dickie et al. 2014), trade-offs become difficult to pin down further hampering clear management objectives. In recognition of these challenges, cost–benefit analysis tools need to account for externalities, uncertainties, and equity considerations as well as to encompass multiple perspectives along with ecological-economic assessments and allow for participatory decision-making processes (De Wit et al. 2001; Wise et al. 2012).

Understanding and managing DE-IAS

Frameworks for evaluating trade-offs

Risk assessment frameworks, although challenging, are essential for developing policies reconciling benefits from an introduced species and the probability that it brings along negative impacts, i.e. curtailing the invasion while minimizing impacts to trade that allow for net economic benefits to be produced (Keller et al. 2009; Springborn et al. 2011). Considering the costs of limiting trade of species with low risk of becoming invasive, it is possible that risk assessments may harm an economy. This is why for risk assessments to be financially justifiable, a high level of accuracy is required, particularly for distinguishing between species likely and unlikely to become invasive (Keller et al. 2007, 2009). In other words, the financial benefits of a risk assessment policy (sum of benefits from trade minus invasion costs, minus the cost of developing and implementing the risk assessment) must exceed those of policies of allowing importation of all or no species (Keller et al. 2009). Indeed, in most circumstances, risk assessment to taxa and trade result in large financial benefits (Keller et al. 2007; Springborn et al. 2011). Beyond predicting the probability of a traded species becoming invasive, for policy makers to decide on import protocols it is necessary to understand potential positive and negative impacts of the species to be imported. This is especially so when either the negative impacts of a potential IAS may be relatively minor thereby justifying their import for their greater benefits for example as a food source, or when likely benefits are very low so that an import ban may still be deemed optimal (particularly considering uncertainties and surprises) (Keller et al. 2009).

In the absence of robust impact assessment tools, it is difficult to reconcile the contrasting views on the management of DE-IAS. Unlike assessment frameworks for IAS such as EICAT, SEICAT and GISS (as well as the weed risk assessment frameworks identified earlier on) or other tools that identify regime shifts, which have in many ways advanced our understanding of impacts (Blackburn et al. 2014; Gaertner et al. 2014; Nentwig et al. 2016; Hui and Richardson 2017; Bacher et al. 2018; Probert et al. 2020), frameworks for the explicit assessment of DE-IAS are generally missing. Exceptions include a few recent efforts (Kumschick et al. 2012; Martinez-Cillero et al. 2019) and frameworks for stakeholder engagement (Gaertner et al. 2016; Novoa et al. 2018) (but see also policy frameworks, such as RSA (2004), that implicitly encompass such considerations). Similarly, IAS assessment frameworks specific to environmental realms such as the Freshwater Invertebrate Invasiveness Scoring Kit (FI-ISK), or the more generic Aquatic Species Invasiveness Screening Kit (AS-ISK), have been very helpful in identifying impacts, prioritizing according to risks entailed and even predicting the effects of climate change on invasiveness of species from these environment types (Copp et al. 2009, 2016; Tricarico et al. 2010). In particular, AS-ISK permits applications among multiple taxonomic groups (for example, amphibians, plants, freshwater and marine fish and invertebrates) and environmental contexts such as climatic zones or salinity regimes. Again, such frameworks are largely missing for DE-IAS; the aforementioned do not include potential benefits and focus exclusively on biological/ecological parameters to determine impacts rather than costs. Different barriers may be hindering the development of such frameworks. For example, at national levels, an important barrier is the potential for differences in goals and perspectives among different resource management agencies or government departments. This can include wishing to enhance economic outputs through increased trade, versus wishing to reduce the risk of invasions to conserve endangered species (Mack et al. 2000; Forseth and Innis 2004; Virtue et al. 2004; Raghu et al. 2006). Nevertheless, the development of such risk assessment frameworks is a key step towards ensuring efficient management approaches which may help determine whether a species should be allowed in the trade, considering the probabilities of escape, establishment, spread and impact (Lodge et al. 2016). They also allow for encompassing both benefits and harms for a potential introduction, bioeconomic analyses integrating ecological and economic processes, as well as consideration of different management options (Epanchin-Niell and Hastings 2010; Hui and Richardson 2017). However, unlike thorough assessments for the release of new drugs, food safety or infectious diseases for humans and livestock, society is more tolerant to risks of IAS (Lodge et al. 2016). Although with a few wealthy nations implementing different risk assessment tools, for example for imports, it could be argued that benefits are indirectly captured, gaps clearly persist, particularly within management once such species become established.

Among at least nine impact assessment frameworks that have been developed over the last two decades, only a third include beneficial impacts of IAS (Kumschick et al. 2012; Katsanevakis et al. 2016; Martinez-Cillero et al. 2019; see details in Vimercati et al. 2020). These gaps limit our understanding of the role of DE-IAS, and hamper a robust evaluation of underlying trade-offs. The few frameworks available for the assessment of beneficial impacts are less frequently used by practitioners or cited by academics (Vimercati et al. 2020), despite the clear evidence that species often have both deleterious impacts and beneficial effects reported (e.g. Katsanevakis et al. 2014). Interestingly, Vimercati et al. (2020) attribute this lack of attention to beneficial aspects of IAS in assessment frameworks to a conscious choice of scholars to exclude benefits from analyses, rather than to denying their existence (see also Gozlan 2008).

Vimercati et al. (2020) also highlight as a major challenge for comprehensive assessments of IAS impacts the distinction between types of benefits, and their relevance to basic and applied science. The key distinction is between values that change in a quantifiable manner and those associated with societal or ethical considerations, such as those affecting human well-being. Additionally, frameworks that formalize and quantify beneficial impacts might be perceived as less urgent compared to those with pernicious effects, and thus might not convey the urgency for action. In addition to the intrinsic difficulties in assessing values -- either positive or negative -- the stakeholders benefiting from certain IAS might not have strong incentives to share information on their gains from otherwise problematic species, which may also play a role in this difference.

Quantification of non-native species' economic benefits can be readily performed for some sectors, which can contribute to the development of assessment frameworks. The Food and Agriculture Organization (FAO), for example, has a statistical framework to record benefits arising from non-native fish farming. Fisheries and aquaculture production globally relies heavily on non-native species, and their economic benefits often far outweigh the costs for farmers and fishers. In many parts of the world where fisheries and fish farming are an essential source of affordable proteins for the local population, there is little incentive to reduce production of non-native species. Such profitable markets therefore flourish (e.g. see Gozlan 2015), especially considering that any costs rarely burden fishing or aquaculture operators, but instead fall upon government agencies and/or the general public. It is important to note though that only a small proportion of DE-IAS are associated with sizable economic benefits, and the majority of these relate to intrinsic values rather than tangible gains. Primary economic sectors, such as forestry and fisheries, may be associated with disproportionately frequent occurrences of invasions with direct monetary benefits. Otherwise, intrinsic values often require non-market valuation methods which can be based on peoples' real-world choices or behaviours (revealed preference methods), or rely on peoples' explicit statements regarding their values or making choices over hypothetical scenarios proposed in surveys (stated preference methods). However, non-market valuation methods, and especially stated preferences methods, often come with limitations and biases owing to people's subjective perceptions (Shackleton et al. 2019a, b).

Stakeholder engagement frameworks have been gaining traction in recent years as they hold significant potential to mitigate conflict, reducing barriers to management, increasing inclusiveness in decision making, acceptance of management and equitable outcomes (Victorian Goverment 2010; Braysher 2017; Crowley et al. 2017a, b; Hui and Richardson 2017; Sarkar and Minteer 2018; Shackleton et al. 2019a; Aley et al. 2020; Palmer et al. 2020; Villarreal-Rosas et al. 2020). Such engagement is crucial given that oftentimes limited literature, research funding, time constraints and different value systems would otherwise hinder the understanding of trade-offs and the diversity of perceptions on DE-IAS (Gaertner et al. 2016; Woodford et al. 2017). For example, Novoa et al. (2018)) developed a 12-step framework for stakeholder engagement focused on principles of collaboration, with the expectation that this can accommodate a diverse range of needs and lead to workable management strategies broadly accepted by stakeholders on both sides. Similarly, Gaertner et al. (2016), with an eye towards urban environments, discuss explicit and transparent consideration of stakeholder perceptions on DE-IAS. Their proposed 3-management approach focussed on tolerance, active engagement and control priorities, helps pragmatic solutions emerge by bringing to surface a diverse range of tangible and intangible perceived costs and benefits.

Management through market mechanisms

Incentive-based mechanisms are increasingly used by resource managers to combat IAS and these generally involve the development of commercial markets, joining existing ones, or the promotion of recreational harvest, bounty programs and contract operations (Pasko and Goldberg 2014). Market-based instruments can create incentives to reduce a high population of a DE-IAS, and in some cases, where the goal is modest population reductions, may be more cost-effective compared to larger, coordinated control operations (Nugent and Choquenot 2004). However, market mechanisms may also exacerbate the invasion problem if the instrument used does not provide sufficient incentives or is not designed appropriately to meet management goals (for example harvesting enough individuals at the right life stage or spatial scale), and the effects may vary based on how influential resource users with positive/negative stakes are. Creating a market for invasions that have potential as economic resources has shown that such practices occasionally generate pressure among local communities to protect the harmful species, and may also create additional pressures to maintain a larger population (Nuñez et al. 2012). Such local support may lead to justifying larger steady state populations where ecosystem damages are ignored or undervalued against direct economic benefits. Additionally, people participating in harvests where the target species evolves as an economic resource might also wish to recreate the market in uninvaded areas (Nuñez et al. 2012).

Importantly, the fact that the harvest of a DE-IAS may command a positive value does not imply that efforts to develop a market for it, or to join an existing one, will contribute towards limiting its spread or controlling its population. Despite the fact that the outcome of attempts to use commercial harvesting for the control of DE-IAS has rarely been clear-cut, scientists and wildlife managers around the world have been increasingly considering commercial harvesting as a control mechanism for invasions. However, in many of those cases a solid understanding of how the invasion evolved is missing, which may lead to misleading policy recommendations. For example, the Norwegian strategy on the red king crab, although unclear regarding its effectiveness in minimizing its spread (Kourantidou and Kaiser 2021), has been suggested as a viable and successful control mechanism for the blue crab (Callinectes sapidus) in the Mediterranean and Black Seas as well as eastern Atlantic coasts of the Iberian Peninsula (Mancinelli et al. 2017). Suggesting that consumers' interest for an IAS and willingness to pay for it can lead to eradication or substitute other control mechanisms, such as in the case of the Asian carp in the U.S. (Varble and Secchi 2013), can be both erroneous and dangerously misleading. Indeed, Tsehaye et al. (2013) find that current fishing practices may reduce biomass, but are unlikely to drive carp populations to extinction unless additional incentives to improve size selectivity and species targeting are put in place. The fact that an IAS is safe to consume does not determine that human consumption can operate as a successful control mechanism as some resource managers and/or scholars suggest (see for example Clark et al. 2009 for the Chinese mitten crab). These types of recommendations should be made with care and ideally through analysis of population dynamics and responses to harvesting. That is because they may carry the risk of turning an IAS into a DE-IAS where incentives are developed to maintain the harmful population, as benefits from commercialization of an IAS tend to grow larger as the invasion develops. This is especially the case if commercial harvesting evolves into an important source of income, potentially supporting livelihoods of those involved in the harvest and/or supporting the development of industries, all of which create complex trade-offs between ecosystem conservation and economic development (Nuñez et al. 2012).

Examples of DE-IAS that contribute to locally important industries include the invasive Australian crayfish redclaw (Cherax quadricarinatus) in Jamaica (Pienkowski et al. 2015), exotic salmonids in Patagonia (Pascual et al. 2009), wattle species (Acacia mearnsii and Acacia dealbata) and Prickly pear (Opuntia ficus-indica) in South Africa (de Neergaard et al. 2005) and Prosopis (Prosopis juliflora) in Sudan, Kenya and other places in Africa (Geesing et al. 2004; Mwangi and Swallow 2005; Laxén 2007; Bokrezion 2008). The example of the redclaw crayfish Cherax quadricarinatus is particularly interesting due to its near global production in astaciculture, where it provides economic opportunities and contributes to food security as an inexpensive and easily accessible source of protein. However, escapes have resulted in feral populations in 22 countries on five continents, posing large risks to the recipient environment as evidenced through impacts to native freshwater biota and multiple ecosystem services in Africa (Madzivanzira et al. 2020) and North America (Mcloughlan 2014). Despite a lack of thorough large-scale impact and risk assessments, the perceived benefits of food security often seem to outweigh these risks, particularly in rural regions. Consequently, redclaw continues to be promoted globally for aquaculture and the pet trade where its high price provides incentives for customers and aquaculture growers (Haubrock et al. 2021a).

Management through bounties has also been implemented both historically and more recently for a range of IAS and also DE-IAS. Because they can inform as to the location and encounter frequencies in a given place and time, bounties can be seen as a ‘passive surveillance’ measure, which helps resource management agencies to control and reduce the expense of searching for invaded sites over a large area (e.g. see Cacho and Hester 2011). However, bounty systems can easily be misused in multiple ways and frequently fail to meet their intended goals as a result of undesirable human behavioral responses that produce unexpected outcomes through the triggering of adverse incentives (Bulte et al. 2003; Vann 2003; Walker 2013; Chapman 2016). Attempts have been made to combine bounty programs with other mechanisms such as the development of commercial markets, again with limited evidence for effectiveness. Norway manages the red king crab as a commercial fishery with quotas in the eastern part of its introduced range, but with an open-access fishery to the west supplemented by a bounty payment scheme for catches of undersized and female crabs to minimize westward spread. It remains unclear whether these measures, aimed at balancing the risk of spread with the benefits from commercial exploitation, successfully meet the management goals. There are fears that fishers may develop incentives to: (a) harvest in the quota-regulated area and land their catch in the open-access area to benefit from the bounties and (b) purposefully introduce crabs in order to maintain the bounty system (Kourantidou 2018).

These examples demonstrate the need to build a thorough understanding of the invaded system and interactions with resource users, since lack of knowledge can hinder effective management approaches. Ideally actions should be taken within an adaptive management framework so that the existence of uncertainty does not result in delays, but allows for management to be adjusted and updated as new information is acquired (Williams et al. 2009; Bodey et al. 2010; Parrott 2017). However, there also remains a need for novel and innovative approaches that successfully combine or reconcile market-based mechanisms with effective management of DE-IAS.

Policy frameworks and persisting knowledge gaps

Policy frameworks

Regulatory and policy frameworks

With a few exceptions, international policy and regulatory frameworks on invasions are generally bereft of comprehensive assessments or guidelines for managing DE-IAS. International legislation frameworks for IAS such as the Convention on Biological Diversity (CBD) do not provide clear frameworks that can help resource managers address DE-IAS in practice. Additionally, there are examples where the timing of when a country ratified the convention, or the convention became effective (if for example post-introduction), has been used to justify limited efforts or obligation to control and/or eradicate (Acoura 2017; Kourantidou and Kaiser 2019b).

Previous work, focusing on the aquaculture sector—a prominent source of DE-IAS—has begun to address the issue. The IMPASSE (2008) project identified and assessed the time, purpose and ecological, social and economic impacts of species introduced for aquaculture across all European countries. The risk assessment protocols, guidelines and recommendations of this project were used as an input by the European Commission for identifying species that can be imported into the EU without a specific license (Annex IV, 'Council Regulation 708/2007 concerning use of alien and locally absent species in aquaculture', (EC) No 708/2007 (EC-ASR)). However, species that have been used in aquaculture for extended periods are generally exempt from the principles of the EU regulation, based on the reasoning of 'no adverse effects'. This includes some commercially important, yet harmful DE-IAS such as the Manila clam (Ruditapes philippinarum) and the pacific oyster (Crassostrea gigas) whose impacts are well-documented (Galil et al. 2013). For example, the commercial exploitation of the pacific oyster, despite its deleterious impacts, is in accordance with EU regulations for prevention and management of the introduction and spread of IAS (EU, No 1143/2014). The long history of aquaculture together with its economic weight makes it possible to list the oyster within Annex IV of the European Commission Council Regulation. It is worth noting though that, despite its caveats, the EU regulation (No 1143/2014) provides a broadly useful framework for DE-IAS; while it does not aim to limit European trade, even if that includes introduction and movement of non-native species, it targets species that have not been deliberately introduced.

The value from commercial operations should be appropriately traded-off against the impacts of an invasion, which may continue and exacerbate through time and space. It is therefore important that sufficiently dynamic frameworks are established that allow for ongoing assessment and updating of knowledge across all aspects to address the potential risks and trade-offs. This can inform lists of both exempt and banned species, acknowledging that listing a species as exempt should not necessarily be a permanent distinction as this fails to consider temporal changes and advances in knowledge through time.

Private industry interests affecting management and policy frameworks

In several cases where valuable DE-IAS have become part of a commercial market, industries or actors engaged in commercial operations have sought to certify their product as 'sustainable'. These may include resources-products of value to some industry, edible resources or goods and services supporting tourism and recreation. Certification of a DE-IAS typically requires considerable financial investments from interested market actors and provides reassurance to consumers that their choice is a responsible one; but potentially leads to misconceptions on overall impact and, in the worst-case scenario, could exacerbate environmental problems.

In 2018, the Marine Stewardship Council certified the Russian red king crab fishery in the Barents Sea, as a ‘sustainable and well-managed fishery’, despite multiple concerns expressed concerning the ecosystem impacts of this invasion (Kourantidou and Kaiser 2019b). However, the Marine Stewardship Council standard does not encompass such broader ecosystem concerns, but rather focuses on narrowly defined principles of sustainability of stocks, environmental impacts of fishing practices and effective management for commercial production. This same principle resulted in certification of the Russian Barents fishery for the invasive snow crab Chionoecetes opilio, for which the impacts are even less understood and largely underplayed in light of high profits and other geopolitical interests in the region (Kaiser et al. 2018; MSC 2020). Similar concerns have been expressed for certification of other fisheries that rely on IAS with well-documented negative impacts on native species and the marine ecosystem, such as the invasive Manila clam (Ruditapes philippinarum) fishery in Ria Arousa, Spain (Galil et al. 2013).

Conflicting economic goals between groups are a common feature of resource management generally, and regularly produce conflicting views on legislative instruments. Slow processes in revising legislation combined with limited scientific knowledge, or stance and motivation on the part of much of the general public, may exacerbate uncertainties along with any deleterious invasion impacts. At the same time, private industry interests, which tend to drive third-party certifications such as the Marine Stewardship Council, and interests that lobby for resource management in favor of sustaining DE-IAS populations, can make it difficult to ensure that outcomes of certification or regulation do not conflict with biodiversity conservation objectives. That is because management of commercially valuable IAS often relies on standards set by the industry. Government initiatives may in theory be more appropriate and mindful of invasion impacts and therefore a more credible source of sustainability at all levels, depending of course on the definitional framework of sustainability (Kourantidou and Kaiser 2019b). In addition to their responsibility to manage natural resources for the public good, state authorities are also bound by national legislation for IAS management and international treaties, such as the CBD. Ideally, government-industry agreements that generate widespread support and provide verifiable codes of practice that inform consumers will produce the best outcomes in managing species that are potential or actual DE-IAS (Hulme et al. 2018; Lockwood et al. 2019).

Knowledge gaps

Knowledge gaps on ecological and economic impacts may allow space for the cultural services from an IAS to dominate other considerations, which can in some cases lead to resistance to managing these species as a regular invasion. Examples of such cultural services include intrinsic, aesthetic, spiritual and recreational benefits along with many other constituents of human well-being (Barr et al. 2002; Bertolino and Genovesi 2003; Kerr and Abell 2014; Bacher et al. 2018; Shackleton et al. 2019b). Emotional or cultural attachment can result in opposition to eradication initiatives as seen with charismatic IAS including the grey squirrel in Ireland, U.K., Italy and South Africa (Shackleton et al. 2019b), the monk parakeet in the U.K and elsewhere (Beever et al. 2019; Crowley et al. 2019), and the colorful jacaranda tree in South Africa (Dickie et al. 2014). Indeed, it is very common for people to forge cultural connections in places they live (for example invasive trees being a local symbol or part of the identity and sense of place, such as Jacaranda for Pretoria, South Africa, Jacaranda festivals in Grafton, Australia or the “Eucalyptus School” of art, in California, USA), which are expected to be pronounced in urban areas where cultural heterogeneity is stronger (Nuñez and Simberloff 2005; Dickie et al. 2014;

Novoa et al. 2017). Oftentimes such cultural attachments may not become evident until there is a legitimate threat to the IAS or invaded landscape (Crowley et al. 2017b).

The valuation of ecosystem goods and services, although complex, at times daunting and subject to criticisms, is a fundamental tenet in decision-making for resource management, including DE-IAS (Schröter et al. 2014; Silvertown 2015; Potschin et al. 2016; Schröter and van Oudenhoven 2016; Temel et al. 2018). Despite these challenges and inherent limitations, the development and use of robust decision-making frameworks that draw on benefits, costs and stakeholder perceptions are a stepping stone in dealing with DE-IAS and the conflicts those species generate (Gaertner et al. 2016). From an anthropocentric standpoint, IAS may provide new ecological, environmental or cultural services or restore previously eroded services. Viewing these services through a narrow lens though, may come in direct conflict with conservation goals and undermine the role of IAS management efforts. Commodification of IAS brings in an additional challenge associated with disparities in the way costs and benefits evolve. For example, while direct market costs (for example crop damaging IAS) and benefits (for example IAS as new crops) adhere to market fluctuations (i.e. supply and demand), that is not the case for non-market costs from lost ecosystem services, biodiversity loss or ecosystem degradation. Market mechanisms yield lower demand (or below optimal levels) since they do not account for such ecosystem components, which are essentially ‘public goods’ (Fisher et al. 2008). Although assigning monetary values to positive and negative impacts of IAS can be helpful in shaping policies on management and conservation, it is equally important to consider how non-market valuation factors or may fail to factor into the equation (Hanley and Roberts 2019). For optimal outcomes, it is necessary to account for non-market values which will shift the demand curve upwards for impacts that are both marketed and non-marketed. Additionally, considering that positive and negative impacts associated with DE-IAS change and evolve through time (van Wilgen and Richardson 2014), it may be more appropriate to consider a gradient between these effects. This could better account for the evolution or degradation of benefits and costs temporally associated with market and non-market mechanisms for impacting species, which is otherwise restricted when considering a binary classification between IAS and DE-IAS. However, for many species, knowledge gaps, context-dependencies and a lack of resolution in the available data presently preclude reliable assignment along such a gradient. Public consultation and stakeholder engagement (Novoa et al. 2018) could bolster such assessments and aid in quantifying perspectives and values, designing policy, and dealing with uncertainties considering multiple values associated with DE-IAS.

Several other knowledge gaps and potential “biases” pertinent to DE-IAS, that we have not explicitly considered in this paper, are also worthy of consideration. These may include consideration for the (dis)proportional scale of negative impacts relative to benefits, and vice versa. In cases where certain groups of stakeholders have vested interests in either costs or benefits of the DE-IAS, advocacy may influence public perceptions to recognize only one over the other. In cases where benefits overshadow costs, this becomes a particularly large concern since negative impacts may be obfuscated and lead to amplification of environmental risks. Generally, it is likely that species with a known high documented value or cost tend to be foci for management and public authorities. In light of the many unknowns and the selective reporting of costs, for several IAS with benefits, identifying trade-offs accurately becomes particularly difficult and calls for nuanced considerations on the determinants of research priorities and decisions on public expenditure for IAS.

In the absence of a complete and comprehensive repository that includes records of DE-IAS with detailed and standardized information on potential cost and benefits, we remain largely agnostic about the existence of biases in reporting. However, efforts to identify such biases in research are critical to ensure that they may not feed into management and future work, through for example altering resource-managers' and policy-makers' perceptions or shaping future research funding priorities.

Outlook

A small but increasing number of scientists have criticized the way IAS are characterized in the ecology literature by highlighting positive aspects of invasions in an ecosystem service framework or by exploring possible biases in the IAS literature and the ways in which invasion biology is communicated to the general public (García-Llorente et al. 2008; Gozlan 2008; Pejchar and Mooney 2009; Stromberg et al. 2009; Gozlan et al. 2013; Warren II et al. 2017; Vimercati et al. 2020). While this literature can help to understand the multiple dimensions of DE-IAS, it is important they do not undermine research efforts for better understanding unexplored negative impacts (Richardson and Ricciardi 2013). Similarly, these efforts should not be diminished by arguments that benefits of IAS have received less attention in the literature compared to damages or losses (Bonanno 2016; Chapman 2016).

The InvaCost database that we used to describe patterns and trends through time, provides a centralised platform for addressing such important knowledge gaps and enables the worldwide reporting of economic costs of invasions in a standardised, comparable format and in multiple languages (Diagne et al. 2020b; Angulo et al. 2021). Nonetheless, invasion costs are simply not reported or accessible for many countries, taxonomic groups and sectors of the economy (Haubrock et al. 2021b; Kourantidou et al. 2021). Most importantly, InvaCost currently offers only a qualitative, binary report of the presence of benefits from invaders, and therefore does not allow for assessing simultaneously, in a quantitative manner, the reported costs and benefits of an invasion. The current configuration of the database only permits the recording of benefits from studies which also report costs, and therefore studies identifying DE-IAS are likely under-estimated. We suggest that further collation of such missing data on the nature and magnitude of DE-IAS benefits, if available, could help to resolve this, clarifying management choices for decision makers, and resolving arguments around the characterization of IAS in general. While efforts are underway to extend existing impact classification systems like the ecological and socio-economic impact classification of alien taxa (Blackburn et al. 2011; Bacher et al. 2018), it remains critical to understand the potential for bias within such assessments based on incomplete reporting, and to not only rigorously test for this, but to ensure such frameworks are sufficiently light-footed to react to and incorporate new information (Vimercati et al. 2020). Similarly, it is important to be equipped with a sufficiently diverse toolbox of policies and strategies that can be adapted to local conditions, as well as to promote the design of management interventions that allow, wherever possible, for capturing any positive impacts while at the same time effectively mitigating negative impacts (García-Díaz et al. 2021).

This study sheds light on key knowledge gaps and highlights the need to advance the way resource managers treat DE-IAS. Improving understanding of the trade-offs for DE-IAS, their management challenges, resistance from various stakeholders and other barriers to optimal management, points at the need for cross-disciplinary scientific inquiries (Keller et al. 2009; Estévez et al. 2015; Crowley et al. 2017a, b). Transcending traditional disciplinary boundaries between social and natural sciences is key in evaluating these trade-offs. Fostering effective utilization of interdisciplinary models (e.g., including economic and ecological theory) and identifying optimal management strategies goes beyond just academia and requires inclusion of societal collaborators with diverse backgrounds. Views and interests of different stakeholder groups, however, may be assigned with different weights (Kumschick et al. 2012) which complicates the process through which conservation managers should seek input from stakeholders within each group. Nevertheless, policies and resource management should be designed mindful of the conditions that underpin overall human and social wellbeing. This stresses the need for a deeper understanding of DE-IAS features, society's perceptions on those species and frameworks that allow for comprehensive assessments, which are also necessary to equip conservation agencies to manage them.

Publisher's Note

Acknowledgements

The authors acknowledge the French National Research Agency (ANR-14-CE02-0021) and the BNP-Paribas Foundation Climate Initiative for funding the Invacost project that allowed the construction of the InvaCost database. Furthermore, the authors wish to thank Dr. Nigel Taylor for useful discussions that informed the conceptualization of this paper. The present work was conducted following a workshop funded by the AXA Research Fund Chair of Invasion Biology and is part of the AlienScenario project funded by BiodivERsA and Belmont-Forum call 2018 on biodiversity scenarios (BMBF/PT DLR 01LC1807C). CD is funded by AlienScenarios and RC by the Alexander von Humboldt Foundation.

Authors' contribution

MK and FC conceptualized the study. MK, TB, PH, RC and FC contributed to the design of the study. MK, PH and RC contributed to processing, analysing and visualising the data. MK led the writing of the manuscript with input from all co-authors. All authors contributed to writing the manuscript and approved its submission.

Availability of data and materials

Not applicable.

Declarations

Conflict of interest: The authors have declared that no competing interests exist.

Supplementary Information

Supplementary file 1 (DOCX 442 KB)

References

Abila RO (2000) The development of the Lake Victoria fishery: a boon or bane for food security? IUCN Report No. 8. Nairobi, Kenya

Acoura (2017) MSC Sustainable fisheries certification. Russia Barents Sea Red King Crab. https://cert.msc.org/FileLoader/FileLinkDownload.asmx/GetFile?encryptedKey=JkTPLYsG+EjOnfFBOHC2m/CJ0dL/LCzfi6dfb3uHn4o2z1yJQzTfqDrdH6TKaQJf. Accessed 7 Dec 2018

Ahmed DA, Hudgins EJ, Cuthbert RN et al (2021) Managing biological invasions: the cost of inaction. Res Sq (pre-print). https://doi.org/10.21203/rs.3.rs-300416/v1

Aley JP, Milfont TL, Russell JC (2020) The pest-management attitude (PMA) scale: a unidimensional and versatile assessment tool. Wildl Res 47:166–176. https://doi.org/10.1071/WR19094

Aloo PA, Njiru J, Balirwa JS, Nyamweya CS (2017) Impacts of Nile perch, *Lates niloticus*, introduction on the ecology, economy and conservation of Lake Victoria, East Africa. Lakes Reserv Res Manag 22:320–333. https://doi.org/10.1111/lre.12192

Angulo E, Diagne C, Ballesteros-Mejia L et al (2021) Non-English languages enrich scientific knowledge: the example of economic costs of biological invasions. Sci Total Environ. https://doi.org/10.1016/j.scitotenv.2020.144441

Bacher S, Blackburn TM, Essl F et al (2018) Socio-economic impact classification of alien taxa (SEICAT). Methods Ecol Evol 9:159–168. https://doi.org/10.1111/2041-210X.12844

Barr JJF, Lurz PWW, Shirley MDF, Rushton SP (2002) Evaluation of immunocontraception as a publicly acceptable form of vertebrate pest species control: the introduced grey squirrel in Britain as an example. Environ Manag 30:342–351. https://doi.org/10.1007/s00267-002-2686-7

Beever EA, Simberloff D, Crowley SL et al (2019) Social–ecological mismatches create conservation challenges in introduced species management. Front Ecol Environ 17:117–125. https://doi.org/10.1002/fee.2000

Bellard C, Cassey P, Blackburn TM (2016) Alien species as a driver of recent extinctions. Biol Lett 12:20150623. https://doi.org/10.1098/rsbl.2015.0623

Bertolino S, Genovesi P (2003) Spread and attempted eradication of the grey squirrel (*Sciurus carolinensis*) in Italy, and consequences for the red squirrel (*Sciurus vulgaris*) in Eurasia. Biol Conserv 109:351–358. https://doi.org/10.1016/S0006-3207(02)00161-1

Billé R, Laurans Y, Mermet L et al (2012) Valuation without action? On the use of economic valuations of ecosystem services. IDDRI Policy Br 7:1–4

Blackburn TM, Pyšek P, Bacher S et al (2011) A proposed unified framework for biological invasions. Trends Ecol Evol 26:333–339. https://doi.org/10.1016/j.tree.2011.03.023

Blackburn TM, Essl F, Evans T et al (2014) A unified classification of alien species based on the magnitude of their environmental impacts. PLoS Biol 12:e1001850. https://doi.org/10.1371/journal.pbio.1001850

Bodey TW, Bearhop S, Roy SS et al (2010) Behavioural responses of invasive American mink *Neovison vison* to an eradication campaign, revealed by stable isotope analysis. J Appl Ecol 47:114–120

Bokrezion H (2008) The ecological and socio-economic role of *Prosopis juliflora* in Eritrea. Acad Diss Johannes Gutenberg-Universität Mainz, Ger (PhD report)

Bonanno G (2016) Alien species: to remove or not to remove? That is the question. Environ Sci Policy 59:67–73. https://doi.org/10.1016/j.envsci.2016.02.011

Bradshaw CJA, Leroy B, Bellard C et al (2016) Massive yet grossly underestimated global costs of invasive insects. Nat Commun 7:1–8. https://doi.org/10.1038/ncomms12986

Braysher M (2017) Managing Australia's pest animals: a guide to strategic planning and effective management. Csiro Publishing, Clayton

Britton JR, Gozlan RE, Copp GH (2011) Managing non-native fish in the environment. Fish Fish 12:256–274. https://doi.org/10.1111/j.1467-2979.2010.00390.x

Brundu G, Pauchard A, Pyšek P et al (2020) Global guidelines for the sustainable use of non-native trees to prevent tree invasions and mitigate their negative impacts. NeoBiota. https://doi.org/10.3897/neobiota.61.58380

Bulte EH, Horan RD, Shogren JF (2003) Is the Tasmanian tiger extinct? A biological-economic re-evaluation. Ecol Econ 45:271–279. https://doi.org/10.1016/S0921-8009(03)00076-4

Cacho OJ, Hester SM (2011) Deriving efficient frontiers for effort allocation in the management of invasive species. Aust J Agric Resour Econ 55:72–89. https://doi.org/10.1111/j.1467-8489.2010.00520.x

Castro-Díez P, Vaz AS, Silva JS et al (2019) Global effects of non-native tree species on multiple ecosystem services. Biol Rev 94:1477–1501. https://doi.org/10.1111/brv.12511

Chapman P (2016) Benefits of invasive species. Mar Pollut Bull 1:1–2. https://doi.org/10.1016/j.marpolbul.2016.04.067

Clark PF, Mortimer DN, Law RJ et al (2009) Dioxin and PCB contamination in Chinese mitten crabs: human consumption as a control mechanism for an invasive species. Environ Sci Technol 43:1624–1629. https://doi.org/10.1021/es802935a

Copp GH, Vilizzi L, Mumford J et al (2009) Calibration of FISK, an invasiveness screening tool for nonnative freshwater fishes. Risk Anal Int J 29:457–467. https://doi.org/10.1111/j.1539-6924.2008.01159.x

Copp GH, Vilizzi L, Tidbury H et al (2016) Development of a generic decision-support tool for identifying potentially invasive aquatic taxa: AS-ISK. Manag Biol Invasions 7:343–350. https://doi.org/10.3391/mbi.2016.7.4.04

Courchamp F, Fournier A, Bellard C et al (2017) Invasion biology: specific problems and possible solutions. Trends Ecol Evol 32:13–22. https://doi.org/10.1016/j.tree.2016.11.001

Courtois P, Mullier C, Salles JM (2012) Managing biological invasions: the good, the bad and the ambivalent. In: 14th BioEcon conference. Cambridge, UK

Crowley SL, Hinchliffe S, McDonald RA (2017a) Invasive species management will benefit from social impact assessment. J Appl Ecol 54:351–357. https://doi.org/10.1111/1365-2664.12817

Crowley SL, Hinchliffe S, McDonald RA (2017b) Conflict in invasive species management. Front Ecol Environ 15:133–141. https://doi.org/10.1002/fee.1471

Crowley SL, Hinchliffe S, McDonald RA (2019) The parakeet protectors: understanding opposition to introduced species management. J Environ Manag 229:120–132. https://doi.org/10.1016/j.jenvman.2017.11.036

Crystal-Ornelas R, Lockwood JL (2020) The ‘known unknowns’ of invasive species impact measurement. Biol Invasions 22:1513–1525. https://doi.org/10.1007/s10530-020-02200-0

de Neergaard A, Saarnak C, Hill T et al (2005) Australian wattle species in the Drakensberg region of South Africa: an invasive alien or a natural resource? Agric Syst 85:216–233. https://doi.org/10.1016/j.agsy.2005.06.009

De Wit MP, Crookes DJ, Van Wilgen BW (2001) Conflicts of interest in environmental management: estimating the costs and benefits of a tree invasion. Biol Invasions 3:167–178. https://doi.org/10.1023/A:1014563702261

Diagne C, Catford JA, Essl F et al (2020a) What are the economic costs of biological invasions? A complex topic requiring international and interdisciplinary expertise. NeoBiota 63:25–37. https://doi.org/10.3897/neobiota.63.55260

Diagne C, Leroy B, Gozlan RE et al (2020b) InvaCost, a public database of the economic costs of biological invasions worldwide. Sci Data 7:1–12. https://doi.org/10.1038/s41597-020-00586-z

Dickie IA, Bennett BM, Burrows LE et al (2014) Conflicting values: ecosystem services and invasive tree management. Biol Invasions 16:705–719. https://doi.org/10.1007/s10530-013-0609-6

Downing AS, Galic N, Goudswaard KPC et al (2013) Was Lates late? A null model for the Nile perch boom in Lake Victoria. PLoS ONE. https://doi.org/10.1371/journal.pone.0076847

Ens E, Fisher J, Costello O (eds) (2015) Indigenous people and invasive species: perceptions, management, challenges and uses. IUCN Commission on Ecosystem Management Community Report

Epanchin-Niell RS (2017) Economics of invasive species policy and management. Biol Invasions 19:3333–3354. https://doi.org/10.1007/s10530-017-1406-4

Epanchin-Niell RS, Hastings A (2010) Controlling established invaders: integrating economics and spread dynamics to determine optimal management. Ecol Lett 13:528–541. https://doi.org/10.1111/j.1461-0248.2010.01440.x

Essl F, Dullinger S, Rabitsch W et al (2011) Socioeconomic legacy yields an invasion debt. Proc Natl Acad Sci 108:203–207. https://doi.org/10.1073/pnas.1011728108

Essl F, Dullinger S, Genovesi P et al (2019) A conceptual framework for range-expanding species that track human-induced environmental change. Bioscience 69:908–919. https://doi.org/10.1093/biosci/biz101

Estévez RA, Anderson CB, Pizarro JC, Burgman MA (2015) Clarifying values, risk perceptions, and attitudes to resolve or avoid social conflicts in invasive species management. Conserv Biol 29:19–30. https://doi.org/10.1111/cobi.12359

Falk-Petersen J, Bøhn T, Sandlund OT (2006) On the numerous concepts in invasion biology. Biol Invasions 8:1409–1424. https://doi.org/10.1007/s10530-005-0710-6

Fisher B, Turner K, Zylstra M et al (2008) Ecosystem services and economic theory: integration for policy-relevant research. Ecol Appl 18:2050–2067. https://doi.org/10.1890/07-1537.1

Fleming PJS, Ballard G, Reid NCH, Tracey JP (2017) Invasive species and their impacts on agri-ecosystems: issues and solutions for restoring ecosystem processes. Rangel J 39:523–535. https://doi.org/10.1071/RJ17046

Forseth IN, Innis AF (2004) Kudzu (Pueraria montana): history, physiology, and ecology combine to make a major ecosystem threat. CRC Crit Rev Plant Sci 23:401–413. https://doi.org/10.1080/07352680490505150

Gaertner M, Biggs R, Te Beest M et al (2014) Invasive plants as drivers of regime shifts: identifying high-priority invaders that alter feedback relationships. Divers Distrib 20:733–744. https://doi.org/10.1111/ddi.12182

Gaertner M, Larson BMH, Irlich UM et al (2016) Managing invasive species in cities: a framework from Cape Town, South Africa. Landsc Urban Plan 151:1–9. https://doi.org/10.1016/j.landurbplan.2016.03.010

Galil BS, Genovesi P, Ojaveer H et al (2013) Mislabeled: eco-labeling an invasive alien shellfish fishery. Biol Invasions 15:2363–2365. https://doi.org/10.1007/s10530-013-0460-9

Garcia De Leaniz C, Gajardo G, Consuegra S (2010) From best to pest: changing perspectives on the impact of exotic salmonids in the southern hemisphere. Syst Biodivers 8:447–459. https://doi.org/10.1080/14772000.2010.537706

García-Díaz P, Cassey P, Norbury G et al (2021) Management policies for invasive alien species: addressing the impacts rather than the species. Bioscience 71:174–185. https://doi.org/10.1093/biosci/biaa139

García-Llorente M, Martín-López B, González JA et al (2008) Social perceptions of the impacts and benefits of invasive alien species: Implications for management. Biol Conserv 141:2969–2983. https://doi.org/10.1016/j.biocon.2008.09.003

Geesing D, Al-Khawlani M, Abba ML (2004) Management of introduced Prosopis species: can economic exploitation control an invasive species. Unasylva 217:36–44

Goodenough AE (2010) Are the ecological impacts of alien species misrepresented? A review of the “native good, alien bad” philosophy. Community Ecol 11:13–21. https://doi.org/10.1556/ComEc.11.2010.1.3

Victorian Goverment (2010) Invasive plants and animals policy framework. Department of Primary Industries, Melbourne

Gozlan RE (2008) Introduction of non-native freshwater fish: is it all bad? Fish Fish 9:106–115. https://doi.org/10.1111/j.1467-2979.2007.00267.x

Gozlan RE (2015) Role and impact of non-native species on inland fisheries: the Janus syndrome. Freshw Fish Ecol. https://doi.org/10.1002/9781118394380.ch53

Gozlan RE (2017) Interference of non-native species with fisheries and aquaculture. In: Vilà M, Hulme P (eds) Impact of biological invasions on ecosystem services. Invading Nature-Springer series in invasion ecology, vol 12. Springer, Cham, pp 119–137

Gozlan RE, Newton AC (2009) Biological invasions: benefits versus risks. Science 324:1015. https://doi.org/10.1126/science.324_1015a

Gozlan RE, Burnard D, Andreou D, Britton JR (2013) Understanding the threats posed by non-native species: public vs. conservation managers. PLoS ONE 8:e53200. https://doi.org/10.1371/journal.pone.0053200

Hamblyn EL (1961) Nile perch Project. EAFFRO Annual Report.26-32, Jinja, Uganda

Han Y (2016) Ecosystem-wide management of invasive species in the face of severe uncertainty. PhD. The University of Queensland

Hanley N, Roberts M (2019) The economic benefits of invasive species management. People Nat. https://doi.org/10.1002/pan3.31

Harris CK, Wiley DS, Wilson DC (1995) Socio-economic impacts of introduced species in Lake Victoria fisheries. In: Pitcher TJ, Hart PJB (eds) The impact of species changes in African lakes. Springer, Berlin, pp 215–242

Haubrock PJ, Oficialdegui FJ, Zeng Y et al (2021a) The redclaw crayfish: a prominent aquaculture species with invasive potential in tropical and subtropical biodiversity. Rev Aquac 1:43. https://doi.org/10.1111/raq.12531

Haubrock PJ, Turbelin A, Cuthbert RN et al (2021b) Economic costs of invasive alien species across Europe. NeoBiota. https://doi.org/10.3897/neobiota.67.58196

Hui C, Richardson DM (2017) Invasion dynamics. Oxford University Press, Oxford

Hulme PE, Brundu G, Carboni M et al (2018) Integrating invasive species policies across ornamental horticulture supply chains to prevent plant invasions. J Appl Ecol 55:92–98. https://doi.org/10.1111/1365-2664.12953

IMPASSE (2008) Final Report Summary—IMPASSE (Environmental impacts of invasive alien species in aquaculture). https://cordis.europa.eu/project/id/44142/reporting

IPBES (2019) Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. IPBES Secretariat, Bonn

Jarić I, Courchamp F, Correia RA et al (2020) The role of species charisma in biological invasions. Front Ecol Environ. https://doi.org/10.1002/fee.2195

Kaiser BA, Kourantidou M, Fernandez LM (2018) A Case for the commons: the snow crab in the Barents. J Environ Manag 210:338–348. https://doi.org/10.1016/j.jenvman.2018.01.007

Katsanevakis S, Wallentinus I, Zenetos A et al (2014) Impacts of invasive alien marine species on ecosystem services and biodiversity: a pan-European review. Aquat Invasions 9:391–423. https://doi.org/10.3391/ai.2014.9.4.01

Katsanevakis S, Tempera F, Teixeira H (2016) Mapping the impact of alien species on marine ecosystems: the Mediterranean Sea case study. Divers Distrib 22:694–707. https://doi.org/10.1111/ddi.12429

Kaufman L (1992) Catastrophic change in species-rich freshwater ecosystems. Bioscience 42:846–858. https://doi.org/10.2307/1312084

Keller RP, Lodge DM, Finnoff DC (2007) Risk assessment for invasive species produces net bioeconomic benefits. Proc Natl Acad Sci 104:203–207. https://doi.org/10.1073/pnas.0605787104

Keller RP, Lodge DM, Lewis MA, Shogren JF (2009) Bioeconomics of invasive species: integrating ecology, economics, policy, and management. Oxford University Press, Oxford

Kerr GN, Abell W (2014) Big game hunting in New Zealand: per capita effort, harvest and expenditure in 2011–2012. N Z J Zool 41:124–138. https://doi.org/10.1080/03014223.2013.870586

Kourantidou M (2018) Stewardship of resources in rapidly evolving Arctic economies and ecosystems: the role of marine invasive species. PhD Thesis Univ South Denmark Dep Sociol Environ Bus Econ. ISBN 978-87-93669-38-3

Kourantidou M, Kaiser BA (2019a) Research agendas for profitable invasive species. J Environ Econ Policy 8:209–230. https://doi.org/10.1080/21606544.2018.1548980

Kourantidou M, Kaiser BA (2019b) Sustainable seafood certifications are inadequate to challenges of ecosystem change. ICES J Mar Sci 76:794–802. https://doi.org/10.1093/icesjms/fsy198

Kourantidou M, Kaiser BA (2021) Allocation of research resources for commercially valuable invasions: Norway's red king crab fishery. Fish Res. https://doi.org/10.1016/j.fishres.2020.105871

Kourantidou M, Cuthbert RN, Haubrock P et al (2021) Economic costs of invasive alien species in the Mediterranean basin. NeoBiota 67:427–458. https://doi.org/10.3897/neobiota.67.58926

Kumschick S, Bacher S, Dawson W et al (2012) A conceptual framework for prioritization of invasive alien species for management according to their impact. NeoBiota 15:69–100. https://doi.org/10.3897/neobiota.15.3323

Latombe G, Lenzner B, Schertler A et al (2020) How moral values influence conservation: a framework to capture different management perspectives. bioRxiv

Laurans Y, Mermet L (2014) Ecosystem services economic valuation, decision-support system or advocacy? Ecosyst Serv 7:98–105. https://doi.org/10.1016/j.ecoser.2013.10.002

Laurans Y, Rankovic A, Billé R et al (2013) Use of ecosystem services economic valuation for decision making: questioning a literature blindspot. J Environ Manage 119:208–219. https://doi.org/10.1016/j.jenvman.2013.01.008

Laxén JPE (2007) Is prosopis a curse or a blessing?: An ecological economic analysis of an invasive alien tree species in Sudan. Viikki Tropical Resources Institute, University of Helsinki Tropical Forestry Reports, PhD Thesis. Viikin tropiikki-instituutti, VITRI

Le Maitre DC, Van Wilgen BW, Chapman RA, McKelly DH (1996) Invasive plants and water resources in the Western Cape Province, South Africa: modelling the consequences of a lack of management. J Appl Ecol. https://doi.org/10.2307/2405025

Leroy et al (2021) Global costs of biological invasions: living figure. https://borisleroy.com/invacost/invacost_livingfigure.html

Ligtvoet W, Witte F, Goldschmidt T et al (1991) Species extinction and concomitant ecological changes in Lake Victoria. Neth J Zool 42:214–232. https://doi.org/10.1163/156854291X00298

Lockwood JL, Welbourne DJ, Romagosa CM et al (2019) When pets become pests: the role of the exotic pet trade in producing invasive vertebrate animals. Front Ecol Environ 17:323–330. https://doi.org/10.1002/fee.2059

Lodge DM, Simonin PW, Burgiel SW et al (2016) Risk analysis and bioeconomics of invasive species to inform policy and management. Annu Rev Environ Resour 41:453–488. https://doi.org/10.1146/annurev-environ-110615-085532

Lonsdale WM (1994) Inviting trouble: introduced pasture species in northern Australia. Aust J Ecol 19:345–354. https://doi.org/10.1111/j.1442-9993.1994.tb00498.x

Mack RN, Simberloff D, Mark Lonsdale W et al (2000) Biotic invasions: causes, epidemiology, global consequences, and control. Ecol Appl 10:689–710. https://doi.org/10.1890/1051-0761(2000)010[0689:BICEGC]2.0.CO;2

Madzivanzira TC, South J, Wood LE et al (2020) A review of freshwater crayfish introductions in Africa. Rev Fish Sci Aquac 29:218–241

Mancinelli G, Chainho P, Cilenti L et al (2017) On the Atlantic blue crab (*Callinectes sapidus* Rathbun 1896) in southern European coastal waters: time to turn a threat into a resource? Fish Res 194:1–8. https://doi.org/10.1016/j.fishres.2017.05.002

Martinez-Cillero R, Willcock S, Perez-Diaz A et al (2019) A practical tool for assessing ecosystem services enhancement and degradation associated with invasive alien species. Ecol Evol 9:3918–3936. https://doi.org/10.1002/ece3.5020

Mazza G, Tricarico E (2018) Invasive species and human health. CABI, Wallingford. https://doi.org/10.1079/9781786390981.0000

Mcloughlan S (2014) The distribution and population characteristics of *Cherax quadricarinatus*, in South East Queensland and Northern New South Wales. Honours Thesis. Griffith School of Environment. Griffith University, Gold Coast, Queensland, Australia, p 87

Messing RH (2000) The impact of nontarget concerns on the practice of biological control. In: Follett PA, Duan JJ (eds) Nontarget effects of biological control. Springer, Boston, pp 45–55

MSC (2020) Russia Barents Sea Opilio Trap Fishery. Public Certification Report. Lloyd's Register. https://cert.msc.org/FileLoader/FileLinkDownload.asmx/GetFile?encryptedKey=ChzB7lIh58cl4tRzwRKEf/t1VKZAs0ostJY7cOWEf85vqsC4UAMDGuNV+AQ+oXEA

Mugidde R, Gichuki J, Rutagemwa D et al (2005) Status of water quality and its implication on fishery production. In: The state of the fisheries resources of Lake Victoria and their management. In: Proceedings of the regional stakeholders' conference. Secretariat, Jinja, pp 106–112

Mwangi E, Swallow B (2005) Invasion of *Prosopis juliflora* and local livelihoods: case study from the lake Baringo area of Kenya. ICRAF Work Pap Nairobi, Kenya World Agrofor Cent 3. https://doi.org/10.5716/WP13657.PDF

Nentwig W, Bacher S, Pyšek P et al (2016) The generic impact scoring system (GISS): a standardized tool to quantify the impacts of alien species. Environ Monit Assess 188:315. https://doi.org/10.1007/s10661-016-5321-4

Ngorima A, Shackleton CM (2019) Livelihood benefits and costs from an invasive alien tree (*Acacia dealbata*) to rural communities in the Eastern Cape, South Africa. J Environ Manag 229:158–165. https://doi.org/10.1016/j.jenvman.2018.05.077

Njiru M, Waithaka E, Muchiri M et al (2005) Exotic introductions to the fishery of Lake Victoria: what are the management options? Lakes Reserv Res Manag 10:147–155. https://doi.org/10.1111/j.1440-1770.2005.00270.x

Novoa A, Dehnen-Schmutz K, Fried J, Vimercati G (2017) Does public awareness increase support for invasive species management? Promising evidence across taxa and landscape types. Biol Invasions 19:3691–3705. https://doi.org/10.1007/s10530-017-1592-0

Novoa A, Shackleton R, Canavan S et al (2018) A framework for engaging stakeholders on the management of alien species. J Environ Manag 205:286–297. https://doi.org/10.1016/j.jenvman.2017.09.059

Nugent G, Choquenot D (2004) Comparing cost-effectiveness of commercial harvesting, state-funded culling, and recreational deer hunting in New Zealand. Wildl Soc Bull 32:481–492. https://doi.org/10.2193/0091-7648(2004)32[481:CCOCHS]2.0.CO;2

Nuñez MA, Simberloff D (2005) Invasive species and the cultural keystone species concept. Ecol Soc. https://doi.org/10.5751/ES-01342-1001r04

Nuñez MA, Kuebbing S, Dimarco RD, Simberloff D (2012) Invasive species: to eat or not to eat, that is the question. Conserv Lett 5:334–341. https://doi.org/10.1111/j.1755-263X.2012.00250.x

Nuñez MA, Chiuffo MC, Torres A et al (2017) Ecology and management of invasive Pinaceae around the world: progress and challenges. Biol Invasions 19:3099–3120. https://doi.org/10.1007/s10530-017-1483-4

Odongkara K, Abila R, Onyango P (2005) Distribution of economic benefits from the fisheries of Lake Victoria. In: The state of the fisheries resources of Lake Victoria and their management. In: Proceedings of the regional stakeholders' conference. Lake Victoria Fisheries Organization Secretariat. Jinja, Uganda, pp 124–31

Oficialdegui FJ, Sánchez MI, Clavero M (2020) One century away from home: how the red swamp crayfish took over the world. Rev Fish Biol Fish 30:121–135. https://doi.org/10.1007/s11160-020-09594-z

Paini DR, Sheppard AW, Cook DC et al (2016) Global threat to agriculture from invasive species. Proc Natl Acad Sci 113:7575–7579. https://doi.org/10.1073/pnas.1602205113

Palmer S, Mercier OR, King-Hunt A (2020) Towards rangatiratanga in pest management? Māori perspectives and frameworks on novel biotechnologies in conservation. Pac Conserv Biol. https://doi.org/10.1071/PC20014

Parrott L (2017) The modelling spiral for solving 'wicked' environmental problems: guidance for stakeholder involvement and collaborative model development. Methods Ecol Evol 8:1005–1011. https://doi.org/10.1111/2041-210X.12757

Pascual MA, Lancelotti JL, Ernst B et al (2009) Scale, connectivity, and incentives in the introduction and management of non-native species: the case of exotic salmonids in Patagonia. Front Ecol Environ 7:533–540. https://doi.org/10.1890/070127

Pasko S, Goldberg J (2014) Review of harvest incentives to control invasive species. Manag Biol Invasions 5:263–277. https://doi.org/10.3391/mbi.2014.5.3.10

Pejchar L, Mooney HA (2009) Invasive species, ecosystem services and human well-being. Trends Ecol Evol 24:497–504. https://doi.org/10.1016/j.tree.2009.03.016

Pienkowski T, Williams S, McLaren K et al (2015) Alien invasions and livelihoods: economic benefits of invasive Australian Red Claw crayfish in Jamaica. Ecol Econ 112:68–77. https://doi.org/10.1016/j.ecolecon.2015.02.012

Potschin MB, Primmer E, Furman E, Haines-Young RH (2016) Have ecosystem services been oversold? A response to Silvertown. Trends Ecol Evol 31:334–335. https://doi.org/10.1016/j.tree.2016.03.008

Probert AF, Volery L, Kumschick S et al (2020) Understanding uncertainty in the Impact Classification for Alien Taxa (ICAT) assessments. NeoBiota 62:387. https://doi.org/10.3897/neobiota.62.52010

Quist MC, Hubert WA (2004) Bioinvasive species and the preservation of cutthroat trout in the western United States: ecological, social, and economic issues. Environ Sci Policy 7:303–313. https://doi.org/10.1016/j.envsci.2004.05.003

Raghu S, Anderson RC, Daehler CC et al (2006) Adding biofuels to the invasive species fire? Science 313:1742. https://doi.org/10.1126/science.1129313

Ricciardi A, Cohen J (2007) The invasiveness of an introduced species does not predict its impact. Biol Invasions 9:309–315. https://doi.org/10.1007/s10530-006-9034-4

Richardson DM, Rejmánek M (2011) Trees and shrubs as invasive alien species—a global review. Divers Distrib 17:788–809. https://doi.org/10.1111/j.1472-4642.2011.00782.x

Richardson DM, Ricciardi A (2013) Misleading criticisms of invasion science: a field guide. Divers Distrib 19:1461–1467. https://doi.org/10.1111/ddi.12150

RSA (2004) Republic of South Africa (RSA), 2004, 'National Environmental Management: Biodiversity Act 10 of 2004'. In: Proceedings of R47/Government Gazette No. 26887/20041008

Sarkar S, Minteer BA (2018) A sustainable philosophy: the work of Bryan Norton. Springer, Berlin

Schlaepfer MA, Sax DF, Olden JD (2011) The potential conservation value of non-native species. Conserv Biol 25:428–437. https://doi.org/10.1111/j.1523-1739.2010.01646.x

Schröter M, van Oudenhoven APE (2016) Ecosystem services go beyond money and markets: reply to Silvertown. Trends Ecol Evol 31:333–334. https://doi.org/10.1016/j.tree.2016.03.001

Schröter M, Van der Zanden EH, van Oudenhoven APE et al (2014) Ecosystem services as a contested concept: a synthesis of critique and counter-arguments. Conserv Lett 7:514–523. https://doi.org/10.1111/conl.12091

Shackleton CM, McGarry D, Fourie S et al (2007) Assessing the effects of invasive alien species on rural livelihoods: case examples and a framework from South Africa. Hum Ecol 35:113–127. https://doi.org/10.1007/s10745-006-9095-0

Shackleton RT, Adriaens T, Brundu G et al (2019a) Stakeholder engagement in the study and management of invasive alien species. J Environ Manag 229:88–101. https://doi.org/10.1016/j.jenvman.2018.04.044

Shackleton RT, Richardson DM, Shackleton CM et al (2019b) Explaining people's perceptions of invasive alien species: a conceptual framework. J Environ Manag 229:10–26. https://doi.org/10.1016/j.jenvman.2018.04.045

Shackleton RT, Shackleton CM, Kull CA (2019c) The role of invasive alien species in shaping local livelihoods and human well-being: a review. J Environ Manag 229:145–157. https://doi.org/10.1016/j.jenvman.2018.05.007

Shaka J (2013) Migingo Island: Kenyan or Ugandan Territory? J Conflictol 4:5. https://doi.org/10.7238/joc.v4i2.1886

Silvertown J (2015) Have ecosystem services been oversold? Trends Ecol Evol 30:641–648. https://doi.org/10.1016/j.tree.2015.08.007

Skonhoft A, Kourantidou M (2021) Managing a natural asset that is both a value and a nuisance: competition vs. cooperation for the Barents Sea Red King Crab. Mar Resour Econ. https://doi.org/10.1086/714416

Sladonja B, Poljuha D, Uzelac M (2018) Non-native invasive species as ecosystem service providers. In: Hufnagel L (ed) Ecosystem services and global ecology. IntechOpen, London, pp 39–59. https://doi.org/10.5772/intechopen.75057

Soto D, Jara F, Moreno C (2001) Escaped salmon in the inner seas, southern Chile: facing ecological and social conflicts. Ecol Appl 11:1750–1762. https://doi.org/10.1890/1051-0761(2001)011[1750:ESITIS]2.0.CO;2

Springborn M, Romagosa CM, Keller RP (2011) The value of nonindigenous species risk assessment in international trade. Ecol Econ 70:2145–2153. https://doi.org/10.1016/j.ecolecon.2011.06.016

Stromberg JC, Chew MK, Nagler PL, Glenn EP (2009) Changing perceptions of change: the role of scientists in Tamarix and river management. Restor Ecol 17:177–186. https://doi.org/10.1111/j.1526-100X.2008.00514.x

Temel J, Jones A, Jones N, Balint L (2018) Limits of monetization in protecting ecosystem services. Conserv Biol 32:1048–1062. https://doi.org/10.1111/cobi.13153

Tricarico E, Vilizzi L, Gherardi F, Copp GH (2010) Calibration of FI-ISK, an invasiveness screening tool for nonnative freshwater invertebrates. Risk Anal Int J 30:285–292. https://doi.org/10.1111/j.1539-6924.2009.01255.x

Tsehaye I, Catalano M, Sass G et al (2013) Prospects for fishery-induced collapse of invasive Asian carp in the Illinois River. Fisheries 38:445–454. https://doi.org/10.1080/03632415.2013.836501

van Kleunen M, Xu X, Yang Q et al (2020) Economic use of plants is key to their naturalization success. Nat Commun 11:1–12. https://doi.org/10.1038/s41467-020-16982-3

van Wilgen BW, Richardson DM (2014) Challenges and trade-offs in the management of invasive alien trees. Biol Invasions 16:721–734. https://doi.org/10.1007/s10530-013-0615-8

van Wilgen BW, Wilson JR (eds) (2018) The status of biological invasions and their management in South Africa. South African National Biodiversity Institute, Kirstenbosch and DST-NRF Centre of Excellence for Invasion Biology, Stellenbosch

Vann MG (2003) Of rats, rice, and race: the great hanoi rat massacre, an episode in French colonial history. French Colon Hist 4:191–203. https://doi.org/10.1353/fch.2003.0027

Varble S, Secchi S (2013) Human consumption as an invasive species management strategy. A preliminary assessment of the marketing potential of invasive Asian carp in the US. Appetite 65:58–67. https://doi.org/10.1016/j.appet.2013.01.022

Vaz AS, Kueffer C, Kull CA et al (2017) Integrating ecosystem services and disservices: insights from plant invasions. Ecosyst Serv 23:94–107. https://doi.org/10.1016/j.ecoser.2016.11.017

Vigliano PH, Alonso MF (2007) Salmonid introductions in Patagonia: a mixed blessing. In: Bert TM (ed) Ecological and genetic implications of aquaculture activities. Springer, Berlin, pp 315–331

Villarreal-Rosas J, Sonter LJ, Runting RK et al (2020) Advancing systematic conservation planning for ecosystem services. Trends Ecol Evol. https://doi.org/10.1016/j.tree.2020.08.016

Vimercati G, Kumschick S, Probert AF et al (2020) The importance of assessing positive and beneficial impacts of alien species. NeoBiota 62:525–545. https://doi.org/10.3897/neobiota.62.52793

Virtue JG, Bennett SJ, Randall RP (2004) Plant introductions in Australia: how can we resolve 'weedy'conflicts of interest. In: Sindel BM, Johnson SB (eds) Proceedings of the 14th Australian weeds conference, pp 42–48

Walker P (2013) Self-defeating regulation. Int'l Zeitschrift 9:31

Walsh JR, Carpenter SR, Vander Zanden MJ (2016) Invasive species triggers a massive loss of ecosystem services through a trophic cascade. Proc Natl Acad Sci 113:4081–4085. https://doi.org/10.1073/pnas.1600366113

Warren RJ II, King JR, Tarsa C et al (2017) A systematic review of context bias in invasion biology. PLoS ONE. https://doi.org/10.1371/journal.pone.0182502

Williams BK, Szaro RC, Shapiro CD (2009) Adaptive management: the US Department of the Interior technical guide. Adaptive Management Working Group. US Department of the Interior, Washington, DC

Wilson JRU (2020) Definitions can confuse: why the “neonative” neologism is bad for conservation. Bioscience 70:110–111. https://doi.org/10.1093/biosci/biz159

Wise RM, Van Wilgen BW, Le Maitre DC (2012) Costs, benefits and management options for an invasive alien tree species: the case of mesquite in the Northern Cape, South Africa. J Arid Environ 84:80–90. https://doi.org/10.1016/j.jaridenv.2012.03.001

Witte F, Goldschmidt T, Wanink J et al (1992) The destruction of an endemic species flock: quantitative data on the decline of the haplochromine cichlids of Lake Victoria. Environ Biol Fishes 34:1–28. https://doi.org/10.1007/BF00004782

Witte F, Kishe-Machumu MA, Mkumbo OC et al (2013) The fish fauna of Lake Victoria during a century of human induced perturbations. In: Snoeks J, Getahun A (eds) Proceedings of the fourth international conference on African Fish and Fisheries, pp 49–66

Woodford DJ, Richardson DM, MacIsaac HJ et al (2016) Confronting the wicked problem of managing biological invasions. https://doi.org/10.3897/neobiota.31.10038

Woodford DJ, Ivey P, Novoa A et al (2017) Managing conflict-generating invasive species in South Africa: challenges and trade-offs. Bothalia-African Biodivers Conserv 47:1–11. https://doi.org/10.4102/abc.v47i2.2160

Wyatt S (2018) Benefits and costs of the wilding pine management programme phase 2

Yongo E, Keizire BB, Mbilinyi HG (2005) Socio-economic impacts of trade. In: The state of the fisheries resources of Lake Victoria and their management. Proceedings of the regional stakeholders’ conference. Lake Victoria Fisheries Organization Secretariat, Jinja, Uganda, pp 124–31

Zivin J, Hueth B, Zilberman D (2000) Managing a multiple-use resource: the case of feral pig management in California rangeland. J Environ Econ Manag 39:189–204. https://doi.org/10.1006/jeem.1999.1101