Recent advancements in the risk screening of freshwater and terrestrial non-native species

Edited by

Daniela Giannetto, Marina Piria,
Ali Serhan Tarkan, Grzegorz Zięba

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EDITORIAL

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Advancing research on alien species and biological invasions

Editorial

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Species introductions are a major concern for ecosystem functioning, socio-economy and human well-being (Vilà et al. 2010; Lockwood et al. 2013; Diagne et al. 2021; Zenni et al. 2021). However, despite measures for prevention and control, a large number of non-native species have been identified in the last decades worldwide in both aquatic and terrestrial environments (IPBES 2019; Lowe et al. 2000; Guo et al. 2021). Although preventing introductions has proved to be the most effective management strategy (Wittenberg and Cock 2001; Pergl et al. 2016), extant non-native species are still expanding their distributional range and new non-native species are being recorded (Seebens et al. 2017). Non-native species introduced into new environments may represent a serious ecological and economical threat, especially if they spread rapidly in a new region and thus become invasive (Ricciardi et al. 2021; Cuthbert et al. 2021). Further, geographical areas that act as biodiversity hotspots with a high level of endemism are especially threatened by invasive species (Ribeiro and Leunda 2012). Hence, the identification of those non-native species that are likely to become invasive may be of crucial importance for the development of prevention measures, which can be achieved by risk screening studies (Adams and Lee 2012).

In the risk analysis process applied to non-native species (as defined in Copp et al. 2005), the first step is risk identification (a.k.a. risk screening), the second step is risk assessment, and the third step is risk management and communication (Canter 1993; UK Defra 2003). The risk screening of non-native species aims to identify which non-native species are likely to be invasive in a given risk assessment area, and the follow-up risk assessment for the highest risk species involves detailed examination of the likelihood and magnitude of risks of: (i) introduction (entry); (ii) establishment (of one or more self-sustaining populations); (iii) dispersal (more widely within the risk assessment area, i.e. so-called secondary spread or introductions); and (iv) impacts (to native biodiversity, ecosystem function and services, and the introduction and transmission of diseases) (see Vilizzi et al. 2022). Identification of potentially invasive species facilitates the development of policy and management procedures with regard to a specified risk assessment area to prevent and/or mitigate the impacts of biological invasions (Copp et al. 2016a).

Electronic decision-support tools for non-native species risk screening are becoming an essential component of government strategies to tackle non-native species invasions. The recent availability of user-friendly and widely deployable multilingual electronic tools (e.g. Copp et al. 2016b, 2021; Vilizzi et al. 2021) can facilitate early detection of potential threats, hence provide useful information to assist environmental managers and policy-makers in making decisions for the appropriate management and conservation of ecosystems. To this end, the Weed Risk Assessment (WRA: Pheloung et al. 1999) developed for terrestrial plants and later adapted to screening aquatic plants (Gordon et al. 2008) is a widely used decision-support tool. The WRA template inspired the '-ISK' (Invasiveness Screening Kit) family of decision-support tools developed for aquatic organisms (Copp et al. 2005; Copp 2013; Vilizzi et al. 2019), which were recently combined into the taxon-generic Aquatic Species Invasiveness Screening Kit (AS-ISK) to screen freshwater, brackish and marine aquatic organisms under current and future climatic conditions (Copp et al. 2016b, 2021; Vilizzi et al. 2021).

Despite the existence of the above-mentioned decision-support tools and a large number of published applications worldwide (e.g. Gordon et al. 2008; Vilizzi et al. 2019, 2021), there remain several knowledge gaps in the risk screening of non-native species with relevance to the following topics: (i) the relative dearth of information on the invasiveness of non-native aquatic species in taxonomic groups other than fishes and aquatic invertebrates; (ii) the paucity of risk screening studies focusing on biodiversity hotspots and/or tropical areas; (iii) the requirement for updated information on species invasiveness within a dynamic risk screening and comparative perspective; and (iv) the need for a taxon-generic decision-support toolkit for screening terrestrial animals and related applications.

All papers in this Special Issue were designed to address at least one of the research topics mentioned above so as to fill current knowledge gaps and provide novel information in the risk screening of freshwater and terrestrial non-native species.

Invasiveness of non-native aquatic plants and pathogens

The use of inconsistent and ambiguous terminology about invasive non-native species, together with the lack of focus on their potential impacts, limit understanding of their biology and role in the invaded ecosystems (Verbrugge et al. 2021). Insufficient understanding also causes a lack of public awareness and a consequent shortage of dedicated studies. **Fachinello et al. (2022)** emphasised this point by applying a scientometric approach to analyse academic documents on non-native plant species in Brazil published between 2002 and 2021. The authors found that only 13% of the 398 examined publications provided a clear definition of 'invasive species'. Of these publications, only 23.8% reported some type of damage caused by the invasive species and only 5% addressed economic or social damage. The authors also showed that only 17% of the publications proposed a method for control and/or mitigation of biological invasions and encouraged the use of further scientometric studies to guide future efforts to support more objective measures for management and decision-making.

There is still a lack of literature and relevant research on the distribution of non-native aquatic plants in some areas, despite their posing a serious threat to native macrophyte community composition by disrupting natural flow dynamics, depleting oxygen and altering food web structure and soil properties. To fill this knowledge gap and with the aim to help prioritisation measures for the proper management of non-native aquatic plants under projected climate conditions, **Piria et al. (2022)** identified and screened 10 extant and 14 aquatic plant species from a horizon scanning for their risk to become invasive in the Pannonian and Mediterranean regions of Croatia. The authors classified 90% and 60% of the extant aquatic plant species as carrying a high risk for the Pannonian and Mediterranean regions, respectively, under current and future climate conditions. Further, 42% of the species from a horizon scanning were classified as high-risk under current climatic conditions, but increased to 78% under a scenario of global warming.

Although most risk analyses in invasion biology have focused on the invasiveness of non-native species, some (dominant) native species can also pose a high risk of becoming invasive, especially under current global change. **Yazlık and Ambarlı (2022)** used an adaptation to Turkey's geographical and climatic conditions of the WRA decision-support tool to evaluate the risk of invasiveness of ten plant species (five non-native and five native) all known to be invasive in several parts of the world. Based on the resulting risk scores, all non-native species were classified as invasive and all native species as 'expanding' for Turkey. The outcomes of the study suggested that species can carry several risk-related traits resulting in high-risk scores irrespective of their origin. The authors also emphasised the importance of including dominant species with high environmental and socio-economic impacts in their habitats as part of priority lists for the implementation of management measures, hence irrespective of the species' origin (i.e. native or non-native).

Introductions of non-native species can drive disease emergence by extending the geographical range of associated parasites and pathogens (Foster et al. 2021), although limited research on this topic is available to date. The aquatic ornamental industry is one of the main introduction pathways for freshwater fish invasions, which can also act as a driver of disease emergence from associated parasites and pathogens by extending their geographical range (Chan et al. 2019). The increase in temperatures projected under future climate change scenarios is likely to increase the probability of survival and establishment of some commonly traded tropical and subtropical non-native ornamental fish species, even in geographical areas such as Northern Europe, which is currently not (yet) climatically suitable for their survival. **Guilder et al. (2022)** screened 24 of the 233 ornamental aquatic species (fishes and invertebrates) identified as traded in the UK for potential parasites and pathogens and reported a total of 155 of them of which the majority were platyhelminths, viruses and bacteria. Some potential parasites and pathogens currently absent from UK waters and with zoonotic potential were also identified, and their presence was highlighted in the context of understanding potential impacts in addition to the provision of evidence to inform risk assessment and mitigation approaches.

Biodiversity hotspots

Biological invasions are considered to be one of the most important threats to global biodiversity (Jeschke et al. 2022), particularly in biodiversity hotspots where non-native species may cause extensive damage to native species and ecosystems (Magalhães and Jacobi 2013). Preserving biodiversity and maintaining ecosystem function is of utmost importance not only in geographically large ecosystems but also in vulnerable biodiversity hotspots, which often host a large number of rare and/or endemic species. The South Caucasus represents one such biodiversity hotspot that includes the countries of Armenia, Azerbaijan and Georgia. **Mumladze et al. (2022)** screened 32 non-native extant and fish species from a horizon scanning for their risk of invasiveness under current and projected climate conditions in this risk assessment area. The number of very high-risk species increased from four (12.5%) under our current climate to 12 (37.5%) under projected climate conditions.

The Balkan Peninsula is also considered an important area for freshwater biodiversity due to the high number of endemic species (Hewitt 2011; Ćaleta et al. 2019). This region is particularly important for the high diversity of salmonid species that are being threatened by the introduction of non-native salmonids (Škraba Jurlina et al. 2020) and for which little is known about their potential risk of invasiveness, especially under predicted climate change conditions. **Marić et al. (2022)** screened 13 extant and four non-native salmonid species from a horizon scanning for their risk of becoming invasive in the Danube and Adriatic basins of four Balkan countries. Six (35%) of the screened species were ranked as high-risk under current climate conditions, although they decreased to three (17%) under projected conditions of global warming. Species ranked as medium-risk under current conditions were also medium-risk under future climate projections, although the relative risk score decreased. The authors concluded that global warming would influence salmonids and that only species with a wider temperature tolerance such as rainbow trout *Oncorhynchus mykiss* will likely prevail.

Comparative perspectives

One of the most important challenges in research (including risk screening studies) conducted simultaneously or repeated by several researchers to obtain reliable and reproducible results is to achieve the maximum possible compliance. A major challenge in risk assessment studies is to collect information on the overall severity and extent of consistency in responses, and empirical information on the factors influencing consistency across assessors is still not fully available. **Bernardo-Madrid et al. (2022)** quantified and compared the consistency in the scores of questions for impact assessment protocols with inter-rater reliability metrics. The authors provided an overview of impact assessment consistency and the factors altering it by evaluating 1,742 impact assessments of 60 terrestrials, freshwater and marine vertebrates, invertebrates and plants conducted using seven protocols applied in Europe. The authors reported that the great majority of assessments (67%) showed high consistency and only a small minority (13%) low consistency. Consistency of responses did not depend on species' identity or the amount of information on their impacts, but partly on the impact type evaluated and the protocol used.

Stable isotope analysis is commonly used to reconstruct species' feeding ecology and their trophic interactions within communities. Therefore, stable isotope analysis has been considered a sensitive and powerful tool to reveal competition and predation processes in food webs and used to quantify the ecological effects of non-native species (Sagouis et al. 2015). **Balzani and Haubrock (2022)** proposed the implementation of stable isotope analysis as an approach for assessment schemes to increase the accuracy in predicting invader impacts as well as the success of reintroductions and assisted migrations. The authors reviewed and discussed possibilities and limitations of using this method and suggested promising and useful applications for scientists and managers.

Development of a screening toolkit for terrestrial animals

Despite the availability of decision support tools for terrestrial animals, they are often in spreadsheet format which can make their usage time-consuming, if not counter-intuitive, to the end user. However, still there is no user-friendly, dialog-driven electronic decision-support tool, such as AS-ISK screening toolkit, available for terrestrial animals. Kopecký et al. (2019) remedied the lack of a dedicated screening tool using the AS-ISK as a 'surrogate' to screen terrestrial reptiles which highlighted need for its development. In this special issue, **Vilizzi et al. (2022)** described the development of a multilingual decision-support tool for screening terrestrial animals, namely the Terrestrial Animal Species Invasiveness Screening Kit (TAS-ISK). Based on the programming architecture of the AS-ISK and the questionnaire template common to the WRA-type toolkits, the TAS-ISK consists of 55 questions of which 49 deal with the species' biogeographical/historical traits and biological/ecological interactions and six are aimed to predicting the potential influence of climate change on the risks of introduction, establishment, dispersal and impact of the screened species. The authors also reported the results of nine trial screenings for each representative species in the main taxonomic groups of terrestrial animals supported by the toolkit: mammals, birds, reptiles, amphibians, annelids, insects, molluscs, nematodes, and platyhelminths.

Conclusions

Although the current research findings may not solve all identified shortcomings related to research in the risk screening of non-native species, all papers in this Special Issue have contributed to fill at least partially the existing gaps. The content of the Special Issue has helped to emphasise the importance not only of using appropriate nomenclature but also of a comprehensive approach to understanding the threat posed by non-native species and to multi-author risk screening studies. Alarming data have arisen on how many non-native species of aquatic plants could pose a threat to local communities, especially under projected conditions of global warming. These data are even more worrying considering the high potential invasiveness emerged also for some native plant species. At the same time, projected conditions of global warming may mitigate the invasiveness risk of some non-native species such as some salmonids that are not tolerant to high temperature fluctuations. The accidental spread of aquatic potential parasites and pathogens is also of concern and especially with regard to the fate of biodiversity hotspots. Finally, the proposal of novel approaches for assessment schemes based on different techniques such as stable isotope analysis together with the availability of the newly developed TAS-ISK decision-support tool for the risk screening of terrestrial animals, is expected to assist researchers and stakeholders and increase accuracy in predicting the impacts of biological invasions.

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References

Adams DC, Lee DJ (2012) Technology adoption and mitigation of invasive species damage and risk: Application to zebra mussels. Journal of Bioeconomics 14(1): 21–40. https://doi.org/10.1007/s10818-011-9117-x

Balzani P, Haubrock PJ (2022) Expanding the invasion toolbox: including stable isotope analysis in risk assessment. In: Giannetto D, Piria M, Tarkan AS, Zięba G (Eds) Recent advancements in the risk screening of freshwater and terrestrial non-native species. NeoBiota 76: 191–210. https://doi.org/10.3897/neobiota.76.77944

Bernardo-Madrid R, González-Moreno P, Gallardo B, Bacher S, Vilà M (2022) Consistency in impact assessments of invasive species is generally high and depends on protocols and impact types. In: Giannetto D, Piria M, Tarkan AS, Zięba G (Eds) Recent advancements in the risk screening of freshwater and terrestrial non-native species. NeoBiota 76: 163–190. https://doi.org/10.3897/neobiota.76.83028

Ćaleta M, Marčić Z, Buj I, Zanella D, Mustafić P, Duplić A, Horvatić S (2019) A Review of Extant Croatian Freshwater Fish and Lampreys. Croatian Journal of Fisheries: Ribarstvo 77(3): 137–234. https://doi.org/10.2478/cjf-2019-0016

Canter LW (1993) Pragmatic suggestions for incorporating risk assessment principles in EIA studies. Environment and Progress 15: 125–138.

Chan FT, Beatty SJ, Gilles Jr AS, Hill JE, Kozic S, Luo D, Morgan DL, Pavia Jr RTB, Therriault TW, Verreycken H, Vilizzi L, Wei H, Yeo DCJ, Zeng Y, Zięba G, Copp GH (2019) Leaving the fish bowl: The ornamental trade as a global vector for freshwater fish invasions. Aquatic Ecosystem Health & Management 22(4): 417–439. https://doi.org/10.1080/14634988.2019.1685849

Copp GH (2013) The Fish Invasiveness Screening Kit (FISK) for non-native freshwater fishes – a summary of current applications. Risk Analysis 33(8): 1394–1396. https://doi.org/10.1111/risa.12095

Copp GH, Garthwaite R, Gozlan RE (2005) Risk identification and assessment of non-native freshwater fishes: concepts and perspectives on protocols for the UK, Cefas Science Technical Report No. 129, Cefas, Lowestoft, 2005. www.cefas.defra.gov.uk/publications/techrep/tech129.pdf

Copp GH, Russell IC, Peeler EJ, Gherardi F, Tricarico E, MacLeod A, Cowx IG, Nunn AD, Occhipinti Ambrogi A, Savini D, Mumford JD, Britton JR (2016a) European Non-native Species in Aquaculture Risk Analysis Scheme – a summary of assessment protocols and decision making tools for use of alien species in aquaculture. Fisheries Management and Ecology 23(1): 1–11. https://doi.org/10.1111/fme.12074

Copp GH, Vilizzi L, Tidbury H, Stebbing PD, Tarkan AS, Miossec L, Goulletquer P (2016b) Development of a generic decision-support tool for identifying potentially invasive aquatic taxa: AS-ISK. Management of Biological Invasions 7(4): 343–350. https://doi.org/10.3391/mbi.2016.7.4.04

Copp GH, Vilizzi L, Wei H, Li S, Piria M, Al-Faisal AJ, Almeida D, Atique U, Al-Wazzan Z, Bakiu R, Bašić T, Bui TD, Canning-Clode J, Castro N, Chaichana R, Çoker T, Dashinov D, Ekmekçi FG, Erős T, Ferincz A, Ferreira T, Giannetto D, Gilles Jr AS, Głowacki Ł, Goulletquer P, Interesova E, Iqbal S, Jakubčinová K, Kanongdate K, Kim JE, Kopecký O, Kostov O, Koutsikos N, Kozic S, Kristan P, Kurita Y, Lee HG, Leuven RSEW, Lipinskaya T, Lukas J, Marchini A, González Martínez AI, Masson L, Memedemin D, Moghaddas SD, Monteiro J, Mumladze L, Naddafi R, Năvodaru I, Olsson KH, Onikura N, Paganelli D, Pavia Jr RT, Perdikaris C, Pickholtz R, Pietraszewski D, Povž M, Preda C, Ristovska M, Rosíková K, Santos JM, Semenchenko V, Senanan W, Simonović P, Smeti E, Števove B, Švolíková K, Ta KAT, Tarkan AS, Top N, Tricarico E, Uzunova E, Vardakas L, Verreycken H, Zięba G, Mendoza R (2021) Speaking their language – Development of a multilingual decision-support tool for communicating invasive species risks to decision makers and stakeholders. Environmental Modelling & Software 135: 104900. https://doi.org/10.1016/j.envsoft.2020.104900

Cuthbert RN, Pattison Z, Taylor NG, Verbrugge L, Diagne C, Ahmed DA, Leroy B, Angulo E, Briski E, Capinha C, Catford JA, Dalu T, Essl F, Gozlan RE, Haubrock PJ, Kourantidou M, Kramer AM, Renault D, Wasserman RJ, Courchamp F (2021) Global economic costs of aquatic invasive alien species. Science of the Total Environment 775: 145238. https://doi.org/10.1016/j.scitotenv.2021.145238

Defra UK (2003) Guidelines for Environmental Risk Assessment and Management. http://www.defra.gov.uk/environment/risk/eramguide/02.htm

Diagne C, Leroy B, Vaissiere AC, Gozlan RE, Roiz D, Jaric I, Salles JM, Bradshaw CJA, Courchamp F (2021) High and rising economic costs of biological invasions worldwide. Nature 592(7855): 571–576. https://doi.org/10.1038/s41586-021-03405-6

Fachinello MC, Romero JHC, Chiba de Castro WA (2022) Defining invasive species and demonstrating impacts of biological invasions: a scientometric analysis of studies on invasive alien plants in Brazil over the past 20 years. In: Giannetto D, Piria M, Tarkan AS, Zięba G (Eds) Recent advancements in the risk screening of freshwater and terrestrial non-native species. NeoBiota 76: 13–24. https://doi.org/10.3897/neobiota.76.85881

Foster R, Peeler E, Bojko J, Clark PF, Morritt D, Roy HE, Stebbing P, Tidbury HJ, Wood LE, Bass D (2021) Pathogens co-transported with invasive non-native aquatic species: Implications for risk analysis and legislation. NeoBiota 69: 79–102. https://doi.org/10.3897/neobiota.69.71358

Gordon DR, Onderdonk DA, Fox AM, Stocker RK (2008) Consistent accuracy of the Australian weed risk assessment system across varied geographies. Diversity & Distributions 14(2): 234–242. https://doi.org/10.1111/j.1472-4642.2007.00460.x

Guilder J, Copp GH, Thrush MA, Stinton N, Murphy D, Murray J, Tidbury HJ (2022) Threats to UK freshwaters under climate change: Commonly traded aquatic ornamental species and their potential pathogens and parasites. In: Giannetto D, Piria M, Tarkan AS, Zięba G (Eds) Recent advancements in the risk screening of freshwater and terrestrial non-native species. NeoBiota 76: 73–108. https://doi.org/10.3897/neobiota.76.80215

Guo Q, Cen X, Song R, McKinney ML, Wang D (2021) Worldwide effects of non-native species on species–area relationships. Conservation Biology 35(2): 711–721. https://doi.org/10.1111/cobi.13573

Hewitt GM (2011) Mediterranean peninsulas: the evolution of hotspots. In: Zachos F, Habel J (Eds) Biodiversity hotspots. Springer, Berlin, Heidelberg, 123–147. https://doi.org/10.1007/978-3-642-20992-5_7

IPBES (2019) Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. In: Brondizio ES, Settele J, Díaz S, Ngo HT (Eds) IPBES secretariat, Bonn, Germany, 1148 pp. https://doi.org/10.5281/zenodo.3831673

Jeschke JM, Liu C, Saul WC, Seebens H (2022) Biological Invasions: Introduction, Establishment and Spread. In: Mehner T, Tockner K (Eds) Encyclopedia of Inland Waters (2nd Edin.), Elsevier, 355–367. https://doi.org/10.1016/B978-0-12-819166-8.00033-5

Kopecký O, Bílková A, Hamatová V, Kňazovická D, Konrádová L, Kunzová B, Slaměníková J, Slanina O, Šmídová T, Zemancová T (2019) Potential invasion risk of pet traded lizards, snakes, crocodiles, and tuatara in the EU on the basis of a Risk Assessment Model (RAM) and Aquatic Species Invasiveness Screening Kit (AS-ISK). Diversity (Basel) 11(9): 164. https://doi.org/10.3390/d11090164

Lockwood JL, Hoopes MF, Marchetti MP (2013) Invasion ecology. 2nd Edn. Wiley, Chichester, 466 pp.

Lowe S, Browne M, Boudjelas S, De Poorter M (2000) 100 of the world's worst invasive alien species: a selection from the global invasive species database. The IUCN Invasive Species Specialist Group (ISSG), Auckland, New Zealand, 1–12.

Magalhães ALB, Jacobi CM (2013) Asian aquarium fishes in a Neotropical biodiversity hotspot: Impeding establishment, spread and impacts. Biological Invasions 15(10): 2157–2163. https://doi.org/10.1007/s10530-013-0443-x

Marić A, Špelić I, Radočaj T, Vidović Z, Kanjuh T, Vilizzi L, Piria M, Nikolić V, Škraba Jurlina D, Mrdak D, Simonović P (2022) Changing climate may mitigate the invasiveness risk of non-native salmonids in the Danube and Adriatic basins of the Balkan Peninsula (south-eastern Europe). In: Giannetto D, Piria M, Tarkan AS, Zięba G (Eds) Recent advancements in the risk screening of freshwater and terrestrial non-native species. NeoBiota 76: 135–161. https://doi.org/10.3897/neobiota.76.82964

Mumladze L, Kuljanishvili T, Japoshvili B, Epitashvili G, Kalous L, Vilizzi L, Piria M (2022) Risk of invasiveness of non-native fishes in the South Caucasus biodiversity and geopolitical hotspot. In: Giannetto D, Piria M, Tarkan AS, Zięba G (Eds) Recent advancements in the risk screening of freshwater and terrestrial non-native species. NeoBiota 76: 109–133. https://doi.org/10.3897/neobiota.76.82776

Pergl J, Sádlo J, Petrusek A, Laštůvka Z, Musil J, Perglová I, Šanda R, Šefrová H, Šíma J, Vohralík V, Pyšek P (2016) Black, Grey and Watch Lists of alien species in the Czech Republic based on environmental impacts and management strategy. NeoBiota 28: 1–37. https://doi.org/10.3897/neobiota.28.4824

Pheloung PC, Williams PA, Halloy SR (1999) A weed risk assessment model for use as a biosecurity tool evaluating plant introductions. Journal of Environmental Management 57(4): 239–251. https://doi.org/10.1006/jema.1999.0297

Piria M, Radočaj T, Vilizzi L, Britvec M (2022) Climate change may exacerbate the risk of invasiveness of non-native aquatic plants: the case of the Pannonian and Mediterranean regions of Croatia. In: Giannetto D, Piria M, Tarkan AS, Zięba G (Eds) Recent advancements in the risk screening of freshwater and terrestrial non-native species. NeoBiota 76: 25–52. https://doi.org/10.3897/neobiota.76.83320

Ribeiro F, Leunda PM (2012) Non-native fish impacts on Mediterranean freshwater ecosystems: Current knowledge and research needs. Fisheries Management and Ecology 19(2): 142–156. https://doi.org/10.1111/j.1365-2400.2011.00842.x

Ricciardi A, Iacarella JC, Aldridge DC, Blackburn TM, Carlton JT, Catford JA, Dick JTA, Hulme PE, Jeschke JE, Liebhold AM, Lockwood JL, MacIsaac HJ, Meyerson LA, Pyšek P, Richardson DM, Ruiz GM, Simberloff D, Vilà M, Wardle DA (2021) Four priority areas to advance invasion science in the face of rapid environmental change. Environmental Reviews 29(2): 119–141. https://doi.org/10.1139/er-2020-0088

Sagouis A, Cucherousset J, Villéger S, Santoul F, Boulêtreau S (2015) Non-native species modify the isotopic structure of freshwater fish communities across the globe. Ecography 38: 001–007. https://doi.org/10.1111/ecog.01348

Seebens H, Blackburn TM, Dyer EE, Genovesi P, Hulme PE, Jeschke JM, Pagad S, Pysek P, Winter M, Arianoutsou M, Bacher S, Blasius B, Brundu G, Capinha C, Celesti-Grapow L, Dawson W, Dullinger S, Fuentes N, Jager H, Kartesz J, Kenis M, Kreft H, Kuhn I, Lenzner B, Liebhold A, Mosena A, Moser D, Nishino M, Pearman D, Pergl J, Rabitsch W, Rojas-Sandoval J, Roques A, Rorke S, Rossinelli S, Roy HE, Scalera R, Schindler S, Stajerova K, Tokarska-Guzik B, van Kleunen M, Walker K, Weigelt P, Yamanaka T, Essl F (2017) No saturation in the accumulation of alien species worldwide. Nature Communications 8(1): e14435. https://doi.org/10.1038/ncomms14435

Škraba Jurlina D, Marić A, Mrdak D, Kanjuh T, Špelić I, Nikolić V, Piria M, Simonović P (2020) Alternative life-history in native trout (*Salmo* spp.) suppresses the invasive effect of alien trout strains introduced into streams in the western part of the Balkans. Frontiers in Ecology and Evolution 8: 188. https://doi.org/10.3389/fevo.2020.00188

Verbrugge LNH, Dawson MI, Gettys LA, Leuven RSEW, Marchante H, Marchante E, Nummi P, Rutenfrans AHM, Schneider K, Vanderhoeven S (2021) Novel tools and best practices for education about invasive alien species. Management of Biological Invasions 12(1): 8–24. https://doi.org/10.3391/mbi.2021.12.1.02

Vilà M, Basnou C, Pyšek P, Josefsson M, Genovesi P, Gollasch S, Nentwig W, Olenin S, Roques A, Roy D, Hulme PE (2010) How well do we understand the impacts of alien species on ecosystem services? A pan-European, cross-taxa assessment. Frontiers in Ecology and the Environment 8(3): 135–144. https://doi.org/10.1890/080083

Vilizzi L, Copp GH, Adamovich B, David A, Chan J, Davison PI, Dembski S, Ekmekçi FG, Ferincz Á, Forneck SC, Hill JE, Kim JE, Koutsikos N, Leuven RSEW, Luna SA, Magalhães F, Marr SM, Mendoza R, Mourão CF, Neal JW, Onikura N, Perdikaris C, Piria M, Poulet N, Puntila R, Range IL, Simonović P, Ribeiro F, Tarkan AS, Troca DFA, Vardakas L, Verreycken H, Vintsek L, Weyl OLF, Yeo DCJ, Zeng Y (2019) A global review and meta-analysis of applications of the freshwater Fish Invasiveness Screening Kit. Reviews in Fish Biology and Fisheries 29(3): 529–568. https://doi.org/10.1007/s11160-019-09562-2

Vilizzi L, Copp GH, Hill JE, Adamovich B, Aislabie L, Akin D, Al-Faisal AJ, Almeida D, Azmai MNA, Bakiu R, Bellati A, Bernier R, Bies JM, Bilge G, Branco P, Bui TD, Canning-Clode J, Cardoso Ramos HA, Castellanos-Galindo GA, Castro N, Chaichana R, Chainho P, Chan J, Cunico AM, Curd A, Dangchana P, Dashinov D, Davison PI, de Camargo MP, Dodd JA, Durland Donahou AL, Edsman L, Ekmekçi FG, Elphinstone-Davis J, Erős T, Evangelista C, Fenwick G, Ferincz Á, Ferreira T, Feunteun E, Filiz H, Forneck SC, Gajduchenko HS, Gama Monteiro J, Gestoso I, Giannetto D, Gilles Jr AS, Gizzi F, Glamuzina B, Glamuzina L, Goldsmit J, Gollasch S, Goulletquer P, Grabowska J, Harmer R, Haubrock PJ, He D, Hean JW, Herczeg G, Howland KL, İlhan A, Interesova E, Jakubčinová K, Jelmert A, Johnsen SI, Kakareko T, Kanongdate K, Killi N, Kim JE, Kırankaya ŞG, Kňazovická D, Kopecký O, Kostov V, Koutsikos N, Kozic S, Kuljanishvili T, Kumar B, Kumar L, Kurita Y, Kurtul I, Lazzaro L, Lee L, Lehtiniemi M, Leonardi G, Leuven RSEW, Li S, Lipinskaya T, Liu F, Lloyd L, Lorenzoni M, Luna SA, Lyons TJ, Magellan K, Malmstrøm M, Marchini A, Marr SM, Masson G, Masson L, McKenzie CH, Memedemin D, Mendoza R, Minchin D, Miossec L, Moghaddas SD, Moshobane MC, Mumladze L, Naddafi R, Najafi-Majd E, Năstase A, Năvodaru I, Neal JW, Nienhuis S, Nimtim M, Nolan ET, Occhipinti-Ambrogi A, Ojaveer H, Olenin S, Olsson K, Onikura N, O'Shaughnessy K, Paganelli D, Parretti P, Patoka J, Pavia Jr RTB, Pellitteri-Rosa D, Pelletier-Rousseau M, Peralta EM, Perdikaris C, Pietraszewski D, Piria M, Pitois S, Pompei L, Poulet N, Preda C, Puntila-Dodd R, Qashqaei AT, Radočaj T, Rahmani H, Raj S, Reeves D, Ristovska M, Rizevsky V, Robertson DR, Robertson P, Ruykys L, Saba AO, Santos JM, Sarı HM, Segurado P, Semenchenko V, Senanan W, Simard N, Simonović P, Skóra ME, Slovák Švolíková K, Smeti E, Šmídová T, Špelić I, Srėbalienė G, Stasolla G, Stebbing P, Števove B, Suresh VR, Szajbert B, Ta KAT, Tarkan AS, Tempesti J, Therriault TW, Tidbury HJ, Top-Karakuş N, Tricarico E, Troca DFA, Tsiamis K, Tuckett QM, Tutman P, Uyan U, Uzunova E, Vardakas L, Velle G, Verreycken H, Vintsek L, Wei H, Weiperth A, Weyl OLF, Winter ER, Włodarczyk R, Wood LE, Yang R, Yapıcı S, Yeo SSB, Yoğurtçuoğlu B, Yunnie ALE, Zhu Y, Zięba G, Žitňanová K, Clarke S (2021) A global-scale screening of non-native aquatic organisms to identify potentially invasive species under current and future climate conditions. Science of the Total Environment 788: 147868. https://doi.org/10.1016/j.scitotenv.2021.147868

Vilizzi L, Piria M, Pietraszewski D, Kopecký O, Špelić I, Radočaj T, Šprem N, Ta KAT, Tarkan AS, Weiperth A, Yoğurtçuoğlu B, Candan O, Herczeg G, Killi N, Lemić D, Szajbert B, Almeida D, Al-Wazzan Z, Atique U, Bakiu R, Chaichana R, Dashinov D, Ferincz Á, Flieller G, Gilles Jr AS, Goulletquer P, Interesova E, Iqbal S, Koyama A, Kristan P, Li S, Lukas J, Moghaddas SD, Monteiro JG, Mumladze L, Olsson KH, Paganelli D, Perdikaris C, Pickholtz R, Preda C, Ristovska M, Švolíková KS, Števove B, Uzunova E, Vardakas L, Verreycken H, Wei H, Zięba G (2022) Development and application of a multilingual electronic decision-support tool for risk screening non-native terrestrial animals under current and future climate conditions. In: Giannetto D, Piria M, Tarkan AS, Zięba G (Eds) Recent advancements in the risk screening of freshwater and terrestrial non-native species. NeoBiota 76: 211–236. https://doi.org/10.3897/neobiota.76.84268

Wittenberg R, Cock MJ (2001) Invasive alien species: a toolkit of best prevent ion and management practices. CABI Publishing, Wallingford, 240 pp. https://doi.org/10.1079/9780851995694.0000

Yazlık A, Ambarlı D (2022) Do non-native and dominant native species carry a similar risk of invasiveness? A case study for plants in Turkey. In: Giannetto D, Piria M, Tarkan AS, Zięba G (Eds) Recent advancements in the risk screening of freshwater and terrestrial non-native species. NeoBiota 76: 53–72. https://doi.org/10.3897/neobiota.76.85973

Zenni RD, Essl F, García-Berthou E, McDermott SM (2021) The economic costs of biological invasions around the world. NeoBiota 67: 1–9. https://doi.org/10.3897/neobiota.67.69971

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Advancing research on alien species and biological invasions

Defining invasive species and demonstrating impacts of biological invasions: a scientometric analysis of studies on invasive alien plants in Brazil over the past 20 years

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Abstract

Despite biological invasions being widely recognised as an important driver of environmental change, lack of consensus regarding the definition of invasive alien species (IAS) and vagueness around the demonstration of their impacts limits knowledge and research in this field. In this study, a scientometric approach was used to analyse academic documents published between 2002 and 2021 in three databases with reference to invasive alien plants in Brazil. Despite the growing body of scientific literature in the area, only 10% of the publications provided some definition of invasive species. Of the 398 publications analysed, 23.6% found some type of damage caused by the invader and, of these, only 5% addressed economic or social damage. Only 17% of the publications proposed a method for controlling and/or mitigating biological invasions. The absence of clear terminology and the lack of focus on impacts limits understanding of IAS of plants in Brazil. Based on the present findings, future studies on IAS of plants should move towards a consensus on the definition of biological invasion, as well as understand the impact caused by these species. In addition, it is recommended that further scientometric studies should guide future efforts to support objective measures for management and decision-making.

Keywords

biodiversity, impact, invasive alien species, management

Introduction

The fundamental property associated with biological invasions is the capacity of some invasive alien species (IAS) to expand and become established outside their native range (Richardson et al. 2000; Valéry et al. 2008). However, there are other important properties of biological invasions that are associated with the term 'impact' (Mack et al. 2000; Simberloff et al. 2005; Pyšek et al. 2012). This is because IAS can have detrimental effects not only on ecosystem function and services and on human well-being (Simberloff and Rejmánek 2011; Paini et al. 2016; Costa et al. 2019; Martinez-Cillero et al. 2019), but also on the economies of the invaded areas (Diagne et al. 2020; Zenni et al. 2021). However, despite wide (and implicit) acceptance in literature, controversy remains over the association between IAS and their negative impacts (Ricciardi and Ryan 2018; Sagoff 2018). Indeed, some studies have recently questioned this tenet suggesting that several factors, including lack of empirical evidence, set-up of poorly-executed experiments and over-emphasis of isolated cases, may bias proper understanding of the impacts of biological invasions (Davis and Chew 2017; Sagoff 2020). Additionally, other studies have questioned the science behind biological invasions and provided scientific evidence in support of this contention (Ricciardi and Ryan 2017; Boltovskoy et al. 2018). Furthermore, the absence of terminological consensus is another factor that has resulted in questioning the impacts of biological invasions (Simberloff 2012).

One of the key definitions of biological invasions was presented by Richardson et al. (2000), who introduced the concepts of introduction of IAS to the recipient area and their subsequent establishment by reproduction and expansion. However, the same authors emphasised that the definition of biological invasion should not be applied to species that cause environmental or economic impacts, but should be based exclusively on ecological and biogeographic criteria (Richardson et al. 2011). In contrast, Richardson and Pyšek (2012) stated that the most prominent invasive plant species were those that reached the highest abundance and had substantial impacts, thereby resulting in high costs to society. More recently, Pyšek et al. (2020) reiterated their definition of biological invasion (i.e. introduction, establishment and dispersion of an alien species), but also suggested that several negative impacts are associated with biological invasions. The same authors added that, according to the International Union for Conservation of Nature, only those alien species that have negative impacts should be classified as invasive. On the other hand, most studies that have either directly or indirectly addressed the topic of biological invasions have not presented a clear definition of 'invasiveness' and many of them have cited previously-published studies for evidence of the invasiveness of the species under study (Pereyra 2016).

The presence of IAS of plants in natural areas has been reported from different regions of the world and, in many cases, the consequences of biological invasions have been devastating (Justo et al. 2019). Ornamental use is recognised as the main route for the introduction of alien and potentially invasive plants to new regions (Hulme et al. 2017; Mayer et al. 2017). In recent years, the mechanisms behind the invasiveness of plant species in the recipient environment have been widely addressed by studies on plant invasions (Fridley 2010; Van Kleunen et al. 2010; Wang et al. 2020). However, there are still several gaps in knowledge about the status of invasive plant establishment in natural areas (Hulme 2018), their impacts (Foxcroft et al. 2019) and related measures for control (Weidlich et al. 2020) and these gaps are even more evident in tropical biomes (Ackerman et al. 2017; Pinto et al. 2020; Xavier et al. 2021).

Using a scientometric approach (Parra et al. 2019), academic articles published in the last 20 years were analysed in this study to understand the main trends and gaps in research on IAS of plants in Brazil. The specific objectives were to: (i) analyse the main methodologies used in the field of biological invasions; (ii) evaluate whether there is a clear and objective definition of IAS; (iii) identify the type of impact caused by IAS; and (iv) investigate methods for management and/or control of biological invasions.

Methods

Scientometrics is a new branch of science that measures and quantifies scientific progress via bibliometric indicators (Parra et al. 2019). Scientometric analysis seeks to observe trends and patterns in scientific production with predictive, prognostic and/or strategic approaches (Ivancheva 2008; Rizzi et al. 2014; Mills and Rahal 2019; Xie et al. 2020). In the context of biological invasions, this represents an important tool in the analysis of conservation issues in mega-diverse natural areas such as Brazilian biomes (Frehse et al. 2016). Scientometric studies play a leading role in surveying the state of the art of scientific literature (Santos et al. 2021), developing explanatory ecological models (Barbosa et al. 2012), forecasting the impacts of invasions in poorly-studied biomes (Pinto et al. 2020) and optimising resources and directing new research (Fonseca et al. 2021).

Following a scientometric approach, a survey of scientific literature was conducted in this study according to the PRISMA statement (Moher et al. 2009) by using two global databases and one regional database that have accessible online platforms, namely: Web of Science (WoS: www.webofknowledge.com), Scopus (www.scopus.com) and SciELO (Scientific Electronic Library Online: https://www.scielo.br/). In each database, search criteria were applied to all available fields (i.e. Title, Abstract, Keywords and other fields depending on the database) and only academic articles (hereafter, 'publications') published in the last 20 years (i.e. 2002–2021) were selected. The combinations of terms used to search the databases are shown in Table 1.

Both English and Portuguese terms were used in order to retrieve more results. The base of the words was retained and "*" was used as a wildcard to expand the search (Table 1). All collected references were compared to check for redundancies between the results of the three databases. After removing duplicate references, all remaining publications were evaluated using the following three inclusion criteria:

- Study carried out in Brazil;
- Study addressing invasive alien plants;
- Study not addressing agribusiness and/or monoculture.

Table 1. Combination of terms used to retrieve publications from the Web of Science, Scopus and SciELO online databases.

Topic	Term	Combination
Biological terminology	Biological invasion	
	Weed	
	Exotic	
	Non-native	
	Non-indigenous	
	Vegetable/Plant	(“Bio*invasion*”, “Invader*”, “Daninha”, “Exotic*”, “Alien”, “Non-native”, “Não nativa”, “Non-indigenous”, and “Weed”) AND (Impact*, Ecosystemic*, OR Economic*, OR Socio*) AND (Plant OR Vegetal) AND (Brasil OR Brazil)
Impact caused	Ecosystem	
	Economic	
	Social	
	Impact	
Location	Brazil	

The title and abstract of each publication were then evaluated and publications were divided into three groups:

1. Within the scope of the present study, i.e. whose title and/or abstract met the inclusion criteria;
2. Potentially within the scope of the present study, i.e. whose title and abstract partially met the inclusion criteria;
3. Not within the scope of the present study, i.e. whose title and abstract did not meet any of the inclusion criteria.

The body of the text was reviewed to explore the adequacy of the inclusion criteria for publications that were potentially within the scope of the present study (Group 2). As a result, all publications in Group 2 were reclassified (hence, redistributed) between Groups 1 or 3, as applicable. Publications in Group 3 were then excluded from the database and those in Group 1 were categorised according to the attributes listed in Table 2. In this step, the abstract of each publication was analysed. If the attribute information in the abstract was insufficient, then the entire publication was analysed.

Results

In total, 7,587 publications were retrieved from the three database searches. After applying the inclusion criteria, 578 publications were selected of which 348 were from Group 1 and 230 from Group 2. After reclassifying the publications in Group 2, 398 publications were obtained (i.e. final sample size), which were used in the scientometric analysis.

A growth trend was observed in the number of publications from 2002 to 2021. In 2002, there was only one publication, whereas there were 35 in 2021

Table 2. Scientometric attributes used in the data collection.

Attribute	Meaning	Category
Year	Publication date	2002–2021
Definition	Referenced definition of invasive alien species (IAS)	1. Definition not provide 2. Definition provided
Category	Main methodology used	1. List of species: field or bibliographic survey of alien species that occur in an area. 2. Ecology: study explored the biological attributes of the alien species and/or their relationship with the environment 3. Perception and ethnobiology: study explored the social perception of biological invasion 4. Scientometrics: statistically analysed publications/studies on plant invasions
Approach	Research environment	1. *In vitro* 2. *In situ* 3. *In silico* (i.e. an experiment performed on computer)
Impact	The study addressed the (negative) impact caused by the species' invasion	0. Did not address impact 1. Ecological (ecosystem damage) 2. Economic (financial loss) 3. Social
Control	The study tested or proposed some method for managing the invasion	1. Yes 2. No

and none in 2007 (Fig. 1A). In total, 36 publications (≈ 10% of the sample) were retrieved that provided some definition of IAS. Amongst these, four publications provided no references for the definition of IAS and 45.7% of those published in 2021 provided a definition (Fig. 1B). Regarding the category of the study, 342 publications (81.4% of the total) addressed the ecology of IAS, 42 (10.5%) provided species lists, 21 (5.2%) consisted of scientometric studies and 11 (2.7%) included studies on perception and ethnobiology. Between 2002 and 2005, only publications on the ecology of IAS were found. The first species list was recorded in 2006, the first scientometric approach in 2008 and the first study on perception in 2012 (Fig. 1C). Overall, 235 (59.2%) of the studies relied on *in situ* methodologies, 115 (28.8%) *in vitro* and 21 (10.3%) *in silico*, with an additional nine studies using both *in vitro* and *in situ* methodologies and with seven (1.8%) publications providing no information about the methodology. The first publication that used an *in silico* methodology was in 2008 (Fig. 1D). In total, 95 (23.6%) publications reported some negative influence caused by plant invasion. Of these, 90 (94.7%) addressed ecological impact, whereas economic impact, ecological and economic impact and ecological and social impact were reported by 4.2%, 1.0% and 1.0% of the publications, respectively (note that no studies addressed only social impact). The first record of a publication that addressed economic impact was from 2017 and of one that addressed social impact from 2018 (Fig. 1E). Of the 398 publications analysed, only 69 mentioned measures for the management of IAS, representing 17.3% of the studies considered (Fig. 1F).

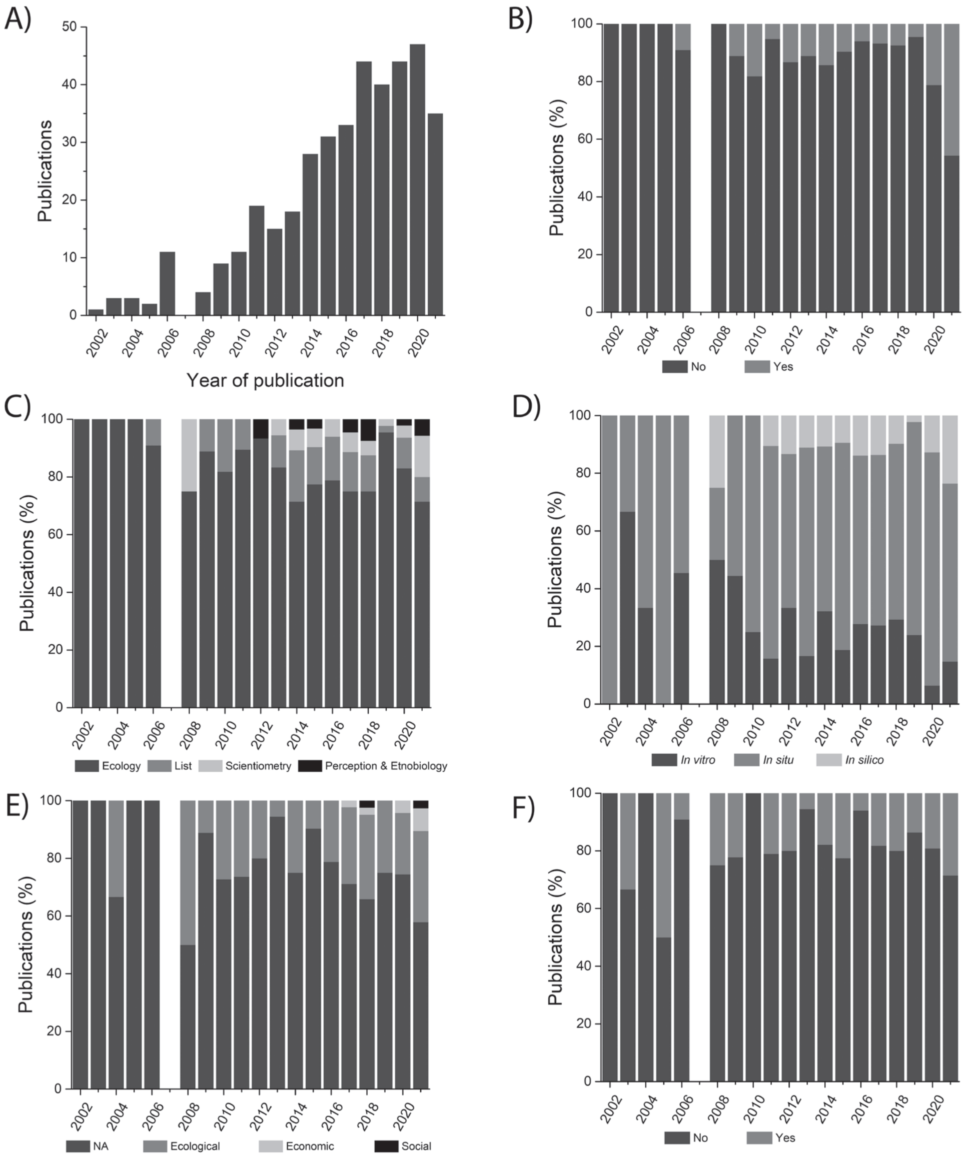

Figure 1. Charts showing the trend in publications (academic articles) on invasive alien species (IAS) of plants in Brazil from 2002 to 2021 as retrieved from the Web of Science, Scopus and SciELO databases (see Table 2) **A** number of publications retrieved per year **B** proportion of publications providing a definition of IAS **C** proportion of categories of publications **D** proportion of methodological approaches (i.e. study environment) **E** proportion of publications that addressed some type of impact **F** proportion of publications presenting measures of management.

Discussion

The methodology adopted in the present study for identifying academic articles by evaluating both title and abstract and, in case of doubt, using the body of the text, has been used in recent environmental scientometric studies (Moral-Muñoz et al. 2020; Pinto et al. 2020; Fonseca et al. 2021). Considering the time range and methodology used, the number of academic documents analysed in the present study has provided a representative sample in this field of research compared to previous similar studies. Thus, Dias et al. (2013) reviewed all articles published in scientific journals (databases not specified) that used the terms "invasive exotic species" or "biological invasion" in "Brazil" before 2012, obtaining 124 publications. Frehse et al. (2016) used WoS to identify studies on biological invasions in Brazil published until 2014, resulting in 354 publications. Romero (2020) searched for publications on plant IAS in Latin American countries between 1945 and 2019, obtaining 373 publications for Brazil.

In the last 20 years, the number of academic articles that have addressed IAS of plants has grown substantially, highlighting the importance of the topic in plant conservation (Paclibar and Tadiosa 2019). The present results indicate that research in Brazil has focused on basic aspects of invasions (> 81% of the publications surveyed), such as biological knowledge of the invasive plants and their ecological relationships with the invaded area. Thus, despite growing knowledge on the number of alien plant species in Brazil (Frehse et al. 2016), there is a lack of research on their establishment, dispersion or impacts (Pinto et al. 2020). Consequently, *in situ* approaches are used because this method provides more accurate perspectives on the behaviour of IAS of plants (Barbet-Massin et al. 2018).

The way in which plant invasions have been studied has also changed. In the last six years, a slight increase in the number of *in silico* studies has been observed. This approach has been used in studies that focus on scientometrics and invasive species. However, the small number of surveys and scientometric studies associated with the limited use of the *in silico* approach indicates a lack of studies on plant invasion management (Zenni et al. 2016) and on the investigation of tropical environments. These approaches are critical for supporting conservation planning in tropical areas (Barbosa et al. 2012). Furthermore, there are gaps in the prediction of the number of alien species (Seebens et al. 2020). Given the need to understand the state of the art of the science of biological invasions and to advance research on the management of IAS, an increase is predicted in the number of publications using the *in silico* approach in the coming years.

Despite explaining biological invasion through invasion attributes (i.e. phenotypic plasticity, allelopathic compounds, invasiveness and invasibility), almost 87% of Brazilian studies presented no clear definition of biological invasion. However, these observations are not restricted to articles from Brazil. Indeed, Pereyra (2016) stated that, between 2011 and 2012, only 13% of academic articles provided a definition of 'invasive species' in the main international scientific journals. Moreover, the lack of consensus on terminology creates doubts regarding the meaning of biological invasion, including when to consider a given situation as an invasion (Moro et al. 2012; Simberloff 2012). Often, terms such as 'invasive' and 'invader', which may sound more attractive than 'exotic' or 'introduced', are used out of correct context (Pereyra 2016). This makes accurate communication of biological invasions difficult, produces mixed results and provides scope for the use of the term 'invader' without a definition or ecological verification. Furthermore, owing to the lack of consensus about terminology within the scientific community and the ecological complexity surrounding biological invasions, society is less aware of this global threat when compared to most other threats that cause biodiversity loss (i.e. agricultural expansion, climate change, hunting: Courchamp et al. 2017). For this reason, it is recommended that future studies should focus on providing a definition of biological invasion in a clear and objective way, i.e. based on scientifically validated literature and/or ecological evidence.

In the present study, only 23.8% of the publications retrieved addressed an impact, which demonstrates a lack of Brazilian studies exploring the consequences of biological invasion. Most studies addressing impacts from IAS in Brazil have focused on losses in agricultural production (Adelino et al. 2021). The lack of relevant data on the impacts of IAS in natural areas creates barriers to effective management (Diagne et al. 2020) and this is especially important given that 'demonstrating' rather than 'assuming' invasiveness/impact is a crucial aspect in the risk analysis of IAS (Vilizzi et al. 2022).

Of the publications that addressed impact(s), only 5% (1% of the total) reported economic impacts caused by invasive species. Recently, Adelino et al. (2021) reported losses of USD 105.53 billion attributed to the biological invasion by only 16 species in Brazil. However, if the current number of invasive species in the country is considered, then the real economic loss due to such invasions might be much higher than published figures. One of the consequences of this lack of studies on the economic impacts of invasions includes low investment in the management of IAS (Zenni et al. 2021). Only 17.3% of the studies retrieved suggested control alternatives for IAS. Hence, despite the increase in the number of studies on the biological and ecological aspects of IAS of plants in Brazil, several gaps remain in the study of the management of these invasions. Lack of awareness about the importance of this topic, difficulties associated with the planning, coordination and organisation of competent public bodies and an inherent lack of research aimed at the control of IAS in developing countries (Weidlich et al. 2020) are major challenges for the management of IAS of plants in Brazil. Therefore, future research in this country should also consider the impacts of IAS of plants with a view to controlling them. Finally, *in silico* studies or studies evaluating IAS metadata in Brazil would allow the identification of trends or behaviour of invasive species in tropical environments and the identification of those with greater impact.

In conclusion, despite the growing body of scientific literature regarding IAS of plants in Brazil, the lack of data on the impact of IAS and the lack of consensus on the definition of invasion limit current understanding of the topic. This has direct implications for the recognition of its importance in natural areas and indirect implications on the understanding of the consequences of biological invasions for society. Consequently, for successful management actions against IAS, this lack of consensus represents an even worse impediment. Additionally, only few scientific documents have addressed or mentioned IAS control or management. For this reason, scientometric studies should be conducted to understand more comprehensively IAS in Brazil so as to provide guidelines for future research. Overall, it is recommended that future studies on IAS of plants should: (i) clarify and establish a consensus on the definition of biological invasion; (ii) understand the negative effects of such invasions using diverse methodologies (i.e. *in situ*, *in vitro* and *in silico* methods); and (iii) identify objective measures for the mitigation and control of the threat posed by biological invasions on biodiversity and ecosystem services.

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References

Ackerman JD, Tremblay RL, Rojas-Sandoval J, Hernández-Figueroa E (2017) Biotic resistance in the tropics: Patterns of seed plant invasions within an island. Biological Invasions 19(1): 315–328. https://doi.org/10.1007/s10530-016-1281-4

Adelino JRP, Heringer G, Diagne C, Courchamp F, Faria LDB, Zenni RD (2021) The economic costs of biological invasions in Brazil: A first assessment. NeoBiota 67: 349–374. https://doi.org/10.3897/neobiota.67.59185

Barbet-Massin M, Rome Q, Villemant C, Courchamp F (2018) Can species distribution models really predict the expansion of invasive species? PLoS ONE 13(3): e0193085. https://doi.org/10.1371/journal.pone.0193085

Barbosa FG, Schneck F, Melo AS (2012) Use of ecological niche models to predict the distribution of invasive species: A scientometric analysis. Brazilian Journal of Biology 72(4): 821–829. https://doi.org/10.1590/S1519-69842012000500007

Boltovskoy D, Sylvester F, Paolucci EM (2018) Invasive species denialism: sorting out facts, beliefs, and definitions. Ecology and Evolution 8: 11190–11198. https://doi.org/10.1002/ece3.4588

Costa RO, Jose CM, Grombone-Guaratini MT, Matos DMS (2019) Chemical characterization and phytotoxicity of the essential oil from the invasive *Hedychium coronarium* on seeds of Brazilian riparian trees. Flora 257: e151411. https://doi.org/10.1016/j.flora.2019.05.010

Courchamp F, Fournier A, Bellard C, Bertelsmeier C, Bonnaud E, Jeschke JM, Russell JC (2017) Invasion Biology: Specific Problems and Possible Solutions. Trends in Ecology & Evolution 32(1): 13–22. https://doi.org/10.1016/j.tree.2016.11.001

Davis MA, Chew MK (2017) "The Denialists Are Coming!" Well, Not Exactly: A Response to Russell and Blackburn. Trends in Ecology & Evolution 32: 229–230. https://doi.org/10.1016/j.tree.2017.02.008

Diagne C, Leroy B, Gozlan RE, Vaissière AC, Assailly C, Nuninger L, Courchamp F (2020) InvaCost, a public database of the economic costs of biological invasions worldwide. Scientific Data 7(1): 277. https://doi.org/10.1038/s41597-020-00586-z

Dias J, da Fonte MA, Baptista R, Mantoani MC, Holdefer DR, Torezan JMD (2013) Invasive alien plants in Brazil: A nonrestrictive revision of academic works. Natureza & Conservação 11(1): 31–35. https://doi.org/10.4322/natcon.2013.004

Fonseca CR, Paterno GB, Guadagnin DL, Venticinque EM, Overbeck GE, Ganade G, Weisser WW (2021) Conservation biology: Four decades of problem- and solution-based research. Perspectives in Ecology and Conservation 19(2): 121–130. https://doi.org/10.1016/j.pecon.2021.03.003

Foxcroft LC, Spear D, van Wilgen NJ, McGeoch MA (2019) Assessing the association between pathways of alien plant invaders and their impacts in protected areas. NeoBiota 4: 1–25. https://doi.org/10.3897/neobiota.43.29644

Frehse FDA, Braga RR, Nocera GA, Vitule JRS (2016) Non-native species and invasion biology in a megadiverse country: Scientometric analysis and ecological interactions in Brazil. Biological Invasions 18(12): 3713–3725. https://doi.org/10.1007/s10530-016-1260-9

Fridley JD (2010) Biodiversity as a bulwark against invasion: conceptual threads since Elton. In: Richardson DM (Ed.) Fifty years of invasion ecology: the legacy of Charles Elton. Wiley-Blackwell, Oxford), 121–130. https://doi.org/10.1002/9781444329988.ch10

Hulme PE (2018) Protected land: Threat of invasive species. Science 361(6402): 561–562. https://doi.org/10.1126/science.aau3784

Hulme PE, Brundu G, Carboni M, Dehnen-Schmutz K, Dullinger S, Early R, Verbrugge LNH (2017) Integrating invasive species policies across ornamental horticulture supply chains to prevent plant invasions. Journal of Applied Ecology 55(1): 92–98. https://doi.org/10.1111/1365-2664.12953

Ivancheva L (2008) Scientometrics today: A methodological overview. Collnet Journal of Scientometrics and Information Management 2(2): 47–56. https://doi.org/10.1080/09737766.2008.10700853

Justo FM, Hofmann GS, Almerão MP (2019) Espécies exóticas invasoras em unidades de conservação na região sul do Brasil. Revista de Ciências Ambientais 13: 57–76. https://revistas.unilasalle.edu.br/index.php/Rbca/article/view/6233

Mack RN, Simberloff D, Mark Lonsdale W, Evans H, Clout M, Bazzaz FA (2000) Biotic invasions: Causes, epidemiology, global consequences, and control. Ecological Applications 10(3): 689–710. https://doi.org/10.1890/1051-0761(2000)010[0689:BICEGC]2.0.CO;2

Martinez-Cillero R, Willcock S, Perez-Diaz A, Joslin E, Vergeer P, Peh KSH (2019) A practical tool for assessing ecosystem services enhancement and degradation associated with invasive alien species. Ecology and Evolution 9(7): 3918–3936. https://doi.org/10.1002/ece3.5020

Mayer K, Haeuser E, Dawson W, Essl F, Kreft H, Pergl J, van Kleunen M (2017) Naturalization of ornamental plant species in public green spaces and private gardens. Biological Invasions 19(12): 3613–3627. https://doi.org/10.1007/s10530-017-1594-y

Mills MC, Rahal C (2019) A scientometric review of genome-wide association studies. Communications Biology 2(1): 1–9. https://doi.org/10.1038/s42003-018-0261-x

Moher D, Liberati A, Tetzlaff J, Altman DG, The PRISMA Group (2009) Preferred reporting items for systematic reviews and meta-analyses: The PRISMA statement. PLoS Medicine 6(7): e1000097. https://doi.org/10.1371/journal.pmed.1000097

Moral-Muñoz JA, Herrera-Viedma E, Santisteban-Espejo EA, Cobo MJ (2020) Software tools for conducting bibliometric analysis in science: An up-to-date review. El Profesional de la Información 29(1): e290103. https://doi.org/10.3145/epi.2020.ene.03

Moro MF, Souza VC, Oliveira-Filho ATD, Queiroz LPD, Fraga CND, Rodal MJN, Martins FR (2012) Alienígenas na sala: O que fazer com espécies exóticas em trabalhos de taxonomia, florística e fitossociologia? Acta Botanica Brasílica 26(4): 991–999. https://doi.org/10.1590/S0102-33062012000400029

Paclibar GCB, Tadiosa ER (2019) Ecological niche modeling of invasive alien plant species in a protected landscape. Global Journal of Environmental Science and Management 5: 371–382. https://doi.org/10.22034/GJESM.2019.03.09

Paini DR, Sheppard AW, Cook DC, De Barro PJ, Worner SP, Thomas MB (2016) Global threat to agriculture from invasive species. Proceedings of the National Academy of Sciences of the United States of America 113(27): 7575–7579. https://doi.org/10.1073/pnas.1602205113

Parra MR, Coutinho RX, Pessano EFC (2019) Um breve olhar sobre a cienciometria: Origem, evolução, tendências e sua contribuição para o ensino de ciências. Revista Contexto & Educação 34(107): 126–141. https://doi.org/10.21527/2179-1309.2019.107.126-141

Pereyra PJ (2016) Revisiting the use of the invasive species concept: An empirical approach. Austral Ecology 41(5): 519–528. https://doi.org/10.1111/aec.12340

Pinto AS, Monteiro FKS, Ramos MB, Araújo RCC, Lopes SF (2020) Invasive plants in the Brazilian Caatinga: A scientometric analysis with prospects for conservation. Neotropical Biology and Conservation 15(4): 503–520. https://doi.org/10.3897/neotropical.15.e57403

Pyšek P, Jarošík V, Hulme PE, Pergl J, Hejda M, Schaffner U, Vilà M (2012) A global assessment of invasive plant impacts on resident species, communities and ecosystems: The interaction of impact measures, invading species' traits and environment. Global Change Biology 18(5): 1725–1737. https://doi.org/10.1111/j.1365-2486.2011.02636.x

Pyšek P, Hulme PE, Simberloff D, Bacher S, Blackburn TM, Carlton JT, Richardson DM (2020) Scientists' warning on invasive alien species. Biological Reviews of the Cambridge Philosophical Society 95(6): 1511–1534. https://doi.org/10.1111/brv.12627

Ricciardi A, Ryan R (2017) The exponential growth of invasive species denialism. Biological Invasions 20(3): 549–553. https://doi.org/10.1007/s10530-017-1561-7

Ricciardi A, Ryan R (2018) Invasive species denialism revisited: Response to Sagoff. Biological Invasions 20(10): 2731–2738. https://doi.org/10.1007/s10530-018-1753-9

Richardson DM, Pyšek P (2012) Naturalization of introduced plants: Ecological drivers of biogeographical patterns. The New Phytologist 196(2): 383–396. https://doi.org/10.1111/j.1469-8137.2012.04292.x

Richardson DM, Pyšek P, Rejmánek M, Barbour MG, Panetta FD, West CJ (2000) Naturalization and invasion of alien plants: Concepts and definitions. Diversity & Distributions 6(2): 93–107. https://doi.org/10.1046/j.1472-4642.2000.00083.x

Richardson DM, Carruthers J, Hui C, Impson FAC, Miller JT, Robertson MP, Wilson JRU (2011) Human-mediated introductions of Australian acacias - a global experiment in biogeography. Diversity & Distributions 17(5): 771–787. https://doi.org/10.1111/j.1472-4642.2011.00824.x

Rizzi F, Van Eck JN, Frey M (2014) The production of scientific knowledge on renewable energies: Worldwide trends, dynamics and challenges and implications for management. Renewable Energy 62: 657–671. https://doi.org/10.1016/j.renene.2013.08.030

Romero JHC (2020) Plantas invasoras na América Latina: avanços, direções e desafios. PHD Thesis. Federal University of São Carlos (São Carlos).

Sagoff M (2018) Invasive species denialism: A reply to Ricciardi and Ryan. Biological Invasions 20(10): 2723–2729. https://doi.org/10.1007/s10530-018-1752-x

Sagoff M (2020) Fact and value in invasion biology. Conservation Biology 34(3): 581–588. https://doi.org/10.1111/cobi.13440

Santos LVR, Camilo JPG, Oliveira CYBD, Nader C, Oliveira CDL (2021) Current status of Brazilian scientific production on non-native species. Ethology Ecology and Evolution 34(2): 187–200. https://doi.org/10.1080/03949370.2020.1870570

Seebens H, Bacher S, Blackburn TM, Capinha C, Dawson W, Dullinger S, Essl F (2020) Projecting the continental accumulation of alien species through to 2050. Global Change Biology 27: 968–969. https://doi.org/10.1111/gcb.15333

Simberloff D (2012) Nature, natives, nativism, and management: Worldviews underlying controversies in invasion biology. Environmental Ethics 34(1): 5–25. https://doi.org/10.5840/enviroethics20123413

Simberloff D, Rejmánek M (2011) Encyclopedia of Biological Invasions. University of California Press, Berkeley, 1–379.

Simberloff D, Parker IM, Windle PN (2005) Introduced Species Policy, Management, and Future Research Needs. Frontiers in Ecology and the Environment 3(1): 12–20. https://doi.org/10.1890/1540-9295(2005)003[0012:ISPMAF]2.0.CO;2

Valéry L, Fritz H, Lefeuvre JC, Simberloff D (2008) In search of a real definition of the biological invasion phenomenon itself. Biological Invasions 10(8): 1345–1351. https://doi.org/10.1007/s10530-007-9209-7

Van Kleunen M, Dawson W, Schlaepfer D, Jeschke JM, Fischer M (2010) Are invaders different? A conceptual framework of comparative approaches for assessing determinants of invasiveness. Ecology Letters 13: 947–958. https://doi.org/10.1111/j.1461-0248.2010.01503.x

Vilizzi L, Hill JE, Piria M, Copp GH (2022) A protocol for screening potentially invasive non-native species using Weed Risk Assessment-type decision-support tools. Science of the Total Environment 832: 154966. https://doi.org/10.1016/j.scitotenv.2022.154966

Wang C, Wei M, Wang S, Wu B, Cheng H (2020) *Erigeron annuus* (L.) Pers. and *Solidago canadensis* L. antagonistically affect community stability and community invasibility under the co-invasion condition. Science of the Total Environment 716: e137128. https://doi.org/10.1016/j.scitotenv.2020.137128

Weidlich EWA, Flórido FG, Sorrini TB, Brancalion P (2020) Controlling invasive plant species in ecological restoration: A global review. Journal of Applied Ecology 57(9): 1806–1817. https://doi.org/10.1111/1365-2664.13656

Xavier CN, Granato-Souza D, Barbosa ACMC, da Silva JRM (2021) Tropical dendrochronology applied to invasive tree species in the Brazilian Atlantic Forest. Journal of Forestry Research 32(1): 91–101. https://doi.org/10.1007/s11676-019-01075-9

Xie H, Zhang Y, Choi Y, Li F (2020) A scientometrics review on land ecosystem service research. Sustainability 12(7): 2959. https://doi.org/10.3390/su12072959

Zenni RD, Dechoum M, Ziller SR (2016) Dez anos do informe brasileiro sobre espécies exóticas invasoras: Avanços, lacunas e direções futuras. Biotemas 29(1): 133–153. https://doi.org/10.5007/2175-7925.2016v29n1p133

Zenni RD, Essl F, García-Berthou E, McDermott SM (2021) The economic costs of biological invasions around the world. NeoBiota 67: 1–9. https://doi.org/10.3897/neobiota.67.69971

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Climate change may exacerbate the risk of invasiveness of non-native aquatic plants: the case of the Pannonian and Mediterranean regions of Croatia

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Abstract

Non-native aquatic plants are amongst the major threats to freshwater biodiversity and climate change is expected to facilitate their further spread and invasiveness. To date, in Croatia, no complete list of non-native extant and horizon aquatic plants has been compiled nor has a risk screening been performed. To address this knowledge gap, 10 extant and 14 horizon aquatic plant species were screened for their risk of invasiveness in the Pannonian and Mediterranean regions of Croatia under current and predicted (future) climate conditions. Overall, 90% and 60% of the extant species were classified as high risk for the Pannonian and Mediterranean regions, respectively, under both climate scenarios. Of the horizon species, 42% were classified as high risk under current conditions and, under climate change, this proportion increased to 78%. The 'top invasive' species (i.e. scored as very high risk) under both climate conditions and for both regions were extant *Elodea nuttallii* and horizon *Lemna aequinoctialis*. The horizon *Hygrophila polysperma* was very high risk for the Mediterranean Region under current climate conditions and for both regions under projected climate conditions. *Azolla filiculoides*, *Elodea canadensis*, *Egeria densa* and *Utricularia gibba* were also classified as high risk under current climate conditions and, after accounting for climate change, they became of very high risk in both regions. Further, *Gymnocoronis spilanthoides* and *Lemna minuta* were found to pose a very high risk under climate change only for the Pannonian Region. It is anticipated that the outcomes of this study will contribute to knowledge of the invasiveness of aquatic plants in different climatic regions and enable prioritisation measures for their control/eradication.

Keywords
Adriatic Sea Basin, AS-ISK, Black Sea Basin, freshwater, risk screening

Introduction

Invasive non-native species pose globally one of the most serious environmental threats due to their adverse impacts on the environment (Blackburn et al. 2014; Essl et al. 2019; Rendeková et al. 2019) and the resulting multiple socio-economic implications (Lovell et al. 2006; Bacher et al. 2018). Freshwater ecosystems are especially vulnerable to the introduction of invasive non-native species, which occurs via several pathways and vectors linked to a large variety of human activities (Banha and Anastácio 2015; Coughlan et al. 2017; Rodriguez-Merino et al. 2018). In the case of non-native aquatic plants, the main introduction vectors include ship fouling, hitchhiking, fish stocking, floods and other natural events, host and vector organisms, the ornamental trade and aquarium waste releases (e.g. Leung et al. 2006; Pollux et al. 2006; Dehnen-Schmutz and Touza 2008; Hussner et al. 2010; Reynolds et al. 2015). Once established, non-native aquatic plants may alter habitat condition and ecosystem function (Rodriguez-Merino et al. 2018), as well as food-web structure (Villamagna and Murphy 2010), increase the risk of flooding events by impeding river flow (Thouvenot et al. 2013), induce oxygen depletion (Caraco et al. 2006), disrupt ecosystem properties such as soil cover, nutrient cycling, fire regimes and hydrology (Weidlich et al. 2020) and change macroinvertebrate and fish species richness and abundance (Strayer 2010). In addition, non-native aquatic plants mainly reproduce and spread by vegetative propagation (Eckert et al. 2016; Crane et al. 2019), which facilitates their transportation by water currents to new water bodies (Hussner et al. 2017).

In the last 100 years, the number and abundance of non-native aquatic plants has considerably increased worldwide (Hussner et al. 2010). This increase has been mainly caused by enhanced trading, higher water turbidity by eutrophication/re-oligotrophication and by climatic factors mostly related to temperature increase (Hussner et al. 2010; Rodríguez-Merino et al. 2018). Yet, the increased threat posed by new introductions of non-native aquatic plants may still be prevented, or at least mitigated, using horizon scanning as an early-warning tool, which helps to identify (potentially) invasive non-native species that are not yet established within some geographical area often of high conservation value (Copp et al. 2007; Amanatidou et al. 2012; Roy et al. 2015).

In Croatia, the first comprehensive list of aquatic plants, including many rare and threatened species, together with information on their historical and recent distribution, was recently produced from herbarium museum sheets and includes 76 species, of which three are non-native, namely *Azolla filiculoides*, *Egeria densa* and *Elodea canadensis* (Zeko et al. 2020). *Elodea canadensis* had already been introduced to Croatia in the 19th century, specifically 60 years before its first official record in 1954, whereas its congener *Elodea nuttallii* was first recorded in 2006 in the drainage channels of the Kopački Rit Nature Park in the Pannonian Region (Black Sea Basin) of Croatia (Kočić et al.

2014; Nikolić 2022). *Elodea nuttallii* is included in the Invasive Alien Species List of European Union concern (EU 2014) together with *Myriophyllum heterophyllum*, which was discovered in 2000 on Krk Island in the Mediterranean Region (Adriatic Sea Basin) of Croatia and with *Ludwigia peploides*, which was found in 2018 in the Pannonian Region (Jasprica et al. 2017; Buzjak and Sedlar 2018). Finally, *Najas graminea* also was discovered recently in the Mediterranean Region (Glasnović et al. 2015).

In addition to the above, extensive monitoring research on non-native species, including aquatic plants, has been conducted in recent years in Croatia (Kutleša et al. 2021). Based on this monitoring programme, 20 aquatic and semi-aquatic non-native plant species have been documented in the Pannonian Region, whereas the Mediterranean Region remains unexplored. As a result, to date no complete list of non-native aquatic plants has been compiled for Croatia nor a risk screening for their invasiveness in the country has been performed. To address this knowledge gap, the aims of this study were: (i) to identify extant non-native aquatic plant species in Croatia and perform a horizon scanning to find which species might enter Croatia in the future from neighbouring countries; and (ii) to evaluate the risk of invasiveness of the identified extant and horizon species under both current and predicted (future) climate conditions in the Pannonian and Mediterranean regions of Croatia, which belong to two different climate zones. It is anticipated that the outcomes of this study will support the prioritisation of future management measures for introduced non-native aquatic plants in Croatia and will help in the identification of the highest-risk species likely to invade Croatian aquatic ecosystems in the near future with the aim to establish rapid control/eradication measures.

Materials and methods

Study area

Croatia is biogeographically divided into the lowland Pannonian Region, the Mediterranean Region (along the Adriatic coast and in its immediate hinterland) and the highland Alpine area (in the elevated Lika and Gorski Kotar). Hydrologically, the Pannonian Region (a.k.a. Pannonian Plain or Hungarian Lowland) includes the Danube River Basin, which is dominated by the large rivers Danube, Drava and Sava, and the karst Mediterranean region, which includes the Adriatic Sea Basin with its immediate and confined basins (Potočki et al. 2021). The karst rivers of the immediate river basins of the Mediterranean Region have direct confluence into the Adriatic Sea, whereas confined basins represent the highland region with karst rivers (mostly intermittent) without direct confluence to the Adriatic Sea (Fig. 1A). The rivers of the Mediterranean Region are short and isolated and often flow through deep canyons, where they create waterfalls and lakes (cf. lentic expansions). These rivers have a seasonal hydrological regime with abundance of water in autumn and spring, but with some of them drying out completely in summer (Bonacci and Andrić 2008; Bonacci and Roje-Bonacci 2012; Bonacci et al. 2014).

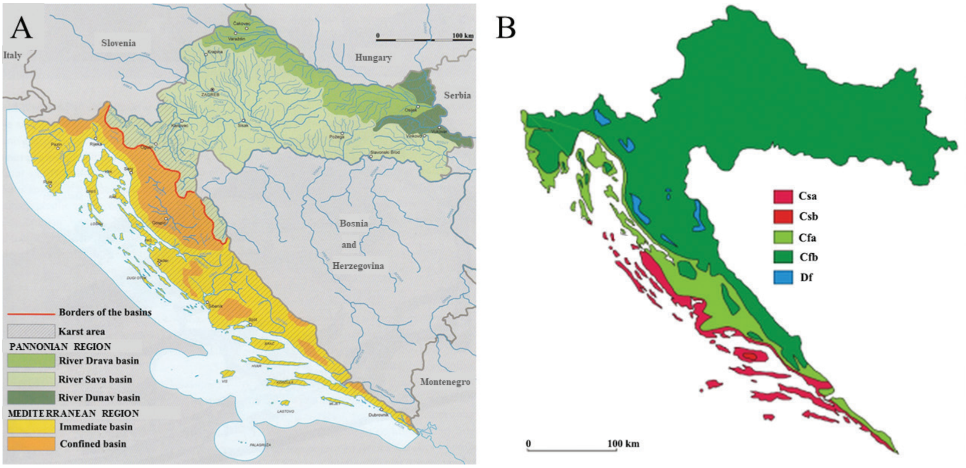

Figure 1. A map of the Pannonian and Mediterranean regions of Croatia representing the two risk assessment areas for the screening of non-native aquatic plants (see Table I) **B** geographical distribution of the climate types in Croatia (according to the Köppen-Geiger climate classification system): *Csa* = warm-temperate with dry and hot summer; *Csb* = warm-temperate with dry and warm summer; *Cfa* = warm-temperate, fully humid, hot summer; *Cfb* = warm-temperate, fully humid, warm summer; *Df* = boreal humid.

Croatia has mostly a temperate rainy climate with average monthly temperature higher than –3 °C and lower than 18 °C in the coldest month. In the Pannonian Region, the warmest month of the year has an average temperature lower than 22 °C, whereas in the Mediterranean Region, it is higher than 22 °C and more than four months in a year have a monthly average temperature higher than 10 °C (Zaninović et al. 2008). According to the Köppen–Geiger climate classification system (Peel et al. 2007), in the Pannonian Region, the *Cfb* climate type (warm-temperate, fully humid, warm summer) prevails, whereas at higher altitudes, this is true of the *Df* climate type (boreal humid). In the Mediterranean Region, the *Cfa* (warm-temperate, fully humid, hot summer) and *Cfb* climate types prevail in the North, whereas the islands and coastal areas of the middle and southern Adriatic Sea are characterised by the *Csa* (warm-temperate with dry and hot summer) and *Csb* (warm-temperate with dry and warm summer) climate types. Finally, the inland area and nearby shores are mostly *Cfa*, changing further from the coast into *Cfb* and *Df* at the highest altitudes (Fig. 1B).

Risk screening

In total, 24 non-native aquatic plant species were selected for risk screening of their potential invasiveness in the Pannonian and Mediterranean regions of Croatia – hereafter, also referred to as the 'risk assessment areas'. The scientific names, authority and more frequently used common names for the screened species are listed in Table 1 and plant nomenclature follows The Plant List (http://www.theplantlist.org/), World Flora Online

Table 1. Extant and horizon non-native aquatic plant species screened for their potential risk of invasiveness in the Pannonian and Mediterranean regions of Croatia. For each species, the Region of establishment (M = Mediterranean; P = Pannonian) is provided together with the *a priori* categorisation outcome into Non-invasive and Invasive (after Vilizzi et al. 2022). GISD = Global Invasive Species Database (www.iucngisd.org); CABI = Centre for Agriculture and Bioscience International Invasive Species Compendium (www.cabi.org/ISC); IESNA = the Invasive and Exotic Species of North America list (www.invasive.org); Gscholar = Google Scholar literature search (whenever applicable). N = no impact/threat; Y = impact/threat; '–' = absent; n.e. = not evaluated (but present in database); n.a. = not applicable.

Species name	*A priori* categorisation						
	Common name	Region	GISD	CABI	IESNA	GScholar	Outcome
Extant							
Azolla cristata Kaulf.	–	P	–	Y	–	n.a.	Invasive
Azolla filiculoides Lam.	Pacific mosquitofern	P	–	Y	–	n.a.	Invasive
Egeria densa Planch.	Brazilian waterweed	M	Y	Y	Y	n.a.	Invasive
Elodea canadensis Michx.	Canadian waterweed	P	Y	Y	–	n.a.	Invasive
Elodea nuttallii (Planch.) H.St John	western waterweed	P	–	Y	–	n.a.	Invasive
Ludwigia peploides (Kunth) P.H.Raven	floating primrose-willow	P	–	Y	–	n.a.	Invasive
Myriophyllum heterophyllum Michx.	twoleaf watermilfoil	M	Y	Y	Y	n.a.	Invasive
Najas graminea Delile	ricefield waternymph	M	–	n.e.	–	N	Non-invasive
Nymphaea candida C.Presl	–	M	–	n.e.	–	N	Non-invasive
Pistia stratiotes L.	water lettuce	P	Y	Y	–	n.a.	Invasive
Horizon							
Cabomba caroliniana A.Gray	Carolina fanwort	–	Y	Y	–	n.a.	Invasive
Gymnocoronis spilanthoides (D.Don ex Hook. & Arn.) DC.	Senegal tea plant	–	Y	Y	–	n.a.	Invasive
Hygrophila polysperma (Roxb.) T.Anderson	Indian swampweed	–	Y	Y	–	n.a.	Invasive
Lemna aequinoctialis Welw.	lesser duckweed	–	–	N	–	N	Non-invasive
Lemna minuta Kunth	least duckweed	–	–	Y	–	n.a.	Invasive
Lemna turionifera Landolt	turion duckweed	–	–	–	–	N	Non-invasive
Najas guadalupensis (Spreng.) Magnus	southern waternymph	–	–	n.e.	–	N	Non-invasive
Nelumbo nucifera Gaertn.	sacred lotus	–	–	n.e.	–	N	Non-invasive
Nymphaea lotus L.	white Egyptian lotus	–	–	N	–	N	Non-invasive
Rotala macrandra Koehne	–	–	–	N	–	N	Non-invasive
Rotala rotundifolia (Buch.-Ham. ex Roxb.) Koehne	dwarf rotala	–	–	–	Y	n.a.	Invasive
Sagittaria subulata (L.) Buchenau	awl-leaf arrowhead	–	–	–	–	N	Non-invasive
Utricularia gibba L.	humped bladderwort	–	Y	N	–	n.a.	Invasive
Vallisneria australis S.W.L.Jacobs & Les	–	–	–	–	–	N	Non-invasive

(http://www.worldfloraonline.org/) and The PLANTS Database (https://plants.usda.gov/home). Selection of emergent, submergent or floating aquatic plants was based on two criteria: (i) extant species, i.e. already present in both risk assessment areas (n = 10) and identified using the MINGOR 2022 (https://invazivnevrste.haop.hr/katalog) and Flora Croatica (http://hirc.botanic.hr/fcd) databases; and (ii) horizon species, i.e. likely to enter the risk assessment areas in the near future (n = 14) and identified with the aid of the Centre for Agriculture and Bioscience International (CABI) scanning tool (www.cabi.org/horizonscanningtool). Screenings were conducted independently by authors TR, MP and MB for both risk assessment areas, with each assessor screening a subset of the total number of species (i.e. nine, eight and seven species, respectively). Notably, the screening of a set of species for a certain risk assessment area by more than one independent assessor has been shown to provide no advantage in terms of increased level of confidence compared to screenings carried out by the same assessors on subsets of the total number of species. This is the approach followed in the present study, which has important implications in terms of allocation of resources and cost-benefit analysis (combination 1IS: Vilizzi et al. 2022).

Risk identification was undertaken using the Aquatic Species Invasiveness Screening Kit (AS-ISK: Copp et al. 2016, 2021), which is available for free download at www.cefas.co.uk/nns/tools/). This taxon-generic decision-support tool complies with the 'minimum standards' for screening non-native species under EC Regulation No. 1143/2014 on the prevention and management of the introduction and spread of invasive alien species (EU 2014). The AS-ISK consists of 55 questions: the first 49 questions comprise the Basic Risk Assessment (BRA) and address the biogeography/invasion history and biology/ecology of the species; the last six questions comprise the Climate Change Assessment (CCA) and require the assessor to predict how future predicted climatic conditions are likely to affect the BRA with respect to risks of introduction, establishment, dispersal and impact. The BRA questions consist of two sections with eight categories: Section A of Biogeography/Invasion History including the categories Domestication/Cultivation (C1), Climate, distribution and introduction risk (C2) and Invasive elsewhere (C3); Section B of Biology/Ecology including the categories Undesirable (or persistence) traits (C4), Resource exploitation (C5), Reproduction (C6), Dispersal mechanisms (C7) and Tolerance attributes (C8); Section C of Climate change including the (implicit) category Climate change (C9) (see Suppl. material 1: Table S1).

To achieve a valid screening, the assessor must provide for each question a response, a confidence level for the response (see below) and a justification, based on literature sources. The outcomes are a BRA score and a (composite) BRA+CCA score, which is obtained after adding or subtracting up to 12 points to the BRA score or leaving it unchanged in case of a CCA score equal to 0. Scores < 1 suggest that the species poses a 'low risk' to become invasive in the risk assessment area, whereas scores ≥ 1 indicate a 'medium risk' or a 'high risk'. The threshold (Thr) value to distinguish between medium-risk (BRA and BRA+CCA score < Thr) and high-risk (BRA and BRA+CCA score ≥ Thr) species for the risk assessment area is obtained by 'calibration', based on the Receiver Operating Characteristic (ROC) curve analysis (see Vilizzi et al. 2022). A measure of the accuracy of the calibration analysis is the area under the curve (AUC) whose values are interpreted as: $0.7 \leq$ AUC < 0.8 = acceptable discriminatory power, $0.8 \leq$ AUC < 0.9 = excellent, $0.9 \leq$ AUC = outstanding (Hosmer et al. 2013). For the species classified as high risk, a distinction was made in this study of the 'very high risk' species, based on an *ad hoc* threshold, weighted according to the range of high-risk score values obtained for the BRA and BRA+CCA. Identification of the (very) high-risk species is useful to prioritise allocation of resources in view of a full risk assessment (Copp et al. 2016). This examines in detail the risks of: (i) introduction (entry); (ii) establishment (of one or more self-sustaining populations); (iii) dispersal (more widely within the risk assessment area, i.e. so-called secondary spread or introductions); and (iv) impacts (to native biodiversity, ecosystem function and services and the introduction and transmission of diseases).

For ROC curve analysis to be implemented, the species selected for screening must be categorised *a priori* as 'non-invasive' or 'invasive' using literature sources. The *a priori* categorisation of species was implemented as per Vilizzi et al. (2022) (Table 1). Confidence levels in the responses to questions in the AS-ISK are ranked using a 1–4 scale and, based on the confidence level (CL) allocated to each response, a confidence factor (CF) is obtained as:

$$\mathrm{CF} = \sum(\mathrm{CL}_{\mathrm{Q}i})/(4 \times 55) \ (i = 1, \ldots, 55)$$

where $\mathrm{CL}_{\mathrm{Q}i}$ is the CL for Q*i*, 4 is the maximum achievable value for confidence (i.e. very high: see above) and 55 is the total number of questions comprising the AS-ISK questionnaire (Vilizzi et al. 2022). The CF ranges from a minimum of 0.25 (i.e. all 55 questions with confidence level equal to 1) to a maximum of 1 (i.e. all 55 questions with confidence level equal to 4). Based on all 55 Qs of the AS-ISK questionnaire, the 49 Qs comprising the BRA and the six Qs comprising the CCA, the $\mathrm{CF}_{\mathrm{Total}}$, $\mathrm{CF}_{\mathrm{BRA}}$ and $\mathrm{CF}_{\mathrm{CCA}}$ are respectively computed.

Implementation of ROC curve analysis followed the protocol described in Vilizzi et al. (2022), with the true/false positive/negative outcome distinction not applied to the medium-risk species, as they can be either included or not into a full (comprehensive) risk assessment depending on priority and/or availability of financial resources. Following ROC analysis, the best threshold value that maximises the true positive rate and minimises the false positive rate was determined using Youden's *J* statistic, whereas the 'default' threshold of 1 was set to distinguish between low-risk and medium-risk species. Fitting of the ROC curve was with package pROC (Robin et al. 2011) for R ×64 v.4.0.5 (R Core Team 2021) using 2000 bootstrap replicates for the confidence intervals of specificities, which were computed along the entire range of sensitivity points (i.e. 0 to 1, at 0.1 intervals). Differences in mean BRA and BRA+CCA scores between risk assessment areas (Pannonian Region, Mediterranean Region) were statistically tested with permutational ANOVA, based on a one-factor design. Differences in CF between risk assessment areas, components (BRA, BRA+CCA) and species status (Extant, Horizon) were also tested with permutational ANOVA, based on a nested-factorial design with factors risk assessment area and Component crossed and factor Status nested within Risk assessment area × Component and with all factors fixed. Analysis was implemented in PERMANOVA+ for PRIMER v.7, with normalisation of the data and using a Bray-Curtis dissimilarity measure, 9999 permutations of the raw data (unrestricted in case of the factorial-nested design) and with statistical effects evaluated at $\alpha = 0.05$.

Results

For the Pannonian Region: the BRA scores ranged from 5.5 to 41.0, with mean = 23.6, median = 22.8 and 5% and 95% CI = 6.3 and 39.9; the BRA+CCA scores ranged from 5.5 to 53.0, with mean = 31.5, median = 32.5 and 5% and 95% CI = 8.4 and 51.9. For the Mediterranean region: the BRA scores ranged from 6.0 to 41.0, with mean = 25.5, median = 32.3 and 5% and 95% CI (confidence interval) = 6.7 and 40.0; the BRA+CCA scores ranged from 6.0 to 53.0, with mean = 32.5, median = 35.8 and 5% and 95% CI = 8.4 and 51.0. There were no differences in the mean BRA scores for the Pannonian and Mediterranean regions ($F^{\#}_{1,46} = 0.245$, $P = 0.640$; # = permutational value) nor in the mean BRA+CCA scores ($F^{\#}_{1,46} = 0.055$, $P = 0.816$).

Risk outcomes

The ROC curve for the Pannonian Region resulted in an AUC of 0.8357 (0.6410–1.0000 95% CI) and for the Mediterranean Region in an AUC of 0.8679 (0.6864–1.0000 95% CI). Both AUCs had, therefore, excellent discriminatory power, hence were able to classify reliably non-invasive and invasive aquatic plant species for the two risk assessment areas. Youden's *J* provided the thresholds of 22.75 and 24.75 for the Pannonian and Mediterranean regions, respectively. These thresholds were used for calibration of the risk outcomes to distinguish between medium-risk and high-risk species (combined AS-ISK report in Suppl. material 2).

For the Pannonian Region (Table 2):

- Based on the BRA outcome scores: 12 (50.0%) species were classified as high risk and 12 (50.0%) as medium risk. Amongst the 14 species categorised *a priori* as invasive, eleven were true positives and amongst the 10 species categorised *a priori* as non-invasive, one was a false positive. Of the 12 medium-risk species, nine were *a priori* non-invasive and three invasive.
- Based on the BRA+CCA outcome scores (hence, after accounting for climate change predictions): 18 (75.0%) species were classified as high risk and six (25.0%) as medium risk. Amongst the *a priori* invasive species, twelve were true positives and amongst the *a priori* non-invasive species, six were false positive. Of the six medium-risk species, four were *a priori* non-invasive and two invasive.

Table 2. Risk outcomes for the non-native aquatic plant species screened with the Aquatic Species Invasiveness Screening Kit (AS-ISK) for the Pannonian and Mediterranean regions of Croatia. For each species, the following information is provided: *a priori* categorisation of invasiveness (N = non-invasive; Y = invasive: see Table 1); BRA and BRA+CCA scores with corresponding risk outcomes (M = Medium; H = High; VH = Very high, based on an *ad hoc* threshold equal to 40: see text for details) and classification (Class: FP = false positive; TP = true positive; '–' = not applicable as medium-risk: see text for details); difference (Delta) between BRA+CCA and BRA scores; confidence factor (CF) for all 55 questions of the AS-ISK (CF_{Total}), for the 49 BRA questions (CF_{BRA}) and for the six CCA questions (CF_{CCA}). Risk outcomes are based on the thresholds (Thr) of 22.75 for the Pannonian Region and 24.75 for the Mediterranean Region. Risk outcomes for the BRA are computed as: M, within the interval [1, Thr, H [Thr, 40[and VH [40, 68]. Risk outcomes for the BRA+CCA are computed as: M [1, Thr, H [Thr, 40[and VH [40, 68] (note the reverse bracket notation indicating in all cases an open interval).

Species name	*A priori*	BRA			BRA+CCA				CF		
		Score	Outcome	Class	Score	Outcome	Class	Delta	Total	BRA	CCA
Pannonian Region											
Azolla cristata	Y	33.0	H	TP	45.0	VH	TP	12.0	0.53	0.51	0.67
Azolla filiculoides	Y	30.0	H	TP	42.0	VH	TP	12.0	0.66	0.67	0.63
Cabomba caroliniana	Y	23.0	H	TP	23.0	H	TP	0.0	0.48	0.48	0.50
Egeria densa	Y	36.0	H	TP	48.0	VH	TP	12.0	0.57	0.58	0.50
Elodea canadensis	Y	39.0	H	TP	51.0	VH	TP	12.0	0.68	0.68	0.63
Elodea nuttallii	Y	41.0	VH	TP	53.0	VH	TP	12.0	0.68	0.67	0.75
Gymnocoronis spilanthoides	Y	28.0	H	TP	40.0	VH	TP	12.0	0.68	0.65	0.92
Hygrophila polysperma	Y	35.5	H	TP	47.5	VH	TP	12.0	0.72	0.73	0.63
Lemna aequinoctialis	N	40.0	VH	FP	52.0	VH	FP	12.0	0.72	0.72	0.71
Lemna minuta	Y	33.0	H	TP	43.0	VH	TP	10.0	0.71	0.73	0.50
Lemna turionifera	N	21.0	M	–	27.0	H	FP	6.0	0.71	0.73	0.58
Ludwigia peploides	Y	22.0	M	–	22.0	M	–	0.0	0.47	0.46	0.58
Myriophyllum heterophyllum	Y	24.0	H	TP	34.0	H	TP	10.0	0.53	0.52	0.58
Najas graminea	N	11.5	M	–	11.5	M	–	0.0	0.42	0.41	0.50
Najas guadalupensis	N	17.0	M	–	29.0	H	FP	12.0	0.48	0.48	0.50
Nelumbo nucifera	N	19.0	M	–	31.0	H	FP	12.0	0.54	0.57	0.29
Nymphaea candida	N	5.5	M	–	5.5	M	–	0.0	0.40	0.38	0.58
Nymphaea lotus	N	14.5	M	–	26.5	H	FP	12.0	0.68	0.69	0.58
Pistia stratiotes	Y	15.0	M	–	23.0	H	TP	8.0	0.53	0.53	0.54
Rotala macrandra	N	8.0	M	–	8.0	M	–	0.0	0.41	0.40	0.50
Rotala rotundifolia	Y	14.0	M	–	14.0	M	–	0.0	0.50	0.51	0.50
Sagittaria subulata	N	6.0	M	–	6.0	M	–	0.0	0.37	0.36	0.50
Utricularia gibba	Y	28.5	H	TP	40.5	VH	TP	12.0	0.65	0.66	0.54
Vallisneria australis	N	22.5	M	–	34.5	H	FP	12.0	0.61	0.61	0.63
Mediterranean Region											
Azolla cristata	Y	32.0	H	TP	44.0	VH	TP	12.0	0.52	0.51	0.63
Azolla filiculoides	Y	39.0	H	TP	51.0	VH	TP	12.0	0.67	0.67	0.63
Cabomba caroliniana	Y	29.5	H	TP	39.5	H	TP	10.0	0.62	0.63	0.58
Egeria densa	Y	36.0	H	TP	48.0	VH	TP	12.0	0.57	0.58	0.50
Elodea canadensis	Y	39.0	H	TP	51.0	VH	TP	12.0	0.68	0.69	0.63
Elodea nuttallii	Y	41.0	VH	TP	53.0	VH	TP	12.0	0.68	0.67	0.75
Gymnocoronis spilanthoides	Y	28.0	H	TP	38.0	H	TP	10.0	0.65	0.63	0.75

Species name	A priori	BRA			BRA+CCA				CF		
		Score	Outcome	Class	Score	Outcome	Class	Delta	Total	BRA	CCA
Hygrophila polysperma	Y	40.0	VH	TP	44.0	VH	TP	4.0	0.69	0.70	0.58
Lemna aequinoctialis	N	40.0	VH	FP	48.0	VH	FP	8.0	0.70	0.72	0.46
Lemna minuta	Y	33.0	H	TP	33.0	H	TP	0.0	0.75	0.78	0.50
Lemna turionifera	N	20.0	M	–	26.0	H	FP	6.0	0.70	0.72	0.54
Ludwigia peploides	Y	26.5	H	TP	36.5	H	TP	10.0	0.52	0.52	0.54
Myriophyllum heterophyllum	Y	27.5	H	TP	37.5	H	TP	10.0	0.53	0.52	0.58
Najas graminea	N	14.5	M	–	14.5	M	–	0.0	0.45	0.44	0.50
Najas guadalupensis	N	13.5	M	–	25.5	H	FP	12.0	0.46	0.44	0.58
Nelumbo nucifera	N	23.0	M	–	35.0	H	FP	12.0	0.54	0.57	0.29
Nymphaea candida	N	6.5	M	–	10.5	M	–	4.0	0.37	0.37	0.38
Nymphaea lotus	N	14.5	M	–	24.5	M	–	10.0	0.67	0.69	0.46
Pistia stratiotes	Y	18.0	M	–	18.0	M	–	0.0	0.51	0.51	0.50
Rotala macrandra	N	8.0	M	–	8.0	M	–	0.0	0.40	0.39	0.50
Rotala rotundifolia	Y	16.0	M	–	16.0	M	–	0.0	0.50	0.50	0.50
Sagittaria subulata	N	6.0	M	–	6.0	M	–	0.0	0.36	0.34	0.50
Utricularia gibba	Y	32.0	H	TP	42.0	VH	TP	10.0	0.65	0.66	0.54
Vallisneria australis	N	21.5	M	–	31.5	H	FP	10.0	0.60	0.59	0.67

The highest-scoring species (BRA and BRA+CCA scores ≥ 40, taken as an *ad hoc* 'very high risk' threshold) were *Elodea nuttallii* and *Lemna aequinoctialis* and, after accounting for the CCA, also *Elodea canadensis*, *Egeria densa*, *Hygrophila polysperma*, *Azolla cristata*, *Lemna minuta*, *Azolla filiculoides*, *Utricularia gibba* and *Gymnocoronis spilanthoides*. The CCA resulted in an increase in the BRA score (cf. BRA+CCA score) for 17 species and in no change for the remaining seven (Table 2).

For the BRA score outcomes, there were discrepancies in risk ranking between the two risk assessment areas only for *Ludwigia peploides*, which was high risk for the Mediterranean Region, but medium risk for the Pannonian Region (Fig. 2A). For the BRA+CCA score outcomes, there were discrepancies for *Cabomba caroliniana*, *Ludwigia peploides* and *Myriophyllum heterophyllum*, which were high risk for the Mediterranean Region and medium risk for the Pannonian Region and for *Lemna minuta*, which was medium risk for the Mediterranean Region and high risk for the Pannonian Region (Fig. 2B).

For the Mediterranean Region (Table 2):

- Based on the BRA outcome scores: 13 (54.2%) species were classified as high risk and 11 (45.8%) as medium risk. Amongst the 14 species categorised *a priori* as invasive, 12 were true positives and, amongst the 10 species categorised *a priori* as non-invasive, one was a false positive. Of the eleven medium-risk species, nine were *a priori* non-invasive and two invasive.
- Based on the BRA+CCA outcome scores (hence, after accounting for climate change predictions): 17 (70.8%) species were classified as high risk and seven (29.2%) as medium risk. Amongst the *a priori* invasive species, 12 were true positives and amongst the *a priori* non-invasive species, five were false positives. Of the seven medium-risk species, five were *a priori* non-invasive and two invasive.

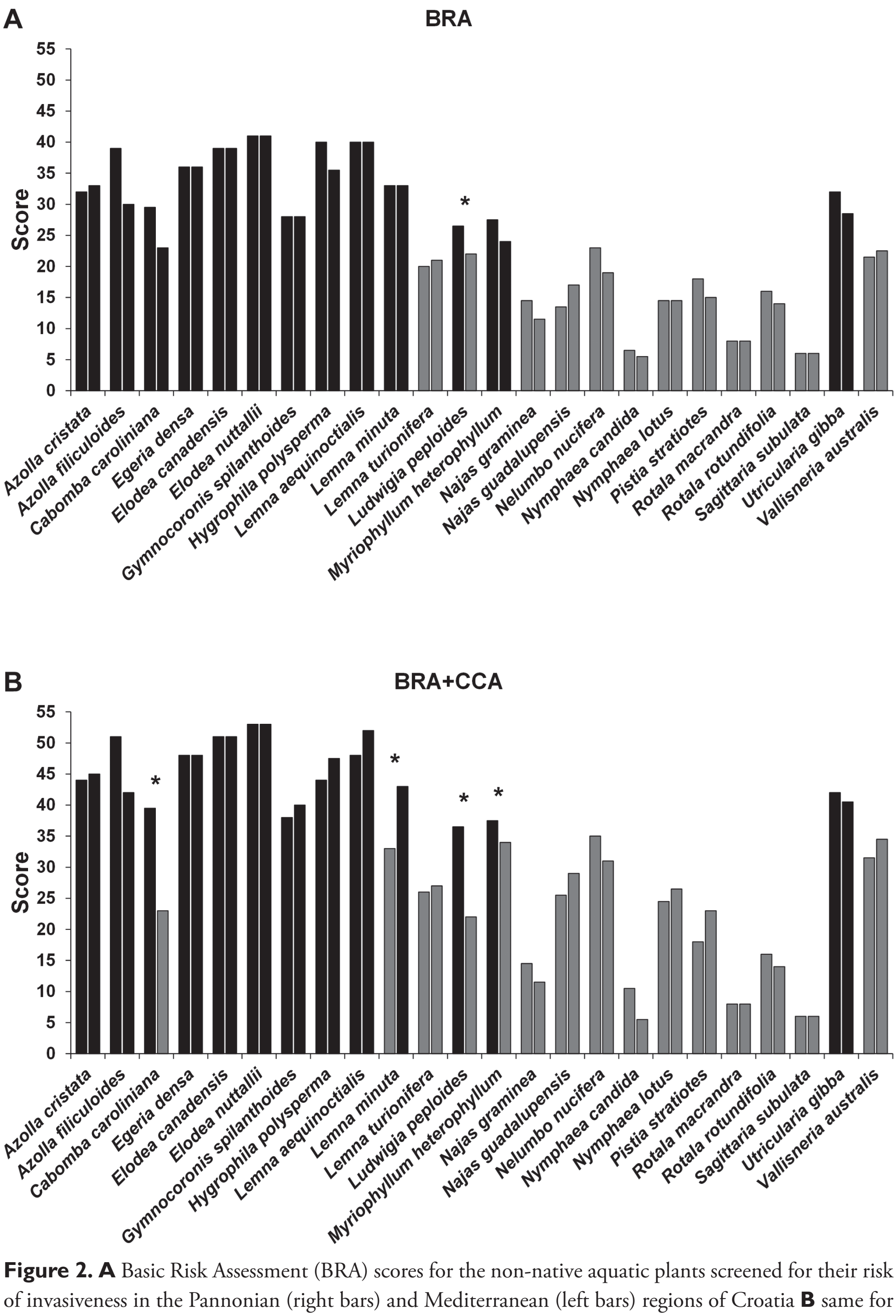

Figure 2. A Basic Risk Assessment (BRA) scores for the non-native aquatic plants screened for their risk of invasiveness in the Pannonian (right bars) and Mediterranean (left bars) regions of Croatia **B** same for the BRA + CCA (Climate Change Assessment) scores. Grey bars = medium risk; black bars = high (or very high) risk (see Table 2). Asterisk indicates species with different risk outcome for the two risk assessment areas.

The highest-scoring species (same very high-risk threshold as for the Pannonian Region) were *Elodea nuttallii*, *Hygrophila polysperma* and *Lemna aequinoctialis* and, after accounting for the CCA, also *Azolla filiculoides*, *Elodea canadensis*, *Egeria densa*, *Azolla cristata* and *Utricularia gibba*. The CCA resulted in an increase in the BRA score (cf. BRA+CCA score) for 18 species and in no change for the remaining six (Table 2).

Confidence and discrepancies in responses

For the Pannonian Region, the mean CF_{Total} was 0.573 ± 0.023 SE, the mean CF_{BRA} 0.573 ± 0.025 SE and the mean CF_{CCA} 0.576 ± 0.024 SE. For the Mediterranean Region, the mean CF_{Total} was 0.573 ± 0.023 SE, the mean CF_{BRA} 0.577 ± 0.025 SE and the mean CF_{CCA} 0.545 ± 0.021 SE. There were no differences in mean CF between risk assessment areas, Components and Status within Risk assessment area × Component (Table 3).

Most discrepancies in the responses, as measured by the number of species for which a different response was provided to a certain question (Q), were for all the Climate, distribution and introduction risk and Climate change questions. There were also discrepancies for four of the 12 Qs related to Undesirable (or persistence) traits, as well as for one Q in each of the Resource exploitation, Reproduction and Dispersal mechanisms sections (Fig. 3).

Table 3. Permutational ANOVA results for the confidence factor (CF) of the non-native aquatic plant species risk screened for the Pannonian and Mediterranean regions of Croatia – the risk assessment areas. Component = BRA, BRA+CCA (see Table 2); Status = Extant, Horizon (see Table 1); # = permutational value.

Source of variation	df	MS	$F^{\#}/t$	$P^{\#}$
Risk assessment area	1	0.326	0.393	0.540
Component	1	0.089	0.108	0.658
Risk assessment area × Component	1	0.562	0.677	0.467
Status (Risk assessment area × Component)	4	0.831	0.808	0.518
Residual	88	1.027		

Discussion

This study is the first calibrated application of the AS-ISK on aquatic plants for a defined risk assessment area (see Vilizzi et al. 2021). Classification of the screened aquatic plant species into medium-risk and high-risk was successfully achieved for the Pannonian and Mediterranean regions of Croatia with a high degree of accuracy (as indicated by the excellent discriminatory power) and the threshold values of 22.75 for the Pannonian Region and 24.75 for the Mediterranean Region are overall in accordance with the global value of 24.5 obtained for freshwater aquatic plants (Vilizzi et al. 2021).

The top invasive species ranked as very high-risk under both current and predicted climate conditions in both risk assessment areas were *Elodea nuttallii* and

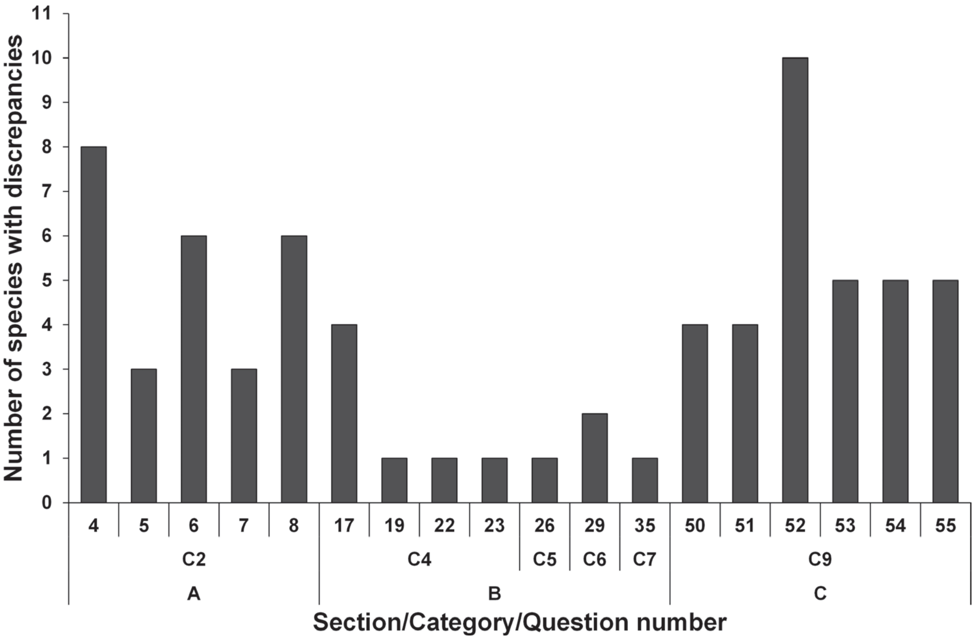

Figure 3. Number of species for which discrepancies in the responses to the AS-ISK questions were found, based on screening for the Pannonian vs. the Mediterranean regions. Section: A = Biogeography/Invasion history; B = Domestication/Cultivation; C = Climate change. Category: C2 = Climate, distribution and introduction risk; C4 = Undesirable (or persistence) traits; C5 = Resource exploitation; C6 = Reproduction; C7 = Dispersal mechanisms; C9 = Climate change. Question codes as per Suppl. material 1: Table S1.

Lemna aequinoctialis. *Elodea nuttallii* is a perennial submerged aquatic plant native to North America and one of the most widespread non-native species in Europe (Hussner 2012) and is found in the Pannonian Region near the Hungarian border (Király et al. 2007; Kočić et al. 2014). Although this species has not yet been recorded in the Mediterranean Region (Nikolić 2022), it is known to be established in its proximity. *Elodea nuttallii* was accidentally introduced as an aquarium plant into Europe, where it was first recorded at the beginning of the 20th century (Cook and Urmi-König 1985). This species has since become widespread throughout the continent (Steen et al. 2019) and, since 2017, has been included in the List of Invasive Alien Species of Union concern (European Union Regulation 1143/2014: EU 2014). The reason for its successful colonisation is attributable to its vegetative reproduction by fragmentation, stem division and production of winter buds as the dominant method of propagation (Thiébaut and Di Nino 2009). In addition, *E. nuttallii* has wide tolerance of habitats and salinity, benefits from anthropogenic pressure, is characterised by vigorous growth (Steen et al. 2019) and is less affected by water temperature fluctuations than native aquatic plants (Fritz et al. 2017). The climate conditions in the Pannonian Region match the species' invaded habitats of Europe, hence rapid spread and colonisation are highly likely under both climate scenarios (Rodríguez-Merino et al. 2018). Although *E. nuttallii* reproduces by vegetative propagation in warm waters (≈ 20 °C) (Hoffmann et al. 2015), it has a temperature limit (Netten et al. 2010). However, as the Mediterranean Region belongs to the Dinaric karst, most of its water bodies are short and their mean annual temperature does not exceed 13 °C (Horvatinčić et al. 2003; Bonacci and Roje-Bonacci 2012; Bonacci et al. 2014); hence, under predicted global warming, the environmental conditions of freshwater ecosystems in this area may become even more suitable for this species (Rodríguez-Merino et al. 2018).

Lemna aequinoctialis is a horizon species for both risk assessment areas that has a broad distribution extending over several continents and has expanded its range to become cosmopolitan (Crawford et al. 2006; Tippery and Les 2020). Although *L. aequinoctialis* is not yet naturalised in most European countries where it is recorded (Hussner et al. 2010), it has high potential to become a new component of European aquatic ecosystems (Crawford et al. 2006). Growth rate of *L. aequinoctialis* under optimal conditions is close to exponential and its frond number may almost double within 24 h, making it one of the fastest-growing flowering plants (Fourounjian et al. 2020). This species can tolerate extreme ranges in pH from 3.2 to more than 9.0 (Thiébaut 2007) and can produce turions (dormant vegetative organs) in response to unfavourable environmental conditions (i.e. decreased temperature, day length or nutrient availability) (Fourounjian et al. 2020; Tippery and Les 2020). The turions overwinter on the bottom water in darkness under hypoxic or anoxic conditions and resume growth once water temperature reaches about 15 °C (Fourounjian et al. 2020; Schweingruber et al. 2020). Despite evidence that *L. aequinoctialis* cannot tolerate temperatures below 0 °C (Vélez-Gavilán 2017), this species seems already naturalised in the Pannonian Region of Hungary (Lukács et al. 2014) and has been reported from France, Germany, Greece and Sweden (Ryman and Anderberg 1999; Thiébaut 2007; Hussner et al. 2010; Lansdown et al. 2016). The climate of the Mediterranean Region under current conditions fully matches the requirements of *L. aequinoctialis* (Vélez-Gavilán 2017), though an increase in mean annual temperatures in both risk assessment areas and particularly in the Pannonian Region, may expedite the species' naturalisation process (Beck et al. 2018; Rodríguez-Merino et al. 2018). Importantly, as *L. aequinoctialis* can be misidentified with native or introduced *Lemna* sp. (Xu et al. 2015), it is highly likely to pose a greater risk of invasiveness than previously assumed.

The horizon species *Hygrophila polysperma* ranked as very high-risk for the Mediterranean Region under current climate conditions and in both risk assessment areas under projected climate conditions. This species naturally occurs in tropical Asia, India and Malaysia and was introduced to Florida and Texas (USA), where it is established (Angerstein and Lemke 1994). However, *H. polysperma* has also recently been found in Europe (i.e. Austria, Germany, Hungary and Poland) in thermally heated waters, where it was probably released from aquaria (Hussner 2012; EPPO 2017). This species has been flagged as high priority in the list of the European Union pest risk analysis (Tanner et al. 2017) due to its high phenotypic plasticity, tolerance of a wide range of habitats, predominant spread via fragments with high regeneration rates and build-up of high biomass densities, which cause it to occupy the entire water column and outcompete by shading with both native and other invasive plant species (EPPO 2017). Under current climate conditions, the Mediterranean coastline of the risk assessment area may be suitable for the naturalisation of *H. polysperma*, whereas under a scenario of climate change, potentially suitable regions for colonisation are the Continental, Black Sea and Atlantic biogeographical regions (EPPO 2017).

Extant *Azolla cristata*, *Azolla filiculoides*, *Elodea canadensis*, *Egeria densa* and horizon *Utricularia gibba* were ranked as high-risk under current climate conditions and, after accounting for climate change, they became very high-risk for both risk assessment areas, whereas horizon *Gymnocoronis spilanthoides* and *Lemna minuta* were ranked as very high-risk under climate change only for the Pannonian Region. *Azolla filiculoides* and *E. canadensis* are the most widespread non-native aquatic plants in Europe (Hussner 2012). The native distributional range of *A. cristata* and *A. filiculoides* extends from North to South America, reflecting their wide temperature and climate tolerance (Troy et al. 2022), whereas *E. canadensis* originates from North America, with most European climates matching its ecological requirements (Duenas-Lopez et al. 2018).

Azolla filiculoides was recorded for the first time in Kopački Rit in 1978 and *Azolla cristata* in 1982 from Osijek in a hydromelioration channel and at Vukovar town in backwaters near the River Danube (Trinajstić and Pavletić 1978; Nikolić 2022). Further records of *Azolla* spp. in Croatia have been related to *A. filiculoides*, which is distributed along the River Drava (Nikolić 2022). However, difficulties in distinguishing between *Azolla* species have led to a long history of misidentifications and taxonomic confusion (Reid et al. 2006). As a result, the finding of *A. cristata* may be either a misidentification for Croatia or its replacement (i.e. by competition/overgrowth) by *A. filiculoides* due to its greater adaptability to eutrophication caused by urbanisation and agricultural activities (Lastrucci et al. 2019). Indeed, in Serbia, which borders Croatia through the River Danube, only *A. filiculoides* has been recognised as present in the aquatic systems of the region (Anđelković et al. 2016). In addition, a survey performed in Czechia revealed that only *A. filiculoides* is present and that *A. cristata* has never occurred in the country (Pyšek et al. 2012). Similar findings have been reported from Portugal (Pereira et al. 2001) and Italy, where *A. cristata* has disappeared from the wild (Lastrucci et al. 2019), with the only reliable record of *A. cristata* in Europe having been confirmed for The Netherlands (Pyšek et al. 2012). Current climate conditions for both risk assessment areas seem suitable for both *Azolla* species (Peel et al. 2007). Nevertheless, *A. filiculoides* under climate change scenarios will extend its distribution northwards and at higher altitudes, so that part of the Mediterranean habitats may become unsuitable (Rodríguez-Merino et al. 2019). However, the boreal climate type in the mountains of the Mediterranean Region of Croatia matches the climate types of northern Europe, including those of the European Alps under both scenarios (Peel et al. 2007; Rubel et al. 2017), making these regions particularly suitable for invasion. On the contrary, *A. cristata* seems to tolerate a higher thermal range (Madeira et al. 2016); hence, under a climate change scenario, this species is likely to adapt to the southern part of the Mediterranean Region.

Elodea canadensis has a long history of establishment in Croatia and, similar to its congener *Elodea nuttallii*, is perennial, has wide ecological tolerance with overwintering in deeper waters, asexual reproduction and relatively fast growth (Barrat-Segretain et al. 2002). *Elodea canadensis* is currently well distributed across the Pannonian Region, but not yet recorded in the Mediterranean Region (Kočić et al. 2014). However, due to its long history of establishment in Croatia and several aquatic species translocations (Pofuk et al. 2017), *E. canadensis* may also have been transferred, but overlooked, particularly because of the reported lack of a full inventory of aquatic plants in the Mediterranean Region. Under climate change predictions, suitable areas for *E. canadensis* extend into several Mediterranean countries and areas next to the Black Sea (Heikkinen et al. 2009), hence making this species highly likely to become more invasive in Croatia.

The native distribution of *Egeria densa* is temperate and sub-tropical South America, whereas the distribution of *Gymnocoronis spilanthoides* and *Lemna minuta* extends to tropical regions of North and South America. *Utricularia gibba* has a mostly pantropical distribution and, apart from North America, occurs in Asia, the Pacific and the western Mediterranean (CABI 2022). All four species have been introduced around the world primarily by the aquarium and pet trade (Saul et al. 2017). Amongst these species, naturalised populations of *E. densa* were recently recorded in the Mediterranean Region of Croatia in the River Neretva Basin in clear, slow flowing, oligohaline waters with high alkalinity and conductivity, where the species surpasses in abundance the indigenous ones (Rimac et al. 2018). The presence of only male specimens of *E. densa* has been observed in Croatia, similar to other countries' part of the species' introduced range of distribution (Thiébaut et al. 2016; Rimac et al. 2018). The principal means of *E. densa* reproduction is vegetatively by fragmentation of stems, which can form dense mats (Thiébaut et al. 2016). This species tolerates a wide range of climate types that overlap with the climates of both risk assessment areas and, under predicted climate change conditions, it is likely to remain as high-risk particularly at higher altitudes of the risk assessment areas (Gillard et al. 2017). In proximity to the risk assessment areas, *G. spilanthoides* and *L. minuta* are found in Hungary, whereas *U. gibba* has also been recorded in Serbia (Hussner 2012; CABI 2022). *Utricularia gibba* is an annual or perennial submerged or free-floating carnivorous aquatic plant that can rapidly colonise new water bodies by stem fragmentation or by its seeds forming a dense mat cover on the water surface (deWinton et al. 2009; CABI 2022). Considering its wide distribution and tolerance of a wide range of temperatures and habitats, this species may become a serious threat under both current and projected climate conditions in the risk assessment areas. *Gymnocoronis spilanthoides* is naturalised in Europe, grows very rapidly, easily reproduces vegetatively by any parts of its stem, forms floating mats that may cover entire water bodies, blocks drainage channels and degrades natural wetlands by displacing native plants and animals – all of these traits make it of higher threat for Europe than initially predicted (Ardenghi et al. 2016). *Lemna minuta* is a small free-floating plant that also forms dense mats on the water surface, reduces light penetration and gas exchange, causes the disappearance of submersed aquatic plants and alters invertebrate community composition and abundance (Ceschin et al. 2020).

This species is established and widespread in several countries and climate types in Europe (Paolacci et al. 2018) but has not yet been recorded in the risk assessment areas. However, *L. minuta* can be easily overlooked and confused with native *Lemna minor*. This is because the only reliable diagnostic character is the vein number, which is not easy to identify in the field: *L. minuta* has just one, whereas *L. minor* has three (Bog et al. 2010; Gérard and Triest 2018). Under current climate conditions, *G. spilanthoides* and *L. minuta* gained a high risk of invasiveness but may become very high-risk in the Pannonian Region likely due to the availability of habitats consisting of slow-flowing or lentic water bodies and wet–marshy soils and wetlands, which are not well represented in the Mediterranean Region.

In both risk assessment areas, *Cabomba caroliniana* and *Myriophyllum heterophyllum* were ranked as high-risk under both climate scenarios, whereas *Ludwigia peploides* was high-risk under both climate scenarios in the Pannonian Region only. The native distribution of *C. caroliniana* covers the eastern part of subtropical and temperate areas of South America (Roberts and Florentine 2022), hence matching current and projected climate conditions in both risk assessment areas (Beck et al. 2018; Rodríguez-Merino et al. 2018). This species is established in Serbia (Vojvodina, near the Croatian border), where it has expanded from Hungary probably through the canal network of the hydrosystem Danube-Tisa-Danube (Anđelković et al. 2016). In Europe, *C. caroliniana* is still not regarded as invasive and is mostly found in localised and scattered populations (Roberts and Florentine 2022), though in The Netherlands, it has been declared as high risk (Matthews et al. 2013). The species' high invasiveness has been reported in its non-native distributional range, primarily due to its high competitiveness, dense and persistent growth, asexual reproduction through stem auto-fragmentation and tolerance of extreme pH ranges from 4.0 to 8.8 (Matthews et al. 2013). The species' population expansion in connected waterways, as occurring in the Pannonian Region, may be facilitated by its long fragments that can get wrapped in boat motors, boating or anglers' equipment (Roberts and Florentine 2022). In addition, vectors like fish re-stocking and spread by birds cannot be ruled out as potential pathways of introduction into the Mediterranean Region. *Myriophyllum heterophyllum* is native to the southeast USA and *L. peploides* to South and Central USA and both species are listed as invasive alien species of EU concern (EU 2014). Their native distribution and preferred climate match both risk assessment areas. *Ludwigia peploides* was recently recorded in the Pannonian Region (River Ilova: Buzjak and Sedlar 2018) and has been recognised as posing a severe problem should it expand its current distributional range (Vuković et al. 2021). Additionally, in Croatia, *M. heterophyllum* was recently found in the Mediterranean Region on the Island of Krk (north-eastern Adriatic) in Lake Ponikve (Starmühler 2009; Jasprica et al. 2017), as well as in the River Neretva Delta (Jasprica et al. 2017). It is likely that both species can spread rapidly due to their fast uncontrolled growth and propagation by fragments (Gérard et al. 2014; Gross et al. 2020). Both *L. peploides* and *M. heterophyllum* are tolerant of a wide range of temperatures, substrata, and water quality (Matrat et al. 2004; Hussner and Jahns 2015). The most suitable habitat in Croatia for *L. peploides* is the eastern Pannonian Region, whereas for *M. heterophyllum*, the Mediterranean Region appears to be more suitable (Rodríguez-Merino et al. 2018).

However, under global warming conditions *L. peploides* may accelerate the time of germination of its seeds (Gillard et al. 2017) and for *M. heterophyllum*, the ecologically suitable habitat is likely to increase in both risk assessment areas (Jasprica et al. 2017).

The horizon species *Lemna turionifera*, *Najas guadalupensis*, *Nelumbo nucifera* and *Vallisneria australis* from medium-risk level areas under current climate conditions were predicted to become high-risk under climate change in both risk assessment areas, whereas a high-risk score under such conditions was obtained for horizon *Nymphaea lotus* and extant *Pistia stratiotes* only for the Mediterranean Region. Naturalised populations of *L. turionifera*, *N. nucifera*, *V. australis* and *P. stratiotes* are found in Europe (Mastrantuono and Mancinelli 1999; Hussner 2012; Mesterházy et al. 2021) and can be expected in the risk assessment areas in the near future. Recently, tropical *P. stratiotes* was found in Croatia (Boršić and Rubinić 2018), although its status remains unknown. This species has been found to be established in Slovenia in thermal springs of the Pannonian Region (Jaklič et al. 2020), where it is capable of explosive vegetative spread in early spring. This indicates that *P. stratiotes* can expand its range in similar habitats of the area (Šajna et al. 2007). In addition, global warming may also assist in the process of the species' expansion.

Other screened species including extant *Najas graminea* and horizon *Nymphaea lotus*, *Nymphaea candida*, *Rotala macrandra*, *Rotala rotundifolia* and *Sagittaria subulata* gained the lowest scores in both risk assessment areas. However, considering that some of those species are naturalised in Europe (e.g. *S. subulata*: Hrivnák et al. 2019), their threat of establishment due to the presence of diverse climate types in the risk assessment areas cannot be ruled out. An example is the established population of *N. graminea* on the Island of Cres in the Mediterranean Region (Nikolić 2022), despite this population being localised.

Conclusions

Research on aquatic plants in Croatia is historically fragmented and has not been conducted systematically (Odak and Treer 2000) with the result that, compared to other groups of aquatic organisms, it is almost non-existent (MINGOR 2022; Nikolić 2022). Most of the available reports have been related to aspects of aquatic plants in aquaculture (Bralić 1969; Debeljak et al. 1992), so that there is no research investigating the impacts of non-native aquatic plants on the aquatic ecosystems and biodiversity of Croatia. The removal and reduction of dense mats of aquatic plants, including those of non-native *Elodea canadensis*, in fishponds of the risk assessment areas have been attempted by the use of herbicides and by the introduction of non-native herbivorous fish species for biological control (e.g. grass carp *Ctenopharyngodon idella*) (Đisalov 1961; Bralić 1969), but without real success. Overall, there have been no systematic attempts to eradicate any species of non-native aquatic plants until recently, when a first attempt was made for *Myriophyllum heterophyllum* in the River Neretva Basin in 2021. However, the outcomes of this eradication programme are not yet known.

Despite the recent establishment of monitoring programmes for invasive alien species co-financed by the European Union Cohesion Fund, in the Croatian Catalogue of alien species, data are currently missing for aquatic non-native plants and in relation to species' description and pathways/vectors of introduction and distribution (MINGOR 2022: https://invazivnevrste.haop.hr/katalog). This suggests that urgent research is necessary in the risk assessment areas of the country to develop management plans for the establishment of rapid control and eradication measures. The present study, therefore, represents the first step towards an increase in the knowledge about the risks poses by the extant and horizon non-native aquatic plants in Croatia, which may allow decision-makers to develop adequate measures for management and control/mitigation.

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References

Amanatidou E, Butter M, Carabias V, Könnölä T, Leis M, Saritas O, Schaper-Rinkel P, van Rij V (2012) On concepts and methods in horizon scanning: Lessons from initiating policy dialogues on emerging issues. Science and Public Policy 39(2): 208–221. https://doi.org/10.1093/scipol/scs017

Anđelković A, Zivković MM, Cvijanović DL, Novković M, Marisavljević D, Pavlović D, Radulović S (2016) The contemporary records of aquatic plants invasion through the Danubian floodplain corridor in Serbia. Aquatic Invasions 11(4): 381–395. https://doi.org/10.3391/ai.2016.11.4.04

Angerstein MB, Lemke DE (1994) First records of the aquatic weed *Hygrophila polysperma* (Acanthaceae) from Texas. SIDA, Contributions to Botany 16: 365–371.

Ardenghi NM, Barcheri G, Ballerini C, Cauzzi P, Guzzon F (2016) *Gymnocoronis spilanthoides* (Asteraceae, Eupatorieae), a new naturalized and potentially invasive aquatic alien in S Europe. Willdenowia 46(2): 265–273. https://doi.org/10.3372/wi.46.46208

Bacher S, Blackburn TM, Essl F, Genovesi P, Heikkilä J, Jeschke JM, Jones G, Keller R, Kenis M, Kueffer C, Martinou AF, Nentwig W, Pergl J, Pyšek P, Rabitsch W, Richardson DM, Roy HE, Saul W-C, Scalera R, Vilà M, Wilson JRU, Kumschick S (2018) Socio-economic impact classification of alien taxa (SEICAT). Methods in Ecology and Evolution 9(1): 159–168. https://doi.org/10.1111/2041-210X.12844

Banha F, Anastácio PM (2015) Live bait capture and crayfish trapping as potential vectors for freshwater invasive fauna. Limnologica 51: 63–69. https://doi.org/10.1016/j.limno.2014.12.006

Barrat-Segretain MH, Elger A, Sagnes P, Puijalon S (2002) Comparison of three life-history traits of invasive *Elodea canadensis* Michx. and *Elodea nuttallii* (Planch.) H. St. John. Aquatic Botany 74(4): 299–313. https://doi.org/10.1016/S0304-3770(02)00106-7

Beck HE, Zimmermann NE, McVicar TR, Vergopolan N, Berg A, Wood EF (2018) Present and future Köppen-Geiger climate classification maps at 1-km resolution. Scientific Data 5(1): e180214. https://doi.org/10.1038/sdata.2018.214

Blackburn TM, Essl F, Evans T, Hulme PH, Jeschke JM, Kühn I, Kumschick S, Marková Z, Mrugała A, Nentwig W, Pergl J, Pyšek P, Rabitsch W, Ricciardi A, Richardson DM, Sendek A, Vilà M, Wilson JRU, Winter M, Genovesi P, Bacher S (2014) A unified classification of alien species based on the magnitude of their environmental impacts. PLoS Biology 12(5): e1001850. https://doi.org/10.1371/journal.pbio.1001850

Bog M, Baumbach H, Schween U, Hellwig F, Landolt E, Appenrot KJ (2010) Genetic structure of the genus *Lemna* L. (Lemnaceae) as revealed by amplified fragment length polymorphism. Planta 232(3): 609–619. https://doi.org/10.1007/s00425-010-1201-2

Bonacci O, Andrić I (2008) Sinking karst rivers hydrology: Case of the Lika and Gacka (Croatia). Acta Carsologica 37(2–3): 185–196. https://doi.org/10.3986/ac.v37i2.146

Bonacci O, Roje-Bonacci T (2012) Impact of grout curtains on karst groundwater behaviour: An example from the Dinaric karst. Hydrological Processes 26(18): 2765–2772. https://doi.org/10.1002/hyp.8359

Bonacci O, Andrić I, Yamashiki Y (2014) Hydrology of Blue Lake in the Dinaric karst. Hydrological Processes 28(4): 1890–1898. https://doi.org/10.1002/hyp.9736

Boršić I, Rubinić T (2018) First record of *Pistia stratiotes* L. (Araceae) in Croatia, with the consideration of possible introduction pathways. 3rd Croatian Symposium on invasive species 26–27. XI 2018, Book of abstracts, Zagreb, Croatia, 1–96.

Bralić V (1969) Weed control in fishponds using herbicides. Ribarstvo Jugoslavije 6: 130–132. [in Croatian]

Buzjak S, Sedlar Z (2018) *Ludwigia peploides* (Kunth.) PH Raven – Floating Water Primrose, a new species in Croatian flora from the list of invasive alien species of Union concern. Natura Croatica 27(2): 351–356. https://doi.org/10.20302/NC.2018.27.25

CABI (2022) Invasive Species Compendium. CAB International, Wallingford. www.cabi.org/isc

Caraco N, Cole J, Findlay S, Wigand C (2006) Vascular plants as engineers of oxygen in aquatic systems. Bioscience 56(3): 219–225. https://doi.org/10.1641/0006-3568(2006)056[0219:VPAEOO]2.0.CO;2

Ceschin S, Ferrante G, Mariani F, Traversetti L, Ellwood NTW (2020) Habitat change and alteration of plant and invertebrate communities in waterbodies dominated by the invasive alien macrophyte *Lemna minuta* Kunth. Biological Invasions 22(4): 1325–1337. https://doi.org/10.1007/s10530-019-02185-5

Cook CDK, Urmi-König K (1985) A revision of the genus *Elodea* (Hydrocharitaceae). Aquatic Botany 21(2): 111–156. https://doi.org/10.1016/0304-3770(85)90084-1

Copp GH, Templeton M, Gozlan RE (2007) Propagule pressure and the invasion risks of non-native freshwater fishes in Europe: A case study of England. Journal of Fish Biology 71: 148–159. https://doi.org/10.1111/j.1095-8649.2007.01680.x

Copp GH, Vilizzi L, Tilbury H, Stebbing P, Tarka AS, Miossec L, Goulletquer P (2016) Development of a generic decision-support tool for identifying potentially invasive aquatic taxa: AS-ISK. Management of Biological Invasions 7(4): 343–350. https://doi.org/10.3391/mbi.2016.7.4.04

Copp GH, Vilizzi L, Wei H, Li S, Piria M, Al-Faisal AJ, Almeida D, Atique U, Al-Wazzan Z, Bakiu R, Bašić T, Bui TD, Canning-Clode J, Castro N, Chaichana R, Çoker T, Dashinov D, Ekmekçi FG, Erős T, Ferincz Á, Ferreira T, Giannetto D, Gilles Jr AS, Głowacki Ł, Goulletquer P, Interesova E, Iqbal S, Jakubčinová K, Kanongdate K, Kim JE, Kopecký

O, Kostov V, Koutsikos N, Kozic S, Kristan P, Kurita Y, Lee HG, Leuven RSEW, Lipinskaya T, Lukas J, Marchini A, González-Martínez AI, Masson L, Memedemin D, Moghaddas SD, Monteiro J, Mumladze L, Naddafi R, Năvodaru I, Olsson KH, Onikura N, Paganelli D, Pavia Jr RT, Perdikaris C, Pickholtz R, Pietraszewski D, Povž M, Preda C, Ristovska M, Rosíková K, Santos JM, Semenchenko V, Senanan W, Simonović P, Smeti E, Števove B, Švolíková K, Ta KAT, Tarkan AS, Top N, Tricarico E, Uzunova E, Vardakas L, Verreycken H, Zięba G, Mendoza R (2021) Speaking their language – Development of a multilingual decision-support tool for communicating invasive species risks to decision makers and stakeholders. Environmental Modelling & Software 135: e104900. https://doi.org/10.1016/j.envsoft.2020.104900

Coughlan NE, Kelly TC, Jansen MA (2017) "Step by step": High frequency short-distance epizoochorous dispersal of aquatic macrophytes. Biological Invasions 19(2): 625–634. https://doi.org/10.1007/s10530-016-1293-0

Crane K, Cuthbert RN, Dick JT, Kregting L, MacIsaac HJ, Coughlan NE (2019) Full steam ahead: Direct steam exposure to inhibit spread of invasive aquatic macrophytes. Biological Invasions 21(4): 1311–1321. https://doi.org/10.1007/s10530-018-1901-2

Crawford DJ, Landolt EL, Les DH, Kimball RT (2006) Speciation in Duckweeds (Lemnaceae): Phylogenetic and ecological inferences. Aliso: A Journal of Systematic and Evolutionary Botany 22: e19. https://doi.org/10.5642/aliso.20062201.19

Debeljak Lj, Bebek Ž, Fašaić K, Mrakovčić M (1992) The influence of water macrophytes on the quantity of oxygen in carp ponds' water. Ribarstvo 1/2: 5–12. [in Croatian]

Dehnen-Schmutz K, Touza J (2008) Plant invasions and ornamental horticulture: pathway, propagule pressure and the legal framework. Floriculture, ornamental and plant biotechnology advances and topical issues 5: 15–21. https://doi.org/10.1111/j.1523-1739.2006.00538.x

deWinton MD, Champion PD, Clayton JS, Wells RD (2009) Spread and status of seven submerged pest plants in New Zealand lakes. New Zealand Journal of Marine and Freshwater Research 43(2): 547–561. https://doi.org/10.1080/00288330909510021

Đisalov N (1961) Attempt of acclimatization of herbivorous fish. Ribarstvo Jugoslavije 16: 51–52. [in Croatian]

Duenas-Lopez MA, Popay I, Dawson H (2018) *Elodea canadensis* (Canadian pondweed). Invasive Species Compendium. CABI, Wallingford. https://doi.org/10.1079/ISC.20759.20203483396

Eckert CG, Dorken ME, Barrett SCH (2016) Ecological and evolutionary consequences of sexual and clonal reproduction in aquatic plants. Aquatic Botany 135: 46–61. https://doi.org/10.1016/j.aquabot.2016.03.006

EPPO (2017) Pest risk analysis for *Hygrophila polysperma*. European and Mediterranean Plant Protection Organization, Paris. http://www.iap-risk.eu/media/files/pra_exp_HYGPO.pdf

Essl F, Dawson W, Kreft H, Pergl J, Pyšek P, Van Kleunen M, Weigelt P, Mang T, Dullinger S, Lenzner B, Moser D, Maurel N, Seebens H, Stein A, Weber E, Chatelain C, Genovesi P, Kartesz J, Morozova O, Nishino M, Nowak PM, Pagad S, Shu WS, Winter M (2019) Drivers of the relative richness of naturalized and invasive plant species on Earth. AoB Plants 11(5): plz051. https://doi.org/10.1093/aobpla/plz051

EU (2014) Regulation (EU) 1143/2014 of the European Parliament and of the Council of 22 October 2014 on the prevention and management of the introduction and spread of invasive alien species. Official Journal of the European Union, L 317: 35–55. http://eur-lex.europa.eu/legalcontent/EN/TXT/?qid=1417443504720&uri=CELEX:32014R1143

Fourounjian P, Fakhoorian T, Cao XH (2020) Importance of duckweeds in basic research and their industrial applications. In: Cao XH, Fourounjian P, Wang W (Eds) The duckweed genomes. Springer Nature Switzerland, 1–18. https://doi.org/10.1007/978-3-030-11045-1_1

Fritz C, Schneider T, Geist J (2017) Seasonal variation in spectral response of submerged aquatic macrophytes: A case study at Lake Starnberg (Germany). Water (Basel) 9(7): e527. https://doi.org/10.3390/w9070527

Gérard J, Triest L (2018) Competition between invasive *Lemna minuta* and native *L. minor* in indoor and field experiments. Hydrobiologia 812(1): 57–65. https://doi.org/10.1007/s10750-016-2754-2

Gérard J, Brion N, Triest L (2014) Effect of water column phosphorus reduction on competitive outcome and traits of *Ludwigia grandiflora* and *L. peploides*, invasive species in Europe. Aquatic Invasions 9(2): 157–166. https://doi.org/10.3391/ai.2014.9.2.04

Gillard M, Grewell BJ, Deleu C, Thiébaut G (2017) Climate warming and water primroses: Germination responses of populations from two invaded ranges. Aquatic Botany 136: 155–163. https://doi.org/10.1016/j.aquabot.2016.10.001

Glasnović P, Novak Š, Behrič S, Fujs N (2015) Towards a checklist of the vascular flora of the Neretva River Delta (Croatia). Natura Croatica 24(2): 163–190. https://doi.org/10.20302/NC.2015.24.11

Gross EM, Groffier H, Pestelard C, Hussner A (2020) Ecology and environmental impact of *Myriophyllum heterophyllum*, an aggressive invader in European waterways. Diversity (Basel) 12(4): e127. https://doi.org/10.3390/d12040127

Heikkinen R, Leikola N, Fronzek S, Lampinen R, Toivonen H (2009) Predicting distribution patterns and recent northward range shift of an invasive aquatic plant: *Elodea canadensis* in Europe. BioRisk 2: 1–32. https://doi.org/10.3897/biorisk.2.4

Hoffmann MA, Raeder U, Melzer A (2015) Influence of environmental conditions on the regenerative capacity and the survivability of *Elodea nuttallii* fragments. Journal of Limnology 74: 12–20. https://doi.org/10.4081/jlimnol.2014.952

Horvatinčić N, Bronić IK, Obelić B (2003) Differences in the 14C age, δ13C and δ18O of Holocene tufa and speleothem in the Dinaric Karst. Palaeogeography, Palaeoclimatology, Palaeoecology 193(1): 139–157. https://doi.org/10.1016/S0031-0182(03)00224-4

Hosmer Jr DW, Lemeshow S, Sturdivant RX (2013) Applied logistic regression. 3rd edn, John Wiley & Sons, UK, 511 pp. https://doi.org/10.1002/9781118548387

Hrivnák R, Medvecká J, Baláži P, Bubíková K, Oťaheľová H, Svitok M (2019) Alien aquatic plants in Slovakia over 130 years: Historical overview, current distribution and future perspectives. NeoBiota 49: 37–56. https://doi.org/10.3897/neobiota.49.34318

Hussner A (2012) Alien aquatic plant species in European countries. Weed Research 52(4): 297–306. https://doi.org/10.1111/j.1365-3180.2012.00926.x

Hussner A, Jahns P (2015) European native *Myriophyllum spicatum* showed a higher HC03- use capacity than alien invasive *Myriophyllum heterophyllum.* Hydrobiologia 746(1): 171–182. https://doi.org/10.1007/s10750-014-1976-4

Hussner A, van de Weyer K, Gross EM, Hilt S (2010) Comments on increasing number and abundance of non-indigenous aquatic macrophyte species in Germany. Weed Research 50(6): 519–526. https://doi.org/10.1111/j.1365-3180.2010.00812.x

Hussner A, Stiers I, Verhofstad MJJM, Bakker ES, Grutters BMC, Haury J, van Valkenburg JLCH, Brundu G, Newman J, Clayton JS, Anderson LWJ, Hofstra D (2017) Management and control methods of invasive alien aquatic plants: A review. Aquatic Botany 136: 112–137. https://doi.org/10.1016/j.aquabot.2016.08.002

Jaklič M, Koren Š, Jogan N (2020) Alien water lettuce (*Pistia stratiotes* L.) outcompeted native macrophytes and altered the ecological conditions of a Sava oxbow lake (SE Slovenia). Acta Botanica Croatica 79(1): 35–42. https://doi.org/10.37427/botcro-2020-009

Jasprica N, Lasić A, Hafner D, Bratoš Cetinić A (2017) *Myriophyllum heterophyllum* Michx. (Haloragaceae) u Hrvatskoj. Natura Croatica 26: 99–103. https://doi.org/10.20302/NC.2017.26.7

Király G, Steták D, Bányász Á (2007) Spread of invasive macrophytes in Hungary. NeoBiota 7: 123–131.

Kočić A, Horvatić J, Jelaska SD (2014) Distribution and morphological variations of invasive macrophytes *Elodea nuttallii* (Planch.) H. St. John and *Elodea canadensis* Michx in Croatia. Acta Botanica Croatica 73(2): 437–446. https://doi.org/10.2478/botcro-2014-0011

Kutleša P, Boršić I, Cigovski Mustafić M, Ješovnik A, Mihinjač T, Slivar S (2021) The results of the first systematic mapping of alien species in Croatia. 4th Croatian Symposium on Invasive Species. Book of abstracts of the 4th Croatian symposium on invasive species. Zagreb, Croatia, 29–30. XI 2021, 1–17.

Lansdown RV, Anastasiu P, Barina Z, Bazos I, Çakan H, Caković D, Delipetrou P, Matevski V, Mitić B, Ruprecht E, Tomović G, Tosheva A, Király G (2016) Review of Alien Freshwater Vascular Plants in South-east Europe. In: Rat M, Trichova T, Scalera R, Tomov R, Uludag A (Eds) ESENIAS Scientific Reports 1. State of the Art of Alien Species in South-Eastern Europe. University of Novi Sad, Serbia; IBER-BAS, Bulgaria; ESENIAS, 137–154.

Lastrucci L, Fiorini G, Lunardi L, Viciani D (2019) Herbarium survey on the genus *Azolla* (Salviniaceae) in Italy: distributive and taxonomic implications. Plant Biosystems-An International Journal Dealing with all Aspects of Plant Biology 153: 710–719. https://doi.org/10.1080/11263504.2018.1549601

Leung B, Bossenbroek JM, Lodge DM (2006) Boats, pathways, and aquatic biological invasions; estimating dispersal potential with gravity models. Biological Invasions 8(2): 241–254. https://doi.org/10.1007/s10530-004-5573-8

Lovell S, Stone S, Fernandez L (2006) the economic impacts of aquatic invasive species: A review of the literature. Agricultural and Resource Economics Review 35(1): 195–208. https://doi.org/10.1017/S1068280500010157

Lukács BA, Mesterházy A, Vidéki AR, Király G (2014) Alien aquatic vascular plants in Hungary (Pannonian ecoregion): Historical aspects, data set and trends, Plant Biosystems 150: 388–395. https://doi.org/10.1080/11263504.2014.987846

Madeira PT, Hill MP, Dray Jr FA, Coetzee JA, Paterson ID, Tipping PW (2016) Molecular identification of *Azolla* invasions in Africa: The *Azolla* specialist, *Stenopelmus rufinasus* proves to be an excellent taxonomist. South African Journal of Botany 105: 299–305. https://doi.org/10.1016/j.sajb.2016.03.007

Mastrantuono L, Mancinelli T (1999) Long-term changes of zoobenthic fauna and submerged vegetation in the shallow Lake Monterosi (Italy). Limnologica 29(2): 160–167. https://doi.org/10.1016/S0075-9511(99)80063-2

Matrat R, Anras L, Vienne L, Hervochon F, Pineau C, Bastian S, Dutartre A, Haury J, Lambert E, Gilet H, Lacroix P, Maman L (2004) Guide technique pour la gestion des plantes exotiques envahissantes en cours d'eau et zones humides. Comité des Pays de la Loire, 68 pp. https://hal.inrae.fr/hal-02584159

Matthews J, Beringen R, Lamers LPM, Odé B, Pot R, Velde G, Valkenburg JLCH, Verbrugge LNH, Leuven LS (2013) Risk analysis of the non-native Fanwort (*Cabomba caroliniana*) in the Netherlands. Series of Reports on Environmental Science, 46 pp. http://www.roelfpot.nl/publicaties/RA_Cabomba_2013.pdf

Mesterházy A, Somogyi G, Efremov A, Verloove F (2021) Assessing the genuine identity of alien *Vallisneria* (Hydrocharitaceae) species in Europe. Aquatic Botany 174: e103431. https://doi.org/10.1016/j.aquabot.2021.103431

MINGOR (2022) Catalogue of alien species. Ministry of Economy and Sustainable Development, Institute for Environment and Nature. https://invazivnevrste.haop.hr/katalog

Netten JJ, Arts GH, Gylstra R, van Nes EH, Scheffer M, Roijackers RM (2010) Effect of temperature and nutrients on the competition between free-floating *Salvinia natans* and submerged *Elodea nuttallii* in mesocosms. Fundamental and Applied Limnology 177(2): 125–132. https://doi.org/10.1127/1863-9135/2010/0177-0125

Nikolić T [Ed.] (2022) Flora Croatica Database. University of Zagreb Faculty of Science. http://hirc.botanic.hr/fcd

Odak T, Treer T (2000) The research of water macrophyte in Croatia. Ribarstvo 58: 101–109. [in Croatian]

Paolacci S, Jansen MA, Harrison S (2018) Competition between *Lemna minuta*, *Lemna minor*, and *Azolla filiculoides*. Growing fast or being steadfast? Frontiers in Chemistry 6: e207. https://doi.org/10.3389/fchem.2018.00207

Peel MC, Finlayson BL, McMahon TA (2007) Updated world map of the Köppen-Geiger climate classification. Hydrology and Earth System Sciences 11(5): 1633–1644. https://doi.org/10.5194/hess-11-1633-2007

Pereira AL, Teixeira G, Sevinate-Pinto I, Antunes T, Carrapiço F (2001) Taxonomic re-evaluation of the *Azolla* genus in Portugal. Plant Biosystems 135: 285–294. https://doi.org/10.1080/11263500112331350920

Pofuk M, Zanella D, Piria M (2017) An overview of the translocated native and non-native fish species in Croatia: Pathways, impacts and management. Management of Biological Invasions 8(3): 425–435. https://doi.org/10.3391/mbi.2017.8.3.16

Pollux BJA, De Jong M, Steegh A, Ouborg NJ, Van Groenendael JM, Klaassen M (2006) The effect of seed morphology on the potential dispersal of aquatic macrophytes by the common carp (*Cyprinus carpio*). Freshwater Biology 51(11): 2063–2071. https://doi.org/10.1111/j.1365-2427.2006.01637.x

Potočki K, Bekić D, Bonacci O, Kulić T (2021) Hydrological Aspects of Nature-Based Solutions in Flood Mitigation in the Danube River Basin in Croatia: Green vs. Grey Approach. The Handbook of Environmental Chemistry. Springer, Berlin, Heidelberg, 26 pp. https://doi.org/10.1007/698_2021_770

Pyšek P, Danihelka J, Sádlo J, Chrtek Jr J, Chytrý M, Jarošík V, Kaplan Z, Krahulec F, Moravcová L, Pergl J, Štajerová K, Tichý L (2012) Catalogue of alien plants of the Czech Republic: Checklist update, taxonomic diversity and invasion patterns. Preslia 84: 155–255.

R Core Team (2021) R: A Language and Environment for Statistical Computing. Vienna, Austria: R Foundation for Statistical Computing. https://www.r-project.org/

Reid JD, Plunkett GM, Peters GA (2006) Phylogenetic relationships in the heterosporous fern genus *Azolla* (Azollaceae) based on DNA sequence data from three noncoding regions. International Journal of Plant Sciences 167(3): 529–538. https://doi.org/10.1086/501071

Rendeková A, Mičieta K, Hrabovský M, Eliašová M, Miškovic J (2019) Effects of invasive plant species on species diversity: Implications on ruderal vegetation in Bratislava City, Slovakia, Central Europe. Acta Societatis Botanicorum Poloniae 88(2): e3621. https://doi.org/10.5586/asbp.3621

Reynolds C, Miranda NAF, Cumming GS, Keller R (2015) The role of waterbirds in the dispersal of aquatic alien and invasive species. Diversity and Distributions 21(7): 744–754. https://doi.org/10.1111/ddi.12334

Rimac A, Stanković I, Alegro A, Gottstein S, Koletić N, Vuković N (2018) The Brazilian elodea (*Egeria densa* Planch.) invasion reaches Southeast Europe. BioInvasions Records 7(4): 381–389. https://doi.org/10.3391/bir.2018.7.4.05

Roberts J, Florentine S (2022) A global review of the invasive aquatic weed *Cabomba caroliniana* [A. Gray] (Carolina fanwort): Current and future management challenges, and research gaps. Weed Research 62(1): 75–84. https://doi.org/10.1111/wre.12518

Robin X, Turck N, Hainard A, Tiberti N, Lisacek F, Sanchez J-C, Müller M (2011) pROC: An open-source package for R and S+ to analyze and compare ROC curves. BMC Bioinformatics 12(1): e77. https://doi.org/10.1186/1471-2105-12-77

Rodriguez-Merino A, Garcia-Murillo P, Cirujano S, Fernandez-Zamudio R (2018) Predicting the risk of aquatic plant invasions in Europe: How climatic factors and anthropogenic activity influence potential species distributions. Journal for Nature Conservation 45: 58–71. https://doi.org/10.1016/j.jnc.2018.08.007

Rodríguez-Merino A, Fernández-Zamudio R, García-Murillo P, Muñoz J (2019) Climatic niche shift during *Azolla filiculoides* invasion and its potential distribution under future scenarios. Plants 8(10): e424. https://doi.org/10.3390/plants8100424

Roy HE, Adriaens T, Aldridge DC, Bacher S, Bishop JDD, Blackburn TM, Branquart E, Brodie J, Carboneras C, Cook EJ, Copp GH, Dean HJ, Eilenberg J, Essl F, Gallardo B, Garcia M, García-Berthou E, Genovesi P, Hulme PE, Kenis M, Kerckhof F, Kettunen M, Minchin D, Nentwig W, Nieto A, Pergl J, Pescott O, Peyton J, Preda C, Rabitsch W, Roques A, Rorke S, Scalera R, Schindler S, Schönrogge K, Sewell J, Solarz W, Stewart A, Tricarico E, Vanderhoeven S, van der Velde G, Vilà M, Wood CA, Zenetos A (2015) Invasive Alien Species - Prioritising prevention efforts through horizon scanning. ENV.B.2/ETU/2014/0016 European Commission. https://ec.europa.eu/environment/nature/invasivealien/docs/Prioritising%20prevention%20efforts%20[th]rough%20horizon%20scanning.pdf

Rubel F, Brugger K, Haslinger K, Auer I (2017) The climate of the European Alps: Shift of very high resolution Köppen-Geiger climate zones 1800–2100. Meteorologische Zeitschrift (Berlin) 26(2): 115–125. https://doi.org/10.1127/metz/2016/0816

Ryman S, Anderberg A (1999) Five species of introduced duckweeds (Fem adventiva andmatsarter). Svensk Botanisk Tidskrift 93: 129–138.

Šajna N, Haler M, Škornik S, Kaligarič M (2007) Survival and expansion of *Pistia stratiotes* L. in a thermal stream in Slovenia. Aquatic Botany 87(1): 75–79. https://doi.org/10.1016/j.aquabot.2007.01.012

Saul W-C, Roy HE, Booy O, Carnevali L, Chen H-J, Genovesi P, Harrower CA, Hulme PE, Pagad S, Pergl J, Jeschke J (2017) Data from: Assessing patterns in introduction pathways of alien species by linking major invasion databases, Dryad, Dataset. https://doi.org/10.1111/1365-2664.12819

Schweingruber FH, Kučerová A, Adamec L, Doležal J (2020) Anatomic atlas of aquatic and wetland plant stems. Springer Nature Switzerland 16–17. https://doi.org/10.1007/978-3-030-33420-8

Starmühler W (2009) Vorarbeiten zu einer „Flora von Istrien", Teil XII. Carinthia II 199(119): 553–600.

Steen B, Cardoso AC, Tsiamis K, Nieto K, Engel J, Gervasini E (2019) Modelling hot spot areas for the invasive alien plant *Elodea nuttallii* in the EU. Management of Biological Invasions 10(1): 151–170. https://doi.org/10.3391/mbi.2019.10.1.10

Strayer DL (2010) Alien species in fresh waters: Ecological effects, interactions with other stressors, and prospects for the future. Freshwater Biology 55: 152–174. https://doi.org/10.1111/j.1365-2427.2009.02380.x

Tanner R, Branquart E, Brundu G, Buholzer S, Chapman D, Ehret P, Fried G, Starfinger U, van Valkenburg J (2017) The prioritisation of a short list of alien plants for risk analysis within the framework of the Regulation (EU) No. 1143/2014. NeoBiota 35: 87–118. https://doi.org/10.3897/neobiota.35.12366

Thiébaut G (2007) Non-indigenous aquatic and semiaquatic plant species in France. In: Gherardi F (Ed.) Biological Invaders in Inland Waters. Profiles, Distribution and Threats 2: 209–229. https://doi.org/10.1007/978-1-4020-6029-8_11

Thiébaut G, Di Nino F (2009) Morphological variations of natural populations of an aquatic macrophyte *Elodea nuttallii* in their native and in their introduced ranges. Aquatic Invasions 4(2): 311–320. https://doi.org/10.3391/ai.2009.4.2.2

Thiébaut G, Gillard M, Deleu C (2016) Growth, regeneration and colonisation of *Egeria densa* fragments: The effect of autumn temperature increases. Aquatic Ecology 50(2): 175–185. https://doi.org/10.1007/s10452-016-9566-3

Thouvenot L, Haury J, Thiebaut G (2013) A success story: Water primroses, aquatic plant pests. Aquatic Conservation 23: 790–803. https://doi.org/10.1002/aqc.2387

Tippery NP, Les DH (2020) Tiny plants with enormous potential: phylogeny and evolution of duckweeds. In: Cao XH, Fourounjian P, Wang W (Eds) The Duckweed genomes. Springer Nature Switzerland, 20–38. https://doi.org/10.1007/978-3-030-11045-1_2

Trinajstić I, Pavletić Z (1978) *Azolla filiculoides* Lam. in the Water-Plant Vegetation of Eastern Croatia. Acta Botanica Croatica 37: 159–162.

Troy W, Werier D, Nelson A (2022) New York flora atlas. New York Flora Association, Albany, New York.

Vélez-Gavilán J (2017) *Lemna aequinoctialis* (lesser duckweed). Invasive Species Compendium. CABI, Wallingford. https://doi.org/10.1079/ISC.121132.20203483098

Vilizzi L, Copp GH, Hill JE, Adamovich B, Aislabie L, Akin D, Al-Faisal AJ, Almeida D, Azmai MNA, Bakiu R, Bellati A, Bernier R, Bies JM, Bilge G, Branco P, Bui TD, Canning-Clode J, Cardoso Ramos HA, Castellanos-Galindo GA, Castro N, Chaichana R, Chainho P, Chan J, Cunico AM, Curd A, Dangchana P, Dashinov D, Davison PI, de Camargo MP, Dodd JA, Durland Donahou AL, Edsman L, Ekmekçi FG, Elphinstone-Davis J, Erős T, Evangelista C, Fenwick G, Ferincz Á, Ferreira T, Feunteun E, Filiz H, Forneck SC, Gajduchenko HS, Gama Monteiro J, Gestoso I, Giannetto D, Gilles Jr AS, Gizzi F, Glamuzina B, Glamuzina L, Goldsmit J, Gollasch S, Goulletquer P, Grabowska J, Harmer R, Haubrock PJ, He D, Hean JW, Herczeg G, Howland KL, İlhan A, Interesova E, Jakubčinová K, Jelmert A, Johnsen SI, Kakareko T, Kanongdate K, Killi N, Kim J-E, Kırankaya ŞG, Kňazovická D, Kopecký O, Kostov V, Koutsikos N, Kozic S, Kuljanishvili T, Kumar B, Kumar L, Kurita Y, Kurtul I, Lazzaro L, Lee L, Lehtiniemi M, Leonardi G, Leuven RSEW, Li S, Lipinskaya T, Liu F, Lloyd L, Lorenzoni M, Luna SA, Lyons TJ, Magellan K, Malmstrøm M, Marchini A, Marr SM, Masson G, Masson L, McKenzie CH, Memedemin D, Mendoza R, Minchin D, Miossec L, Moghaddas SD, Moshobane MC, Mumladze L, Naddafi R, Najafi-Majd E, Năstase A, Năvodaru I, Neal JW, Nienhuis S, Nimtim M, Nolan ET, Occhipinti-Ambrogi A, Ojaveer H, Olenin S, Olsson K, Onikura N, O'Shaughnessy K, Paganelli D, Parretti P, Patoka J, Pavia RTB, Jr Pellitteri-Rosa D, Pelletier-Rousseau M, Peralta EM, Perdikaris C, Pietraszewski D, Piria M, Pitois S, Pompei L, Poulet N, Preda C, Puntila-Dodd R, Qashqaei AT, Radočaj T, Rahmani H, Raj S, Reeves D, Ristovska M, Rizevsky V, Robertson DR, Robertson P, Ruykys L, Saba AO, Santos JM, Sarı HM, Segurado P, Semenchenko V, Senanan W, Simard N, Simonović P, Skóra ME, Slovák Švolíková K, Smeti E, Šmídová T, Špelić I, Srėbalienė G, Stasolla G, Stebbing P, Števove B, Suresh VR, Szajbert B, Ta KAT, Tarkan AS, Tempesti J, Therriault TW, Tidbury HJ, Top-Karakuş N, Tricarico E, Troca DFA, Tsiamis K, Tuckett QM, Tutman P, Uyan U, Uzunova E, Vardakas L, Velle G, Verreycken H, Vintsek L, Wei H, Weiperth A, Weyl OLF, Winter ER, Włodarczyk R, Wood LE, Yang R, Yapıcı S, Yeo SSB, Yoğurtçuoğlu B, Yunnie ALE, Zhu Y, Zięba G, Žitňanová K, Clarke S (2021) A global-scale screening of non-native aquatic organisms to identify potentially invasive species under current and future climate conditions. Science of the Total Environment 788: e147868. https://doi.org/10.1016/j.scitotenv.2021.147868

Vilizzi L, Hill JE, Piria M, Copp GH (2022) A protocol for screening potentially invasive non-native species using Weed Risk Assessment-type decision-support toolkits. Science of the Total Environment 832: e154966. https://doi.org/10.1016/j.scitotenv.2022.154966

Villamagna AM, Murphy BR (2010) Ecological and socio-economic impacts of invasive water hyacinth (*Eichhornia crassipes*): A review. Freshwater Biology 55(2): 282–298. https://doi.org/10.1111/j.1365-2427.2009.02294.x

Vuković N, Šegota V, Rimac A, Koletić N, Alegro A (2021) New records of alien plants–*Ludwigia peploides* (Kunth) PH Raven, *Reynoutria sachalinensis* (F. Schmidt) Nakai and *Nicotiana glauca* Graham in Croatia. Natura Croatica 30(1): 27–35. https://doi.org/10.20302/NC.2021.30.3

Weidlich EW, Flórido FG, Sorrini TB, Brancalion PH (2020) Controlling invasive plant species in ecological restoration: A global review. Journal of Applied Ecology 57(9): 1806–1817. https://doi.org/10.1111/1365-2664.13656

Xu Y, Ma S, Huang M, Peng M, Bog M, Sree KS, Appenroth K-J, Zhang J (2015) Species distribution, genetic diversity and barcoding in the duckweed family (Lemnaceae). Hydrobiologia 743(1): 75–87. https://doi.org/10.1007/s10750-014-2014-2

Zaninović K, Gajić-Čapka M, Perčec Tadić M, Vučetić M, Milković J, Bajić A, Cindrić K, Cvitan L, Katušin Z, Kaučić D, Likso T, Lončar E, Lončar Ž, Mihajlović D, Pandžić K, Patarčić M, Srnec L, Vučetić V (2008) Climate atlas of Croatia 1961–1990, 1971–2000. Meteorological and Hydrological Service, 1–172.

Zeko A, Šegota V, Vilović T, Koletić N, Alegro A (2020) Aquatic plants of Croatia: Data derived from the ZA herbarium collection. Natura Croatica 29(2): 205–216. https://doi.org/10.20302/NC.2020.29.27

Supplementary material 1

Table S1

Authors: Marina Piria, Tena Radočaj, Lorenzo Vilizzi, Mihaela Britvec

Data type: docx file

Explanation note: List of the 55 questions making up the Aquatic Species Invasiveness Screening Kit (AS-ISK).

Link: https://doi.org/10.3897/neobiota.76.83320.suppl1

Supplementary material 2

Combined AS-ISK report for the 24 non-native aquatic plant species screened for their potential risk of invasiveness in the Pannonian and Mediterranean regions of Croatia.

Authors: Marina Piria, Tena Radočaj, Lorenzo Vilizzi, Mihaela Britvec

Data type: pdf file

Explanation note: Combined AS-ISK report for the 24 non-native aquatic plant species screened for their potential risk of invasiveness in the Pannonian and Mediterranean regions of Croatia.

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RESEARCH ARTICLE

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Do non-native and dominant native species carry a similar risk of invasiveness? A case study for plants in Turkey

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Abstract

Most risk analysis studies in invasion biology have focused on the invasiveness of non-native species, even though some native species also can pose a high risk to the environment and human well-being. This is especially true under current global change, which may cause dominant native species to expand their range of distribution and have substantial effects on the ecosystem. In this study, the risk of invasiveness of five non-native and five native plant species in Turkey was evaluated using a standard risk screening protocol. All ten species selected for screening are known to be invasive in several parts of the world, i.e. non-native *Ailanthus altissima*, *Cuscuta campestris*, *Phytolacca americana*, *Robinia pseudoacacia* and *Sicyos angulatus*, and native *Cirsium arvense*, *Hedera helix*, *Onopordum acanthium*, *Phragmites australis* and *Sorghum halepense*. The Australian Weed Risk Assessment decision-support tool adapted to Turkey's geographical and climatic conditions was used for screening the study species based on their biological traits, ecology and management approaches. All species were classified as high-risk, with *R. pseudoacacia* among non-natives and *P. australis* among natives achieving the highest scores followed by *S. halepense*, *C. campestris*, *C. arvense*, *O. acanthium*, *P. americana*, *S. angulatus*, *A. altissima* and *H. helix*. Based on their risk scores, all non-native species were classified as invasive and all native species as 'expanding' for Turkey. An ordination based on the risk scores showed similarities between invasive and expanding species. The outcomes of this study indicate that species can have several risk-related traits resulting in high risk scores irrespective of their origin. Such species can modify their environment and interact with other species with severe consequences for biodiversity. It is argued that dominant species with highly negative environmental and socioeconomic impacts in their habitats should be included in priority lists for management measures irrespective of their origin (i.e. native or non-native). More studies are needed to evaluate the magnitude and prevalence of the present findings for other regions worldwide.

Keywords
Alien species, expansion, invasion, management, risk screening, Türkiye

Introduction

In the last decades, increased travel, trade and tourism in connection with globalisation and human population expansion have facilitated the deliberate and/or unintentional transport of plant and animal species beyond their natural biogeographical barriers (Hulme 2009; Şekercioğlu et al. 2011; Pyšek et al. 2017, 2020; Essl et al. 2019; Zenni et al. 2021). This has resulted in the introduction of non-native invasive species into new regions with consequent negative environmental and socioeconomic impacts (Pyšek et al. 2020). Under the challenging conditions of global change, a major task of invasion biology is to identify those high-risk species that are more likely to cause negative impacts. Usually, species that are either non-native invasive or 'expanding native' (Simberloff et al. 2012; Díaz et al. 2019; Essl et al. 2019; Simberloff 2022; Yazlık and Üremiş 2022) and that become dominant in natural habitats may exert direct or indirect impacts on community structure and composition, species interactions and ecosystem functions, all of which can result in a 'domino effect' (Hawkins et al. 2015; Pyšek et al. 2017, 2020; Hulme and Bernard-Verdier 2018; Díaz et al. 2019; Brundu et al. 2020).

Identifying species posing a high risk of invasiveness is sometimes challenging due to knowledge gaps in their biology/ecology, and this represents a limitation for the implementation of effective management and control measures (Hulme 2009; Hulme and Bernard-Verdier 2018; Yazlık et al. 2018; Pyšek et al. 2020). This is a crucial aspect in risk screening/identification (i.e. the first step in risk analysis followed by risk assessment,risk management and communication: e.g. Vilizzi et al. 2022) especially given current debate on whether non-native species can be considered as a contribution to the biodiversity of the invaded regions (Simberloff 2011; Simberloff et al. 2012; Pauchard et al. 2018), hence in contrast to evidence for their environmental and socioeconomic impacts (Hawkins et al. 2015; Nentwig et al. 2016; Rumlerová et al. 2016; Bacher et al. 2018; Yazlık et al. 2018; Starfinger and Schrader 2021). For this reason, the first step in the identification of potentially high-risk invasive species is to find out their native or non-native status in the regions where they are found (e.g. Uludağ et al. 2017) and then determine their potential environmental and socioeconomic impacts (Hawkins et al. 2015; Bacher et al. 2018; Pauchard et al. 2018; Tanner and Fried 2020; Yazlık and Albayrak 2020; EPPO 2021). This provides for an opportunity to select those species more likely to be selected for risk analysis (Hawkins et al. 2015; Nentwig et al. 2016; Rumlerová et al. 2016; Bacher et al. 2018; Yazlık et al. 2018).

An effective means to identify high-risk invasive species is by the use of risk screening decision-support tools (see Vilizzi et al. 2022). These allow to carry out follow-up risk assessment after identification of the species classified as carrying a high risk of invasiveness for a certain risk assessment area (Díaz et al. 2019; Lenzner et al. 2019; Pyšek et al. 2020). At the same time, the drivers of global change, such as climate and land-use activities and accessibility, can also cause an increase in the range of expansion and abundance of the native species, which are then referred to as expanding species. Some examples are the expansion of tall grass plants in the absence of large herbivores (Corazza et al. 2016), liana infestations in tropical forests following disturbance (Schnitzer and Bongers 2011), and graminoids and shrubs expanding in tundra as a result of climate change (McManus et al. 2012).

Several mechanisms including the availability of free niches and increased competitive ability are involved in the invasion process by non-native species (Catford et al. 2009; Hiero and Callaway 2021). Yet, several plant species within their native range behave like invasive plants (Pyšek et al. 2004; Simberloff 2011; Simberloff et al. 2012; Hejda et al. 2021; Yazlık and Üremiş 2022). Although there is an ongoing debate as to whether the impacts of non-native invasive plants differ from those of expanding native (dominant) plants (Simberloff et al. 2012, 2013; Hejda et al. 2021), there is solid evidence that both non-native species' invasions and the spread of dominant native species may pose threats to biodiversity and sustainability (Hejda et al. 2021; Yazlık and Üremiş 2022). This is also because native dominant plant species are likely to be invasive outside their native range (Pyšek et al. 2009; Phillips et al. 2010; Hejda et al. 2021). However, there are very few native dominant plant species that have been compared with non-native invasive plant species in terms of their fast spread and negative impacts on vegetation (e.g. Hejda 2013). It is therefore argued that dominant expanding species should be evaluated in a similar way to non-native species by risk analysis in order to understand the threats they may pose to the ecosystem (Sohrabi et al. 2020; Jan et al. 2022). Importantly, identifying potential invasion/expansion of these species by risk analysis will play an important role in preventing/mitigating environmental and socioeconomic impacts, especially in terms of biodiversity loss.

The aim of this study was to show that some dominant native plant species can pose a high risk of invasiveness as much as non-native plant species using a dataset from Turkey. To this end, a risk screening was conducted on ten plant species in Turkey that are registered as non-native invasive in several geographical regions worldwide. The specific objectives were to: (i) determine the invasion/expansion status of the study species in Turkey, and (ii) search for a relationship between the risk status of these species and their origin. The purpose of this study is to emphasise the necessity of approaching expanding species from an invasiveness perspective.

Methods

Species selection

Four criteria were used for selection of the plant species for screening. Firstly, species were selected that have a wide distribution in three biogeographic regions of Turkey, namely the Euro-Siberian, Iran-Turanian and Mediterranean (Bizim Bitkiler 2020). Secondly, species were selected for which no risk analysis studies have been conducted in Turkey, but are defined as non-native invasive plants in different parts of the world

Table 1. Information on the species screened for their risk of invasiveness in Turkey. EPPO code: code used for plant taxa by the European and Mediterranean Plant Protection Organization.

Species	Family	Origin	Lifetime and form	EPPO code
Non-native				
Ailanthus altissima (Mill.) Swingle	Simaroubaceae	China	Perennial tree	AILAL
Cuscuta campestris Yunck.	Convolvulaceae	America	Parasitic; climbing annual or perennial herb	CVCCA
Phytolacca americana L.	Phytolaccaceae	America	Polycarpic perennial herb	PHTAM
Robinia pseudoacacia L.	Fabaceae	America	Perennial tree	ROBPS
Sicyos angulatus L.	Cucurbitaceae	America	Climbing or creeping annual herb	SIYAN
Native				
Cirsium arvense (L.) Scop.	Asteraceae	Turkey	Polycarpic perennial herb	CIRAR
Hedera helix L.	Araliaceae	Turkey	Climbing or creeping perennial woody	HEEHE
Onopordum acanthium L.	Asteraceae	Turkey	Annual or biennial herb	ONRAC
Phragmites australis (Cav.) Trin. ex Steud.	Poaceae	Turkey	Perennial herb	PHRCO
Sorghum halepense (L.) Pers.	Poaceae	Turkey	Perennial herb	SORHA

(GISD 2022). Of note, this type of selection has been proposed for studies comparing invasive non-native species with native species (van Kleunen et al. 2010). These first two criteria enabled the selection of species with a high potential for impacts whilst their risk of invasiveness was not known beforehand. Thirdly, species were selected that have biological traits of invasiveness. To this end, the species' life-history, biological, morphological and physiological traits were evaluated and the following were considered: adaptation to different habitats, soil type, pH range, competitive abilities, presence of below- and above-ground structures, and high generative and/or vegetative capacity. Lastly, species were selected that have high environmental and socioeconomic impacts, such as negative effects on natural vegetation, allelochemical contents, and toxic and/or injurious to humans and animals (Yazlık et al. 2017; Yazlık et al. 2018; Yazlık and Albayrak 2020; Aksan and Yazlık 2021). Conducting a risk analysis on non-native species and determining their invasiveness status was suggested in previous studies for Turkey (Uludağ et al. 2017; Yazlık et al. 2018; Yazlık 2022). Notably, the study species were not limited to pairs of native and non-native species with certain traits or habitat features, which would make drawing generalisable conclusions more difficult. On the contrary, the objective was to select species with a similar level of invasiveness but different origin and habitat. As a result, five non-native and five native species were selected: *Ailanthus altissima*, *Cuscuta campestris*, *Phytolacca americana*, *Robinia pseudoacacia* and *Sicyos angulatus* as non-native, and *Cirsium arvense*, *Hedera helix*, *Onopordum acanthium*, *Phragmites australis* and *Sorghum halepense*as as native (Table 1).

Risk screening

For risk screening, a decision-support tool adapted from the Australian Weed Risk Assessment (WRA: Pheloung et al. 1999) was used accounting for the geographical and climatic conditions of Turkey, namely the Türkiye Weed Risk Assessment: TR-WRA (Suppl. material 1: Table S1). The screening protocol for the TR-WRA involves 49 questions dealing with the species' biological traits, environmental impacts and management planning. The following modifications were done to the original set of questions (Qs): (i) 'suitability of the species to Australian climate' was changed to 'suitability of the species to the climate in Turkey' (Q 2.1); (ii) 'native or naturalised in regions with extended dry periods' was changed to 'native or naturalised in regions with a mild climate' (Q 2.4); (iii) 'presence of effective natural enemies in Australia' was changed to 'presence of effective natural enemies in Turkey' (Q 8.5). For each answered question, the species is assigned a score between –2 and 2, and the Q-specific scores are then summed to produce a total risk score (RS), which ranges from a minimum of –14 to a maximum of 29. However, in the question about the quality of climate matching data (Q 2.2), as all screened species scored high (i.e. with 2 points) and their natural ranges are well known (Table 1), the scores were not included in the RS, and these scores were not shown in the risk analysis table (Suppl. material 1: Table S1). In addition, 'no' or 'unknown' was added to the choice of some questions that were not related to the study species or for unknown risks (Suppl. material 1: Table S1).

As no RS thresholds for invasiveness identification were set by the authors who designed the protocols for the A-WRA test (Pheloung et al. 1999; Andreu and Vilà 2010), after accounting for similar risk outcomes scoring higher than the maximum value (e.g. Morais et al. 2017), the RS was modified to being ≥ 29. Also, at least ten answers are required for the evaluation of a species (Andreu and Vilà 2010; Morais et al. 2017). Overall, following Andreu and Vilà (2010), the TR-WRA scoring system can be used to classify species into three groups according to their level of risk: (i) species' occurrence in the risk assessment area acceptable (score < 1); (ii) species introduction in the risk assessment area prohibited (score > 6); or (iii) further work needed for a reliable risk screening outcome (score between 1 and 6). If a native species is identified in the second group, this implies that species management is required.

Data collection and statistical analysis

The information required to answer each question was obtained from national and international literature. Search for literature was conducted in Google Scholar, Web of Science, Scopus and ULAKBİM (Suppl. material 1: Table S2). In addition, a monograph (Tanner and Fried 2020), data sheets (EPPO 2010, 2021), one 'grey literature' reference (Köstekçi 2010), and online databases (European Project DAISIE: http://www.europe-aliens.org/; USDA Plants database http://plants.usda.gov; International Survey of Herbicide Resistant Weeds: http://www.weedscience.org; Global Invasive Species Database: http://www.issg.org/database) were used.

Multiple Correspondence Analysis (MCA), which is suitable for ordination of categorical data (Abdi and Williams 2010), was employed to visualise variation in the species' risk scores and their relationship with the species' origin (i.e. non-native vs native), and to identify similar species in terms of scores. The output of MCA can be interpreted similar to a quantitative ordination, with species closer to each other having higher similarities in their scores. Function *mca* of package *FactoMineR* (Le et al. 2008) was used to implement MCA in R version 3.5.0 (R Core Team 2020). Before analysis, the scores for a total of 17 questions with the same scores for all species were omitted from the dataset as they did not carry useful information for an ordination. These questions included: naturalised or not (Q 1.2 in Suppl. material 1: Table S1), climatic suitability (Q 2.1), climate match (Q 2.2), environmental versatility (Q 2.3), repeated introductions (Q 2.5), garden/amenity/disturbance weed (Q 3.2), agricultural weed (Q 3.3), environmental weed (Q 3.4), allelopathy (Q 4.2), host for unwanted species (Q 4.6), plant of infertile soils (Q 4.10), geophyte (Q 5.4), reproductive failure (Q 6.1), viable seeds (Q 6.2) pollinator requirement (Q 6.5), unintentional dispersion (Q 7.1) and prolific seed production (Q 8.1). Furthermore, Q 6.4 with 'unknown' as an answer was removed from the dataset.

Results

Following risk screening, all ten species were found to carry a high risk of invasiveness (Suppl. material 1: Table S1). The species with the highest scores were *R. pseudoacacia* (RS = 32) among non-natives and *P. australis* (40) among natives, followed by *S. halepense* (33), *C. arvense* and *C. campestris* (31), *O. acanthium* and *P. americana* (30), *S. angulatus* (29), *A. altissima* and *H. helix* (28). Based on these scores, all non-native species were risk-ranked as invasive and all native species as expanding for Turkey. All species were recorded in various habitats, predominantly agricultural but also sandy, saline, rocky and ruderal (Table 2).

Although these species have very different characteristics from each other, similar scores were achieved in the sub-categories related to their dominant characters. For example, when the dispersal mechanism (Section 7: Suppl. material 1: Table S1) was analysed, the total score range of the species changed between 5 and 8 according to the character-

Table 2. Habitats in Turkey of the species under study (for evidence, see Section 3 in Suppl. material 1: Table S1).

Habitat	*A. altissima*	*C. arvense*	*C. campestris*	*H. helix*	*O. acanthium*	*P. americana*	*P. australis*	*R. pseudoacacia*	*S. angulatus*	*S. halepense*
Arable	*	*	*	*	*	*	*	*	*	*
Dryland	-	*	*	-	*	-	-	-	-	-
Forest	*	-	-	*	-	*	*	*	*	*
Grassland	*	*	*	-	*	*	*	*	-	*
Riparian	*	*	*	*	*	*	*	*	*	*
Rocky	*	*	*	*	*	-	*	*	-	*
Ruderal	*	*	*	*	*	*	*	*	*	*
Saline	*	*	*	*	*	*	*	*	-	*
Sandy	*	*	*	*	*	*	*	*	-	*

istics of the plant (Q 7.4: Suppl. material 1: Table S1). In addition, the grading of questions or the absence of a certain feature affected the total score. For example, while *C. arvense*, *O. acanthium*, *P. australis* and *R. pseudoacacia* achieved the highest score of 2 in this section, *H. helix* and *S. halepense* achieved a score of −1, and other species were scored in the range of 0 to 1. Thus, a species that is known to have definite spread by wind and a species that is likely to drift to a limited area in very strong winds are not given the same score. Although the question-specific risk scores for the native plant species were mostly either negative or 0 because of their origin, this did not affect their total (high-risk) score. For example, since natural enemies of *C. arvense* and *O. acanthium* are in a limited range and not used as biological control agents, both plants scored −1 instead of the lowest score of −2 for the question (Q 8.5) related to the presence of natural enemies.

The two dimensions of the MCA analysis explained ≈39% of the variation in the data (Fig. 1; Suppl. material 1: Table S3). Amongst the 69 attributes included, those with the highest contribution to the first axis of variation were absence of natural enemies and naturalisation outside the native range (Suppl. material 1: Table S4). For the second axis, those variables were no wind dispersal and properties of propagule banks (Suppl. material 1: Table S4 and Fig. S1). The first axis represented a gradient from two native species with no naturalisation but natural enemies (i.e. *C. arvense* and *O. acanthium*) to non-native species, which were all located in the negative part of the second axis. Non-native *C. campestris* was very close to the above two natives, indicating similarity in scores. The three other native species (i.e. *H. helix*, *P. australis* and *S. halepense*) were located far from the other natives along the secondary axis, indicating weak similarity. Based on the answers to the risk screening questionnaire, native and non-native species were not clearly grouped in the ordination space. Overall, MCA showed that similarity in terms of risk can be high amongst species of native and non-native species and low between two native species.

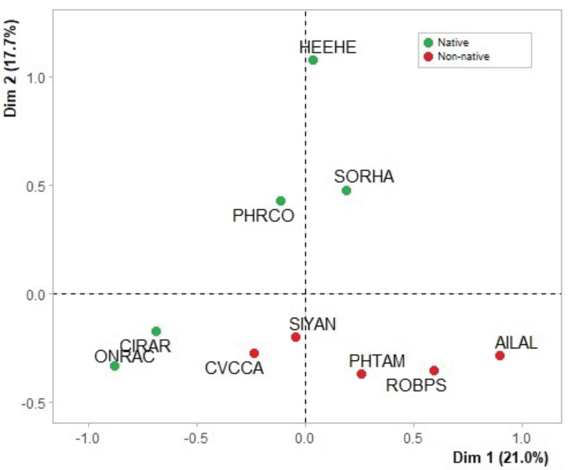

Figure 1. Multiple correspondence analysis factor map of the 10 plant species screened for their risk of invasiveness for Turkey according to their risk scores. Species labelled with their EPPO codes (see Table 1).

Discussion

This study has shown similar risk levels for non-native and native species with high potential of exerting negative impacts on both ecosystems and human well-being. In addition, this study is the first to provide a dataset of national evaluation for Turkey on the invasion/expansion status of ten dominant plant species that are registered as non-native invasive plants in different geographies (GISD 2022), but whose risk status has so far been unknown in this country. The present results showed that the invasion/expansion status of dominant plants may be independent of their local range, thereby emphasising the importance to evaluate species not only according to their biogeographical origin but also to their biological, morphological and physiological characteristics as well as environmental and socioeconomic impacts. Clearly, more studies relying on larger sample sizes are needed to quantify the magnitude and prevalence of this first evidence provided for Turkey.

The ten species under study were interpreted in two groups by accounting for their local distributional range, risk scores and human-induced dispersal. Accordingly, native species were considered as 'expanding apophytic', which are quite aggressive, spread rapidly and affect vegetation (Yücel et al. 2019; Hejda et al. 2021), whereas non-native species were considered as 'invasive anthropophytes' (*sensu* Yücel et al. 2019). All ten species have a dominant distribution in various habitats (i.e. agriculture, coastal, forest: Table 2) and human influence has a high share in their spread. In this respect, the most important factors are the 'weed' status of these species in agricultural habitats combined with their competitive abilities such as morphological characteristics (Yazlık and Tepe 2001; Kaçan and Boz 2015; Uludağ 2015; Üstüner et al. 2015; Sezer and Kolören 2019; Terzioğlu and Ergül Bozkurt 2020; Yazlık and Albayrak 2020; Aksan and Yazlık 2021). Specifically, clonal growth (Bímová et al. 2003), high biomass (van Kleunen et al. 2010; Hejda 2013; Canavan et al. 2019) and a large number of branches/tillerings (Hejda 2013; Yazlık and Üremiş 2022) were all traits associated with high invasiveness. In addition, serious problems have been reported regarding the presence of these species in their habitats (Table 2), which is a major reason for their high-risk scores, hence irrespective of their origin. Below, details are provided as to why these non-native and native species were found to carry a similar level of risk.

Native *O. acanthium* has negative impacts due to its superior competitiveness, spread and unpalatability based on its thorny structure, seed volatiles and re-sprout from root shoots, all of which cause vegetation degradation, decrease in agricultural production, injury in animals,deterioration of livestock nutrition, and labour costs (Pınar et al. 2018; Aksan et al. 2019; Aksan and Yazlık 2021). This is similar to native *C. arvense*, *P. australis* and *S. halepense*, which have dominant generative and vegetative propagation abilities (Suppl. material 1: Table S1). These species also create dense populations in habitats such as beaches and sand dunes, especially in agricultural and pasture lands, causing serious negative impacts on vegetation (Yazlık and Tepe 2001; Köstekçi 2010; Meyerson et al. 2010; Aksan et al. 2019; Aksan and Yazlık 2021; Erbaş and Doğan 2022; Jan et al. 2022; Yazlık and Üremiş 2022).

Hedera helix is present primarily in forests and urban habitats (Table 2) but also in agricultural habitats (e.g. nurseries, hazelnut orchards: Yazlık et al. 2019; Aksoy and Çelik 2020; Güneş Özkan et al. 2020). One of the main factors for the prevalence of this species in urban habitats is its use as an ornamental plant in parks or home gardens, while at the same time this species has a major impact on plant community composition in forests. *Cuscuta campestris*, *P. americana* and *S. angulatus* are naturalised non-native plants in Turkey that appear to occupy more than one habitat (Table 2). A parasitic plant with a wide host range, *C. campestris*, which is one of the species with the highest impacts worldwide (Yazlık et al. 2017), exerts major negative impacts by infecting cultivated plants in agricultural habitats, affecting rail ballast in railways, increasing fire risk, and being toxic to humans and animals (Yazlık and Albayrak 2020). Finally, non-native *P. americana* and *S. angulatus* are found in agricultural habitats that are generally considered to pose serious problems to agricultural production (Terzioglu and Ansin 1999; Korkmaz et al. 2016; Sezer and Kolören 2019).

The present risk screening study also determined the potential of non-native species to cause indirectly high risks in terms of plant diseases and nematode transmission in the areas where they are found (Suppl. material 1: Table S1). For example, *P. americana* is reported to provide suitable host conditions for five different nematodes (i.e. *Meloidogyne arenaria*, *M. floridensis*, *M. incognita*, *M. javanica* and *M. mayaguensis*: Kaur et al. 2007). Although there is no record of nematodes that are a problem for this species in Turkey, three nematodes reported by Kaur et al. (2007) are present in the country, namely *M. arenaria*, *M. incognita* and *M. javanica* (Özarslandan and Elekçioğlu 2010). Therefore, the interaction of *P. americana* with existing nematodes in the habitats of Turkey may create secondary problems by enhancing their further spread. This is especially important for arable lands, as there is evidence of damage by nematodes on cultivated plants (e.g. Özarslandan and Elekçioğlu 2010). Conversely, in terms of host or vector status of disease agents, *C. campestris* is a vector for virus and phytoplasma diseases (Yazlık and Albayrak 2020), whereas *S. angulatus* poses a high risk by being host to the watermelon mosaic virus (WMV-2: Korkmaz et al. 2016). Another example of a host is for *R. pseudoacacia*, which has the host status of *Viscum album* – a most problematic weed for many orchards in Turkey (Üstüner et al. 2015). Therefore, this non-native plant can contribute to the distribution of this parasitic plant.

Human-mediated dispersal was an important factor for the high risk of invasiveness identified in this study. Evidence shows that some of the screened species have often been reported as problematic weeds in agricultural areas and their prevalence may be due to their dispersal via contaminated agricultural tools and equipment with plant parts (Suppl. material 1: Table S1). Furthermore, transportation via road corridors can be an important channel for plant invasions (Lemke et al. 2021), as in the case of *C. campestris* (Yazlık and Albayrak 2020). Moreover, cultivation of *R. pseudoacacia*, which started 70 years ago in Turkey, is supported on the basis that it provides important socioeconomic benefits, such as erosion control, honey production with increased nectar provision, and timber use (BOEP 2013; Onur and Acar 2017). Therefore, the dispersal of some non-native plants, including *A. altissima*, can occur with direct human contribution due to their economic value. This is in agreement with the contextual assessment made by Vítková et al. (2020) in the decision to cultivate *R. pseudoacacia* in its non-native ranges. Therefore, the decision to continue the cultivation of high-risk non-native plants in Turkey as discussed in this study should be considered depending on the regional, ecological, conservation and socio-economic context.

The long-term presence of the study species were considered as another factor supporting their widespread distribution. For instance, *A. altissima*, *P. australis*, *R. pseudoacacia* and *S. halepense* not only in Turkey but also in several other regions worldwide is known to be widespread (POWO 2021). This was reflected by these species' high-risk scores because many species with long residence time are more likely to have a niche and geographic spread (Sychrová et al. 2022). At this stage, it should be taken into account that long residence time may also create problems in control studies of related species, even if native. For example, it has been reported that the herbicide Glyphosate applied at the edges of irrigation canals was not fully successful to combat *P. australis* in the Aydın plain, which is one of the most important polyculture crop production plains in the Aegean region of Turkey. This is because this species has a long-term persistent population in those ruderal habitats and integrated applications by mowing along the canal sides also cannot be made (Erbaş and Doğan 2022).

The species screened in this study are also affected by human activities (intentionally and/or unintentionally) besides spread and establishment in various habitats (Table 2). Amongst the different habitats, it has been emphasised that arable land is the most occupied by non-native plants, whereas natural and semi-natural grasslands are less invaded (Chytrý et al. 2008; Pyšek et al. 2009; Jauni and Hyvönen 2010). For instance, among the study species, *P. americana* has been reported from agricultural, forest and coastal habitats but as problematic especially in arable lands, due to shading and harvesting difficulties, such as for kiwi fruit (Sezer and Kolören 2019) and tea (Terzioğlu and Ergül Bozkurt 2020). Similarly, the screened native species have also been reported in several habitats including arable lands (Yazlık and Tepe 2001; Kaçan and Boz 2015; Yazlık and Üremiş 2022). For instance, the incidence of *P. australis* was determined as 48% and the density as 12 plants/m^2 in traditional vineyards of Manisa province in the Aegean Region of Turkey (Kaçan and Boz 2015).

Dominant native species can also cause demographic issues as a result of human-induced changes to the environment (Valéry et al. 2009; Simberloff 2011; Méndez et al. 2014; Sohrabi et al. 2020; Jan et al. 2022) thereby posing management challenges under current scenarios of global change (Simberloff 2011; Méndez et al. 2014). Nevertheless, *P. australis* (the native species with the highest risk score in this study) has also socioeconomic aspects on the country's trade and local people in the Sultan Marsh Nature Park, which is included in the List of Class A Wetlands in accordance to the second and third articles of the International Ramsar Convention in Turkey (Ramsar site no. 661 - https://rsis.ramsar.org/ris/661). Approximately 1500 tons of reeds (i.e. *P. australis* and *Typha* spp.) are cut annually by the local people in Sultan Marsh with most of the cut reeds being exported. The amount of thatch exported is approximately

300,000–400,000 bundles per year, and in 1995 a reed tying and storage facility was established in the town of Sindelhöyük. In addition, reeds (especially *P. australis*, which is a pure community represented by almost a single species in Yay Lake in the south and southwest areas of the Sultan marshes: Hamzaoğlu and Aksoy 2006) are used as roofing material (thatched roof) and animal feed in the region, where they represent an important source of income (Karadeniz 2000; Hamzaoğlu and Aksoy 2006; Sarısoy 2015). As a result, it is recommended that native (*P. australis*) and non-native (*A. altissima* and *R. pseudoacacia*) high-risk species with socioeconomic contributions should be monitored across Turkey and context-dependent prevention and management approaches should be developed in case of local adverse impacts.

The presence of natural enemies to native species is another important criterion to determine their risk of invasiveness (Q 8.5: Suppl. material 1: Table S1). For example, despite the existence of natural enemies for *O. acanthium* such as *Homoeosoma nebulellum* (Lepidoptera, Pyralidae) and *Larinus latus* (Coleoptera, Curculionidae) (Gültekin 2008; Yücel and Çobanoğlu 2016), the potential of these insects as biological control agents is limited (Gültekin 2008; Yücel and Çobanoğlu 2016). This is also true of *C. arvense*, whose natural enemies are recorded in its local distributional range (Kedici et al. 1994). Therefore, control of these plant species by such natural enemies may be limited to areas where these agents are present. For this reason, it is suggested that studies should be carried out to investigate the role of such natural enemies for an effective control and to identify related plant species in Turkey as biological control agents.

Due to their high risk of invasiveness, all species screened in this study (and regardless of their origin) should be listed as priority species. Sustainability of existing native species and reducing or stopping the negative impacts of invasive/expanding species can be possible by prevention. To achieve this objective, awareness-raising activities, training and effective species-specific management programmes (including the use of clean equipment in production areas, human-induced transportation of plant parts, Integrated Weed Management (IWM) application methods, and the use of non-native ornamental plants) should be organised based on the species' habitat. Effective management programmes are also important in terms of setting precautionary measures in plant transitions from Turkey to different geographies, as indicated by the large number of weed species originating from Turkey and being invasive or naturalised in different geographies/continents worldwide (A. Yazlık, unpublished data). To this end, implementation of effective biosecurity measures and cooperation amongst stakeholder groups would help in such efforts (Guo 2006; Lenzner et al. 2019; Pyšek et al. 2020; Wallingford et al. 2020; Yang et al. 2021).

Overall, if high-risk species disperse into areas other than their native habitats or geographic regions, additional risks may arise and the extent of the resulting impacts may increase. Further environmental and socioeconomic impacts can be expected in range-shifting non-native species due to hybridisation (Essl et al. 2019; Wallingford et al. 2020; Seebens et al. 2021). However, this requires an understanding of their potential interactions in new environments (Guo 2006; Wallingford et al. 2020; Seebens et al. 2021) as well as of the extent of such impacts (Wallingford et al. 2020; Simberloff 2022). All of this would require monitoring programmes and gathering local ecological information. For these reasons, it is believed that the present study can broaden the perspective about native and non-native species and add new data to the knowledge of related plants.

Conclusions

The present study has provided evidence for how both non-native and native species can result in high-risk scores of invasiveness independent of their native range. This suggests that further studies should be carried out on the extent and size of the impact exerted by such species. As research on invasiveness has been strongly focused on non-native species, it is hoped that the present study will point to the necessity of working on dominant native (expanding) species. Considering the results of the ten species investigated, it is suggested that further studies in risk analysis should include not only non-native species but also all dominant species that are known to cause high impacts. This is because damage to natural ecosystems is in most cases an irreversible process (Křivánek and Pyšek 2006; Brundu et al. 2020). Moreover, considering all aspects of socioeconomic and environmental changes at the national level provides a resource to monitor more effectively the potential developments of future biological invasions (Latombe et al. 2022). Therefore, it is suggested that invasive/expanding species lists should be created on a regional basis in view of risk analysis studies. At the same time, it is recommended that priority should be given to the establishment of management programmes (Brundu et al. 2020) and the implementation of effective biosecurity measures (Latombe et al. 2022) for species whose invasive/expanding status has been determined by risk analysis. Given the presence of the species screened in this study in different habitats across Turkey, appropriate management programmes should be implemented by taking into account the IWM principle. In particular, it is recommended that research institutes working on biological control in Turkey (e.g. Adana Biological Control Research Institute, which carries out studies on mass insect production) should consider the research on the natural enemies mentioned in this study. Finally, considering urban habitats, public awareness should be raised and decision-makers should be informed about the use of high-risk plants such as *A. altissima*, *H. helix* and *R. pseudoacacia*, which are sold and used as ornamental and/or landscape plants country-wide.

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References

Abdi H, Williams JL (2010) Principal component analysis. John Wiley and Sons, Inc. WIREs Computational Statistics 2(4): 433–459. https://doi.org/10.1002/wics.101

Aksan UA, Yazlık A (2021) The plant species and their impacts in pasture areas: A case study from Düzce central district. Akademik Ziraat Dergisi 10: 81–96. https://doi.org/10.29278/azd.797748 [In Turkish]

Aksan UA, Kuşkapan Ö, Yazlık A (2019) The impacts of wild plant species on animals in the meadow - pasture areas. International Conference on Agriculture and Rural Development (ISPEC), Bildiriler Kitabı, 10–12 Haziran, Siirt, Türkiye, 16–36. [In Turkish]

Aksoy A, Çelik J (2020) Vascular plant diversity of the Alanya Castle walls and their ecological effects. Biological Diversity and Conservation 13: 9–18. https://doi.org/10.46309/biodicon.2020.731423

Andreu J, Vilà M (2010) Risk analysis of potential invasive plants in Spain. Journal of Nature Conservation 18(1): 34–44. https://doi.org/10.1016/j.jnc.2009.02.002

Bacher S, Blackburn TM, Essl F, Jeschke JM, Genovesi P, Heikkilä J, Jones G, Keller R, Kenis M, Kueffer C, Martinou AF, Nentwig W, Pergl J, Pyšek P, Rabitsch W, Richardson DM, Roy HE, Saul W-C, Scalera R, Vilà M, Wilson JRU, Kumschick S (2018) Socio-economic impact classification of alien taxa (SEICAT). Methods in Ecology and Evolution 9(1): 159–168. https://doi.org/10.1111/2041-210X.12844

Bímová K, Mandák B, Pyšek P (2003) Experimental study of vegetative regeneration in four invasive *Reynoutria* taxa (Polygonaceae). Plant Ecology 166(1): 1–11. https://doi.org/10.1023/A:1023299101998

Bizim Bitkiler (2020) Bizim Bitkiler Version 3.1. https://www.bizimbitkiler.org.tr

BOEP (2013) Honey Forest Action Plan 2013–2017. T.C. Orman ve Su İşleri Bakanlığı Orman Genel Müdürlüğü. https://www.ogm.gov.tr/ekutuphane/Yayinlar

Brundu G, Pauchard A, Pyšek P, Pergl J, Bindewald AM, Brunori A, Canavan S, Campagnaro T, Celesti-Grapow L, Dechoum MdeS, Dufour-Dror J-M, Essl F, Flory SL, Genovesi P, Guarino F, Guangzhe L, Hulme PE, Jäger H, Kettle CJ, Krumm F, Langdon B, Lapin K, Lozano V, Le Roux JJ, Novoa A, Nuñez MA, Porté AJ, Silva JS, Schaffner U, Sitzia T, Tanner R, Tshidada N, Vítková M, Westergren M, Wilson JRU, Richardson DM (2020) Global guidelines for the sustainable use of non-native trees to prevent tree invasions and mitigate their negative impacts. NeoBiota 61: 65–116. https://doi.org/10.3897/neobiota.61.58380

Canavan S, Meyerson LA, Packer JG, Pyšek P, Maurel N, Lozano V, Richardson DM, Brundu G, Canavan K, Cicatelli A, Čuda J, Dawson W, Essl F, Guarino F, Guo WY, van Kleunen M, Kreft H, Lambertini C, Pergl J, Skálová H, Soreng RJ, Visser V, Vorontsova MS, Weigelt P, Winter M, Wilson JRU (2019) Tall-statured grasses: A useful functional group for invasion science. Biological Invasions 21(1): 37–58. https://doi.org/10.1007/s10530-018-1815-z

Catford JA, Jansson R, Nilsson C (2009) Reducing redundancy in invasion ecology by integrating hypotheses into a single theoretical framework. Diversity & Distributions 15(1): 22–40. https://doi.org/10.1111/j.1472-4642.2008.00521.x

Chytrý M, Maskell LC, Pino J, Pyšek P, Vilà M, Font X, Smart SM (2008) Habitat invasions by alien plants: A quantitative comparison among Mediterranean, subcontinental and oceanic regions of Europe. Journal of Applied Ecology 45(2): 448–458. https://doi.org/10.1111/j.1365-2664.2007.01398.x

Corazza M, Tardella FM, Ferrari C, Catorci A (2016) Tall grass invasion after grassland abandonment influences the availability of palatable plants for wild herbivores: Insight into the conservation of the apennine chamois *Rupicapra pyrenaica ornata*. Environmental Management 57(6): 1247–1261. https://doi.org/10.1007/s00267-016-0679-1

Díaz S, Settele J, Brondízio ES, Ngo HT, Guèze M, Agard J, Arneth A, Balvanera P, Brauman KA, Butchart SHM, Chan KMA, Garibaldi LA, Liu IJ, Subramanian SM, Midgley GF, Miloslavich P, Molnár Z, Obura D, Pfaff A, Polasky S, Purvis A, Razzaque J, Reyers B, Chowdhury RR, Shin YJ, Visseren-Hamakers IJ, Willis KJ, Zayas C (2019) Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES). https://doi.org/10.5281/zenodo.3553579

EPPO (2010) PM 9/12 (1): *Sicyos angulatus*. EPPO Bulletin 40(3): 396–398. https://doi.org/10.1111/j.1365-2338.2010.02413.x

EPPO (2021) EPPO Invasive alien plants. http://www.eppo.org/

Erbaş F, Doğan N (2022) The effect of Glyphosate applications on *Phragmites australis* (Cav.) Trin. ex Steud (Common reed) at different periods. Adnan Menderes Üniversitesi Ziraat Fakültesi Dergisi 19: 131–137. https://doi.org/10.25308/aduziraat.1062893 [In Turkish]

Essl F, Dullinger S, Genovesi P, Hulme PE, Jeschke JM, Katsanevakis S, Kühn I, Lenzner B, Pauchard A, Pyšek P, Rabitsch W, Richardson DM, Seebens H, van Kleunen M, van der Putten WH, Vilà M, Bacher S (2019) A conceptual framework for range-expanding species that track human-induced environmental change. Bioscience 69(11): 908–919. https://doi.org/10.1093/biosci/biz101

GISD (2022) Global Invasive Species Database. http://www.issg.org/database

Gültekin L (2008) Host plants of *Larinus latus* (Herbst 1784) in eastern Turkey (Coleoptera, Curculionidae). Weevil News 40: 1–7.

Güneş Özkan N, Yazlık A, Jabran K (2020) Naturally distributed *Heracleum* L. taxa, their habitats and floristic composition of these habitats in Düzce. Eurasian Journal of Forest Science 8: 264–284. https://doi.org/10.31195/ejejfs.784797 [In Turkish]

Guo Q (2006) Intercontinental biotic invasions: What can we learn from native populations and habitats? Biological Invasions 8(7): 1451–1459. https://doi.org/10.1007/s10530-005-5834-1

Hamzaoğlu E, Aksoy A (2006) A phytosological study on the halophytic communities of Sultansazlığı (Inner Anatolia-Turkey). Ekoloji 15: 8–15. [In Turkish]

Hawkins CL, Bacher S, Essl F, Hulme PE, Jeschke JM, Kühn I, Kumschick S, Nentwig W, Pergl J, Pyšek P, Rabitsch W, Richardson DM, Vilà M, Wilson JRU, Genovesi P, Blackburn TM (2015) Framework and guidelines for implementing the proposed IUCN Environmental Impact Classification for Alien Taxa (EICAT). Diversity & Distributions 21(11): 1360–1363. https://doi.org/10.1111/ddi.12379

Hejda M (2013) Do species differ in their ability to coexist with the dominant alien *Lupinus polyphyllus*? A comparison between two distinct invaded ranges and a native range. NeoBiota 17: 39–55. https://doi.org/10.3897/neobiota.17.4317

Hejda M, Sádlo J, Kutlvašr J, Petřík P, Vítková M, Vojík M, Pyšek P, Pergl J (2021) Impact of invasive and native dominants on species richness and diversity of plant communities. Preslia 93(3): 181–201. https://doi.org/10.23855/preslia.2021.181

Hiero JL, Callaway RM (2021) The ecological importance of allelopathy. Annual Review of Ecology, Evolution, and Systematics 52(1): 25–45. https://doi.org/10.1146/annurev-ecolsys-051120-030619

Hulme PE (2009) Trade, transport and trouble: Managing invasive species pathways in an era of globalization. Journal of Applied Ecology 46(1): 10–18. https://doi.org/10.1111/j.1365-2664.2008.01600.x

Hulme PE, Bernard-Verdier M (2018) Comparing traits of native and alien plants: Can we do better? Functional Ecology 32(1): 117–125. https://doi.org/10.1111/1365-2435.12982

Jan I, Yaqoob S, Reshi ZA, Rashid I, Shah MA (2022) Risk assessment and management framework for rapidly spreading species in a Kashmir Himalayan Ramsar site. Environmental Monitoring and Assessment 194(3): e175. https://doi.org/10.1007/s10661-022-09764-5

Jauni M, Hyvönen T (2010) Invasion level of alien plants in semi-natural agricultural habitats in boreal region. Agriculture, Ecosystems & Environment 138(1–2): 109–115. https://doi.org/10.1016/j.agee.2010.04.007

Kaçan K, Boz Ö (2015) The comparison and determination of the weed species in conventional and organic vineyards. Ege Üniversitesi Ziraat Fakültesi Dergisi 52: 169–179. https://doi.org/10.20289/euzfd.18678 [In Turkish]

Karadeniz N (2000) Sultansazlığı, Ramsar Site in Turkey. Humedales Mediterráneos 1: 107–114. https://doi.org/10.1080/14683840008721223 [In Turkish]

Kaur R, Brito JA, Rich JR (2007) Host suitability of selected weed species to five Meloidogyne species. Nematropica 37: 107–120.

Kedici R, Melan K, Erciş A, Ural H (1994) Phytophagous insects detected in important weeds in the cereal fields of Ankara and Çankırı provinces and their evaluation in terms of biological control. Türkiye 3. Biyolojik Mücadele Kongresi (25–28 Ocak 1994), İzmir, Türkiye, 309–320. [In Turkish]

Korkmaz F, Karaca K, Özaslan C, Yanar Y, Önen H (2016) *Sicyos angulatus*: A natural host of Watermelon Mosaic Virus (WMV-2). Turkish Journal of Weed Science 19: 1–5. [In Turkish]

Köstekçi S (2010) Comparative morphological research on *Cirsium* Mill. Sect. *Cirsium* species distributed in Turkey. Master Thesis (with English Abstract), İnönü Üniversitesi, Malatya. [In Turkish]

Křivánek M, Pyšek P (2006) Predicting invasions by woody species in a temperate zone: A test of three risk assessment schemes in the Czech Republic (Central Europe). Diversity & Distributions 12(3): 319–327. https://doi.org/10.1111/j.1366-9516.2006.00249.x

Latombe G, Seebens H, Lenzner B, Courchamp F, Dullinger S, Golivets M, Kühn I, Leung B, Roura-Pascual N, Cebrian E, Dawson W, Diagne C, Jeschke JM, Pérez-Granados C, Moser D, Turbelin A, Visconti P, Essl F (2022) Capacity of countries to reduce biological invasions. Sustainability Science. https://doi.org/10.1007/s11625-022-01166-3

Le S, Josse J, Husson F (2008) FactoMineR: An R Package for Multivariate Analysis. Journal of Statistical Software 25(1): 1–18. https://doi.org/10.18637/jss.v025.i01

Lemke A, Buchholz S, Kowarik I, Starfinger U, von der Lippe M (2021) Interaction of traffic intensity and habitat features shape invasion dynamics of an invasive alien species (*Ambrosia artemisiifolia*) in a regional road network. NeoBiota 64: 55–175. https://doi.org/10.3897/neobiota.64.58775

Lenzner B, Leclère D, Franklin O, Seebens H, Roura-Pascual N, Obersteiner M, Dullinger S, Essl F (2019) A framework for global twenty-first century scenarios and models of biological invasions. Bioscience 69(9): 697–710. https://doi.org/10.1093/biosci/biz070

McManus KM, Morton DC, Masek JG, Wang D, Sexton JO, Nagol JR, Ropars P, Boudreau S (2012) Satellite-based evidence for shrub and graminoid tundra expansion in northern Quebec from 1986 to 2010. Global Change Biology 18(7): 2313–2323. https://doi.org/10.1111/j.1365-2486.2012.02708.x

Méndez M, Escudero A, Iriondo JM, Viejo RM (2014) Demography gone wild in native species: Four reasons to avoid the term "native invaders. Web Ecology 14(1): 85–87. https://doi.org/10.5194/we-14-85-2014

Meyerson LA, Viola DV, Brown RN (2010) Hybridization of invasive *Phragmites australis* with a native subspecies in North America. Biological Invasions 12(1): 103–111. https://doi.org/10.1007/s10530-009-9434-3

Morais M, Marchante E, Marchante H (2017) Big troubles are already here: Risk assessment protocol shows high risk of many alien plants present in Portugal. Journal for Nature Conservation 35: 1–12. https://doi.org/10.1016/j.jnc.2016.11.001

Nentwig W, Bacher S, Pyšek P, Vilà M, Kumschick S (2016) The Generic Impact Scoring System (GISS): A standardized tool to quantify the impacts of alien species. Environmental Monitoring and Assessment 188(5): e315. https://doi.org/10.1007/s10661-016-5321-4

Onur M, Acar C (2017) Investigation of important plants for honey forests in Trabzon region. The Journal of Academic Social Science 5(48): 435–444. https://doi.org/10.16992/ASOS.12440 [In Turkish]

Özarslandan A, Elekcioğlu H (2010) Investigation on virulence of *Meloidogyne incognita* (Kofoid & White, 1919), *M. arenaria* (Neal, 1889) ve *M. javanica* (Treub, 1885) (Tylenchida: Meloidogynidae) populations on resistant and susceptible tomato cultivars. Turkiye Entomoloji Dergisi 34: 495–502. https://dergipark.org.tr/tr/pub/entoted/issue/5689/76083 [In Turkish]

Pauchard A, Meyerson LA, Bacher S, Blackburn TM, Brundu G, Cadotte MW, Courchamp F, Essl F, Genovesi P, Haider S, Holmes ND, Hulme PH, Jeschke JM, Lockwood JL, Novoa A, Nuñez MA, Peltzer D, Pyšek P, Richardson DM, Simberloff D, Smith K, van Wilgen BW, Vilà M, Wilson JRU, Winter M, Zenni RD (2018) Biodiversity assessments: Origin matters. PLoS Biology 16(11): e2006686. https://doi.org/10.1371/journal.pbio.2006686

Pheloung PC, Williams PA, Halloy SR (1999) A weed risk assessment model for use as a 791 biosecurity tool evaluating plant introductions. Journal of Environmental Management 57(4): 239–251. https://doi.org/10.1006/jema.1999.0297

Phillips ML, Murray BR, Pyšek P, Pergl J, Jarošík V, Chytrý M, Kühn I (2010) Plant species of the Central European flora as aliens in Australia. Preslia 82: 465–482.

Pınar SM, Behçet L, Fidan M, Eroğlu H (2018) A new record for the flora of Turkey: *Onopordum cinereum* Grossh (Asteraceae). Erzincan University Journal of Science and Technology 11: 85–91. [In Turkish]

POWO [Plants of the World Online] (2021) Facilitated by the Royal Botanic Gardens, Kew. http://www.plantsoftheworldonline.org/

Pyšek P, Richardson DM, Rejmánek M, Webster G, Williamson M, Kirschner J (2004) Alien plants in checklists and floras: Towards better communication between taxonomists and ecologists. Taxon 53(1): 131–143. https://doi.org/10.2307/4135498

Pyšek P, Lambdon PW, Arianoutsou M, Kühn I, Pino J, Winter M (2009) Alien vascular plants of Europe. In: Handbook of alien species in Europe. Invading Nature – Springer Series in Invasion Ecology, vol 3. Springer, Dordrecht, 43–61. https://doi.org/10.1007/978-1-4020-8280-1_4

Pyšek P, Pergl J, Essl F, Lenzner B, Dawson W, Kreft H, Weigelt P, Winter M, Kartesz J, Nishino M, Antonova LA, Barcelona JF, Cabezas FJ, Cárdenas D, Cárdenas-Toro J, Castańo N, Chacón E, Chatelain C, Dullinger S, Ebel AL, Figueiredo E, Fuentes N, Genovesi P, Groom QJ, Henderson L, Inderjit, Kupriyanov A, Masciadri S, Maurel N, Meerman J, Morozova O, Moser D, Nickrent D, Nowak PM, Pagad S, Patzelt A, Pelser PB, Seebens H, Shu W, Thomas J, Velayos M, Weber E, Wieringa JJ, Baptiste MP, van Kleunen M ((2017) Naturalized alien fora of the world: Species diversity, taxonomic and phylogenetic patterns, geographic distribution and global hotspots of plant invasion. Preslia 89: 203–274. https://doi.org/10.23855/preslia.2017.203

Pyšek P, Hulme PE, Simberloff D, Bacher S, Blackburn TM, Carlton JT, Dawson W, Essl F, Foxcroft LC, Genovesi P, Jeschke JM, Kühn I, Liebhold AM, Mandrak NE, Meyerson LA, Pauchard A, Pergl J, Roy HE, Seebens H, van Kleunen M, Vilà M, Wingfield MJ, Richardson DM (2020) Scientists' warning on invasive alien species. Biological Reviews of the Cambridge Philosophical Society 6(6): 1511–1534. https://doi.org/10.1111/brv.12627

R Core Team (2020) R: A language and environment for statistical computing. https://www.R-project.org/

Rumlerová Z, Vilà M, Pergl J, Nentwig W, Pyšek P (2016) Scoring environmental and socioeconomic impacts of alien plants invasive in Europe. Biological Invasions 18(12): 3697–3711. https://doi.org/10.1007/s10530-016-1259-2

Sarısoy M (2015) The ecosystem geography of the Sultan reed basin. Master Thesis, Istanbul Üniversitesi, İstanbul. [In Turkish with English abstract]

Schnitzer SA, Bongers F (2011) Increasing liana abundance and biomass in tropical forests: Emerging patterns and putative mechanisms. Ecology Letters 14(4): 397–406. https://doi.org/10.1111/j.1461-0248.2011.01590.x

Seebens H, Bacher S, Blackburn TM, Capinha C, Dawson W, Dullinger S, Genovesi P, Hulme PE, van Kleunen M, Kühn I, Jeschke JM, Lenzner B, Liebhold AM, Pattison Z, Pergl J, Pyšek P, Winter M, Essl F (2021) Projecting the continental accumulation of alien species through to 2050. Global Change Biology 27(5): 970–982. https://doi.org/10.1111/gcb.15333

Şekercioğlu ÇH, Anderson S, Akçay E, Bilgin R, Can ÖE, Semiz G, Anderson S, Akçay E, Bilgin R, Can ÖE, Semiz G, Tavşanoğlu Ç, Yokeş MB, Soyumert A, İpekdal K, Sağlam İK, Yücel M, Dalfes HN (2011) Turkey's globally important biodiversity in crisis. Biological Conservation 144(12): 2752–2769. https://doi.org/10.1016/j.biocon.2011.06.025

Sezer A, Kolören O (2019) Determination of weed species, their frequency and general coverage areas in kiwifruit orchards in Eastern Black Sea Region of Turkey. Akademik Ziraat Dergisi 8: 227–236. https://doi.org/10.29278/azd.598855 [In Turkish]

Simberloff D (2011) Native invaders. In: Simberloff D, Rejmánek M (Eds) Encyclopedia of biological invasions. University of California Press, Berkeley and Los Angeles, 472–475. https://doi.org/10.1525/9780520948433-106

Simberloff D (2022) A future planet of weeds? In: Clements, DR, Upadhyaya MK, Joshi S, Shrestha A (Eds) Global plant invasions. Springer, Cham, 361–373. https://doi.org/10.1007/978-3-030-89684-3_17

Simberloff D, Souza L, Nuñez MA, Barrios-Garcia MN, Bunn W (2012) The natives are restless, but not often and mostly when disturbed. Ecology 93(3): 598–607. https://doi.org/10.1890/11-1232.1

Simberloff D, Martin JL, Genovesi P, Maris V, Wardle DA, Aronson J, Courchamp F, Galil B, Berthou EG, Pascal M, Pyšek P, Sousa R, Tabacchi E, Vilà M (2013) Impacts of biological invasions: What's what and the way forward. Trends in Ecology & Evolution 28(1): 58–66. https://doi.org/10.1016/j.tree.2012.07.013

Sohrabi S, Downey PO, Gherekhloo J, Hassanpour-bourkheili S (2020) Testing the Australian Post-Border Weed Risk Management (WRM) system for invasive plants in Iran. Journal for Nature Conservation 53: 125780. https://doi.org/10.1016/j.jnc.2019.125780

Starfinger U, Schrader G (2021) Invasive alien plants in plant health revisited: Another 10 years. Bulletin OEPP. EPPO Bulletin. European and Mediterranean Plant Protection Organisation 51(3): 632–638. https://doi.org/10.1111/epp.12787

Sychrová M, Divíšek J, Chytrý M, Pyšek P (2022) Niche and geographical expansions of North American trees and tall shrubs in Europe. Journal of Biogeography 49(6): 1151–1161. https://doi.org/10.1111/jbi.14377

Tanner R, Fried G (2020) *Phytolacca americana* L. Risk assessment template developed under the "Study on Invasive Alien Species - Development of risk assessments to tackle priority species and enhance prevention". Contract No 07.0202/2018/788519/ETU/ENV.D.21. https://circabc.europa.eu

Terzioglu S, Ansin R (1999) A contribution to exotic plants of Turkey: *Sicyos angulatus* L. Turkish Journal of Agriculture and Forestry 23: 359–362. https://journals.tubitak.gov.tr/agriculture/vol23/iss3/13 [In Turkish]

Terzioğlu S, Ergül Bozkurt A (2020) The weed flora of Turkish tea plantations. Gümüşhane Üniversitesi Fen Bilimleri Enstitüsü Dergisi [Gümüşhane University Journal of Science and Technology] 10(3): 621–630. https://doi.org/10.17714/gumusfenbil.655157

Uludağ A (2015) *Ailanthus altissima*. Önen H (Ed.) Invasive plants catalog of Turkey. Gıda Tarım ve Hayvancılık Bakanlığı, 148–155. [In Turkish]

Uludağ A, Aksoy N, Yazlık A, Arslan ZF, Yazmış E, Üremiş I, Cossu T, Groom Q, Pergl J, Pyšek P, Brundu G (2017) Alien flora of Turkey: Checklist, taxonomic composition and ecological attributes. NeoBiota 35: 61–85. https://doi.org/10.3897/neobiota.35.12460

Üstüner T, Düzenli S, Kitiş YE (2015) Determination of infection rate of mistletoe (*Viscum album*) on hosts in Niğde province. Turkish Journal of Weed Science 18: 6–14. https://dergipark.org.tr/en/download/article-file/618846 [In Turkish]

Valéry L, Fritz H, Lefeuvre JC, Simberloff D (2009) Invasive species can also be native. Trends in Ecology & Evolution 24(11): 585. https://doi.org/10.1016/j.tree.2009.07.003

van Kleunen M, Weber E, Fischer M (2010) A meta-analysis of trait differences between invasive and non-invasive plant species. Ecology Letters 13(2): 235–245. https://doi.org/10.1111/j.1461-0248.2009.01418.x

Vilizzi L, Hill JE, Piria M, Copp GH (2022) A protocol for screening potentially invasive non-native species using Weed Risk Assessment-type decision-support toolkits. Science of the Total Environment 832: 154966. https://doi.org/10.1016/j.scitotenv.2022.154966

Vítková M, Sádlo J, Roleček J, Petřík P, Sitzia T, Müllerová J, Pyšek P (2020) *Robinia pseudoacacia*-dominated vegetation types of Southern Europe: Species composition, history, distribution and management. Science of the Total Environment 707: 134857. https://doi.org/10.1016/j.scitotenv.2019.134857

Wallingford PD, Morelli TL, Allen JM, Beaury EM, Blumenthal DM, Bradley BA, Dukes JS, Early R, Fusco EJ, Goldberg DE, Ibáñez I, Laginhas BB, Vilà M, Sorte CJB (2020) Adjusting the lens of invasion biology to focus on the impacts of climate-driven range shifts. Nature Climate Change 10(5): 398–405. https://doi.org/10.1038/s41558-020-0768-2

Yang Q, Weigelt P, Fristoe TS, Weigelt P, Fristoe TS, Zhang Z, Kreft H, Stein A, Seebens H, Dawson W, Essl F, König C, Lenzner B, Pergl J, Pouteau R, Pyšek P, Winter M, Ebel AL, Fuentes N, Giehl ELH, Kartesz J, Krestov P, Kukk T, Nishino M, Kupriyanov A, Villaseñor JL, Wieringa JJ, Zeddam A, Zykova E, van Kleunen M (2021) The global loss of floristic uniqueness. Nature Communications 12(1): e7290. https://doi.org/10.1038/s41467-021-27603-y

Yazlık A (2022) Invasive alien plants and their impacts. In: Mennan H, Pala F (Eds) Current issues in weed science. IKSAD VII, 263–293. [In Turkish]

Yazlık A, Tepe I (2001) The studies on weeds in apple and pear orchards in Van province and their distributions. Turkish Journal of Weed Science 4: 11–20. [In Turkish]

Yazlık A, Pergl J, Pyšek P (2017) Global assessment of alien plant impacts using the Environmental Impact Classification for Alien Taxa (EICAT). In: Máguas C, Crous C, Costa C (Eds) Ecology and management of alien plant invasions. Syntheses, challenges and new opportunities Book of Abstracts. 4–8 September 2017. Lisboa, Portugal.

Yazlık A, Pergl J, Pyšek P (2018) Impact of alien plants in Turkey assessed by the Generic Impact Scoring System. NeoBiota 39: 31–51. https://doi.org/10.3897/neobiota.39.23598

Yazlık A, Çöpoğlu E, Özçelik A, Tembelo B, Yiğit M, Albayrak B, Baykuş M, Aydınlı V (2019) Weed species and their impacts: Fruit nursery area sample in Düzce. Tekirdag Ziraat Fakültesi Dergisi 16: 389–401. https://doi.org/10.33462/jotaf.578999 [In Turkish]

Yazlık A, Albayrak B (2020) Dodder taxa in Turkey and their impacts. Turkish Journal of Biodiversity 3(2): 95–106. https://doi.org/10.38059/biodiversity.763460 [In Turkish]

Yazlık A, Üremiş İ (2022) Impact of *Sorghum halepense* (L.) Pers. on the species richness in native range. Phytoparasitica. https://doi.org/10.1007/s12600-022-00992-6

Yücel C, Çobanoğlu S (2016) Feral host plants of the European sunflower moth (*Homoeosoma nebulellum* Den. et Schiff.) in Ankara. Tekirdag Ziraat Fakültesi Dergisi 13(4): 124–130. [In Turkish]

Yücel M, Sögüt Z, Türkmen N, Çolakkadıoğlu D, Kahveci B, Çeliktaş V (2019) Determination of the effect of increasing settlement on flora in Çukurova University campus. JENAS Journal of Environmental and Natural Studies 22: 310–322. https://doi.org/10.18016/ksutarimdoga.vi.541325 [In Turkish]

Zenni RD, Essl F, García-Berthou E, McDermott SM (2021) The economic costs of biological invasions around the world. In: Zenni RD, McDermott S, García-Berthou E, Essl F (Eds) The economic costs of biological invasions around the world. NeoBiota 67: 1–9. https://doi.org/10.3897/neobiota.67.69971

Supplementary material 1

Tables S1–S3, Figure S1

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Data type: Tables and figure (docx. file)

Explanation note: **Table S1.** The risk analysis of the native and alien taxa. From left to right columns show question categories, questions and possible scores, scores for each non-native and native species, notes for yes/no. **Table S2.** List of references used for scoring the impact of the study species. **Table S3.** The proportion of variances retained by the dimensions of MCA. **Table S4.** Contribution of each variable to the MCA dimensions. Only the first five dimensions were presented. Variables indicated in Suppl. material 1: Table S1 are shortened in the first column. As the variables are categorical, their values with respect to a specific attribute are indicated with numbers at the end of variable labels. **Figure S1.** MCA ordination with variables. As the variables are categorical, their values with respect to a specific attribute are indicated with numbers at the end of variable labels. To prevent overlapping labels, small lines are used but still some attributes cannot be visualised due to overlaps.

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Threats to UK freshwaters under climate change: Commonly traded aquatic ornamental species and their potential pathogens and parasites

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Abstract

The aquatic ornamental industry, whilst providing socio-economic benefits, is a known introduction pathway for non-native species, which if invasive, can cause direct impacts to native species and ecosystems and also drive disease emergence by extending the geographic range of associated parasites and pathogens and by facilitating host-switching, spillover and spill-back. Although current UK temperatures are typically below those necessary for the survival and establishment of commonly-traded tropical, and some sub-tropical, non-native ornamental species, the higher water temperatures predicted under climate-change scenarios are likely to increase the probability of survival and establishment. Our study aimed primarily to identify which of the commonly-traded non-native ornamental aquatic species (fish and invertebrates), and their pathogens and parasites, are likely to benefit in terms of survival and establishment in UK waters under predicted future climate conditions. Out of 233 ornamental species identified as traded in the UK, 24 were screened, via literature search, for potential parasites and pathogens (PPPs) due to their increased risk of survival and establishment under climate change. We found a total of 155 PPPs, the majority of which were platyhelminths, viruses and bacteria. While many of the identified PPPs were already known to occur in UK waters, PPPs currently absent from UK waters and with zoonotic potential were also identified. Results are discussed in the context of understanding potential impact, in addition to provision of evidence to inform risk assessment and mitigation approaches.

Keywords
Alien species, disease emergence, horizon scanning, invasive non-native species, risk screening

Introduction

The global trade in aquatic ornamental species is extensive, involving over 140 countries (Evers et al. 2019; Hood et al. 2019). Its value is estimated to be in the region of 15–30 billion US dollars annually, with a trade of 1.3 billion ornamental fishes reported (Evers et al. 2019; King 2019). The total value of live ornamental fishes imported into the UK in 2020 was £16.2 million (OATA 2020). The industry includes trade of both freshwater and marine ornamentals, however 76% of the 1244 metric tonnes of live fishes imported into the UK in 2020 were tropical freshwater fishes (OATA 2020). Though generally less well studied, invertebrates (including Mollusca and Crustacea) also represent an important group of traded aquatic ornamentals (Keller and Lodge 2007; Ng et al. 2016).

While the ornamental industry clearly provides economic and social benefits, it is a known pathway for the introduction of non-native species (NNS), pathogens and parasites, which pose a potential threat to aquatic biodiversity if they become invasive (Padilla and Williams 2004; Copp et al. 2005a; Peeler et al. 2011; Hood et al. 2019). Ornamental species are typically kept in closed systems, isolated from open waterways; but deliberate introduction into the wild, often the result of animals overbreeding or getting too large to house, or accidental introduction following escape, is known to occur (e.g. Courtenay 1999; Crossman and Cudmore 1999; Padilla and Williams 2004; Copp et al. 2005b; Duggan et al. 2006; Wood et al. 2022). Introductions of NNS can drive disease emergence by extending the geographic range of associated parasites and pathogens, and by facilitating host-switching or via spillover and spill-back (Peeler et al. 2011). Outbreaks of Koi Herpes Virus (KHV) and Spring Viraemia of Carp (SVC) in UK fisheries, which resulted in substantial mortalities in common carp *Cyprinus carpio carpio* L. 1758 and economic losses, have been linked to the introduction of koi carp *C. carpio koi*, an ornamental variety of common carp (Taylor et al. 2010, 2011, 2013).

In recognition of the threat posed by live non-native fishes, legislation that restricts the keeping of live fishes is in place in the UK. Key legislation includes 'The Prohibition of Keeping or Release of Live Fish (Specified Species) (England) Order 2014' (and its predecessors in 1998 and 2003) implemented under the 'Import of Live Fish (England and Wales) Act 1980', and 'The Keeping and Introduction of Fish (England and River Esk Catchment Area) Regulations 2015'. These legislative instruments apply primarily (if not exclusively) to freshwater fishes, prohibiting their keeping in England without a licence, with similar powers applying in Wales, Scotland and Northern Ireland. The original 1998 Order listed only species considered to be of concern at that time, with the 2003 Order extending the list to include some additional species. These orders were perhaps the most advanced of their kind in Europe and North America (Copp et al. 2005a). The 2014 Order took a much wider approach, with the schedule listing all taxonomic Orders that contained freshwater fish species and stating that all non-native freshwater fishes required a licence to be kept with the exception of those (primarily native species) listed in the 2014 Order's annexes. However, two general licences have been issued permitting the keeping of fishes in garden ponds and/or indoor aquaria. The first, defined here as the 'garden pond fish list' (UK Government 2021a) details standard ornamental pond fishes, namely koi carp, goldfish *Carassius auratus* L. 1758, orfe *Leuciscus idus* L. 1758, grass carp *Ctenopharyngodon idella* Cuvier & Valenciennes, 1844 and sturgeon (*Acipenser* spp.), permitted to be kept in aquaria or secure outdoor garden ponds. The second, defined as the 'ornamental fish list' (UK Government 2019), comprises mainly tropical and subtropical genera (and in some cases species), which are considered to pose a low risk of becoming established or invasive in UK waters, and permitted to be kept in indoor aquaria only. Parallel legislation exists in relation to crayfish in the form of the 'Prohibition of Keeping of Live Fish (Crayfish) Order 1996', which only permits the keeping of red-clawed crayfish *Cherax quadricarinatus* Von Martens, 1868 for ornamental purposes under a general licence and for the use of all non-native fishes in fisheries in the form of 'The Keeping and Introduction of Fish (England and River Esk Catchment Area) Regulations 2015'. Legislation also exists to prevent the introduction of NNS in to the wild (e.g. Wildlife and Countryside Act 1981), and to limit activities associated with specific NNS (e.g. Invasive Alien Species (Enforcement and Permitting) Order 2019). Further, in relation to aquatic animal disease risk specifically, under 'The Aquatic Animal Health (England and Wales) Regulations 2009', live aquatic animal imports require certification.

Currently, UK temperatures are typically below those necessary for the survival and establishment of the commonly-traded tropical, and some sub-tropical, ornamental fishes (species on the ornamental fish list). However, elevated water temperatures (a 2 °C increase) forecasted by future climate change scenarios are predicted to increase the probability of survival and establishment for some existing fish species (Britton et al. 2010). Hence there may be an increased risk of pathogen and parasite introductions, transmissions and disease emergence events (Marcos-López et al. 2010). Climate modelling under four different Representative Concentration Pathway (RCP) scenarios, *sensu* Moss et al. (2010), indicates that global mean surface temperature may increase by between 0.4 and 2.6 °C by the mid-21st century (Moss et al. 2010; Van Vuuren et al. 2011; Nazarenko et al. 2015). An understanding of the trade in ornamental species, and potential disease threats associated with commonly-traded species that may have an increased risk of establishment in the wild under future climate conditions, is essential to mitigate threats and protect aquatic biodiversity, and ensure the sustainability of an important industry.

The aim of our study was to identify commonly-traded non-native (NN) ornamental fishes, crustacea and molluscs at increased risk of survival and establishment in UK waters under elevated temperatures predicted by climate change forecasting. Further, applying the workflow proposed by Foster et al. (2021), organisms known, or with potential to be, pathogenic or parasitic that have been observed associating with these NN ornamental species at any point in the ornamental fish trade pathway were identified. Applications of the outcomes to inform risk assessment and mitigation measures to protect and sustain the ornamental industry in the long term are discussed.

Methods

Records of commonly imported species, such as packing lists that document details of ornamental species imported via the Heathrow Border Control Post (BCP), were not available for use during this study. Therefore, to identify the NN freshwater ornamental species most commonly-imported into UK, a proxy measure was adopted that combined outputs from three different, but complementary methodologies: expert elicitation, eBay retailer search and Google search.

Expert elicitation involved use of the list, provided by the Ornamental Aquatic Trade Association (OATA; https://ornamentalfish.org/), of ornamental species/genera considered by OATA to be the most commonly traded in the UK (by volume). Further, a short list of those NN ornamental species most likely to establish in UK, i.e. species from warm temperate or sub-tropical climatic zones, was provided by the Fish Health Inspectorate (FHI) for England and Wales.

A list of ornamental live-fish retailers was constructed from an eBay search carried out on 8 October 2020 using the term 'live fish'. Search results were filtered for NN fishes that fell under the water type categories of 'fresh', 'pond', 'all water types' and 'not specified'. Species were recorded from all listings between 9 September and 8 October 2020, inclusive. The total number of listings for each NN fish species was used as a proxy measure of trade volume. A separate eBay search using the term 'live invertebrates' was carried out on 13 October 2020. Search results were filtered for NN invertebrates that fell under the water type categories of 'fresh', 'pond', 'all water types' and 'not specified'. Initial results indicated that significantly fewer invertebrate species were listed compared to fish species. Therefore, all NN invertebrate species listings returned by the search, with no restrictions on the date, were recorded. The number of listings per NN invertebrate species was not recorded and species listed multiple times were recorded only once.

A Google search was carried out on 20 October 2020 using the term 'fish species for cold or unheated aquaria", and this provided information on popular ornamental fish species likely to be traded in the UK. Although returning primarily temperate species, it also included tropical fish species with wide temperature tolerances, which therefore do not require heated aquaria, e.g. Endler's livebearer *Poecilia wingei* Poeser, Kempkes & Isbrücker, 2005, and zebra danio *Danio rerio* Hamilton, 1822 (López-Olmeda and Sánchez-Vázquez 2011). In the search for cold or unheated aquaria species, the most popular NN fish species listed in the first 20 websites or blogs (See Suppl. material 1: List S1) were used to represent commonly-traded species. All species mentioned were recorded only once.

A master list of species' common and scientific names was developed. If the species scientific name was absent in the eBay listing or on the website/blog, then it was searched for (using the common name) on FishBase (www.fishbase.se/search.php) for all fish species or via a google search for the invertebrate species. Where fishes and invertebrates were not identified to species level, the entry was removed from the master list. Species recorded via any of the methodologies were collated into the single master list (See Suppl. material 1: Table S1).

The master list was refined by removal of species, based on the following criteria: i) the NN species is present on the 'garden pond fish list' or is not present on the 'ornamental fish' list under The Prohibition of Keeping or Release of Live Fish (Specified Species) (England) Order 2014; and ii) the NN species is recorded as present within UK waters on the Global Biodiversity Information Facility (GBIF; www.gbif.org). Although climate change may increase the risk of some of these NNS, either increasing their current range or establishing new populations as a result of further introductions, the associated pathogen risk was considered to exist already because the species is already present.

Data analysis and modelling

To aid the selection of species for potential pathogen and parasite (PPP) screening, a high-level estimation of climate suitability for each NNS on the master list was undertaken using a species distribution modelling (SDM) approach. Note that the term potential 'pathogen or parasite' is used as a catch-all term, given that evidence for pathogenic or parasitic association was not extensively reviewed in the present study and in fact is often unavailable, in particular for novel environments or hosts. The development of SDMs involved selection of temperature variables under the current climate (2020) and, under future climate, represented by an intermediate climate change scenario, the RCP 4.5 scenario, which predicts stabilisation of radiative forcing (Van Vuuren et al. 2011) and an increase in global annual mean surface air temperature of between 1 and 2 °C (Nazarenko et al. 2015).

The global distribution for each species on the master list was obtained from the GBIF. The climatic zone classification sub-tropical or temperate and the native continent(s) were determined using FishBase. No equivalent database to FishBase exists for invertebrates, so the native range of each invertebrate species was determined via a Google search, and the climatic zone of each range was then climate classified by matching the invertebrate species with fish species from a similar range. Species classified as subtropical or temperate, or with an occurrence record on the GBIF that was outwith the tropical bands (i.e. between the tropics of Capricorn and Cancer), were selected for further analysis (See Suppl. material 1: Table S2). The total number of geo-referenced occurrences for the selected species was recorded. To reduce bias of repeated sampling or multiple reports of a species within the same location, only one record per coordinate was included in the analysis. Species with <100 geo-referenced records were excluded from further analysis (See Suppl. material 1: Table S1). A threshold of 100 geo-referenced records balanced the accuracy of suitability model outputs and number of species for which models could be run. For each species, occurrence data were cleaned using the CoordinateCleaner package in R (Zizka et al. 2019); this package includes a wrapper function that identifies and removes potential errors in the data based on: country and coordinate mismatches, coordinates at sea, zero coordinates, coordinates assigned to country centroids and significant outliers.

Global climate variables at a spatial resolution of ten arc-minutes were downloaded for the present day from the WorldClim dataset (http://worldclim.org) – these are observed data that have been interpolated from current climatic conditions recorded by weather stations (Hijmans et al. 2005). Six temperature variables were then selected for the species distribution models: 1) Annual mean temperature; 2) Mean diurnal range (Mean of monthly: max temp – min temp); 3) Max temperature of warmest month; 4) Min temperature of coldest month; 5) Mean temperature of warmest quarter; and 6) Mean temperature of coldest quarter. Future climate projections were downloaded from the WorldClim dataset. These are derived from five bias-corrected CMIP5 Global Climate Models (GDFL-CM3, HadGEM2-CC, MIROC5, INM-CM4.0, and CSSM4) which specifically related to the 2050 projection (mean for 2041–2060) of the RCP 4.5 climate change scenario.

Species distribution models (SDMs) were employed for NNS on the refined master list to predict the potential suitability of the UK climate for the NNS with respect to the selected temperature variables, both under current (2020) and future climate conditions (in 2050), as represented by climate change scenario RCP 4.5 (Moss et al. 2010). The SDMs were run in R using the SDM package (Naimi and Araújo 2016). Occurrences used in the SDMs were limited to a maximum of 1000 per species. Pseudo-absences were then assigned to each species. The number of pseudo-absences was not fixed across species, rather the number of pseudo-absences was equal to the number of occurrence records for that species as suggested for classification techniques by Barbet-Massin et al. (2012). No pseudo-absence was assigned to a coordinate representing a presence occurrence. Both presences and pseudo-absences input into the SDMs were restricted to the continent within which the native range of the species occurs. For species with distributions that extend across more than one continent, presences and pseudo-absences from all relevant continents were input into the SDMs.

Ensemble models were built for current climate conditions by using two different machine learning methods (boosted-regression trees and random forests). These models estimated the effects of the selected temperature variables, for the present day, on the distribution of each species within the continent of their native range. As no data were available to evaluate the model predictions independently, data were split at random into training (70%) and test data (30%). This random split of the data was repeated five times. To account for the influence of pseudo-absences on model outputs, five random and independent pseudo-absence sets were generated. In total, 50 model replicates were run (two modelling techniques × five pseudo absences × five split samplings) for each species. A geographical representation of the UK was created by cropping a rectangular area using the drawExtent function, which was split into 3828 ten arc-min grid cells (≈ 340 km^2). A suitability score for each species, was predicted for each grid cell using the un-weighted ensemble models, with scores ranging from 0 to 1. A suitability score of 1 indicates that the model predicts the presence of the species in a given location and a score of 0 indicates that the model predicts the absence of the species in that location, based solely on temperature predictors. An overall UK suitability score for each species, for both the present day and under the 2050 scenario, was then calculated by taking the mean of all grid cell suitability scores. Species with a mean suitability score of ≥0.15 in 2050 were selected for parasite/symbiont screening.

Consistent with the framework outlined by Foster et al. (2021), PPP screening consisted of (Table 1): i) organisms associated with each species, i.e. those classified as pathogens, parasites or symbionts, were searched for on PubMed; ii) the associated organisms revealed in the first two pages of a Google search result (using the following terms and format: "*species name*" AND pathogen OR parasite OR commensal OR symbiont OR protist OR bacteria OR virus) were screened for information supplementary to that provided through the PubMed search; and iii) search results were categorised into two groups – PPPs reported to be associated with (or infect) species at any point within the ornamental trade pathway, including wild sourcing (i.e. natural host-PPP interactions), and PPPs known to infect the species through laboratory challenge studies only. Where possible, whether the host-PPP interaction was associated with sub-clinical or asymptomatic infection, clinical signs of disease and/or mortality were noted. Additionally, the country of observation and the point in the ornamental fish trade pathway (e.g. wild, farm, retailer, hobbyist etc.) were recorded.

Table 1. The process undertaken to find information on pathogens, parasites and symbionts associated with each species, with a suitability score of ≥0.15, on PubMed. Only steps one to three were carried out in the present study (adapted from Foster et al. 2021).

Step 1	Search full species name in [All Fields]. If 0, then go to 2, if ≥1, then go to 3
Step 2	Search genus name in [All Fields]. If 0, then decide whether continuing at a higher taxonomic level, is appropriate. If ≥1, go to 4.
Step 3	Conduct search using the criteria: (*Species name* [All Fields]) AND (microbiome[Title/Abstract] OR symbio*[Title/Abstract] OR pathogen*[Title/Abstract] OR parasit*[Title/Abstract] OR protist[Title/Abstract] OR protozoa[Title/Abstract] OR bacteria*[Title/Abstract] OR virus[Title/Abstract] OR host[Title/Abstract] OR reservoir[Title/Abstract] OR vector[Title/Abstract] OR infection [Title/Abstract])
	Scan papers for pathogen/symbiont reports and IDs and record
Step 4	Conduct search using criteira: (G*enus name* [All Fields]) AND (microbiome[Title/Abstract] OR symbio*[Title/Abstract] OR pathogen*[Title/Abstract] OR parasit*[Title/Abstract] OR protist[Title/Abstract] OR protozoa[Title/Abstract] OR bacteria*[Title/Abstract] OR virus[Title/Abstract] OR host[Title/Abstract] OR reservoir[Title/Abstract] OR vector[Title/Abstract] OR infection [Title/Abstract])
	Scan papers for pathogen/symbiont reports and IDs and record
Step 5	Engage with taxon group specialists, as appropriate, to sense check & compile additional information.

Results

Commonly-traded species

The master list of species commonly imported to the UK contained 193 species of ornamental fish, and 40 species of ornamental invertebrate, (total 233, See Suppl. material 1: Table S1). A total of 160 fish species were removed from the master list, and not subject to suitability scoring, because they met at least one of the following criteria: i) present on the list of 'garden pond' fishes, ii) absent from the list of 'ornamental fish' species or recorded as present in the UK on the GBIF, iii) low number (<100) of GBIF records, or iv) distribution was restricted to within the tropical bands (See Suppl. material 1: Table S1).

A total of 32 invertebrate species were removed from the master list because they met at least one of the following criteria: i) species list or recorded as present in the UK on GBIF, ii) low number (<100) of GBIF records, iii) distribution was restricted to within the tropical bands. One further invertebrate species, *Pomacea maculata* Perry, 1810 (synon. *P. insularum*), was removed from the master list because all *Pomacea* spp. were banned from import into the UK (OATA 2021; UK Government 2021b) at the time of our study (See Suppl. material 1: Table S1). Therefore, the refined master list subjected to suitability analysis comprised 33 fish and seven invertebrate species (Table 2).

The majority of fish species (30.3%; *n* = 10) on the refined master list belong to the Order Cypriniformes, which includes the loaches, carps, barbs and minnows; taxa that are common in the aquarium trade. A notable proportion of the fish species on the refined master list (21%; *n* = 7) are the smaller ray-finned fishes of the Order Cyprinodontiformes, such as killifishes and livebearers (e.g. mollies, guppies), which are popular aquarium fishes. Also common on the refined master list are species of the taxonomic orders Siluriformes (18.1%; *n* = 6), representing the catfishes, and Cichliformes (15.1%; *n* = 5), representing the cichlids and angelfishes. Invertebrate species on the refined master list comprise snails, crabs, shrimps and a crayfish. Three were of the taxonomic Order Decapoda (Table 2). Species within orders Notostraca and Cycloneritida are also present on the list.

Species estimates of UK temperature suitability

Mean UK suitability scores for the fish species ranged from 0.08 to 0.59 under current climate conditions and 0.08 to 0.62 under future (2050) climate conditions (Table 2). The highest suitability score in 2050 was seen for the dojo loach *Misgurnus anguillicaudatus* Cantor, 1842, and the Japanese rice fish *Oryzias latipes* Temminck & Schlegel, 1846 (Table 2, also see Suppl. material 1: Fig. S1). Although there was no difference in suitability between current day and 2050 for 15 fish species, suitability increased (mean increase in suitability of 0.05) for 15 fish species. The greatest increase

Table 2. Outputs of species distribution models (SDMs), using UK temperatures under current and future climate conditions (i.e. 2050, under Representative Concentration Pathway, RCP 4.5, scenario), for ornamental freshwater fish and invertebrate species identified via eBay and Google searches in addition to expert elicitation as commonly traded in the UK (ordered by decreasing mean RCP 4.5 suitability score, then by mean current day suitability score and then by native continent. Also given is the number of records (*n*) from the Global Biodiversity Information Facility (GBIF; www.GBIF.org) used to carry out the SDMs (after selection of 1000 random points, removal of duplicates and cleaning). Species in bold had a mean suitability score of ≥ 0.15 under RCP4.5 2050 and were therefore subject to pathogen and parasite screening.

Taxon group/Scientific name	Common name	Native Continent	n	Current	RCP 4.5
FISHES					
Misgurnus anguillicaudatus	dojo loach	Asia	781	0.59	0.62
Oryzias latipes	Japanese rice fish	Asia	247	0.53	0.58
Aphanius mento	pearl-spotted killifish	Asia	135	0.35	0.38
Rhodeus ocellatus	rosy bitterling	Asia	311	0.23	0.32
Pimephales promelas	fathead minnow	North America	859	0.19	0.31
Enneacanthus chaetodon	black banded sunfish	North America	177	0.23	0.29
Misgurnus mizolepis	Chinese muddy loach	Asia	244	0.26	0.28
Garra rufa	red garra	Asia and Europe	234	0.25	0.28
Notropis chrosomus	rainbow shiner	North America	376	0.26	0.27
Amatitlania nigrofasciata	convict cichlid	North America	487	0.21	0.26
Xiphophorus variatus	variable platy	North America	276	0.15	0.20
Pethia conchonius	rosy barb	Asia	128	0.17	0.19
Xiphophorus hellerii	swordtail	North America	943	0.13	0.17
Paracheirodon axelrodi	cardinal tetra	South America	129	0.17	0.17
Corydoras paleatus	pepper corydoras	South America	126	0.15	0.16
Barbodes semifasciolatus	gold barb	Asia	141	0.16	0.15
Astronotus ocellatus	oscar	South America	241	0.16	0.15
Osteoglossum bicirrhosum	arawana	South America	126	0.15	0.15
Phractocephalus hemioliopterus	redtail catfish	South America	120	0.14	0.14
Pethia ticto	ticto barb	Asia	113	0.13	0.13
Hypostomus plecostomus	suckermouth catfish	South America	277	0.13	0.13
Hypseleotris compressa	empire gudgeon	Australasia	855	0.11	0.12
Pygocentrus nattereri	red bellied piranha	South America	532	0.12	0.12
Poecilia reticulata	guppy	North & South America	936	0.11	0.11
Corydoras aeneus	bronze corydoras	South America	278	0.12	0.11
Melanotaenia nigrans	black-banded rainbowfish	Australasia	212	0.10	0.10
Amphilophus citrinellus	midas cichlid	North America	193	0.10	0.10
Cyprinella lutrensis	red shiner	North America	902	0.10	0.10
Rocio octofasciata	Jack Dempsey	North America	595	0.10	0.10
Pterophyllum scalare	angelfish	South America	152	0.10	0.10
Poecilia velifera	sail-fin molly	North America	175	0.09	0.09
Vieja melanura	redhead cichlid	North America	706	0.09	0.09
Poecilia sphenops	common molly	North & South America	519	0.08	0.08
INVERTEBRATES					
Palaemonetes paludosus	ghost shrimp	North America	249	0.31	0.35
Tarebia granifera	quilted melania	Asia & Australasia	160	0.26	0.28
Cherax quadricarinatus	redclaw crayfish (blue lobster)	Australasia	108	0.22	0.22
Triops australiensis	tadpole shrimp	Australasia	145	0.21	0.21
Neritina pulligera	dusky nerite	Africa, Asia & Australasia	111	0.18	0.19
Marisa cornuarietis	Colombian ramshorn apple snail	North & South America	195	0.15	0.17
Metasesarma aubryi	red apple crab	Asia	312	0.13	0.13

in suitability score was seen for the fathead minnow *Pimephales promelas* Rafinesque, 1820, whose score increased by nearly 63% from 0.19 to 0.31. For three fish species, suitability scores reduced between the current day and 2050, though by only a small amount (Table 2): gold barb *Barbodes semifasciolatus* Günther, 1886, bronze corydoras *Corydoras aeneus* Gill, 1858 and the oscar, *Astronotus ocellatus* Agassiz, 1831.

Mean suitability scores for the invertebrate species ranged from 0.15 to 0.33 and from 0.17 to 0.44 under current day and 2050, respectively (Table 2). The highest suitability score in 2050 was seen for the ghost shrimp *Palaemonetes paludosus* Gibbes, 1850. In total, four invertebrate species showed a small increase in suitability score between the present day and 2050. No difference in suitability score between current day and 2050 was seen for three species: red-clawed crayfish, tadpole shrimp *Triops australiensis* Spencer & Hall, 1895, and red apple crab *Metasesarma aubryi* A. Milne-Edwards, 1869. In contrast to fishes, a reduction in suitability in 2050 was not seen for any of the listed invertebrate species.

Potential pathogen and parasite screen

In total, 18 fish and six invertebrate host species were screened for potential pathogens and parasites based on their suitability score of ≥0.15 under RCP 45 2050. A total of 504 records were returned from the literature (PubMed and Google) search. The number of records against each screened host species ranged between 0 and 144, with four species (tadpole shrimp; black banded sunfish *Enneacanthus chaetodon* Baird, 1855; rainbow shiner *Notropis chrosomus* Jordan, 1877; dusky nerite *Neritina pulligera* L., 1767) returning no records. A total of 243 records were deemed unsuitable for the PPP screen following review of the abstract to assess whether or not the publication included both the host species and/or a PPP. In total, 163 records documented natural interactions between hosts and PPPs (Table 3) and 98 records reported host species susceptibility to PPP infection under laboratory conditions (See Suppl. material 1: Table S2).

In total, 155 PPPs across four biological kingdoms (Animalia, Fungi, Prokaryotes and Protists) and two domains (Bacteria and Viruses) were identified as associated with the screened host species. The majority belonged to phyla within the Animalia kingdom (66%; n = 100), specifically Acanthocephala (2%, n = 3), Annelida (1%, n = 2), Arthropoda (6%, n = 10), Cnidaria (1%, n = 2), Nematoda (10%, n = 16), and Platyhelminthes (43%, n = 67) (Table 3).

Viruses represented 12% of the total PPPs identified as associated with screened host species, and they belonged to a number of RNA virus families, including *Rhabdoviridae, Birnaviridae*, as well as the DNA virus family, the iridioviruses (Table 3). Evidence suggests that a large proportion of the viruses identified can cause clinical disease (72%) and/or mortality (56%) in potential hosts screened. Sub-clinical infection by some viruses was also reported to be present in some of the screened potential hosts. Bacterial PPPs represented 11% of PPPs associated with screened hosts and belonged to a number of groups including Aeromonads, *Mycobacterium*, *Vibrio* and *Streptococcus*, and these were largely opportunistic bacteria, which are commonly associated with disease (88%) and mortality (84%) across a wide range of species (Table 3).

The greatest number of PPPs were reported for fathead minnow, with 27 in total (Table 3). Screening also highlighted 25 PPPs associated with dojo loach, 17 PPPs associated with red-claw crayfish and 12 PPPs associated with red garra *Garra rufa* Heckel, 1845.

Many of the PPPs found to be associated with the screened host species are known to already occur in UK waters. In particular, species of bacteria associated with screened hosts have a wide global distribution and are likely already associated with disease in aquatic organisms in the UK. Also known to cause disease in the UK is the protist *Ichthyophthirius multifiliis* Fouquet, 1876, commonly known as 'Ich' – the causative agent of white-spot disease. This protozoan was identified in several screened ornamental fishes including: swordtail *Xiphophorus helleri* Heckel, 1848, oscar, arawana *Osteoglossum bicirrhosum* Cuvier, 1829 and cardinal tetra *Paracheirodon axelrodi* Schultz, 1956. In addition, *Trichodina* Ehrenberg, 1838, another widespread protozoan genus already found in the UK, was identified as associated with several of the screened ornamental species. *Aphanomyces astaci* Schikora, 1906, which is widely distributed throughout Europe and the causative agent of the crayfish plague, was associated with the redclaw crayfish. Arthropoda PPPs, common in the UK, associated with listed species included *Argulus japonicus* Thiele, 1900 and *Argulus foliaceus* L., 1758.

However, PPPs not known to occur in UK waters were identified. For example, infection of fathead minnow by viral haemorrhagic septicaemia virus (VHSv), the aetiological agent of OIE-listed Viral Haemorrhagic Septicaemia, and of the oscar by infectious spleen and necrosis virus (ISKNv) were reported. In addition, the protist *Aphanomyces invadans* David & Kirk, 1997, the aetiological agent of OIE listed Epizootic Elcerative Syndrome, was reported as associated with the rosy barb *Pethia conchonius* Hamilton, 1822. Further, the fungi *Pseudoloma neurophilia* was reported to cause mortality in the fathead minnow. Finally, the Cnidarian, *Myxobolus axelrodi*, was associated with the cardinal tetra and was also reported to cause mortalities.

Also identified were PPPs with zoonotic potential, including two trematodes, *Isthmiophora hortensis* Asata, 1926 and *Clinostomum complanatum* Rudolphi, 1814 were reported as associated with the dojo loach and rosy bitterling *Rhodeus ocellatus* Kner, 1868, respectively. One cestode, *Schyzocotyle acheilognathi* Yamaguti, 1934, also with known zoonotic potential, was reported as associated with swordtail. Bacterial PPPs known to infect both fishes and humans were also identified as associated with screened fishes, including: *Acinetobacter pittii* Nemec et al., 2011, *Aeromonas veronii* Hickman-Brenner et al., 1987, *A. hydrophila* Chester, 1901, *Vibrio cholerae* Pacini, 1854, *Shewanella putrefaciens* MacDonell & Colwell, 1986, *Mycobacterium marinum* Aronson, 1926 and *Mycobacterium goodii* Brown et al., 1999. Antimicrobial resistance was reported for some bacterial strains identified in screened species, including a strain of *Aeromonas sobria* Popoff & Vron, 1981 (in swordtail and dojo loach).

Table 3. List of potential pathogens and parasites reported as natural infections of ornamental fish and invertebrate species traded into the UK , whereby literature evidence was found (Y = Yes) for: Disease = clinical signs or disease in the animal caused by the associated pathogen or parasite; Mort. = mortalities in the animal as a result of the associated pathogen or parasite; Sub. = sub-clinical or asymptomatic infection in the animal. Location types: 'Aquarium' includes reports on specimens held in aquaria by hobbyists, public aquaria, vets or laboratories; 'Retail (Pet shop)' (Retail P) includes ornamental fish shops, both stand-alone and within ornamental markets; 'Border' refers to Import/Border Border Control Inspection Posts; 'Retail B' = Retail bait shop; 'Retail S' = Retail Spa.

Type & name of Disease Agent	Ornamental Species	Disease	Mort.	Sub.	Country	Location type	Reference
Viruses							
Aquatic birnavirus	*Garra rufa*		Y		Ireland	Retail S	(Ruane et al. 2013)
Athtabvirus	*Cherax quadricarinatus*	Y	Y		Australia	Farm	(Sakuna et al. 2018)
Chequa iflavirus	*Cherax quadricarinatus*	Y	Y	Y	Australia	Farm	(Sakuna, et al. 2017)
Cherax quadricarinatus iridovirus	*Cherax quadricarinatus*	Y	Y		China	Farm	(Xu et al. 2016)
Decapod ambidensovirus[1]	*Cherax quadricarinatus*	Y	Y		Australia	Farm	(Bochow et al. 2015)
Fathead minnow calicivirus	*Pimephales promelas*	Y		Y	USA	Retail B	(Mor et al. 2017)
Fathead minnow nidovirus	*Pimephales promelas*	Y		Y	USA	Retail B	(Batts et al. 2012)
Fathead minnow picornavirus	*Pimephales promelas*	Y			USA	Retail B	(Phelps et al. 2014)
Fathead minnow rhabdovirus	*Pimephales promelas*	Y	Y		USA	Farm	(Iwanowicz and Goodwin 2002)
Golden shiner reovirus	*Pimephales promelas*	Y		Y	USA	Retail B	(Boonthai et al. 2018)
Hepatopancreatic reovirus	*Cherax quadricarinatus*	Y	Y	Y	Australia	Farm	(Edgerton et al. 2000)
ISK necrosis virus	*Astronotus ocellatus*				Australia	Retail P	(Go et al. 2016)
ISK necrosis virus	*Astronotus ocellatus*			Y	India	Retail P	(Girisha et al. 2021)
Loach birnavirus	*Misgurnus anguillicaudatus*	Y			Taiwan	Farm	(Chou et al. 1993)
Parvo-like virus	*Cherax quadricarinatus*	Y	Y		Australia	Farm	(Bowater et al. 2002)
South American cichlid iridovirus	*Astronotus ocellatus*	Y	Y		USA	Retail P	(Koda et al. 2018)
Spawner-isolated mortality virus	*Cherax quadricarinatus*		Y	Y	Australia	Farm	(Owens and McElnea 2000)
Turbot reddish body iridovirus	*Astronotus ocellatus*				USA	Retail P	(Go et al. 2016)
Viral Haemorraghic Septicaemia[2]	*Pimephales promelas*			Y	USA	Wild	(Cornwell et al. 2013)
Bacteria							
Acinetobacter pittii	*Misgurnus anguillicaudatus*	Y	Y		China	Farm	(Wang et al. 2019)
Aeromonas hydrophila	*Garra rufa*	Y	Y	Y	Italy	Retail S	(Volpe et al. 2019)
Aeromonas hydrophila	*Misgurnus anguillicaudatus*	Y	Y		South Korea	Farm	(Jun et al. 2010)
Aeromonas sobria	*Corydoras paleatus*			Y	Italy	Wholesaler	(Sicuro et al. 2020)
Aeromonas sobria	*Garra rufa*	Y	Y		Slovakia	Farm	(Majtán et al. 2012)
Aeromonas sobria	*Misgurnus anguillicaudatus*			Y	Italy	Wholesaler	(Sicuro et al. 2020)
Aeromonas sobria	*Misgurnus mizolepis*	Y	Y		South Korea	Farm	(Yu et al. 2015)
Aeromonas sobria	*Xiphophorus hellerii*			Y	Italy	Wholesaler	(Sicuro et al. 2020)
Aeromonas veronii	*Astronotus ocellatus*	Y	Y		India	Farm	(Sreedharan et al. 2011)
Aeromonas veronii	*Garra rufa*	Y	Y	Y	Italy	Retail S	(Volpe et al. 2019)

Type & name of Disease Agent	Ornamental Species	Disease	Mort.	Sub.	Country	Location type	Reference
Bacteria							
Chryseobacterium cucumeris	*Misgurnus anguillicaudatus*	Y	Y		South Korea	Farm	(Kim et al. 2020)
Citrobacter freundii	*Garra rufa*	Y	Y		South Korea	Farm	(Baeck et al. 2009)
Edwardsiella ictaluri	*Pethia conchonius*	Y	Y	Y	Australia	Border	(Humphrey et al. 1986)
Listonella anguillarum	*Misgurnus anguillicaudatus*	Y	Y		China	Farm	(Qin et al. 2014)
Mycobacterium abscessus	*Xiphophorus variatus*	Y	Y		Italy	Border	(Zanoni et al. 2008)
Mycobacterium fortuitum	*Xiphophorus variatus*	Y	Y		Italy	Aquarium	(Zanoni et al. 2008)
Mycobacterium goodii	*Garra rufa*	Y	Y	Y	Italy	Retail S	(Volpe et al. 2019)
Mycobacterium gordonae	*Cherax quadricarinatus*	Y	Y		Israel	Farm	(Davidovich et al. 2019)
Mycobacterium marinum	*Garra rufa*	Y	Y	Y	Italy	Retail S	(Volpe et al. 2019)
Rickettsia-like organism	*Cherax quadricarinatus*	Y	Y		Ecuador	Farm	(Romero et al. 2000)
Shewanella putrefaciens	*Garra rufa*	Y	Y	Y	Italy	Retail S	(Volpe et al. 2019)
Shewanella putrefaciens	*Misgurnus anguillicaudatus*	Y	Y		China	Farm	(Qin et al. 2014)
Streptococcus agalactiae	*Garra rufa*	Y	Y		Ireland	Retail S	(Ruane et al. 2013)
Streptococcus iniae	*Astronotus ocellatus*	Y			Iran	Aquarium	(Tukmechi et al. 2009)
Vibrio cholerae	*Garra rufa*	Y	Y	Y	Italy	Retail S	(Volpe et al. 2019)
Protists							
Achlya sp.	*Astronotus ocellatus*	Y			Iran	Aquarium	(Peyghan et al. 2019)
Aphanomyces astaci	*Cherax quadricarinatus*	Y	Y		Taiwan	Farm	(Hsieh et al. 2016)
Aphanomyces invadans	*Pethia conchonius*	Y	Y		India	Wild	(Pradhan et al. 2014)
Dermocystidium salmonis	*Paracheirodon axelrodi*	Y	Y		Germany	Aquarium	(Langenmayer et al. 2015)
Ichthyobodo necator	*Xiphophorus hellerii*				USA		(Callahan et al. 2005)
Ichthyophthirius multifiliis	*Astronotus ocellatus*				Brazil	Wild	(Neves et al. 2013; Tavares-Dias and Neves 2017)
Ichthyophthirius multifiliis	*Osteoglossum bicirrhosum*				Brazil	Wild	(Rodrigues et al. 2014)
Ichthyophthirius multifiliis	*Paracheirodon axelrodi*				Brazil	Retail P	(Hoshino et al. 2018)
Ichthyophthirius multifiliis	*Xiphophorus hellerii*				Australia	Wild	(Dove and Ernst 1998)
Piscinoodinium pillulare	*Astronotus ocellatus*				Brazil	Wild	(Neves et al. 2013; Tavares-Dias and Neves 2017)
Piscinoodinium pillulare	*Osteoglossum bicirrhosum*				Brazil	Wild	(Rodrigues et al. 2014)
Tokophrya huangmeiensis	*Cherax quadricarinatus*				China	Farm	(Tahir et al. 2017)
Trichodina acuta	*Misgurnus anguillicaudatus*				China	Farm	(Wang et al. 2017)
Trichodina acuta	*Xiphophorus hellerii*				Brazil	Farm	(Piazza et al. 2006; Garcia et al. 2009)
Trichodina heterodentata	*Xiphophorus hellerii*				Australia	Wild	(Dove 2000)
Trichodina lechriodentata	*Misgurnus anguillicaudatus*				China		(Zhao and Tang 2007)
Trichodina modesta	*Misgurnus anguillicaudatus*				China		(Zhao and Tang 2007)
Trichodina sp.	*Paracheirodon axelrodi*				Brazil	Wild	(Tavares-Dias et al. 2010)
Trichodina sp.	*Pimephales promelas*				USA	Wild	(Weichman and Janovy 2000)
Trichodina sp.	*Xiphophorus hellerii*				Sri Lanka	Farm	(Thilakaratne et al.
Fungi							
Apotaspora heleios	*Palaemonetes paludosus*	Y			USA	Wild	(Sokolova and Overstreet 2018)
Exophiala pisciphila	*Paracheirodon axelrodi*	Y			Czechia	Aquarium	(Rehulka et al. 2017)
Glugea pimephales	*Pimephales promelas*	Y			Canada	Wild	(Forest, et al. 2009)
Pleistophora hyphessobryconis	*Paracheirodon axelrodi*				Czechia	Aquarium	(Novotný and Dvořák 2001)
Pleistophora sp.	*Pimephales promelas*			Y	USA	Farm	(Ruehl-Fehlert et al. 2005)

Type & name of Disease Agent	Ornamental Species	Disease	Mort.	Sub.	Country	Location type	Reference
Fungi							
Pseudoloma neurophilia	*Pimephales promelas*	Y	Y		UK	Aquarium	(Sanders et al. 2016)
Animal Kingdom							
Acanthocephala							
Acanthochepalan polymorphus sp.	*Astronotus ocellatus*				Brazil	Wild	(Azevedo et al. 2007)
Neoechinorhynchus panucensis	*Amatitlania nigrofasciata*				Mexico	Wild	(Salgado-Maldonado 2013)
Triaspiron aphanii	*Aphanius mento*				Turkey	Wild	(Smales et al. 2012)
Annelida							
Chaetogaster limnaei	*Tarebia granifera*				Jamaica	Wild	(McKoy et al. 2011)
sp.	*Astronotus ocellatus*				Brazil	Wild	(Neves et al. 2013)
Glossiphonidae gen. sp.	*Astronotus ocellatus*				Brazil	Wild	(Neves et al. 2013)
Arthropoda							
Argulus foliaceus	*Astronotus ocellatus*				Turkey		(Toksen 2006)
Argulus japonicus	*Rhodeus occellatus*				Japan	Wild	(Yamauchi and Shimizu 2013)
Argulus multicolor	*Astronotus ocellatus*				Brazil	Wild	(Tavares-Dias and Neves 2017)
Dolops nana	*Astronotus ocellatus*				Brazil	Wild	(Neves et al. 2013)
Ergasilus ceylonensis	*Xiphophorus hellerii*				Sri Lanka	Farm	(Thilakaratne et al. 2003)
Lamproglena monodi	*Astronotus ocellatus*				Brazil	Wild	(Azevedo et al. 2012)
Lernaea cyprinacea	*Corydoras paleatus*				Argentina	Wild	(Plaul et al. 2010)
Lernaea cyprinacea	*Pimephales promelas*				USA	Wild	(Marcogliese 1991)
Lernaea cyprinacea	*Rhodeus ocellatus*				Japan	Wild	(Nagasawa and Torii 2014)
Lernaea cyprinacea	*Xiphophorus hellerii*				Iran	Farm	(Mirzaei 2015)
Neoergasilus japonicus	*Pimephales promelas*				USA	Wild	(Hudson and Bowen 2002)
Probopyrus pandalicola	*Palaemonetes paludosus*				USA	Wild	(Beck 1980)
Sebekia mississippiensis	*Xiphophorus helleri*				USA	Retail P	(Boyce et al. 1987)
Cnidaria							
Myxobolus axelrodi	*Paracheirodon axelrodi*		Y				(Camus et al. 2017)
Thelohanellus misgurni	*Misgurnus mizolepis*						(Kwon and Kim 2011)
Nematoda							
Anguillicoloides crassus	*Amatitlania nigrofasciata*				Germany	Wild	(Emde et al. 2016)
Camallanus acaudatus	*Osteoglossum bicirrhosum*				Brazil	Wild	(Rodrigues et al. 2014)
Camallanus cotti	*Amatitlania nigrofasciata*				Germany	Wild	(Emde et al. 2016)
Camallanus cotti	*Misgurnus anguillicaudatus*				Canada	Aquarium	(Moravec and Justine 2006)
Camallanus cotti	*Xiphophorus hellerii*				USA	Wild	(Vincent and Font 2003)
Camallanus sp.	*Paracheirodon axelrodi*				Brazil	Retail P	(Hoshino et al. 2018)
Contracaecum bancrofti	*Misgurnus anguillicaudatus*				Australia	Wild	(Shamsi et al. 2019)
Contracaecum sp.	*Astronotus ocellatus*				Brazil	Wild	(Neves et al. 2013; Tavares-Dias and Neves 2017)
Contracaecum sp.	*Osteoglossum bicirrhosum*				Brazil	Wild	(Oliveira et al. 2019)
Contracaecum sp.	*Pimephales promelas*				USA	Wild	(Martins et al. 2017)
Eustrongylides excisus	*Aphanius mento*				Turkey	Wild	(Aydo_du et al. 2011)
Eustrongylides sp.	*Osteoglossum bicirrhosum*				Brazil	Wild	(Oliveira et al. 2019)
Gnathostoma nipponicum	*Misgurnus anguillicaudatus*				South Korea	Retail P	(Sohn et al. 1993)
Mexiconema cichlasomae	*Xiphophorus hellerii*				Mexico	Wild	(Moravec et al. 1998)
Procamallanus inopinatus	*Astronotus ocellatus*				Brazil	Wild	(Tavares-Dias, Sousa and Neves 2014)
Procamallanus pintoi	*Corydoras paleatus*				Argentina	Wild	(Ailán-Choke et al. 2018)
Procamallanus sp.	*Paracheirodon axelrodi*				Brazil	Wild	(Tavares-Dias et al. 2010)

Type & name of Disease Agent	Ornamental Species	Disease	Mort.	Sub.	Country	Location type	Reference
Nematoda							
Procamallanus spiculastriatus	*Astronotus ocellatus*				Brazil	Wild	(Pinheiro et al.2019)
Pseudocapillaria margolisi	*Pethia conchonius*				India	Wild	(De and Maity 1996)
Pseudoproleptus sp.	*Astronotus ocellatus*				Brazil	Wild	(Pinheiro et al. 2019)
Platyhelminthes							
Acanthatrium hitaense	*Tarebia granifera*				Thailand	Wild	(Dechruska and Krailas 2007)
Acanthostomum sp.	*Paracheirodon axelrodi*				Brazil	Retail P	(Hoshino et al. 2018)
Caballerotrema aruanense	*Osteoglossum bicirrhosum*				Brazil	Wild	(Oliveira et al. 2019)
Centrocestus formosanus	*Barbodes semifasciolatus*				Vietnam	Wild	(Chai et al.2012)
Centrocestus formosanus	*Melanoides tuberculata*				USA	Wild	(Tolley-Jordan and Chadwick 2018)
Centrocestus formosanus	*Osteoglossum bicirrhosum*				Iran	Wild	(Mood et al. 2010)
Centrocestus formosanus	*Tarebia granifera*				USA	Wild	(Tolley-Jordan and Chadwick 2018)
Centrocestus formosanus	*Tarebia granifera*				Thailand	Wild	(Dechruska and Krailas 2007)
Clinostomum complanatum	*Misgurnus anguillicaudatus*				Taiwan	Farm	(Wang et al. 2017)
Clinostomum complanatum	*Rhodeus ocellatus*				Japan	Wild	(Aohagi et al. 1992)
Clinostomum marginatum	*Astronotus ocellatus*				Brazil	Wild	(Tavares-Dias and Neves 2017)
Clonorchis sinensis	*Misgurnus anguillicaudatus*				China	Wild	(Zhang et al. 2014)
Clonorchis sinensis	*Misgurnus anguillicaudatus*				South Korea	Wild	(Shin 1964)
Clonorchis sinensis	*Rhodeus ocellatus*				South Korea	Wild	(Rhee et al. 1983)
Craspedella pedum	*Cherax quadricarinatus*				South Africa	Wild	(Tavakol et al. 2016)
Crassiphiala bulboglossa	*Pimephales promelas*				USA	Wild	(Wisenden et al. 2012)
Dactylogyrus olfactorius	*Pimephales promelas*				USA	Wild	(Lari et al. 2016)
Dactylogyrus simplex	*Pimephales promelas*				USA	Wild	(Knipes and Janovy 2009)
Dactylogyrus bychowskyi	*Pimephales promelas*				USA	Wild	(Knipes and Janovy 2009)
Dactylogyrus pectenatus	*Pimephales promelas*				USA	Wild	(Knipes and Janovy 2009)
Dactylogyrus ostraviensis	*Pethia conchonius*			Y	Australia	Border	(Trujillo-Gonz‡lez et al. 2019)
Dactylogyrus sp.	*Garra rufa*				Iraq	Wild	(Abdullah 2017)
Dactylogyrus sp.	*Xiphophorus hellerii*				Sri Lanka	Farm	(Thilakaratne et al. 2003)
Diceratocephala boschmai	*Cherax quadricarinatus*				Thailand	Wild	(Ngamniyom et al. 2019)
Diceratocephala boschmai	*Cherax quadricarinatus*				South Africa	Wild	(Tavakol et al. 2016)
Didymorchis sp.	*Cherax quadricarinatus*				South Africa	Wild	(Tavakol et al. 2016)
Diplostomidae sp.	*Paracheirodon axelrodi*				Brazil	Retail P	(Hoshino et al. 2018)
Echinostoma cinetorchis	*Misgurnus anguillicaudatus*				South Korea	Retail P	(Seo et al. 1984)
Echinostoma sp.	*Melanoides tuberculata*				Philippines	Wild	(Paller et al. 2019)
Gonocleithrum aruanae	*Osteoglossum bicirrhosum*				Brazil	Wild	(Tavares-Dias et al. 2014)
Gonocleithrum coenoideum	*Osteoglossum bicirrhosum*				Brazil	Wild	(Rodrigues et al. 2014)
Gonocleithrum cursitans	*Osteoglossum bicirrhosum*				Iran	Retail P	(Mehdizadeh et al. 2016)
Gonocleithrum planacrus	*Osteoglossum bicirrhosum*				Brazil	Wild	(Rodrigues et al. 2014)
Gussevia asota	*Astronotus ocellatus*				Peru	Wild	(Mendoza-Franco et al. 2010)
Gussevia asota	*Astronotus ocellatus*				Panama	Wild	(Mendoza-Franco et al. 2007)
Gussevia asota	*Astronotus ocellatus*				South Korea	Farm	(Kim et al. 2002)
Gussevia astronii	*Astronotus ocellatus*				Brazil	Wild	(Neves et al. 2013; Tavares-Dias and Neves 2017)

Type & name of Disease Agent	Ornamental Species	Disease	Mort.	Sub.	Country	Location type	Reference
Nematoda							
Gussevia rogersi	*Astronotus ocellatus*				Brazil	Wild	(Neves et al. 2013)
Gyrodactylus anisopharynx	*Corydoras paleatus*				Argentina	Wild	(Rauque et al. 2018)
Gyrodactylus anisopharynx	*Corydoras paleatus*				Brazil	Wild	(Boeger et al. 2005)[3]
Gytrodactylus bullatarudis	*Xiphophorus hellerii*				Australia	Wild	(Dove and Ernst 1998)
Gyrodactylus cichlidarum	*Astronotus ocellatus*				Iran	Retail P	(Mousavi et al. 2013)
Gyrodactylus corydori	*Corydoras paleatus*				Brazil	Wild	(Bueno-silva et al. and Boeger 2009)
Gyrodactylus macracanthus	*Misgurnus anguillicaudatus*				Australia	Wild	(Dove and Ernst 1998)
Gyrodactylus medaka	*Oryzias latipes*				Japan	Wild	(Nitta and Nagasawa 2018)
Gyrodactylus samirae	*Corydoras paleatus*				Brazil	Wild	(Popazoglo and Boeger 2016)
Gyrodactylus sp.	*Misgurnus anguillicaudatus*				USA	Wild	(Reyda et al. 2020)
Gyrodactylus sp.	*Paracheirodon axelrodi*				Brazil	Wild	(Tavares-Dias et al. 2010)
Gyrodactylus sp.	*Xiphophorus hellerii*				Sri Lanka	Farm	(Thilakaratne et al. 2003)
Gyrodactylus superbus	*Corydoras paleatus*				Argentina	Wild	(Rauque et al. 2018)
Haematoloechus similis	*Tarebia granifera*				Thailand	Wild	(Dechruska and Krailas 2007)
Haplorchis pumilio	*Barbodes semifasciolatus*				Vietnam	Wild	(Chai et al. 2012)
Haplorchis pumilio	*Melanoides tuberculata*				USA	Wild	(Tolley-Jordan and Chadwick 2018)
Haplorchis pumilio	*Melanoides tuberculata*				Thailand	Wild	(Dechruska and Krailas 2007)
Haplorchis pumilio	*Tarebia granifera*				USA	Wild	(Tolley-Jordan and Chadwick 2018)
Haplorchis sp.	*Tarebia granifera*				Laos	Wild	(Ditrich et al. 1990)
Haplorchis taichui	*Melanoides tuberculata*				Thailand	Wild	(Chontananarth and Wongsawad 2010)
Haplorchis taichui	*Melanoides tuberculata*				Laos	Wild	(Nawa et al. 2001)
Haplorchis taichui	*Tarebia granifera*				Thailand	Wild	(Chontananarth and Wongsawad 2010)
Haplorchis taichui	*Tarebia granifera*				Laos	Wild	(Nawa et al. 2001)
Herpetodiplostomum sp.	*Astronotus ocellatus*				Brazil	Wild	(Neves et al. 2013)
Isthmiophora hortensis[4]	*Misgurnus anguillicaudatus*				China	Wild	(Qiu et al. 2017)
Isthmiophora hortensis[4]	*Misgurnus anguillicaudatus*				South Korea	Wild	(Ryang 1990)
Isthmiophora hortensis[4]	*Misgurnus anguillicaudatus*				South Korea	Retail P	(Jong-Yil Chai et al. 1985)
Loxogenoides bicolor	*Melanoides tuberculata*				Thailand	Wild	(Ukong et al. 2007)
Loxogenoides bicolor	*Tarebia granifera*				Thailand	Wild	(Ukong et al. 2007)
Massaliatrema misgurni	*Misgurnus anguillicaudatus*				Japan	Retail P	(Ohyama et al. 2001)
Megulurous sp.	*Melanoides tuberculata*				Philippines	Wild	(Paller et al. 2019)
Metorchis orientalis	*Rhodeus ocellatus*				China	Wild	(Qiu et al. 2017)
Notocotylid sp.	*Tarebia granifera*				Jamaica	Wild	(McKoy et al. 2011)
Ornithodiplostomum ptychocheilus	*Pimephales promelas*				USA	Wild	(Wisenden et al. 2012)
Ornithodiplostomum ptychocheilus	*Pimephales promelas*				Canada	Wild	(Sandland and Goater 2001; Sandland et al. 2001)[5]
Paracaryophyllaeus gotoi	*Misgurnus anguillicaudatus*				Japan	Wild	(Scholz et al. 2001)
Parapleurophocercous sp.	*Melanoides tuberculata*				Philippines	Wild	(Paller et al. 2019)
Parapleurophocercous sp.	*Tarebia granifera*				Philippines	Wild	(Paller et al. 2019)
Philophthalmus gralli	*Melanoides tuberculata*				USA	Wild	(Tolley-Jordan and Chadwick 2018)

Type & name of Disease Agent	Ornamental Species	Disease	Mort.	Sub.	Country	Location type	Reference
Nematoda							
Philophthalmus gralli	*Tarebia granifera*				USA	Wild	(Tolley-Jordan and Chadwick 2018)
Philophthalmus sp.	*Tarebia granifera*				Jamaica	Wild	(McKoy et al. 2011)
Posthodiplostomum minimum	*Pimephales promelas*				Canada	Wild	(Schleppe and Goater 2004)
Posthodiplostomum minimum	*Pimephales promelas*				USA	Farm	(Mitchell et al. 1982)
Posthodiplostomum sp.	*Astronotus ocellatus*				Brazil	Wild	(Neves et al. 2013)[6]
Pimephales promelas	*Posthodiplostomum* sp.				USA	Wild	(Weichman and Janovy 2000)
Proteocephalus gibsoni	*Astronotus ocellatus*				Brazil	Wild	(Tavares-Dias and Neves 2017)
Proteocephalus misgurni	*Misgurnus anguillicaudatus*				Russia	Wild	(Scholz et al. 2014)
Prototransversotrema steeri	*Xiphophorus hellerii*				Sri Lanka	Farm	(Dove 2000)
Pseudolevinseniella anenteron	*Cherax quadricarinatus*				Thailand	Wild	(Ngamniyom et al. 2019)
Paradiplozoon bingolensis	*Garra rufa*				Turkey	Wild	(Civanova et al. 2013)
Schyzocotyle acheilognathi	*Pimephales promelas*				USA	Retail B	(Boonthai et al. 2017)
Schyzocotyle acheilognathi	*Xiphophorus hellerii*				USA	Wild	(Vincent and Font 2003)
Stellantchasmus falcatus	*Tarebia granifera*				Thailand	Wild	(Chontananarth et al. 2018)
Temnosewellia sp.	*Cherax quadricarinatus*				Thailand	Wild	(Ngamniyom et al. 2019)
Tetracotyle wayanadensis	*Pethia conchonius*				India	Wild	(Jithila and Prasadan 2018)
Thometrema sp.	*Astronotus ocellatus*				Brazil	Wild	(Tavares-Dias and Neves 2017)
Uvulifer ambloplitis	*Pimephales promelas*				USA	Wild	(Weichman and Janovy 2000)
Telethecium nasalis	*Osteoglossum bicirrhosum*				Brazil	Wild	(Kritsky et al. 1996)

[1] variant *Cherax quadricarinatus densovirus*; [2] Rhabdovirus; [3] Also Pie and Boeger (2006) and Bueno-silva and Boeger (2009); [4] Also Reported under synonym *Echinostoma hortense* ; [5] Aso Schleppe and Goater (2004); [6] Also Tavares-Dias and Neves (2017) and Pinheiro et al. (2019).

Discussion

Trade in live aquatic ornamental species is vast, benefitting from globalisation and improved transport in recent decades. Over 140 countries are involved in the international trade of more than 1500 fish and 300 aquatic invertebrate species (Weir et al. 2012; Hood et al. 2019). Our study provides an assessment of freshwater ornamental trade in the UK in which commonly traded species and their likelihood of establishment in UK waters under current and future climate conditions, with their potential pathogens and parasites also identified..

These data on commonly traded species are a snapshot in time, which potentially limits accuracy and prevents the assessment of seasonal and annual variations. That said, the six species identified were listed amongst the 30 species reported to predominate the global trade in ornamental freshwater organisms in a relatively recent review: Endler's livebearer, goldfish, zebra danio, neon tetra *Paracheirodon innesi* Myers, 1936, angel fish *Pterophyllum scalare* Schultze, 1823, and discus *Symphysodon aequifasciatus* Pellegrin, 1904 (Dey 2016).

Access to robust ornamental trade data, in particular with respect to species traded and import origin, is fundamental to fill knowledge gaps and inform risk screenings and the risk analysis process (Copp et al. 2016; Chan et al. 2019). A comprehensive understanding of spatial and temporal trade patterns to species level will increase capacity to identify high risk links, facilitating targeted and cost-effective surveillance. Species-level information will also support analyses from a conservation point of view, particularly for marine species that are wild sourced, and ensure there is increased transparency in the source, quantity and sustainability of trade for each species (Andersson et al. 2021). In the UK, aquatic species imports from third party countries are electronically recorded on the 'Import of Animals Food and Feed System' (IPAFFS). Historically, non-susceptible aquatic species imported from the EU for ornamental use were not recorded on any system. As of January 2021, all aquatic imports must be recorded on IPAFFS, but records on this database are categorised by international commodity codes (www.uktradeinfo.com/find-commodity-data/help-with-classifying-goods/). However, the purpose of these codes is to enable the application of appropriate tariffs to imported goods and they do not provide sufficient resolution to identify consignments to species level, so cannot be used to inform disease susceptibility or invasive potential. Import data at the species-level for the UK currently exist in paper format only (on the invoices that accompany all other relevant import certification), though there are periods for which species-level electronic data have been available for freshwater fishes, including ornamental varieties, imported to England between 2000 and 2004 inclusive (see Copp et al. 2007). Limitations with respect to access to detailed live ornamental import data are not unique to the UK (Rhyne et al. 2012, 2017; Leal et al. 2016; Pinnegar and Murray 2019; Biondo and Burki 2020). Creation of an import data App, or extension of an existing, established App, is an opportunity to capture detailed import data, integrating species trade information with other crucial information such as invasiveness potential, associated disease threats and conservation status. Such a system would enhance capacity for real-time monitoring and analysis, at the point of exporter application for import, allowing trade in high risk species to be tracked and incidences of illicit trade, such as the import of prohibited species, to be detected in a timely manner (Rhyne et al. 2017).

Legislative instruments restrict the keeping of many temperate species but do allow the keeping of numerous commonly traded tropical and sub-tropical species. The application of SDMs indicated that, while establishment of commonly traded species if released into the wild is unlikely in the UK under current conditions, predicted temperature increases associated with climate change may increase risks of survival and establishment. The mean increase in temperature suitability of 2.4% and 1.8% for fish and invertebrates, respectively, by 2050 under RCP 4.5 demonstrated in our study may seem a small increase in 'risk', but RCP 4.5 represents a moderate climate-change scenario, and temperature increases may be greater than this scenario predicts. Although a broad scale indication of the change in suitability under climate change is provided, careful interpretation of SDM outputs may be required. For instance, the red shiner *Cyprinella lutrensis* Baird & Girard, 1853, is widespread across the USA and has been identified, using the Fish Invasiveness Screening Kit, as posing a medium risk of being invasive in England and Wales (Copp et al. 2009). However, the red shiner had a relatively low suitability score (0.10) under current day conditions in our study (Table 2). The Chinese muddy loach *Misgurnus mizolepis* Guenther, 1888 had a suitability score of 0.26 under current day conditions, yet there has been only one record of a reproducing population in the UK (in southern England), which was subsequently eradicated (Zięba et al. 2010).

Although outside the scope of our study, there will be value in assessing species suitability at a finer spatial resolution, for example accounting for differences in conditions across the entirety of the UK (Thrush and Peeler 2013), for example, the notably warmer conditions in the South compared to the North UK, affecting the risks of survival and establishment of sub-tropical and tropical species (see Suppl. material 1: Figs S1, S2). Further, extension of the SDMs to incorporate environmental factors such as rainfall, habitat type or elevation (Logez et al. 2012), species' life-history traits such as size or fecundity (Copp and Fox 2007; Liu and Olden 2017), and consideration of the likelihood of release of each species (i.e. potential propagule pressure) will be of value. For example, inclusion of elevation will account for species with native distributions at higher altitudes within the tropical zone that may be more adapted to temperature conditions more similar to the temperate zone species due to Rapoport's latitudinal rule to altitude (Stevens 1992).

In total, 155 PPPs were found to be associated with the screened ornamental fishes and invertebrates. Despite following a standardised approach for each host species, the number of PPPs identified in the literature may be skewed by the research effort applied to a species and affected by the use of different accepted names or synonyms. One of the key drivers of impacts associated with NN aquatic species (Peeler et al. 2011) are PPPs, with disease emergence events resulting from NNS introductions being well documented (Taraschewski 2006; Peeler et al. 2011; Lymbery et al. 2014). Disease emergence can be driven solely by a switch in geographical range, because a new environment may favour increased PPP virulence, or by host switching and pathogen/parasite spillover (Peeler et al. 2011; Foster et al. 2021). Thermal tolerances of a PPP may also determine the likelihood and impact of disease emergence resulting from co-transportation, particularly for PPPs that can survive outside a host or have free-living stages (Barber et al. 2016).

Even if the long-term survival of an ornamental species is not supported by future UK temperatures, the host species may persist long enough to transmit a PPP to a native susceptible host or introduce a free-living stage which can survive. Temperature may also determine the likelihood of PPPs causing disease and morbidity in infected hosts. For example, KHV is thought to only cause clinical signs and mortality between 16 °C and 25 °C (OIE 2019), whereas outbreaks of VHS rarely occur above 15 °C (Baillon et al. 2020). While beyond the scope of this study there will be value in building on the present study by examining the environmental tolerances of the PPPs and the likely impact of climate change. Indeed, PPPs introduced via the ornamental pathway may cause wider impact, for example causing mortality and yield loss in aquaculture systems and affecting human health, although strong biosecurity and health and safety precautions can mitigate against such risks.

The PPPs that cause mortalities or clinical expression of disease in the traded host species are more likely to be detected via visual inspection or quarantine at border control posts or other stages in the ornamental trade pathway (Table 3). However, PPPs that live symbiotically with a host, or those PPPs that have sub-clinical or latent infection stages, provide a greater challenge to detection (Gomez et al. 2006; Becker et al. 2014). Traditionally, the testing of host species for PPP presence often requires destructive sampling, which limits the number of specimens and ultimately reduces the probability of PPP detection. However, new methodologies that incorporate molecular techniques at border control posts, such as environmental DNA, may present a good non-lethal option (Trujillo-González et al. 2019b; Brunner 2020) for improved PPP detection. Measures such as heat ramping (gradually increasing the water temperature over a minimum period) can be used during quarantine to detect latent infections (Eide et al. 2011); however, this may not appropriate for all PPPs, when surveillance aims to target multiple PPPs and where there is a need to adhere to animal welfare laws and guidance.

Though the remit of our study was to undertake a high-level screening to identify all PPPs associated with commonly traded ornamental species, rather than novel threats *per se*, we note that while some of the identified PPPs are known in the UK, others are not. Some PPPs are already known within the ornamental fish trade industry, and do not cause widespread impact or can be successfully treated to minimise impact. However, the abundance and diversity of PPPs increases potential for future disease outbreaks under changing environmental conditions. Even where a PPP has not yet been implicated in any mortality events, the changing climate and alterations to host communities (e.g. due to species introductions) may provide the perfect storm for disease emergence into the future. Next steps should aim to assess the risk associated with each PPP, focussing on the interplay between the PPP, all potential hosts and changing environmental conditions.

In conclusion, the ornamental fish trade is largely free from serious and untreatable diseases. However, through screening of a small subset of ornamental freshwater species, our study highlights the abundance and diversity of PPPs present in ornamental species commonly traded in the UK. An understanding of hazards associated with PPPs, in particular under changing ecological and environmental conditions, is crucial to determine and communicate risks and enhance risk awareness amongst stakeholders and the general public, thereby enabling mitigation through management actions (Britton et al. 2011) to ensure a safe and sustainable ornamental aquatics industry into the future (Copp et al. 2016).

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References

Abdullah SMA (2017) First record of *Dactylogyrus rectotrabus* (Monogenetic Trematoda) from *Garra rufa* from Greater Zab River, North of Iraq, regarding its ecological aspects. Egypt Journal of Aquatic Biology and Fish 11: 1029–1040.

Ailán-Choke LG, Ramallo G, Davies D (2018) Further study on *Procamallanus* (*Spirocamallanus*) *pintoi* (Kohn et Fernandes, 1988) (Nematoda: Camallanidae) in *Corydoras paleatus* and *Corydoras micracanthus* (Siluriformes: Callichthyidae) from Salta, Argentina, with a key to congeneric species. Acta Parasitologica 63(3): 595–604. https://doi.org/10.1515/ap-2018-0068

Andersson AA, Tilley HB, Lau W, Dudgeon D, Bonebrake TC, Dingle C (2021) CITES and beyond: Illuminating 20 years of global, legal wildlife trade. Global Ecology and Conservation 26: e01455. https://doi.org/10.1016/j.gecco.2021.e01455

Aohagi Y, Shibahara T, Machida N, Yamaga Y, Kagota K (1992) *Clinostomum complanatum* (Trematoda: Clinostomatidae) in five new fish hosts in Japan. Journal of Wildlife Diseases 28(3): 467–469. https://doi.org/10.7589/0090-3558-28.3.467

Baeck GW, Kim JH, Choresca C, Gomez DK, Shin SP, Han JE, Park SC (2009) Mass mortality of doctor fish (*Garra rufa obtusa*) caused by *Citrobacter freundii* infection. Journal of Veterinary Clinics 26: 150–154. https://www.koreascience.or.kr/article/JAKO200916762902489.pdf

Baillon L, Mérour E, Cabon J, Louboutin L, Vigouroux E, Alencar ALF, Cuenca A, Blanchard Y, Olesen NJ, Panzarin V, Morin T, Brémont M, Biacchesi S (2020) The Viral Hemorrhagic Septicemia Virus (VHSV) markers of virulence in rainbow trout (*Oncorhynchus mykiss*). Frontiers in Microbiology 11: 1–17. https://doi.org/10.3389/fmicb.2020.574231

Barber I, Berkhout BW, Ismail Z (2016) Thermal change and the dynamics of multi-host parasite life cycles in aquatic ecosystems. Integrative and Comparative Biology 56(4): 561–572. https://doi.org/10.1093/icb/icw025

Barbet-Massin M, Jiguet F, Albert CH, Thuiller W (2012) Selecting pseudo-absences for species distribution models: How, where and how many? Methods in Ecology and Evolution 3(2): 327–338. https://doi.org/10.1111/j.2041-210X.2011.00172.x

Batts WN, Goodwin AE, Winton JR (2012) Genetic analysis of a novel nidovirus from fathead minnows. Journal of General Virology 93(6): 1247–1252. https://doi.org/10.1099/vir.0.041210-0

Beck JT (1980) Life history relationships between the bopyrid isopod *Probopyrus pandalicola* and of its freshwater shrimp hosts *Palaemonetes paludosus.* American Midland Naturalist 104(1): 135–154. https://doi.org/10.2307/2424966

Becker JA, Tweedie A, Rimmer A, Landos M, Lintermans M, Whittington RJ (2014) Incursions of Cyprinid herpesvirus 2 in goldfish populations in Australia despite quarantine practices. Aquaculture (Amsterdam, Netherlands) 432: 53–59. https://doi.org/10.1016/j.aquaculture.2014.04.020

Biondo MV, Burki RP (2020) A systematic review of the ornamental fish trade with emphasis on coral reef fishes-An impossible task. Animals (Basel) 10(11): 1–21. https://doi.org/10.3390/ani10112014

Bochow S, Condon K, Elliman J, Owens L (2015) First complete genome of an Ambidensovirus; Cherax quadricarinatus densovirus, from freshwater crayfish *Cherax quadricarinatus.* Marine Genomics 24: 305–312. https://doi.org/10.1016/j.margen.2015.07.009

Boeger WA, Kritsky D, Pie MR, Engers KB (2005) Mode of transmission, host switching, and escape from the red queen by viviparous gyrodactylids (Monogenoidea). The Journal of Parasitology 9(5): 1000–1007. https://doi.org/10.1645/GE-515R.1

Boonthai T, Herbst SJ, Whelan GE, Van Deuren MG, Loch TP, Faisal M (2017) The Asian fish tapeworm *Schyzocotyle acheilognathi* is widespread in baitfish retail stores in Michigan, USA. Parasites & Vectors 10(1): 1–11. https://doi.org/10.1186/s13071-017-2541-6

Boonthai T, Loch TP, Zhang Q, Van Deuren MG, Faisal M, Whelan GE, Herbst SJ (2018) Retail baitfish in Michigan Harbor serious fish viral pathogens. Journal of Aquatic Animal Health 30(4): 253–263. https://doi.org/10.1002/aah.10034

Bowater RO, Wingfield M, Fisk A, Condon KML, Reid A, Prior H, Kulpa EC (2002) A parvo-like virus in cultured redclaw crayfish *Cherax quadricarinatus* from Queensland, Australia. Diseases of Aquatic Organisms 50: 79–86. https://doi.org/10.3354/dao050079

Boyce WM, Kazacos EA, Kazacos KR, Engelhardt JA (1987) Pathology of pentastomid infections (*Sebekia mississippiensis*) in fish. Journal of Wildlife Diseases 23(4): 689–692. https://doi.org/10.7589/0090-3558-23.4.689

Britton JR, Cucherousset J, Davies GD, Godard MJ, Copp GH (2010) Non-native fishes and climate change: Predicting species responses to warming temperatures in a temperate region. Freshwater Biology 55(5): 1130–1141. https://doi.org/10.1111/j.1365-2427.2010.02396.x

Britton JR, Copp GH, Brazier M, Davies GD (2011) A modular assessment tool for managing introduced fishes according to risks of species and their populations, and impacts of management actions. Biological Invasions 13(12): 2847–2860. https://doi.org/10.1007/s10530-011-9967-0

Brunner JL (2020) Pooled samples and eDNA-based detection can facilitate the "clean trade" of aquatic animals. Scientific Reports 10(1): 1–11. https://doi.org/10.1038/s41598-020-66280-7

Bueno-Silva M, Boeger WA (2009) Neotropical Monogenoidea. 53. *Gyrodactylus corydori* sp. n. and redescription of *Gyrodactylus anisopharynx* (Gyrodactylidea: Gyrodactylidae), parasites of *Corydoras* spp. (Siluriformes: Callichthyidae) from southern Brazil. Folia Parasitologica 56(1): 13–20. https://doi.org/10.14411/fp.2009.003

Bueno-silva M, Boeger WA, Pie MR (2011) Choice matters: Incipient speciation in *Gyrodactylus corydori* (Monogenoidea: Gyrodactylidae). International Journal for Parasitology 41(6): 657–667. https://doi.org/10.1016/j.ijpara.2011.01.002

Callahan HA, Litaker RW, Noga EJ (2005) Genetic relationships among members of the Ichthyobodo necator complex: Implications for the management of aquaculture stocks. Journal of Fish Diseases 28(2): 111–118. https://doi.org/10.1111/j.1365-2761.2004.00603.x

Camus AC, Dill JA, Rosser TG, Pote LM, Griffin MJ (2017) *Myxobolus axelrodi* n. sp. (Myxosporea: Myxobolidae) a parasite infecting the brain and retinas of the cardinal tetra *Paracheirodon axelrodi* (Teleostei: Characidae). Parasitology Research 116(1): 387–397. https://doi.org/10.1007/s00436-016-5301-1

Chai JY, Hong SJ, Sohn WM (1985) Studies on intestinal trematodes in Korea. XVL. Infection status of loaches with the metacercariae of *Echinostoma hortense.* Korean Journal of Parasitology 23(1): 18–23. https://doi.org/10.3347/kjp.1985.23.1.18

Chai JY, Van De N, Sohn WM (2012) Foodborne trematode metacercariae in fish from northern Vietnam and their adults recovered from experimental hamsters. Korean Journal of Parasitology 50(4): 317–325. https://doi.org/10.3347/kjp.2012.50.4.317

Chan FT, Beatty SJ, Gilles Jr AS, Hill JE, Kozic S, Du L, Morgan DL, Pavia RTB, Therriault TW, Verreycken H, Vilizzi L, Wei H, Yeo DCJ, Zeng Y, Zięba G, Copp GH (2019) Leaving the fishbowl: The ornamental trade as a global vector for freshwater fish invasions. Aquatic Ecosystem Health & Management 22(4): 417–439. https://doi.org/10.1080/14634988.2019.1685849

Chontananarth T, Wongsawad C (2010) Prevalence of *Haplorchis taichui* in field-collected snails: A molecular approach. Korean Journal of Parasitology 48(4): 343–346. https://doi.org/10.3347/kjp.2010.48.4.343

Chontananarth T, Anucherngchai S, Tejangkura T (2018) The rapid detection method by polymerase chain reaction for minute intestinal trematodes: *Haplorchis taichui* in intermediate snail hosts based on 18s ribosomal DNA. Journal of Parasitic Diseases: Official Organ of the Indian Society for Parasitology 42(3): 423–432. https://doi.org/10.1007/s12639-018-1020-0

Chou HY, Lo CF, Tung MC, Wang CH, Fukuda H, Sano T (1993) The General Characteristics of a birnavirus isolated from cultured loach (*Misgurnus anguillicaudatus*) in Taiwan. Fish Pathology 28(1): 1–7. https://doi.org/10.3147/jsfp.28.1

Civanova K, Koyun M, Faculty L, Republic C (2013) The molecular and morphometrical description of a new diplozoid species from the gills of the *Garra rufa* (Heckel, 1843) (Cyprinidae) from Turkey – including a commentary on taxonomic division of Diplozoidae. 112: 3053–3062. https://doi.org/10.1007/s00436-013-3480-6

Copp GH, Bianco PG, Bogutskaya NG, Erős T, Falka I, Ferreira MT, Fox MG, Freyhof J, Gozlan RE, Grabowska J, Kováč V, Moreno-Amich R, Naseka AM, Peňáz M, Povž M, Przybylski M, Robillard M, Russell IC, Stakėnas S, Šumer S, Vila-Gispert A, Wiesner C (2005a) To be, or not to be, a non-native freshwater fish? Journal of Applied Ichthyology 21(4): 242–262. https://doi.org/10.1111/j.1439-0426.2005.00690.x

Copp GH, Wesley KJ, Vilizzi L (2005b) Pathways of ornamental and aquarium fish introductions into urban ponds of Epping Forest (London, England): The human vector. Journal of Applied Ichthyology 21(4): 263–274. https://doi.org/10.1111/j.1439-0426.2005.00673.x

Copp GH, Templeton M, Gozlan RE (2007) Propagule pressure and the invasion risks of non-native freshwater fishes in Europe: A case study of England. Journal of Fish Biology 71(Supplement D): 148–159. https://doi.org/10.1111/j.1095-8649.2007.01680.x

Copp GH, Vilizzi L, Mumford JD, Fenwick GV, Godard MJ, Gozlan RE (2009) Calibration of FISK, an invasive-ness screening tool for non-native freshwater fishes. Risk Analysis 29(3): 457–467. https://doi.org/10.1111/j.1539-6924.2008.01159.x

Copp GH, Russell IC, Peeler EJ, Gherardi F, Tricarico E, MacLeod A, Cowx IG, Nunn AD, Occhipinti Ambrogi A, Savini D, Mumford JD, Britton JR (2016) European Non-native Species in Aquaculture Risk Analysis Scheme – a summary of assessment protocols and decision making tools for use of alien species in aquaculture. Fisheries Management and Ecology 23(1): 1–11. https://doi.org/10.1111/fme.12074

Cornwell ER, Bellmund CA, Groocock GH, Wong PT, Hambury KL, Getchell RG, Bowser PL (2013) Fin and gill biopsies are effective nonlethal samples for detection of viral hemorrhagic septicemia virus genotype IVb. Journal of Veterinary Diagnostic Investigation 25(2): 203–209. https://doi.org/10.1177/1040638713476865

Courtenay Jr WR (1999) Aquariums and water gardens as vectors of introduction. In: Claudi R, Leach JH (Eds) Nonindigenous Freshwater Organisms: Vectors, Biology, and Impacts. Lewis Publishers, Boca Raton, Florida, 127–128.

Crossman EJ, Cudmore BC (1999) Summary of North American fish introductions through the aquarium/horti- culture trade. In: Claudi R, Leach JH (Eds) Nonindige- nous Freshwater Organisms: Vectors, Biology, and Impacts. Lewis Publishers, Boca Raton, 129–134.

Davidovich N, Tobia P, Blum SE, Zina B, Rona G, Kaidar-Shwartz H, Zeev D, Efrat R (2019) Mycobacterium gordonae infecting redclaw crayfish *Cherax quadricarinatus.* Diseases of Aquatic Organisms 135(2): 169–174. https://doi.org/10.3354/dao03392

De NC, Maity RN (1996) *Pseudocapillaria* (*Discocapillaria*) *margolisi* n. subg., n. sp. (Nematoda: Trichuroidea) from freshwater fishes of West Bengal, India. Systematic Parasitology 34(1): 49–52. https://doi.org/10.1007/BF01531210

de Azevedo RK, Abdallah VD, Luque JL (2007) Community ecology of metazoan parasites of apaiarí *Astronotus ocellatus* (Cope, 1872) (Perciformes: Cichlidae) from Guandu river, State of Rio de Janeiro, Brazil. Revista Brasileira de Parasitologia Veterinária 16: 15–20.

de Azevedo RK, Abdallah VD, da Silva RJ, de Azevedo TMP, Martins ML, Luque JL (2012) Expanded description of *Lamproglena monodi* (Copepoda: Lernaeidae), parasitizing native and introduced fishes in Brazil. Revista Brasileira de Parasitologia Veterinária 21(3): 263–269. https://doi.org/10.1590/S1984-29612012000300015

Dechruska W, Krailas D (2007) Trematode infections of the freshwater snail family Thiaridae in the Khek River, Thailand. The Southeast Journal of Tropical Medicine and Public Health 38: 1016–1028.

Dey VK (2016) The global trade in ornamental fish. Infofish International 4: 52–55. https://www.bassleer.com/ornamentalfishexporters/wp-content/uploads/sites/3/2016/12/GLOBAL-TRADE-IN-ORNAMENTAL-FISH.pdf

Ditrich O, Scholz T, Giboda M (1990) Occurrence of some medically important flukes (Trematoda: Opisthorchiidae and Heterophyidae) in Nam Ngum water reservoir, Laos. The Southeast Asian Journal of Tropical Medicine and Public Health 21: 482–488.

Dove ADM (2000) Richness patterns in the parasite communities of exotic poeciliid fishes. Parasitology 120(6): 609–623. https://doi.org/10.1017/S0031182099005958

Dove ADM, Ernst I (1998) Concurrent invaders - Four exotic species of Monogenea now established on exotic freshwater fishes in Australia. International Journal for Parasitology 28(11): 1755–1764. https://doi.org/10.1016/S0020-7519(98)00134-9

Duggan IC, Rixon CA, MacIsaac HJ (2006) Popularity and propagule pressure: Determinants of introduction and establishment of aquarium fish. Biological Invasions 8(2): 377–382. https://doi.org/10.1007/s10530-004-2310-2

Edgerton BF, Webb R, Anderson IG, Kulpa EC (2000) Description of a presumptive hepatopancreatic reovirus, and a putative gill parvovirus, in the freshwater crayfish *Cherax quadricarinatus.* Diseases of Aquatic Organisms 41: 83–90. https://doi.org/10.3354/dao041083

Eide KE, Miller-Morgan T, Heidel JR, Kent ML, Bildfell RJ, LaPatra S, Watson G, Jin L (2011) Investigation of Koi Herpesvirus Latency in Koi. Journal of Virology 85(10): 4954–4962. https://doi.org/10.1128/JVI.01384-10

Emde S, Kochmann J, Kuhn T, Dörge DD, Plath M, Miesen FW, Klimpel S (2016) Cooling water of power plant creates "hot spots" for tropical fishes and parasites. Parasitology Research 115(1): 85–98. https://doi.org/10.1007/s00436-015-4724-4

Evers HG, Pinnegar JK, Taylor MI (2019) Where are they all from? – sources and sustainability in the ornamental freshwater fish trade. Journal of Fish Biology 94: 909–916. https://doi.org/10.1111/jfb.13930

Forest JJH, King SD, Cone DK (2009) Occurrence of Glugea pimephales in planktonic larvae of fathead minnow in Algonquin Park, Ontario. Journal of Aquatic Animal Health 21(3): 164–166. https://doi.org/10.1577/H08-057.1

Foster R, Peeler E, Bojko J, Clark PF, Morritt D, Roy HE, Stebbing P, Tidbury HJ, Wood LE, Bass D (2021) Pathogens co-transported with invasive non-native aquatic species: Implications for risk analysis and legislation. NeoBiota 69: 79–102. https://doi.org/10.3897/neobiota.71358

Garcia F, Fujimoto RY, Martins ML, Moraes FR (2009) Parasitos protozoários de *Xiphophorus* sp. (Poeciliidae) e a relação deles com as características da água. Arquivo Brasileiro de Medicina Veterinária e Zootecnia 61: 156–162. https://doi.org/10.1590/S0102-09352009000100022

Girisha SK, Kushala KB, Nithin MS, Puneeth TG, Naveen Kumar BT, Vinay TN, Suresh T, Ajay SK, Venugopal MN, Ramesh KS (2021) First report of the infectious spleen and kidney necrosis virus (ISKNV) infection in ornamental fishes in India. Transboundary and Emerging Diseases 68(2): 964–972. https://doi.org/10.1111/tbed.13793

Go J, Waltzek TB, Subramaniam K, Yun SC, Groff JM, Anderson IG, Chong R, Shirley I, Schuh JCL, Handlinger JH, Tweedie A, Whittington RJ (2016) Detection of infectious spleen and kidney necrosis virus (ISKNV) and turbot reddish body iridovirus (TRBIV) from archival ornamental fish samples. Diseases of Aquatic Organisms 122(2): 105–123. https://doi.org/10.3354/dao03068

Gomez DK, Dong JL, Gun WB, Hee JY, Nam SS, Hwa YY, Cheol YH, Jun HP, Se CP (2006) Detection of betanodaviruses in apparently healthy aquarium fishes and invertebrates. Journal of Veterinary Science (Suwon-si, Korea) 7(4): 369–374. https://doi.org/10.4142/jvs.2006.7.4.369

Hijmans RJ, Cameron SE, Parra JL, Jones PG, Jarvis A (2005) Very high resolution interpolated climate surfaces for global land areas. International Journal of Climatology 25(15): 1965–1978. https://doi.org/10.1002/joc.1276

Hood Y, Sadler J, Poldy J, Starkey CS, Robinson AP (2019) Biosecurity system reforms and the development of a risk-based surveillance and pathway analysis system for ornamental fish imported into Australia. Preventive Veterinary Medicine 167: 159–168. https://doi.org/10.1016/j.prevetmed.2018.11.006

Hoshino ÉM, Hoshino MDFG, Tavares-Dias M (2018) Parasites of ornamental fish commercialized in Macapá, Amapá State (Brazil). Revista Brasileira de Parasitologia Veterinaria 27: 75–80. https://doi.org/10.1590/S1984-29612018002

Hsieh C, Huang C, Pan Y (2016) Crayfish plague *Aphanomyces astaci* detected in redclaw crayfish, *Cherax quadricarinatus* in Taiwan. Journal of Invertebrate Pathology 136: 117–123. https://doi.org/10.1016/j.jip.2016.03.015

Hudson PL, Bowen CA II (2002) First Record of *Neoergasilus japonicus* (Poecilostomatoida : Ergasilidae), a parasitic copepod new to the Laurentian Great Lakes. The Journal of Parasitology 88(4): 657–663. https://doi.org/10.1645/0022-3395(2002)088[0657:FRONJP]2.0.CO;2

Humphrey JD, Lancaster C, Gudkovs N, McDonald W (1986) Exotic bacterial pathogens Edwardsiella tarda and Edwardsiella ictaluri from imported ornamental fish *Betta splendens* and *Puntius conchonius*, respectively: Isolation and quarantine significance. Australian Veterinary Journal 63(11): 369–371. https://doi.org/10.1111/j.1751-0813.1986.tb02900.x

Iwanowicz LR, Goodwin AE (2002) A new bacilliform fathead minnow rhabdovirus that produces syncytia in tissue culture. Archives of Virology 147(5): 899–915. https://doi.org/10.1007/s00705-001-0793-z

Jithila PJ, Prasadan PK (2018) Description of *Tetracotyle wayanadensis* n. sp. (Digenea: Strigeidae) metacercaria infecting six species of freshwater fishes from Western Ghats, India. Journal of Parasitic Diseases: Official Organ of the Indian Society for Parasitology 42(2): 226–231. https://doi.org/10.1007/s12639-018-0988-9

Jun JW, Kim JH, Gomez DK, Choresca Jr CH, Han JE, Shin SP, Park SC (2010) Occurrence of tetracycline-resistant *Aeromonas hydrophila* infection in Korean cyprinid loach (*Misgurnus anguillicaudatus*). African Journal of Microbiological Research 4: 849–855.

Keller RP, Lodge DM (2007) Species invasions from commerce in live aquatic organisms: Problems and possible solutions. Bioscience 57(5): 428–436. https://doi.org/10.1641/B570509

Kim JH, Hayward CJ, Joh SJ, Heo GJ (2002) Parasitic infections in live freshwater tropical fishes imported to Korea. Diseases of Aquatic Organisms 52: 169–173. https://doi.org/10.3354/dao052169

Kim SG, Giri SS, Kim SW, Kwon J, Lee S, Park SC (2020) First isolation and characterization of *chryseobacterium cucumeris* sknucl01, isolated from diseased pond loach (*Misgurnus anguillicaudatus*) in korea. Pathogens (Basel, Switzerland) 9(5): 1–15. https://doi.org/10.3390/pathogens9050397

King TA (2019) Wild caught ornamental fish: A perspective from the UK ornamental aquatic industry on the sustainability of aquatic organisms and livelihoods. Journal of Fish Biology 94(6): 925–936. https://doi.org/10.1111/jfb.13900

Knipes AK, Janovy Jr J (2009) Community structure and seasonal dynamics of *Dactylogyrus* Spp. (monogenea) on the fathead minnow (*Pimephales promelas*) from the salt valley watershed, Lancaster County, Nebraska. The Journal of Parasitology 95(6): 1295–1305. https://doi.org/10.1645/GE-2166.1

Koda SA, Subramaniam K, Francis-Floyd R, Yanong RP, Frasca Jr S, Groff JM, Popov VL, Fraser WA, Yan A, Mohan S, Waltzek TB (2018) Phylogenomic characterization of two novel members of the genus *Megalocytivirus* from archived ornamental fish samples. Diseases of Aquatic Organisms 130(1): 11–14. https://doi.org/10.3354/dao03250

Kritsky DC, Van Every LR, Boeger WA (1996) Neotropical monogenoidea. 27. Two new species of *Telethecium* gen. n. from the nasal cavities of central amazonian fishes and a redescription of *Kritskyia moraveci* Kohn, 1990 (Dactylogyridae, Ancyrocephalinae). Comparative Parasitology 63: 35–41.

Kwon SR, Kim HJ (2011) *Thelohanellus misgurni* (Kudo, 1919) infection on the fins of Chinese muddy loach *Misgurnus mizolepis.* Journal of Fish Pathology 24(2): 167–171. https://doi.org/10.7847/jfp.2011.24.2.167

Langenmayer MC, Lewisch E, Gotesman M, Hoedt W, Schneider M, El-Matbouli M, Hermanns W (2015) Cutaneous infection with *Dermocystidium salmonis* in cardinal tetra, *Paracheirodon axelrodi* (Schultz, 1956). Journal of Fish Diseases 38(5): 503–506. https://doi.org/10.1111/jfd.12281

Lari E, Adams RV, Cone DK, Goater CP, Pyle GG (2016) *Dactylogyrus olfactorius* n. sp. (Monogenea, Dactylogyridae) from the olfactory chamber of the fathead minnow, *Pimephales promelas* Rafinesque (Cyprinidae). Systematic Parasitology 93(6): 575–581. https://doi.org/10.1007/s11230-016-9649-5

Leal MC, Vaz MCM, Puga J, Rocha RJM, Brown C, Rosa R, Calado R (2016) Marine ornamental fish imports in the European Union: An economic perspective. Fish and Fisheries 17(2): 459–468. https://doi.org/10.1111/faf.12120

Liu C, Olden JD (2017) Heads you win, tails you lose: Life-history traits predict invasion and extinction risk of the world's freshwater fishes. Aquatic Conservation 27(4): 773–779. https://doi.org/10.1002/aqc.2740

Logez M, Bady P, Pont D (2012) Modelling the habitat requirement of riverine fish species at the European scale: Sensitivity to temperature and precipitation and associated uncertainty. Ecology Freshwater Fish 21(2): 266–282. https://doi.org/10.1111/j.1600-0633.2011.00545.x

López-Olmeda JF, Sánchez-Vázquez FJ (2011) Thermal biology of zebrafish (*Danio rerio*). Journal of Thermal Biology 36(2): 91–104. https://doi.org/10.1016/j.jtherbio.2010.12.005

Lymbery AJ, Morine M, Kanani HG, Beatty SJ, Morgan DL (2014) Co-invaders: The effects of alien parasites on native hosts. International Journal for Parasitology. Parasites and Wildlife 3(2): 171–177. https://doi.org/10.1016/j.ijppaw.2014.04.002

Majtán J, Černy J, Ofúkana A, Takáč P, Kozánek M (2012) Mortality of therapeutic fish *Garra rufa* caused by *Aeromonas sobria.* Asian Pacific Journal of Tropical Biomedicine 2(2): 85–87. https://doi.org/10.1016/S2221-1691(11)60197-4

Marcogliese DJ (1991) Seasonal occurrence of *Lernaea cyprinacea* on fishes in Belews Lake, North Carolina. The Journal of Parasitology 77(2): 326–327. https://doi.org/10.2307/3283108

Marcos-López M, Gale P, Oidtmann BC, Peeler EJ (2010) Assessing the impact of climate change on disease emergence in freshwater fish in the United Kingdom. Transboundary and Emerging Diseases 57(5): 293–304. https://doi.org/10.1111/j.1865-1682.2010.01150.x

Martins ML, Tavares-Dias M, Janik AJ, Kent ML, Jerônimo GT (2017) Hematology and condition factor of tui chub and fathead minnow parasitized by nematode from Upper Klamath Lake, Oregon, USA. Diseases of Aquatic Organisms 126(3): 257–262. https://doi.org/10.3354/dao03168

McKoy SA, Hyslop EJ, Robinson RD (2011) Associations between two trematode parasites, an ectosymbiotic annelid, and thiara (*Tarebia*) *granifera* (Gastropoda) in Jamaica. The Journal of Parasitology 97(5): 828–832. https://doi.org/10.1645/GE-2494.1

Mendoza-Franco EF, Aguirre-Macedo ML, Vidal-Martínez VM (2007) New and previously described species of dactylogyridae (Monogenoidea) from the gills of Panamanian freshwater fishes (Teleostei). The Journal of Parasitology 93(4): 761–771. https://doi.org/10.1645/GE-1068R.1

Mendoza-Franco EF, Scholz T, Rozkošná P (2010) *Tucunarella* n. gen. and other dactylogyrids (monogenoidea) from cichlid fish (perciformes) from peruvian Amazonia. Journal of Parasitology 96(3): 491–498. https://doi.org/10.1645/GE-2213.1

Mirzaei M (2015) Prevalence and histopathologic study of *Lernaea cyprinacea* in two species of ornamental fish (*Poecilia latipinna* and *Xiphophorus helleri*) in Kerman, South-East Iran. Turkiye Parazitoloji Dergisi 39(3): 222–226. https://doi.org/10.5152/tpd.2015.3960

Mitchell AJ, Smith CE, Hoffman GL (1982) Pathogenicity and histopathology of an unusually intense infection of white grubs (*Posthodiplostomum m. minimum*) in the fathead minnow (*Pimephales promelas*). Journal of Wildlife Diseases 18(1): 51–57. https://doi.org/10.7589/0090-3558-18.1.51

Mood SM, Mousavi HAE, Mokhayer BA, Ahmadi M, Soltani M, Sharifpour I (2010) *Centrocestus formosanus* metacercarial infection of four ornamental fish species imported into Iran. Bulletin of the European Association of Fish Pathologists 30: 146–149.

Mood S M., Rassouli M (2016) Monogenean infestations of arowana (*Osteoglossum bicirrhosum*) and cat fish (*Hypostomus plecostomus*). Iranian Journal of Fisheries Science 15: 606–612. http://hdl.handle.net/1834/37643

Mor SK, Phelps NBD, Ng TFF, Subramaniam K, Primus A, Armien AG, McCann R, Puzach C, Waltzek TB, Goyal SM (2017) Genomic characterization of a novel calicivirus, FHMCV-2012, from baitfish in the USA. Archives of Virology 162(12): 3619–3627. https://doi.org/10.1007/s00705-017-3519-6

Moravec F, Justine JL (2006) *Camallanus cotti* (Nematoda: Camallanidae), an introduced parasite of fishes in New Caledonia. Folia Parasitologica 53(4): 287–296. https://doi.org/10.14411/fp.2006.035

Moravec F, Jiménez-García MI, Salgado-Maldonado G (1998) New observations on *Mexiconema cichlasomae* (Nematoda: Dracunculoidea) from fishes in Mexico. Parasite (Paris, France) 5(3): 289–293. https://doi.org/10.1051/parasite/1998053289

Moss RH, Edmonds JA, Hibbard KA, Manning MR, Rose SK, Van Vuuren DP, Carter TR, Emori S, Kainuma M, Kram T, Meehl GA, Mitchell JFB, Nakicenovic N, Riahi K, Smith SJ, Stouffer RJ, Thomson AM, Weyant JP, Wilbanks TJ (2010) The next generation of scenarios for climate change research and assessment. Nature 463(7282): 747–756. https://doi.org/10.1038/nature08823

Mousavi HAE, Omidzahir S, Soltani M, Shayan P, Ebrahimzadeh E, Mousavi S, Hoseini M (2013) Morphometrical and molecular characterization of *Gyrodactylus cichlidarum* (Gyrodactylidae) from *Astronotus ocellatus* (Cichlidae) in Iran. Comparative Clinical Pathology 22(6): 1093–1097. https://doi.org/10.1007/s00580-012-1534-2

Nagasawa K, Torii R-I (2014) The parasitic copepod *Lernaea cyprinacea* from freshwater fishes, including alien species (*Gambusia affinis* and *Rhodeus ocellatus ocellatus*), in central Japan. Biosphere Science 53: 27–31.

Naimi B, Araújo MB (2016) Sdm: A reproducible and extensible R platform for species distribution modelling. Ecography 39(4): 368–375. https://doi.org/10.1111/ecog.01881

Nawa Y, Noda S, Uchiyama-Nakamura F, Ishiwata K (2001) Current status of food-borne parasitic zoonoses in Japan. The Southeast Asian Journal of Tropical Medicine and Public Health 32: 4–7.

Nazarenko L, Schmidt GA, Miller RL, Tausnev N, Kelley M, Reudy R, Russell GL, Aleinov I, Bauer M, Bauer S, Bleck R, Canuto V, Cheng Y, Clune TL, Del Genio AD, Faluvegi G, Hansen JE, Healy RJ, Kiang NY, Koch D, Lacis AA, LeGrande AN, Lerner J, Lo KK, Menon S, Oinas V, Perlwitz J, Puma MJ, Rind D, Romanou A, Sato M, Shindell DT, Sun S, Tsigaridis K, Unger N, Voulgarakis A, Yao MS, Zhang J (2015) Future climate change under RCP emission scenarios with GISS ModelE2. Journal of Advances in Modeling Earth Systems 7(1): 244–267. https://doi.org/10.1002/2014MS000403

Neves LR, Pereira FB, Tavares-Dias M, Luque JL (2013) Seasonal influence on the parasite fauna of a wild population of *Astronotus ocellatus* (Perciformes: Cichlidae) from the Brazilian Amazon. The Journal of Parasitology 99(4): 718–721. https://doi.org/10.1645/12-84.1

Ng TH, Tan SK, Wong WH, Meier R, Chan SY, Tan HH, Yeo DCJ (2016) Molluscs for sale: Assessment of freshwater gastropods and bivalves in the ornamental pet trade. PLoS ONE 11(8): 1–23. https://doi.org/10.1371/journal.pone.0161130

Ngamniyom A, Sriyapai T, Sriyapai P, Panyarachun B (2019) Contributions to the knowledge of *Pseudolevinseniella* (Trematoda: Digenea) and temnocephalans from alien cray fish in natural freshwaters of Thailand. Heliyon 5(12): e02990. https://doi.org/10.1016/j.heliyon.2019.e02990

Nitta M, Nagasawa K (2018) *Gyrodactylus medaka* n. sp. (Monogenea: Gyrodactylidae) parasitic on wild and laboratory-reared medaka *Oryzias latipes* (Beloniformes: Adrianichthyidae) in Japan. Parasitology International 67(5): 651–658. https://doi.org/10.1016/j.parint.2018.06.010

Novotný L, Dvořák P (2001) Manifestation of mycobacteriosis in cardinal tetras *Paracheirodon axelrodi* (Schultz,1956) during the *Pleistophora hyphessobryconis* (Schaperclaus, 1941) infection. Folia Veterinaria 50: 80–82.

OATA (2020) Annual Review 2019/20: Reflecting on a changing world.

OATA (2021) Apple snails can return to GB Aquariums after restrictions lift. https://ornamentalfish.org/apple-snails-can-return-to-gb-after-restriction

Ohyama F, Okino T, Ushirogawa H (2001) *Massaliatrema misgurni* n. sp. (Trematoda: Heterophyidae) whose metacercariae encyst in loaches (*Misgurnus anguillicaudatus*). Parasitology International 50(4): 267–271. https://doi.org/10.1016/S1383-5769(01)00084-8

OIE (Office International des Epizooties) (2019) Infection with koi herpesvirus. Manual of diagnostic tests for aquatic animals. https://www.oie.int/index.php?id=2439&L=0&htmfile=chapitre_koi_herpesvirus.htm

Oliveira MSB, Corrêa LL, Tavares-Dias M (2019) Helminthic endofauna of four species of fish from lower Jari river, a tributary of the Amazon basin in Brazil. Boletim do Instituto de Pesca 45(1): e393. https://doi.org/10.20950/1678-2305.2019.45.1.393

Owens L, McElnea C (2000) Natural infection of the redclaw crayfish *Cherax quadricarinatus* with presumptive spawner-isolated mortality virus. Diseases of Aquatic Organisms 40: 219–223. https://doi.org/10.3354/dao040219

Padilla D, Williams S (2004) Beyond ballast water: Aquarium and ornamental trades as sources of invasive species in aquatic ecosystems. Frontiers in Ecology and the Environment 2(3): 131–138. https://doi.org/10.1890/1540-9295(2004)002[0131:BBWAAO]2.0.CO;2

Paller VGV, Macaraig JRM, Verona RT, Estaño LA (2019) Cercarial fauna of freshwater snails in selected agricultural areas in Laguna, Philippines. Helminthologia (Poland) 56(1): 81–86. https://doi.org/10.2478/helm-2018-0040

Peeler EJ, Oidtmann BC, Midtlyng PJ, Miossec L, Gozlan RE (2011) Non-native aquatic animals introductions have driven disease emergence in Europe. Biological Invasions 13(6): 1291–1303. https://doi.org/10.1007/s10530-010-9890-9

Peyghan R, Rahnama R, Dezfuly ZT, Shokoohmand M (2019) *Achlya* infection in an oscar (*Astronotus ocellatus*) with typical symptoms of saprolegniosis. Veterinary Research Forum 10: 89–92.

Phelps NBD, Mor SK, Armien AG, Batts W, Goodwin AE, Hopper L, McCann R, Ng TFF, Puzach C, Waltzek TB, Delwart E, Winton J, Goyal SM (2014) Isolation and molecular characterization of a novel picornavirus from baitfish in the USA. PLoS ONE 9(2): e87593. https://doi.org/10.1371/journal.pone.0087593

Piazza RS, Martins ML, Guiraldelli L, Yamashita MM (2006) Parasitic diseases of freshwater ornamental fishes commercialized in Florianópolis, Santa Catarina, Brazil. Boletim do Instituto de Pesca 32: 51–57.

Pie MR, Boeger WA (2006) Density-dependent topographical specialization in *Gyrodactylus anisopharynx* (Monogenoidea, Gyrodactylidae): Boosting transmission or evading competition? Journal of Parasitology 92(3): 459–463. https://doi.org/10.1645/GE-641.1

Pinder AC, Gozlan RE (2003) Sunbleak and topmouth gudgeon – two new additions to Britain's freshwater fishes. British Wildlife (December): 77–83.

Pinheiro RHDS, Tavares-Dias M, Giese EG (2019) Helminth parasites in two populations of *Astronotus ocellatus* (Cichliformes: Cichlidae) from the eastern amazon, northern brazil. Revista Brasileira de Parasitologia Veterinaria 28: 425–431. https://doi.org/10.1590/S1984-29612019052

Pinnegar JK, Murray JM (2019) Understanding the United Kingdom marine aquarium trade – a mystery shopper study of species on sale. Journal of Fish Biology 94: 917–924. https://doi.org/10.1111/jfb.13941

Plaul SE, Romero NG, Barbeito CG (2010) Distribution of the exotic parasite, *Lernaea cyprinacea* (Copepoda, Lernaeidae) in Argentina. Bulletin of the European Association of Fish Pathologists 30: 65–73.

Popazoglo F, Boeger WA (2016) Neotropical Monogenoidea 37. Redescription of *Gyrodactylus superbus* (Szidat, 1973) comb. n. and description of two new species of *Gyrodactylus* (Gyrodactylidea: Gyrodactylidae) from *Corydoras paleatus* and *C. ehrhardti* (Teleostei: Siluriformes: Callichthyidae) of Southern Brazil. Folia Parasitologica 47(2): 105–110. https://doi.org/10.14411/fp.2000.022

Pradhan PK, Rathore G, Sood N, Swaminathan TR, Yadav MK, Verma DK, Chaudhary DK, Abidi R, Punia P, Jena JK (2014) Emergence of epizootic ulcerative syndrome: Large-scale mortalities of cultured and wild fish species in Uttar Pradesh, India. Current Science 106: 1711–1718. http://www.jstor.org/stable/24103006

Qin L, Zhu M, Xu J (2014) First report of *Shewanella* sp. and *Listonella* sp. infection in freshwater cultured loach, *Misgurnus anguillicaudatus.* Aquaculture Research 45(4): 602–608. https://doi.org/10.1111/j.1365-2109.2012.03260.x

Qiu JH, Zhang Y, Zhang XX, Gao Y, Li Q, Chang QC, Wang CR (2017) Metacercaria infection status of fishborne zoonotic trematodes, except for *Clonorchis sinensis* in fish from the Heilongjiang Province, China. Foodborne Pathogens and Disease 14(8): 440–446. https://doi.org/10.1089/fpd.2016.2249

Rauque C, Viozzi G, Flores V, Vega R, Waicheim A, Salgado-maldonado G (2018) Helminth parasites of alien freshwater fishes in Patagonia (Argentina). International Journal for Parasitology: Parasites and Wildlife 7(3): 369–379. https://doi.org/10.1016/j.ijppaw.2018.09.008

Řehulka J, Kolařík M, Hubka V (2017) Disseminated infection due to *Exophiala pisciphila* in Cardinal tetra, *Paracheirodon axelrodi*. Journal of Fish Diseases 40(8): 1015–1024. https://doi.org/10.1111/jfd.12577

Reyda FB, Wells SM, Ermolenko AV, Zietara MS, Lumme JI (2020) Global parasite trafficking: Asian *Gyrodactylus* (Monogenea) arrived to the U.S.A. via invasive fish *Misgurnus anguillicaudatus* as a threat to amphibians. Biological Invasions 22(2): 391–402. https://doi.org/10.1007/s10530-019-02097-4

Rhee JK, Baek BK, Lee SB, Koh HB (1983) Epidemiological studies of *Clonorchis Sinensis* in Mangyeong riverside areas in Korea. Korean Journal of Parasitology 21(2): 157–166. https://doi.org/10.3347/kjp.1983.21.2.157

Rhyne AL, Tlusty MF, Schofield PJ, Kaufman L, Morris JA, Bruckner AW (2012) Revealing the appetite of the marine aquarium fish trade: The volume and biodiversity of fish imported into the United States. PLoS ONE 7(5): e35808. https://doi.org/10.1371/journal.pone.0035808

Rhyne AL, Tlusty MF, Szczebak JT, Holmberg RJ (2017) Expanding our understanding of the trade in marine aquarium animals. PeerJ 5: e2949. https://doi.org/10.7717/peerj.2949

Rodrigues MNG, Dias MKR, Marinho RDB, Tavares-Dias M (2014) Parasite Diversity Of *Osteoglossum Bicirrhosum*, An Osteoglossidae Fish From Amazon. Neotropical Helminthology 8.

Romero X, Turnbull JF, Jiménez R (2000) Ultrastructure and cytopathology of a rickettsia-like organism causing systemic infection in the redclaw crayfish, *Cherax quadricarinatus* (Crustacea: Decapoda), in Ecuador. Journal of Invertebrate Pathology 76(2): 95–104. https://doi.org/10.1006/jipa.2000.4952

Ruane NM, Collins EM, Geary M, Swords D, Hickey C, Geoghegan F (2013) Isolation of *Streptococcus agalactiae* and an aquatic birnavirus from doctor fish *Garra rufa* L. Irish Veterinary Journal 66(1): 2–5. https://doi.org/10.1186/2046-0481-66-16

Ruehl-Fehlert C, Bomke C, Dorgerloh M, Palazzi X, Rosenbruch M (2005) *Pleistophora* infestation in fathead minnows, *Pimephales promelas* (Rafinesque). Journal of Fish Diseases 28(11): 629–637. https://doi.org/10.1111/j.1365-2761.2005.00661.x

Ryang YS (1990) Studies on *Echinostoma* spp. in the Chungju Reservoir and upper streams of the Namhan River. Korean Journal of Parasitology 28(4): 221–233. https://doi.org/10.3347/kjp.1990.28.4.221

Sakuna K, Elliman J, Owens L (2017) Discovery of a novel Picornavirales, Chequa iflavirus, from stressed redclaw crayfish (*Cherax quadricarinatus*) from farms in northern Queensland, Australia. Virus Research 238: 148–155. https://doi.org/10.1016/j.virusres.2017.06.021

Sakuna K, Elliman J, Tzamouzaki A, Owens L (2018) A novel virus (order Bunyavirales) from stressed redclaw crayfish (*Cherax quadricarinatus*) from farms in northern Australia. Virus Research 250: 7–12. https://doi.org/10.1016/j.virusres.2018.03.012

Salgado-Maldonado G (2013) Redescription of *Neoechinorhynchus* (*Neoechinorhynchus*) *golvani* Salgado-Maldonado, 1978 (Acanthocephala: Neoechinorhynchidae) and description of a new species from freshwater cichlids (Teleostei: Cichlidae) in Mexico. Parasitology Research 112(5): 1891–1901. https://doi.org/10.1007/s00436-013-3374-7

Sanders JL, Watral V, Stidworthy MF, Kent ML (2016) Expansion of the known host range of the *Microsporidium*, *Pseudoloma neurophilia.* Zebrafish 13(S1): S102–S106. https://doi.org/10.1089/zeb.2015.1214

Sandland GJ, Goater CP (2001) Parasite-Induced variation in host morphology: Brain-encysting trematodes in fathead minnows. The Journal of Parasitology 87(2): 267–272. https://doi.org/10.1645/0022-3395(2001)087[0267:PIVIHM]2.0.CO;2

Sandland GJ, Goater CP, Danylchuk AJ (2001) Population dynamics of *Ornithodiplostomum ptychocheilus* metacercariae in fathead minnows (*Pimephales promelas*) from four northern-Alberta lakes. Journal of Parasitology 87(4): 744–748. https://doi.org/10.1645/0022-3395(2001)087[0744:PDOOPM]2.0.CO;2

Schleppe JL, Goater CP (2004) Comparative life histories of two diplostomid trematodes, *Ornithodiplostomum ptychocheilus* and *Posthodiplostomum minimum.* Journal of Parasitology 90(6): 1387–1390. https://doi.org/10.1645/GE-274R

Scholz T, Shimazu T, Olson PD, Nagasawa K (2001) Caryophyllidean tapeworms (Platyhelminthes: Eucestoda) from freshwater fishes in Japan. Folia Parasitologica 48(4): 275–288. https://doi.org/10.14411/fp.2001.046

Scholz T, Oros M, Bazsalovicsová E, Brabec J, Waeschenbach A, Xi BW, Aydoğdu A, Besprozvannykh V, Shimazu T, Králová-Hromadová I, Littlewood DTJ (2014) Molecular evidence of cryptic diversity in *Paracaryophyllaeus* (Cestoda: Caryophyllidea), parasites of loaches (Cobitidae) in Eurasia, including description of *P. vladkae* n. sp. Parasitology International 63(6): 841–850. https://doi.org/10.1016/j.parint.2014.07.015

Seo BS, Park YH, Chai JY, Hong SJ, Lee SH (1984) Studies on intestinal trematodes in Korea xiv. infection status of loaches with metacercariae of *Echinostoma cinetorchis* and their development in albino rats. Korean Journal of Parasitology 22(2): 181–189. https://doi.org/10.3347/kjp.1984.22.2.181

Shamsi S, Stoddart A, Smales L, Wassens S (2019) Occurrence of *Contracaecum bancrofti* larvae in fish in the Murray-Darling Basin. Journal of Helminthology 93(05): 574–579. https://doi.org/10.1017/S0022149X1800055X

Shin DS (1964) Epidemiological studies of *Clonorchis sinensis* prevailed in the peoples of Kyungpook Province. Korean Journal of Parasitology 2(1): 1–13. https://doi.org/10.3347/kjp.1964.2.1.1

Sicuro B, Pastorino P, Barbero R, Barisone S, Dellerba D, Menconi V, Righetti M, De Vita V, Prearo M (2020) Prevalence and antibiotic sensitivity of bacteria isolated from imported ornamental fish in Italy: A translocation of resistant strains? Preventive Veterinary Medicine 175: e104880. https://doi.org/10.1016/j.prevetmed.2019.104880

Smales LR, Aydogdu A, Emre Y (2012) Pomphorhynchidae and Quadrigyridae (Acanthocephala), including a new genus and species (Pallisentinae), from freshwater fishes, cobitidae and cyprinodontidae, in Turkey. Folia Parasitologica 59(3): 162–166. https://doi.org/10.14411/fp.2012.022

Sohn W-M, Kho W-G, Lee SH (1993) Larval *Gnathostoma nipponicum* found in the imported Chinese loaches. Korean Journal of Parasitology 31(4): 347–352. https://doi.org/10.3347/kjp.1993.31.4.347

Sokolova YY, Overstreet RM (2018) A new microsporidium, *Apotaspora heleios* n. g., n. sp., from the Riverine grass shrimp *Palaemonetes paludosus* (Decapoda: Caridea: Palaemonidae). Journal of Invertebrate Pathology 157: 125–135. https://doi.org/10.1016/j.jip.2018.05.007

Sreedharan K, Philip R, Singh ISB (2011) Isolation and characterization of virulent *Aeromonas veronii* from ascitic fluid of oscar *Astronotus ocellatus* showing signs of infectious dropsy. Diseases of Aquatic Organisms 94(1): 29–39. https://doi.org/10.3354/dao02304

Stevens GC (1992) The elevational gradient in altitudinal range : An extension of Rapoport's latitudinal rule to altitude. American Naturalist 140(6): 893–911. https://doi.org/10.1086/285447

Tahir UBIN, Deng Q, Li SEN, Liu Y, Wang ZHE, Gu Z (2017) First record of a new epibionts suctorian ciliate *Tokophrya huangmeiensis* sp. n. (Ciliophora, Phyllopharyngea) from redclaw crayfish *Cherax quadricarinatus* von Martens 1868. Zootaxa 4269(2): 287–295. https://doi.org/10.11646/zootaxa.4269.2.7

Taraschewski H (2006) Hosts and parasites as aliens. Journal of Helminthology 80(2): 99–128. https://doi.org/10.1079/JOH2006364

Tavakol S, Luus-Powell WJ, Smit WJ, Baker C, Hoffman A, Halajian A (2016) First introduction of two Australian temnocephalan species into Africa with an alien host: Double trouble. Journal of Parasitology 102(6): 653–658. https://doi.org/10.1645/15-936

Tavares-Dias M, Neves LR (2017) Diversity of parasites in wild *Astronotus ocellatus* (Perciformes, cichlidae), an ornamental and food fish in Brazil. Anais da Academia Brasileira de Ciências 89(3 suppl): 2305–2315. https://doi.org/10.1590/0001-3765201720160700

Tavares-Dias M, Lemos JRG, Martins ML (2010) Parasitic fauna of eight species of ornamental freshwater fish species from the middle Negro River in the Brazilian Amazon Region. Revista Brasileira de Parasitologia Veterinária 19(2): 29–33. https://doi.org/10.1590/S1984-29612010000200007

Tavares-Dias M, Sousa TJSM, Neves LR (2014) Infecções parasitárias em dois peixes bentopelágicos da amazônia: O aruanã *Osteoglossum bicirrhosum* (Osteoglossidae) e apaiari *Astronotus ocellatus* (Cichlidae). Bioscience Journal 30: 546–555.

Taylor NGH, Way K, Jeffery KR, Peeler EJ (2010) The role of live fish movements in spreading koi herpesvirus throughout England and Wales. Journal of Fish Diseases 33(12): 1005–1007. https://doi.org/10.1111/j.1365-2761.2010.01198.x

Taylor NGH, Norman RA, Way K, Peeler EJ (2011) Modelling the koi herpesvirus (KHV) epidemic highlights the importance of active surveillance within a national control policy. Journal of Applied Ecology 48(2): 348–355. https://doi.org/10.1111/j.1365-2664.2010.01926.x

Taylor NGH, Peeler EJ, Denham KL, Crane CN, Thrush MA, Dixon PF, Stone DM, Way K, Oidtmann BC (2013) Spring viraemia of carp (SVC) in the UK: The road to freedom. Preventive Veterinary Medicine 111(1–2): 156–164. https://doi.org/10.1016/j.prevetmed.2013.03.004

Thilakaratne IDSIP, Rajapaksha G, Hewakopara A, Rajapakse RPVJ, Faizal ACM (2003) Parasitic infections in freshwater ornamental fish in Sri Lanka. Diseases of Aquatic Organisms 54: 157–162. https://doi.org/10.3354/dao054157

Thrush MA, Peeler EJ (2013) A model to approximate lake temperature from gridded daily air temperature records and its application in risk assessment for the establishment of fish diseases in the uk. Transboundary and Emerging Diseases 60(5): 460–471. https://doi.org/10.1111/j.1865-1682.2012.01368.x

Tokşen E (2006) *Argulus foliacesus* (Crustacea: Branchiura) infestation on oscar, *Astronotus ocellatus* (Cuvier, 1829) and its treatment. Ege Journal of Fisheries & Aquatic Sciences 23: 177–179.

Tolley-Jordan LR, Chadwick MA (2018) Effects of parasite infection and host body size on habitat associations of invasive aquatic snails: Implications for environmental monitoring. Journal of Aquatic Animal Health 31(1): 121–128. https://doi.org/10.1002/aah.10059

Trujillo-González A, Becker JA, Vaughan DB, Hutson KS (2019a) Monogenean parasites infect ornamental fish imported to Australia. Parasitology Research 118(1): 383–384. https://doi.org/10.1007/s00436-018-6156-4

Trujillo-González A, Edmunds RC, Becker JA, Hutson KS (2019b) Parasite detection in the ornamental fish trade using environmental DNA. Scientific Reports 9(1): 1–10. https://doi.org/10.1038/s41598-019-41517-2

Tukmechi A, Hobbenaghi R, Rahmati Holasoo H, Morvaridi A (2009) *Streptococcosis* in a pet fish, *Astronotus ocellatus*: A case study. World Academy of Science, Engineering and Technology 37: 14–15.

UK Government (2019) Copy of ILFA Aquarium Ornamentals Alphabetic Checklist. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/919683/Copy_of_ILFA._Aquarium_Ornamentals._Alphabetic_checklist_for_BIPs.__1_.ods

UK Government (2021a) Introduce or keep non-native fish and shellfish. www.gov.uk/guidance/introduce-or-keep-non-native-fish-and-shellfish

UK Government (2021b) UK Risk Register Details for *Pomacea* spp. https://secure.fera.defra.gov.uk/phiw/riskRegister/viewPestRisks.cfm?cslref=6289

Ukong S, Krailas D, Dangprasert T, Channgarm P (2007) Studies on the morphology of cercariae obtained from freshwater snails at Erawan Waterfall, Erawan National Park, Thailand. The Southeast Journal of Tropical Medicine and Public Health 38: 303–312.

Van Vuuren DP, Edmonds J, Kainuma M, Riahi K, Thomson A, Hibbard K, Hurtt GC, Kram T, Krey V, Lamarque JF, Masui T, Meinshausen M, Nakicenovic N, Smith SJ, Rose SK (2011) The representative concentration pathways: An overview. Climatic Change 109(1–2): 5–31. https://doi.org/10.1007/s10584-011-0148-z

Vincent AG, Font WF (2003) Seasonal and yearly population dynamics of two exotic helminths, *Camallanus cotti* (nematoda) and *Bothriocephalus acheilognathi* (cestoda), parasitizing exotic fishes in Waianu Stream, O'ahu, Hawaii. The Journal of Parasitology 89(4): 756–760. https://doi.org/10.1645/GE-90R

Volpe E, Mandrioli L, Errani F, Serratore P, Zavatta E, Rigillo A, Ciulli S (2019) Evidence of fish and human pathogens associated with doctor fish (*Garra rufa*, Heckel, 1843) used for cosmetic treatment. Journal of Fish Diseases 42(12): 1637–1644. https://doi.org/10.1111/jfd.13087

Wang Z, Zhou T, Gu Z (2017a) New data of two trichodinid ectoparasites (Ciliophora: Trichodinidae) from farmed freshwater fishes in Hubei, China. European Journal of Protistology 60: 50–59. https://doi.org/10.1016/j.ejop.2017.04.002

Wang ML, Chen HY, Shih HH (2017b) Occurrence and distribution of yellow grub trematodes (*Clinostomum complanatum*) infection in Taiwan. Parasitology Research 116(6): 1761–1771. https://doi.org/10.1007/s00436-017-5457-3

Wang X, Li J, Cao X, Wang W, Luo Y (2019) Isolation, identification and characterisation of an emerging fish pathogen, *Acinetobacter pittii*, from diseased loach (*Misgurnus anguillicaudatus*) in China. Antonie van Leeuwenhoek 113(1): 21–32. https://doi.org/10.1007/s10482-019-01312-5

Weichman MA, Janovy Jr J (2000) Parasite community structure in *Pimephales promelas* (pisces: Cyprinidae) from two converging streams. Journal of Parasitology 86(3): 654–656. https://doi.org/10.1645/0022-3395(2000)086[0654:PCSIPP]2.0.CO;2

Weir M, Rajić A, Dutil L, Cernicchiaro N, Uhland FC, Mercier B, Tuševljak N (2012) Zoonotic bacteria, antimicrobial use and antimicrobial resistance in ornamental fish: A systematic review of the existing research and survey of aquaculture-allied professionals. Epidemiology and Infection 140(2): 192–206. https://doi.org/10.1017/S0950268811001798

Wisenden BD, Martinez-Marquez JY, Gracia ES, McEwen DC (2012) High intensity and prevalence of two species of trematode metacercariae in the fathead minnow (*Pimephales promelas*) with no compromise of minnow anti-predator competence. Journal of Parasitology 98(4): 722–727. https://doi.org/10.1645/GE-2454.1

Wood LE, Guilder J, Brennan ML, Birland NJ, Taleti V, Stinton N, Taylor NG, Thrush MA (2022) Biosecurity and the ornamental fish trade: A stakeholder perspective in England. Journal of Fish Biology 100(2): 352–365. https://doi.org/10.1111/jfb.14928

Xu L, Wang T, Li F, Yang F (2016) Isolation and preliminary characterization of a new pathogenic iridovirus from redclaw crayfish *Cherax quadricarinatus.* Diseases of Aquatic Organisms 120(1): 17–26. https://doi.org/10.3354/dao03007

Yamauchi T, Shimizu M (2013) New host and distribution records for the freshwater fish ectoparasite *Argulus japonicus* (Crustacea: Branchiura: Argulidae). Comparative Parasitology 80(1): 136–137. https://doi.org/10.1654/4554.1

Yera H, Kuchta R, Brabec J, Peyron F, Dupouy-Camet J (2013) First identification of eggs of the Asian fish tapeworm *Bothriocephalus acheilognathi* (Cestoda: Bothriocephalidea) in human stool. Parasitology International 62(3): 268–271. https://doi.org/10.1016/j.parint.2013.02.001

Yu J, Koo BH, Kim DH, Kim DW, Park SW (2015) *Aeromonas sobria* infection in farmed mud loach (*Misgurnus mizolepis*) in Korea, a bacteriological survey. Majallah-i Tahqiqat-i Dampizishki-i Iran 16: 194–201.

Zanoni RG, Florio D, Fioravanti ML, Rossi M, Prearo M (2008) Occurrence of *Mycobacterium* spp. in ornamental fish in Italy. Journal of Fish Diseases 31(6): 433–441. https://doi.org/10.1111/j.1365-2761.2008.00924.x

Zhang Y, Zhang Y, Na L, Wang WT, Xu WW, Gao DZ, Liu ZX, Wang CR, Zhu XQ (2014) Prevalence of *Clonorchis sinensis* infection in freshwater fishes in northeastern China. Veterinary Parasitology 204(3–4): 209–213. https://doi.org/10.1016/j.vetpar.2014.05.007

Zhao Y, Tang F (2007) Trichodinid ectoparasites (Ciliophora: Peritricha) from *Misgurnus anguillicaudatus* (Cantor) and *Anodonta woodiana* (Lea) in China, with descriptions of two new species of Trichodina Ehrenberg, 1838. Systematic Parasitology 67(1): 65–72. https://doi.org/10.1007/s11230-006-9070-6

Zięba G, Copp G, Davies G, Stebbing P, Wesley K, Britton R (2010) Recent releases and dispersal of non-native fishes in England and Wales, with emphasis on sunbleak *Leucaspius delineatus* (Heckel, 1843). Aquatic Invasions 5(2): 155–161. https://doi.org/10.3391/ai.2010.5.2.04

Zizka A, Silvestro D, Andermann T, Azevedo J, Duarte Ritter C, Edler D, Farooq H, Herdean A, Ariza M, Scharn R, Svantesson S, Wengström N, Zizka V, Antonelli A (2019) CoordinateCleaner: Standardized cleaning of occurrence records from biological collection databases. Methods in Ecology and Evolution 10(5): 744–751. https://doi.org/10.1111/2041-210X.13152

Supplementary material 1

Threats to UK freshwaters under climate change: Commonly traded aquatic ornamental species and their potential pathogens and parasites

Authors: Guilder J, Copp GH, Thrush M, Stinton N, Murphy D, Murray J, Tidbury HJ

Data type: Lists, Tables and Maps

Explanation note: This supplementary file provides a list of all websites used in the Google search, a table of all fish and invertebrate species observed as being sold in the UK and criteria for further analysis and a table of results for the pathogen and parasite screen based on laboratory studies (and the reference list for this table).

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RESEARCH ARTICLE

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Risk of invasiveness of non-native fishes in the South Caucasus biodiversity and geopolitical hotspot

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Abstract

Aquatic invasions are one of the major threats for freshwater ecosystems. However, in developing countries, knowledge of biological invasions, essential for the implementation of appropriate legislation, is often limited if not entirely lacking. In this regard, the identification of potentially invasive non-native species by risk screening, followed by a full risk assessment of the species ranked as higher risk, enables decision-makers to be informed about the extent of the threats posed to the recipient (risk assessment) area. In this study, 32 non-native extant and horizon fish species were screened for their risk of invasiveness under current and predicted climate conditions for the South Caucasus – a biodiversity and geopolitical hotspot that includes the countries of Armenia, Azerbaijan and Georgia. Overall, the number of very high-risk species increased from four (12.5%) under current climate conditions to 12 (37.5%) under predicted climate conditions. The highest-risk species under both conditions included the already established gibel carp *Carassius gibelio* and topmouth gudgeon *Pseudorasbora parva*, the locally translocated pikeperch *Sander lucioperca* and the horizon North African catfish *Clarias gariepinus*. Under predicted climate conditions, a very high risk of invasiveness was predicted also for the translocated three-spined stickleback *Gasterosteus aculeatus* and Eurasian perch *Perca fluviatilis*, for the already established eastern mosquitofish *Gambusia holbrooki*, ruffe *Gymnocephalus cernua*, sharpbelly *Hemiculter leucisculus* and Nile tilapia *Orechromis niloticus*, and for the horizon pumpkinseed *Lepomis gibbosus* and largemouth bass *Micropterus salmoides*. Future research on the non-native species in the South Caucasus should be conducted both country- and region-wide and should account not only for the high biodiversity, but also for the critical geopolitical situation affecting the study area.

Keywords
Aquatic invasions, AS-ISK, Black Sea, Caspian Sea, climate change, extant, horizon

Introduction

Biological invasions are a major threat to global biodiversity and pose a considerable challenge for human well-being (Mazza et al. 2014; Shackleton et al. 2018). Protecting biodiversity and maintaining ecosystem function involves the allocation of considerable financial resources due to the growing trend of invasion events and the resulting rate of pressure exerted by invasive non-native species (Seebens et al. 2017; Diagne et al. 2020). Additionally, the management and eradication of invasive non-native species once they become established are far more demanding endeavours in terms of costs and related challenges than prevention and early detection (Simberloff et al. 2013). The latter can be achieved through the establishment of country-based regulations to control species translocations and introductions within/amongst countries, supported by actions for eradication (Simberloff et al. 2013). Yet, concerted efforts to mitigate the non-native species invasion process and promote management actions pose overall a challenge at the global scale (Genovesi et al. 2013; Tittensor et al. 2014; CBD 2018).

One of the main reasons hindering effective national and cross-national strategic plans against invasive non-native species is the absence of quality biological data for several countries (Latombe et al. 2017). This is especially true of most developing countries that lack an exhaustive list of invasive non-native species, do not have monitoring capacities and/or have not yet adopted strategies for dealing with such species, which are usually identified long after their entry and establishment in the recipient area (Early et al. 2016). A good example is the South Caucasus biodiversity hotspot, which comprises the countries of Armenia, Azerbaijan and Georgia (Mittermeier et al. 2004). Despite preliminary efforts, there are currently no effective nation-wide initiatives in these countries to provide an inventory of non-native species, nor are there any related monitoring programmes in place. In fact, only incidental academic studies have so far provided some authoritative, albeit in most cases still partial, inventories of non-native plants (Kikodze et al. 2010; Fayvush and Tamanyan 2014; Sharabidze et al. 2018; Abdiyeva 2019), molluscs (Mumladze et al. 2019), insects (Aleksidze et al. 2021) and fishes (Kuljanishvili et al. 2021b). Yet, none of the aforementioned inventories (except for fishes) is complete or up-to-date enough to be useful at the national or cross-national level. Moreover, the impacts of invasive non-native species on the biodiversity and ecosystems of the South Caucasus have not yet been evaluated even for a single species, despite some of the non-native species therein having already caused environmental and economic losses (Diagne et al. 2020).

The South Caucasus is widely recognised as a biodiversity hotspot characterised by a great diversity of landscapes and climate zones that shelter a highly diverse plant and animal biota. Freshwater biodiversity is the most understudied ecological aspect of the South Caucasus (Mumladze et al. 2020), whose watercourses are exploited for hydropower production (Japoshvili et al. 2021) and secondarily for drinking water uptake, fisheries, irrigation and recreational activities. However, the potential threats faced by the freshwater biodiversity and ecosystems of the South Caucasus have not yet been evaluated, hence remain overall poorly understood. Only in a recent study have the effects of existing and planned hydropower plants on the connectivity of fish communities in Georgia been investigated in some detail (Japoshvili et al. 2021). Additionally, in another recent study, an attempt has been made to summarise the diversity, distribution and introduction history of non-native fishes in the South Caucasus based on a literature review and social-media data (Kuljanishvili et al. 2021b).

To understand the potential risk of invasiveness posed by non-native fishes in the South Caucasus, the aims of the present study were to: (i) screen both extant and horizon species and (ii) discuss the resulting species-specific risk ranks of invasiveness also within the current geopolitical situation affecting the study area with a view to implementing future legislation. Notably, this study represents the first risk screening for the South Caucasus and Georgia in particular. It is anticipated that the outcomes of this study will provide for an important step forward in the understanding of the impacts and related risks of environmental/economic losses caused by invasive non-native fishes in this biodiversity and geopolitical hotspot.

Methods

Risk assessment area

The South Caucasus (hereafter, also the 'risk assessment area') is located south of the Great Caucasus mountain range and stretches across the Black and Caspian seas with 80% of its area belonging to the Kura-Aras drainage basin (Caspian Sea Basin) shared with Turkey and Iran and the remaining 20% (western part) to the Black Sea Basin (Fig. 1). The South Caucasus is politically subdivided into the three independent countries of Armenia, Azerbaijan and Georgia. However, the South Caucasus is also an individually distinct geographic unit bordered by the Great Caucasus mountain range in the north (an impassable barrier for most animal species) and the Black and Caspian seas in the west and east, respectively. Whereas the southern border of the South Caucasus, albeit less distinctively identifiable, coincides with the two large rivers Kura-Aras and Chorokh, which originate from the Anatolian Plateau in Turkey. The Kura-Aras River is the longest watercourse and flows into the Caspian Sea, with most of the eastern South Caucasian rivers draining into this river's basin, except for a few short watercourses in the extreme north-east part of Azerbaijan. Unlike Armenia and Azerbaijan, in western Georgia, several small- to medium-sized rivers drain into the

Figure 1. Map of the South Caucasus (Armenia, Azerbaijan, Georgia), representing the risk assessment area, and neighbouring countries.

Black Sea and, at higher altitudes, there are several isolated mountainous lakes with their own independent basins. Amongst these, Lake Sevan in Armenia is the largest and harbours endemic taxa.

The climate of the South Caucasus is continental-mesophilic with strong local variation due to its complex topography. According to the updated Köppen-Geiger climate map (Beck et al. 2018), the warm-temperate climate types without dry season *Cfa* and *Cfb* (with warm and hot summer, respectively) are predominant in the westernmost part of the risk assessment area, which stretches along the entire eastern Black Sea coast and extends through to the Cholchis lowland. In this area, precipitation can be as high as 4000 mm annually (Adjara Region, south-western Georgia). Going further east, at higher altitudes (700–2000 m a.s.l.) the climate changes from cold with no dry season *Dfa* (but with hot summer) to *Dfb* (with warm summer) and *Dfc* (cold summer). Further east and up to the Caspian Sea, the climate becomes drier and corresponds predominantly to the *Bsk* (cold, arid steppe) and *Csa* (temperate dry and hot summer) types, with areas of cold semi-arid climate *Bwk*. In the eastern part, there is also a climate 'island' along the foothills of the southern Great Caucasus corresponding to the *Cfa* type. At the higher altitudes (above 2500 m) of the northern Great Caucasus and southern Lesser Caucasus, the climate changes sharply from cold with no dry season and cold summer (*Dfc*) to polar (*ET*).

Currently, there are 121 freshwater and anadromous fish species known from the South Caucasus (Kuljanishvili et al. 2021b) of which nearly 30% are endemic. The Kura-Aras River is the richest with at least 16 endemic species alone, though some watercourses of the Black Sea Basin also are species-rich, for example, the River Rioni, which is the last spawning ground for at least four sturgeon species (Beridze et al. 2022). Amongst the 121 fish species, ten are currently considered established non-native, whereas an additional five species are intra-regionally translocated (Kuljanishvili et al. 2021b). However, the conservation status of these species is largely unknown and, at the time of writing, there are only 16 species listed in the IUCN as globally threatened under various conservation statuses, with the remaining species still waiting for a comprehensive assessment.

Species selection

In total, 32 freshwater fish taxa (hereafter, for simplicity 'species') were selected for risk screening in the South Caucasus (Table 1). Notably, the marine/brackish water fishes that are frequently entering the lower reaches of rivers in the risk assessment area (i.e. golden grey mullet *Chelon auratus*, leaping mullet *Chelon saliens* and flathhead grey mullet *Mugil cephalus*) or that can survive in isolated freshwater bodies (i.e. black-striped pipefish *Syngnathus abaster*) (Elanidze 1956, 1983; Kuljanishvili et al. 2021c) will also be regarded in this study as 'freshwater species'. The criteria for species selection were as follows (Table 1):

1. Translocated species (n = 8);
2. Non-native species already present in the risk assessment area (n = 14);
3. Non-native 'horizon' species established in neighbouring countries or countries of similar climate to the risk assessment area (n = 5);
4. Non-native species recorded in the risk assessment area, but in the wild (n = 5).

Selection of species based on the first three criteria was according to the most recent non-native species list published by Kuljanishvili et al. (2021b), whereas selection based on the latter criterion relied on literature resources.

Risk screening

Risk screening was undertaken using the Aquatic Species Invasiveness Screening Kit (AS-ISK: Copp et al. 2016b, 2021), which is available for free download at www.cefas.co.uk/nns/tools. This taxon-generic decision-support tool consists of 55 questions: the first 49 questions comprise the Basic Risk Assessment (BRA) and address the biogeography/invasion history and biology/ecology of the species under screening; the last six questions comprise the Climate Change Assessment (CCA) and require the assessor to predict how future predicted climatic conditions are likely to affect the BRA with respect to risks of introduction, establishment, dispersal and impact. For the purposes of the CCA component of the screening protocol, an increase in temperature on average by 2 °C relative to current conditions is predicted for the SCR (Hansen et al. 2010; Beck et al. 2018). Screenings were undertaken on all species by three independent assessors, i.e. the authors BJ, GE and TK (combination 3IA: Vilizzi et al. 2022).

To achieve a valid screening, the assessor must provide for each question a response, a level of confidence for the response (see below) and a justification based on

Table 1. Freshwater fish taxa (for simplicity, 'species') screened for their potential risk of invasiveness in the South Caucasus – the risk assessment area. For each species, the following information is provided: criterion (Crit.) for selection (1 = translocated species; 2 = non-native species already present in the risk assessment area; 3 = non-native 'horizon' species established in neighbouring countries or countries of similar climate to the risk assessment area; 4 = non-native species recorded in the risk assessment area, but in the wild); a priori categorisation outcome into Non-invasive or Invasive. For the a priori categorisation, the results of the related protocol (after Vilizzi et al. 2022) are indicated: (i) FishBase (www.fishbase.org); (ii) Centre for Agriculture and Bioscience International Invasive Species Compendium (CABI: www.cabi.org/ISC) and Global Invasive Species Database (GISD: www.iucngisd.org); (iii) Invasive and Exotic Species of North America list (IESNA: www.invasive.org); (iv) Google Scholar literature search. N = no impact/threat; Y = impact/threat; '–' = absent; n.e. = not evaluated (but present in database); n.a. = not applicable.

Species name	Common name	A priori categorisation						
		Crit.	FishBase	CABI	GISD	IESNA	Google Scholar	Outcome
Ameiurus melas	black bullhead	3	Y	Y	–	–	n.a.	Invasive
Anguilla anguilla	European eel	4	N	Y	–	–	n.a.	Invasive
Carassius gibelio	gibel carp	2	Y	Y	–	–	n.a.	Invasive
Chelon auratus	golden grey mullet	1	N	–	–	–	N	Non-invasive
Chelon saliens	leaping mullet	1	N	–	–	–	N	Non-invasive
Clarias gariepinus	North African catfish	3	Y	Y	Y	–	n.a.	Invasive
Coregonus albula	vendace	2	N	Y	–	–	n.a.	Invasive
Coregonus sp.*	–	2	N	–	–	–	N	Non-invasive
Ctenopharyngodon idella	grass carp	2	Y	Y	Y	Y	n.a.	Invasive
Gambusia holbrooki	eastern mosquitofish	2	Y	Y	Y	–	n.a.	Invasive
Gasterosteus aculeatus	three-spined stickleback	1	N	–	–	–	N	Non-invasive
Gobio artvinicus	Artvin gudgeon	1	N	–	–	–	N	Non-invasive
Gymnocephalus cernua	ruffe	2	–	Y	–	–	n.a.	Invasive
Hemiculter leucisculus	sharpbelly	2	Y	N	–	–	n.a.	Invasive
Hypophthalmichthys molitrix	silver carp	2	Y	Y	Y	Y	n.a.	Invasive
Hypophthalmichthys nobilis	bighead carp	2	Y	Y	Y	Y	n.a.	Invasive
Ictalurus punctatus	channel catfish	4	Y	Y	–	–	n.a.	Invasive
Lepomis gibbosus	pumpkinseed	3	Y	N	–	–	n.a.	Invasive
Micropterus salmoides	largemouth bass	3	Y	Y	Y	–	n.a.	Invasive
Mugil cephalus	flathead grey mullet	4	N	–	–	–	N	Non-invasive
Mylopharyngodon piceus	black carp	4	Y	Y	–	Y	n.a.	Invasive
Oncorhynchus kisutch	coho salmon	4	N	–	–	Y	n.a.	Invasive
Oncorhynchus mykiss	rainbow trout	2	Y	Y	Y	Y	n.a.	Invasive
Oreochromis niloticus	Nile tilapia	2	Y	Y	Y	Y	n.a.	Invasive
Perca fluviatilis	Eurasian perch	1	Y	Y	Y	–	n.a.	Invasive
Pseudorasbora parva	topmouth gudgeon	2	Y	Y	–	–	n.a.	Invasive
Rhinogobius lindbergi	Lin's goby	2	N	–	–	–	N	Non-invasive
Salmo ischchan	Sevan trout	1	–	–	–	–	n.a.	Non-invasive
Salmo trutta	brown trout	2	Y	Y	Y	Y	n.a.	Invasive
Salvelinus fontinalis	brook trout	3	Y	Y	Y	–	n.a.	Invasive
Sander lucioperca	pikeperch	1	Y	Y	–	–	n.a.	Invasive
Syngnathus abaster	black-striped pipefish	1	N	N	–	–	N	Non-invasive

* Reference species for the a priori categorisation: European whitefish *Coregonus lavaretus*.

literature sources. The outcomes are a BRA score and a (composite) BRA+CCA score, which is obtained after adding or subtracting up to 12 points to the BRA score or leaving it unchanged in case of a CCA score equal to 0. Scores < 1 suggest that the species poses a 'low risk' of becoming invasive in the risk assessment area, whereas scores

≥ 1 indicate a 'medium risk' or a 'high risk'. The threshold (Thr) value to distinguish between medium-risk (BRA and BRA+CCA score < Thr) and high-risk (BRA and BRA+CCA score ≥ Thr) species for the risk assessment area is obtained by 'calibration' based on the Receiver Operating Characteristic (ROC) curve analysis (see Vilizzi et al. 2022). A measure of the accuracy of the calibration analysis is the area under the curve (AUC) whose values are interpreted as: 0.7 ≤ AUC < 0.8 = acceptable discriminatory power, 0.8 ≤ AUC < 0.9 = excellent, 0.9 ≤ AUC = outstanding (Hosmer et al. 2013). For the species ranked as high risk, a distinction was made in this study of the 'very high-risk' species, based on an ad hoc threshold weighted according to the range of high-risk scores obtained for the BRA and BRA+CCA. Identification of the (very) high-risk species is useful to prioritise allocation of resources in view of a full risk assessment (Copp et al. 2016a). This examines in detail the risks of: (i) introduction (entry); (ii) establishment (of one or more self-sustaining populations); (iii) dispersal (more widely within the risk assessment area, i.e. so-called secondary spread or introductions); and (iv) impacts (to native biodiversity, ecosystem function and services, and the introduction and transmission of diseases).

For the ROC curve analysis to be implemented, the species selected for screening must be categorised a priori as 'non-invasive' or 'invasive' using literature sources. The a priori categorisation of the species was implemented as per Vilizzi et al. (2022) (Table 1). Confidence levels in the responses to questions in the AS-ISK are ranked using a 1–4 scale and based on the confidence level allocated to each response, a confidence factor (CF) is obtained that ranges from a minimum of 0.25 (i.e. all 55 questions with confidence level equal to 1) to a maximum of 1 (i.e. all 55 questions with confidence level equal to 4). Based on all 55 Qs of the AS-ISK questionnaire, the 49 Qs comprising the BRA and the six Qs comprising the CCA, the CF_{Total}, CF_{BRA} and CF_{CCA} are respectively computed (Vilizzi et al. 2022).

Implementation of the ROC curve analysis followed the protocol described in Vilizzi et al. (2022), with the true/false positive/negative outcome distinction not applied to the medium-risk species, as they can be either included or not into a full (comprehensive) risk assessment depending on priority and/or availability of financial resources. The ROC curve fitting was in two steps. Firstly, separate ROC curves were generated for each of the three independent assessors and differences amongst the resulting three AUCs were statistically tested (Mann-Whitney *U*-statistic, $\alpha = 0.05$; applet StAR available at http://melolab.org/star/home.php: Vergara et al. 2008). As differences between assessor-specific AUCs were not found, in the second step, a single ROC curve was generated, based on the average species-specific BRA scores of the three assessors. Following ROC analysis, the best threshold value that maximises the true positive rate and minimises the false positive rate was determined using Youden's *J* statistic, whereas the 'default' threshold of 1 was set to distinguish between low-risk and medium-risk species. Fitting of the ROC curve was with package pROC (Robin et al. 2011) for R x64 v.4.0.5 (R Core Team 2021) using 2000 bootstrap replicates for the confidence intervals of specificities, which were computed along with the entire range of sensitivity points (i.e. 0 to 1, at 0.1 intervals). Differences in CF between components (i.e. BRA and BRA+CCA) were tested with permutational ANOVA. Analysis was implemented in PERMANOVA+ for PRIMER v.7, with normalisation of the data and using a Bray-Curtis dissimilarity measure, 9999 unrestricted permutations of the raw data and with statistical effects evaluated at α = 0.05.

Results

There were no differences between the AUCs resulting from the three assessor-specific ROC curves (BJ vs. GE: *P* = 0.912; BJ vs. TK: *P* = 0.090; GE vs. TK: *P* = 0.287): this justified computation of one ROC curve based on the mean BRA scores for the screened species. Accordingly, the ROC curve resulted in an AUC of 0.8213 (0.6310–1.000 95% CI), which indicated that the risk screening was able to distinguish with excellent discriminatory power between invasive and non-invasive fish species for the risk assessment area. Youden's *J* provided the threshold of 18, which was used to calibrate the risk outcomes to distinguish between medium-risk and high-risk species. The AS-ISK report for the 32 screened species is provided as Suppl. material 1.

Based on the BRA outcome scores (Table 2, Fig. 2A):

- 21 (65.6%) species were ranked as high risk and eleven (34.4%) as medium risk;
- Amongst the nine species categorised a priori as non-invasive, one was a false positive (three-spined stickleback *Gasterosteus aculeatus*);
- Amongst the 23 species categorised a priori as invasive, 20 were true positives;
- Of the 11 medium-risk species, eight were a priori non-invasive and three invasive.
- Based on the BRA+CCA outcome scores, hence after accounting for climate change predictions (Table 2, Fig. 2B):
- 23 (71.9%) species were ranked as high risk, eight (25.0%) as medium risk and one (3.1%) as low risk (*Coregonus* sp.);
- Amongst the a priori non-invasive species, four were false positives (*Chelon auratus*, *Gasterosteus aculeatus*, Lin's goby *Rhinogobius lindbergi*, *Syngnathus abaster*) and one a true negative (*Coregonus* sp.);
- Amongst the a priori invasive species, 19 were true positives;
- Of the nine medium-risk species, four were a priori non-invasive and four invasive.

The highest-scoring ('top invasive') species (based on an ad hoc 'very high risk' threshold = 40) were gibel carp *Carassius gibelio*, North African catfish *Clarias gariepinus*, topmouth gudgeon *Pseudorasbora parva* and pikeperch *Sander lucioperca* for both the BRA and BRA+CCA, and eastern mosquitofish *Gambusia holbrooki*, three-spined stickleback *Gasterosteus aculeatus*, ruffe *Gymnocephalus cernua*, sharpbelly *Hemiculter leucisculus*, pumpkinseed *Lepomis gibbosus*, largemouth bass *Micropterus salmoides*, Nile tilapia *Oreochromis niloticus* and Eurasian perch *Perca fluviatilis* for the BRA+CCA only. Overall, the number of very high-risk species increased from four (12.5%) under the BRA to 12 (37.5%) under the BRA+CCA (Figs 2A and B). The CCA resulted in an increase in the BRA score (cf. CCA) for 26 species and in a decrease for the remaining six species (Table 2). Across the three assessors: differences in BRA scores ranged from

Table 2. Risk outcomes for the freshwater fish species screened with the Aquatic Species Invasiveness Screening Kit (AS-ISK) for the South Caucasus. For each species, the following information is provided: a priori categorisation for invasiveness (N = non-invasive; Y = invasive: see Table 1); min, max and mean Basic Risk Assessment (BRA) and BRA + Climate Change Assessment (BRA+CCA) scores with corresponding risk ranks (L = Low; M = Medium; H = High; VH = Very high, based on an ad hoc threshold = 40) and classifications (Class: FP = false positive; TN = true negative; TP = true positive; '–' = not applicable as medium risk: see text for details); difference (Delta) between mean BRA+CCA and BRA scores. Risk ranks are based on a threshold of 18 and computed as: L, with score within the interval [−20, 1[, M [1, 18[, H [18, 40[, VH [40, 68] for the BRA; L [−32, 1[; M [1, 18[, H [18, 40[, VH [40, 80] for the BRA+CCA (note the reverse bracket notation indicating in all cases an open interval).

Species name	A priori	BRA					BRA+CCA								
		Score					Score						CF		
		Min	Max	Mean	Rank	Class	Min	Max	Mean	Rank	Class	Delta	Total	BRA	CCA
Ameiurus melas	Y	26.5	35.0	31.5	H	TP	24.5	47.0	36.8	H	TP	5.3	0.70	0.71	0.61
Anguilla anguilla	Y	1.0	15.0	9.3	M	–	−5.0	17.0	8.0	M	–	−1.3	0.72	0.72	0.71
Carassius gibelio	Y	36.0	52.0	44.0	VH	TP	48.0	64.0	55.3	VH	TP	11.3	0.74	0.75	0.67
Chelon auratus	N	14.0	25.0	17.7	M	–	16.0	20.0	18.3	H	FP	0.7	0.70	0.72	0.46
Chelon saliens	N	13.0	23.0	16.7	M	–	15.0	20.0	17.3	M	–	0.7	0.65	0.68	0.42
Clarias gariepinus	Y	38.0	45.0	40.3	VH	TP	46.0	55.0	49.7	VH	TP	9.3	0.66	0.68	0.51
Coregonus albula	Y	5.0	19.5	11.2	M	–	−3.0	7.5	1.2	M	–	−10.0	0.71	0.72	0.63
Coregonus sp.	N	1.0	18.0	9.0	M	–	−7.0	6.0	−0.3	L	TN	−9.3	0.70	0.71	0.61
Ctenopharyngodon idella	Y	18.0	23.5	20.7	H	TP	14.5	31.5	24.7	H	TP	4.0	0.69	0.72	0.49
Gambusia holbrooki	Y	31.5	38.0	34.5	H	TP	37.5	48.0	43.2	VH	TP	8.7	0.70	0.72	0.51
Gasterosteus aculeatus	N	37.0	38.0	37.7	H	FP	38.0	44.0	41.0	VH	FP	3.3	0.67	0.69	0.50
Gobio artvinicus	N	5.0	14.0	8.7	M	–	7.0	15.0	12.0	M	–	3.3	0.59	0.61	0.44
Gymnocephalus cernua	Y	34.0	46.0	39.3	H	TP	46.0	58.0	50.7	VH	TP	11.3	0.63	0.65	0.50
Hemiculter leucisculus	Y	32.0	35.0	33.8	H	TP	42.0	45.0	43.8	VH	TP	10.0	0.71	0.73	0.58
Hypophthalmichthys molitrix	Y	20.5	24.0	22.8	H	TP	18.5	34.0	28.8	H	TP	6.0	0.65	0.67	0.51
Hypophthalmichthys nobilis	Y	25.5	28.0	26.8	H	TP	19.5	38.0	30.8	H	TP	4.0	0.67	0.68	0.61
Ictalurus punctatus	Y	26.0	33.0	29.0	H	TP	32.0	45.0	39.0	H	TP	10.0	0.64	0.66	0.47
Lepomis gibbosus	Y	25.5	36.0	29.8	H	TP	37.5	46.0	40.5	VH	TP	10.7	0.71	0.72	0.63
Micropterus salmoides	Y	22.0	38.5	31.2	H	TP	34.0	50.5	41.2	VH	TP	10.0	0.70	0.72	0.56
Mugil cephalus	N	6.0	22.0	11.3	M	–	12.0	18.0	14.7	M	–	3.3	0.63	0.66	0.42
Mylopharyngodon piceus	Y	20.0	24.0	22.0	H	TP	28.0	34.0	32.0	H	TP	10.0	0.66	0.69	0.44
Oncorhynchus kisutch	Y	4.0	15.0	11.2	M	–	8.0	17.0	11.8	M	–	0.7	0.64	0.66	0.50
Oncorhynchus mykiss	Y	15.0	26.5	20.2	H	TP	15.0	18.5	16.8	M	–	−3.3	0.63	0.66	0.35
Oreochromis niloticus	Y	24.0	38.0	32.7	H	TP	34.0	48.0	42.0	VH	TP	9.3	0.65	0.68	0.49
Perca fluviatilis	Y	17.0	51.0	32.0	H	TP	23.0	63.0	41.3	VH	TP	9.3	0.66	0.68	0.50
Pseudorasbora parva	Y	32.0	47.0	40.0	VH	TP	44.0	57.0	49.3	VH	TP	9.3	0.77	0.78	0.71
Rhinogobius lindbergi	N	16.0	17.5	16.5	M	–	26.0	28.0	27.2	H	FP	10.7	0.54	0.53	0.63
Salmo ischchan	N	5.0	25.0	16.7	M	–	−7.0	20.0	10.0	M	–	−6.7	0.64	0.65	0.58
Salmo trutta	Y	34.0	39.0	36.0	H	TP	31.0	40.0	36.7	H	TP	0.7	0.63	0.66	0.46
Salvelinus fontinalis	Y	17.0	33.0	23.7	H	TP	17.0	29.0	23.0	H	TP	−0.7	0.70	0.75	0.31
Sander lucioperca	Y	30.0	50.0	43.0	VH	TP	38.0	59.0	46.3	VH	TP	3.3	0.69	0.70	0.58
Syngnathus abaster	N	9.0	27.0	16.3	M	–	5.0	37.0	20.3	H	FP	4.0	0.70	0.72	0.50

0 to 34, with a mean of 11.8, a median of 11.0 and 5% and 95% CIs of 5.1 and 20.2, respectively (Fig. 3A); differences in BRA+CCA scores ranged from 4 to 35, with a mean of 16.1, a median of 13.5 and 5% and 95% CIs of 5.8 and 32.0, respectively (Fig. 3B).

In terms of confidence in responses, the mean CL_{Total} was 2.69 ± 0.03 SE, the mean CL_{BRA} 2.76 ± 0.03 SE and the mean CL_{CCA} 2.11 ± 0.07 SE (hence, indicating

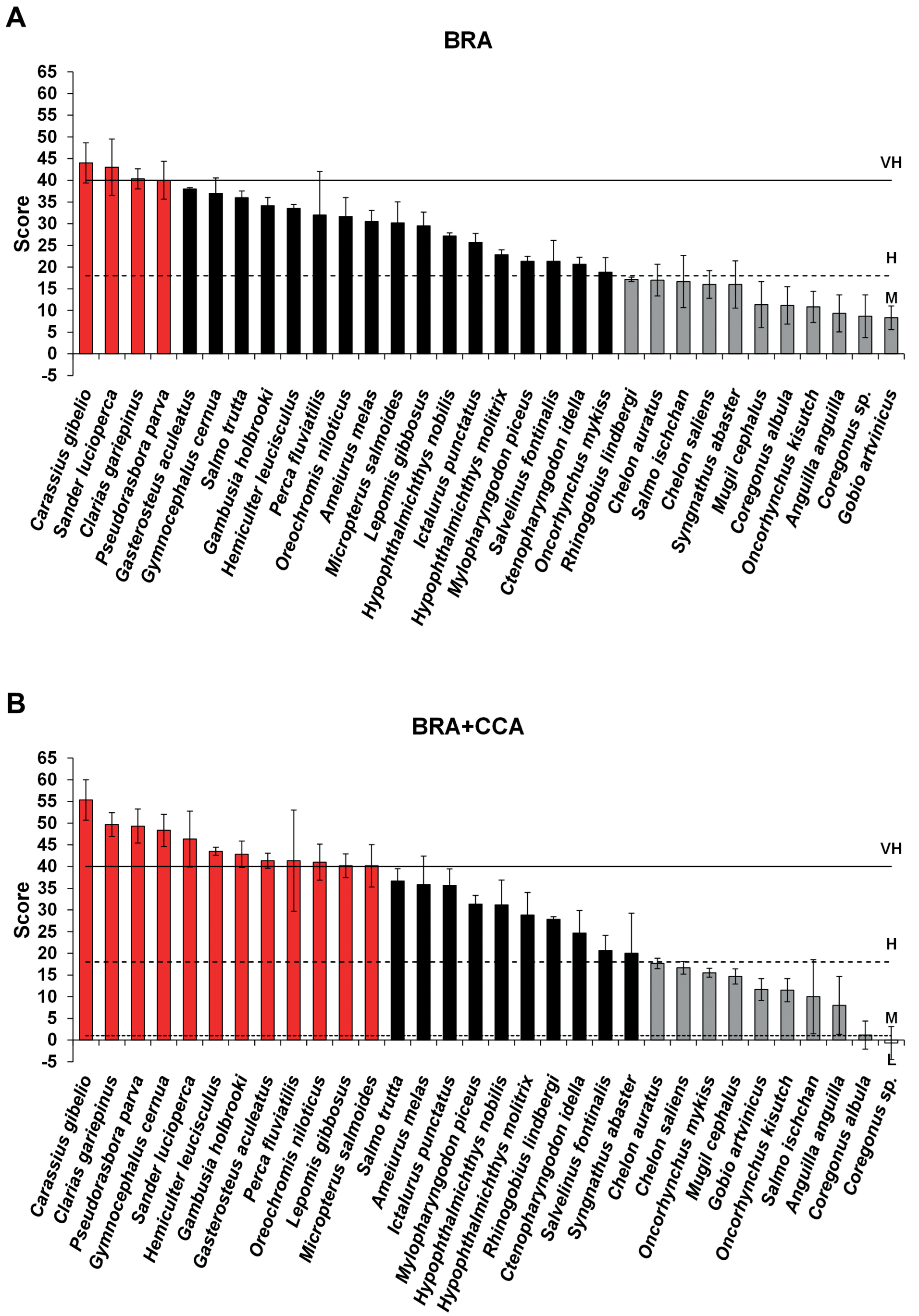

Figure 2. Aquatic Species Invasiveness Screening Kit (AS-ISK) mean outcome scores (± SE) for the species screened for the South Caucasus: **A** Basic Risk Assessment (BRA) scores **B** BRA+CCA (Climate Change Assessment) scores. Red bars = very high-risk species; Black bars = high-risk species; Grey bars = medium-risk species; White bars = low-risk (L) species. Solid line = very high-risk (VH) threshold; Hatched line = high-risk (H) threshold; Dotted line = medium-risk (M) threshold (thresholds as per Table 2).

a medium confidence level). The mean CF_{Total} was 0.673 ± 0.008 SE, the mean CF_{BRA} 0.691 ± 0.008 SE and the mean CF_{CCA} 0.527 ± 0.017 SE. Statistically, the CL_{BRA} was higher than the CL_{CCA} ($F^{\#}_{1,62}$ = 75.44, P < 0.001; # = permutational value).

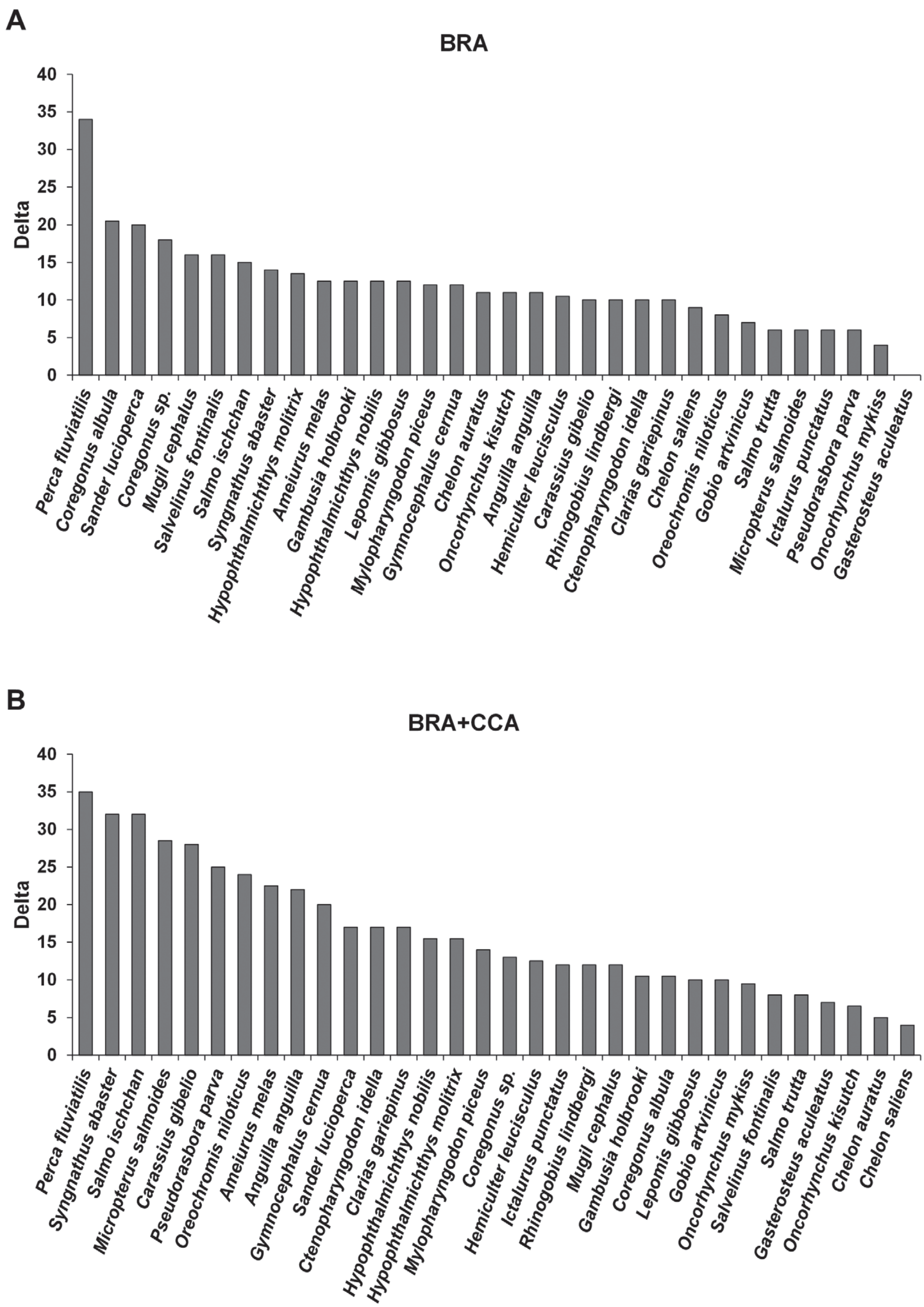

Figure 3. a Between-assessor differences in the BRA scores for the species screened for the South Caucasus **b** same for the BRA+CCA scores. See also Table 2.

Discussion

Risk outcomes

The present study, which is the first to conduct a risk screening for the South Caucasus, was able to identify with excellent discriminatory power the level of risk of invasiveness of the non-native fish species under evaluation. The calibrated threshold value (Thr = 18) in this study can, therefore, be used for future screening of additional non-native fish species in the risk assessment area, as required. Further, this threshold could be refined subject to availability of new biological data on the species screened in this study and/or additional species that may be identified as horizon or recorded in the risk assessment area by future surveys and/or based on more up-to-date climate change scenarios – this is in line with risk analysis as a dynamic, 'work-in-progress' applied field of science (see Vilizzi et al. 2021). Finally, the replication of screenings by three independent assessors in this study to account for the inherent potential bias with expert-based evaluations and the resulting lack of differences in assessor-specific AUCs has further strengthened the reliability of the species-specific risk ranks.

Amongst the screened species, 20 were ranked as carrying a high or very high risk of invasiveness under both current (BRA) and predicted climate conditions (BRA+CCA) (Table 2). These species included the false positive *Gasterosteus aculeatus*, whose high-risk ranking in the present study can, however, be justified with reference to the risk assessment area. In this respect, *G. aculeatus* is a widespread circum-arctic/temperate freshwater fish that naturally occurs in the River Danube mouth and on the Black Sea coast (Piria et al. 2018). This species has expanded its range to the Caspian Sea via the Volga-Don Channel and is now widely established along the Caspian Sea coast (eastern South Caucasus), where it actively enters the lower reaches of rivers (Ibrahimov and Mustafayev 2015). Due to its high tolerance of salinity and temperature and its reproductive and foraging characteristics (e.g. Roch et al. 2018; Candolin 2019), *G. aculeatus* can, therefore, be regarded as a high-risk species for the risk assessment area, despite its a priori non-invasive status elsewhere.

Overall, the a priori invasive species found to carry a high or very high risk of invasiveness (Table 2) may represent a threat to the native species and ecosystems of the South Caucasus. Some of these species are already established in the risk assessment area (Table 1) and, amongst these, *Carassius gibelio* and *Pseudorasbora parva* (both identified as very high risk) are already recognised as posing a serious threat (Kuljanishvili et al. 2021b). *Carassius gibelio* was introduced unintentionally (and probably several times) since the 1980s mainly as a contaminant of stockings of young-of-the-year common carp *Cyprinus carpio* in Lake Paliastomi and in the lakes of the Javakheti Plateau (Japoshvili et al. 2013; Kuljanishvili et al. 2021b). However, other species, including Chinese carps (i.e. bighead carp *Aristichthys nobilis*, *Ctenopharyngodon idella* and silver carp *Hypophthalmichthys molitrix*) and *Cyprinus carpio*, have also been intensively and mostly illegally introduced to other water bodies of the South Caucasus by local anglers for recreational purposes resulting in accidental introductions of *C. gibelio*

(Japoshvili et al. 2013; Kuljanishvili et al. 2021b). Concerning *P. parva*, this species is a typical hitchhiker without any economic value that is spreading on its own through the watercourses of the South Caucasus and is still being translocated as part of cyprinid farming practices (Gozlan et al. 2010; Kuljanishvili et al. 2021b). Overall, both species are currently the most widespread in the risk assessment area, where, in most cases, they form dense populations that dominate in abundance the local fish community (Shoniya et al. 2011; Japoshvili et al. 2013; Pipoyan and Arakelyan 2015; Kuljanishvili et al. 2021b).

A very high risk of invasiveness was also attributed to *Clarias gariepinus* and to locally translocated *Sander lucioperca*. *Clarias gariepinus* is found in neighbouring Turkey and is an invasive predator species that can easily spread once established (Ellender et al. 2015; Weyl et al. 2016). Detrimental effects of this species on the native fauna are, therefore, expected to occur in the South Caucasus as observed elsewhere (Kadye and Booth 2012; Ellender et al. 2015) and can be exacerbated by climate change (i.e. increase in temperatures) as revealed by the augmented BRA+CCA score for this species (Table 2). *Sander lucioperca* is a sought-after species with anglers that is actively translocated within the risk assessment area (Kuljanishvili et al. 2021b). This species has been reported to alter severely the invaded ecosystems in multiple ways including predation, hybridisation and disease transmission (Godard and Copp 2011). However, the species' impact on the native freshwater fauna of the risk assessment area remains unknown.

The threats posed by other established species ranked as carrying a high risk (BRA) or very high risk (BRA+CCA) of invasiveness, such as *Gambusia holbrooki*, *Gymnocephalus cernua* and *Hemiculter leucisculus* and by locally-translocated *Perca fluviatilis*, are still not clearly understood. *Gambusia holbrooki* was one of the first non-native species to be introduced in the South Caucasus for mitigation of the malaria disease (Barach 1941; Elanidze 1983). Since its introduction, this species has formed dense populations in most of the still water bodies of western Georgia and the eastern South Caucasus (Kuljanishvili et al. 2021b). The other two species *G. cernua* and *H. leucisculus* are cryptic invaders for which no invasion/establishment history exists. The former species is far more widespread in the risk assessment area than previously thought (G. Epitashvili, unpublished data), whereas the latter was first detected from the River Rioni in western Georgia in 2020 based on DNA barcoding (Epitashvili et al. 2020). *Perca fluviatilis* has been repeatedly introduced to water bodies of the South Caucasus for recreational angling, although no data are available on its introduction/establishment history, population dynamics or range of expansion (Kuljanishvili et al. 2021b). However, given its predatory lifestyle, this species can severely alter the native fish communities of the risk assessment area (review in Rowe et al. 2008).

Other species ranked as high (or very high) risk included those that are regularly stocked in the risk assessment area, but have not yet established self-sustaining populations, namely grass carp *Ctenopharyngodon idella*, *Hypophthalmichthys molitrix*, bighead carp *Hypophthalmichthys nobilis*, channel catfish *Ictalurus punctatus*, black carp *Mylopharyngodon piceus*, rainbow trout *Oncorhynchus mykiss*, Nile tilapia *Orechromis niloticus* and brown trout *Salmo trutta*. Amongst these species, *O. niloticus* is currently considered a hitchhiker in the South Caucasus, where no information on its deliberate farming is available. This species has been recorded only once in the wild (River Alazani, eastern Georgia), though no established population has been confirmed (Kuljanishvili et al. 2021a). Overall, all of the above species could pose a substantial threat to the local native ecosystems once established (e.g. Dibble and Kovalenko 2009; Martín-Torrijos et al. 2016; DeBoer et al. 2018; Faria et al. 2019), so their stocking should either be avoided altogether or strictly monitored.

The horizon species black bullhead *Ameiurus melas*, *Lepomis gibbosus*, *Micropterus salmoides* and brook trout *Salvelinus fontinalis* were also ranked as high (or very high) risk. However, neither of them has so far been recorded from the risk assessment area, although they are all well known to have expanded their range worldwide as a result of introductions for recreational and aquaculture purposes. Although a proper understanding of their impact on the invaded ecosystems is limited, these species are known to pose substantial threats to the native fish faunas (e.g. Cucherousset et al. 2008; Leunda et al. 2008; Drake 2009; Almeida et al. 2014; Copp et al. 2017).

Of the eleven species found to carry a medium risk of invasiveness (based on the BRA), vendace *Coregonus* sp., *Coregonus albula*, coho salmon *Oncorhynchus kisutch* and *Rhinogobius lindbergi* deserve some special consideration. Both *Coregonus* sp. (a putative hybrid known as *C. lavaretus sevanicus*: Dadikyan 1964, 1986) and *C. albula* have been regularly stocked in the mountainous lakes of Georgia (Javakheti Plateau) and in Lake Sevan, where they are thought to have established self-reproducing populations and are also highly valued commercially (Dadikyan 1964; Japoshvili 2012; Kuljanishvili et al. 2021b). In contrast, *O. kisutch* was released only once into the Caspian Sea in the 1980s and no further information on this species has since been available (Musayev et al. 2004). The small gobiid fish *R. lindbergi*, which is already widespread in the south-western Caspian Sea Basin (Sadeghi et al. 2019; Japoshvili et al. 2020), is not known to pose any risk to the native fauna, although this may be only a provisional expectation. This is because *R. lindbergi* resembles native gobies and occupies a similar ecological niche. Additionally, being a cryptic invader (e.g. Epitashvili et al. 2020), *R. lindbergi* may pose a threat to the native fauna; hence, it should be subject to future monitoring.

The remaining species carrying a medium risk of invasiveness are all native to the South Caucasus and translocated, except for *Mugil cephalus*. Four of these species, namely the mullets *Chelon auratus*, *Chelon saliens* and *Mugil cephalus*, as well as *Syngnathus abaster*, are primarily marine/brackish water species regularly occurring in estuaries or in the lower stretches of rivers. Mullets, which are economically valuable and naturally occur in the Black Sea, were introduced to the Caspian Sea in the early 20th century and, amongst them, *C. saliens* and *C. auratus* have established dense and abundant populations in river mouths (Bogutskaya et al. 2013). Risk screening for the latter two species has also been conducted for the neighbouring Anzali Wetland Complex (Caspian Sea Basin, Iran), with a similar medium-risk rank for *C. saliens*, but a low-risk rank for *C. auratus* (Moghaddas et al. 2021). This was mainly due to the fact that these species are known to reproduce in the Caspian Sea and to use associated estuaries and river mouths only temporarily for feeding (Coad 2017). As per *M. cephalus*, this species is currently not known to be established in the risk assessment area. However, given its high tolerance for water temperature and salinity, *M. cephalus* could reach the Caspian Sea Basin either on its own (i.e. via the Volga-Don Channel) or by translocation for aquaculture purposes (Abo-Taleb et al. 2021). The other eurihaline translocated species *S. abaster*, which is native to the Black and Caspian sea coasts (and with a threatened status in the Caspian Sea: Kolangi-Miandare et al. 2013), was introduced into Tbilisi Reservoir in the 1980s and has since resulted in the establishment of a dense population (Kuljanishvili et al. 2021b). Although not regarded as invasive, *S. abaster* is known for its ability to establish easily in freshwater habitats and affect zooplankton communities by selective feeding (Didenko et al. 2018).

The remaining translocated species ranked as medium risk included migratory European eel *Anguilla anguilla* and the resident species Artvin gudgeon *Gobio artvinicus* and *Salmo ischchan*. *Anguilla anguilla* was recorded from the Caspian Sea Basin in 1964 (Kuljanishvili et al. 2021b) and has been reported to enter various rivers of the risk assessment area, where it occurs, however, at low densities (Abdurakhmanov 1966; Ibrahimov and Mustafayev 2015; Pipoyan 2015). *Gobio arthvinicus* was unintentionally translocated from the Black Sea Basin to the River Kura, where it has most probably spread as a hitchhiker, though it is not used either for recreational or aquaculture purposes. This species is already established in the River Kura Basin (Kuljanishvili et al. 2021b) and its distribution has expanded to other watercourses. In contrast, *S. ischchan* was introduced to water bodies of Azerbaijan and Georgia from Lake Sevan and self-sustaining populations have been reported to cause negative impacts on native Caspian trout *Salmo caspius* (Elanidze 1983; Musayev et al. 2004; Yusifov et al. 2017; Kuljanishvili et al. 2021b).

Overall, under predicted climate change, 12 species in total were ranked as very high risk (Table 2, Fig. 1B). Of these species, eight are already established in the risk assessment area of which three are translocated (i.e. *Gasterosteus aculeatus*, *Perca fluviatilis*, *Sander lucioperca*) and six introduced (i.e. *Carassius gibelio*, *Gambusia holbrooki*, *Gymnocephalus cernua*, *Hemiculter leucisculus*, *Oreochromis niloticus*, *Pseudorasbora parva*) and three are horizon (i.e. *Clarias gariepinus*, *Lepomis gibbosus*, *Micropterus salmoides*). The current lack of legislation for non-native species in the South Caucasus, hence strategies for dealing with the impacts of current invasions, is therefore an issue of even higher concern given the predicted increase in number of very high risk species because of climate change (see below).

Recommendations for future research

In this study, the South Caucasus has been treated as a distinct biogeographic unit rather than a politically defined entity at the country level, hence in line with the preferred approach to the definition of a risk assessment area (Vilizzi et al. 2022). This is because the Kura-Aras Basin is shared amongst all the South Caucasus countries of Armenia, Azerbaijan and Georgia, making establishment of invasive non-native species in any of them likely to result in future potential ecological impacts for all three countries. In addition, the Kura-Aras Basin is also shared between Turkey and the South Caucasus and the same is true for the River Chorokhi between Turkey and western Georgia.

Consequently, any evaluation of the impact of potentially invasive non-native species and the establishment of regulations for their introduction and management should ideally be agreed upon and implemented across all of the South Caucasus countries (if not beyond: cf. Turkey), rather than independently for each of them. Unfortunately, the South Caucasus biodiversity hotspot is also a 'geopolitical hotspot' subject to permanent military tensions, which are likely to be exacerbated in recent times (Muradov 2022). For this reason, it is hardly possible to communicate and discuss the outcomes of environmental issues not only between countries (i.e. Armenia vs Azerbaijan), but also at the country level (e.g. 20% of the territory of Georgia is inaccessible due to Russian occupation), with the result that these long-unresolved political tensions are further aggravating the extent of non-native species management plans across the South Caucasus.

In the European Union, policies, legislation and management approaches have been developed to address the issue of non-native species, based on Regulation (EU) no. 1143/2014 of the European Parliament and of the Council on the prevention and management of the introduction and spread of invasive alien species (Piria et al. 2017, 2021). However, countries outside of the EU do not have an obligation to follow these rules and usually lack national legislation, which represents an additional problem for non-native species management (Piria et al. 2021). Whilst there is no region-wide agreement/management policy related to non-native aquatic species, there is also a major lack of relevant national-level legislation in the three South Caucasus countries (Kuljanishvili et al. 2021b). These all are parties to the Convention on Biological Diversity (CBD: https://www.cbd.int/) and Georgia is, in addition, a party to two other conventions, namely the Convention on the Protection of the Black Sea Against Pollution (http://www.blacksea-commission.org/) and the Convention for the Control and Management of Ships' Ballast Water and Sediments (https://www.imo.org/en). Within these Conventions (and particularly under the CBD), the South Caucasus countries are regularly developing National Biodiversity Strategy and Action Plans (https://www.cbd.int/nbsap/) in which the lack of data/infrastructure dealing with invasive non-native species is being emphasised and relevant targets (e.g. development of invasive non-native species lists, identification of introduction pathways, evaluation of impacts, implementation of legislative measures) are being proposed. However, none of the targets related to (freshwater) non-native species has so far been fulfilled (see, for example, the fifth national reports for Armenia, Azerbaijan and Georgia: https://www.cbd.int/).

Overall, to date, none of the South Caucasus countries has achieved a clear understanding of non-native species management within a national legislation plan (Kuljanishvili et al. 2021b), so that the import/farming of aquatic non-native species for recreational/aquaculture purposes is simply allowed under permission of governmental bodies. In fact, there is no legislative means for banning any particular aquatic non-native species (including highly invasive ones), checking for hitchhikers or restricting translocations. It is, therefore, strongly advocated that both country- and region-wide (i.e. South Caucasus) strategies, legislation acts and related actions for freshwater non-native species should be urgently developed. In this respect, it is recommended that such a strategy should adopt the following overarching conceptual goals:

1. Full risk assessment of any potentially invasive species should focus on those ranked as high risk (or very high risk, depending on availability of resources). Thus, whenever possible, a comprehensive risk screening, as achieved in this study, should be conducted and species-specific risk-rank outcomes presented to decision-makers. In this study, 12 very high-risk species in total (after accounting for climate change predictions) were identified that should be prioritised for follow-up full risk assessment.

2. Knowledge gap analysis and improvement of the legal basis for species introductions related to aquaculture/game fisheries and the pet trade. Ideally, this should be jointly agreed upon by the SCR countries to be fully effective.

3. Early detection and communication of freshwater non-native species is a process already under way, with researchers publishing results about new introductions of potentially invasive non-native species and citizen science platforms regularly receiving data from the general public on the identification of new non-native species. However, data and knowledge developing over time must be standardised in order to be rapidly communicated to stakeholders and decision-makers. In addition, adequate measures should be taken to enhance data collection from all potential sources. For instance, there is currently no information on non-native species available from local markets.

4. Continuous development of an in-depth monitoring scheme (including infrastructure for data collection based on fieldwork, barcoding/metabarcoding approaches, data management and presentation). This is a critical step to understand the history of non-native species colonisation and the accompanying processes related to community perception, including associated costs for damage/mitigation.

5. Prevention of introductions (cf. 'blacklists' of species). Since there is a large amount of data on freshwater non-native species worldwide, it would be straightforward to develop a list of potentially invasive non-native species for the South Caucasus (see Roy et al. 2019). In this regard, risk screening could help further refinement of the taxa list and via a similar exercise given in the present study. Control of the introduction of the most (if not all) threatening species could then be implemented by the local governments. As such practice already exists elsewhere (Essl et al. 2011; Gederaas et al. 2012; Poeta et al. 2017; Battisti et al. 2019), it should be discussed and adopted also within the countries of the South Caucasus.

6. Impact assessment research, including that related to already established invasive non-native species. Currently, there is no evaluation of the economic/environmental costs related to freshwater non-native species, nor for terrestrial ones. This makes it difficult to assess the effects of non-native species on local communities and to manage in an optimal way available resources to prevent/mitigate non-native species introductions. The results of this study can, therefore, be used to prioritise the list of fish species in the South Caucasus to be evaluated for impact assessment.

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References

Abdiyeva RT (2019) Invasive flora in the ecosystems of the Greater Caucasus (Azerbaijan part). Plant & Fungal Research 2: 15–22. https://doi.org/10.29228/plantfungalres.44

Abdurakhmanov UA (1966) Fauna of Azerbaijan: Fishes. Publishing house of the Azerbaijan Academy of Science, Baku, 223 pp. [In Russian]

Abo-Taleb HA, El-feky MM, Azab AM, Mabrouk MM, Elokaby MA, Ashour M, Mansour T, Abdelzaher OF, Abualnaja KM, Sallam AE (2021) Growth performance, feed utilization, gut integrity, and economic revenue of grey mullet, *Mugil cephalus*, fed an increasing level of dried zooplankton biomass meal as fishmeal substitutions. Fishes 6(3): e38. https://doi.org/10.3390/fishes6030038

Aleksidze G, Japaridze G, Kavtaradze G, Barjadze S (2021) Invasive alien species of Georgia. In: Pullaiah T, Ielmini MR (Eds) Invasive alien species: observations and issues from around the world. John Wiley & Sons Ltd., 88–123. https://doi.org/10.1002/9781119607045

Almeida D, Merino-Aguirre R, Vilizzi L, Copp GH (2014) Interspecific aggressive behaviour of invasive pumpkinseed *Lepomis gibbosus* in Iberian fresh waters. PLoS ONE 9(2): e88038. https://doi.org/10.1371/journal.pone.0088038

Barach G (1941) Freshwater fishes. In: Fauna of Georgia (Volume 1), Metsniereba, Tbilisi, 1–288. [In Russian]

Battisti C, Staffieri E, Poeta G, Sorace A, Luiselli L, Amori G (2019) Interactions between anthropogenic litter and birds: A global review with a ‘black-list’ of species. Marine Pollution Bulletin 138: 93–114. https://doi.org/10.1016/j.marpolbul.2018.11.017

Beck HE, Zimmermann NE, McVicar TR, Vergopolan N, Berg A, Wood EF (2018) Present and future Köppen-Geiger climate classification maps at 1-km resolution. Scientific Data 5(1): e180214. https://doi.org/10.1038/sdata.2018.214

Beridze T, Boscari E, Scheele F, Edisherashvili T, Anderson C, Congiu L (2022) Interspecific hybridization in natural sturgeon populations of the Eastern Black Sea: The consequence of drastic population decline? Conservation Genetics 23(1): 211–216. https://doi.org/10.1007/s10592-021-01413-7

Bogutskaya N, Kijashko P, Naseka AM, Orlova MI (2013) Identification keys for fish and invertebrates of the Caspian Sea. Vol. 1. Fish and molluscs. Tovarishestvo Naucnikh Izdanii KMK, Moscow, 1–443. [Russian]

Candolin U (2019) The threespine stickleback (*Gasterosteus aculeatus*) as a modifier of ecological disturbances. Evolutionary Ecology Research 20: 167–191. http://www.evolutionary-ecology.com/issues/v20/n02/eear3167.pdf

CBD (2018) Analysis of the contribution of targets established by parties and progress towards the Aichi biodiversity targets. CBD/SBI/2/2/Add.2. https://www.cbd.int/doc/c/e24a/347c/a8b84521f326b90a198b1601/sbi-02-02-add2-en.pdf

Coad BW (2017) Review of the freshwater mullets of Iran (Family Mugilidae). Iranian Journal of Ichthyology 4: 75–130. https://doi.org/10.7508/iji.2016.0

Copp GH, Russell IC, Peeler EJ, Gherardi F, Tricarico E, Macleod A, Cowx IG, Nunn AD, Occhipinti-Ambrogi A, Savini D, Mumford J, Britton JR (2016a) European Non-native Species in Aquaculture Risk Analysis Scheme - a summary of assessment protocols and decision support tools for use of alien species in aquaculture. Fisheries Management and Ecology 23(1): 1–11. https://doi.org/10.1111/fme.12074

Copp GH, Vilizzi L, Tidbury H, Stebbing PD, Tarkan AS, Moissec L, Goulletquer P (2016b) Development of a generic decision-support tool for identifying potentially invasive aquatic taxa: AS-ISK. Management of Biological Invasions 7(4): 343–350. https://doi.org/10.3391/mbi.2016.7.4.04

Copp GH, Britton JR, Guo Z, Edmonds-Brown VR, Pegg J, Vilizzi L, Davison PI (2017) Trophic consequences of non-native pumpkinseed *Lepomis gibbosus* for native pond fishes. Biological Invasions 19(1): 25–41. https://doi.org/10.1007/s10530-016-1261-8

Copp GH, Vilizzi L, Wei H, Li S, Piria M, Al-Faisal AJ, Almeida D, Atique U, Al-Wazzan Z, Bakiu R, Bašić T, Bui TD, Canning-Clode J, Castro N, Chaichana R, Çoker T, Dashinov D, Ekmekçi FG, Erős T, Ferincz Á, Ferreira T, Giannetto D, Gilles Jr AS, Głowacki Ł, Goulletquer P, Interesova E, Iqbal S, Jakubčinová K, Kanongdate K, Kim JE, Kopecký O, Kostov V, Koutsikos N, Kozic S, Kristan P, Kurita Y, Lee HG, Leuven RSEW, Lipinskaya T, Lukas J, Marchini A, González-Martínez AI, Masson L, Memedemin D, Moghaddas SD, Monteiro J, Mumladze L, Naddafi R, Năvodaru I, Olsson KH, Onikura N, Paganelli D, Pavia Jr RT, Perdikaris C, Pickholtz R, Pietraszewski D, Povž M, Preda C, Ristovska M, Rosíková K, Santos JM, Semenchenko V, Senanan W, Simonović P, Smeti E, Števove B, Švolíková K, Ta KAT, Tarkan AS, Top N, Tricarico E, Uzunova E, Vardakas L, Verreycken H, Zięba G, Mendoza R (2021) Speaking their language – Development of a multilingual decision-support tool for communicating invasive species risks to decision makers and stakeholders. Environmental Modelling & Software 135: e104900. https://doi.org/10.1016/j.envsoft.2020.104900

Cucherousset J, Aymes JC, Poulet N, Santoul F, Céréghino R (2008) Do native brown trout and non-native brook trout interact reproductively? Naturwissenschaften 95(7): 647–654. https://doi.org/10.1007/s00114-008-0370-3

Dadikyan MG (1964) Towards the results of introduction of Coregonids (*Coregonus lavaretus maraenoides* Poljakow, *C. lavaretus ludoga* Poljakow) in the Lake Sevan. Proceedings of the Academy of Science of Armenian SSR 17: 41–48. [In Russian]

Dadikyan MG (1986) Fishes of Armenia. National Academy of Science of Armenian SSR, Yerevan, 245 pp. [In Russian]

DeBoer JA, Anderson AM, Casper AF (2018) Multi-trophic response to invasive silver carp (*Hypophthalmichthys molitrix*) in a large floodplain river. Freshwater Biology 63(6): 597–611. https://doi.org/10.1111/fwb.13097

Diagne C, Leroy B, Gozlan RE, Vaissière AC, Assailly C, Nuninger L, Roiz D, Jourdain F, Jarić I, Courchamp F (2020) InvaCost, a public database of the economic costs of biological invasions worldwide. Scientific Data 7(1): 1–12. https://doi.org/10.1038/s41597-020-00586-z

Dibble ED, Kovalenko K (2009) Ecological impact of grass carp: A review of the available data. Journal of Aquatic Plant Management 47: 1–15.

Didenko A, Kruzhylina S, Gurbyk A (2018) Feeding patterns of the black-striped pipefish *Syngnathus abaster* in an invaded freshwater habitat. Environmental Biology of Fishes 101(6): 917–931. https://doi.org/10.1007/s10641-018-0747-x

Drake A (2009) *Micropterus salmoides*. In: Invasive Species Compendium. CAB International, Wallingford. https://www.cabi.org/isc/datasheet/74846

Early R, Bradley BA, Dukes JS, Lawler JJ, Olden JD, Blumenthal DM, Gonzalez P, Grosholz ED, Ibanez I, Miller LP, Sorte CJB, Tatem AJ (2016) Global threats from invasive alien species in the twenty-first century and national response capacities. Nature Communications 7(1): 1–9. https://doi.org/10.1038/ncomms12485

Elanidze R (1956) Ichtyofauna of the Rioni River. Proceedings of the Institute of Zoology 15: 111–168. [In Georgian]

Elanidze R (1983) Ichthyofauna of the rivers and lakes of Georgia. Metsniereba, Tbilisi, 320 pp. [In Russian]

Ellender BR, Woodford DJ, Weyl OL (2015) The invasibility of small headwater streams by an emerging invader, *Clarias gariepinus.* Biological Invasions 17(1): 57–61. https://doi.org/10.1007/s10530-014-0744-8

Epitashvili G, Geiger MF, Astrin JJ, Herder F, Japoshvili B, Mumladze L (2020) Towards retrieving the Promethean treasure: A first molecular assessment of the freshwater fish diversity of Georgia. Biodiversity Data Journal 8: e57862. https://doi.org/10.3897/BDJ.8.e57862

Essl F, Nehring S, Klingenstein F, Milasowszky N, Nowack C, Rabitsch W (2011) Review of risk assessment systems of IAS in Europe and introducing the German–Austrian Black List Information System (GABLIS). Journal for Nature Conservation 19(6): 339–350. https://doi.org/10.1016/j.jnc.2011.08.005

Faria L, Alexander ME, Vitule JR (2019) Assessing the impacts of the introduced channel catfish *Ictalurus punctatus* using the comparative functional response approach. Fisheries Management and Ecology 26(6): 570–577. https://doi.org/10.1111/fme.12353

Fayvush G, Tamanyan K (2014) Invasive and expanding plant species of Armenia. Institute of Botany of Armenian National Academy of Science, Yerevan, 272 pp. [In Russian]

Gederaas L, Loennechen Moen T, Skjelseth S, Larsen LK (2012) Alien species in Norway-with the Norwegian Black List 2012. The Norwegian Biodiversity Information Centre, Trondheim, 214 pp.

Genovesi P, Butchart SHM, McGeoch MA, Roy DB (2013) Monitoring trends in biological invasion, its impact and policy responses. In: Collen B, Pettorelli N, Baillie JEM, Durant SM (Eds) Biodiversity monitoring and conservation: bridging the gap between global commitment and local action. Wiley, Chichester, 138–158.

Godard M, Copp G (2011) *Sander lucioperca*. Invasive Species Compendium. CAB International, Wallingford. https://www.cabi.org/isc/datasheet/65338

Gozlan RE, Andreou D, Asaeda T, Beyer K, Bouhadad R, Burnard D, Caiola N, Cakisc P, Djikanovic V, Esmaeili HR, Falka I, Glicher D, Harka A, Jeney G, Kovac V, Musil J, Nocita A, Povz M, Poulet N, Virbickas T, Wolter C, Tarkan AS, Tricario E, Trichkova T, Verreycken H, Witkowski A, Zhang CG, Zweimueller I, Britton RJ (2010) Pan-continental invasion of *Pseudorasbora parva*: Towards a better understanding of freshwater fish invasions. Fish and Fisheries 11(4): 315–340. https://doi.org/10.1111/j.1467-2979.2010.00361.x

Hansen J, Ruedy R, Sato M, Lo K (2010) Global surface temperature change. Reviews of Geophysics 48(4): RG4004. https://doi.org/10.1029/2010RG000345

Hosmer Jr DW, Lemeshow S, Sturdivant RX (2013) Applied logistic regression. John Wiley & Sons, Hoboken, 511 pp. https://doi.org/10.1002/9781118548387

Ibrahimov SR, Mustafayev NJ (2015) Current status of Azerbaijan ichthyofauna. Proceedings of Azerbaijan Institute of Zoology 33: 58–68. [In Azeri]

Japoshvili B (2012) Long-term assessment of a vendace (*Coregonus albula* L.) stock in Lake Paravani, South Georgia. Advances in Limnology 63: 363–369. https://doi.org/10.1127/advlim/63/2012/363

Japoshvili B, Mumladze L, Küçük F (2013) Invasive *Carassius* carp in Georgia: Current state of knowledge and future perspectives. Current Zoology 59(6): 732–739. https://doi.org/10.1093/czoolo/59.6.732

Japoshvili B, Lipinskaya T, Gajduchenko H, Sinchuk A, Bikashvili A, Mumladze L (2020) First DNA-based records of new alien freshwater species in the Republic of Georgia. Acta Zoologica Bulgarica 72: 545–551.

Japoshvili B, Couto TBA, Mumladze L, Epitashvili G, McClain ME, Jenkins CN, Anderson EP (2021) Hydropower development in the Republic of Georgia and implications for freshwater biodiversity conservation. Biological Conservation 263: e109359. https://doi.org/10.1016/j.biocon.2021.109359

Kadye WT, Booth AJ (2012) Detecting impacts of invasive non-native sharptooth catfish, *Clarias gariepinus*, within invaded and non-invaded rivers. Biodiversity and Conservation 21(8): 1997–2015. https://doi.org/10.1007/s10531-012-0291-5

Kikodze D, Memiadze N, Kharazishvili D, Manvelidze Z, Mueller-Schaerer H (2010) The alien flora of Georgia. 2nd Edn. Swiss National Science Foundation, Swiss Agency for Development and Cooperation and SCOPES (project number IB73A0–110830): 1–36.

Kolangi-Miandare H, Askari G, Fadakar D, Aghilnegad M, Azizah S (2013) The biometric and cytochrome oxidase sub unit I (COI) gene sequence analysis of *Syngnathus abaster* (Teleostei: Syngnathidae) in Caspian Sea. Molecular Biology Research Communications 2: 133–142. https://doi.org/10.22099/MBRC.2013.1821

Kuljanishvili T, Epitashvili G, Japoshvili B, Patoka J, Kalous L (2021a) Finding of Nile tilapia *Oreochromis niloticus* (Cichliformes: Cichlidae) in Georgia, the South Caucasus. In: Kurniawan E (Ed.) IOP Conf. Series: Earth and Environmental Science 744: International Symposium on Aquatic Sciences and Resources Management, Bogor (Indonesia), November 2020: 1–5. https://doi.org/10.1088/1755-1315/744/1/012036

Kuljanishvili T, Mumladze L, Japoshvili B, Mustafayev N, Ibrahimov S, Patoka J, Pipoyan S, Kalous L (2021b) The first unified inventory of non-native fishes of the South Caucasian countries, Armenia, Azerbaijan, and Georgia. Knowledge and Management of Aquatic Ecosystems 422(422): 1–16. https://doi.org/10.1051/kmae/2021028

Kuljanishvili T, Patoka J, Bohatá L, Rylková K, Japoshvili B, Kalous L (2021c) Evaluation of the potential establishment of black-striped pipefish transferred by cultural drivers. Inland Waters 11(3): 278–285. https://doi.org/10.1080/20442041.2021.1909374

Latombe G, Pyšek P, Jeschke JM, Blackburn TM, Bacher S, Capinha C, Costello MJ, Fernandez M, Gregory RD, Hobern D, Hui C, Jetz W, Kumschick S, McGrannachan C, Pergl

J, Roy HE, Scalera R, Squires ZE, Wilson JRU, Winter M, Genovesi P, McGeoch MA (2017) A vision for global monitoring of biological invasions. Biological Conservation 213: 295–308. https://doi.org/10.1016/j.biocon.2016.06.013

Leunda PM, Oscoz J, Elvira B, Agorreta A, Perea S, Miranda R (2008) Feeding habits of the exotic black bullhead *Ameiurus melas* (Rafinesque) in the Iberian Peninsula: First evidence of direct predation on native fish species. Journal of Fish Biology 73(1): 96–114. https://doi.org/10.1111/j.1095-8649.2008.01908.x

Martín-Torrijos L, Sandoval-Sierra JV, Muñoz J, Diéguez-Uribeondo J, Bosch J, Guayasamin JM (2016) Rainbow trout (*Oncorhynchus mykiss*) threaten Andean amphibians. Neotropical Biodiversity 2(1): 26–36. https://doi.org/10.1080/23766808.2016.1151133

Mazza G, Tricarico E, Genovesi P, Gherardi F (2014) Biological invaders are threats to human health: An overview. Ethology Ecology and Evolution 26(2–3): 112–119. https://doi.org/10.1080/03949370.2013.863225

Mittermeier R, Gil P, Hoffman M, Pilgrim J, Brooks T, Mittermeier C, Lamoreux J, da Fonseca GAB (2004) Hotspots revisited: Earth's biologically richest and most endangered terrestrial ecoregions. Camex, Mexico, 391 pp.

Moghaddas SD, Abdoli A, Kiabi BH, Rahmani H, Vilizzi L, Copp GH (2021) Identifying invasive fish species threats to RAMSAR wetland sites in the Caspian Sea region–A case study of the Anzali Wetland Complex (Iran). Fisheries Management and Ecology 28(1): 28–39. https://doi.org/10.1111/fme.12453

Mumladze L, Bikashvili A, Japoshvili B, Anistratenko VV (2019) New alien species *Mytilopsis leucophaeata* and *Corbicula fluminalis* (Mollusca, Bivalvia) recorded in Georgia and notes on other non-indigenous molluscs invaded the South Caucasus. Vestnik Zoologii 53(3): 187–194. https://doi.org/10.2478/vzoo-2019-0019

Mumladze L, Japoshvili B, Anderson EP (2020) Faunal biodiversity research in the Republic of Georgia: A short review of trends, gaps, and needs in the Caucasus biodiversity hotspot. Biologia 75(9): 1385–1397. https://doi.org/10.2478/s11756-019-00398-6

Muradov I (2022) The Russian hybrid warfare: The cases of Ukraine and Georgia. Defence Studies 1–24. https://doi.org/10.1080/14702436.2022.2030714

Musayev MA, Quliyev ZM, Rehimov DB (2004) Vertebrates, volume III. In: Musayev MA, (Ed.) The Animal World of Azerbaijan. Elm, Baku, 1–316. [In Azeri]

Pipoyan SKH (2015) Discovery of black eel *Anguilla Anguilla* in Armenian waters. Biological Journal of Armenia 3: 104–106.

Pipoyan SK, Arakelyan AS (2015) The distribution of topmouth gudgeon *Pseudorasbora parva* (Temminck et Schlegel, 1846) (Actinopterygii: Cyprinidae) in water bodies of Armenia. Russian Journal of Biological Invasions 6(3): 179–183. https://doi.org/10.1134/S2075111715030030

Piria M, Copp GH, Dick JT, Duplić A, Groom Q, Jelić D, Lucy FE, Roy HE, Sarat E, Simonović P, Tomljanović T, Tricarico E, Weinlander M, Adámek Z, Bedolfe S, Coughlan NE, Davis E, Dobrzycka-Krahel A, Grgić Z, Kırankaya ŞG, Ekmekçi FG, Lajtner J, Lukas JAY, Koutsikos N, Mennen GJ, Mitić B, Pastorino P, Ruokonen TJ, Skóra ME, Smith ERC, Šprem N, Tarkan AS, Treer T, Vardakas L, Vehanen T, Vilizzi L, Zanella D, Caffrey JM (2017) Tackling invasive alien species in Europe II: Threats and opportunities until 2020. Management of Biological Invasions 8(3): 273–286. https://doi.org/10.3391/mbi.2017.8.3.02

Piria M, Simonović P, Kalogianni E, Vardakas L, Koutsikos N, Zanella D, Ristovska M, Apostolou A, Adrović A, Mrdak D, Tarkan AS, Milošević D, Zanella LN, Bakiu R, Ekmekçi FG, Povž M, Korro K, Nikolić V, Škrijelj R, Kostov V, Gregori A, Joy MK (2018) Alien freshwater fish species in the Balkans - Vectors and pathways of introduction. Fish and Fisheries 19(1): 138–169. https://doi.org/10.1111/faf.12242

Piria M, Stroil BK, Giannetto D, Tarkan AS, Gavrilović A, Špelić I, Radočaj T, Killi N, Filiz H, Uysal UT, Aldemir C, Kamberi E, Hala E, Bakiu R, Kolitari J, Buda E, Bakiu SD, Sadiku E, Bakrač A, Mujić E, Avdić S, Doumpas N, Giovos I, Dinoshi I, Ušanović L, Kalajdžić A, Pešić A, Ćetković I, Marković O, Milošević D, Mrdak D, Sará G, Belmar MB, Marchessaux G, Trajanovski S, Zdraveski K (2021) An assessment of regulation, education practices and socio-economic perceptions of non-native aquatic species in the Balkans. Journal of Vertebrate Biology 70(4): e21047. https://doi.org/10.25225/jvb.21047

Poeta G, Staffieri E, Acosta AT, Battisti C (2017) Ecological effects of anthropogenic litter on marine mammals: A global review with a "black-list" of impacted taxa. Hystrix, The Italian Journal of Mammalogy 28: 253–264. https://doi.org/10.4404/hystrix-00003-2017

R Core Team (2021) R: A Language and Environment for Statistical Computing. Vienna, Austria: R Foundation for Statistical Computing. https://www.r-project.org/

Robin X, Turck N, Hainard A, Tiberti N, Lisacek F, Sanchez JC, Müller M (2011) pROC: An open-source package for R and S+ to analyze and compare ROC curves. BMC Bioinformatics 12(1): e77. https://doi.org/10.1186/1471-2105-12-77

Roch S, von Ammon L, Geist J, Brinker A (2018) Foraging habits of invasive three-spined sticklebacks (*Gasterosteus aculeatus*) - impacts on fisheries yield in Upper Lake Constance. Fisheries Research 204: 172–180. https://doi.org/10.1016/j.fishres.2018.02.014

Rowe DK, Moore A, Giorgetti A, Maclean C, Grace P, Wadhwa S, Cooke J (2008) Review of the impacts of gambusia, redfin perch, tench, roach, yellowfin goby and streaked goby in Australia. Prepared for the Australian Government Department of the Environment, Water, Heritage and the Arts. https://www.awe.gov.au/sites/default/files/documents/introduce-fish.pdf

Roy HE, Bacher S, Essl F, Adriaens T, Aldridge DC, Bishop JDD, Blackburn TM, Branquart E, Brodie J, Carboneras C, Cottier-Cook EJ, Copp GH, Dean HJ, Eilenberg J, Gallardo B, Garcia M, García-Berthou E, Genovesi P, Hulme PE, Kenis M, Kerckhof F, Kettunen M, Minchin D, Nentwig W, Nieto A, Pergl J, Pescott OL, Peyton JM, Preda C, Roques A, Rorke SL, Scalera R, Schindler S, Schönrogge K, Sewell J, Solarz W, Stewart AJA, Tricarico E, Vanderhoeven S, van der Velde G, Vilà M, Wood CA, Zenetos A, Rabitsch W (2019) Developing a list of invasive alien species likely to threaten biodiversity and ecosystems in the European Union. Global Change Biology 25: 1032–1048. https://doi.org/10.1111/gcb.14527

Sadeghi R, Esmaeili HR, Zarei F, Esmaeili A, Abbasi K (2019) The taxonomic status of an introduced freshwater goby of the genus Rhinogobius to Iran (Teleostei: Gobiidae). Zoology in the Middle East 65(1): 51–58. https://doi.org/10.1080/09397140.2018.1540149

Seebens H, Blackburn TM, Dyer EE, Genovesi P, Hulme PE, Jeschke JM, Pagad S, Pyšek P, Winter M, Arianoutsou M, Bacher S, Blasius B, Brundu G, Capinha C, Celesti-Grapow L, Dawson W, Dullinger S, Fuentes N, Jäger H, Kartesz J, Kenis M, Kreft H, Kühn I, Lenzner B, Liebhold A, Mosena A, Moser D, Nishino M, Pearman D, Pergl J, Rabitsch W, Rojas-Sandoval J, Roques A, Rorke S, Rossinelli S, Roy HE, Scalera R, Schindler S, Štajerová K, Tokarska-Guzik B, Van Kleunen M, Walker K, Weigelt P, Yamanaka T, Essl F (2017) No saturation in the accumulation of alien species worldwide. Nature Communications 8(1): 1–9. https://doi.org/10.1038/ncomms14435

Shackleton RT, Biggs R, Richardson DM, Larson BM (2018) Social-ecological drivers and impacts of invasion-related regime shifts: Consequences for ecosystem services and human well-being. Environmental Science & Policy 89: 300–314. https://doi.org/10.1016/j.envsci.2018.08.005

Sharabidze A, Mikeladze I, Gvarishvili N, Davitadze M (2018) Invasion of foreign origin (alien) woody plants in Seaside Adjara. Biological Forum: An International Journal 10: 109–113.

Shoniya L, Dzhaposhvili B, Kokosadze T (2011) The invasive species *Pseudorasbora parva* (Teleostei, Cyprynidae) in the ecosystem of lake Bazalety. Zoologicheskii. Zoological Journal 90(10): 1277–1280.

Simberloff D, Martin JL, Genovesi P, Maris V, Wardle DA, Aronson J, Courchamp F, Galil B, García-Berthou E, Pascal M, Pyšek P, Sousa R, Tabacchi E, Vilà M (2013) Impacts of biological invasions: What's what and the way forward. Trends in Ecology & Evolution 28(1): 58–66. https://doi.org/10.1016/j.tree.2012.07.013

Tittensor DP, Walpole M, Hill S, Boyce D, Britten GL, Burgess N, Butchart SHM, Reagan EC, Alkemade R, Baumung R, Bellard C, Bouwman L, Boles-Newark NJ, Chenery AM, Cheung WWL, Christensen V, Cooper HD, Crowther AR, Dixon MJR, Galli A, Gaveau V, Gregory RD, Gutierrez NL, Hirsch TL, Höft R, Januchowsky-Hartley SR, Karmann M, Krug CB, Leverington FJ, Loh J, Lojenga RK, Malsch K, Marques A, Morgan DHW, Mumby PJ, Newbold T, Noonan-Mooney K, Pagad SN, Parks BC, Pereira HM, Robertson T, Rondinini C, Santini L, Scharlemann JPW, Schindler S, Sumaila UR, Teh LSL, van Kolck J, Visconti P, Ye Y (2014) A mid-term analysis of progress towards international biodiversity targets. Science 346(6206): 241–244. https://doi.org/10.1126/science.1257484

Vergara IA, Norambuena T, Ferrada E, Slater AW, Melo F (2008) StAR: A simple tool for the statistical comparison of ROC curves. BMC Bioinformatics 9(1): 265. https://doi.org/10.1186/1471-2105-9-265

Vilizzi L, Copp GH, Hill JE, Adamovich B, Aislabie L, Akin D, Al-Faisal AJ, Almeida D, Azmai MNA, Bakiu R, Bellati A, Bernier R, Bies JM, Bilge G, Branco P, Bui TD, Canning-Clode J, Cardoso Ramos HA, Castellanos-Galindo GA, Castro N, Chaichana R, Chainho P, Chan J, Cunico AM, Curd A, Dangchana P, Dashinov D, Davison PI, de Camargo MP, Dodd JA, Durland Donahou AL, Edsman L, Ekmekçi FG, Elphinstone-Davis J, Erős T, Evangelista C, Fenwick G, Ferincz Á, Ferreira T, Feunteun E, Filiz H, Forneck SC, Gajduchenko HS, Gama Monteiro J, Gestoso I, Giannetto D, Gilles AS, Gizzi Jr F, Glamuzina B, Glamuzina L, Goldsmit J, Gollasch S, Goulletquer P, Grabowska J, Harmer R, Haubrock PJ, He D, Hean JW, Herczeg G, Howland KL, İlhan A, Interesova E, Jakubčinová K, Jelmert A, Johnsen SI, Kakareko T, Kanongdate K, Killi N, Kim J-E, Kırankaya ŞG, Kňazovická D, Kopecký O, Kostov V, Koutsikos N, Kozic S, Kuljanishvili T, Kumar B, Kumar L, Kurita Y, Kurtul I, Lazzaro L, Lee L, Lehtiniemi M, Leonardi G, Leuven RSEW, Li S, Lipinskaya T, Liu F, Lloyd L, Lorenzoni M, Luna SA, Lyons TJ, Magellan K, Malmstrøm M, Marchini A, Marr SM, Masson G, Masson L, McKenzie CH, Memedemin D, Mendoza R, Minchin D, Miossec L, Moghaddas SD, Moshobane MC, Mumladze L, Naddafi R, Najafi-Majd E, Năstase A, Năvodaru I, Neal JW, Nienhuis S, Nimtim M, Nolan ET, Occhipinti-Ambrogi A, Ojaveer H, Olenin S, Olsson K, Onikura N, O'Shaughnessy K, Paganelli D, Parretti P, Patoka J, Pavia RTB, Jr Pellitteri-Rosa D, Pelletier-Rousseau M, Peralta EM, Perdikaris C, Pietraszewski D, Piria M, Pitois S, Pompei L, Poulet N, Preda C, Puntila-Dodd R, Qashqaei AT, Radočaj T, Rahmani H, Raj S, Reeves D, Ristovska M, Rizevsky V, Robertson DR, Robertson P, Ruykys L, Saba AO, Santos JM, Sarı HM, Segurado P, Semenchenko V, Senanan W, Simard N, Simonović P, Skóra ME, Slovák Švolíková K, Smeti E, Šmídová T, Špelić I, Srėbalienė G, Stasolla G, Stebbing P, Števove B, Suresh VR, Szajbert B, Ta KAT, Tarkan AS, Tempesti J, Therriault TW, Tidbury HJ, Top-Karakuş N, Tricarico E, Troca DFA, Tsiamis K, Tuckett QM, Tutman P, Uyan U, Uzunova E, Vardakas L, Velle G, Verreycken H, Vintsek L, Wei H, Weiperth A, Weyl OLF, Winter ER, Włodarczyk R, Wood LE, Yang R, Yapıcı S, Yeo SSB, Yoğurtçuoğlu B, Yunnie ALE, Zhu Y, Zięba G, Žitňanová K, Clarke S (2021) A global-scale screening of non-native aquatic organisms to identify potentially invasive species under current and future climate conditions. Science of the Total Environment 788: e147868. https://doi.org/10.1016/j.scitotenv.2021.147868

Vilizzi L, Hill JE, Piria M, Copp GH (2022) A protocol for screening potentially invasive non-native species using Weed Risk Assessment-type decision-support toolkits. Science of the Total Environment 832: e154966. https://doi.org/10.1016/j.scitotenv.2022.154966

Weyl OLF, Daga VS, Ellender BR, Vitule JRS (2016) A review of *Clarias gariepinus* invasions in Brazil and South Africa. Journal of Fish Biology 89(1): 386–402. https://doi.org/10.1111/jfb.12958

Yusifov EF, Alekperov IK, Ibrahimov SR, Aliyev AR, Guliyev GN, Mustafayev NJ (2017) About the biological diversity of inland water ecosystems in Azerbaijan. Proceedings of the Azerbaijan National Academy of Science 72: 74–91.

Supplementary material 1

Combined AS-ISK report including the 96 screenings for the 32 fish species screened for the South Caucasus

Authors: Levan Mumladze, Tatia Kuljanishvili, Bella Japoshvili, Giorgi Epitashvili, Lukáš Kalous, Lorenzo Vilizzi, Marina Piria

Data type: Pdf file

Explanation note: Combined AS-ISK report including the 96 screenings for the 32 fish species screened for the South Caucasus region.

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Changing climate may mitigate the invasiveness risk of non-native salmonids in the Danube and Adriatic basins of the Balkan Peninsula (south-eastern Europe)

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Abstract

Salmonids are an extensively hatchery-reared group of fishes that have been introduced worldwide mainly for their high commercial and recreational value. The Balkan Peninsula (south-eastern Europe) is characterised by an outstanding salmonid diversity that has become threatened by the introduction of non-native salmonids whose potential risk of invasiveness in the region remains unknown and especially so under predicted climate change conditions. In this study, 13 extant and four horizon non-native salmonid species were screened for their risk of invasiveness in the Danube and Adriatic basins of four Balkan countries. Overall, six (35%) of the screened species were ranked as carrying a high risk of invasiveness under current climate conditions, whereas under predicted conditions of global warming, this number decreased to three (17%). Under current climate conditions, the very high risk ('top invasive') species were rainbow trout *Oncorhynchus mykiss* and brown trout *Salmo trutta* (*sensu stricto*), whereas under predicted climate change, this was true only of *O. mykiss*. A high risk was also attributed to horizon vendace *Coregonus albula* and lake charr *Salvelinus namaycush,* and to extant Atlantic salmon *Salmo salar* and brook trout *Salvelinus fontinalis*, whose risk of invasiveness, except for *S. fontinalis*, decreased to medium. For the other eleven medium-risk species, the risk score decreased under predicted climate change, but still remained medium. The outcomes of this study reveal that global warming will influence salmonids and that only species with wider temperature tolerance, such as *O. mykiss* will likely prevail. It is anticipated that the present results may contribute to the implementation of appropriate management plans to prevent the introduction and translocation of non-native salmonids across the Balkan Peninsula. Additionally, adequate measures should be developed for aquaculture facilities to prevent escapees of non-native salmonids with a high risk of invasiveness, especially into recipient areas of high conservation value.

Keywords
AS-ISK, extant, fish, horizon, invasive, risk screening

Introduction

Following the exponential increase in recent years in the number of introduced species worldwide (Vilà et al. 2010; Gesundheit and Macias Garcia 2018; Boer et al. 2020; Hughes et al. 2020), biological invasions have become a leading driver of global biodiversity loss (Butchart et al. 2010; Pyšek et al. 2020), posing a serious threat to native biota, including aquatic ecosystems (Piria et al. 2018). In inland waters, freshwater fishes are one of the most frequently introduced groups of organisms (García-Berthou et al. 2005; Cucherousset et al. 2008) that may seriously disrupt ecosystem function through competition, predation, disease, and pest transmission and hybridisation (García-Berthou et al. 2005; Gozlan et al. 2005; Hughes et al. 2020).

Amongst freshwater fishes, salmonids are one of the most widely introduced groups (Buoro et al. 2016), mainly due to their high commercial and recreational value (e.g. Simonović et al. 2015; Piria et al. 2017). As an extensively hatchery-reared group of fishes, salmonids require particular attention since intensive stocking pressure by fishery managers and anglers may threaten the genetic diversity of indigenous salmonid species (Araki and Schmid 2010; Pinter et al. 2019). In this respect, interbreeding between introduced and native salmonids inevitably leads to 'genetic contamination', which may affect either a single population (Crisp 2000) or an entire species, including its evolutionary potential (Pinter et al. 2019). Furthermore, the stocking of salmonids in inland waters is usually done when specimens are ready to consume larger prey; this makes predation one of the principal impacts of salmonids on native aquatic organisms, both vertebrates and invertebrates (Cadwallader 1996; Piria et al. 2020; Čanak Atlagić et al. 2021).

Located in south-eastern Europe, the Balkan Peninsula was a glacial refugium for a large number of endemic species (Bănărescu 2004; Oikonomou et al. 2014) and is currently recognised as one of the world's 35 biodiversity hotspots (Hewitt 2011). In regions that are so important from a conservation perspective, the introduction of new predators can affect the abundance of native species and increase their risk of extinction (Pyšek et al. 2020), which is of even more concern for valuable and vulnerable endemic species. Examples are Australia and New Zealand, where native galaxiids have been threatened to the brink of extinction by introduced rainbow trout *Oncorhynchus mykiss* and brown trout *Salmo trutta* (McIntosh et al. 1994; McIntosh and Townsend 1995; Glova 2003; Joy et al. 2019).

The exact period of first introduction, re-introduction and translocation of salmonids in the Balkan Peninsula remains unknown, though in the past century these activities have intensified considerably as a result of re-stocking for recreational fishing (Piria et al. 2018). However, besides impacting on the endemic fauna, such practices in the Danube and Adriatic basins of the region may lead to biotic homogenisation of native salmonids amongst which native *Salmo trutta* is known to be particularly threatened (Škraba Jurlina et al. 2020). This taxon is often considered a complex of distinct species concordant with their matching two distinct haplogroups (Bernatchez 2001), whereas the number of species contained within the *Salmo trutta* complex remains debatable (Kalayci et al. 2018; Makhrov and Lajus 2018). However, the introduction of stream-dwelling *Salmo trutta* (*sensu stricto*) of Atlantic origin and of Macedonian trout *Salmo macedonicus* of Adriatic mitochondrial haplotype, originating from the Aegean Basin, has made this unresolved taxonomy even more complicated (Latiu et al. 2020), primarily due to long-term hybridisation (Škraba Jurlina et al. 2020). Additionally, several repeated translocations of salmonids have taken place, mainly from the Danube Basin into the Adriatic Basin of the Balkan Peninsula involving European grayling *Thymallus thymallus* and *Salmo trutta* of Danube origin. Finally, *Hucho hucho* and endemic soft-muzzled trout *Salmo obtusirostris* have also been translocated at different locations within the same basin (Pofuk et al. 2017), with both species flagged as endangered in the IUCN Red List (https://www.iucnredlist.org/).

Previous risk screenings have been carried out for salmonid species partly covering the Danube and Adriatic basins of the Balkan Peninsula (Simonović et al. 2013), as well as for eleven non-native trout species and strains from Serbia (Simonović et al. 2015). However, those screenings did not account for climate change predictions, nor did they include any horizon species, i.e. species present in nearby regions but not yet found in the risk assessment area. A risk screening study accounting for climate change predictions was recently carried out for seven extant and five horizon salmonids (Radočaj et al. 2021), but covered only the northern part of the Danube and Adriatic basins. Hence, the full potential risk posed by extant and horizon salmonid species on the diverse and vulnerable freshwater biota of the Balkan Peninsula remains unknown, especially given climate change predictions of global warming.

To fill the above knowledge gap, the aims of this study were to: (i) identify the translocated and introduced salmonid species of the Danube and Adriatic basins of the Balkan Peninsula; (ii) identify by horizon scanning which non-native salmonid species might enter the Balkan Peninsula in the (near) future from neighbouring countries; and (iii) evaluate the risk of invasiveness of both the identified extant and horizon salmonids under current and future (predicted) climate conditions for the risk assessment area.

Given their extensive use in aquaculture, regular monitoring of the invasiveness of non-native salmonids is crucial to achieve better management of the native freshwater biota of the Balkan Peninsula with the aim of improving appropriate conservation measures.

Methods

Risk assessment area

The risk assessment area includes the Danube and Adriatic basins of Bosnia and Herzegovina, Croatia, Montenegro and Serbia (including Kosovo) (Fig. 1). According to the updated Köppen-Geiger climate map (Rubel et al. 2017), the warm-temperate climate types without dry season *Cfa* and *Cfb* (with warm and hot summer, respectively) are predominant in the risk assessment area and especially in the Danube Basin. Specifically, the *Cfa* type is characteristic of the lower-lying areas of the Danube Basin and of the north-western coastal part of the Adriatic Basin, whereas the *Cfb* type is predominant in the higher-lying continental areas of both basins. The south-eastern coastal part of the Adriatic Basin belongs to the warm-temperate climate types with dry summer *Csa* and *Csb* (warm and hot summer, respectively). Finally, the boreal climate types without dry season *Dfb* and *Dfc* (warm and cold summer, respectively) are found only at the highest elevations of the mountain ranges of the region, namely the Dinaric Alps, Rhodopes, Carpathians and Balkan mountains.

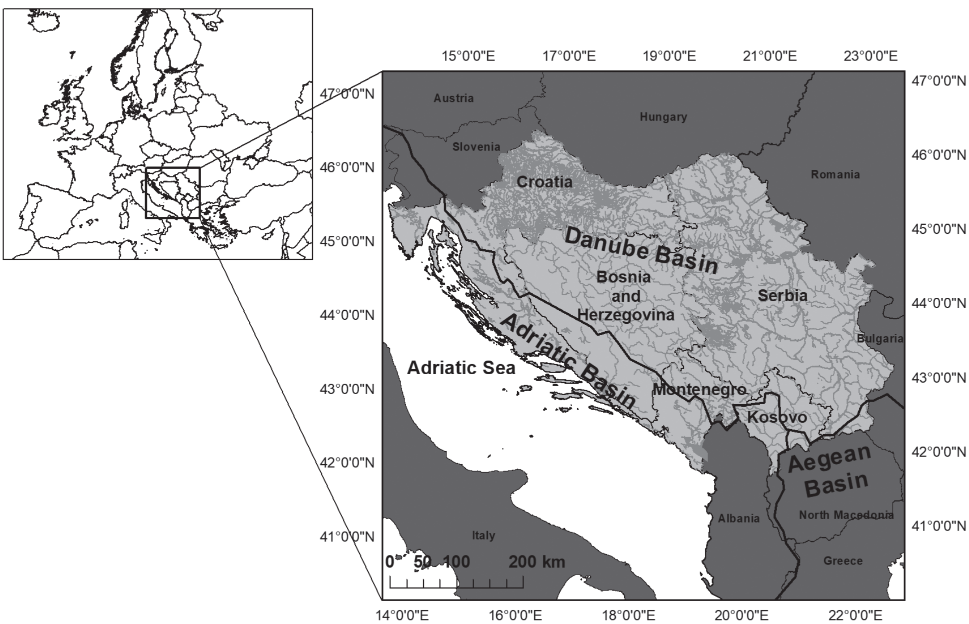

Figure 1. Map of the risk assessment area (Danube and Adriatic Basins of Bosnia and Herzegovina, Croatia, Montenegro and Serbia with Kosovo) and neighbouring countries for evaluating the potential invasiveness of non-native salmonids.

The Danube Basin includes large lowland rivers, amongst which the most important, besides the River Danube, are the River Sava (Bosnia and Herzegovina, Croatia, Serbia) and the River Tisa (Serbia). The largest river of the Adriatic Basin is the River Neretva (Bosnia and Herzegovina, Croatia). Several other large rivers are present, though the main characteristic of the Adriatic Basin's hydrology is the presence of numerous karst-sinking rivers, springs and perennial streams (Jelić et al. 2016; Piria et al. 2017).

The Balkan Peninsula is characterised by a remarkable diversity of native salmonids, especially in the countries of Bosnia and Herzegovina (Škraba et al. 2017), Croatia (Sušnik et al. 2007; Buj et al. 2021), Montenegro (Mrdak et al. 2012) and Serbia (Simonović et al. 2017). At the same time, freshwater salmonid aquaculture in the aforementioned Balkan countries has a long tradition dating back to the late 19th century. The predominantly farmed non-native salmonid species in both the Danube and Adriatic Basins of these countries is *Oncorhynchus mykiss* which, together with *Salmo trutta* (*sensu stricto*) of Atlantic origin, represents the main food fish for inland water re-stocking (Piria et al. 2018). Other salmonid species reared in aquaculture are also found, namely Arctic charr *Salvelinus alpinus*, brook trout *Salvelinus fontinalis* and huchen *Hucho hucho*, although they are mostly used for re-stocking purposes (Kapetanović et al. 2010; Muhamedagić and Habibović 2013; Piria et al. 2018).

Species selection

In total, 17 salmonid species were included as part of the risk screening (Table 1). Selection of the species for screening was according to the following Criteria (where Criteria 1, 2 and 5 are the same as those defined in Piria et al. 2016 and Radočaj et al. 2021):

1. Native species translocated from the Danube Basin to the Adriatic Basin (*n* = 2: *Thymallus thymallus* and *Salmo labrax*, which also includes the tentative *Salmo taleri*);
2. Native species translocated outside their native range, but within the Danube Basin (*n* = 1: *Hucho hucho*);
3. Native species translocated outside their native range, but within the Adriatic Basin (*n* = 1: *Salmo obtusirostris*);
4. Native species translocated from the Aegean Basin to the Danube Basin (*n* = 1: *Salmo macedonicus*).
5. Non-native species already present and naturalised/acclimatised in one or more drainage basins (*n* = 8: European whitefish *Coregonus lavaretus*, peled *Coregonus peled*, *Oncorhynchus mykiss*, Ohrid trout *Salmo letnica*, *Salmo salar*, *Salmo trutta* (*sensu stricto*), Arctic charr *Salvelinus alpinus*, brook trout *Salvelinus fontinalis*);
6. Horizon species, i.e. not yet reported, but likely to enter the risk assessment area in the near future (*n* = 4: lake trout *Salvelins namaycush*, lake charr *Salvelinus umbla*, Chinook salmon *Oncorhynchus tshawytscha*, vendace *Coregonus albula*). These species were selected by using the CABI scanning tool (www.cabi.org/horizonscanningtool) for each country in the risk assessment area separately and by literature searches (e.g. Ventura et al. 2017; Radočaj et al. 2021), including studies in the native language and 'grey' literature.

Risk screening

Risk screening was undertaken using the Aquatic Species Invasiveness Screening Kit (AS-ISK: Copp et al. 2016, 2021), which is available for free download at www.cefas.co.uk/nns/tools. This taxon-generic decision-support tool consists of 55 questions: the first 49 questions comprise the Basic Risk Assessment (BRA) and address the biogeography/invasion history and biology/ecology of the species under screening; the last six questions comprise the Climate Change Assessment (CCA) and require the assessor to predict how future predicted climatic conditions are likely to affect the BRA with respect to risks of introduction, establishment, dispersal and impact. In this study, for the CCA component, local warming scenarios for the Danube and Adriatic Basins of the Balkan Peninsula were used. Accordingly, temperatures are expected to increase from 2030 to 2060 by 1.1–1.7 °C in the Danube Basin (Stagl and Hattermann 2016) and by 1.5–2.5 °C in the Adriatic Basin (Karleuša et al. 2018). In addition, in the Adriatic Basin, an expected decrease in precipitation by 25 mm per decade (5–20% by 2050) would result in a 15% decrease of freshwaters in the Balkan Peninsula (Karleuša et al. 2018). The temperature tolerance ranges for each screened species were searched for in literature, although data were often incomplete and varied depending on the source. Screenings were undertaken on all species initially by four independent assessors (authors AM, IŠ, TK, TR), but with the final screenings based on three assessors (combination 3IA: Vilizzi et al. 2022) (see Results).

To achieve a valid screening, the assessor must provide for each question a response, a level of confidence for the response (see below) and a justification based on literature sources. The outcomes are a BRA score and a (composite) BRA+CCA score, which is obtained after adding or subtracting up to 12 points to the BRA score or leaving it unchanged in case of a CCA score equal to 0. Scores < 1 suggest that the species poses a 'low risk' to become invasive in the risk assessment area, whereas scores ≥ 1 indicate a 'medium risk' or a 'high risk'. The threshold (Thr) value to distinguish between medium-risk (BRA and BRA+CCA score < Thr) and high-risk (BRA and BRA+CCA score ≥ Thr) species for the risk assessment area is obtained by 'calibration' based on the Receiver Operating Characteristic (ROC) curve analysis (see Vilizzi et al. 2022). A measure of the accuracy of the calibration analysis is the area under the curve (AUC) whose values are interpreted as: 0.7 ≤ AUC < 0.8 = acceptable discriminatory power, 0.8 ≤ AUC < 0.9 = excellent, 0.9 ≤ AUC = outstanding (Hosmer et al. 2013). For the species classified as high risk, a distinction was made in this study of the 'very high risk' species, based on an *ad hoc* threshold weighted according to the range of high-risk score values obtained for the BRA and BRA+CCA. Identification of the (very) high-risk species is useful to prioritise allocation of resources in view of a full risk assessment (Copp et al. 2016). This examines in detail the risks of: (i) introduction (entry); (ii) establishment (of one or more self-sustaining populations); (iii) dispersal (more widely within the risk assessment area, i.e. so-called secondary spread or introductions); and (iv) impacts (to native biodiversity, ecosystem function and services, and the introduction and transmission of diseases).

For the ROC curve analysis to be implemented, the species selected for screening must be categorised *a priori* as 'non-invasive' or 'invasive' using literature sources. The

Table 1. Extant and horizon non-native salmonids evaluated for their potential risk of invasiveness in the Danube and Adriatic Basins of Bosnia and Herzegovina, Croatia, Montenegro and Serbia (including Kosovo) – the risk assessment area. The criteria for selection of species are: 1 = Native species translocated from the Danube Basin to the Adriatic Basin; 2 = Native species translocated outside their native range but within the Danube Basin; 3 = Native species translocated outside their native range, but within the Adriatic Basin; 4 = Native species translocated from the Aegean Basin to the Danube Basin; 5 = Non-native species already present and naturalised/acclimatised in one or more drainage basins; 6 = Horizon species, i.e. not yet reported but likely to enter the risk assessment area in the near future. For extant species, details about the native distribution area are provided including the location and year of introduction. For all species, the *a priori* categorisation outcome into Non-invasive and Invasive is provided, based on a multi-tiered protocol (after Vilizzi et al. 2022) relying on FishBase (www.fishbase.org), the Global Invasive Species Database (GISD: www.iucngisd.org), the Centre for Agriculture and Bioscience International Invasive Species Compendium (CABI: www.cabi.org/ISC), the Invasive and Exotic Species of North America list (IESNA: www.invasive.org) and a Google Scholar literature search whenever applicable. N = no impact/threat; Y = impact/threat; '–' = absent; n.e. = not evaluated (but present in database); n.a. = not applicable.

Taxon name	Common name	Criterion	Distribution area			*A priori* categorisation					
			Native	Introduced	Year	FishBase	GISD	CABI	IESNA	GScholar	Outcome
Extant											
Coregonus lavaretus	European whitefish	5	Northern Europe	Plitvice lakes, Peruča Reservoir, River Cetina	1937	N	–	–	–	N	Non-invasive
Coregonus peled	peled	5	Northern Europe	Plitvice Lakes, Peruča reservoir, River Cetina	1937	–	–	–	–	N	Non-invasive
Hucho hucho	huchen	2	Europe	Rivers Đetinja, Jerma, Nišava, Mlava, Moravica	2001	N	–	–	–	N	Non-invasive
Oncorhynchus mykiss	rainbow trout	5	North America	Vlasina Reservoir	1792	Y	Y	Y	Y	n.a.	Invasive
Salmo labrax	Black Sea salmon	1	Eurasia	Rivers Gacka, Vrijeka	1948	N	–	–	–	N	Non-invasive
Salmo letnica	Ohrid trout	5	Europe, Lake Ohrid	Vlasina Reservoir	1950	N	n.e.	–	–	N	Invasive
Salmo macedonicus	Macedonian trout	4	Central Europe	River Jerma	2000	N	–	–	–	N	Non-invasive
Salmo obtusirostris	soft-muzzled trout	3	Europe, Adriatic Basin	River Žrnovnica	1970s	N	–	–	–	N	Non-invasive
Salmo salar	Atlantic salmon	5,6	Northern Europe	Krka Estuary, rivers Sava and Drava	1980	N	N	Y	–	n.a.	Invasive
Salmo trutta (*sensu stricto*)	brown trout	5	Western Europe	Rivers Gacka, Gradac, Vratna	1970	Y	Y	Y	Y	n.a.	Invasive
Salvelinus alpinus	Arctic charr	5	Northern Europe	Plitvice lakes, River Neretva, Peruča accumulation, Lake Kokin Brod	1963	N	–	–	–	N	Non-invasive
Salvelinus fontinalis	brook trout	5	North America	Plitvice lakes, River Neretva, Peruča accumulation, Lake Kokin Brod	1960	Y	Y	Y	–	n.a.	Invasive
Thymallus thymallus	grayling	1	Eastern Europe	Rivers Cetina, Gacka, Istria, Neretva, Rude	1960	N	–	–	–	N	Non-invasive
Horizon											
Coregonus albula	vendace	6	–	–	–	N	Y	–	–	n.a.	Invasive
Oncorhynchus tshawytscha	chinook salmon	5	–	–	–	–	N	–	–	N	Non-invasive
Salvelinus namaycush	lake charr	6	–	–	–	N	Y	Y	–	n.a.	Invasive
Salvelinus umbla	Alpine charr	6	–	–	–	N	–	–	–	N	Non-invasive

a priori categorisation of species was implemented as per Vilizzi et al. (2022) (Table 1). Confidence levels in the responses to questions in the AS-ISK are ranked using a 1–4 scale and, based on the confidence level (CL) allocated to each response, a confidence factor (CF) is obtained as:

$$CF = \sum(CL_{Qi})/(4 \times 55)\ (i = 1, \ldots, 55)$$

where CL_{Qi} is the CL for Q*i*, 4 is the maximum achievable value for confidence (i.e. very high: see above) and 55 is the total number of questions comprising the AS-ISK questionnaire (Vilizzi et al. (2022). The CF ranges from a minimum of 0.25 (i.e. all 55 questions with confidence level equal to 1) to a maximum of 1 (i.e. all 55 questions with confidence level equal to 4). Based on all 55 Qs of the AS-ISK questionnaire, the 49 Qs comprising the BRA and the six Qs comprising the CCA, the CF_{Total}, CF_{BRA} and CF_{CCA} are respectively computed.

Implementation of the ROC curve analysis followed the protocol described in Vilizzi et al. (2022), with the true/false positive/negative outcome distinction not applied to the medium-risk species, as they can be either included or not into a full (comprehensive) risk assessment depending on priority and/or availability of financial resources. The ROC curve fitting was in two steps. Firstly, separate ROC curves were generated for each of the four independent assessors and differences amongst the resulting four AUCs were statistically tested (Mann-Whitney *U*-statistic, $\alpha = 0.05$; applet StAR available at http://melolab.org/star/home.php: Vergara et al. 2008). As differences between assessor-specific AUCs were found, in the second step, a single ROC curve was generated, based on the average scores of those assessors whose AUC was above the acceptable discriminatory power. Following ROC analysis, the best threshold value that maximises the true positive rate and minimises the false positive rate was determined using Youden's *J* statistic; whereas the 'default' threshold of 1 was set to distinguish between low-risk and medium-risk species. Fitting of the ROC curve was with package pROC (Robin et al. 2011) for R x64 v.4.0.5 (R Core Team 2021) using 2000 bootstrap replicates for the confidence intervals of specificities, which were computed along the entire range of sensitivity points (i.e. 0 to 1, at 0.1 intervals). Differences in outcome scores and CF between components (BRA and BRA+CCA) and assessors (AM, IŠ, TK, TR for the scores; AM, IŠ, TR for the CF) were tested with permutational ANOVA, based on a two-factor design with factors Component and Assessor crossed and both fixed. Analysis was implemented in PERMANOVA+ for PRIMER v.7, with normalisation of the data and using a Bray-Curtis dissimilarity measure, 9999 unrestricted permutations of the raw data and with statistical effects evaluated at $\alpha = 0.05$, including *a posteriori* pair-wise comparisons in case of significance.

Results

Across all four assessors (Fig. 2): the BRA scores ranged from 0 to 38.0, with mean = 18.3, median = 16.8 and 5% and 95% CI (confidence interval) = 3.1 and 36.5; the BRA+CCA scores ranged from –6.0 to 48.0, with mean = 13.5, median = 11.5 and 5% and 95%

CI = –2.0 and 33.0. The mean BRA score was significantly higher than the mean BRA+CCA score (18.3 ± 9.6 SE vs. 13.5 ± 11.8) and the overall scores (i.e. BRA and BRA+CCA) for assessor AM (21.7 ± 10.9) were significantly higher than those for the

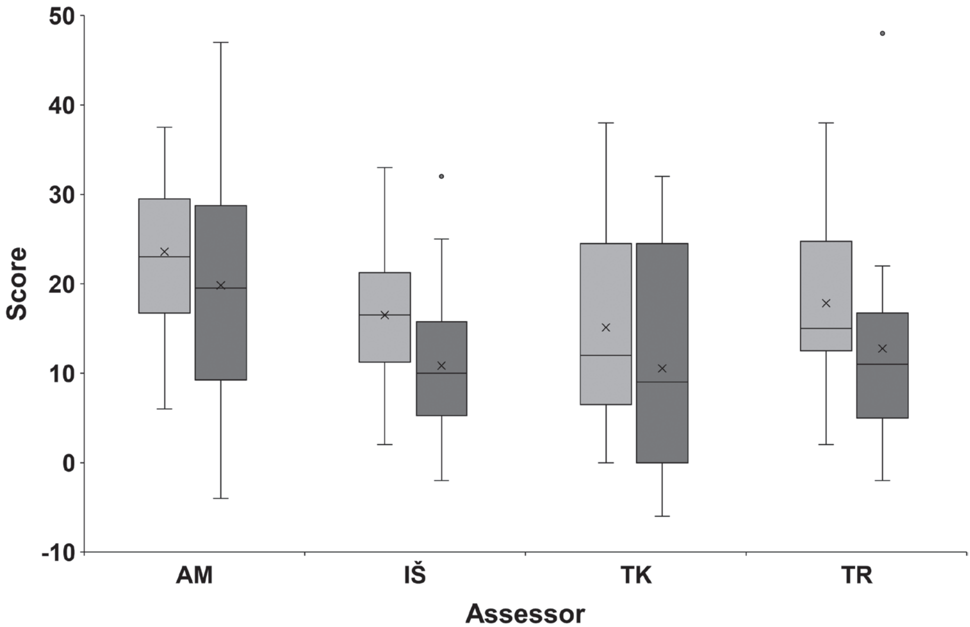

Figure 2. Box-and-whisker plots showing the Aquatic Species Invasiveness Screening Kit (AS-ISK) outcome scores (Basic Risk Assessment, BRA: light grey; BRA + Climate Change Assessment, BRA+CCA: dark grey) for the four assessors (AM = Ana Marić; IŠ = Ivan Špelić; TK = Tamara Kanjuh; TR = Tena Radočaj) screening the non-native salmonids for the risk assessment area (see Fig. 1).

Table 2. Permutational ANOVA results for the Aquatic Species Invasiveness Screening Kit (AS-ISK) outcome scores and for the confidence factor (CF) of the non-native salmonids screened for the risk assessment area. Component = BRA, BRA+CCA (see Table 3).

Source of variation	df	MS	$F^{\#}/t$	$P^{\#}$
Scores				
Component	1	6.431	7.177	0.009
Assessor	3	4.578	5.109	0.002
AM vs. IŠ	1	–	3.402	**< 0.001**
AM vs. TK	1	–	3.228	**0.003**
AM vs. TR	1	–	2.522	**0.014**
IŠ vs. TK	1	–	0.343	0.734
IŠ vs. TR	1	–	0.717	0.476
TK vs. TR	1	–	0.928	0.352
Component × Assessor	3	0.045	0.050	0.984
Residual	128	0.896		
CF				
Component	1	10.540	24.515	**< 0.001**
Assessor	2	23.664	55.040	**< 0.001**
AM vs. IŠ	1	–	5.058	**< 0.001**
AM vs. TR	1	–	10.111	**< 0.001**
IŠ vs. TR	1	–	5.604	**< 0.001**
Component × Assessor	2	0.929	2.162	0.123
Residual	96	0.430		

other assessors (13.7 ± 8.8 for IŠ, 12.8 ± 11.9 for TK, 15.3 ± 10.3 for TR). However, there was no interaction term, indicating that the BRA and BRA+CCA scores did not differ between each other depending on the assessor (Table 2).

There were differences in AUCs between AM and TK ($P < 0.01$), whose AUC had a much lower value (i.e. 0.6143, hence below acceptable discriminatory power) compared to the AUCs from AM, IŠ and TR (i.e. 0.9143, 0.8000 and 0.8786, respectively, hence with excellent to outstanding discriminatory power). As a result, the BRA score outcomes from TK were removed from subsequent analyses and the threshold value was computed, based on the mean BRA scores from AM, IŠ and TR. The ROC curve resulted in an AUC of 0.9286 (0.7810–1.0000 95% CI), which indicated outstanding discriminatory power. Youden's J provided the threshold of 19.25, which was used for calibration of the risk outcomes. Accordingly, based on the BRA scores, the threshold allowed the distinction of medium-risk species with scores within the interval [1, 19.25[from high-risk species with scores within [19.25, 68]; based on the BRA+CCA scores, the threshold allowed the distinction of medium-risk species with scores within the interval [1, 19.25[from high-risk species with scores within [19.25, 80]. Low-risk species had BRA scores within [−20, 1[and BRA+CCA scores within [−32, 1[(see Table 2; combined AS-ISK report in Suppl. material 1). Using the above threshold:

- Based on the BRA outcome scores (Table 3): six (35.3%) species were classified as high risk and eleven (64.7%) as medium risk. Amongst the seven species categorised *a priori* as invasive, six were true positives (*Coregonus albula*, *Oncorhynchus mykiss*, *Salmo salar*, *Salmo trutta*, *Salvelinus fontinalis*, *Salvelinus namaycush*). Of the eleven medium-risk species, ten were *a priori* non-invasive and one invasive.
- Based on the BRA+CCA outcome scores, hence after accounting for climate change predictions (Table 3): three (17.6%) species were classified as high risk, 13 (76.5%) as medium risk and one (5.9%) as low risk (*Hucho hucho*). Amongst the *a priori* invasive species, three were true positives (*Oncorhynchus mykiss*, *Salmo trutta*, *Salvelinus fontinalis*) and, amongst the ten species categorised *a priori* as non-invasive, one was a truer negative (*Hucho hucho*). Of the 13 medium-risk species, nine were *a priori* non-invasive and four invasive.

The highest-scoring species (BRA and BRA+CCA scores > 30, taken as an *ad hoc* 'very high risk' threshold) were *Oncorhynchus mykiss* and *Salmo trutta* for both the BRA and BRA+CCA and *Oncorhynchus mykiss* only for the CCA. The CCA resulted in a slight increase in the BRA score for only one species (*Oncorhynchus mykiss*), in no change for another species (*Salmo macedonicus*) and in a decrease for the remaining 15 species (Table 3).

The mean CF_{Total} was 0.707 ± 0.017 SE, the mean CF_{BRA} 0.720 ± 0.018 and the mean CF_{CCA} 0.593 ± 0.020. Across the three assessors (i.e. AM, IŠ and TR), the mean CF_{BRA} was significantly higher than the mean CF_{CCA} and the overall CF (i.e. for the BRA and CCA) for assessor AM (0.792 ± 0.112) was significantly higher than that for assessors IŠ (0.663 ± 0.135) and TR (0.515 ± 0.147), which also differed significantly. However, there was no interaction term, indicating that CF_{BRA} and CF_{CCA} did not differ between each other depending on the assessor (Table 3).

Table 3. Risk outcomes for the non-native salmonids screened with AS-ISK for the risk assessment area. For each species, the following information is provided: *a priori* categorisation of invasiveness (N = non-invasive; Y = invasive: see Table 1); BRA and BRA+CCA scores with corresponding risk outcomes (M = Medium; H = High; VH = Very high based on *ad hoc* threshold of 30: see text for details) and classification (Class: TN = true negative; TP = true positive; '–' = not applicable as medium-risk: see text for details); difference (Delta) between BRA+CCA and BRA scores; confidence factor (CF) for all 55 questions of the AS-ISK (CF_{Total}), for the 49 BRA questions (CF_{BRA}) and for the six CCA questions (CF_{CCA}). Risk outcomes are based on a threshold of 19.25 and computed as: L, within the interval [–20, 1[, M [1, 19.25[, H [19.25, 30[and VH [30, 68] for the BRA; L [–32, 1[, M [1, 19.25[, H [19.25, 30[and VH [30, 80] for the BRA+CCA (note the reverse bracket notation indicating in all cases an open interval).

Taxon name	*A priori*	BRA			BRA+CCA			Delta	CF		
		Score	Outcome	Class	Score	Outcome	Class		Total	BRA	CCA
Coregonus albula	Y	19.3	H	TP	10.0	M	–	–9.3	0.64	0.64	0.58
Coregonus lavaretus	N	18.7	M	–	14.0	M	–	–4.7	0.74	0.77	0.54
Coregonus peled	N	14.5	M	–	10.5	M	–	–4.0	0.73	0.75	0.57
Hucho hucho	N	10.0	M	–	0.0	L	TN	–10.0	0.75	0.76	0.61
Oncorhynchus mykiss	Y	33.7	VH	TP	42.3	VH	TP	8.7	0.86	0.88	0.72
Oncorhynchus tshawytscha	N	17.5	M	–	13.5	M	–	–4.0	0.70	0.70	0.68
Salmo labrax	N	19.2	M	–	15.2	M	–	–4.0	0.70	0.71	0.64
Salmo letnica	Y	15.8	M	–	11.2	M	–	–4.7	0.64	0.65	0.60
Salmo macedonicus	N	18.3	M	–	17.0	M	–	–1.3	0.57	0.58	0.44
Salmo obtusirostris	N	8.0	M	–	2.0	M	–	–6.0	0.72	0.72	0.75
Salmo salar	Y	22.2	H	TP	17.5	M	–	–4.7	0.65	0.68	0.44
Salmo trutta	Y	32.8	VH	TP	26.8	H	TP	–6.0	0.76	0.78	0.58
Salvelinus alpinus	N	19.2	M	–	13.2	M	–	–6.0	0.72	0.73	0.61
Salvelinus fontinalis	Y	29.8	H	TP	24.5	H	TP	–5.3	0.76	0.79	0.53
Salvelinus namaycush	Y	24.5	H	TP	15.8	M	–	–8.7	0.66	0.67	0.57
Salvelinus umbla	N	9.8	M	–	3.8	M	–	–6.0	0.63	0.63	0.63
Thymallus thymallus	N	14.8	M	–	8.8	M	–	–6.0	0.80	0.82	0.58

Discussion

Risk outcomes

In this study, the risk of invasiveness of 17 salmonids was determined with a very high level of accuracy (cf. discriminatory power), based on independent assessors. According to the threshold value of 19.25, based on the BRA, only six (35%) species were classified as carrying a high risk of invasiveness for the risk assessment area, whereas based on the BRA+CCA, this number decreased to three (17%). A similar decrease in score for salmonids under predicted climate change scenarios has been observed for Croatia and Slovenia (Radočaj et al. 2021), Turkey (Yoğurtçuoğlu et al. 2021) and even for regions with colder climate ranging from humid continental to sub-arctic as found in the West Siberian Plain (Interesova et al. 2020). In this study, the mean CF was lower for the CCA compared to the BRA, which agrees with other AS-ISK applications (e.g. Bilge et al. 2019; Interesova et al. 2020; Radočaj et al. 2021) and reflects the uncertainty in climate change predictions generally due to a dearth of literature for several of the screened species. On the contrary, for a species like *Oncorhynchus mykiss* for which the impact of climate change has been largely investigated (e.g. Benjamin et al. 2013; Stanković et al.

2015), the CF value for the CCA was the highest amongst all species in this study (0.72: Table 3), similar to screenings for this species in other risk assessment areas compared to other salmonids (e.g. Tarkan et al. 2017: 0.74; Moghaddas et al. 2021: 0.77).

Of the screened species, seven were found to pose a high to very high risk of invasiveness for the RA area under current climate conditions (BRA). However, after accounting for predicted climate change conditions (CCA), for four of these species, the risk of invasiveness decreased from high to medium (Table 3). Specifically, only *Oncorhynchus mykiss* was classified as very high risk for both the BRA and BRA+CCA, whereas *Salmo trutta*, which was classified as very high risk for the BRA, became of high risk after accounting for climate change. Both species belong to the List of the 100 World's Worst Invasive Alien Species (GISD 2021), likely as a result of their vagility, life history, phenotypic plasticity, broad water temperature tolerance and highly adaptive behaviour, as documented worldwide (Crowl et al. 1992; Hardy 2002; Hasegawa 2020). Finally, *Salvelinus fontinalis* was the only species classified as high risk for both the BRA and BRA+CCA.

Oncorhynchus mykiss is a top predator whose negative effects in its introduced range resulting from its carnivorous diet have been documented worldwide (Skelton 1987; Young et al. 2010; Juncos et al. 2013). In the risk assessment area, this species' impact is mostly reflected on the endemic minnow-like fishes (Zupančič et al. 2008), which has led to the near-extinction of *Telestes metohiensis* from the River Ljuta near Dubrovnik in Croatia (Piria et al. 2016). In its native range, *O. mykiss* is an anadromous species that can tolerate high salinities and a wide range of water velocities (Leitwein et al. 2017), and for this reason it is found even in catches of commercial fishers from the Adriatic Sea (M. Piria, pers. obs.). Although it is generally presumed that *O. mykiss* cannot establish viable populations in the risk assessment area, there are some documented cases of its reproduction dating back to the early 20th century in Slovenia (Franke 1913; Mršić 1935), the early 1970s in Croatia (MacCrimmon 1971), plus several more recent reports (e.g. Stanković et al. 2015; Mihinjač et al. 2019). In addition, there is evidence of reproduction in a population of *O. mykiss* in the Međimurje area (P. Simonović, pers. obs.), in the rivers of southern Croatia (D. Zanella, pers. obs.) and in southern Greece on the Island of Crete (Koutsikos et al. 2012; Stoumboudi et al. 2017).

Salmo trutta (*sensu stricto*) is one of the most attractive recreational salmonids in the risk assessment area that, however, poses a major threat to the native salmonids because of genetic contamination. Introgression of alien Atlantic haplotypes into the indigenous *Salmo labrax* and *Salmo obtusirostris* gene pool has already been documented (Simonović et al. 2014, 2015; Tošić et al. 2016; Škraba et al. 2017; Kanjuh et al. 2020, 2021; Škraba Jurlina et al. 2020), with the size of the intact native populations of these two species still remaining unknown. Although the score for *Salmo trutta* decreased as a result of the CCA, this species remains at high risk for the risk assessment area probably because of its dispersal mechanisms, which are often deliberate through farming and stocking for recreational purposes. In this respect, the first stocking of this species in the risk assessment area occurred in early 20th century and has become quite intensive in recent times (Simonović et al. 2014; Piria et al. 2018).

Salvelinus fontinalis is a valuable species for angling both in the risk assessment area and worldwide (Lenhardt et al. 2011; CABI 2021). There are known cases where the presence of introduced *S. fontinalis* has negatively affected populations of native amphibians in France and Spain (Orizaola and Braña 2006). In addition, this species has been found to overlap its diet with native *Salmo trutta* populations in southern France with which it also interferes in terms of reproductive success and hybridisation (Cucherousset et al. 2007, 2008), though resulting in sterile offspring (Hisar et al. 2003). There is evidence that *S. fontinalis* may exert detrimental impacts on native *Salmo trutta* in Sweden, leading to extinction of some native populations (Spens et al. 2007). All of this confirms that this species carries several undesirable life-history traits, which is in line with its high-risk ranking.

The three *a priori* invasive species *Coregonus albula*, *Salmo salar* and *Salvelinus namaycush* gained a high risk of invasiveness under current climate conditions (cf. BRA) whereas under the BRA+CCA, their risk became medium. *Coregonus albula* and *Salmo salar* are characterised by behavioural and developmental plasticity, which makes them capable to react and potentially adapt to variation in environmental conditions. However, there are limitations to these capacities, especially over short periods of time (Garcia de Leaniz et al. 2007; Muir et al. 2013; Karjalainen et al. 2015, 2016). In this regard, water temperature is fundamental in regulating fish physiology and environmental variation during development can play a crucial role in generating variability in offspring through phenotypic plasticity (Little et al. 2020). Migratory fishes, such as *S. salar*, are particularly vulnerable to warming environments as the appropriate time of transition between habitats is fine-tuned to specific environmental cues (Crozier et al. 2008), with the success of these transition periods having consequences on subsequent survival. *Salvelinus namaycush* is known to migrate to deep cold-water habitats and generally occupies temperatures within its optimum range (10 °C ± 2 °C: Plumb and Blanchfield 2009). This is aside from brief forays into shallow warm-water habitats to forage (Morbey et al. 2006) or to avoid limiting oxygen conditions at high depths (Guzzo and Blanchfield 2017). Therefore, increasing temperatures would drastically affect populations of *S. namaycush* by reducing suitable summer thermal habitats and by increasing exposure to sub-optimal temperatures and thermal stress (Ficke et al. 2007; Guzzo and Blanchfield 2017), thereby limiting growth and condition (Plumb et al. 2014; Guzzo et al. 2017).

Salmo letnica was the only *a priori* invasive species found to carry a medium risk of invasiveness likely due to its low dispersal mechanism traits, but also to the scarce data available to answer the AS-ISK questions about 'undesirable traits' (see Copp et al. 2016). *Coregonus lavaretus*, *Coregonus peled*, *Oncorhynchus tshawytscha*, *Salmo labrax*, *Salmo macedonicus*, *Salmo obtusirostris*, *Salvelinus alpinus*, *Salvelinus umbla* and *Thymallus thymallus* were all classified as medium-risk for both the BRA and BRA+CCA, with the risk for *Hucho hucho* becoming low after accounting for the CCA. The latest outcome is expected because *Hucho hucho* is an already threatened species due to the relatively long period of time to reach maturity during which it is intolerant of pollution and damming (Weiss and Schenekar 2016).

Climate change

As cold-water species, salmonids are likely to be strongly affected by climate change. An increase in temperature and a decrease in precipitation can directly influence water levels in rivers and lakes (e.g. Schindler 2001), with consequent changes in other water-related characteristics, such as food amount and composition, acidity and other chemical parameters (Cochrane et al. 2009). These changes could trigger a range of negative responses in salmonid fishes and especially in those species with complex life-histories consisting of several developmental stages (Crozier et al. 2008). Studies on the potential effects of climate change on salmonids have shown complex behavioural responses in *Oncorhynchus mykiss* exposed to different seasonal temperatures, acidity, nitrogen and food supply (Morgan et al. 2001; Ficke et al. 2007). Higher air temperatures could affect productivity or even cause mortality in aquaculture ponds via increased water temperatures, especially for salmonids with a narrow water temperature range (Cochrane et al. 2009). Although most salmonid ponds have a flow-through system with frequent water exchange that can mitigate increases in temperature, climate change can affect water regime by causing drought or flood events. With global warming, more precipitation events occurs as rainfall instead of snowfall, snow melts earlier and there is increased runoff and risk of flooding in early spring, but increased risk of drought in summer, especially in continental areas (Trenberth 2011; Karleuša et al. 2018). Overall, warming conditions in temperate regions across the Globe will probably not only narrow the distribution of wild salmonid stocks, but also reduce the number of appropriate sites for salmonid farming (Cochrane et al. 2009).

Implications for aquaculture

The most suitable streams for salmonid farming in the risk assessment area are in Montenegro, western Croatia and Bosnia-Herzegovina because of the presence of extensive areas with higher altitudes and boreal climate conditions. Interestingly, all salmonid farming in the risk assessment area and surrounding countries (i.e. Albania, Bulgaria, North Macedonia) is based on non-native species with *Oncorhynchus mykiss* being predominant (Koutsikos et al. 2019), followed by *Salmo trutta* (*sensu stricto*) (Piria et al. 2018). Other non-native farmed salmonid species include *Coregonus lavaretus*, *Coregonus peled*, *Salvelinus alpinus* and *Salvelinus fontinalis*. However, due to current aquaculture strategies and proposed diversification of species, farmers are trying to diversify their production with more profitable species (Ministry of Agriculture 2020). For example, in Croatia, there is an attempt to introduce *Salmo salar* in aquaculture. However, because of: (i) Regulation (EU) no. 1143/2014 of the European Parliament and of the Council on the prevention and management of the introduction and spread of invasive alien species and (ii) Council Regulation (EC) No 708/2007 of 11 June 2007 concerning the use of alien and locally absent species in aquaculture, plus (iii) national law, the introduction of new species for aquaculture in countries which are part of the European Union is becoming increasingly difficult (Piria et al. 2017, 2021a).

Overall, it is advised that non-native species introductions should be brought to a minimum or avoided altogether and that every introduction of a new species should be conducted only after a full risk assessment (e.g. Tarkan et al. 2020, 2022), because any new fish species in aquaculture carries a risk of escape (De Silva 2012). However, in countries that are not part of the EU (e.g. Bosnia-Herzegovina and Serbia), hence do not need to abide by the above EU regulations (Piria et al. 2021a), the introduction of new species in aquaculture relies only on local laws and regulations, which do not include any risk assessment. If an escape eventually occurs, monitoring programmes could be used as an early-warning system before the new species becomes established. This is especially true of large river systems, where an introduced species can be detected early so that any adverse impact can be contained (Radočaj et al. 2021). Furthermore, accidental escapees from fish farms can be a source of pathogen transmission to wild stocks (Krkošek et al. 2007; Rosenberg 2008) and this is another important threat still understudied (Wood et al. 2021). Despite tight trade measures, established customs and quarantine methods and protocols related to transboundary aquatic diseases in the Member States of the EU, introductions of new pathogens into aquaculture are still occurring (Peters et al. 2018; Pofuk 2021). Similarly, biosecurity regulations in the countries of the risk assessment area are well-developed for aquaculture, although inspection and control do not function well, whereas regulations remained completely undeveloped for the purposes of open-water re-stocking (Pofuk 2021).

Management actions

In the countries of the risk assessment area, freshwater fishing is regulated by different fisheries acts. For example, in Serbia, stocking is limited by law to native species only (Official Gazette 2018) with penalties for misdemeanours as in the case of stocking occurring not under professional control (Official Gazette 2005). Similarly, in Croatia, Montenegro and Bosnia-Herzegovina, local fisheries acts and mandatory management plans regulate stocking activities and prevent or limit possibilities for the stocking of non-native species (Vehanen et al. 2020; Piria et al. 2021a). Despite existing legislation in the countries of the risk assessment area to prevent stocking of rivers and streams with non-native fish, there are still possible pathways that mediate new (unauthorised) introductions by anglers and escapees from aquaculture (Britton et al. 2011; Cerri et al. 2018). These pathways of introduction are especially important for salmonids because of their value for local aquaculture and angling (Simonović et al. 2015). In particular, in the risk assessment, area stocking with non-native species is still possible into isolated water bodies without access to inland waters, where such species are already naturalised and have been present for a long time. Such introductions are still legally supported and prescribed in anglers' management plans. The best example of this practice is in the karst region of the River Lika in Croatia, where more than 90% of fishes are of non-native origin (i.e. mostly translocated from another basin but within the same country) and anglers' management plans are based on re-stocking with 'native' fish species, which in fact have never been native to the region (Piria et al. 2021b). On the contrary, in other connected river systems and inland waters of the risk assessment area, this practice is prohibited, so that all recent introductions with non-native fishes (if any) are considered illegal and there is no available information on such practices.

Possibly the most challenging (and still unrecognised) problem for the Balkan Peninsula is the legal stocking of salmonid streams with *Salmo trutta* (*sensu stricto*), which poses a threat to native genetic integrity (Kanjuh et al. 2020; Piria et al. 2020; Buj et al. 2021). *Salmo trutta* (*sensu stricto*) is considered a native species by law in all countries of the risk assessment area. This is because management plans for salmonid re-stocking require performing obligatory stocking by *S. trutta*, although without any specification of which lineage. In the risk assessment area, only *S. trutta* (*sensu stricto*) is found for aquaculture and there are no producers of native *Salmo* sp. for re-stocking (Piria et al. 2020). If anglers do not re-stock based on this management plan, a misdemeanour report by the inspectorate will follow. Thus, decision-makers cannot prohibit re-stocking with a species that is prescribed to be re-stocked, even if it belongs to a different lineage. Clearly, the problem of genetic contamination is still not well recognised by decision-makers and stakeholders and currently, in the risk assessment area, *S. trutta* (*sensu stricto*) interacts with *Salmo obtusirostris*, *Salmo labrax* and *Salmo macedonicus* by changing their original gene pool.

Control and containment of introduced salmonids, once established, is the only advisable approach, since eradication is virtually impossible in river systems and large lakes (Britton et al. 2011). However, containment (e.g. by artificial barriers preventing migration) and control (e.g. by gillnets and electrofishing) can be very costly endeavours and may sometimes conflict with local legislation, thereby making them not feasible across the risk assessment area. A possible solution for the control of established populations of salmonids in the long term could be to encourage anglers to remove non-native salmonids from the wild. Another solution could be the obligation by fishing clubs to use exclusively native lineages of salmonids for stocking local river systems, although in this case it would be necessary to encourage farmers to produce indigenous salmonids. Decision-makers may follow for example the farming of *Hucho hucho* in Bosnia and Herzegovina and in Slovenia for stocking in rivers where the species is indigenous (Andreji and Stráňai 2013; Muhamedagić and Habibović 2013) or of marble trout *Salmo marmoratus* by banning stocking of *Salmo trutta* (*sensu stricto*). This could be achieved by revising fishing regulations for anglers and genetically testing brood stock from hatcheries for stocking phenotypic and pure young-of-the-year *S. marmoratus* (Berrebi et al. 2022).

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References

Andreji J, Stráňai I (2013) Growth parameters of huchen *Hucho hucho* (L.) in the wild and under culture conditions. Fisheries & Aquatic Life 21: 179–188. https://doi.org/10.2478/aopf-2013-0015

Araki H, Schmid C (2010) Is hatchery stocking a help or harm? Evidence, limitations, and future directions in ecological and genetic surveys. Aquaculture 308: S2–S11. https://doi.org/10.1016/j.aquaculture.2010.05.036

Bănărescu PM (2004) Distribution pattern of the aquatic fauna of the Balkan Peninsula. In: Griffiths HI, Kryštufek B, Reed JM (Eds) Balkan biodiversity – Pattern and process in the European hotspot. Springer Science Business Media, Dordrecht, 203–218. https://doi.org/10.1007/978-1-4020-2854-0_12

Benjamin JR, Connolly PJ, Romine JG, Perry RW (2013) Potential effects of changes in temperature and food resources on life history trajectories of juvenile *Oncorhynchus mykiss.* Transactions of the American Fisheries Society 142(1): 208–220. https://doi.org/10.1080/00028487.2012.728162

Bernatchez L (2001) The evolutionary history of brown trout (*Salmo trutta* L.) inferred from phylogeographic, nested clade, and mismatch analyses of mitochondrial DNA variation. Evolution; International Journal of Organic Evolution 55(2): 351–379. https://doi.org/10.1111/j.0014-3820.2001.tb01300.x

Berrebi P, Jesenšek D, Laporte M, Crivelli AJ (2022) Restoring marble trout genes in the Soča River (Slovenia). Conservation Genetics. https://doi.org/10.1007/s10592-022-01430-0

Bilge G, Filiz H, Yapici S, Tarkan AS, Vilizzi L (2019) A risk screening study on the potential invasiveness of Lessepsian fishes in the south-western coasts of Anatolia. Acta Ichthyologica et Piscatoria 49(1): 23–31. https://doi.org/10.3750/AIEP/02422

Boer P, Loss AC, Bakker F, Beentjes K, Fisher B (2020) *Monomorium sahlbergi* Emery, 1898 (*Formicidae*, *Hymenoptera*): A cryptic globally introduced species. ZooKeys 979: 87–97. https://doi.org/10.3897/zookeys.979.55342

Britton JR, Gozlan RE, Copp GH (2011) Managing non-native fish in the environment. Fish and Fisheries 12(3): 256–274. https://doi.org/10.1111/j.1467-2979.2010.00390.x

Buj I, Raguž L, Marčić Z, Ćaleta M, Duplić A, Zanella D, Mustafić P, Ivić L, Horvatić S, Karlović R (2021) Plitvice Lakes National park harbors ancient, yet endangered diversity of trout (genus *Salmo*). Journal of Applied Ichthyology 37(1): 20–37. https://doi.org/10.1111/jai.14120

Buoro M, Olden JD, Cucherousset J (2016) Global Salmonidae introductions reveal stronger ecological effects of changing intraspecific compared to interspecific diversity. Ecology Letters 19(11): 1363–1371. https://doi.org/10.1111/ele.12673

Butchart SHM, Walpole M, Collen B, Van Strien A, Scharlemann JPW, Almond REA, Baillie JEM, Bomhard B, Brown C, Bruno J, Carpenter KE, Carr GM, Chanson J, Chenery AM, Csirke J, Davidson NC, Dentener F, Foster M, Galli A, Galloway JN, Genovesi P, Gregory RD, Hockings M, Kapos V, Lamarque J-F, Leverington F, Loh J, McGeoch MA, McRae L, Minasyan A, Hernández Morcillo M, Oldfield TEE, Pauly D, Quader S, Revenga C, Sauer JR, Skolnik B, Spear D, Stanwell-Smith D, Stuart

SN, Symes A, Tierney M, Tyrrell TD, Vié J-C, Watson R (2010) Global biodiversity: Indicators of recent declines. Science 328(5982): 1164–1168. https://doi.org/10.1126/science.1187512

CABI (2021) Invasive species compendium. CAB International, Wallingford. www.cabi.org/isc

Cadwallader PL (1996) Overview of the impacts of introduced salmonids on Australian native fauna. Australian Nature Conservation Agency, Canberra, 64 pp.

Čanak Atlagić J, Marić A, Tubić B, Andjus S, Đuknić J, Marković V, Paunović M, Simonović P (2021) What's on the menu for the resident brown trout in a rich limestone stream? Water (Basel) 13(18): e2492. https://doi.org/10.3390/w13182492

Cerri J, Ciappelli A, Lenuzza A, Zaccaroni M, Nocita A (2018) Recreational angling as a vector of freshwater invasions in Central Italy: Perceptions and prevalence of illegal fish restocking. Knowledge and Management of Aquatic Ecosystems 419(419): e38. https://doi.org/10.1051/kmae/2018028

Cochrane K, De Young C, Soto D, Bahri T (2009) Climate change implications for fisheries and aquaculture. FAO Fisheries and Aquaculture Technical Paper 530: 1–212. http://www.lis.edu.es/uploads/07483fb7_72a2_45ca_b8e7_48bf74072fd3.pdf

Copp GH, Vilizzi L, Tidbury H, Stebbing PD, Tarkan AS, Miossec L, Goulletquer P (2016) Development of a generic decision-support tool for identifying potentially invasive aquatic taxa: AS-ISK. Management of Biological Invasions 7(4): 343–350. https://doi.org/10.3391/mbi.2016.7.4.04

Copp GH, Vilizzi L, Wei H, Li S, Piria M, Al-Faisal AJ, Almeida D, Atique U, Al-Wazzan Z, Bakiu R, Bašić T, Bui TD, Canning-Clode J, Castro N, Chaichana R, Çoker T, Dashinov D, Ekmekçi FG, Erős T, Ferincz Á, Ferreira T, Giannetto D, Gilles Jr AS, Głowacki Ł, Goulletquer P, Interesova E, Iqbal S, Jakubčinová K, Kanongdate K, Kim JE, Kopecký O, Kostov V, Koutsikos N, Kozic S, Kristan P, Kurita Y, Lee HG, Leuven RSEW, Lipinskaya T, Lukas J, Marchini A, González-Martínez AI, Masson L, Memedemin D, Moghaddas SD, Monteiro J, Mumladze L, Naddafi R, Năvodaru I, Olsson KH, Onikura N, Paganelli D, Pavia Jr RT, Perdikaris C, Pickholtz R, Pietraszewski D, Povž M, Preda C, Ristovska M, Rosíková K, Santos JM, Semenchenko V, Senanan W, Simonović P, Smeti E, Števove B, Švolíková K, Ta KAT, Tarkan AS, Top N, Tricarico E, Uzunova E, Vardakas L, Verreycken H, Zięba G, Mendoza R (2021) Speaking their language – Development of a multilingual decision-support tool for communicating invasive species risks to decision makers and stakeholders. Environmental Modelling & Software 135: e104900. https://doi.org/10.1016/j.envsoft.2020.104900

Crisp DT (2000) Trout and salmon: Ecology, conservation and rehabilitation. Blackwell Science, 212 pp. https://doi.org/10.1002/9780470999776

Crowl TA, Townsend CR, Mcintosh AR (1992) The impact of introduced brown and rainbow trout on native fish: The case of Australasia. Reviews in Fish Biology and Fisheries 2(3): 217–241. https://doi.org/10.1007/BF00045038

Crozier LG, Hendry AP, Lawson PW, Quinn TP, Mantua NJ, Battin J, Shaw RG, Huey RB (2008) Potential responses to climate change in organisms with complex life histories: Evolution and plasticity in Pacific salmon. Evolutionary Applications 1(2): 252–270. https://doi.org/10.1111/j.1752-4571.2008.00033.x

Cucherousset J, Aymes JC, Santoul F, Céréghino R (2007) Stable isotope evidence of trophic interactions between introduced brook trout *Salvelinus fontinalis* and native brown trout *Salmo trutta* in a mountain stream of south-west France. Journal of Fish Biology 71: 210–223. https://doi.org/10.1111/j.1095-8649.2007.01675.x

Cucherousset J, Aymes JC, Poulet N, Santoul F, Céréghino R (2008) Do native brown trout and non-native brook trout interact reproductively? Naturwissenschaften 95(7): 647–654. https://doi.org/10.1007/s00114-008-0370-3

De Silva SS (2012) Aquaculture—A newly emergent food production sector- and perspectives of its impacts on biodiversity and conservation. Biodiversity and Conservation 21(12): 3187–3220. https://doi.org/10.1007/s10531-012-0360-9

Ficke AD, Myrick CA, Hansen LJ (2007) Potential impacts of global climate change on freshwater fisheries. Reviews in Fish Biology and Fisheries 17(4): 581–613. https://doi.org/10.1007/s11160-007-9059-5

Franke I (1913) Colourful trout (*Trutta iridea*) in red trout (*Salmo fontinalis*). Lovec 4: 130–131. [in Croatian]

García-Berthou E, Alcaraz C, Pou-Rovira Q, Zamora L, Coenders G, Feo C (2005) Introduction pathways and establishments rates of invasive aquatic species in Europe. Canadian Journal of Fisheries and Aquatic Sciences 62(2): 453–463. https://doi.org/10.1139/f05-017

Garcia de Leaniz C, Fleming IA, Einum S, Verspoor E, Jordan WC, Consuegra S, Aubin-Horth N, Lajus D, Letcher BH, Youngson AF, Webb JH, Vøllestad LA, Villanueva B, Ferguson A, Quinn TP (2007) A critical review of adaptive genetic variation in Atlantic salmon: Implications for conservation. Biological Reviews of the Cambridge Philosophical Society 82(2): 173–211. https://doi.org/10.1111/j.1469-185X.2006.00004.x

Gesundheit P, Macias Garcia C (2018) The role of introduced species in the decline of a highly endemic fish fauna in Central Mexico. Aquatic Conservation 6(6): 1384–1395. https://doi.org/10.1002/aqc.2927

GISD (2021) The Invasive Species Specialist Group ISSG – Global Invasive Species Database. http://www.issg.org/database

Glova GJ (2003) A test for interaction between brown trout (*Salmo trutta*) and inanga (*Galaxias maculatus*) in an artificial stream. Ecology of Freshwater Fish 12(4): 247–253. https://doi.org/10.1046/j.1600-0633.2003.00019.x

Gozlan RE, St-Hilaire S, Feist SW, Martin P, Kent ML (2005) Disease threat to European fish. Nature 435(7045): e1046. https://doi.org/10.1038/4351046a

Guzzo MM, Blanchfield PJ (2017) Climate change alters the quantity and phenology of habitat for lake trout (*Salvelinus namaycush*) in small Boreal Shield lakes. Canadian Journal of Fisheries and Aquatic Sciences 74(6): 871–884. https://doi.org/10.1139/cjfas-2016-0190

Guzzo MM, Blanchfield PJ, Rennie MD (2017) Behavioral responses to annual temperature variation alter the dominant energy pathway, growth, and condition of a cold-water predator. Proceedings of the National Academy of Sciences 114(37): 9912–9917. https://doi.org/10.1073/pnas.1702584114

Hardy RW (2002) Rainbow trout, *Oncorhynchus mykiss*. In: Webster CD, Lim C (Eds) Nutrient requirements and feeding of finfish for aquaculture. CAB International, Oxon, 184–202. https://doi.org/10.1079/9780851995199.0184

Hasegawa K (2020) Invasions of rainbow trout and brown trout in Japan: A comparison of invasiveness and impact on native species. Ecology of Freshwater Fish 29(3): 419–428. https://doi.org/10.1111/eff.12534

Hewitt GM (2011) Mediterranean peninsulas: the evolution of hotspots. In: Zachos F, Habel J (Eds) Biodiversity hotspots. Springer, Berlin, Heidelberg, 123–147. https://doi.org/10.1007/978-3-642-20992-5_7

Hisar O, Yanik T, Aras-Hisar S (2003) Hatchery and growth performance of two trout pure breeds, *Salvelinus alpinus* and *Salmo trutta fario*, and their hybrid. Israeli Journal of Aquaculture - Bamidgeh 55: 154–159. https://doi.org/10.46989/001c.20357

Hosmer Jr DW, Lemeshow S, Sturdivant RX (2013) Applied logistic regression. 3rd edn., John Wiley & Sons, UK, 511 pp. https://doi.org/10.1002/9781118548387

Hughes KA, Pescott OL, Peyton J, Adriaens T, Cottier-Cook EJ, Key G, Rabitsch W, Tricarico E, Barnes DKA, Baxter N, Belchier M, Blake D, Convey P, Dawson W, Frohlich D, Gardiner LM, González-Moreno P, James R, Malumphy C, Martin S, Martinou AF, Minchin D, Monaco A, Moore N, Morley SA, Ross K, Shanklin J, Turvey K, Vaughan D, Vaux AGC, Werenkraut V, Winfield IJ, Roy HE (2020) Invasive non-native species likely to threaten biodiversity and ecosystems in the Antarctic Peninsula region. Global Change Biology 26(4): 2702–2716. https://doi.org/10.1111/gcb.14938

Interesova E, Vilizzi L, Copp GH (2020) Risk screening of the potential invasiveness of non-native freshwater fishes in the River Ob basin (West Siberian Plain, Russia). Regional Environmental Change 20(2): 1–10. https://doi.org/10.1007/s10113-020-01644-3

Jelić D, Špelić I, Žutinić P (2016) Introduced species community over-dominates endemic ichthyofauna of high Lika plateau (Central Croatia) over a 100 year period. Acta Zoologica Academiae Scientiarum Hungaricae 62(2): 191–216. https://doi.org/10.17109/AZH.62.2.191.2016

Joy MK, Foote KJ, McNie P, Piria M (2019) The decline of New Zealand's freshwater fish fauna; the influence of land-use. Marine and Freshwater Research 70(1): 114–124. https://doi.org/10.1071/MF18028

Juncos R, Beauchamp DA, Vigliano PH (2013) Modeling prey consumption by native and nonnative piscivorous fishes: Implications for competition and impacts on shared prey in an ultraoligotrophic lake in Patagonia. Transactions of the American Fisheries Society 142(1): 268–281. https://doi.org/10.1080/00028487.2012.730109

Kalayci G, Ozturk RC, Capkin E, Altinok I (2018) Genetic and molecular evidence that brown trout *Salmo trutta* belonging to the Danubian lineage are a single biological species. Journal of Fish Biology 93(5): 792–804. https://doi.org/10.1111/jfb.13777

Kanjuh T, Marić A, Piria M, Špelić I, Maguire I, Simonović P (2020) Diversity of brown trout, *Salmo trutta* (Actinopterygii: Salmoniformes: Salmonidae), in the Danube River basin of Croatia revealed by mitochondrial DNA. Acta Ichthyologica et Piscatoria 50(3): 291–300. https://doi.org/10.3750/AIEP/02939

Kanjuh T, Tomić T, Marić A, Škraba Jurlina D, Nikolić V, Simonović P (2021) Trout *Salmo* spp. (Salmoniformes: Salmonidae) molecular diversity in streams on the southern slopes of the Stara Planina Mts. in Serbia. Acta Zoologica Bulgarica 73: 425–429. http://acta-zoologica-bulgarica.eu/2021/002487.pdf

Kapetanović D, Vardić I, Valić D, Teskeredžić E (2010) Furunculosis in cultured Arctic charr (*Salvelinus alpinus*) in Croatia. Aquaculture Research 41: e719–e721. https://doi.org/10.1111/j.1365-2109.2010.02589.x

Karjalainen J, Keskinen T, Pulkkanen M, Marjomaki TJ (2015) Climate change alters the egg development dynamics in cold-water adapted coregonids. Environmental Biology of Fishes 98(4): 979–991. https://doi.org/10.1007/s10641-014-0331-y

Karjalainen J, Jokinen L, Keskinen T, Marjomaki TJ (2016) Environmental and genetic effects on larval hatching time in two coregonids. Hydrobiologia 780(1): 135–143. https://doi.org/10.1007/s10750-016-2807-6

Karleuša B, Rubinić J, Radišić M, Krvavica N (2018) Analysis of climate change impact on water supply in Northern Istria (Croatia). Technical Gazette 25(Suppl. 2): 366–374. https://doi.org/10.17559/TV-20170809140304

Koutsikos N, Zogaris S, Vardakas L, Tachos V, Kalogianni E, Sanda R, Chatzinikolaou Y, Giakoumi S, Economidis PS, Economou AN (2012) Recent contributions to the distribution of the freshwater ichthyofauna in Greece. Mediterranean Marine Science 13(2): 268–277. https://doi.org/10.12681/mms.308

Koutsikos N, Vardakas L, Zogaris S, Perdikaris C, Kalantzi OI, Economou AN (2019) Does rainbow trout justify its high rank among alien invasive species? Insights from a nationwide survey in Greece. Aquatic Conservation 29(3): 409–423. https://doi.org/10.1002/aqc.3025

Krkošek M, Ford JS, Morton A, Lele S, Myers RA, Lewis MA (2007) Declining wild salmon populations in relation to parasites from farm salmon. Science 318(5857): 1772–1773. https://doi.org/10.1126/science.1148744

Latiu C, Cocan D, Uiuiu P, Ihut A, Nicula SA, Constantinescu R, Mireșan V (2020) The Black Sea trout, *Salmo labrax* Pallas, 1814 (Pisces: Salmonidae) in Romanian waters. Bulletin of University of Agricultural Sciences and Veterinary Medicine Cluj-Napoca. Animal Science and Biotechnologies 77(2): 9–19. https://doi.org/10.15835/buasvmcn-asb:2020.0017

Leitwein M, Garza JC, Pearse DE (2017) Ancestry and adaptive evolution of anadromous, resident, and adfluvial rainbow trout (*Oncorhynchus mykiss*) in the San Francisco bay area: Application of adaptive genomic variation to conservation in a highly impacted landscape. Evolutionary Applications 10(1): 56–67. https://doi.org/10.1111/eva.12416

Lenhardt M, Marković G, Hegediš A, Maletin S, Ćirković M, Marković Z (2011) Non-native and translocated fish species in Serbia and their impact on the native ichthyofauna. Reviews in Fish Biology and Fisheries 21(3): 407–421. https://doi.org/10.1007/s11160-010-9180-8

Little AG, Loughland I, Seebacher F (2020) What do warming waters mean for fish physiology and fisheries? Journal of Fish Biology 97(2): 328–340. https://doi.org/10.1111/jfb.14402

MacCrimmon HR (1971) World distribution of rainbow trout (*Salmo gairdneri*). Journal of the Fisheries Board of Canada 28(5): 663–704. https://doi.org/10.1139/f71-098

Makhrov AA, Lajus DL (2018) Postglacial colonization of the North European seas by Pacific fishes and lamprey. Contemporary Problems of Ecology 11(3): 247–258. https://doi.org/10.1134/S1995425518030071

McIntosh AR, Townsend CR (1995) Contrasting predation risks presented by introduced brown trout and native common river galaxias in New Zealand streams. Canadian Journal of Fisheries and Aquatic Sciences 52(9): 1821–1833. https://doi.org/10.1139/f95-175

McIntosh AR, Crowl TA, Townsend CR (1994) Size-related impacts of introduced brown trout on the distribution of native common river galaxias. New Zealand Journal of Marine and Freshwater Research 28(2): 135–144. https://doi.org/10.1080/00288330.1994.9516602

Mihinjač T, Sučić I, Špelić I, Vucić M, Ješovnik A (2019) Non-native freshwater fish species in Croatia. Ministarstvo okoliša i energetike Udruga Hyla, 102 pp. [in Croatian]

Ministry of Agriculture (2020) Implementation plan of the aquaculture sector transformation strategy 2020–2030. https://poljoprivreda.gov.hr/UserDocsImages/dokumenti/novosti/Nacrt_strategije_razvoja_akvakulture_2020_2030_.pdf [in Croatian]

Moghaddas SD, Abdoli A, Kiabi BH, Rahmani H, Vilizzi L, Copp GH (2021) Identifying invasive fish species threats to RAMSAR wetland sites in the Caspian Sea region—A case study of the Anzali Wetland Complex (Iran). Fisheries Management and Ecology 28(1): 28–39. https://doi.org/10.1111/fme.12453

Morbey Ye, Addison P, Shuter BJ, Vascotto K (2006) Within-population heterogeneity of habitat use by lake trout *Salvelinus namaycush.* Journal of Fish Biology 69(6): 1675–1696. https://doi.org/10.1111/j.1095-8649.2006.01236.x

Morgan IJ, McDonald DG, Wood CM (2001) The cost of living for freshwater fish in a warmer, more polluted world. Global Change Biology 7(4): 345–355. https://doi.org/10.1046/j.1365-2486.2001.00424.x

Mrdak D, Nikolić V, Tošić A, Simonović P (2012) Molecular and ecological features of the soft-muzzled trout *Salmo obtusirostris* (Heckel, 1852) in the Zeta River, Montenegro. Biologia 67(1): 222–233. https://doi.org/10.2478/s11756-011-0150-y

Mršić V (1935) Experiences with domestication of rainbow trout in Yugoslavia. Ribarski vijesnik 35: 5–7. [in Croatian]

Muhamedagić S, Habibović E (2013) The state and perspective of Danube huchen (*Hucho hucho*) in Bosnia and Herzegovina. Archives of Polish Fisheries 22(3): 155–160. https://doi.org/10.2478/aopf-2013-0012

Muir AM, Vecsei P, Pratt TC, Krueger CC, Power M, Reist JD (2013) Ontogenetic shifts in morphology and resource use of cisco *Coregonus artedi.* Journal of Fish Biology 82(2): 600–617. https://doi.org/10.1111/jfb.12016

Official Gazette (2005) Fisheries Law [Serbia]. http://demo.paragraf.rs/WebParagrafDemo/ZAKON-O-RIBARSTVU-Sl.-glasnik-RS,-br.-35-94,-38-94-ispr.-i-101-2005-djr.-zakon.htm

Official Gazette (2018) The law of protection and sustainable use of fish fund. https://www.pravno-informacioni-sistem.rs/SlGlasnikPortal/eli/rep/sgrs/skupstina/zakon/2014/128/2/reg [Serbian]

Oikonomou A, Leprieur F, Leonardos ID (2014) Biogeography of freshwater fishes of the Balkan Peninsula. Hydrobiologia 738(1): 205–220. https://doi.org/10.1007/s10750-014-1930-5

Orizaola G, Braña F (2006) Effect of salmonid introduction and other environmental characteristics on amphibian distribution and abundance in mountain lakes of northern Spain. Animal Conservation 9(2): 171–178. https://doi.org/10.1111/j.1469-1795.2006.00023.x

Peters L, Spatharis S, Dario MA, Dwyer T, Roca IJ, Kintner A, Kanstad-Hanssen Ø, Llewellyn MS, Praebel K (2018) Environmental DNA: A new low-cost monitoring tool for pathogens in salmonid aquaculture. Frontiers in Microbiology 9: e3009. https://doi.org/10.3389/fmicb.2018.03009

Pinter K, Epifanio J, Unfer G (2019) Release of hatchery-reared brown trout (*Salmo trutta*) as a threat to wild populations? A case study from Austria. Fisheries Research 219: e105296. https://doi.org/10.1016/j.fishres.2019.05.013

Piria M, Povž M, Vilizzi L, Zanella D, Simonović P, Copp GH (2016) Risk screening of non-native freshwater fishes in Croatia and Slovenia using the Fish Invasiveness Screening Kit. Fisheries Management and Ecology 23(1): 21–31. https://doi.org/10.1111/fme.12147

Piria M, Copp GH, Dick JT, Duplić A, Groom Q, Jelić D, Lucy FE, Roy HE, Sarat E, Simonović P, Tomljanović T, Tricarico E, Weinlander M, Adámek Z, Bedolfe S, Coughlan NE, Davis E, Dobrzycka-Krahel A, Grgić Z, Kırankaya ŞG, Ekmekçi FG, Lajtner J, Lukas JAY, Koutsikos N, Mennen GJ, Mitić B, Pastorino P, Ruokonen TJ, Skóra ME, Smith ERC, Šprem N, Tarkan AS, Treer T, Vardakas L, Vehanen T, Vilizzi L, Zanella D, Caffrey JM (2017) Tackling invasive alien species in Europe II: Threats and opportunities until 2020. Management of Biological Invasions 8(3): 273–286. https://doi.org/10.3391/mbi.2017.8.3.02

Piria M, Simonović P, Kalogianni E, Vardakas L, Koutsikos N, Zanella D, Ristovska M, Apostolou A, Adrović A, Mrdak D, Tarkan AS, Milošević D, Zanella LN, Bakiu R, Ekmekçi FG, Povž M, Korro K, Nikolić V, Škrijelj R, Kostov V, Gregori A, Joy MK (2018) Alien freshwater fish species in the Balkans – Vectors and pathways of introduction. Fish and Fisheries 19(1): 138–169. https://doi.org/10.1111/faf.12242

Piria M, Špelić I, Rezić A, Šprem N (2020) Morphological traits and condition of brown trout *Salmo trutta* from Žumberak and Samobor mountain streams. Journal of Central European Agriculture 21: 231–245. https://doi.org/10.5513/JCEA01/21.2.2460

Piria M, Stroil BK, Giannetto D, Tarkan AS, Gavrilović A, Špelić I, Radočaj T, Killi N, Filiz H, Uysal TU, Aldemir C, Kamberi E, Hala E, Bakiu R, Kolitari J, Buda E, Durmishaj Bakiu S, Sadiku E, Bakrač A, Mujić E, Avdić S, Doumpas N, Giovos I, Dinoshi I, Ušanović L, Kalajdžić A, Pešić A, Ćetković I, Marković O, Milošević D, Mrdak D, Sarà G, Bosch Belmar M, Marchessaux G, Trajanovski S, Zdraveski K (2021a) An assessment of regulation, education practices and socio-economic perceptions of non-native aquatic species in the Balkans. Journal of Vertebrate Biology 70(4): e21047. https://doi.org/10.25225/jvb.21047

Piria M, Radočaj T, Špelić I, Vilizzi L (2021b) The truth behind the fish diversity of the Lika River and its tributaries: are all management efforts worth it? In: 4th Croatian symposium on invasive species with International Participation, Book of Abstracts, 29–30 November 2021, Zagreb, Croatia, 1–38.

Plumb JM, Blanchfield PJ (2009) Performance of temperature and dissolved oxygen criteria to predict habitat use by lake trout (*Salvelinus namaycush*). Canadian Journal of Fisheries and Aquatic Sciences 66(11): 2011–2023. https://doi.org/10.1139/F09-129

Plumb JM, Blanchfield PJ, Abrahams MV (2014) A dynamic-bioenergetics model to assess depth selection and reproductive growth by lake trout (*Salvelinus namaycush*). Oecologia 175(2): 549–563. https://doi.org/10.1007/s00442-014-2934-6

Pofuk M (2021) Non-Indigenous parasites of fish in inland waters of Croatia. Croatian Journal of Fisheries: Ribarstvo 79(4): 187–204. https://doi.org/10.2478/cjf-2021-0020

Pofuk M, Zanella D, Piria M (2017) An overview of the translocated native and non-native fish species in Croatia: pathways, impacts and management. Management of Biological invasions 8: 425–435. https://doi.org/10.3391/mbi.2017.8.3.16

Pyšek P, Hulme PE, Simberloff D, Bacher S, Blackburn TM, Carlton JT, Dawson W, Essl F, Foxcroft LC, Genovesi P, Jeschke JM, Kühn I, Liebhold AM, Mandrak NE, Meyerson LA, Pauchard A, Pergl J, Roy HE, Seebens H, van Kleunen M, Vilà M, Wingfield MJ, Richardson DM (2020) Scientists' warning on invasive alien species. Biological Reviews of the Cambridge Philosophical Society 95(6): 1511–1534. https://doi.org/10.1111/brv.12627

R Core Team (2021) R: A language and environment for statistical computing. Vienna,Austria: R Foundation for Statistical Computing. https://www.r-project.org/

Radočaj T, Špelić I, Vilizzi L, Povž M, Piria M (2021) Identifying threats from introduced and translocated non-native freshwater fishes in Croatia and Slovenia under current and future climatic conditions. Global Ecology and Conservation 27: e01520. https://doi.org/10.1016/j.gecco.2021.e01520

Robin X, Turck N, Hainard A, Tiberti N, Lisacek F, Sanchez J-C, Müller M (2011) pROC: An open-source package for R and S+ to analyze and compare ROC curves. BMC Bioinformatics 12(1): e77. https://doi.org/10.1186/1471-2105-12-77

Rosenberg AA (2008) The price of lice. Nature 451(7174): 23–24. https://doi.org/10.1038/451023a

Rubel F, Brugger K, Haslinger K, Auer I (2017) The climate of the European Alps: Shift of very high resolution Köppen-Geiger climate zones 1800–2100. Meteorologische Zeitschrift (Berlin) 26(2): 115–125. https://doi.org/10.1127/metz/2016/0816

Schindler DW (2001) The cumulative effects of climate warming and other human stresses on Canadian freshwaters in the new millennium. Canadian Journal of Fisheries and Aquatic Sciences 58(1): 18–29. https://doi.org/10.1139/f00-179

Simonović P, Tošić A, Vassilev M, Apostolou A, Mrdak D, Ristovska M, Kostov V, Nikolić V, Škraba D, Vilizzi L, Copp GH (2013) Risk identification of non-native freshwater fishes in four countries of the Balkans Region using FISK. Mediterranean Marine Science 14: 369–376. https://doi.org/10.12681/mms.337

Simonović P, Tošić A, Škraba D, Mrdak D, Grujić S, Nikolić V (2014) Effects of stocking with brood fish to manage resident stream dwelling brown trout *Salmo* cf. *trutta* L. stock. Journal of Fisheries Sciences 8: 139–152. https://doi.org/10.3153/jfscom.201418

Simonović P, Vidović Z, Tošić A, Škraba D, Čanak-Atlagić J, Nikolić V (2015) Risks to stocks of native trout of the genus *Salmo* (Actinopterygii: Salmoniformes: Salmonidae) of Serbia and management for their recovery. Acta Ichthyologica et Piscatoria 45(2): 161–173. https://doi.org/10.3750/AIP2015.45.2.06

Simonović P, Tošić A, Škraba Jurlina D, Nikolić V, Piria M, Tomljanović T, Šprem N, Mrdak D, Milošević D, Bećiraj A, Dekić R, Povž M (2017) Diversity of brown trout *Salmo* cf. *trutta* in the River Danube basin of western Balkans as assessed from the structure of their mitochondrial control region haplotypes. Journal of Ichthyology 57(4): 603–616. https://doi.org/10.1134/S0032945217040154

Skelton PH (1987) South African Red Data Book. Fishes. South African National Scientific Programmes Report 137. Council for Scientific and Industrial Research, Pretoria.

Škraba D, Bećiraj A, Šarić I, Ićanović I, Džaferović A, Piria M, Dekić R, Tošić A, Nikolić V, Simonović P (2017) Genotypization of brown trout (*Salmo trutta* L.) populations from River Una drainage area in Bosnia and Herzegovina and implications for conservation and fishery management. Acta Zoologica Bulgarica 69: 25–30.

Škraba Jurlina D, Marić A, Mrdak D, Kanjuh T, Špelić I, Nikolić V, Piria M, Simonović P (2020) Alternative life-history in native trout (*Salmo* spp.) suppresses the invasive effect of alien trout strains introduced into streams in the western part of the Balkans. Frontiers in Ecology and Evolution 8: e188. https://doi.org/10.3389/fevo.2020.00188

Spens J, Alanärä A, Eriksson LO (2007) Nonnative brook trout (*Salvelinus fontinalis*) and the demise of native brown trout (*Salmo trutta*) in northern boreal lakes: Stealthy, long-term patterns? Canadian Journal of Fisheries and Aquatic Sciences 64(4): 654–664. https://doi.org/10.1139/f07-040

Stagl J, Hattermann F (2016) Impacts of climate change on riverine ecosystems: Alterations of ecologically relevant flow dynamics in the Danube River and its major tributaries. Water (Basel) 8(12): e566. https://doi.org/10.3390/w8120566

Stanković D, Crivelli AJ, Snoj A (2015) Rainbow trout in Europe: Introduction, naturalization, and impacts. Reviews in Fisheries Science & Aquaculture 23(1): 39–71. https://doi.org/10.1080/23308249.2015.1024825

Stoumboudi MT, Barbieri R, Kalogianni E (2017) First report of an established population of *Oncorhynchus mykiss* (Walbaum, 1792) (Salmonidae) on the Island of Crete, Greece. Acta Zoologica Bulgarica 9: 99–104.

Sušnik S, Weiss S, Odak T, Delling B, Treer T, Snoj A (2007) Reticulate evolution: ancient introgression of the Adriatic brown trout mtDNA in softmouth trout *Salmo obtusirostris* (Teleostei: Salmonidae). Biological Journal of the Linnean Society 90(1): 139–152. https://doi.org/10.1111/j.1095-8312.2007.00717.x

Tarkan AS, Vilizzi L, Top N, Ekmekçi FG, Stebbing PD, Copp GH (2017) Identification of potentially invasive freshwater fishes, including translocated species, in Turkey using the Aquatic Species Invasiveness Screening Kit (AS-ISK). International Review of Hydrobiology 102(1–2): 47–56. https://doi.org/10.1002/iroh.201601877

Tarkan AS, Yoğurtçuoğlu B, Ekmekçi FG, Clarke SA, Wood LE, Vilizzi L, Copp GH (2020) First application in Turkey of the European Non-native Species in Aquaculture Risk Assessment Scheme to evaluate farmed non-native striped catfish *Pangasianodon hypophthalmus*. Fisheries Management and Ecology 27(2): 123–131. https://doi.org/10.1111/fme.12387

Tarkan AS, Emiroğlu Ö, Aksu S, Başkurt S, Aksu İ, Vilizzi L, Yoğurtçuoğlu B (2022) Coupling molecular analysis with risk assessment to investigate the origin, distribution and potential impact of non-native species: Application to ruffe *Gymnocephalus cernua* in Turkey. The European Zoological Journal 89: 102–114. https://doi.org/10.1080/24750263.2021.2022222

Tošić A, Škraba D, Nikolić V, Čanak Atlagić J, Mrdak D, Simonović P (2016) Haplotype diversity of brown trout in the broader Iron Gate area. Turkish Journal of Zoology 40: 655–662. https://doi.org/10.3906/zoo-1510-54

Trenberth KE (2011) Changes in precipitation with climate change. Climate Research 47(1): 123–138. https://doi.org/10.3354/cr00953

Vehanen T, Piria M, Kubečka J, Skov C, Kelly F, Pokki H, Eskelinen P, Rahikainen M, Keskinen T, Artell J, Romakkaniemi A, Suić J, Adámek Z, Heimlich R, Chalupa P, Ženíšková H, Lyach R, Berg S, Birnie-Gauvin K, Jepsen N, Koed A, Ingemann Pedersen M, Rasmussen G, Gargan P, Roche W (2020) Systems and methodologies of data collection in inland fisheries of Europe. FAO Fisheries and Aquaculture Technical Paper 649: 1–168. https://doi.org/10.4060/ca7993en

Ventura M, Tiberti R, Buchaca T, Bunay D, Sabas I, Miro A (2017) Why Should We Preserve Fishless High Mountain Lakes? In: Catalan J, Ninot JM, Aniz MM (Eds) High mountain conservation in a changing world. Advances in Global Change Research 62: 181–207. https://doi.org/10.1007/978-3-319-55982-7_8

Vergara IA, Norambuena T, Ferrada E, Slater AW, Melo F (2008) StAR: A simple tool for the statistical comparison of ROC curves. BMC Bioinformatics 9(1): e265. https://doi.org/10.1186/1471-2105-9-265

Vilà M, Basnou C, Pyšek P, Josefsson M, Genovesi P, Gollasch S, Nentwig W, Olenin S, Roques A, Roy D, Hulme PE, Partners DAISIE (2010) How well do we understand the impacts of alien species on ecosystem services? A pan-European, cross-taxa assessment. Frontiers in Ecology and the Environment 8(3): 135–144. https://doi.org/10.1890/080083

Vilizzi L, Hill JE, Piria M, Copp GH (2022) A protocol for screening potentially invasive non-native species using Weed Risk Assessment-type decision-support toolkits. Science of the Total Environment 832: e154966. https://doi.org/10.1016/j.scitotenv.2022.154966

Weiss S, Schenekar T (2016) Genetic evaluation of the self-sustaining status of a population of the endangered Danube salmon, *Hucho hucho.* Hydrobiologia 775(1): 153–165. https://doi.org/10.1007/s10750-016-2726-6

Wood LE, Guilder J, Brennan ML, Birland NJ, Taleti V, Stinton N, Taylor NGH, Thrush MA (2021) Biosecurity and the ornamental fish trade: A stakeholder perspective in England. Journal of Fish Biology 100(2): 352–365. https://doi.org/10.1111/jfb.14928

Yoğurtçuoğlu B, Bucak T, Ekmekçi FG, Kaya C, Tarkan AS (2021) Mapping the establishment and invasiveness potential of rainbow trout (*Oncorhynchus mykiss*) in Turkey: With special emphasis on the conservation of native salmonids. Frontiers in Ecology and Evolution 8: e599881. https://doi.org/10.3389/fevo.2020.599881

Young KA, Dunham JB, Stephenson JF, Terreau A, Thailly AF, Gajardo G, Garcia de Leaniz C (2010) A trial of two trouts: Comparing the impacts of rainbow and brown trout on a native galaxiid. Animal Conservation 13(4): 399–410. https://doi.org/10.1111/j.1469-1795.2010.00354.x

Zupančič P, Tisaj D, Lah Lj (2008) Rare and endangered freshwater fishes of Croatia, Slovenia and Bosnia and Herzegovina - Adriatic basin. Dolsko AZV, 79 pp.

Supplementary material 1

Combined AS-ISK report including the 68 screenings for the 17 salmonid species screened for the Danube and Adriatic basins of Bosnia and Herzegovina, Croatia, Montenegro and Serbia (including Kosovo)

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Consistency in impact assessments of invasive species is generally high and depends on protocols and impact types

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Abstract

Impact assessments can help prioritising limited resources for invasive species management. However, their usefulness to provide information for decision-making depends on their repeatability, i.e. the consistency of the estimated impact. Previous studies have provided important insights into the consistency of final scores and rankings. However, due to the criteria to summarise protocol responses into one value (e.g. maximum score observed) or to categorise those final scores into prioritisation levels, the real consistency at the answer level remains poorly understood. Here, we fill this gap by quantifying and comparing the consistency in the scores of protocol questions with inter-rater reliability metrics. We provide an overview of impact assessment consistency and the factors altering it, by evaluating 1,742 impact assessments of 60 terrestrial, freshwater and marine vertebrates, invertebrates and plants conducted with seven protocols applied in Europe (EICAT; EPPO; EPPO prioritisation; GABLIS; GB; GISS; and Harmonia+). Assessments include questions about diverse impact types: environment, biodiversity, native species interactions, hybridisation, economic losses and human health. Overall, the great majority of assessments (67%) showed high consistency; only a small minority (13%) presented low consistency. Consistency of responses did not depend on species identity or the amount of information on their impacts, but partly depended on the impact type evaluated and the protocol used, probably due to linguistic uncertainties (pseudo-R^2 = 0.11 and 0.10, respectively). Consistency of responses was highest for questions on ecosystem and human health impacts and lowest for questions regarding biological interactions amongst alien and native species. Regarding protocols, consistency was highest with Harmonia⁺ and GISS and lowest with EPPO. The presence of few, but very low, consistent assessments indicates that there is room for improvement in the repeatability of assessments. As no single factor explained largely the variance in consistency, low values can rely on multiple factors. We thus endorse previous studies calling for diverse and complementary actions, such as improving protocols and guidelines or consensus assessment to increase impact assessment repeatability. Nevertheless, we conclude that impact assessments were generally highly consistent and, therefore, useful in helping to prioritise resources against the continued relentless rise of invasive species.

Keywords
Alien species policy, biological invasions, ecological impact, epistemic uncertainty, inter-rater reliability, linguistic uncertainty, repeatability, socio-economic impact

Introduction

Invasive alien species are one of the greatest threats to biodiversity, economy and public health (Bellard et al. 2016; Mazza and Tricarico 2018; Diagne et al. 2020; Pyšek et al. 2020; Smith 2020). Concern about invasive species is growing due to the relentless increase in introductions and their spread, mostly associated with environmental change and increasing trade (Seebens et al. 2015, 2017; Chapman et al. 2017; Sardain et al. 2019). Although there are significant national and international efforts to reduce introductions, spreads and their impacts (Keller and Perrings 2011; Turbelin et al. 2017), human operational capacity to avert new invasions is limited (Genovesi and Shine 2004; Keller et al. 2007; Early et al. 2016). Thus, reliable tools to prioritise and underpin invasive species research, management and policy are required (Roberts et al. 2018; Booy et al. 2020). Under this urgent need, systematic semi-quantitative impact assessment protocols, based on available scientific evidence to rank and prioritise management of alien species are of paramount usefulness (e.g. Genovesi and Shine 2004; McGeoch et al. 2016; Vilà et al. 2019; Vilizzi et al. 2021).

The large number of protocols developed with similar objectives, as well as the substantial body of research comparing their outputs, shows the pivotal role of protocol choice in assessments (Glamuzina et al. 2017; Turbé et al. 2017; Vilà et al. 2019; Sohrabi et al. 2021). While this is important, there are also other crucial and more undervalued aspects in impact assessments. Previous studies have frequently illustrated the varying consistency of results when evaluating the same species with the same protocols (McGeoch et al. 2012; Almeida et al. 2013; Lawson et al. 2015; Turbé et al. 2017; González-Moreno et al. 2019; Vilizzi et al. 2019; Clarke et al. 2021; but see Volery et al. 2021). This finding raises doubts as to whether the choice of the evaluator can affect management prioritisations and, thus, whether risk assessments are reliable for providing information for decision-making. The fluctuating consistency is partly to be expected as assessments in the end rely on the judgement of experts (Regan et al. 2002; Burgman et al. 2011; McGeoch et al. 2012; Vanderhoeven et al. 2017) which depend on their experience, i.e. amount and bias of knowledge and their subjective interpretation of evidence (Kumschick et al. 2017; Dorrough et al. 2018; Bindewald et al. 2020; Clarke et al. 2021). Certainly, there have been advances to control this subjectiveness (e.g. refinement of guidelines and protocol questions, as well as peer review and consensus process; Hawkins et al. 2015; Matthews et al. 2017; Vanderhoeven et al. 2017; Dorrough et al. 2018; Volery et al. 2020). However, information on the overall severity and extent of consistency in responses is still missing. For instance, information on the factors underlying different degrees of consistency is mostly theoretical (Regan et al. 2002; Vanderhoeven et al. 2017; Latombe et al. 2019; Probert et al. 2020), while the limited empirical information focuses mainly on consistency in final scores and rankings (e.g. Perdikaris et al. 2016; González-Moreno et al. 2019). However, the protocol's criteria for synthesising scores (e.g. the mean or maximum value for choosing a final score) and the subjective threshold value for ranking species into different categories (Almeida et al. 2013; D'hondt et al. 2015; González-Moreno et al. 2019; Vilà et al. 2019) add unintuitive noise to the real consistency in answers. To date, studies focused on protocol questions are limited to a single taxon and a single protocol (e.g. Clarke et al. 2021; Volery et al. 2021). Thus, empirical information on the factors influencing the consistency across assessors remains poorly understood.

To fill both knowledge gaps, we addressed two objectives. Objective 1: To provide generalisable results on consistency in individual protocol questions, we evaluated consistency when assessing a wide range of taxa (invasive plants, vertebrates and invertebrates), as well as when using multiple protocols. We measured consistency in scores of protocol questions using inter-rater reliability metrics (Hallgren 2012; Gwet 2014) benefiting from one of the most comprehensive datasets on impact assessment of invasive species in Europe (described in González-Moreno et al. 2019). By exploring a wide range of taxa and protocols, our results will provide information for the overall reliability of impact assessments that support decision-making. Objective 2: To evaluate which factors may influence the consistency of responses, we evaluated the relationship between the consistency and the protocol choice, impact type (e.g. environmental, socio-economic), taxonomic group, species identity and the amount of scientific literature available about species impacts. The evaluation of these factors, except for protocols, aims to answer if consistency varies due to epistemic uncertainties, such as if assessors had different knowledge about impacts or responded with greater subjectivity (e.g. due to bias, limited or inconsistent knowledge; McGeoch et al. 2012, 2016; Kumschick et al. 2017). The evaluation of protocol choice aims to detect if consistency is associated with protocol properties (e.g. number of questions per protocol and of responses per question) or with linguistic uncertainties (e.g. clarity or vagueness of the questions). For details on epistemic and linguistic uncertainties, see Regan et al. (2002), Leung et al. (2012), Latombe et al. (2019) and Probert et al. (2020). Altogether, these results can form the basis of future studies to provide information for the design or update of impact assessment protocols for invasive species.

Materials and methods

Assessors, species and impact assessment protocols

Within the Alien Challenge COST Action, 78 assessors with variable experience in biological invasions (PhD or PhD candidates; hereafter assessors) evaluated 60 invasive species with seven different risk assessment protocols (hereafter protocols) to provide information about the agreement of scores in protocols (González-Moreno et al. 2019). In total, we used 1,742 of those impact assessments.

Assessors were grouped according to their taxonomic expertise, under the coordination of a taxonomic leader. Assessors selected by consensus a list of 60 invasive species that covered a wide range of habitat types and biological characteristics: terrestrial plants (n = 10), freshwater plants (5), terrestrial vertebrates (10), terrestrial insects (13), other terrestrial invertebrates (4), freshwater invertebrates (6), freshwater fish (3), marine invertebrates (6) and marine vertebrates (3). See details in Suppl. material 1: Table S1. In our analyses, we focused on the level of species and the three higher taxonomic groups: vertebrates (n = 29 species), invertebrates (16) and plants (15).

Each assessor scored a minimum of three and a maximum of nine species (median = 3) and each species was assessed by a minimum of three and a maximum of eight evaluators (median = 5). Not all assessors evaluated all species of their expertise group; thus, the study design was neither crossed nor nested, an important point in understanding how to measure consistency (see below).

The seven protocols used were developed or applied in Europe: European Plant Protection Organisation-Environmental Impact Assessment for plants (EPPO Brunel et al. 2010); EPPO-Prioritisation scheme (EPPO-Prioritisation; Kenis et al. 2012); German-Austrian Black List Information System (GABLIS; Essl et al. 2011); Great Britain Non-native Species Risk Assessment (GB-NNRA; Baker et al. 2008; Mumford et al. 2010); Generic Impact Scoring System (GISS; Nentwig et al. 2010, 2016); Belgian risk screening tools for potentially invasive plants and animals (Harmonia^{+}; D'hondt et al. 2015) and Environmental Impact Classification of Alien Taxa (called at that time GISS IUCN and now EICAT; Blackburn et al. 2014). The selection of protocols does not consider updates that have become available after 2015 (e.g. for EICAT Volery et al. 2020). For details on protocols and the template used, see González-Moreno et al. (2019).

Before filling the spreadsheets, the assessors read the protocol guidelines and asked questions directly to the protocol developers, if needed. To conduct the assessments, experts decided on their own sources of information (i.e. scientific literature, own expertise or alternative sources). The assessors considered Europe as the risk assessment area. We provided the scores provided by each assessor in each impact assessments, i.e. combination of protocol and species, in Suppl. material 2 as an R list object called "list_impact_assessments.RData".

Table 1. Number of questions regarding different types of impacts of invasive species considered by the seven impact assessment protocols considered. Range of levels indicates the minimum and maximum number of available responses for each question of a given protocol. P-V-I = number of plant, vertebrate and invertebrate species evaluated with each protocol. See the questions and their classification in Suppl. material 1: Table S2. For details on protocols, see González-Moreno et al. (2019).

Protocol	Ecosystem	Biodiversity	Species interaction	Hybridi-sation	Economic losses	Human health	Range of levels	P-V-I
EICAT	2	3	4	1	1	0	5-5	15-16-29
EPPO	4	2	1	1	0	0	3-3	15-0-0
EPPO-Prioritisation	1	1	1	0	2	1	3-3	15-0-0
GABLIS	1	2	2	1	1	1	3-4	15-16-29
GB-NNRA	3	2	2	2	1	1	5-5	15-16-29
GISS	1	2	3	1	5	1	6-6	15-16-29
Harmonia+	1	4	6	2	5	3	3-6	15-16-29

Classification of impact types

Even if some protocols assessed all four components of the invasion process: introduction, establishment, spread and impacts, we only evaluated the latter. To evaluate whether consistency in responses systematically varies across impact types, we grouped the questions into six categories: ecosystem processes, biodiversity, species interactions, hybridisation with native species, economic losses and human health (Table 1 and Suppl. material 1: Table S2). These impacts were further grouped into two coarse impact types: environmental (i.e. biodiversity, species interaction, hybridisation, ecosystems) and socio-economic (i.e. economic losses and human heath).

Quantifying consistency

We measured the consistency of responses across assessors with inter-rater reliability metrics, which quantify the proportion of the variance in the scores associated with assessors (Furr 2021). The values range from 0 to 1 intuitively indicating a low or high consistency in the responses, respectively. For instance, a value of 0.8 would indicate that 20% of the variance observed is due to assessor choice (Hallgren 2012). See an overview on inter-rater reliability metrics provided by Hallgren (2012) and Gwet (2014).

Estimation of inter-rater reliability metrics is influenced by the structure of the data (i.e. which assessors evaluated which species; Putka et al. 2008; Koo and Li 2016). As our study design was neither crossed nor nested, we used the coefficient G (Putka et al. 2008). This coefficient G is based on generalisability theory (G-theory; Brennan 2001; Putka et al. 2008), which is focused on disentangling the sources of error using analyses of variance methods (Brennan 2001). To calculate the coefficient G, we first require estimating the variance associated with raters (e.g. assessors) and ratees (e.g. protocol questions). We did it with a mixed model using the identities of the raters and ratees as random variables (Putka et al. 2008). To address our objectives, we calculated two types of coefficient G: one for the consistency of assessors scoring each question of a protocol for a given species, i.e. the overall consistency in an impact assessment (hereafter $G_{Prot-Spp}$) and a second coefficient G for the consistency of assessors scoring a given impact (i.e. protocol question) across all species of a given taxonomic group (hereafter $G_{Quest-Taxon}$). We differentiated between taxonomic groups, because impact knowledge may vary across them. We used the coefficient $G_{Prot-Spp}$ to provide information on the general consistency in a particular impact assessment (Objective 1), as well as to disentangle the effect of species identity, taxonomic groups and amount of published scientific articles on species impacts in consistency (Objective 2). We used $G_{Quest-Taxon}$ to disentangle the effect of impact types (Objective 2). In addition, we used both $G_{Prot-Spp}$ and $G_{Quest-Taxon}$ in complementary analyses to unravel whether the influence of protocols relies on methodological aspects, such as the number of questions per protocol and of available answers per question or whether the variability could be potentially more associated with linguistic uncertainties (Objective 2). See Table 2 for details.

In the following sections, we explain the calculations of the coefficient $G_{Prot-Spp}$ and $G_{Quest-Taxon}$. We advance that some mixed models to estimate the variance associated with raters and ratees had convergence issues (e.g. identifiability and singularity) and failed to calculate some coefficients G. We also explain in different sections the methodological approximations to disentangle the influence of each factor on consistency of scores.

Calculation of coefficient $G_{Prot-Spp}$

We calculated a $G_{Prot-Spp}$ for each combination of protocol and species (i.e. an impact assessment). A way to visualise the data required is a two-dimension array, where the columns are the assessors evaluating a given species, the rows the impact questions of a given protocol and the values within the matrix the scores estimated. For each array, we performed a mixed model to extract the variance associated with the assessors and the protocol questions (Putka et al. 2008; see Table 2). Second, following Putka et al. (2008), we used the estimated variances to calculate the coefficient G. See mathematical details of the coefficient G in Putka et al. 2008 and our R code (Suppl. material 2).

Table 2. Interpretation and use of $G_{Prot-Spp}$ and $G_{Quest-Taxon}$. Linear mixed models = formulation used to estimate the variances required for the calculation of the coefficients G. The formulation is the one used to run the models with the R function *lmer* of the R package *lme4*.

Metric	Interpretation	Linear mixed models	Use
$G_{Prot-Spp}$	Level of agreement in each impact assessment. (Protocol-Species combination).	*Scores ~ (1|ID Question) + (1|ID assessor)*	Objective 1: To quantify the general consistency of assessors in impact assessments. Objective 2: To evaluate if the consistency varies with the taxonomic group or species evaluated, the amount of published information on species impacts and the protocol choice or the number of questions per protocol.
$G_{Quest-Taxon}$	Level of agreement in each question of a given protocol. (Question-Taxonomic group combination)	*Scores ~ (1|ID Species) + (1|ID assessor)*	Objective 2: To evaluate if the consistency varies with the impact types and the number of available responses per protocol question.

In calculating the $G_{Prot\text{-}Spp}$ values of 330 combinations of species and protocols, we found convergence issues in the mixed models for 66 cases, reflecting in 65 cases of singular models. These issues were not systematically related to species (Chi-squared = 58.69, p-value = 0.52; Chi-squared test with Monte Carlo simulations), but were related to specific protocols (Chi-squared = 53.51, p-value < 0.001; specifically, to EPPO Priorisation and GABLIS protocols). We performed our subsequent analyses with the remaining 264 $G_{Prot\text{-}Spp}$ values. However, to ensure that excluding values from models with singularity issues had no effects on our inferences, we also evaluated differences in $G_{Prot\text{-}Spp}$ between taxonomic groups and protocols without removing the 65 values of the singular models (i.e. sensitivity analysis), which showed similar results.

Calculation of coefficient $G_{Quest\text{-}Taxon}$

We calculated $G_{Quest\text{-}Taxon}$ to evaluate the association between different impact types and levels of consistency. As consistency in answering the diverse impact types can vary across taxonomic groups, we calculated a $G_{Quest\text{-}Taxon}$ for each combination of taxonomic group, protocol and question of each protocol. A way to visualise the data required is a two-dimension array, where the columns are the assessors evaluating a given impact question for any species of a given taxonomic group, the rows, the species of a given taxonomic group and the values within the matrix, the scores estimated. Thus, for the same impact question, we have one to three databases depending on whether the impact can be applied to some or all taxonomic groups (i.e. plants, invertebrates and vertebrates; Table 1). For each array, we performed a linear mixed model to extract the variance associated with the assessors and species identity. Later, we used those variances to calculate $G_{Quest\text{-}Taxon}$ (Putka et al. 2008).

In calculating the $G_{Quest\text{-}Taxon}$ values of the 188 combinations of taxonomic groups, protocols and questions, we found convergence issues in the mixed models for 22 cases. These issues were not systematically associated with protocols (Chi-squared = 5.78, p-value = 0.45), neither impact types (six impact types: Chi-squared = 3.21, p-value = 0.65; two higher impact types: Chi-squared = 0.25, p-value = 0.70). As there was no systematic removal of protocols or impact types, unlike $G_{Prot\text{-}Spp}$, we did not perform sensitivity analyses including the values with warnings about singularity. We performed our subsequent analyses with the remaining 166 $G_{Prot\text{-}Quest}$ values: 64 on plant impacts, 59 on invertebrate impacts and 43 on vertebrate impacts.

Generality and extent of consistency in impact assessments

To interpret $G_{Prot\text{-}Spp}$ values, we classified them into three decision-meaningful categories: low, medium and high consistency in impact assessments. We followed Krippendorff (1980), who considered that impact assessments should be discarded for decision-making if G values were lower than 0.67, impact assessments can tentatively be used for decision-making if G values were between 0.67 and 0.80 and impact assessments can definitively be used for decision-making if G values were above 0.80 (low, medium and high, respectively). To provide information on the general consistency in impact assessments, we discussed the relative frequency of these three categories.

Species

Testing for differences in the consistency of scores between species is challenging due to the relative low amount of protocols and, thus, of $G_{Prot-Spp}$ values per species. The number of available protocols for each vertebrate and invertebrate species is five and seven for plant species (Table 1). Moreover, for some species, the number of $G_{Prot-Spp}$ values was lower due to convergence issues (see Table S3 for $G_{Prot-Spp}$ values estimated). Therefore, we conducted two complementary approximations to test expectations of the influence of species from different perspectives. We called these analyses: permutation test and descriptive analysis. The permutation test is a statistical analyses focused on the proportion of low consistent assessments, while the descriptive analyses is focused on the distribution of raw values.

In the permutation test, we statistically tested if low consistent assessments were associated with few specific species. If true, the number of observed species with a large proportion of low consistent assessments ($G_{Prot-Spp}$ < 0.67) should be lower than those expected by chance. We focused on the proportion of low consistent assessments, instead of using the correlation with all $G_{Prot-Spp}$ values, since that is the subset challenging the reliability and usefulness of impact assessments. To test it, we performed 1,000 permutations swapping the $G_{Prot-Spp}$ between species and protocols at random but maintaining the number of $G_{Prot-Spp}$ values per species and protocol. We later compared, between the observed data and permuted data, the frequency of species with a proportion above 50% of low consistent assessments ($G_{Prot-Spp}$ < 0.67). We looked for statistical differences using the unconditional Boschloo's test with the function *exact.test* of the R package *Exact* (Calhoun 2021). We performed inferences, based on the distribution of the 1,000 p-values. To ensure that our results did not depend on thresholds when calculating the frequency of species with low consistent assessments, we also used the thresholds 30 and 40% to calculate the proportions of low consistent assessments. When sample size is reduced, small variations in the frequency of events have important effects on proportions. We, therefore, conducted the permutation tests with those species with four or more assessments.

In the descriptive analysis, we visually assessed the mean and standard deviations of $G_{Prot-Spp}$ across species. If consistency depends on species identity, we expect to observe species with different means and non-overlapping standard deviations. Complementarily, large standard deviations (> 0.20), reflecting that the consistency in impact assessments for a same species are in different categories (low, medium and high), support the influence of factors associated with the protocols (e.g. linguistic differences or impact types asked). See Suppl. material 1: Table S4 for a summary of the goals and expectations of all analysis (Permutation test = Target 1; Descriptive analyses = Target 2 in Suppl. material 1: Table S4).

Amount of information available on species impacts

We examined the relationship between the proportions of assessments with low consistency per species ($G_{Prot-Spp}$ < 0.67) with the number of scientific articles on impacts per species recorded in the Web of Science (hereafter correlation test). We expected that the number of articles per species reflects the amount and diversity of knowledge on species impacts and should, therefore, correlate negatively with the proportion of assessments with low consistency (Target 3 in Suppl. material 1: Table S4). We used a generalised linear model using the Poisson family with the R package (Bates et al. 2015; Wickham et al. 2019; R Core Team 2021). To search the scientific articles, we used the advanced search of ISI Web of Science (11 July 2020). We used a query with three complementary sections. Two sections were fixed and indicated terms for searching (TS) papers about invasive species and their impacts, while the other section indicated synonyms of a species. See the following example: TS = ("*Cameraria oridella*" OR "*Cameraria ohridella*") AND TS = ("Alien" OR "Invasive" OR "Non-native" OR "Non native" OR "Invasion") AND TS = ("Impact" OR "Damage" OR "Harm"). See Suppl. material 1: Table S5 for details on the searches of each species.

Taxonomic groups and protocols

To statistically test whether consistency in assessments varied across taxonomic groups and protocols, we modelled $G_{Prot\text{-}Spp}$ with beta regression models using the R package *glmmTMB* (Brooks et al. 2017; Targets 4 and 5 in Suppl. material 1: Table S4). We modelled both the mean and the precision in the models. While the mean refers to the effect we are interested in, the precision considers a variable dispersion along the explanatory variables (see details in Cribari-Neto and Zeileis 2009; Ferrari et al. 2011; Zhao et al. 2014). When modelling, we also considered that $G_{Prot\text{-}Spp}$ may be influenced by other factors beyond our interest, such as the number of assessors considered in the calculation of the inter-rater metric (Hallgren 2012). Although we did not include the number of protocol questions due to convergence issues, we performed additional analyses to explore the relationship between the variables protocol and number of questions per protocol (see below; Target 6 in Suppl. material 1: Table S4). To model the mean, we used models representing all combinations of three explanatory variables: the taxonomic group to which the species belongs, the protocol used in the impact assessment and the number of assessors who evaluated each species with each protocol. To model the precision, we included all combinations of two variables: the taxonomic group and protocol identity. Additionally, we controlled the non-independency of data by including in all models the species identity as a random intercept. In total, we performed 28 models. See all models in Suppl. material 1: Table S6. For our inferences, we considered the models with $\Delta AICc \leq 4$ (models with an AICc equal or lower than the minimum observed AICc plus 4). See Targets 4 and 5 in Suppl. material 1: Table S4 for a summary of the main and sensitivity analyses.

We interpreted that statistical differences between taxonomic groups reflect diverse epistemic uncertainties across taxa. In contrast, statistical differences between protocols may reflect linguistic uncertainties, but also three other factors: the number of questions per protocol, the number of responses per question or the impact types evaluated in each protocol. To discuss the origin of protocol variability, we jointly interpreted the results of these beta regression models with three complementary analyses: one focused on $G_{Prot\text{-}Spp}$ (number of questions in a protocol) and two focused on $G_{Quest\text{-}Taxon}$ (the number of responses in the questions and the impact type evaluated; see following sections; Targets 6, 8 and 9 in Suppl. material 1: Table S4). We considered that differences in consistency when using different protocols that are not explained by the number of questions, number of responses per question or the impact types, might support the influence of linguistic uncertainties. In the complementary analyses to quantify the influence of the number of questions per protocol, we repeated the previous 28 beta regression models (Suppl. material 1: Table S6), but exchanging the variable protocol for the variable number of impact questions per protocol. We later compared the marginal pseudo-R^2 associated with the variable protocol and number of questions per protocol (Target 6 in Suppl. material 1: Table S4).

Impact types

To evaluate the influence of impact type, we used $G_{Quest-Taxon}$, i.e. the metric providing information on the consistency when scoring a given protocol question across the species of a particular taxonomic group (Table 2). For each combination of protocol question and taxonomic taxon, we have its $G_{Quest-Taxon}$ value and its association with a detailed or coarse impact (see above; Suppl. material 1: Table S2). As some questions fell into several categories of impact types, we controlled this pseudo-replication in subsequent analyses (see below). In total, we analysed 76 $G_{Quest-Taxon}$ values for plants, 71 $G_{Quest-Taxon}$ values for invertebrates and 51 $G_{Prot-Quest}$ values for vertebrates.

We modelled $G_{Quest-Taxon}$ in relation to impact types and taxonomic groups to consider differences in the knowledge of impact types across taxonomic groups. In the analyses, we controlled four co-variables that can also affect $G_{Quest-Taxon}$ values: the number of species used to calculate $G_{Quest-Taxon}$, the number of assessors used to calculate $G_{Quest-Taxon}$, the protocol to which each question belongs and the specificity of the question (if it asked about one or more types of impact; binomial). In total, we used six variables to study variability in $G_{Quest-Taxon}$. The number of combinations of our four categorical variables were relatively large for our amount of data (166 $G_{Quest-Taxon}$ values for 252 combinations of levels; impact type = 6 levels; taxonomic group = 3; protocols = 7 and specificity = 2). To reduce overparametrisation, we conducted two nested models. First, we modelled the variance associated with the four co-variables (two categorical and two continuous variables; hereafter, first nested model). Later, we modelled its residuals with the impact type and taxonomic group (hereafter, second nested model). We avoided overparametrisation, but assigned to the co-variables any potential variance shared with our variables of interest. Therefore, the detected effect of the taxonomic group and impact types may be conservative.

These first nested models were beta regressions since $G_{Quest-Taxon}$ values ranges from 0 to 1. We modelled $G_{Quest-Taxon}$ with all combinations of the four co-variables, in the mean and precision parameter. We chose the best model, based on the corrected Akaike's Information Criterion approach (AICc; Target 10 in Suppl. material 1: Table S4). We then extracted its residuals and modelled them with the taxonomic group and impact types by using a linear mixed model. We explored five models: a) interactive effects of the impact types and taxonomic groups; b) additive effects of the impact types and taxonomic groups; c) single effect of impact types; d) single effect of taxonomic groups; and e) null model with just the intercept (Target 11 in Suppl. material 1: Table S4). Since the same question can be answered for multiple taxonomic groups (Table 1), we also included the identity of the question as a random effect. For our inferences, we considered the models within the ΔAICc ≤ 4. We obtained the proportion of the variance in $G_{Quest\text{-}Taxon}$ explained by the explanatory variables by applying the function *summary* of the R package *base* to the output of the models (R Core Team 2021). See Suppl. material 1: Table S7 for details on models.

To account for pseudo-replication due to the classification of some questions into multiple impact types (Suppl. material 1: Table S2), we repeated the previous steps 1,000 times by choosing in each one a single impact type per question at random. We called these tests randomisation tests. Note the difference with the permutation tests where we swapped $G_{Prot\text{-}Spp}$ (see sensitivity analyses in Targets 10 and 11 in Suppl. material 1: Table S4). To consider the uncertainty in the results, we calculated the proportion of times each of the five models from the second nested model were selected: (i) interactive effect; (ii) additive effect; (iii) single effect of impact type; (iv) single effect of taxonomic group; or (v) just the intercept (Suppl. material 1: Table S7). Later, we calculated the averaged estimated marginal means of the models included in each of the five sets. We conducted these analyses twice, once considering the detailed impact types and another considering the coarse impact types.

Complementarily, we considered that evaluating questions that are not common across the three taxonomic groups limits our ability to quantify the influence of the impact type and taxonomic group. Thus, we also repeated all the previous steps, but using only the common questions across the taxonomic groups (see sensitivity analyses in Targets 10 and 11 in Suppl. material 1: Table S4).

We ran the beta regression models with the R package *glmmTMB* to include random effects (Brooks et al. 2017). We extracted the residuals with the R package *stats* (R Core Team 2021). We performed the linear mixed models with the R package *lme4* (Bates et al. 2015). We evaluated the performance of models by evaluating their residuals with the function *simulateResiduals* of the R package *DHARMa* (Hartig 2020; see its vignette for details). We evaluated the differences between the different variables by studying the estimated marginal means. We calculated the estimated marginal means of its factor levels in all iterations by using the functions *emmeans* and *immeans* of the R package *emmeans*, depending on whether the model has single effects or interactive effects. We performed the Tukey post-hoc test with the function *pairs* of the R package *emmeans* (Lenth 2021). We quantified the proportion of the variance explained by the model as the pseudo-R provided by Johnson (2014).

Factors associated with protocols

We also used $G_{Quest\text{-}Taxon}$ to complement the main analyses on the protocol variable (Target 5 in Suppl. material 1: Table S4). We explored whether the potential signal in the variable protocol can reflect the different impact types evaluated or number of responses in the questions in each protocol (Targets 8 and 9 in Suppl. material 1: Table S4). We calculated the variance partitioning of two sets of beta regressions modelling $G_{Quest\text{-}Taxon}$. In one set of beta regressions, we modelled the mean and the dispersion with the variables protocol and number of responses. In the second set of beta regressions, we modelled the mean and the dispersion with the variables protocol and impact types. In both models, we calculated the pseudo-R^2 of the saturated model and compared it with the pseudo-R^2 of the models containing only one of the variables. We considered that no shared variance supports the influence of linguistic uncertainties in explaining the consistency in responses between protocols.

Results

Species

The mean $G_{Prot\text{-}Spp}$ was high for 40 out of 60 species ($G_{Prot\text{-}Spp} \geq 0.8$; 19 invertebrates, 12 plants and nine vertebrates), medium for 13 species ($G_{Prot\text{-}Spp} \geq 0.67$ and < 0.8); seven invertebrates, five vertebrates and one plant) and low for seven species ($G_{Prot\text{-}Spp} < 0.67$; three invertebrates, two plants and two vertebrates; Fig. 1). Only in five assessments, assessors scored impacts with a very low consistency ($G_{Prot\text{-}Spp} < 0.3$; *Hydrocotyle verticillata* and *Percnon gibbesi*, both evaluated with GABLIS; *Craspedacusta sowerbii* and *Phasianus colchicus* with GB; and *Solanum elaeagnifolium* with EPPO). See all $G_{Prot\text{-}Spp}$ values in Suppl. material: Table S3. In some cases, $G_{Prot\text{-}Spp}$ varied largely (standard deviations > 0.2). Species with low mean $G_{Prot\text{-}Spp}$ values tended to have larger standard deviations (Spearman correlation between the mean and the standard deviation = -0.82; Fig. 1). However, in general, the standard deviations of the different species overlapped. See Target 2 in Table 3.

The permutation tests showed that the concentration of low consistent assessments ($G_{Prot\text{-}Spp} < 0.67$) could be observed by chance, indicating that assessments with low consistency were not associated with few specific species (Target 1 in Table 3). The p-value of unconditional Boschloo's test was below 0.05 in 0 cases of the 1,000 randomisations, independently of the threshold used to calculate the proportions (30%, 40% and 50%).

Amount of information available on species impacts

The correlation test showed a negative relationship between the proportion of low consistent assessments and the number of published articles on species impact (Estimate = -1.85; Z-value = -14.49; p-value < 0.001). However, the variance explained was low (pseudo-$R^2 \approx 0.05$).

Taxonomic groups and protocols

From the 28 beta regression models used to evaluate the influence of the taxonomic group or the protocols, we identified three best models (Suppl. material 1: Table S8). We focused our results on the model with the variables protocol and taxonomic group because it is the simpler and included our two variables of interest. Nevertheless, the results of the common variable protocol were similar to those of the best models (Suppl. material 1: Tables S9 and S10).

The analyses of the residuals showed no significant deviations from uniformity and homogeneity assumptions for the variable taxonomic group (Kolmogorov-Smirnov test: D = 0.10, p-value = 0.30; uniformity test of each level had a p-value > 0.08; Levene's

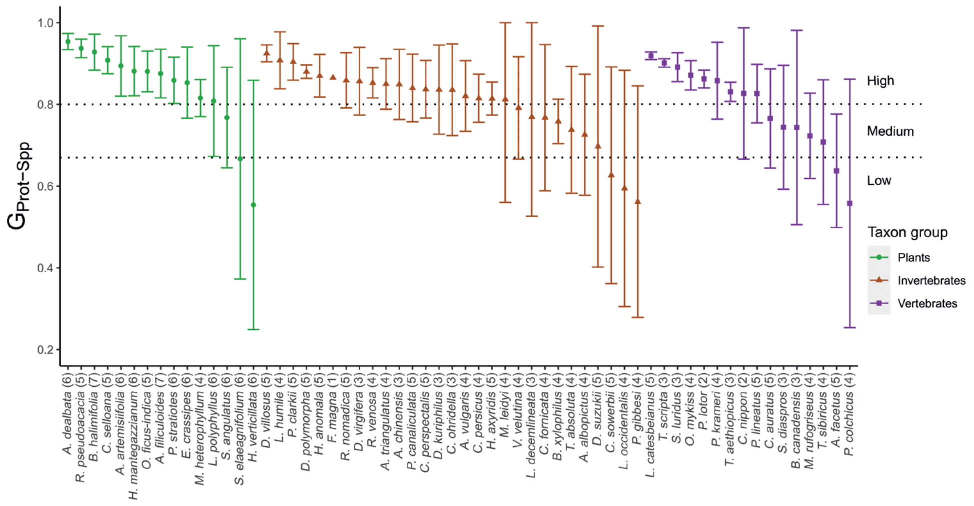

Figure 1. Mean ± standard deviations of the degree of assessor consistency when scoring the impacts of the same species across different protocols ($G_{Prot\text{-}Spp}$). The colours represent different taxonomic groups (green = plants, brown = invertebrates, purple = vertebrates). The number of protocols used to assess each species is indicated between brackets. See complete names of species in Suppl. material 1: Table S3.

test for homogeneity of variance: F value = 0.14, p-value = 0.87) or the variable protocol (Kolmogorov-Smirnov test: D = 0.14, p-value = 0.30; uniformity test of each level had a p-value > 0.20; Levene's test for homogeneity of variance: F value = 0.85, p-value = 0.54). The variable protocol explained greater variance in $G_{Prot\text{-}Spp}$ than the taxonomic group (marginal pseudo-R^2 ≈ 0.10 and ≈ 0.03, respectively). See Targets 4 and 5 in Table 3.

Assessors tended to score plant impacts with high consistency, while invertebrate and vertebrate impacts were moderately consistent, although confidence intervals overlap with G = 0.80 (Fig. 2A). There were statistical differences between plants and vertebrates (Estimate = -0.551, SE = 0.174, p-value = 0.005) and plants and invertebrates (Estimate = -0.422, SE = 0.155, p-value = 0.019), but not between vertebrates and invertebrates (Estimate = -0.129, SE = 0.164, p-value = 0.711). For the protocols, assessors tended to score impacts highly and consistently when using Harmonia$^+$, GISS and EICAT protocols, moderately with GB, moderately-low with GABLIS and low consistently with EPPO. Consistency when using EPPO prioritisation, a protocol that only considered three questions on impacts and with many singularities issues when estimating $G_{Prot\text{-}Spp}$, was highly variable (Fig. 2B; see statistical differences between pairs of protocols in Suppl. material 1: Table S9; Tukey post-hoc test).

The sensitivity analysis, i.e. a repetition of the beta regressions, but also including the $G_{Prot\text{-}Spp}$ values from the mixed models with a warning about singularity, showed greater differences between the levels of the variables protocol and taxonomic group (Suppl. material 3: Fig. S1). However, uniformity and homoscedasticity assumptions were violated.

On the other hand, our complementary analysis to evaluate whether the variable protocol reflected variations in the number of questions per protocol (Target 6 in Suppl. Material 1: Table S4), showed that a model including the variable number of

Table 3. Summary of the main results. Target = Factor evaluated. See details on hypotheses and expectations in Suppl. material 1: Table S4.

Target	Analyses	Result	Interpretation
1) Species	Permutation test	The frequency of species with large proportions of low-consistent assessments can be obtained by chance.	There is no evidence that low-consistent assessments are associated with particular species and, thus, no evidence of clear epistemic uncertainty on species.
2) Species	Descriptive analyses	Visually, the standard deviations overlap across species.	There are no differences in the consistency of responses when assessing different species.
3) Species	Correlation test	Negative correlation between the number of published articles and the proportion of low-consistent assessments. The pseudo-R^2 was low (pseudo-$R^2 \approx 0.05$).	The number of published articles is of little relevance for explaining differences observed.
4) Taxon group	Beta regression	Consistency evaluating plants tended to be larger than when evaluating vertebrates and invertebrates. However, variance explained is small (pseudo-$R^2 \approx 0.03$).	Factors associated with taxonomic groups (e.g. epistemic uncertainties) are not relevant to explain the consistency in assessments.
5) Protocol	Beta regression	Consistency in assessments varied when using different protocols. The protocol explained a low, but relevant 10% of the variance.	Factors associated with protocols are partly relevant to explain the consistency in assessments.
6) Protocol (number of questions per protocol)	Beta regression	The number of protocol questions explains half as much variance as the protocol variable.	Factors associated with protocols are important to some extent. However, some relevance of the protocols is unrelated to the number of questions per protocol (e.g. linguistic uncertainties; see complementary analyses in Targets 8 and 9).
7) Protocol	Descriptive analyses	Some species showed large standard deviations	Factors associated with protocols are important for the impact assessments of some species.
8) Protocol (number of responses per question)	Beta regression	Small variance shared between the number of response questions and the protocol.	The signal observed in protocol (target 5) is not due to number of responses per question and could be caused by linguistic uncertainties.
9) Protocol (Impact type)	Beta regression	Small variance shared between the impact types and the protocol.	The signal observed in protocol (target 5) is not due to the impact types asked in each protocol and could be caused by linguistic uncertainties.
10) Impact types	Beta regression (Nested 1)	Not interesting result. Analysis to avoid overparameterisation. See results on nested linear models 2 (Target 11).	
11) Impact types	Linear model (Nested 2)	As for the coarse impacts, the 1,000 iterations selected as the best model is the one including just the intercept. As for the detailed impacts, only 12.7% of the models showed a statistical signal on impact types. In those cases, impact type explained ≈ 10% of the variance. Sensitivity analyses When using only the common questions for the three taxonomic groups, there is no signal on impact types.	Impact type partly explains the variance in consistency. However, the disappearance of the signal when using the common questions to the three taxonomic groups, suggests the importance of questions specific for each taxon.

questions was worse ($AICc_{questions}$ -387.21 Vs $AICc_{Protocol}$ -416.53). In addition, the marginal pseudo-R^2 of the model including the number of questions was approximately half of the model including the protocol.

Impact types

Our analyses found no statistical differences in $G_{Quest\text{-}Taxon}$ between questions on the coarser impacts (i.e. environmental vs. socio-economic). However, when focusing

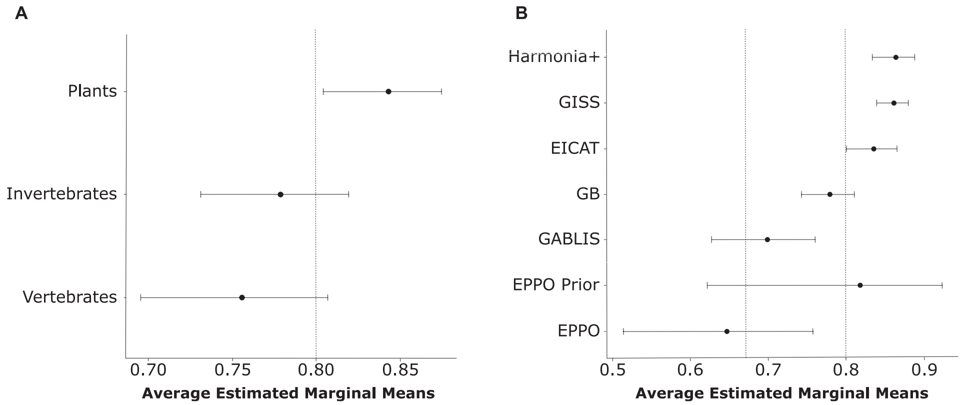

Figure 2. Estimated inter-rater reliability ($G_{Prot\text{-}Spp}$ values) when scoring species belonging to different taxonomic groups (**A**) or using different protocols (**B**). Values averaged over the levels of the variable taxonomic group and protocol, **A** and **B**, respectively, included in the beta regression model (i.e. average estimated marginal means). The dot depicts the mean and the brackets the confidence level at 95%. X-axis values apply the R function *emmeans* with type 'response'. The vertical dotted lines represent the thresholds used to categorise the coefficients G as low, medium and high consistent.

on the detailed impacts, there were no statistical differences in 87.3% of the 1,000 randomisations, i.e. the best model included just the intercept, but there were some differences in the remaining 12.7%. In this reduced subset of models, the consensus of average estimated marginal means showed that assessors most consistently scored questions about impacts on ecosystems and human health and least consistently scored questions about hybridisation and biological interaction amongst species (Fig. 3; see consensus Tukey posthoc-test in Suppl. material 1: Table S11). The single effect of the impact types explained on average 11.4% of the variance in $G_{Prot\text{-}Quest}$. In our sensitivity analyses using only the common questions amongst the three taxonomic groups, there were neither statistical differences between taxonomic groups nor impact types at the coarse and detailed levels. See Targets 10 and 11 in Table 3.

Our complementary analyses to unravel if the signal about the protocol reflected differences in the number of responses per question or the impact types asked in each protocol, showed that the variable protocol shared an irrelevant variance with the variables number of responses per protocol question or the impact types asked (see variance partitioning in Suppl. material 1: Table S12; see Targets 8 and 9 in Table 3).

For similarity with results on $G_{Prot\text{-}Spp}$, we indicated which questions had the highest and lowest consistency ($G_{Quest\text{-}taxon}$). The questions with the highest consistency ($G_{Quest\text{-}taxon} > 0.80$) belonged to protocols Harmonia$^+$ (20 combinations of questions and taxonomic group), GB (20), GISS (20), EICAT (10), GABLIS (4) and EPPO (1); while those with the lowest consistency ($G_{Quest\text{-}taxon} < 0.30$) belonged to protocols Harmonia$^+$ (8), EICAT (2) and GABLIS (2). See the complete list of $G_{Prot\text{-}Spp}$ and $G_{Quest\text{-}taxon}$ values in Suppl. material 1: Tables S3 and S13.

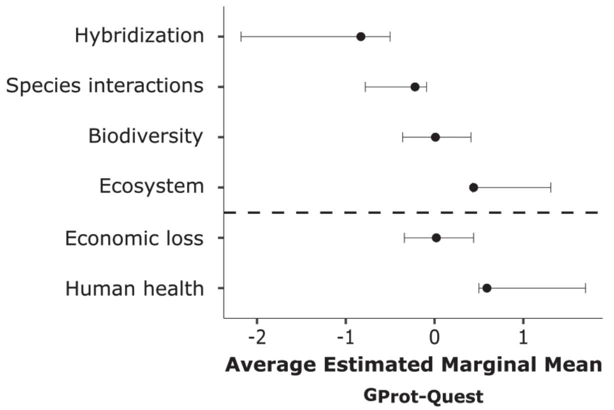

Figure 3. Assessor consistency when scoring different impact types. Results from the 12.7% of the 1,000 randomisations, i.e. models including only the single effect of the detailed impact types as explanatory variable, when using the dataset including all protocol questions on impact ($G_{Quest-Taxon}$). The unit of the x-axis is residuals; note that these estimates are from a model using the residuals of a previous model as dependent variable. The dot depicts the mean and the brackets the confidence level at 95%. See consensus Tukey adhoc-test in Suppl. material 1: Table S11.

Discussion

We provide the first empirical overview of the consistency amongst assessors in scoring particular questions of invasive species impacts in risk assessment. The broad coverage of this study (60 species from three major taxonomic groups and seven protocols) makes our results highly generalisable, while the focus on particular questions, beyond final scores and rankings, provided accurate estimates of the importance of the assessor in risk assessment, as well as evidence on the importance of the drivers, such as the impact types evaluated. In summary, this study provides new and essential information on one of the many sides of the complex prism that is repeatability in impact assessments.

Our most important finding is that assessor consistency was generally high, with up to 67% of the species studied showing high consistency. Thus, it is reasonable to conclude that impact assessments are largely reproducible and reliable. Our results both support and contrast with those of the limited number of existing studies on the consistency of assessments protocols at the answer level (Volery et al. 2021 and Clarke et al. 2021, respectively). However, comparisons are difficult because of the focus of previous studies on a single protocol (EICAT) and taxonomic group (Volery et al. 2021 = alien ungulates; Clarke et al. 2021 = insects), as well as because of the differences in the number of assessors involved or in the guidelines used (Volery et al. 2021 = similar number; Clarke et al. 2021 = two assessors). Another important point is that the methods for calculating consistency vary and the criteria for considering responses as high or low consistent were not explicit as here. Therefore, to move forward with confidence in this field of knowledge, we call for an intuitive and general criterion for measuring the consistency of impact assessments, such as the inter-rater reliability metric, as well as to set standards for the values at which consistency is considered high enough to underpin management.

No species had all its assessments with low consistency and the number of species with a large proportion of low-consistent assessments could have been caused by chance (Targets 1 and 2 in Table 3). This lack of support for the importance of epistemic uncertainties may contrast a priori with the observed negative correlation between the number of published articles on species impacts and the proportion of low-consistent assessments in those species or by the different consistency of assessors scoring impacts of the diverse taxonomic groups (Targets 3 and 4 in Table 3). However, the variance explained by both was very low. Thus, although the invasive species analysed here are not a random subset of all alien species, but arbitrarily selected, epistemic factors associated with particular species and taxonomic groups may be less relevant than expected (Leung et al. 2012; McGeoch et al. 2012).

As for impact types, a small fraction of our nested randomised models (12.7%) suggested that assessors scored questions on ecosystem and human health impacts more consistently than questions on hybridisation and biological interactions with native species (Target 11 in Table 3). These results may be surprising as previous studies have shown how scientific evidence for plant impacts on species is greater and more consistent than for ecosystems (Vilà et al. 2011). Our results also support the fact that, although information on economic impacts is sometimes relatively detailed or more readily available than on ecological ones (e.g. Pimentel et al. 2005; Vilà et al. 2010; Roberts et al. 2018; Diagne et al. 2020), the consistency when answering impacts may not be one of the highest due to the also frequent knowledge gaps (McLaughlan et al. 2014; Renault et al. 2021) and context dependency (Haubrock et al. 2021). Human health impact questions showed the highest consistency, which might be related to the well-known health impact of certain species (e.g. hay fever and disease transmission; Mazza and Tricarico 2018). However, these inferences must be taken with care as most of the nested randomised models (87.3%) did not show statistical differences amongst impact types (i.e. the best second nested model included just the intercept). Moreover, the complete disappearance of the signal in the impact types when considering only the common questions across the three taxonomic groups (sensitivity analyses) can also support that variability in consistency can depend on impacts associated with particular taxa. Therefore, these results can highlight the need for quantitative species-specific evidence (Hulme et al. 2013) and for evaluating the degree of confidence on taxon-specific tools (Glamuzina et al. 2017).

As for protocols, our results support previous studies observing high consistency in assessments using the Harmonia^{+}, GISS and EICAT protocols (Essl et al. 2011; Kenis et al. 2012; Turbé et al. 2017; Volery et al. 2021), while EPPO and GABLIS protocols showed less consistency (Target 5 in Table 3). Our complementary analyses to discern the source of the variability associated with the protocols showed that a relative important part of the variance associated with protocols was not explained by the number of questions per protocol, the number of responses per question or the impact types asked in each protocol (Targets 6, 8 and 9 in Table 3). Potentially, the ability of some protocols to consider knowledge gaps in their responses can partly explain differences in consistency when using alternative protocols (a hypothesis that we did not explore statistically). However, if that is the case, the protocols GABLIS and GISS should have the highest consistency, as they are the only ones considering the response "unknown impact". While this is true for GISS, we, however, observed the contrary result for GABLIS. Thus, our results open the door to the possibility that some variability associated with protocols may be due to linguistic factors, such as the form of the question and language clarity (Turbé et al. 2017; White et al. 2019; Clarke et al. 2021). Although our analyses provide some insights into the role of linguistic uncertainties for consistency, their unravelling would require multidisciplinary collaboration (between ecologists and sociologists). In the meantime, our results call into question whether uncertainty in the alien species lists is almost exclusively epistemic (McGeoch et al. 2012) and support the view that there is still room for improvement of protocols and guidelines (Hawkins et al. 2015; Kumschick et al. 2017; Sandvik et al. 2019; Volery et al. 2020).

Despite the commented differences when scoring different impact types or when using diverse protocols, we note that most impact assessments were highly consistent and that no single factor explained variance to a large extent, important points to prioritise efforts against invasive species. The lack of a clear major factor may suggest that the variability in consistency may be due to different causes and that increasing consistency requires multiple and complementary approaches. To explore this possibility, we conducted additional visual and non-statistical inspections of the nature of the disagreements amongst assessors of the raw data. We observed that the reason of inconsistencies in $G_{Prot\text{-}Spp}$ were diverse, such as the awareness of impacts (e.g. unknown vs. known impacts; GABLIS protocol) or the severity (e.g. low vs. medium in EPPO and GB protocols). Similarly, we observed that low consistencies in $G_{Quest\text{-}taxon}$ were due to assessors disagreeing on the impact severity (e.g. EICAT), the strength of evidence (e.g. "yes" vs. "evidence-based assumption"; GABLIS), or applying the guidelines wrongly (e.g. inapplicable vs. low; Harmonia+). These observations, not shown here, support that the lack of consistency can be due to multiple factors already commented upon in literature (McGeoch et al. 2012; Turbé et al. 2017; White et al. 2019; Probert et al. 2020; Clarke et al. 2021).

Although addressing this question adequately requires analyses beyond the goal of our study, the consistency in scores may be increased by following recommendations from literature. At the assessors group level, it may be promoted by the organisation of iteration-consensus meetings amongst assessors within taxa and across taxa (e.g. horizon scanning; Roy et al. 2014; Gallardo et al. 2016), the use of the same information (Volery et al. 2020), the use of working groups and of peer review panels with clear feedback between assessors and reviewers (Burgman et al. 2011; D'hondt et al. 2015; Vanderhoeven et al. 2017; Volery et al. 2021). At the assessor level, information gathered from scientific literature can be requested to support scores (Vanderhoeven et al. 2017; Vilà et al. 2019) or promote the training of assessors (González-Moreno et al. 2019), an aspect not considered in our dataset, but currently done in some assessments (e.g. EICAT). At the protocol level, it would be desirable to provide clear explanations and guidelines on the information requested for scoring impacts (D'hondt et al. 2015; Turbé et al. 2017; Vilà et al. 2019; Volery et al. 2020), to foment closed-ended questions and improve their wording to avoid ambiguity (Turbé et al. 2017; Vilà et al. 2019) and, at the level of the information used, to foment studies without the presence of confounding factors and with details on data quality and type of the impact observation (see more details in Volery et al. 2020).

In summary, there is still room for improvement in impact assessments and may require multiple and complementary approaches, such as those described above. However, impact assessments are highly consistent and, therefore, reliable to underpin decision-making. This is a positive and hopeful message, since in view of the expected increase in non-native species introductions (Seebens et al. 2021), we will have to prioritise management and tools, such as impact assessments, will play a key role.

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References

Almeida D, Ribeiro F, Leunda PM, Vilizzi L, Copp GH (2013) Effectiveness of FISK, an invasiveness screening tool for non-native freshwater fishes, to perform risk identification assessments in the Iberian Peninsula. Risk Analysis 33(8): 1404–1413. https://doi.org/10.1111/risa.12050

Baker R, Black R, Copp GH, Haysom K, Hulme PE, Thomas M, Ellis M (2008) The UK risk assessment scheme for all non-native species. In: Rabitsch W, Essl F, Klingenstein F (Eds) Biological invasions: from ecology to conservation. Institute of Ecology of the TU Berlin, Berlin, 46–57.

Bates D, Mächler M, Bolker B, Walker S (2015) Fitting Linear Mixed-Effects Models Using lme4. Journal of Statistical Software 67(1): 1–48. https://doi.org/10.18637/jss.v067.i01

Bellard C, Cassey P, Blackburn TM (2016) Alien species as a driver of recent extinctions. Biology Letters 12(2): 20150623. https://doi.org/10.1098/rsbl.2015.0623

Bindewald A, Michiels HG, Bauhus J (2020) Risk is in the eye of the assessor: Comparing risk assessments of four non-native tree species in Germany. Forestry. International Journal of Forestry Research 93(4): 519–534. https://doi.org/10.1093/forestry/cpz052

Blackburn TM, Essl F, Evans T, Hulme PE, Jeschke JM, Kühn I, Kumschick S, Marková Z, Mrugała A, Nentwig W, Pergl J, Pyšek P, Rabitsch W, Ricciardi A, Richardson DM, Sendek A, Vilà M, Wilson JRU, Winter M, Genovesi P, Bacher S (2014) A unified classification of alien species based on the magnitude of their environmental impacts. PLoS Biology 12(5): e1001850. https://doi.org/10.1371/journal.pbio.1001850

Booy O, Robertson PA, Moore N, Ward J, Roy HE, Adriaens T, Shaw R, Van Valkenburg J, Wyn G, Bertolino S, Blight O, Branquart E, Brundu G, Caffrey J, Capizzi D, Casaer J, De Clerck O, Coughlan NE, Davis E, Dick JTA, Essl F, Fried G, Genovesi P, González-Moreno P, Huysentruyt F, Jenkins SR, Kerckhof F, Lucy FE, Nentwig W, Newman J, Rabitsch W, Roy S, Starfinger U, Stebbing PD, Stuyck J, Sutton-Croft M, Tricarico E, Vanderhoeven S, Verreycken H, Mill AC (2020) Using structured eradication feasibility assessment to prioritize the management of new and emerging invasive alien species in Europe. Global Change Biology 26(11): 6235–6250. https://doi.org/10.1111/gcb.15280

Brennan RL (2001) Generalizability Theory. Springer New York, NY. https://doi.org/10.1007/978-1-4757-3456-0

Brooks ME, Kristensen K, Van Benthem KJ, Magnusson A, Berg CW, Nielsen A, Skaug HJ, Machler M, Bolker BM (2017) glmmTMB balances speed and flexibility among packages for zero-inflated generalized linear mixed modeling. The R Journal 9(2): 378–400. https://doi.org/10.32614/RJ-2017-066

Brunel S, Branquart E, Fried G, Van Valkenburg J, Brundu G, Starfinger U, Buholzer S, Uludag A, Joseffson M, Baker R (2010) The EPPO prioritization process for invasive alien plants. Bulletin OEPP. EPPO Bulletin. European and Mediterranean Plant Protection Organisation 40(3): 407–422. https://doi.org/10.1111/j.1365-2338.2010.02423.x

Burgman MA, McBride M, Ashton R, Speirs-Bridge A, Flander L, Wintle B, Fidler F, Rumpff L, Twardy C (2011) Expert status and performance. PLoS ONE 6(7): e22998. https://doi.org/10.1371/journal.pone.0022998

Calhoun P (2021). Exact: Unconditional Exact Test. R package version 3.1. https://CRAN.R-project.org/package=Exact

Chapman D, Purse BV, Roy HE, Bullock JM (2017) Global trade networks determine the distribution of invasive non-native species. Global Ecology and Biogeography 26(8): 907–917. https://doi.org/10.1111/geb.12599

Clarke DA, Palmer DJ, McGrannachan C, Burgess TI, Chown SL, Clarke RH, Kumschick S, Lach L, Liebhold AM, Roy HE, Saunders ME, Yeates DK, Zalucki MP, McGeoch MA (2021) Options for reducing uncertainty in impact classification for alien species. Ecosphere 12(4): e03461. https://doi.org/10.1002/ecs2.3461

Cribari-Neto F, Zeileis A (2009) Beta Regression in R. Research Report Series. Department of Statistics and Mathematics, 98. https://doi.org/10.18637/jss.v034.i02

D'hondt B, Vanderhoeven S, Roelandt S, Mayer F, Versteirt V, Adriaens T, Ducheyne E, San Martin G, Grégoire J-C, Stiers I, Quoilin S, Cigar J, Heughebaert A, Branquart E (2015) Harmonia+ and Pandora+: Risk screening tools for potentially invasive plants, animals and their pathogens. Biological Invasions 17(6): 1869–1883. https://doi.org/10.1007/s10530-015-0843-1

Diagne C, Catford JA, Essl F, Nuñez MA, Courchamp F (2020) 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

Dorrough J, Oliver I, Wall J (2018) Consensus when experts disagree: A priority list of invasive alien plant species that reduce ecological restoration success. Management of Biological Invasions 9(3): 329–341. https://doi.org/10.3391/mbi.2018.9.3.15

Early R, Bradley BA, Dukes JS, Lawler JJ, Olden JD, Blumenthal DM, Gonzalez P, Grosholz ED, Ibañez I, Miller LP, Sorte CJB, Tatem AJ (2016) Global threats from invasive alien species in the twenty-first century and national response capacities. Nature Communications 7(1): 1–9. https://doi.org/10.1038/ncomms12485

Essl F, Nehring S, Klingenstein F, Milasowszky N, Nowack C, Rabitsch W (2011) Review of risk assessment systems of IAS in Europe and introducing the German–Austrian Black List Information System (GABLIS). Journal for Nature Conservation 19(6): 339–350. https://doi.org/10.1016/j.jnc.2011.08.005

Ferrari SL, Espinheira PL, Cribari-Neto F (2011) Diagnostic tools in beta regression with varying dispersion. Statistica Neerlandica 65(3): 337–351. https://doi.org/10.1111/j.1467-9574.2011.00488.x

Furr RM (2021) Psychometrics: an introduction. SAGE publications, Thousand Oaks, California.

Gallardo B, Zieritz A, Adriaens T, Bellard C, Boets P, Britton JR, Newman JR, van Valkenburg JLCH, Aldridge DC (2016) Trans-national horizon scanning for invasive non-native species: A case study in western Europe. Biological Invasions 18(1): 17–30. https://doi.org/10.1007/s10530-015-0986-0

Genovesi P, Shine C (2004) European strategy on invasive alien species: Convention on the Conservation of European Wildlife and Habitats (Bern Convention). Council of Europe.

Glamuzina B, Tutman P, Nikolić V, Vidović Z, Pavličević J, Vilizzi L, Copp GH, Simonović P (2017) Comparison of taxon-specific and taxon-generic risk screening tools to identify potentially invasive non-native fishes in the River Neretva Catchment (Bosnia and Herzegovina and Croatia). River Research and Applications 33(5): 670–679. https://doi.org/10.1002/rra.3124

González-Moreno P, Lazzaro L, Vilà M, Preda C, Adriaens T, Bacher S, Brundu G, Copp GH, Essl F, García-Berthou E, Katsanevakis S, Moen TL, Lucy FE, Nentwig W, Roy HE, Srėbalienė G, Talgø V, Vanderhoeven S, Andjelković A, Arbačiauskas K, Auger-Rozenberg M-A, Bae M-J, Bariche M, Boets P, Boieiro M, Borges PA, Canning-Clode J, Cardigos F, Chartosia N, Cottier-Cook EJ, Crocetta F, D'hondt B, Foggi B, Follak S, Gallardo B, Gammelmo Ø, Giakoumi S, Giuliani C, Guillaume F, Jelaska LŠ, Jeschke JM, Jover M, Juárez-Escario A, Kalogirou S, Kočić A, Kytinou E, Laverty C, Lozano V, Maceda-Veiga A, Marchante E, Marchante H, Martinou AF, Meyer S, Minchin D, Montero-Castaño A, Morais MC, Morales-Rodriguez C, Muhthassim N, Nagy ZÁ, Ogris N, Onen H, Pergl J, Puntila R, Rabitsch W, Ramburn TT, Rego C, Reichenbach F, Romeralo C, Saul W-C, Schrader G, Sheehan R, Simonović P, Skolka M, Soares AO, Sundheim L, Tarkan AS, Tomov R, Tricarico E, Tsiamis K, Uludağ A, van Valkenburg J, Verreycken H, Vettraino AM, Vilar L, Wiig Ø, Witzell J, Zanetta A, Kenis M (2019) Consistency of impact assessment protocols for non-native species. NeoBiota 44: 1–25. https://doi.org/10.3897/neobiota.44.31650

Gwet KL (2014) Handbook of inter-rater reliability: The definitive guide to measuring the extent of agreement among raters. Advanced Analytics, LLC.

Hallgren KA (2012) Computing inter-rater reliability for observational data: An overview and tutorial. Tutorials in Quantitative Methods for Psychology 8(1): 23–34. https://doi.org/10.20982/tqmp.08.1.p023

Hartig F (2020) DHARMa: residual diagnostics for hierarchical regression models. Compr. R Arch. Netw.

Haubrock PJ, Turbelin AJ, Cuthbert RN, Novoa A, Taylor NG, Angulo E, Ballesteros-Mejia L, Bodey TW, Capinha C, Diagne C, Essl F, Golivets M, Kirichenko N, Kourantidou M, Leroy B, Renault D, Verbrugge L, Courchamp F (2021) Economic costs of invasive alien species across Europe. NeoBiota 67: 153–190. https://doi.org/10.3897/neobiota.67.58196

Hawkins CL, Bacher S, Essl F, Hulme PE, Jeschke JM, Kühn I, Kumschick S, Nentwig W, Pergl J, Pyšek P, Rabitsch W, Richardson DM, Vilà M, Wilson JRU, Genovesi P, Blackburn TM (2015) Framework and guidelines for implementing the proposed IUCN Environmental Impact Classification for Alien Taxa (EICAT). Diversity & Distributions 21(11): 1360–1363. https://doi.org/10.1111/ddi.12379

Hulme PE, Pyšek P, Jarošík V, Pergl J, Schaffner U, Vila M (2013) Bias and error in understanding plant invasion impacts. Trends in Ecology & Evolution 28(4): 212–218. https://doi.org/10.1016/j.tree.2012.10.010

Johnson PC (2014) Extension of Nakagawa & Schielzeth's R2GLMM to random slopes models. Methods in Ecology and Evolution 5(9): 944–946. https://doi.org/10.1111/2041-210X.12225

Keller RP, Perrings C (2011) International policy options for reducing the environmental impacts of invasive species. Bioscience 61(12): 1005–1012. https://doi.org/10.1525/bio.2011.61.12.10

Keller RP, Lodge DM, Finnoff DC (2007) Risk assessment for invasive species produces net bioeconomic benefits. Proceedings of the National Academy of Sciences of the United States of America 104(1): 203–207. https://doi.org/10.1073/pnas.0605787104

Kenis M, Bacher S, Baker RHA, Branquart E, Brunel S, Holt J, Hulme PE, MacLeod A, Pergl J, Petter F, Pyšek P, Schrader G, Sissons A, Starfinger U, Schaffner U (2012) New protocols to assess the environmental impact of pests in the EPPO decision-support scheme for pest risk analysis. Bulletin OEPP. EPPO Bulletin. European and Mediterranean Plant Protection Organisation 42(1): 21–27. https://doi.org/10.1111/j.1365-2338.2012.02527.x

Koo TK, Li MY (2016) A guideline of selecting and reporting intraclass correlation coefficients for reliability research. Journal of Chiropractic Medicine 15(2): 155–163. https://doi.org/10.1016/j.jcm.2016.02.012

Krippendorff K (1980) Validity in content analysis. In: Mochmann E (Eds) Computerstrategien fur die kommunikationsanalyse, 69–112.

Kumschick S, Vimercati G, De Villiers FA, Mokhatla MM, Davies SJ, Thorp CJ, Rebelo AD, Measey GJ (2017) Impact assessment with different scoring tools: How well do alien amphibian assessments match? NeoBiota 33: 53–66. https://doi.org/10.3897/neobiota.33.10376

Latombe G, Canavan S, Hirsch H, Hui C, Kumschick S, Nsikani MM, Potgieter LJ, Robinson TB, Saul W-C, Turner SC, Wilson JRU, Yannelli FA, Richardson DM (2019) A four-component classification of uncertainties in biological invasions: Implications for management. Ecosphere 10(4): e02669. https://doi.org/10.1002/ecs2.2669

Lawson LL, Hill JE, Hardin S, Vilizzi L, Copp GH (2015) Evaluation of the fish invasiveness screening kit (FISK v2) for peninsular Florida. Management of Biological Invasions 6(4): 413–422. https://doi.org/10.3391/mbi.2015.6.4.09

Lenth RV (2021) Estimated marginal means, aka least-squares means [R Package Emmeans Version 1.6. 0]. Comprehensive R Archive Network (CRAN).

Leung B, Roura-Pascual N, Bacher S, Heikkilä J, Brotons L, Burgman MA, Dehnen-Schmutz K, Essl F, Hulme PE, Richardson DM, Sol D, Vilà M (2012) TEASIng apart alien species risk assessments: A framework for best practices. Ecology Letters 15(12): 1475–1493. https://doi.org/10.1111/ele.12003

Matthews J, van der Velde G, Collas FP, de Hoop L, Koopman KR, Hendriks AJ, Leuven RS (2017) Inconsistencies in the risk classification of alien species and implications for risk assessment in the European Union. Ecosphere 8(6): e01832. https://doi.org/10.1002/ecs2.1832

Mazza G, Tricarico E (2018) Invasive species and human health. CABI, 210 pp. https://doi.org/10.1079/9781786390981.0000

McGeoch MA, Spear D, Kleynhans EJ, Marais E (2012) Uncertainty in invasive alien species listing. Ecological Applications 22(3): 959–971. https://doi.org/10.1890/11-1252.1

McGeoch MA, Genovesi P, Bellingham PJ, Costello MJ, McGrannachan C, Sheppard A (2016) Prioritizing species, pathways, and sites to achieve conservation targets for biological invasion. Biological Invasions 18(2): 299–314. https://doi.org/10.1007/s10530-015-1013-1

McLaughlan C, Gallardo B, Aldridge DC (2014) How complete is our knowledge of the ecosystem services impacts of Europe's top 10 invasive species? Acta Oecologica 54: 119–130. https://doi.org/10.1016/j.actao.2013.03.005

Mumford JD, Booy O, Baker RHA, Rees M, Copp GH, Black K, Holt J, Leach AW, Hartley M (2010) Invasive non-native species risk assessment in Great Britain. Aspects of Applied Biology: 49–54.

Nentwig W, Kühnel E, Bacher S (2010) A generic impact-scoring system applied to alien mammals in Europe. Conservation Biology 24(1): 302–311. https://doi.org/10.1111/j.1523-1739.2009.01289.x

Nentwig W, Bacher S, Pyšek P, Vilà M, Kumschick S (2016) The generic impact scoring system (GISS): A standardized tool to quantify the impacts of alien species. Environmental Monitoring and Assessment 188(5): 315. https://doi.org/10.1007/s10661-016-5321-4

Perdikaris C, Koutsikos N, Vardakas L, Kommatas D, Simonović P, Paschos I, Detsis V, Vilizzi L, Copp GH (2016) Risk screening of non-native, translocated and traded aquarium freshwater fishes in Greece using Fish Invasiveness Screening Kit. Fisheries Management and Ecology 23(1): 32–43. https://doi.org/10.1111/fme.12149

Pimentel D, Zuniga R, Morrison D (2005) Update on the environmental and economic costs associated with alien-invasive species in the United States. Ecological Economics 52(3): 273–288. https://doi.org/10.1016/j.ecolecon.2004.10.002

Probert AF, Volery L, Kumschick S, Vimercati G, Bacher S (2020) Understanding uncertainty in the Impact Classification for Alien Taxa (ICAT) assessments. NeoBiota 62: 387–405. https://doi.org/10.3897/neobiota.62.52010

Putka DJ, Le H, McCloy RA, Diaz T (2008) Ill-structured measurement designs in organizational research: Implications for estimating interrater reliability. Journal of Applied Psychology 93(5): 959–981. https://doi.org/10.1037/0021-9010.93.5.959

Pyšek P, Hulme PE, Simberloff D, Bacher S, Blackburn TM, Carlton JT, Dawson W, Essl F, Foxcroft LC, Genovesi P, Jeschke JM, Kühn I, Liebhold AM, Mandrak NE, Meyerson LA, Pauchard A, Pergl J, Roy HE, Seebens H, Kleunen M, Vilà M, Wingfield MJ, Richardson DM (2020) Scientists' warning on invasive alien species. Biological Reviews of the Cambridge Philosophical Society 95(6): 1511–1534. https://doi.org/10.1111/brv.12627

R Core Team (2021) R: A language and environment for statistical computing. R Foundation for Statistical Computing. Vienna. https://www.R-project.org/

Regan HM, Colyvan M, Burgman MA (2002) A taxonomy and treatment of uncertainty for ecology and conservation biology. Ecological Applications 12(2): 618–628. https://doi.org/10.1890/1051-0761(2002)012[0618:ATATOU]2.0.CO;2

Renault D, Manfrini E, Leroy B, Diagne C, Ballesteros-Mejia L, Angulo E, Courchamp F (2021) Biological invasions in France: Alarming costs and even more alarming knowledge gaps. NeoBiota 67: 191–224. https://doi.org/10.3897/neobiota.67.59134

Roberts M, Cresswell W, Hanley N (2018) Prioritising invasive species control actions: Evaluating effectiveness, costs, willingness to pay and social acceptance. Ecological Economics 152: 1–8. https://doi.org/10.1016/j.ecolecon.2018.05.027

Roy HE, Peyton J, Aldridge DC, Bantock T, Blackburn TM, Britton R, Clark P, Cook E, Dehnen-Schmutz K, Dines T, Dobson M, Edwards F, Harrower C, Harvey MC, Minchin D, Noble DG, Parrott D, Pocock MJO, Preston CD, Roy S, Salisbury A, Schönrogge K, Sewell J, Shaw RH, Stebbing P, Stewart AJA, Walker KJ (2014) Horizon scanning for invasive alien species with the potential to threaten biodiversity in Great Britain. Global Change Biology 20(12): 3859–3871. https://doi.org/10.1111/gcb.12603

Sandvik H, Hilmo O, Finstad AG, Hegre H, Moen TL, Rafoss T, Skarpaas O, Elven R, Sandmark H, Gederaas L (2019) Generic ecological impact assessment of alien species (GEIAA): The third generation of assessments in Norway. Biological Invasions 21(9): 2803–2810. https://doi.org/10.1007/s10530-019-02033-6

Sardain A, Sardain E, Leung B (2019) Global forecasts of shipping traffic and biological invasions to 2050. Nature Sustainability 2(4): 274–282. https://doi.org/10.1038/s41893-019-0245-y

Seebens H, Essl F, Dawson W, Fuentes N, Moser D, Pergl J, Pyšek P, van Kleunen M, Weber E, Winter M, Blasius B (2015) Global trade will accelerate plant invasions in emerging economies under climate change. Global Change Biology 21(11): 4128–4140. https://doi.org/10.1111/gcb.13021

Seebens H, Blackburn TM, Dyer EE, Genovesi P, Hulme PE, Jeschke JM, Pagad S, Pyšek P, Winter M, Arianoutsou M, Bacher S, Blasius B, Brundu G, Capinha C, Celesti-Grapow L, Dawson W, Dullinger S, Fuentes N, Jäger H, Kartesz J, Kenis M, Kreft H, Kühn I, Lenzner B, Liebhold A, Mosena A, Moser D, Nishino M, Pearman D, Pergl J, Rabitsch W, Rojas-Sandoval J, Roques A, Rorke S, Rossinelli S, Roy HE, Scalera R, Schindler S, Štajerová K, Tokarska-Guzik B, van Kleunen M, Walker K, Weigelt P, Yamanaka T, Essl F (2017) No saturation in the accumulation of alien species worldwide. Nature Communications 8(1): 14435. https://doi.org/10.1038/ncomms14435

Seebens H, Bacher S, Blackburn TM, Capinha C, Dawson W, Dullinger S, Genovesi P, Hulme PE, Kleunen M, Kühn I, Jeschke JM, Lenzner B, Liebhold AM, Pattison Z, Pergl J, Pyšek P, Winter M, Essl F (2021) Projecting the continental accumulation of alien species through to 2050. Global Change Biology 27(5): 970–982. https://doi.org/10.1111/gcb.15333

Smith K (2020) The IUCN Red List and invasive alien species: an analysis of impacts on threatened species and extinctions. IUCN.

Sohrabi S, Pergl J, Pyšek P, Foxcroft LC, Gherekhloo J (2021) Quantifying the potential impact of alien plants of Iran using the Generic Impact Scoring System (GISS) and Environmental Impact Classification for Alien Taxa (EICAT). Biological Invasions 23(8): 1–15. https://doi.org/10.1007/s10530-021-02515-6

Turbé A, Strubbe D, Mori E, Carrete M, Chiron F, Clergeau P, González-Moreno P, Le Louarn M, Luna A, Menchetti M, Nentwig W, Pârâu LG, Postigo J-L, Rabitsch W, Senar JC, Tollington S, Vanderhoeven S, Weiserbs A, Shwartz A (2017) Assessing the assessments: Evaluation of four impact assessment protocols for invasive alien species. Diversity & Distributions 23(3): 297–307. https://doi.org/10.1111/ddi.12528

Turbelin AJ, Malamud BD, Francis RA (2017) Mapping the global state of invasive alien species: Patterns of invasion and policy responses. Global Ecology and Biogeography 26(1): 78–92. https://doi.org/10.1111/geb.12517

Vanderhoeven S, Branquart E, Casaer J, D'hondt B, Hulme PE, Shwartz A, Strubbe D, Turbé A, Verreycken H, Adriaens T (2017) Beyond protocols: Improving the reliability of expert-based risk analysis underpinning invasive species policies. Biological Invasions 19(9): 2507–2517. https://doi.org/10.1007/s10530-017-1434-0

Vilà M, Basnou C, Pyšek P, Josefsson M, Genovesi P, Gollasch S, Nentwig W, Olenin S, Roques A, Roy D, Hulme PE (2010) How well do we understand the impacts of alien species on ecosystem services? A pan-European, cross-taxa assessment. Frontiers in Ecology and the Environment 8(3): 135–144. https://doi.org/10.1890/080083

Vilà M, Espinar JL, Hejda M, Hulme PE, Jarošík V, Maron JL, Pergl J, Schaffner U, Sun Y, Pyšek P (2011) Ecological impacts of invasive alien plants: A meta-analysis of their effects on species, communities and ecosystems. Ecology Letters 14(7): 702–708. https://doi.org/10.1111/j.1461-0248.2011.01628.x

Vilà M, Gallardo B, Preda C, García-Berthou E, Essl F, Kenis M, Roy HE, González-Moreno P (2019) A review of impact assessment protocols of non-native plants. Biological Invasions 21(3): 709–723. https://doi.org/10.1007/s10530-018-1872-3

Vilizzi L, Copp GH, Adamovich B, Almeida D, Chan J, Davison PI, Dembski S, Ekmekçi FG, Ferincz Á, Forneck SC, Hill JE, Kim J-E, Koutsikos N, Leuven RSEW, Luna SA, Magalhães F, Marr SM, Mendoza R, Mourão CF, Neal JW, Onikura N, Perdikaris C, Piria M, Poulet N, Puntila R, Range IL, Simonović P, Ribeiro F, Tarkan AS, Troca DFA, Vardakas L, Verreycken H, Vintsek L, Weyl OLF, Yeo DCJ, Zeng Y (2019) A global review and meta-analysis of applications of the freshwater Fish Invasiveness Screening Kit. Reviews in Fish Biology and Fisheries 29(3): 529–568. https://doi.org/10.1007/s11160-019-09562-2

Vilizzi L, Copp GH, Hill JE, Adamovich B, Aislabie L, Akin D, Al-Faisal AJ, Almeida D, Azmai MNA, Bakiu R, Bellati A, Bernier R, Bies JM, Bilge G, Branco P, Bui TD, Canning-Clode J, Cardoso Ramos HA, Castellanos-Galindo GA, Castro N, Chaichana R, Chainho P, Chan J, Cunico AM, Curd A, Dangchana P, Dashinov D, Davison PI, de Camargo MP, Dodd JA, Durland Donahou AL, Edsman L, Ekmekçi FG, Elphinstone-Davis J, Erős T, Evangelista C, Fenwick G, Ferincz Á, Ferreira T, Feunteun E, Filiz H, Forneck SC, Gajduchenko HS, Gama Monteiro J, Gestoso I, Giannetto D, Gilles Jr AS, Gizzi F, Glamuzina B, Glamuzina L, Goldsmit J, Gollasch S, Goulletquer P, Grabowska J, Harmer R, Haubrock PJ, He D, Hean JW, Herczeg G, Howland KL, İlhan A, Interesova E, Jakubčinová K, Jelmert A, Johnsen SI, Kakareko T, Kanongdate K, Killi N, Kim J-E, Kırankaya ŞG, Kňazovická D, Kopecký O, Kostov V, Koutsikos N, Kozic S, Kuljanishvili T, Kumar B, Kumar L, Kurita Y, Kurtul I, Lazzaro L, Lee L, Lehtiniemi M, Leonardi G, Leuven RSEW, Li S, Lipinskaya T, Liu F, Lloyd L, Lorenzoni M, Luna SA, Lyons TJ, Magellan K, Malmstrøm M, Marchini A, Marr SM, Masson G, Masson L, McKenzie CH, Memedemin D, Mendoza R, Minchin D, Miossec L, Moghaddas SD, Moshobane MC, Mumladze L, Naddafi R, Najafi-Majd E, Năstase A, Năvodaru I, Neal JW, Nienhuis S, Nimtim M, Nolan ET, Occhipinti-Ambrogi A, Ojaveer H, Olenin S, Olsson K, Onikura N, O'Shaughnessy K, Paganelli D, Parretti P, Patoka J, Pavia Jr RTB, Pellitteri-Rosa D, Pelletier-Rousseau M, Peralta EM, Perdikaris C, Pietraszewski D, Piria M, Pitois S, Pompei L, Poulet N, Preda C, Puntila-Dodd R, Qashqaei AT, Radočaj T, Rahmani H, Raj S, Reeves D, Ristovska M, Rizevsky V, Robertson DR, Robertson P, Ruykys L, Saba AO, Santos JM, Sarı HM, Segurado P, Semenchenko V, Senanan W, Simard N, Simonović P, Skóra ME, Slovák Švolíková K, Smeti E, Šmídová T, Špelić I, Srėbalienė G, Stasolla G, Stebbing P, Števove B, Suresh VR, Szajbert B, Ta KAT, Tarkan AS, Tempesti J, Therriault TW, Tidbury HJ, Top-Karakuş N, Tricarico E, Troca DFA, Tsiamis K, Tuckett QM, Tutman P, Uyan U, Uzunova E, Vardakas L, Velle G, Verreycken H, Vintsek L, Wei H, Weiperth A, Weyl OLF, Winter ER, Włodarczyk R, Wood LE, Yang R, Yapıcı S, Yeo SSB, Yoğurtçuoğlu B, Yunnie ALE, Zhu Y, Zięba G, Žitňanová K, Clarke S (2021) A global-scale screening of non-native aquatic organisms to identify potentially invasive species under current and future climate conditions. Science of the Total Environment 788: 147868. https://doi.org/10.1016/j.scitotenv.2021.147868

Volery L, Blackburn TM, Bertolino S, Evans T, Genovesi P, Kumschick S, Roy HE, Smith KG, Bacher S (2020) Improving the Environmental Impact Classification for Alien Taxa (EICAT): A summary of revisions to the framework and guidelines. NeoBiota 62: 547–567. https://doi.org/10.3897/neobiota.62.52723

Volery L, Jatavallabhula D, Scillitani L, Bertolino S, Bacher S (2021) Ranking alien species based on their risks of causing environmental impacts: A global assessment of alien ungulates. Global Change Biology 27(5): 1003–1016. https://doi.org/10.1111/gcb.15467

White RL, Strubbe D, Dallimer M, Davies ZG, Davis AJS, Edelaar P, Groombridge J, Jackson HA, Menchetti M, Mori E, Nikolov BP, Pârâu LG, Pečnikar ŽF, Pett TJ, Reino L, Tollington S, Turbé A, Shwartz A (2019) Assessing the ecological and societal impacts of alien parrots in Europe using a transparent and inclusive evidence-mapping scheme. NeoBiota 48: 45–69. https://doi.org/10.3897/neobiota.48.34222

Wickham H, Averick M, Bryan J, Chang W, McGowan L, François R, Grolemund G, Hayes A, Henry L, Hester J, Kuhn M, Pedersen T, Miller E, Bache S, Müller K, Ooms J, Robinson D, Seidel D, Spinu V, Takahashi K, Vaughan D, Wilke C, Woo K, Yutani H (2019) Welcome to the Tidyverse. Journal of Open Source Software 4(43): 1686. https://doi.org/10.21105/joss.01686

Zhao W, Zhang R, Lv Y, Liu J (2014) Variable selection for varying dispersion beta regression model. Journal of Applied Statistics 41(1): 95–108. https://doi.org/10.1080/02664763.2013.830284

Supplementary material 1

Tables S1–S13

Author: Rubén Bernardo-Madrid

Data type: Tables.

Explanation note: **Table S1.** Species evaluated with impact assessments. **Table S2.** Classification of the impact questions into the different impact types. **Table S3.** GProt-Spp per impact assessment. Inter-rater reliability using all impact questions of the protocol. **Table S4.** Summary of the principal and sensitivity analyses performed to study the influence of different factors on the consistency of responses in protocol questions. **Table S5.** Queries used to search scientific articles in Web of Science. **Table S6.** Models used to evaluate the influence of the protocol and taxonomic group in assessor consistency. **Table S7.** Saturated models for the two nested model to unravel the influence of impact types and their potential interaction with the taxonomic groups. **Table S8.** The 10 regression models with the lowest AICc to evaluate the influence of the protocol and the taxonomic groups. **Table S9.** Tukey post-hoc for the variable protocol in the model including the variable taxonomic group. **Table S10.** Tukey post-hoc for the variable protocol in the model including the number of assessors. **Table S11.** Consensus Tukey post-hoc for the variable impact type. **Table S12.** Variance partitioning of the models to unravel the shared variance of the variable protocol with the number of responses per protocol question and impact types. **Table S13.** GProt-Quest per protocol question. Inter-rater reliability per question when considering the impact scores of all species of the same taxonomic group.

Link: https://doi.org/10.3897/neobiota.76.83028.suppl1

Supplementary material 2

Impact assessments and function to calculate G coefficient

Author: Rubén Bernardo-Madrid

Data type: R objects.

Explanation note: An R list object containing the used impact assessments in the study An R function to calculate the coefficient G (inter-rater reliability metric).

Link: https://doi.org/10.3897/neobiota.76.83028.suppl2

Supplementary material 3

Figure S1

Author: Rubén Bernardo-Madrid

Data type: Figure.

Explanation note: Consistency in impact assessments of invasive species is generally high and depends on protocols and impact types.

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DISCUSSION PAPER

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Expanding the invasion toolbox: including stable isotope analysis in risk assessment

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Abstract
Species introductions are a major concern for ecosystem functioning, socio-economic wealth, and human well-being. Preventing introductions proved to be the most effective management strategy, and various tools such as species distribution models and risk assessment protocols have been developed or applied to this purpose. These approaches use information on a species to predict its potential invasiveness and impact in the case of its introduction into a new area. At the same time, much biodiversity has been lost due to multiple drivers. Ways to determine the potential for successful reintroductions of once native but now extinct species as well as assisted migrations are yet missing. Stable isotope analyses are commonly used to reconstruct a species' feeding ecology and trophic interactions within communities. Recently, this method has been used to predict potentially arising trophic interactions in the absence of the target species. Here we propose the implementation of stable isotope analysis as an approach for assessment schemes to increase the accuracy in predicting invader impacts as well as the success of reintroductions and assisted migrations. We review and discuss possibilities and limitations of this methods usage, suggesting promising and useful applications for scientists and managers.

Keywords
Impacts, mixing models, modelling, prediction, screening, stable isotope analysis

Introduction

Species introductions are an increasing concern for global biodiversity conservation (Doherty et al. 2016; Ricciardi et al. 2017; Bradley et al. 2019). This includes foremost the introduction of alien species, i.e. those species accidentally or intentionally moved outside their natural geographic range by humans, which shows no sign of saturation (Seebens et al. 2017). Introduced species often interact (Veselý et al. 2021), in many cases facilitating each other's establishment (Simberloff and Von Holle 1999), increasing their impacts due to interactions with anthropogenic stressors such as pollution (Crooks et al. 2011) and climate change (Hellmann et al. 2008; Rahel and Holden 2008; Beaury et al. 2020) and thereby become invasive. In addition to impacts on ecosystems (Ehrenfeld 2010), negative effects on human health (Mazza and Tricarico 2018) and ecosystem services (Walsh et al. 2016), as well as increasing economic costs due to direct damages (Ahmed et al. 2021a, b; Angulo et al. 2021) and associated management (Bradshaw et al. 2016; Diagne et al. 2021) are increasingly recognized.

To be effective, efforts to control invasive species should follow the hierarchical approach of the 2002 Convention on Biological Diversity, with prevention as the best option (Simberloff et al. 2013; Roy et al. 2019). For this purpose, risk screenings that identify which species should undergo a comprehensive risk assessment as well as standardized risk assessment protocols to identify potentially arising new threats have been developed (Essl et al. 2011; Vilizzi et al. 2021). These primarily aim at the identification of the most harmful species, pathways, and susceptible (invadable) sites whose protection should be prioritized (McGeoch et al. 2016). Risk assessments are usually designed for specific taxonomic groups (Copp et al. 2009; Brunel et al. 2010), vectors (Gollasch and Leppäkoski 2007), ecosystems (Leidenberger et al. 2015), or geographic regions (Baker et al. 2008), although more comprehensive protocols have been proposed (Copp et al 2016; Davidson et al. 2017). Overall, these protocols attribute a total score summing the separate scores of assessments of species traits, ecological impacts, distribution, and control feasibility, for each of which an uncertainty level is provided (Dahlstrom et al. 2011; Srėbalienė et al. 2019).

Together with predictive models, which use the ecological niche of an invasive species to probabilistically predict its future invaded range (Mainali et al. 2015; Uden et al. 2015), risk assessments are the main tools used to inform decision-makers and wildlife managers and provide fundamental information for legislations that prevent further invasions (Fournier et al. 2019), also in the context of future climate change scenarios (Chai et al. 2016). More recently, risk assessments and species distribution models have been used in combination, increasing the realism and accuracy of the predictions (Chapman et al. 2019; Yoğurtçuoğlu et al. 2021). Alternative taxa-specific approaches are trait-based models that scan a species list using ecological traits from known invasive species to identify potential new invaders (Howeth et al. 2016; Fournier et al. 2019). However, information is often missing, scarce, or anecdotal, particularly on impacts, leading to the assignment of "no potential impact" (Davidson and Hewitt 2014; Davidson et al. 2017) or "data deficient" (Kumschick et al. 2020).

On the other hand, species reintroductions, i.e. the translocation of individuals to areas in which a species became extinct with the aim of re-establishing a self-sustaining population, are of considerable value for conservation efforts (Haase and Pilotto 2019) but rarely successful – mostly due to life-cycle complexity or unpredictable external stressors (abiotic stress; biotic stress such as competition and predation; for a detailed account see Jourdan et al. 2019). Indeed, following the local extinction of a species, multiple factors can inhibit the occurrence of natural recolonization (Kail et al. 2012). For this reason, habitat restoration projects are often undertaken (Loch et al. 2020), although they too may fail due to various unforeseen factors (Bond and Lake 2003; Roni et al. 2018). Nevertheless, reintroductions are commonly used as a tool for wildlife rehabilitation (Armstrong and Seddon 2008), while the effectiveness of such reintroduction attempts themselves will depend on several intrinsic and extrinsic factors (Jourdan et al. 2019). Particularly, the interactions with other unwanted co-occurring species (i.e. alien species) can lead to failure of the reintroduction efforts (Cochran-Biederman et al. 2015).

Similar hurdles are faced by new conservation methods that have been proposed, like assisted migrations, i.e. the translocation of individuals to areas where they are predicted to move according to climate change but are not able to do so naturally (Hällfors et al. 2017). Some threatened species that could naturally move into new areas in accordance with their environmental and ecological requirements are inhibited to do so by limited time or human disturbances. For example, the presence of artificial barriers to natural dispersion can impede the ability of a species to respond to climate change and maintain its populations (Schwartz et al. 2012). Assisted migrations allow individuals to overstep such barriers in reasonable times to aid the species avoiding extinction (Schwartz et al. 2012). Although this approach has already been used (Willis et al. 2009), its application is still largely debated (Pérez et al. 2012; Schwartz et al. 2012) and depends on a trade-off between costs and benefits (Hoegh-Guldberg et al. 2008). For example, assisted migrations may lead to conservation paradoxes of species considered as endangered in their native range but recognized as invasive in their introduced range (Marchetti and Engstrom 2016; Marková et al. 2020). Accordingly, other criteria like feasibility of the translocation and collateral impacts (including the arising biotic interactions) need to be considered (Richardson et al. 2009; Hällfors et al. 2017). Therefore, the implementation of biotic outcomes prediction is crucial to assess whether assisted migration is an advantageous conservation strategy (Peterson and Bode 2021).

In all these cases – ranging from biological invasions to conservation biology – the arising biotic interactions, such as trophic relationships, are difficult to predict, while representing a crucial point for effective forecasting. In particular, what is still lacking is a fine-scale prediction of potential trophic impacts (in terms of predation and competition) on the recipient community and trophic pressures that focal species will encounter. Here we propose the use of stable isotope analysis as a tool for assessment schemes to predict such trophic relationships, and discuss the requirements, advantages, and assumptions of such an approach.

Stable isotope analysis (SIA)

General description

Stable isotope analysis (SIA) of carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) can reveal long-term and time-mediated information of a community's trophic structure and connectance (Boecklen et al. 2011; Layman et al. 2012; Middelburg 2014). Moreover, SIA can be used to quantify ecological niches, reveal trophic interactions as well as feeding preferences (Kelly 2000; Newsome et al. 2007), and enable the estimation of trophic levels (Post 2002; Quezada-Romegialli et al. 2018). Carbon signatures relate to the major energy sources, while nitrogen to the trophic position of a consumer within a food web (Fry 2006; Layman et al. 2012) due to predictable changes in the isotopic signal from prey to consumer, being enriched by 1‰ for carbon and by 2.5–5‰ for nitrogen between consecutive trophic levels (Post 2002; Vanderklift and Ponsard 2003). Using mixing models, it is also possible to determine the contributions of different prey items to the diet of a consumer (Phillips et al. 2014), with the possibility of including literature-based information or diet analysis data as priors to increase the analysis accuracy (Parnell et al. 2013).

Stable isotope analysis has been proven to be a useful tool in the field of invasion ecology (Vander Zanden et al. 1999). It is often used to assess the impacts of introduced species on other taxa in term of predation (Haubrock et al. 2019a; Gaiotto et al. 2020; Oe et al. 2020) and competition with native (Balzani et al. 2016) and other alien species (Balzani et al. 2020; Haubrock et al. 2020a). Moreover, it can be used to reveal the role of new alien prey in the diet of resident predators (Juarez-Sanchez et al. 2019; Stellati et al. 2019), compare trophic levels between introduced and native populations of invasive species (Balzani et al. 2021), as well as to disentangle trophic relationships among alien species in invaded communities (Haubrock et al. 2019a; Bissattini et al. 2021). Finally, SIA can be used to identify links between terrestrial and aquatic environments and depict changes in either one following alterations in the other (Gregs et al. 2014). However, the potential of SIA in this research field has not been fully explored yet (Bodey et al. 2011), and new applications have recently been suggested (McCue et al. 2020).

Predicting biotic interactions

Recently, SIA and associated mixing models have been proposed as a new and versatile approach in assessing potential risks arising from feeding pressure by invasive species, thus enabling to forecast the possible outcomes of the reintroduction of once native species (Haubrock et al. 2019b) and unravelling the role of species introductions on native species extinction (Haubrock et al. 2020b).

These seminal studies were carried out in an aquatic system, namely the model community of Lake Arreo, Northern Spain, which is currently dominated by alien species (Haubrock et al. 2018). In the first study (Haubroch et al. 2019b), the isotopic niche of the European eel *Anguilla anguilla* from a German lake with a similar community composition (Dörner et al. 2009) was projected onto the isotopic community structure of Lake Arreo, where this fish species was once native. The aim was to assess the effectiveness of this predator as biocontrol agent for the aquatic alien species. To allow comparisons, data from both eel and the Arreo community were standardized using the baseline organism (primary producer, *Phragmites australis*) that occurs in both ecosystems. In the other study (Haubrock et al. 2020b), isotopic data from a vertebrate (the common tench *Tinca tinca*) and one invertebrate (a whirligig beetle *Gyrinus* sp.) species once native but now locally extinct, were extrapolated from suitable literature and projected in the community to model biological effects (predation, competition) that potentially lead to their demise.

As such, these studies determined a considerable biotic pressure, mostly driven by both predation from the occurring (introduced) top predator, the largemouth bass *Micropterus salmoides*, and competition with native and other introduced species. Furthermore, the potentially arising trophic web was conceptually depicted considering the potentially consumed prey of the reintroduced eel, and thus, its effect on the recipient community. These studies thereby highlighted the opportunity for a new research line that exploits the potential of isotopic data to assess specific impacts at local scales. For example, the analysis of isotopic niches and resource utilization can be used to predict interspecific interactions (i.e. either competition or predation) after the potential introduction of a global invader in other areas or locations.

Here, we propose the application of this approach as a tool to use within risk analysis frameworks, including horizon scanning and risk screening and further assessments, to prevent new invasions, and to optimize reintroduction as well as assisted migration efforts by assessing the probable trophic relationships arising. Therefore, we discuss the requirements, advantages, and assumptions of this application.

SIA impact assessment

Requirements

Trophic links between species are the result of specific local conditions, as for example the number of trophic levels and the biomass within each – and, thus, prey availability and competition – depend on the productivity of ecosystems (Leibold et al. 1997). Each community differs and is unique due to various factors such as species composition and abundance, behavioural differences and local adaptations, different energy pathways as well as connectance with the surrounding ecosystems, and ultimately abiotic variables (e.g. substrate, altitude, climate). Although consumers show a certain degree of behavioural and dietary plasticity (Lehmann et al. 2013; Svanbäck et al. 2015; Mavraki et al. 2020), it can be assumed that under similar abiotic conditions, communities with similar species would reflect similar trophic positions and structures (Haubrock et al. 2020b). Therefore, it is important to accurately choose data from communities as similar as possible to the focal community (Haubrock et al. 2019b). Moreover, the standardization of isotopic data using local baselines (i.e. primary producers or, preferably, primary consumers) is needed to make data comparable (McMahon and McCarthy 2016). However, the two data sources should rely on the same energy pathway (i.e. terrestrial vs. aquatic, C3 plants vs. C4 plants), otherwise the result would lead to a meaningless confounding effect (Haubrock et al. 2020b). The goodness of the similarity can be checked by testing whether the projected data falls into the community total hull area (Layman et al. 2007) after being standardized (Haubrock et al. 2019b).

Beside spatial, also temporal differences in species diet and relative abundance (e.g. insects boosts) and consequently in community structure must be considered when choosing data. This refers to natural seasonal changes but also to the time since introduction for already established populations, as the diet of a species can change during its invasion process (Tillberg et al. 2007), depending on resource availability (Ruffino et al. 2011).

Finally, it is well known that the carbon isotopic signature is depleted by some compounds, mostly lipids (Post et al. 2007). A plethora of methods such as lipid chemical extraction have been used to deal with this bias (Arostegui et al. 2019). Therefore, isotopic data from samples treated in the same way are needed, whereas untreated and lipid-extracted samples should not be compared, because this could lead to misinterpretations due to the incorrect topology of the projected data in the isotopic space.

Advantages

There are three main advantages that this predictive method can offer. First, the impact of a potentially introduced alien species on the native community in terms of predation can be estimated using mixing models (Parnell et al. 2013). Knowing the species composition of the host community, mixing models will allow to estimate which taxa could be mostly predated. If some of the mainly potentially predated taxa are of conservation concern, this will result in a high potential impact. On the other hand, if sensitive taxa are not likely to be heavily predated, this will reduce the risk associated with the potential introduction of a species.

Another important advantage is the estimation of feeding competition potentially occurring with other, already present, species. The overlapping resource use can be inferred by using models that investigate the proportion (Stasko et al. 2015) of potentially arising isotopic niche overlap (e.g. Bayesian or corrected standardized ellipse area or Kernel isotopic niche, Jackson et al. 2011; Eckrich et al. 2020). Isotopic niches are a multivariate (usually bidimensional) representation of the Hutchinson's n-dimensional ecological niche (Newsome et al. 2007). As discussed above, the niche defined by $\delta^{13}C$ and $\delta^{15}N$ reflects the trophic niche, so the degree of trophic niche overlap provides an index of potential competition between species. When niches are overlapped to some degree, competition is likely to occur, particularly when trophic resources become limiting (Pianka 1981).

Finally, hurdles for the reintroduction of native species as well as assisted migration projects can be identified and addressed a priori given the composition of the community where the reintroduction is planned (Haubrock et al. 2020b), informing the choice of most suitable sites where these actions will be most likely to succeed, minimizing the management costs and maximizing the success probability.

Assumptions

The most important assumption this approach relies on is the niche conservatism of the focal species. The trophic niche of a species in different ecosystems can vary (especially for generalist species) depending e.g. on the availability of different resources, community composition and habitat type (Balzani et al. 2021; Haubrock et al. 2021a,b) and ultimately on climate change (Bestion et al. 2019). Moreover, invasive species are known for their plasticity (Courant et al. 2017; Loureiro et al. 2019; Rolla et al. 2020), thus limiting the reliability of predictive modelling. Despite this, consistent patterns in feeding preferences of introduced populations have been well documented in some invasive species (Tillberg et al. 2007; Wilder et al. 2011). To address this issue, we suggest the use of data from other invasive populations, when available, as these can provide more reliable predictions (Barbet-Massin et al. 2018). However, these assumptions are the same as for other existing tools used in the prevention of potentially invasive species. Indeed, both predictive models and risk assessment protocols use information (e.g. behavioural or biological traits, impacts) on a species from its native or introduced ranges and project this information to predict potential impacts and geographic spread that could arise (Bacher et al. 2018; Roy et al. 2019; Liu et al. 2020).

Second, species should be at equilibrium in their new range and maintain their ecological niche (Gallien et al. 2012; Hattab et al. 2017). Moreover, the output is highly sensitive to uncertainties, errors, and deficient data (Katsanevakis and Moustakas 2018). Even the suggested species distribution model implementation using eDNA similarly presents some limits and potential biases (Muha et al. 2017). Alternative approaches have been proposed, such as comparative functional responses (Dick et al. 2017a,b), that showed predictive power across multiple study systems comprising different taxonomic groups and geographic regions (Cuthbert et al. 2019). Further, this approach allows the rapid assessment of ecological impacts, while incorporating context-dependencies such as warming (Haubrock et al. 2020c) and can be combined with field abundances and reproductive traits to scale-up and predict population-level impacts (Dick et al. 2017a; Dickey et al. 2020). However, the general applicability of this method to measure the impacts of a species remains debated (Dick et al. 2017c; Vonesh et al. 2017a, b). Nevertheless, these tools can provide good predictions, especially when data are derived from other invasive populations (e.g. Barbet-Massin et al. 2018).

True limitations are linked to stable isotope data availability, however with the increase in SIA studies, the available data are rapidly increasing, offering new opportunities. Pauli et al. (2015, 2017) have called for a global stable isotope database, which would prove very useful in this context, together with open access publications and data repositories. If data are available, further information can be considered to refine the predictions. For instance, stable isotope data of a potential prey species could be partitioned according to size classes to improve the resolution of applied mixing models, and predators' diet, gape size or habitat use could be used as priors in Bayesian mixing models. Such a repository for isotopic data (IsoBank) has recently been launched (https://isobank.tacc.utexas.edu/), making feasible all the possibilities above discussed.

Application and potential outlook

With all the discussed potential insights provided by SIA-based risk assessments to improve management programmes, this approach potentially presents a unique way to inform practitioners in the fields of biological invasions and conservation biology to better inform stakeholders and governmental institutions. In practical terms, SIA-based risk assessments could be integrated in already existing tools such as EICAT and/or SEICAT as well as AS-ISK (Hawkins et al. 2015; Copp et al. 2016; Bacher et al. 2018), which have been widely adopted (also in combination, see Haubrock et al. 2021c), and new *ad hoc* tools can also be developed.

Other future developments could derive from this conceptually simple framework. For example, the availability of present and past environmental data, as well as future predictions (under climate change scenarios), integrated with SIA on museum samples will allow to include a temporal view on this approach, considerably improving its accuracy.

One interesting avenue that will surely show its potential in the invasion ecology field is the compound-specific stable isotope analysis (CS-SIA) of amino acids. In the context of our theoretic framework, this promising recent methodology will undoubtedly help in solving the issue of data standardisation and availability. This technique allows a more precise estimation of a consumer's trophic position based solely on the consumer's amino acid isotopic ratios (Chikaraishi et al. 2009). This releases the isotopic data from the need to be referenced by a correct baseline to be useful for projections. Indeed, the baseline presents potentially large spatial and temporal variations that are reflected in primary producers and, consequently, in upper trophic levels along the food web (Ishikawa 2018). CS-SIA makes the isotopic data from different populations directly comparable and increases the usable datasets (i.e. including those without baseline data available). Another advantage of CS-SIA is that different tissues do not present different isotopic signatures (e.g. Cherel et al. 2019), leading to an "absolute" isotopic signature of the animal. This also favours the usability and comparability of data from different tissues, without the need of utilising the same tissue. Further, CS-SIA on museum specimens can be used to reconstruct past food webs, helping in management and restoration efforts (Blanke et al. 2018). Although this technique is still costly, the decreasing costs of eDNA analysis suggest similar price reductions for the application of CS-SIA in the near future.

Conclusions

Projecting stable isotope data onto the isotopic space of the focal community has the potential to predict impacts accompanying a newly introduced species as well as the success of species reintroduction and assisted migration. Despite some required assumptions, the approach can have high utility from a scientific as well as management perspective by identifying trophic biological impacts of a wide range of taxonomic groups and habitats. Such results can thus be used to inform risk-based management programmes and make an important contribution to impact assessments, allowing a better prioritisation. Finally, optimising the chances of success of reintroduction as well as assisted migration efforts, will turn in a considerably better resource utilization.

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References

Ahmed DA, Hudgins EJ, Cuthbert RN, Haubrock PJ, Renault D, Bonnaud E, Diagne C, Courchamp F (2021a) Modelling the damage costs of invasive alien species. https://doi.org/10.21203/rs.3.rs-380351/v1

Ahmed DA, Hudgins EJ, Cuthbert RN, Kourantidou M, Diagne C, Haubrock PJ, Leung B, Liu C, Leroy B, Petrovskii S, Courchamp F (2021b) Managing biological invasions: the cost of inaction. Biological Invasions. https://doi.org/10.21203/rs.3.rs-300416/v1

Angulo E, Hoffmann B, Ballesteros-Mejia L, Taheri A, Balzani P, Renault D, Cordonnier M, Bellard C, Diagne C, Ahmed DA, Watari Y, Courchamp F (2021) Economic costs of invasive alien ants worldwide. Biological Invasions. https://doi.org/10.21203/rs.3.rs-346306/v1

Armstrong DP, Seddon PJ (2008) Directions in reintroduction biology. Trends in Ecology & Evolution 23(1): 20–25. https://doi.org/10.1016/j.tree.2007.10.003

Arostegui MC, Schindler DE, Holtgrieve GW (2019) Does lipid-correction introduce biases into isotopic mixing models? Implications for diet reconstruction studies. Oecologia 191(4): 745–755. https://doi.org/10.1007/s00442-019-04525-7

Bacher S, Blackburn TM, Essl F, Genovesi P, Heikkilä J, Jeschke JM, Jones G, Keller R, Kenis M, Kueffer C, Martinou AF, Nentwig W, Pergl J, Pyšek P, Rabitsch W, Richardson DM, Roy HE, Saul W-C, Scalera R, Vilà M, Wilson JRH, Kumschick S (2018) Socio-economic impact classification of alien taxa (SEICAT). Methods in Ecology and Evolution 9(1): 159–168. https://doi.org/10.1111/2041-210X.12844

Baker RHA, Black R, Copp GH, Haysom KA, Hulme PE, Thomas MB, Brown A, Brown M, Cannon RJC, Ellis J, Ellis M, Ferris R, Glaves P, Gozlan RE, Holt J, Howe L, Knight JD, MacLeod A, Moore NP, Mumford JD, Murphy ST, Parrott D, Sansford CE, Smith GC, St-Hilaire S, Ward NL (2008) The UK risk assessment scheme for all alien species. In: Rabitsch W, Essl F, Klingensten F (Eds) Biological invasions from ecology to conservation. NeoBiota 7: 46–57.

Balzani P, Vizzini S, Santini G, Masoni A, Ciofi C, Ricevuto E, Chelazzi G (2016) Stable isotope analysis of trophic niche in two co-occurring native and invasive terrapins, *Emys orbicularis* and *Trachemys scripta elegans*. Biological Invasions, 18(12): 3611–3621. https://doi.org/10.1007/s10530-016-1251-x

Balzani P, Gozlan RE, Haubrock PJ (2020) Overlapping niches between two co-occurring invasive fish: the topmouth gudgeon *Pseudorasbora parva* and the common bleak *Alburnus alburnus*. Journal of Fish Biology 97(5): 1385–1392. https://doi.org/10.1111/jfb.14499

Balzani P, Vizzini S, Frizzi F, Masoni A, Lessard JP, Bernasconi C, Francoeur A, Ibarra-Isassi J, Brassard F, Cherix D, Santini G (2021) Plasticity in the trophic niche of an invasive ant explains establishment success and long-term coexistence. Oikos 130(5): 691–696. https://doi.org/10.1111/oik.08217

Barbet-Massin M, Rome Q, Villemant C, Courchamp F (2018) Can species distribution models really predict the expansion of invasive species? PLoS ONE 13(3): e0193085. https://doi.org/10.1371/journal.pone.0193085

Beaury EM, Fusco EJ, Jackson MR, Laginhas BB, Morelli TL, Allen JM, Pasquarella VJ, Bradley BA (2020) Incorporating climate change into invasive species management: insights from managers. Biological Invasions 22(2): 233–252. https://doi.org/10.1007/s10530-019-02087-6

Bestion E, Soriano-Redondo A, Cucherousset J, Jacob S, White J, Zinger L, Fourtune L, Di Gesu L, Teyssier A, Cote J (2019) Altered trophic interactions in warming climates: consequences for predator diet breadth and fitness. Proceedings of the Royal Society B 286(1914): e20192227. https://doi.org/10.1098/rspb.2019.2227

Bissattini AM, Haubrock PJ, Buono V, Balzani P, Borgianni N, Stellati L, Inghilesi AF, Tancioni L, Martinoli M, Tricarico E, Vignoli L (2021) Trophic structure of a pond community dominated by an invasive alien species: Insights from stomach content and stable isotope analyses. Aquatic Conservation: Marine and Freshwater Ecosystems 31(4): 948–963. https://doi.org/10.1002/aqc.3530

Blanke C, Chikaraishi Y, Vander Zanden MJ (2018) Historical niche partitioning and long-term trophic shifts in Laurentian Great Lakes deepwater coregonines. Ecosphere 9(1): e02080. https://doi.org/10.1002/aqc.3530

Bodey TW, Bearhop S, McDonald RA (2011) Invasions and stable isotope analysis–informing ecology and management. In: Veitch CR, Clout MN, Towns DR (Eds) Island invasives: eradication and management. IUCN, Gland, Switzerland: 148–151.

Boecklen WJ, Yarnes CT, Cook BA, James AC (2011) On the use of stable isotopes in trophic ecology. Annual Review of Ecology, Evolution, and Systematics 42: 411–440. https://doi.org/10.1146/annurev-ecolsys-102209-144726

Bond NR, Lake PS (2003) Local habitat restoration in streams: constraints on the effectiveness of restoration for stream biota. Ecological Management and Restoration 4(3): 193–198. https://doi.org/10.1046/j.1442-8903.2003.00156.x

Bradley BA, Laginhas BB, Whitlock R, Allen JM, Bates AE, Bernatchez G, Diez JM, Early R, Lenoir J, Vilà M, Sorte CJB (2019) Disentangling the abundance–impact relationship for invasive species. Proceedings of the National Academy of Sciences 116(20): 9919–9924. https://doi.org/10.1073/pnas.1818081116

Bradshaw CJ, Leroy B, Bellard C, Roiz D, Albert C, Fournier A, Barbet-Massin M, Salles JM, Simard F, Courchamp F (2016) Massive yet grossly underestimated global costs of invasive insects. Nature Communications 7(1): 1–8. https://doi.org/10.1038/ncomms12986

Brunel S, Branquart E, Fried G, Van Valkenburg J, Brundu G, Starfinger U, Buholzer S, Uludag A, Joseffson M, Baker R (2010) The EPPO prioritization process for invasive alien plants. EPPO bulletin 40(3): 407–422. https://doi.org/10.1111/j.1365-2338.2010.02423.x

Chai SL, Zhang J, Nixon A, Nielsen S (2016) Using risk assessment and habitat suitability models to prioritise invasive species for management in a changing climate. PLoS ONE 11(10): e0165292. https://doi.org/10.1371/journal.pone.0165292

Chapman D, Pescott OL, Roy HE, Tanner R (2019) Improving species distribution models for invasive non-native species with biologically informed pseudo-absence selection. Journal of Biogeography 46(5): 1029–1040. https://doi.org/10.1111/jbi.13555

Cherel Y, Bustamante P, Richard P (2019) Amino acid $\delta^{13}C$ and $\delta^{15}N$ from sclerotized beaks: a new tool to investigate the foraging ecology of cephalopods, including giant and colossal squids. Marine Ecology Progress Series 624: 89–102. https://doi.org/10.3354/meps13002

Chikaraishi Y, Ogawa NO, Kashiyama Y, Takano Y, Suga H, Tomitani A, Miyashita H, Kitazato H, Ohkouchi N (2009) Determination of aquatic food-web structure based on compound-specific nitrogen isotopic composition of amino acids. Limnology and Oceanography: methods 7(11): 740–750. https://doi.org/10.4319/lom.2009.7.740

Cochran-Biederman JL, Wyman KE, French WE, Loppnow GL (2015) Identifying correlates of success and failure of native freshwater fish reintroductions. Conservation Biology 29(1): 175–186. https://doi.org/10.1111/cobi.12374

Copp GH, Vilizzi L, Mumford J, Fenwick GV, Godard MJ, Gozlan RE (2009) Calibration of FISK, an invasiveness screening tool for nonnative freshwater fishes. Risk Analysis: An International Journal 29(3): 457–467. https://doi.org/10.1111/j.1539-6924.2008.01159.x

Copp GH, Vilizzi L, Tidbury H, Stebbing PD, Tarkan AS, Miossec L, Goulletquer P (2016) Development of a generic decision-support tool for identifying potentially invasive aquatic taxa: AS-ISK. Management of Biological Invasions 7(4): 343–350. https://doi.org/10.3391/mbi.2016.7.4.04

Courant J, Vogt S, Marques R, Measey J, Secondi J, Rebelo R, De Villiers A, Ihlow F, De Busschere C, Backeljau T, Rödder D, Herrel A (2017) Are invasive populations characterized by a broader diet than native populations? PeerJ 5: e3250. https://doi.org/10.7717/peerj.3250

Crooks JA, Chang AL, Ruiz GM (2011) Aquatic pollution increases the relative success of invasive species. Biological Invasions 13(1): 165–176. https://doi.org/10.1007/s10530-010-9799-3

Cuthbert RN, Dickey JW, Coughlan NE, Joyce PW, Dick JT (2019) The Functional Response Ratio (FRR): advancing comparative metrics for predicting the ecological impacts of invasive alien species. Biological Invasions 21(8): 2543–2547. https://doi.org/10.1007/s10530-019-02002-z

Dahlstrom A, Hewitt CL, Campbell ML (2011) A review of international, regional and national biosecurity risk assessment frameworks. Marine Policy 35(2): 208–217. https://doi.org/10.1016/j.marpol.2010.10.001

Davidson AD, Hewitt CL (2014) How often are invasion-induced ecological impacts missed? Biological Invasions 16(5): 1165–1173. https://doi.org/10.1007/s10530-013-0570-4

Davidson A, Fusaro A, Sturtevant RA, Kashian DR (2017) Development of a risk assessment framework to predict invasive species establishment for multiple taxonomic groups and vectors of introduction. Management of Biological Invasions 8(1): 25–36. https://doi.org/10.3391/mbi.2017.8.1.03

Diagne C, Leroy B, Vaissière AC, Gozlan RE, Roiz D, Jarić I, Salles J-M, Bradshaw CJA, Courchamp F (2021) High and rising economic costs of biological invasions worldwide. Nature 592(7855): 571–576. https://doi.org/10.1038/s41586-021-03405-6

Dick JT, Laverty C, Lennon JJ, Barrios-O'Neill D, Mensink PJ, Britton JR, Médoc V, Boets P, Alexander ME, Taylor NG, Dunn AM, Hatcher MJ, Rosewarne PJ, Crookes S, MacIsaac HJ, Xu M, Ricciardi A, Wasserman RJ, Ellender BR, Weyl OLF, Lucy FE, Banks PB, Dodd JA, MacNeil C, Penk MR, Aldridge DC, Caffrey JM (2017a) Invader Relative Impact Potential: a new metric to understand and predict the ecological impacts of existing, emerging and future invasive alien species. Journal of Applied Ecology 54(4): 1259–1267. https://doi.org/10.1111/1365-2664.12849

Dick JT, Alexander ME, Ricciardi A, Laverty C, Downey PO, Xu M, Jeschke JM, Saul W-C, Hill MP, Wasserman R, Barrios-O'Neill D, Weyl OLF, Shaw RH (2017b) Functional responses can unify invasion ecology. Biological Invasions 19(5): 1667–1672. https://doi.org/10.1007/s10530-016-1355-3

Dick JT, Alexander ME, Ricciardi A, Laverty C, Downey PO, Xu M, Jeschke JM, Saul W-C, Hill MP, Wasserman R, Barrios-O'Neill D, Weyl OLF, Shaw RH (2017c) Fictional responses from Vonesh et al. Biological Invasions 19(5): 1677–1678. https://doi.org/10.1007/s10530-016-1360-6

Dickey JW, Cuthbert RN, South J, Britton JR, Caffrey J, Chang X, Crane K, Coughlan N, Fadaei E, Farnsworth K, Ismar-Rebitz S, Julius M, Laverty C, Lucy FE, MacIsaac HJ, McCard M, McGlade C, Reid N, Ricciardi A, Wasserman RJ, Dick JT (2020) On the RIP: using Relative Impact Potential to assess the ecological impacts of invasive alien species. NeoBiota 55: 27–60. https://doi.org/10.3897/neobiota.55.49547

Doherty TS, Glen AS, Nimmo DG, Ritchie EG, Dickman CR (2016) Invasive predators and global biodiversity loss. Proceedings of the National Academy of Sciences 113(40): 11261–11265. https://doi.org/10.1073/pnas.1602480113

Dörner H, Benndorf J (2003) Piscivory by large eels on young-of-the-year fishes: its potential as a biomanipulation tool. Journal of Fish Biology 62(2): 491–494. https://doi.org/10.1046/j.1095-8649.2003.00035.x

Eckrich CA, Albeke SE, Flaherty EA, Bowyer RT, Ben-David M (2020) rKIN: Kernel-based method for estimating iso topic niche size and overlap. Journal of Animal Ecology 89: 757–771. https://doi.org/10.1111/1365-2656.13159

Ehrenfeld JG (2010) Ecosystem consequences of biological invasions. Annual review of ecology, evolution, and systematics 41: 59–80. https://doi.org/10.1146/annurev-ecolsys-102209-144650

Essl F, Nehring S, Klingenstein F, Milasowszky N, Nowack C, Rabitsch W (2011) Review of risk assessment systems of IAS in Europe and introducing the German–Austrian Black List Information System (GABLIS). Journal for Nature Conservation 19(6): 339–350. https://doi.org/10.1016/j.jnc.2011.08.005

Fournier A, Penone C, Pennino MG, Courchamp F (2019) Predicting future invaders and future invasions. Proceedings of the National Academy of Sciences 116(16): 7905–7910. https://doi.org/10.1073/pnas.1803456116

Fry B (2006) Stable isotope ecology. Springer, Berlin. https://doi.org/10.1007/0-387-33745-8

Gaiotto JV, Abrahão CR, Dias RA, Bugoni L (2020) Diet of invasive cats, rats and tegu lizards reveals impact over threatened species in a tropical island. Perspectives in Ecology and Conservation 18(4): 294–303. https://doi.org/10.1016/j.pecon.2020.09.005

Gallien L, Douzet R, Pratte S, Zimmermann NE, Thuiller W (2012) Invasive species distribution models–how violating the equilibrium assumption can create new insights. Global Ecology and Biogeography 21(11): 1126–1136. https://doi.org/10.1111/j.1466-8238.2012.00768.x

Gergs R, Koester M, Schulz RS, Schulz R (2014) Potential alteration of cross-ecosystem resource subsidies by an invasive aquatic macroinvertebrate: implications for the terrestrial food web. Freshwater Biology 59(12): 2645–2655. https://doi.org/10.1111/fwb.12463

Gollasch S, Leppäkoski E (2007) Risk assessment and management scenarios for ballast water mediated species introductions into the Baltic Sea. Aquatic Invasions 2(4): 313–340. https://doi.org/10.3391/ai.2007.2.4.3

Haase P, Pilotto F (2019) A method for the reintroduction of entire benthic invertebrate communities in formerly degraded streams. Limnologica 77: e125689. https://doi.org/10.1016/j.limno.2019.125689

Hällfors MH, Aikio S, Schulman LE (2017) Quantifying the need and potential of assisted migration. Biological Conservation 205: 34–41. https://doi.org/10.1016/j.biocon.2016.11.023

Hattab T, Garzón-López CX, Ewald M, Skowronek S, Aerts R, Horen H, Brasseur B, Gallet-Moron E, Spicher F, Decocq G, Feilhauer H, Honnay O, Kempeneers P, Schmidtlein S, Somers B, Van De Kerchove R, Rocchini D, Lenoir J (2017) A unified framework to model the potential and realized distributions of invasive species within the invaded range. Diversity and Distributions 23(7): 806–819. https://doi.org/10.1111/ddi.12566

Haubrock PJ, Criado A, Monteoliva AP, Monteoliva JA, Santiago T, Inghilesi AF, Tricarico E (2018) Control and eradication efforts of aquatic alien fish species in Lake Caicedo Yuso-Arreo. Management of Biological Invasions 9(3): 267–278. https://doi.org/10.3391/mbi.2018.9.3.09

Haubrock PJ, Balzani P, Azzini M, Inghilesi AF, Veselý L, Guo W, Tricarico E (2019a). Shared histories of co-evolution may affect trophic interactions in a freshwater community dominated by alien species. Frontiers in Ecology and Evolution 7: e355. https://doi.org/10.3389/fevo.2019.00355

Haubrock PJ, Balzani P, Criado A, Inghilesi AF, Tricarico E, Monteoliva AP (2019b) Predicting the effects of reintroducing a native predator (European eel, *Anguilla anguilla*) into a freshwater community dominated by alien species using a multidisciplinary approach. Management of Biological Invasions 10(1): 171–191. https://doi.org/10.3391/mbi.2019.10.1.11

Haubrock PJ, Azzini M, Balzani P, Inghilesi AF, Tricarico E (2020a) When alien catfish meet—Resource overlap between the North American *Ictalurus punctatus* and immature European *Silurus glanis* in the Arno River (Italy). Ecology of Freshwater Fish 29(1): 4–17. https://doi.org/10.1111/eff.12481

Haubrock PJ, Balzani P, Britton JR, Haase P (2020b) Using stable isotopes to analyse extinction risks and reintroduction opportunities of native species in invaded ecosystems. Scientific Reports 10(1): 1–11. https://doi.org/10.1038/s41598-020-78328-9

Haubrock PJ, Cuthbert RN, Veselý L, Balzani P, Baker NJ, Dick JT, Kouba A (2020c) Predatory functional responses under increasing temperatures of two life stages of an invasive gecko. Scientific Reports 10(1): 1–10. https://doi.org/10.1038/s41598-020-67194-0

Haubrock PJ, Balzani P, Hundertmark I, Cuthbert RN (2021a) Spatial and size variation in dietary niche of a non-native freshwater fish. Ichthyology & Herpetology 109(2): 501–506. https://doi.org/10.1643/i2020099

Haubrock PJ, Balzani P, Matsuzaki SIS, Tarkan AS, Kourantidou M, Haase P (2021b) Spatio-temporal niche plasticity of a freshwater invader as a harbinger of impact variability. Science of The Total Environment 777: e145947. https://doi.org/10.1016/j.scitotenv.2021.145947

Haubrock PJ, Copp GH, Johović I, Balzani P, Inghilesi AF, Nocita A, Tricarico E (2021c) North American channel catfish, *Ictalurus punctatus*: a neglected but potentially invasive freshwater fish species?. Biological Invasions 23(5): 1563–1576. https://doi.org/10.1007/s10530-021-02459-x

Hawkins CL, Bacher S, Essl F, Hulme PE, Jeschke JM, Kühn I, Kumschick S, Nentwig, Pergl J, Pyšek P, Rabitsch W, Richardson DM, Vilà M, Wilson JRU, Genovesi P, Blackburn TM (2015) Framework and guidelines for implementing the proposed IUCN Environmental Impact Classification for Alien Taxa (EICAT). Diversity and Distributions 21(11): 1360–1363. https://doi.org/10.1111/ddi.12379

Hellmann JJ, Byers JE, Bierwagen BG, Dukes JS (2008) Five potential consequences of climate change for invasive species. Conservation Biology 22(3): 534–543. https://doi.org/10.1111/j.1523-1739.2008.00951.x

Hoegh-Guldberg O, Hughes L, McIntyre S, Lindenmayer DB, Parmesan C, Possingham HP, Thomas CD (2008) Ecology. Assisted colonization and rapid climate change. Science 321(5887): 345–346. https://doi.org/10.1126/science.1157897

Howeth JG, Gantz CA, Angermeier PL, Frimpong EA, Hoff MH, Keller RP, Mandrak NE, Marchetti MP, Olden JD, Romagosa CM, Lodge DM (2016) Predicting invasiveness of species in trade: climate match, trophic guild and fecundity influence establishment and impact of non-native freshwater fishes. Diversity and Distributions 22(2): 148–160. https://doi.org/10.1111/ddi.12391

Ishikawa NF (2018) Use of compound-specific nitrogen isotope analysis of amino acids in trophic ecology: assumptions, applications, and implications. Ecological Research 33(5): 825–837. https://doi.org/10.1007/s11284-018-1616-y

Jackson AL, Inger R, Parnell AC, Bearhop S (2011) Comparing isotopic niche widths among and within communities: SIBER–Stable Isotope Bayesian Ellipses in R. Journal of Animal Ecology 80(3): 595–602. https://doi.org/10.1111/j.1365-2656.2011.01806.x

Jourdan J, Plath M, Tonkin JD, Ceylan M, Dumeier AC, Gellert G, Graf W, Hawkins CP, Kiel E, Lorenz AW, Matthaei CD, Verdonschot PFM, Verdonschot RCM, Haase P (2019) Reintroduction of freshwater macroinvertebrates: challenges and opportunities. Biological Reviews 94(2): 368–387. https://doi.org/10.1111/brv.12458

Juarez-Sanchez D, Blake JG, Hellgren EC (2019) Variation in Neotropical river otter (*Lontra longicaudis*) diet: Effects of an invasive prey species. PLoS ONE 14(10): e0217727. https://doi.org/10.1371/journal.pone.0217727

Kail J, Arle J, Jähnig SC (2012) Limiting factors and thresholds for macroinvertebrate assemblages in European rivers: empirical evidence from three datasets on water quality, catchment urbanization, and river restoration. Ecological Indicators 18: 63–72. https://doi.org/10.1016/j.ecolind.2011.09.038

Katsanevakis S, Moustakas A (2018) Uncertainty in marine invasion science. Frontiers in Marine Science 5: e38. https://doi.org/10.3389/fmars.2018.00038

Kelly JF (2000) Stable isotopes of carbon and nitrogen in the study of avian and mammalian trophic ecology. Canadian Journal of Zoology 78(1): 1–27. https://doi.org/10.1139/z99-165

Kumschick S, Bacher S, Bertolino S, Blackburn TM, Evans T, Roy HE, Smith K (2020) Appropriate uses of EICAT protocol, data and classifications. NeoBiota 62: 193–212. https://doi.org/10.3897/neobiota.62.51574

Layman CA, Arrington DA, Montaña CG, Post DM (2007) Can stable isotope ratios provide for community-wide measures of trophic structure? Ecology 88(1): 42–48. https://doi.org/10.1890/0012-9658(2007)88[42:CSIRPF]2.0.CO;2

Layman CA, Araujo MS, Boucek R, Hammerschlag-Peyer CM, Harrison E, Jud ZR, Matich P, Rosenblatt AE, Vaudo JJ, Yeager LA, Post DM, Bearhop S (2012) Applying stable isotopes to examine food-web structure: an overview of analytical tools. Biological Reviews 87(3): 545–562. https://doi.org/10.1111/j.1469-185X.2011.00208.x

Lehmann D, Mfune JKE, Gewers E, Cloete J, Brain C, Voigt CC (2013) Dietary plasticity of generalist and specialist ungulates in the Namibian desert: a stable isotopes approach. PLoS ONE 8(8): e72190. https://doi.org/10.1371/journal.pone.0072190

Leibold MA, Chase JM, Shurin JB, Downing AL (1997) Species turnover and the regulation of trophic structure. Annual Review of Ecology and Systematics 28(1): 467–494. https://doi.org/10.1146/annurev.ecolsys.28.1.467

Leidenberger S, Obst M, Kulawik R, Stelzer K, Heyer K, Hardisty A, Bourlat SJ (2015) Evaluating the potential of ecological niche modelling as a component in marine non-indigenous species risk assessments. Marine Pollution Bulletin 97(1–2): 470–487. https://doi.org/10.1016/j.marpolbul.2015.04.033

Liu C, Wolter C, Xian W, Jeschke JM (2020) Most invasive species largely conserve their climatic niche. Proceedings of the National Academy of Sciences 117(38): 23643–23651. https://doi.org/10.1073/pnas.2004289117

Loch JM, Walters LJ, Cook GS (2020) Recovering trophic structure through habitat restoration: A review. Food Webs 25: e00162. https://doi.org/10.1016/j.fooweb.2020.e00162

Loureiro TG, Anastácio PM, de Siqueira Bueno SL, Woodd CT, Araujod PB (2019). Food matters: Trophodynamics and the role of diet in the invasion success of *Procambarus clarkii* in an Atlantic Forest conservation area. Limnologica 79: e125717. https://doi.org/10.1016/j.limno.2019.125717

Mainali KP, Warren DL, Dhileepan K, McConnachie A, Strathie L, Hassan G, Karki D, Shrestha BB, Parmesan C (2015) Projecting future expansion of invasive species: comparing and improving methodologies for species distribution modeling. Global Change Biology 21(12): 4464–4480. https://doi.org/10.1111/gcb.13038

Marchetti MP, Engstrom T (2016) The conservation paradox of endangered and invasive species. Conservation Biology 30(2): 434–437. https://doi.org/10.1111/cobi.12642

Marková J, Jerikho R, Wardiatno Y, Kamal MM, Magalhães ALB, Bohatá L, Kalous L, Patoka J (2020) Conservation paradox of giant arapaima *Arapaima gigas* (Schinz, 1822)(Pisces: Arapaimidae): endangered in its native range in Brazil and invasive in Indonesia. Knowledge & Management of Aquatic Ecosystems 421: e47. https://doi.org/10.1051/kmae/2020039

Mavraki N, De Mesel I, Degraer S, Moens T, Vanaverbeke J (2020) Resource niches of co-occurring invertebrate species at an offshore wind turbine indicate a substantial degree of trophic plasticity. Frontiers in Marine Science 7: 379. https://doi.org/10.3389/fmars.2020.00379

Mazza G, Tricarico E [Eds] (2018) Invasive species and human health (Vol. 10). CABI. https://doi.org/10.1079/9781786390981.0000

McCue MD, Javal M, Clusella-Trullas S, Le Roux JJ, Jackson MC, Ellis AG, Richardson DM, Valentine AJ, Terblanche JS (2020) Using stable isotope analysis to answer fundamental questions in invasion ecology: Progress and prospects. Methods in Ecology and Evolution 11(2): 196–214. https://doi.org/10.1111/2041-210X.13327

McGeoch MA, Genovesi P, Bellingham PJ, Costello MJ, McGrannachan C, Sheppard A (2016) Prioritizing species, pathways, and sites to achieve conservation targets for biological invasion. Biological Invasions 18(2): 299–314. https://doi.org/10.1007/s10530-015-1013-1

McMahon KW, McCarthy MD (2016) Embracing variability in amino acid $\delta^{15}N$ fractionation: mechanisms, implications, and applications for trophic ecology. Ecosphere 7(12): e01511. https://doi.org/10.1002/ecs2.1511

Middelburg JJ (2014) Stable isotopes dissect aquatic food webs from the top to the bottom. Biogeosciences 11(8): 2357–2371. https://doi.org/10.5194/bg-11-2357-2014

Muha TP, Rodríguez-Rey M, Rolla M, Tricarico E (2017) Using environmental DNA to improve species distribution models for freshwater invaders. Frontiers in Ecology and Evolution 5: e158. https://doi.org/10.3389/fevo.2017.00158

Newsome SD, Martinez del Rio C, Bearhop S, Phillips DL (2007) A niche for isotopic ecology. Frontiers in Ecology and the Environment 5(8): 429–436. https://doi.org/10.1890/060150.1

Oe S, Sashika M, Fujimoto A, Shimozuru M, Tsubota T (2020) Predation impacts of invasive raccoons on rare native species. Scientific reports 10(1): 1–12. https://doi.org/10.1038/s41598-020-77016-y

Parnell AC, Phillips DL, Bearhop S, Semmens BX, Ward EJ, Moore JW, Jackson AL, Grey J, Kelly DJ, Inger R (2013) Bayesian stable isotope mixing models. Environmetrics 24(6): 387–399. https://doi.org/10.1002/env.2221

Pauli JN, Steffan SA, Newsome SD (2015) It is time for IsoBank. BioScience 65(3): 229–230. https://doi.org/10.1093/biosci/biu230

Pauli JN, Newsome SD, Cook JA, Harrod C, Steffan SA, Baker CJ, Ben-David M, Bloom D, Bowen GJ, Cerling TE, Cicero C, Cook C, Dohm M, Dharampal PS, Graves G, Gropp R, Hobson KA, Jordan C, MacFadden B, Birch SP, Poelen J, Ratnasingham S, Russell L, Stricker CA, Uhen MD, Yarnes CT, Hayden B (2017) Opinion: Why we need a centralized repository for isotopic data. Proceedings of the National Academy of Sciences 114(12): 2997–3001. https://doi.org/10.1073/pnas.1701742114

Pérez I, Anadón JD, Díaz M, Nicola GG, Tella JL, Giménez A (2012) What is wrong with current translocations? A review and a decision-making proposal. Frontiers in Ecology and the Environment 10(9): 494–501. https://doi.org/10.1890/110175

Peterson K, Bode M (2021) Using ensemble modeling to predict the impacts of assisted migration on recipient ecosystems. Conservation Biology 35(2): 678–687. https://doi.org/10.1111/cobi.13571

Phillips DL, Inger R, Bearhop S, Jackson AL, Moore JW, Parnell AC, Semmens BX, Ward EJ (2014) Best practices for use of stable isotope mixing models in food-web studies. Canadian Journal of Zoology 92(10): 823–835. https://doi.org/10.1139/cjz-2014-0127

Pianka ER (1981) Competition and niche theory. In: May RM (Ed) Theoretical ecology: principles and applications. Blackwell, Oxford, 167–196.

Post DM (2002) Using stable isotopes to estimate trophic position: Models, methods, and assumptions. Ecology 83(3): 703–718. https://doi.org/10.1890/0012-9658(2002)083[0703:USITET]2.0.CO;2

Post DM, Layman CA, Arrington DA, Takimoto G, Quattrochi J, Montana CG (2007) Getting to the fat of the matter: models, methods and assumptions for dealing with lipids in stable isotope analyses. Oecologia 152: 179–189. https://doi.org/10.1007/s00442-006-0630-x

Quezada-Romegialli C, Jackson AL, Hayden B, Kahilainen KK, Lopes C, Harrod C (2018) tRophicPosition, an R package for the Bayesian estimation of trophic position from consumer stable isotope ratios. Methods in Ecology and Evolution 9(6): 1592–1599. https://doi.org/10.1111/2041-210X.13009

Rahel FJ, Olden JD (2008) Assessing the effects of climate change on aquatic invasive species. Conservation Biology 22(3): 521–533. https://doi.org/10.1111/j.1523-1739.2008.00950.x

Ricciardi A, Blackburn TM, Carlton JT, Dick JT, Hulme PE, Iacarella JC, Jeschke JM, Liebhold AM, Lockwood JL, MacIsaac HJ, Pyšek P, Richardson DM, Ruiz GM, Simberloff D, Sutherland WJ, Wardle DA, Aldridge DC (2017) Invasion science: a horizon scan of emerging challenges and opportunities. Trends in Ecology & Evolution 32(6): 464–474. https://doi.org/10.1016/j.tree.2017.03.007

Richardson DM, Hellmann JJ, McLachlan JS, Sax DF, Schwartz MW, Gonzalez P, Brennan EJ, Camacho A, Root TL, Sala OE, Schneider SH, Ashe DM, Rappaport Clark J, Early R, Etterson JR, Fielder ED, Gill JL, Minteer BA, Polasky S, Safford HD, Thompson AR, Vellend M (2009) Multidimensional evaluation of managed relocation. Proceedings of the National Academy of Sciences 106(24): 9721–9724. https://doi.org/10.1073/pnas.0902327106

Rolla M, Consuegra S, Garcia de Leaniz C (2020) Trophic plasticity of the highly invasive topmouth gudgeon (*Pseudorasbora parva*) inferred from stable isotope analysis. Frontiers in Ecology and Evolution 8: e212. https://doi.org/10.3389/fevo.2020.00212

Roni P, Åberg U, Weber C (2018) A review of approaches for monitoring the effectiveness of regional river habitat restoration programs. North American Journal of Fisheries Management 38(5): 1170–1186. https://doi.org/10.1002/nafm.10222

Roy HE, Bacher S, Essl F, Adriaens T, Aldridge DC, Bishop JD, Blackburn TM, Branquart E, Brodie J, Carboneras C, Cottier-Cook EJ, Copp GH, Dean HJ, Eilenberg J, Gallardo B, Garcia M, García-Berthou E, Genovesi P, Hulme PE, Kenis M, Kerckhof F, Kettunen M, Minchin D, Nentwig W, Nieto A, Pergl J, Pescott OL, Peyton JM, Preda C, Roques A, Rorke SL, Scalera R, Schindler S, Schönrogge K, Sewell J, Solarz W, Stewart AJA, Tricarico E, Vanderhoeven S, van der Velde G, Vilà M, Wood CA, Zenetos A, Rabitsch W (2019) Developing a list of invasive alien species likely to threaten biodiversity and ecosystems in the European Union. Global Change Biology 25(3): 1032–1048. https://doi.org/10.1111/gcb.14527

Ruffino L, Russell JC, Pisanu B, Caut S, Vidal E (2011) Low individual-level dietary plasticity in an island-invasive generalist forager. Population Ecology 53(4): 535–548. https://doi.org/10.1007/s10144-011-0265-6

Schwartz MW, Hellmann JJ, McLachlan JM, Sax DF, Borevitz JO, Brennan J, Camacho AE, Ceballos G, Clark JR, Doremus H, Early R, Etterson JR, Fielder D, Gill JL, Gonzalez P, Green N, Hannah L, Jamieson DW, Javeline D, Minteer BA, Odenbaugh J, Polasky S, Richardson DM, Root TL, Safford HD, Sala O, Schneider SH, Thompson AR, Williams JW, Vellend M, Vitt P, Zellmer S (2012) Managed relocation: integrating the scientific, regulatory, and ethical challenges. BioScience 62(8): 732–743. https://doi.org/10.1525/bio.2012.62.8.6

Seebens H, Blackburn TM, Dyer EE, Genovesi P, Hulme PE, Jeschke JM, Pagad S, Pyšek P, Winter M, Arianoutsou M, Bacher S, Blasius B, Brundu G, Capinha C, Celesti-Grapow L, Dawson W, Dullinger S, Fuentes N, Jäger H, Kartesz J, Kenis M, Kreft H, Kühn I, Lenzner B, Liebhold A, Mosena A, Moser D, Nishino M, Pearman D, Pergl J, Rabitsch W, Rojas-Sandoval J, Roques A, Rorke S, Rossinelli S, Roy HE, Scalera R, Schindler S, Štajerová K, Tokarska-Guzik B, van Kleunen M, Walker K, Weigelt P, Yamanaka T, Essl F (2017) No saturation in the accumulation of alien species worldwide. Nature Communications 8(1): 1–9. https://doi.org/10.1038/ncomms14435

Simberloff D, Von Holle B (1999) Positive interactions of nonindigenous species: invasional meltdown? Biological Invasions 1(1): 21–32. https://doi.org/10.1023/A:1010086329619

Simberloff D, Martin JL, Genovesi P, Maris V, Wardle DA, Aronson J, Courchamp F, Galil B, García-Berthou E, Pascal M, Pyšek P, Sousa R, Tabacchi E, Vilà M (2013) Impacts of biological invasions: what's what and the way forward. Trends in Ecology & Evolution 28(1): 58–66. https://doi.org/10.1016/j.tree.2012.07.013

Srėbalienė G, Olenin S, Minchin D, Narščius A (2019) A comparison of impact and risk assessment methods based on the IMO Guidelines and EU invasive alien species risk assessment frameworks. PeerJ 7: e6965. https://doi.org/10.7717/peerj.6965

Stasko AD, Johnston TA, Gunn JM (2015) Effects of water clarity and other environmental factors on trophic niches of two sympatric piscivores. Freshwater Biology 60(7): 1459–1474. https://doi.org/10.1111/fwb.12581

Stellati L, Borgianni N, Bissattini AM, Buono V, Haubrock PJ, Balzani P, Tricarico E, Inghilesi AF, Tancioni L, Martinoli M, Luiselli L, Vignoli L (2019) Living with aliens: suboptimal ecological condition in semiaquatic snakes inhabiting a hot spot of allodiversity. Acta Oecologica 100: e103466. https://doi.org/10.1016/j.actao.2019.103466

Svanbäck R, Quevedo M, Olsson J, Eklöv P (2015) Individuals in food webs: the relationships between trophic position, omnivory and among-individual diet variation. Oecologia 178(1): 103–114. https://doi.org/10.1007/s00442-014-3203-4

Tillberg CV, Holway DA, LeBrun EG, Suarez AV (2007) Trophic ecology of invasive Argentine ants in their native and introduced ranges. Proceedings of the National Academy of Sciences 104(52): 20856–20861. https://doi.org/10.1073/pnas.0706903105

Uden DR, Allen CR, Angeler DG, Corral L, Fricke KA (2015) Adaptive invasive species distribution models: a framework for modeling incipient invasions. Biological Invasions 17(10): 2831–2850. https://doi.org/10.1007/s10530-015-0914-3

Vander Zanden MJ, Casselman JM, Rasmussen JB (1999) Stable isotope evidence for the food web consequences of species invasions in lakes. Nature 401(6752): 464–467. https://doi.org/10.1038/46762

Vanderklift MA, Ponsard S (2003) Sources of variation in consumer-diet $\delta^{15}N$ enrichment: a meta-analysis. Oecologia 136(2): 169–182. https://doi.org/10.1007/s00442-003-1270-z

Veselý L, Ruokonen TJ, Weiperth A, Kubec J, Szajbert B, Guo W, Ercoli F, Bláha M, Buřič M, Hämäläinen H, Kouba A (2021) Trophic niches of three sympatric invasive crayfish of EU concern. Hydrobiologia 848(3): 727–737. https://doi.org/10.1007/s10750-020-04479-5

Vilizzi L, Copp GH, Hill JE, Adamovich B, Aislabie L, Akin D, Al-Faisalh AJ, Almeida D, Azmai MNA, Bakiu R, Bellati A, Bernier R, Bies JM, Bilge G, Branco P, Bui TD, Canning-Clode J, Ramos HAC, Castellanos-Galindo GA, Castro N, Chaichana R, Chainho P, Chan J, Cunico AM, Curd A, Dangchana P, Dashinov D, Davison PI, de Camargo MP, Dodd JA, Durland Donahou AL, Edsman L, Ekmekçi FG, Elphinstone-Davis J, Erős T, Evangelista C, Fenwick G, Ferincz A, Ferreira T, Feunteun E, Filiz H, Forneck SC, Gajduchenko HS, Monteiro JG, Gestoso I, Giannetto D, Gilles Jr AS, Gizzi F, Glamuzina B, Glamuzina L, Goldsmit J, Gollasch S, Goulletquer P, Grabowska J, Harmer R, Haubrock PJ, He D, Hean JW, Herczeg G, Howland KL, İlhan A, Interesova E, Jakubčinová K, Jelmert A, Johnsen SI, Kakareko T, Kanongdate K, Killi N, Kim J-E, Kırankaya SG, Kňazovická D, Kopecký O, Kostov V, Koutsikos N, Kozic S, Kuljanishvili T, Kumar B, Kumar L, Kurita Y, Kurtul I, Lazzaro L, Lee L, Lehtiniemi M, Leonardi G, Leuven RSEW, Li S, Lipinskaya T, Liu F, Lloyd L, Lorenzoni M, Luna SA, Lyons TJ, Magellan K, Malmstrøm M, Marchini A, Marr SM, Masson G, Masson L, McKenzie CH, Memedemin D, Mendoza R, Minchin D, Miossec L, Moghaddas SD, Moshobane MC, Mumladze L, Naddafi R, Najafi-Majd E, Năstase A, Năvodaru I, Neal JW, Nienhuis S, Nimtim M, Nolan ET, Occhipinti-Ambrogi A, Ojaveer H, Olenin S, Olsson K, Onikura N, O'Shaughnessy K, Paganelli D, Parretti P, Patoka J, Pavia Jr RTB, Pellitteri-Rosa D, Pelletier-Rousseau M, Peralta EM, Perdikaris C, Pietraszewski D, Piria M, Pitois S, Pompei L, Poulet N, Preda C, Puntila-Dodd R, Qashqaei AT, Radočaj T, Rahmani H, Raj S, Reeves D, Ristovska M, Rizevsky V, Robertson DR, Robertson P, Ruykys L, Saba AO, Santos JM, Sarı HM; Segurado P, Semenchenko V, Senanan W, Simard N, Simonović P, Skóra ME, Švolíková KS, Smeti E, Šmídová T, Špelić I, Srėbalienė G, Stasolla G, Stebbing P, Števove B, Suresh VR, Szajbertb B, Ta KAT, Tarkan AS, Tempesti J, Therriault TW, Tidbury HJ, Top-Karakuş N, Tricarico E, Troca

DFA, Tsiamis K, Tuckett QM, Tutman P, Uyan U, Uzunova E, Vardakas L, Velle G, Verreycken H, Vintsek L, Wei H, Weiperth A, Weyl OLF, Winter ER, Włodarczyk R, Wood LE, Yang R, Yapıcı S, Yeo SSB, Yoğurtçuoğlu B, Yunnie ALE, Zhu Y, Zięba G, Žitňanová K, Clarke S (2021) A global-scale screening of non-native aquatic organisms to identify potentially invasive species under current and future climate conditions. Science of the Total Environment 788: e147868. https://doi.org/10.1016/j.scitotenv.2021.147868

Vonesh J, McCoy M, Altwegg R, Landi P, Measey J (2017a) Functional responses can't unify invasion ecology. Biological Invasions 19(5): 1673–1676. https://doi.org/10.1007/s10530-016-1356-2

Vonesh J, McCoy M, Altwegg R, Landi P, Measey J (2017b) Rather than unifying invasion biology, Dick et al.'s approach rests on subjective foundations. Biological Invasions 19(5): 1679–1680. https://doi.org/10.1007/s10530-016-1361-5

Walsh JR, Carpenter SR, Vander Zanden MJ (2016) Invasive species triggers a massive loss of ecosystem services through a trophic cascade. Proceedings of the National Academy of Sciences 113(15): 4081–4085. https://doi.org/10.1073/pnas.1600366113

Wilder SM, Holway DA, Suarez AV, LeBrun EG, Eubanks MD (2011) Intercontinental differences in resource use reveal the importance of mutualisms in fire ant invasions. Proceedings of the National Academy of Sciences 108(51): 20639–20644. https://doi.org/10.1073/pnas.1115263108

Willis SG, Hill JK, Thomas CD, Roy DB, Fox R, Blakeley DS, Huntley B (2009) Assisted colonization in a changing climate: a test-study using two UK butterflies. Conservation Letters 2(1): 46–52. https://doi.org/10.1111/j.1755-263X.2008.00043.x

Yoğurtçuoğlu B, Bucak T, Ekmekçi FG, Kaya C, Tarkan AS (2021) Mapping the Establishment and Invasiveness Potential of Rainbow Trout (*Oncorhynchus mykiss*) in Turkey: With Special Emphasis on the Conservation of Native Salmonids. Frontiers in Ecology and Evolution 8: 599881. https://doi.org/10.3389/fevo.2020.599881.

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Development and application of a multilingual electronic decision-support tool for risk screening non-native terrestrial animals under current and future climate conditions

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Abstract

Electronic decision-support tools are becoming an essential component of government strategies to tackle non-native species invasions. This study describes the development and application of a multilingual electronic decision-support tool for screening terrestrial animals under current and future climate conditions: the Terrestrial Animal Species Invasiveness Screening Kit (TAS-ISK). As an adaptation of the widely employed Aquatic Species Invasiveness Screening Kit (AS-ISK), the TAS-ISK question template inherits from the original Weed Risk Assessment (WRA) and related WRA-type toolkits and complies with the 'minimum requirements' for use with the recent European Regulation on invasive alien species of concern. The TAS-ISK consists of 49 basic questions on the species' biogeographical/historical traits and its biological/ecological interactions, and of 6 additional questions to predict how climate change is likely to influence the risks of introduction, establishment, dispersal and impact of the screened species. Following a description of the main features of this decision-support tool as a turnkey software application and of its graphical user interface with support for 32 languages, sample screenings are provided in different risk assessment areas for one representative species of each of the main taxonomic groups of terrestrial animals supported by the toolkit: mammals, birds, reptiles, amphibians, annelids, insects, molluscs, nematodes, and platyhelminths. The highest-scoring species were the red earthworm *Lumbricus rubellus* for the Aegean region of Turkey and the New Zealand flatworm *Arthurdendyus triangulatus* for Croatia. It is anticipated that adoption of this toolkit will mirror that of the worldwide employed AS-ISK, hence allowing to share information and inform decisions for the prevention of entry and/or dispersal of (high-risk) non-native terrestrial animal species – a crucial step to implement early-stage control and eradication measures as part of rapid-response strategies to counteract biological invasions.

Keywords
AS-ISK, biological invasions, decision-makers, turnkey application, TAS-ISK, WRA

Introduction

The steady increase in recent times in the number of invasive non-native species worldwide and its implications for wildlife conservation emphasise the importance of developing user-friendly decision-support tools for scientists to inform decision-makers about the prioritisation of management actions in response to non-native species' impacts (Dana et al. 2014; González-Moreno et al. 2019). The identification and assessment of hazards is a crucial aspect of environmental risk analysis, which consists of three steps: risk screening (identification), risk assessment, and risk communication and management (Canter 1993; UK Defra 2003; Booy et al. 2017; Robertson et al. 2021). In the risk analysis process applied to non-native species, risk screening identifies which non-native species are likely to be invasive in a given risk assessment area. This facilitates the development of policy and management procedures for that risk assessment area to prevent and/or mitigate the impacts of biological invasions (Copp et al. 2016a). In particular, risk screening of non-native species assists decision-makers in the allocation of resources to predict which species pose an elevated threat to native species and ecosystems and therefore require full (follow-up) risk assessment. This involves detailed examination of the likelihood and magnitude of risks of introduction, establishment, dispersal and impacts of a non-native species (Copp et al. 2005, 2016a; Baker et al. 2008; Mumford et al. 2010). To this end, it is crucial to distinguish between risk screening and risk assessment: this distinction is often overlooked in environmental risk analysis, where decision-support tools are often compared and evaluated together (e.g. González-Moreno et al. 2019; Marcot et al. 2019; see also Hill et al. 2020). In this regard, the present study will focus on the first step of the risk analysis process, i.e. the risk screening, and this will include discussion of any related decision-support tools.

Decision-support tools have been developed for screening aquatic and terrestrial non-native species as well as pathogens (Pheloung et al. 1999; Copp et al. 2005, 2009, 2016b, 2021; D'hondt et al. 2015; Drolet et al. 2016). Amongst the most widely applied is the Weed Risk Assessment (WRA) for terrestrial plants (Pheloung et al. 1999) and its adaptations to various biogeographic regions and to the screening of aquatic plants (Gordon et al. 2008). The WRA question template formed the basis to create the Fish Invasiveness Screening Kit (FISK) for freshwater fish (Copp et al. 2005; Vilizzi et al. 2019) and its 'sister' -ISK toolkits for other aquatic organisms (Copp 2013). More recently, the -ISK toolkits were combined into the taxon-generic Aquatic Species Invasiveness Screening Kit (AS-ISK) to screen freshwater, brackish and marine aquatic organisms under current and future climate conditions (Copp et al. 2016b; Vilizzi et al. 2021). Other risk screening tools include Harmonia$^+$ and Pandora$^+$ (D'hondt et al. 2015) for plants, animals and their pathogens, and the Canadian Marine Invasive Screening Tool (CMIST: Drolet et al. 2016) for marine organisms.

A common feature of these risk screening tools is their availability in spreadsheet format, but with the AS-ISK only being designed as a 'turnkey' application (Copp et al. 2016b). This is contrary to the 'automated workbook' format of the other toolkits, which can make their usage time-consuming, if not counter-intuitive, to the end user. For this reason, the recent development of the AS-ISK as a user-friendly, dialog-driven electronic decision-support tool (Copp et al. 2016b) has resulted not only in a shortening of the risk screening process and, possibly, the follow-up decision-making (Matthies et al. 2007) but has also ensured exchangeability and seamless deployment of data and information across users (Copp et al. 2021). A 'fully fledged' electronic decision-support tool such as the AS-ISK, however, is currently available only for the screening of aquatic organisms. In contrast, for terrestrial organisms the (semi-automated) spreadsheet-based WRA (and its various adaptations: Gordon et al. 2008) is the only available tool for screening weeds (Dana et al. 2014). At the same time, most decision-support tools have been developed mainly in English (see Copp et al. 2021). This limitation increases the linguistic uncertainty associated with risk screenings undertaken by non-native English assessors (scientists) who ultimately need to communicate the risk outcomes to decision-makers in the country's native/official language. To meet these requirements, the 32 languages available to users of the AS-ISK are meant to enhance communication of non-native species' risks to local authorities and within/amongst non-English-speaking countries (Copp et al. 2021).

Despite the successful adoption and implementation of the WRA-type toolkits worldwide (Gordon et al. 2008; Vilizzi et al. 2019, 2021), there is currently no similar decision-support tool for screening terrestrial animals, as exemplified by the recent use of the AS-ISK as a 'surrogate' for screening terrestrial reptiles (Kopecký et al. 2019). To address this gap, this paper describes the development and application of a 'sibling' toolkit to the AS-ISK that will allow to share information and inform decision-makers about the prevention of entry and/or dispersal of (high-risk) non-native terrestrial animal species – a crucial step to implement early-stage control and eradication measures as part of rapid-response strategies to counteract biological invasions (Piria et al. 2017; Copp et al. 2021). The aims of this study were threefold: (i) to develop a turnkey application based on the AS-ISK template to produce the Terrestrial Animal Species Invasiveness Screening Kit (TAS-ISK) and describe the main elements of the toolkit's interface and functionality (including some additional features introduced since the release of AS-ISK v1: Copp et al. 2016b); (ii) to review the questions and guidance for aquatic species in the AS-ISK template for adaptation to non-native terrestrial animal species in the TAS-ISK; and (iii) to implement a trial screening of the TAS-ISK on one representative species for each of the main terrestrial animal taxonomic groups supported by this new toolkit.

Methodology

Toolkit features

As an 'offshoot' of the AS-ISK, the TAS-ISK is also designed to comply with the 'minimum standards' for screening non-native species under EC Regulation No. 1143/2014 on the prevention and management of the introduction and spread of invasive alien species (EU 2014). The TAS-ISK consists of 55 questions (Qs). The first 49 Qs comprise the Basic Risk Assessment (BRA) and address the biogeography/invasion history and biology/ecology of the screened species. The last 6 Qs include the Climate Change Assessment (CCA) and require the assessor to predict how predicted (future) climatic conditions are likely to affect the BRA with respect to risks of introduction, establishment, dispersal and impact. The BRA questions consist of two sections with eight categories: Section A Biogeography/Invasion History including Categories Domestication/Cultivation, Climate, distribution and introduction risk, and Invasive elsewhere; Section B Biology/Ecology, including Categories Undesirable (or persistence) traits, Resource exploitation, Reproduction, Dispersal mechanisms, and Tolerance attributes. The CCA questions comprise Section C (and Category) Climate change (see Suppl. material 1: Table S1).

To achieve a valid screening, the assessor must provide for each question a response, a level of confidence for the response (see below), and a justification based on literature sources. The outcomes are a BRA score, which ranges from –20 to 68, and a (composite) BRA+CCA score, which ranges from –32 to 80 (i.e. after adding or subtracting up to 12 points to the BRA score or leaving it unchanged in case of a CCA score equal to 0). Confidence levels in the responses to questions are ranked using a 1–4 scale (1 = low; 2 = medium; 3 = high; 4 = very high) as per the Intergovernmental Panel on Climate Change (see Copp et al. 2016a). Based on the confidence level (CL) allocated to each response, a confidence factor (CF) is obtained as:

$$CF = \sum(CL_{Qi})/(4 \times 55) \ (i = 1, \ldots, 55)$$

where CL_{Qi} is the CL for Qi, 4 is the maximum achievable value for confidence (i.e. very high: see above) and 55 is the total number of questions comprising the TAS-ISK questionnaire. The CF ranges from a minimum of 0.25 (i.e. all 55 Qs with a confidence level equal to 1) to a maximum of 1 (i.e. all 55 Qs with a confidence level equal to 4). For the CF, the CF_{Total}, CF_{BRA} and CF_{CCA} (based on all 55 Qs, on the 49 Qs comprising the BRA, and on the 6 Qs comprising the CCA, respectively) are computed. For further details about implementation of the overall risk screening process, see Vilizzi et al. (2022).

Toolkit development

Questions and related guidance of the AS-ISK v2.3.x template (noting that this toolkit is now available in its release v2.3.2: www.cefas.co.uk/nns/tools) were critically reviewed for application to terrestrial animal taxa. Following modification to the relevant questions and related guidance for adaptation to terrestrial animals, the resulting template was finalised by a consensus meeting to improve clarity, conciseness and accuracy in the text of both questions and guidance. The final template was then circulated amongst the author-translators (see below) for translation into the corresponding native language of the parts of text modified relative to the original AS-ISK template.

Similar to the AS-ISK, the TAS-ISK is designed as a 'turnkey application' (*sensu* Walkenbach 2007). This represents the most advanced level of Excel VBA software development as it allows complete distinction (separation) between graphical user interface, business logic, and data access/storage tiers. This is ensured by separating the data (i.e. the spreadsheet) and the graphical user interface (consisting of tightly controlled dialogs) from the underlying code. All these features offer major benefits: (i) for the end user, by allowing the assessor to work seamlessly on the database spreadsheet(s) located on the local computer or accessible from a network (e.g. under a 'cloud system'); and (ii) for the developer, by facilitating provision of feedback and support by software updates that will replace previous releases of the toolkit whilst ensuring full backward compatibility in data access. The TAS-ISK graphical user interface is available in 32 languages, which allows it to be used in some 161 countries worldwide (see also Copp et al. 2021): English, Albanian, Arabic, Bulgarian, Chinese (simplified), Croatian, Czech, Dutch, Filipino, French, Georgian, German, Greek, Hebrew, Hungarian, Italian, Japanese, Korean, Macedonian, Persian, Polish, Portuguese, Romanian, Russian, Slovak, Slovenian, Spanish, Swedish, Thai, Turkish, Urdu, Vietnamese. This extent of language support is the most advanced allowed by the Excel VBA code (Walkenbach 2007), as it includes support of right-to-left languages (i.e. Arabic, Hebrew, Persian, Urdu) and double-byte-character-set languages (i.e. Chinese, Japanese, Korean).

The TAS-ISK is available for download at www.cefas.co.uk/nns/tools in its release v2.3.2. This first release number of the toolkit mirrors that of the latest version of the AS-ISK (see above), with which the TAS-ISK, as already emphasised, shares most of the underlying code. The TAS-ISK allows the screening of nine taxonomic groups of terrestrial animals (classification mainly after Zoological Record indexing service: https://www.ebsco.com/products/research-databases/zoological-record): Mammals, Birds, Reptiles, Amphibians, Annelids, Insects, Molluscs, Nematodes, Platyhelminths, Other arthropods, Other eukaryote taxa.

Trial screenings

Trial screenings were conducted for one representative taxon (hereafter, for simplicity 'species') of each of the main taxonomic groups of terrestrial animals (i.e. except for 'Other arthropods' and 'Other eukaryote taxa'). In total, eight experts (= assessors) were involved in the resulting nine screenings, with seven species screened each by a single assessor, one species screened by two joint assessors and another species screened by three joint assessors. One assessor screened two species and another assessor four species (Table 1). Notably, each assessor chose the non-native species for screening in which they were more knowledgeable in terms of its environmental biology and risk assessment area.

Each species was categorised *a priori* into non-invasive or invasive based on a search made of: (i) the Centre for Agriculture and Bioscience International Invasive Species Compendium (CABI ISC: www.cabi.org/); (ii) the Global Invasive Species Database (GISD: www.iucngisd.org); and (iii) the Invasive and Exotic Species of North America list (IESNA: www.invasive.org). If the species was not categorised as invasive in any (or all) of the previous three databases, a Google Scholar (literature) search was performed to check whether at least one peer-reviewed reference was found that 'demonstrates' (hence, not 'assumes') invasiveness/impact. The latter was then taken as 'sufficient evidence' for categorising the species as invasive; whereas, if no evidence was found in this last step, then the species was categorised as non-invasive (see also Vilizzi et al. 2022).

As a result of the *a priori* categorisation, there were eight species categorised *a priori* as invasive: the aoudad/Barbary sheep *Ammotragus lervia* (Mammals), the common pheasant *Phasianus colchicus* (Birds), the common house gecko *Hemidactylus frenatus* (Reptiles), the red earthworm *Lumbricus rubellus* (Annelids), the western corn rootworm *Diabrotica virgifera virgifera* (Insects), the Spanish slug *Arion vulgaris* (Molluscs),

Table 1. Taxa evaluated with the Terrestrial Animal Species Invasiveness Screening Kit (TAS-ISK) for their potential risk of invasiveness in different risk assessment areas. For each species, the *a priori* categorisation outcome into Non-invasive and Invasive is provided (after Vilizzi et al. 2022).

Taxonomic group	Taxon name	Common name	Assessor(s)	Risk assessment area	*A priori* categorisation
Mammals	*Ammotragus lervia*	aoudad/Barbary sheep	NS, TR, MP	Europe	Invasive
Birds	*Phasianus colchicus*	common pheasant	TR	Croatia	Invasive
Reptiles	*Hemidactylus frenatus*	common house gecko	BS, MP	Pannonian region of Hungary	Invasive
Amphibians	*Bombina variegata*	yellow-bellied toad	OC	Anatolia (Turkey)	Non-invasive
Annelids	*Lumbricus rubellus*	red earthworm	NK	Aegean region of Turkey	Invasive
Insects	*Diabrotica virgifera virgifera*	western corn rootworm	DL	Croatia	Invasive
Molluscs	*Arion vulgaris*	Spanish slug	IŠ	Croatia	Invasive
Nematodes	*Ditylenchus destructor*	potato rot nematode	MP	Croatia	Invasive
Platyhelminths	*Arthurdendyus triangulatus*	New Zealand flatworm	MP	Croatia	Invasive

the potato rot nematode *Ditylenchus destructor* (Nematodes), and the New Zealand flatworm *Arthurdendyus triangulates* (Platyhelminths). The only species categorised *a priori* as non-invasive was the yellow-bellied toad *Bombina variegata* (Amphibians). For seven species the risk assessment area was Europe or part of it, and for two species it was Anatolia and Aegean regions of Turkey in Asia (Table 2).

Differences in CF between components (BRA, BRA+CCA) were tested with permutational ANOVA. Analysis was implemented in PERMANOVA+ for PRIMER v7, with normalisation of the data and using a Bray-Curtis dissimilarity measure, 9999 permutations of the raw data, and with statistical effects evaluated at $\alpha = 0.05$.

Results

Toolkit development

Modification of the original AS-ISK questionnaire (template) for adaptation to terrestrial animals resulted in changes only to the text for one question, only to the guidance for 14 questions, and to both text and guidance for 10 questions. This resulted in 25 questions being modified out of the 55 in total (i.e. 45.5%), with changes to the text involving all Sections and Categories therein except for the six climate change questions for which only a minor removal of text from the guidance to Q53 was sufficient. In particular: for Domestication/Cultivation, changes involved the guidance for Qs 1 and 2; for Climate, distribution and introduction risk, only the guidance for Q8; for Invasive elsewhere, the text and guidance for Q11 and guidance for Q13; for Undesirable (or persistence) traits, the text and guidance for Qs 15 and 23, text for Q18, and guidance for Qs 19, 22 and 24; for Resource exploitation, the guidance for Q26; for Reproduction, the guidance for Qs 28, 32 and 34; for Dispersal mechanisms, the text and guidance for Qs 36–39 and guidance for Q41; for Tolerance attributes, the text and guidance for Qs 44, 45 and 48 and guidance for Q47; for Climate change, the guidance for Q53 (Suppl. material 1: Table S1).

The graphical user interface of the TAS-ISK consists of six 'dialogs' (i.e. user interface elements that enable communication and interaction between the user and the software program). Below, a concise description of the dialogs is provided (for a full description see the User Guide downloadable with the toolkit):

- **Start** – TAS-ISK requires a spreadsheet (Database tab) and offers the options of opening either an Existing or a New spreadsheet. The user can select to carry out the screening in any of the 32 available Language options, noting that the toolkit will open by default in the language of the Excel version installed on the local computer. The Colour scheme of choice (seven options) can also be selected. Two new features (relative to AS-ISK v1) are the Background (tab) shading (light to dark) and the size of the Dialogs view (tab), which automatically resize to adapt to low-resolution screens.
- **Main Assessment Workspace** – This is the core dialog (launched from Start) where all screening-related data information is displayed and data manipulations can

be performed (i.e. Wizard, Assessment, Thresholds, Report, Utilities tabs). As a new feature (relative to AS-ISK v1), the Report tab offers the option to generate the report for the screened species in Excel spreadsheet format, PDF or MHTML.

- **Wizard** – This new dialog (relative to AS-ISK v1) allows the assessor to generate the basic template quickly for one or (usually) more screenings as part of the risk screening of several species for the risk assessment area under study.
- **New/Edit** – In this dialog, the assessor provides all details of the screened species, either by creating a new screening, editing an existent screening, or batch-editing multiple screenings.
- **Replicate** – In this dialog, replication of a screening selected from the Main Assessment Workspace is generally performed as part of the risk screening of several species for the risk assessment area under study.
- **Q&A** – In this dialog, the screening for the species selected from the Main Assessment Workspace is carried out by responding to the 55 questions, ranking the level of confidence/certainty associated with the response, and providing references and/or other information as justification for each question-related response.

Trial screenings

The highest scoring (*a priori* invasive) species were *Lumbricus rubellus* for the Aegean region of Turkey and *Arthurdendyus triangulatus* for Croatia (Table 2). Both species were recognised as 'invasive elsewhere' and obtained the highest score amongst all screened species for the Biology/Ecology section, with *Arthurdendyus triangulatus* also achieving the highest possible increase (+12 points) for the CCA. The other *a priori* invasive species *Arion vulgaris*, *Diabrotica virgifera virgifera*, *Ditylenchus destructor* and *Phasianus colchicus*, all screened for Croatia, and *Ammotragus lervia*, screened for Europe, obtained BRA scores ≥ 22. These species have been recognised as invasive elsewhere and gained overall high scores for their Undesirable (or persistence) traits. The CCA increased the BRA score for *Ammotragus lervia*, *Diabrotica virgifera virgifera* and *Ditylenchus destructor*, but decreased that of *Arion vulgaris*. At the same time, there was no change in outcome score relative to the BRA (cf. BRA+CCA) for *Phasianus colchicus*. For *Hemidactylus frenatus* screened for the Pannonian region of Hungary, there was a substantial increase in the BRA+CCA relative to the BRA score. Finally, the *a priori* non-invasive *Bombina variegata* screened for Anatolia (Turkey) obtained the lowest outcome score of all species (Table 2). The TAS-ISK combined report for the nine screened species is provided as Suppl. material 2.

The highest confidence factor in responses for the BRA was found for *Diabrotica virgifera virgifera* and *Ditylenchus destructor*, and for the CCA for *Ammotragus lervia* and *Arion vulgaris*. *Bombina variegata* and *Phasianus colchicus* had confidence factors for both components below 0.60 (Table 2). The mean CF_{Total} was 0.697 ± 0.034 SE, the mean CF_{BRA} 0.699 ± 0.036 SE, and the mean CF_{CCA} 0.672 ± 0.043 SE, and there were no differences in CF between BRA and CCA (F^{MC} = 0.002, P^{MC} = 0.970; MC = Monte Carlo permutational value, best for small sample sizes).

Table 2. Scoring output for the terrestrial animal taxa screened with the TAS-ISK (see Table 1). BRA = Basic Risk Assessment; CCA = Climate Change Assessment. See also Suppl. material 1: Table S1.

Section/Category	*Ammotragus lervia*	*Phasianus colchicus*	*Hemidactylus frenatus*	*Bombina variegata*	*Lumbricus rubellus*	*Diabrotica virgifera virgifera*	*Arion vulgaris*	*Ditylenchus destructor*	*Arthurdendyus triangulatus*
A. Biogeography/Historical	12	15.5	9	5	24	21	15	19	17
1. Domestication/ Cultivation	2	4	4	2	4	2	0	0	–2
2. Climate, distribution and introduction risk	1	1	2	1	2	1	1	1	1
3. Invasive elsewhere	9	10.5	3	2	18	18	14	18	18
B. Biology/Ecology	17	12	15	3	24	9	7	12	19
4. Undesirable (or persistence) traits	8	8	5	3	6	9	4	7	5
5. Resource exploitation	5	5	7	0	5	0	0	7	2
6. Reproduction	1	5	1	1	6	–3	6	–2	4
7. Dispersal mechanisms	–2	–2	2	–3	4	0	–1	–1	2
8. Tolerance attributes	5	–4	0	2	3	3	–2	1	6
BRA Score	**29**	**27.5**	**24**	**8**	**48**	**30**	**22**	**31**	**36**
C. Climate change	4	0	8	2	6	4	–10	2	12
BRA+CCA Score	**33**	**27.5**	**32**	**10**	**54**	**34**	**12**	**33**	**48**
Confidence									
BRA	0.76	0.57	0.59	0.59	0.61	0.82	0.79	0.81	0.78
CCA	0.92	0.58	0.71	0.50	0.54	0.71	0.79	0.63	0.67
Total (BRA+CCA)	0.77	0.57	0.61	0.58	0.60	0.81	0.79	0.79	0.76

Discussion

Toolkit development

The successful employment of the WRA-type toolkits for screening weeds (cf. WRA and its derivatives) and aquatic organisms (cf. WRA, -ISK toolkits and AS-ISK) is testified by the vast number of applications worldwide (Gordon et al. 2008; Vilizzi et al. 2019, 2021, 2022). An additional value of these risk screening applications is the high degree of accuracy (cf. discriminatory power *sensu* Hosmer et al. 2013) achieved in the classification of low-to-medium- and high-risk species for a variety of risk assessment areas in different climates and biogeographic regions and, since the development of the AS-ISK, under both current and predicted future climate conditions (Vilizzi et al. 2019, 2021, 2022).

The advantages of a multilingual decision-support toolkit have been described in detail in Copp et al. (2021). In the case of the screening of terrestrial animals with the TAS-ISK, the same benefits are expected in terms of enhanced communication of species-specific risk outcomes between assessors (scientists) and decision-makers by providing screening reports in the native language. This has already been exemplified by some of the AS-ISK applications conducted in the native language of the country's risk assessment area (Vilizzi et al. 2021), including publication and discussion of the corresponding risk outcomes also in the native language (i.e. Moghaddas et al. 2020; IAVH 2021; Li et al. 2021; Wei et al. 2021b).

Trial screenings

The risk outcomes for the nine non-native terrestrial animal species screened with the TAS-ISK highlighted which species are likely to pose the greatest threat of invasiveness (e.g. *Lumbricus rubellus* and *Arthurdendyus triangulatus*), hence should be prioritised for full (follow-up) risk assessment and potentially targeted by prevention measures and related management strategies (Copp et al. 2016a). Confidence in the BRA questions was similar to that in the CCA questions, which reflected the large availability of literature resources for the screened species and the overall knowledge/expertise by the assessors in both the screened species and related risk assessment areas.

Lumbricus rubellus was the highest scoring of the species screened – a finding that is likely to apply to risk assessment areas with warm-temperate and continental climate other than Anatolia (Tiunov et al. 2006). *Lumbricus rubellus* has been introduced in many continents outside its native range in Western Europe, but it is considered invasive only in North America and New Zealand (Greiner et al. 2012; Kim et al. 2015). The species' native distribution is still unclear, as it may originate from the Pyrenees, with its native range extending across France, southern Germany, Austria, Hungary and Romania (Gates 1972). The uncertainty about the origin of *L. rubellus* is to be ascribed to the extensive agricultural and fishing activities that have occurred over the last 2000 years involving the unintentional transport of this species in the soil (i.e. by transportation of plants rooted in soil contaminated with different life stages of this species) and as fish bait (Keller et al. 2007; Crumsey et al. 2014). *Lumbricus rubellus* is harmful in forest ecosystems (Crumsey et al. 2014) and its introduction may change soil structure and chemistry, nutrient dynamics, microbial community content, and even plant community composition (Greiner et al. 2012). Furthermore, the species' hermaphroditism, tolerance of low pH (3.0–7.7) and resistance to low temperatures are all traits that increase the chance for its successful colonisation of novel environments (Tiunov et al. 2006; Wironen and Moore 2006; Kopp et al. 2012). Climate change appears to increase the competitiveness of *L. rubellus* because of its high tolerance of a wide range of temperatures, though not of a reduction in soil water content (Singh et al. 2019).

The second highest scoring species *Arthurdendyus triangulatus* is not yet found in Croatia (the risk assessment area in this study). The species' high risk of invasiveness confirms recent findings using a different risk assessment tool (Thunnissen et al. 2022) and justifies its inclusion in the Invasive Alien Species of Union Concern C/2019/5360 (European Commission 2019). *Arthurdendyus triangulatus* is a free-living terrestrial flatworm native to New Zealand introduced mainly by trade in containerised plants to the British Isles and the Faroe Islands (Murchie and Gordon 2013). This species is considered harmful mainly due to its predation on earthworms with consequent reduction of soil fertility and earthworm-feeding wildlife (Thunnissen et al. 2022). Based on the Köppen-Geiger climate classification system (Peel et al. 2007), *A. triangulatus* could become established in the northern part of Europe including The Netherlands, Denmark, Sweden and also Iceland due to its tolerance of the *Cfb*-type (warm-temperate, fully humid, warm summer) climate (Boag and Yeates 2001; Thunnissen et al. 2022). As this species prefers *Cs*-type (i.e. warm-temperate) climate conditions (typical of its native range on the South Island of New Zealand), it is very likely to establish in Croatia, where a similar climate is present. Although *A. triangulatus* is expected to become less widespread in the U.K. due to climate change (Hulme 2017), in Croatia it may considerably increase its establishment success as winter temperatures in New Zealand are milder compared to other areas of similar latitude (Sturman and Wanner 2001).

The two agricultural pests *Ditylenchus destructor* (not yet present in Croatia) and *Diabrotica virgifera virgifera* (already introduced to Croatia) gained similarly high BRA and BRA+CCA scores. *Ditylenchus destructor* and *D. virgifera virgifera* may cause severe crop damage resulting in financial losses and management expenditures (Tinsley et al. 2013; Benjamin et al. 2018). *Ditylenchus destructor* is a harmful endoparasite of roots and underground-modified plant parts in Europe and North America and is characterised by behavioural plasticity (Spencer et al. 2009; EFSA Panel on Plant Health 2016). Economically, it is the most important pest of the potato *Solanum tuberosum*, although it acts also as a pest of the sweet potato *Bulbous iris*, cultivated mushrooms, garlic *Allium sativum*, and several other cultivated plants (EFSA Panel on Plant Health 2016; Dobosz et al. 2020). Although the impact of *D. destructor* on crops in Europe is negligible due to precautionary measures, in Australia this species is regarded as posing a potentially high risk of invasiveness (Singh et al. 2015; EFSA PLH Panel 2016). Plants for potting are a pathway for the introduction and spread of *D. destructor*, which may cause severe impacts on their intended use. Climate conditions in Europe are favourable to the completion of the species' life cycle, and all of its developmental stages can overwinter successfully throughout Europe (EFSA Panel on Plant Health 2016). *Diabrotica virgifera virgifera* was introduced by at least five independent events from northern USA into Europe (Ciosi et al. 2008), where it is currently successfully established, including in the risk assessment area of Croatia (Lemic et al. 2015). This species is a major pest of corn *Zea mays* but may also affect alternative host species such as soybean *Glycine max* or crops of pumpkin *Cucurbita* sp. (Manole et al. 2017a, b). *Diabrotica virgifera virgifera* poses a challenge to management actions because of its invasive nature and adaptability (Toepfer and Kuhlmann 2006; Toth et al. 2020). Climate is one of the most critical environmental factors for the species' colonisation success (Aragón et al. 2010; Dupin et al. 2011), and as a result of climate change the future distribution of this species may extend northward with the resulting risk of outbreaks at higher latitudes (Aragón and Lobo 2012).

Ammotragus lervia is native to North Africa and established in Croatia, Czechia, Italy and Spain following intentional introductions for hunting purposes (Šprem et al. 2020). *Phasianus colchicus*, partly native to Europe, has a long history of introductions and re-introductions with populations established across the continent (Ashrafzadeh et al. 2021). Both *A. lervia* and *P. colchicus* are highly adaptable and plastic in their use of available food resources, resulting in their distribution expanding rapidly (Hoodless et al. 2001; Šprem et al. 2020). *Phasianus colchicus* is already widespread across Europe including the risk assessment area (Croatia), where it may be favoured by proximity to human-affected land cover (i.e. agriculture, orchards and plantation forests; Ashoori et al. 2018). It has been observed that populations of *P. colchicus* in Croatia have been declining for the past 30 years. However, intended population reinforcements with captive-bred individuals may have negatively affected population size by outbreeding depression, introduction and fast spread of diseases and parasites from birds introduced from foreign sources (Ashrafzadeh et al. 2021). As a result, it seems that further population expansion of this species is not to be expected under current conditions. Also, the distributional range of *P. colchicus* already covers a variety of climate conditions and habitats (Ashoori et al. 2018); hence, further benefits in terms of range expansion under climate change conditions in the risk assessment area remain low. On the contrary, the intense desertification process that is taking place in Mediterranean regions (cf. south-east Spain) as a result of lowered rainfall regimes and increased mean annual temperatures, may result in substantial habitat changes that may favour the expansion of a desert caprid such as *A. lervia* (Acevedo et al. 2007). Thus, particularly in the Mediterranean region of European countries, the threat posed by *A. lervia* population expansion under future climate conditions may become higher.

The native distributional range of *Arion vulgaris* is still uncertain as this species is thought to be native to the Iberian Peninsula (Zemanova et al. 2016) and southern France (Zając et al. 2020). *Arion vulgaris* has extended its distributional range to several European countries (Zemanova et al. 2016) and is classified as one of the 100 most invasive terrestrial invertebrate species in Europe (Vilà et al. 2009). *Arion vulgaris* may pose severe damage to agriculture and horticulture, is responsible for the defoliation of wild plants and trees and has also caused severe impacts in terms of decline in abundance and also disappearance of its congener red slug *A. rufus* as a result of hybridisation (Zemanova et al. 2017). However, mitochondrial diversity of *A. vulgaris* is lower than that of its congeners with a weak association of genetic structuring amongst geographically distant populations in Europe, which suggests a human contribution to the species' ongoing expansion (Zemanova et al. 2016). Based on predicted future temperature increase scenarios for Europe, the broad range of suitable areas for the establishment of *A. vulgaris* may slightly decrease (Zemanova et al. 2018).

There is still no evidence of established populations in Europe of *Hemidactylus frenatus*, which is native to Southeast Asia, although specimens have been recorded in Italy and Portugal as hitchhikers (Weterings and Vetter 2018). This species has been classified as highly invasive in tropical regions of America, Africa, Asia and Australasia (Lei and Booth 2014) due to its competition for food and space with native geckos and transmission of endo- and ecto-parasitic mites (Dame and Petren 2006; Diaz et al. 2020). Recently, several adult specimens of *H. frenatus* were found in Hungary (B. Szajbert, unpulbished data) but it was assumed that this species cannot overwinter outdoors due to its intolerance to the low winter temperatures present in the Pannonian region (Lei and Booth 2014). However, it was recently noted that *H. frenatus* captured in winter has cold tolerances 1–2 °C lower than those captured in summer, suggesting that tropical invaders can adjust their temperature tolerance downwards via phenotypic plasticity (Lapwong et al. 2021). Such changes may allow tropical invaders to expand their geographic range into colder regions of their non-native ranges (Lapwong et al. 2021). This could increase the probability of establishment of *H. frenatus* in the Pannonian region of Hungary under future climate change conditions (Rödder et al. 2008).

The lowest scoring species *Bombina variegata* is protected under the EU Habitat Directive and has been classified as 'Least concern' in the IUCN Red List of Threatened Species since 2004 (Kuzmin et al. 2009). The Atlantic and continental populations of *B. variegata* are classified as in 'bad' condition and others in 'poor' condition, with only a Greek lineage of this species being reported as self-sustaining on a long-term basis and classified as in 'good' condition (https://eunis.eea.europa.eu/species/638#threat_status). The *B. variegata* lineage (subspecies *B. variegata scabra*) originating from Greece (Sotiropoulos 2020) has recently extended its distributional range to Kurtkaya-Enez (Edirne) in Turkey, where it has established self-sustaining populations (Bülbül et al. 2016). According to the Köppen-Geiger climate system, areas with suitable climate conditions will increase in the risk assessment area of Anatolia (Rubel and Kottek 2010), thereby favouring the dispersal of *B. variegata*. This species has been introduced to Great Britain and Northern Ireland (Roy et al. 2020), where no detrimental impacts have been observed. The lowest score amongst the screened species obtained by *B. variegata* in this study is a further indicator of the applicability and reliability of the newly released TAS-ISK.

Conclusions

Given the current dearth of risk screening applications for non-native terrestrial animals (but see Baiwy et al. 2015; Schaffner and Ries 2019; Ries et al. 2021; Thunnissen et al. 2022), it is anticipated that the availability of the TAS-ISK as a multilingual turnkey application will allow for a 'quantum leap' in this field of research in conservation biology. Accordingly, prospective applications of this newly released decision-support tool may focus on: (i) lists of potentially invasive non-native species (both extant and horizon) for selected risk assessment areas, which would allow for local 'calibration' (i.e. setting of a threshold to distinguish between low-to-medium and high-risk species) (e.g. Clarke et al. 2020; Interesova et al. 2020; Killi et al. 2020; Uyan et al. 2020; Li et al. 2021; Moghaddas et al. 2021; Radočaj et al. 2021; Ruykys et al. 2021; Wei et al. 2021a, b), (ii) global (meta-analytical) studies for setting taxonomic group and/or climate-specific thresholds (e.g. Tarkan et al. 2021; Vilizzi et al. 2021), and (iii) individual non-native and (potentially) invasive species regarded as 'high priority' in terms of e.g. importation/commercial exploitation/evaluation of existing impacts for a specific risk assessment area (e.g. Castellanos-Galindo et al. 2018; Suresh et al. 2019; Baduy et al. 2020; Zięba et al. 2020; Haubrock et al. 2021; Kumar et al. 2021; Yoğurtçuoğlu et al. 2021).

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References

Acevedo P, Cassinello J, Hortal J, Gortázar C (2007) Invasive exotic aoudad (*Ammotragus lervia*) as a major threat to native Iberian ibex (*Capra pyrenaica*): A habitat suitability model approach. Diversity and Distributions 13(5): 587–597. https://doi.org/10.1111/j.1472-4642.2007.00374.x

Aragón P, Lobo JM (2012) Predicted effect of climate change on the invasibility and distribution of the Western corn root-worm. Agricultural and Forest Entomology 14(1): 13–18. https://doi.org/10.1111/j.1461-9563.2011.00532.x

Aragón P, Baselga A, Lobo JM (2010) Global estimation of invasion risk zones for the western corn rootworm *Diabrotica virgifera virgifera*: Integrating distribution models and physiological thresholds to assess climatic favourability. Journal of Applied Ecology 47(5): 1026–1035. https://doi.org/10.1111/j.1365-2664.2010.01847.x

Ashoori A, Kafash A, Varasteh Moradi H, Yousefi M, Kamyab H, Behdarvand N, Mohammadi S (2018) Habitat modeling of the common pheasant *Phasianus colchicus* (Galliformes: Phasianidae) in a highly modified landscape: application of species distribution models in the study of a poorly documented bird in Iran. The European Zoological Journal 85(1): 372–380. https://doi.org/10.1080/24750263.2018.1510994

Ashrafzadeh MR, Khosravi R, Fernandes C, Aguayo C, Bagi Z, Lavadinović VM, Szendrei L, Beuković D, Mihalik B, Kusza S (2021) Assessing the origin, genetic structure and demographic history of the common pheasant (*Phasianus colchicus*) in the introduced European range. Scientific Reports 11(1): e21721. https://doi.org/10.1038/s41598-021-00567-1

Baduy F, Saraiva JL, Ribeiro F, Canario AV, Guerreiro PM (2020) Distribution and risk assessment of potential invasiveness of *Australoheros facetus* (Jenyns, 1842) in Portugal. Fishes 5(1): e3. https://doi.org/10.3390/fishes5010003

Baiwy E, Schockert V, Branquart E (2015) Risk analysis of the Fox squirrel, *Sciurus niger*. Risk analysis report of non-native organisms in Belgium. Cellule interdépartementale sur les Espèces invasives (CiEi), DGO3, SPW / Editions, updated version, 34 pp.

Baker RHA, Black R, Copp GH, Haysom KA, Hulme PE, Thomas MB, Brown A, Brown M, Cannon RJC, Ellis J, Ellis M, Ferris R, Glaves P, Gozlan RE, Holt J, Howe L, Knight JD, MacLeod A, Moore NP, Mumford JD, Murphy ST, Parrott D, Sansford CE, Smith GC, St-Hilaire S, Ward NL (2008) The UK risk assessment scheme for all non-native species. In: Rabitsch W, Essl F, Klingenstein F (Eds) Biological Invasions – from Ecology to Conservation. Neobiota 7: 46–57.

Benjamin EO, Grabenweger G, Strasser H, Wesseler J (2018) The socioeconomic benefits of biological control of western corn rootworm *Diabrotica virgifera virgifera* and wireworms *Agriotes* spp. in maize and potatoes for selected European countries. Journal of Plant Diseases and Protection 125(3): 273–285. https://doi.org/10.1007/s41348-018-0156-6

Boag B, Yeates GW (2001) The potential impact of the New Zealand flatworm, a predator of earthworms, in western Europe. Ecological Applications 11(5): 1276–1286. https://doi.org/10.1890/1051-0761(2001)011[1276:TPIOTN]2.0.CO;2

Booy O, Mill AC, Roy HE, Hiley A, Moore N, Robertson P, Baker S, Brazier M, Bue M, Bullock R, Campbell S, Eyre D, Foster F, Hatton-Ellis M, Long J, Macadam C, Morrison-Bell C, Mumford J, Newman J, Parrott D, Payne R, Renals T, Rodgers E, Spencer M, Stebbing P, Sutton-Croft M, Walker KJ, Ward A, Whittaker S, Wyn G (2017) Risk management to prioritise the eradication of new and emerging invasive non-native species. Biological Invasions 19(8): 2401–2417. https://doi.org/10.1007/s10530-017-1451-z

Bülbül U, Kurnaz M, Eroğlu Aİ, Szymura JM, Koç H, Kutrup B (2016) First record of *Bombina variegata* (L., 1758) (Anura: Bombinatoridae) from Turkey. Turkish Journal of Zoology 40: 630–636. https://doi.org/10.3906/zoo-1508-40

Canter LW (1993) Pragmatic suggestions for incorporating risk assessment principles in EIA studies. Environment and Progress 15: 125–138.

Castellanos-Galindo G, Moreno X, Robertson R (2018) Risks to eastern Pacific marine ecosystems from sea-cage mariculture of alien Cobia. Management of Biological Invasions 9(3): 323–327. https://doi.org/10.3391/mbi.2018.9.3.14

Ciosi M, Miller NJ, Kim KS, Giordano R, Estoup A, Guillemaud T (2008) Invasion of Europe by the western corn rootworm, *Diabrotica virgifera virgifera*: Multiple transatlantic introductions with various reductions of genetic diversity. Molecular Ecology 17(16): 3614–3627. https://doi.org/10.1111/j.1365-294X.2008.03866.x

Clarke SA, Vilizzi L, Lee L, Wood LE, Cowie WJ, Burt JA, Mamiit RJ, Hassina A, Davison PI, Fenwick GV, Harmer R, Skóra ME, Kozic S, Aislabie LR, Kennerley A, Le Quesne WJF, Copp GH, Stebbing PD (2020) Identifying potentially invasive non-native marine and brackish water species for the Arabian Gulf and Sea of Oman. Global Change Biology 26(4): 2081–2092. https://doi.org/10.1111/gcb.14964

Copp GH (2013) The Fish Invasiveness Screening Kit (FISK) for non-native freshwater fishes – a summary of current applications. Risk Analysis 33(8): 1394–1396. https://doi.org/10.1111/risa.12095

Copp GH, Garthwaite R, Gozlan RE (2005) Risk identification and assessment of non-native freshwater fishes: concepts and perspectives on protocols for the UK, Cefas Science Technical Report No. 129, Cefas, Lowestoft, 36 pp.

Copp GH, Vilizzi L, Mumford J, Fenwick GM, Godard MJ, Gozlan RE (2009) Calibration of FISK, an invasiveness screening tool for non-native freshwater fishes. Risk Analysis 29(3): 457–467. https://doi.org/10.1111/j.1539-6924.2008.01159.x

Copp GH, Russell IC, Peeler EJ, Gherardi F, Tricarico E, MacLeod A, Cowx IG, Nunn AD, Occhipinti Ambrogi A, Savini D, Mumford JD, Britton JR (2016a) European Non-native Species in Aquaculture Risk Analysis Scheme – a summary of assessment protocols and decision making tools for use of alien species in aquaculture. Fisheries Management and Ecology 23(1): 1–11. https://doi.org/10.1111/fme.12074

Copp GH, Vilizzi L, Tidbury H, Stebbing PD, Tarkan AS, Miossec L, Goulletquer P (2016b) Development of a generic decision-support tool for identifying potentially invasive aquatic taxa: AS-ISK. Management of Biological Invasions 7(4): 343–350. https://doi.org/10.3391/mbi.2016.7.4.04

Copp GH, Vilizzi L, Wei H, Li S, Piria M, Al-Faisal AJ, Almeida D, Atique U, Al-Wazzan Z, Bakiu R, Bašić T, Bui TD, Canning-Clode J, Castro N, Chaichana R, Çoker T, Dashinov D, Ekmekçi FG, Erős T, Ferincz Á, Ferreira T, Giannetto D, Gilles Jr AS, Głowacki Ł, Goulletquer P, Interesova E, Iqbal S, Jakubčinová K, Kanongdate K, Kim JE, Kopecký O, Kostov V, Koutsikos N, Kozic S, Kristan P, Kurita Y, Lee HG, Leuven RSEW, Lipinskaya T, Lukas J, Marchini A, González-Martínez AI, Masson L, Memedemin D, Moghaddas SD, Monteiro J, Mumladze L, Naddafi R, Năvodaru I, Olsson KH, Onikura N, Paganelli D, Pavia Jr RT, Perdikaris C, Pickholtz R, Pietraszewski D, Povž M, Preda C, Ristovska M, Rosíková K, Santos JM, Semenchenko V, Senanan W, Simonović P, Smeti E, Števove B, Švolíková K, Ta KAT, Tarkan AS, Top N, Tricarico E, Uzunova E, Vardakas L, Verreycken H, Zięba G, Mendoza R (2021) Speaking their language – Development of a multilingual decision-support tool for communicating invasive species risks to decision makers and stakeholders. Environmental Modelling & Software 135: e104900. https://doi.org/10.1016/j.envsoft.2020.104900

Crumsey JM, Le Moine JM, Vogel CS, Nadelhoffer KJ (2014) Historical patterns of exotic earthworm distributions inform contemporary associations with soil physical and chemical factors across a northern temperate forest. Soil Biology and Biochemistry 68: 503–514. https://doi.org/10.1016/j.soilbio.2013.10.029

D'hondt B, Vanderhoeven S, Roelandt S, Mayer F, Versteirt V, Adriaens T, Ducheyne E, San Martin G, Grégoire J-C, Stiers I, Quoilin S, Cigar J, Heughebaert A, Branquart E (2015) Harmonia+ and Pandora+: Risk screening tools for potentially invasive plants, animals and their pathogens. Biological Invasions 17(6): 1869–1883. https://doi.org/10.1007/s10530-015-0843-1

Dame EA, Petren K (2006) Behavioural mechanisms of invasion and displacement in Pacific island geckos (*Hemidactylus*). Animal Behaviour 71(5): 1165–1173. https://doi.org/10.1016/j.anbehav.2005.10.009

Dana E, Jeschke J, García-de-Lomas J (2014) Decision tools for managing biological invasions: Existing biases and future needs. Oryx 48(1): 56–63. https://doi.org/10.1017/S0030605312001263

Defra UK (2003) Guidelines for Environmental Risk Assessment and Management. http://www.defra.gov.uk/environment/risk/eramguide/02.htm

Díaz JA, Torres RA, Paternina LE, Santana DJ, Miranda RJ (2020) Traveling with an invader: ectoparasitic mites of *Hemidactylus frenatus* (Squamata: Gekkonidae) in Colombia. Cuadernos de Herpetología 34(1): 79–82. https://doi.org/10.31017/CdH.2020.(2019-027)

Dobosz R, Rybarczyk-Mydłowska K, Winiszewska G (2020) Occurrence of *Ditylenchus destructor* Thorne, 1945 on a sand dune of the Baltic Sea. Journal of Plant Protection Research 60: 31–40.

Drolet D, DiBacco C, Locke A, McKenzie CH, McKindsey CW, Moore AM, Webb JL, Therriault TW (2016) Evaluation of a new screening-level risk assessment tool applied to non-indigenous marine invertebrates in Canadian coastal waters. Biological Invasions 18(1): 279–294. https://doi.org/10.1007/s10530-015-1008-y

Dupin M, Reynaud P, Jarošík V, Baker R, Brunel S, Eyre D, Pergl J, Makowski D (2011) Effects of the training dataset characteristics on the performance of nine species distribution models: Application to *Diabrotica virgifera virgifera.* PLoS ONE 6(6): e20957. https://doi.org/10.1371/journal.pone.0020957

EFSA Panel on Plant Health (PLH), Jeger M, Bragard C, Caffier D, Candresse T, Chatzivassiliou E, Dehnen-Schmutz K, Gilioli G, Grégoire J-C, Jaques Miret JA, MacLeod A, Navajas Navarro M, Niere B, Parnell S, Potting R, Rafoss T, Rossi V, Van Bruggen A, Van Der Werf W, West J, Winter S, Mosbach-Schulz O, Urek G (2016) Scientific opinion on the risk to plant health of *Ditylenchus destructor* for the EU territory. EFSA Journal 14: e04602. https://doi.org/10.2903/j.efsa.2016.4602

EU (2014) Regulation (EU) 1143/2014 of the European Parliament and of the Council of 22 October 2014 on the prevention and management of the introduction and spread of invasive alien species. Official Journal of the European Union, L 317: 35–55. http://eur-lex.europa.eu/legalcontent/EN/TXT/?qid=1417443504720&uri=CELEX:32014R1143

European Commission (2019) Commission Implementing Regulation (EU) 2019/1262 of 25 July 2019 amending Implementing Regulation (EU) 2016/1141 to update the list of invasive alien species of Union concern C/2019/5360. Official Journal of the European Union, L 199(1): 1–4.

Gates GE (1972) Burmese earthworms–An introduction to the systematics and biology of megadrile oligochaetes with special reference to southeast Asia. Transactions of the American Philosophical Society 62(pt. 7): 1–326. https://doi.org/10.2307/1006214

González-Moreno P, Lazzaro L, Vilà M, Preda C, Adriaens T, Bacher S, Brundu G, Copp GH, Essl F, García-Berthou E, Katsanevakis S, Moen TL, Lucy FE, Nentwig W, Roy HE, Srėbalienė G, Talgø V, Vanderhoeven S, Andjelković A, Arbačiauskas K, Auger-Rozenberg M-A, Bae M-J, Bariche M, Boets P, Boieiro M, Borges PA, Canning-Clode J, Cardigos F, Chartosia N, Cottier-Cook EJ, Crocetta F, D'hondt B, Foggi B, Follak S, Gallardo B, Gammelmo Ø, Giakoumi S, Giuliani C, Guillaume F, Jelaska LŠ, Jeschke JM, Jover M, Juárez-Escario A, Kalogirou S, Kočić A, Kytinou E, Laverty C, Lozano V, Maceda-Veiga A, Marchante E, Marchante H, Martinou AF, Meyer S, Minchin D, Montero-Castaño A, Morais MC, Morales-Rodriguez C, Muhthassim N, Nagy ZÁ, Ogris N, Onen H, Pergl J, Puntila R, Rabitsch W, Ramburn TT, Rego C, Reichenbach F, Romeralo C, Saul W-C, Schrader

G, Sheehan R, Simonović P, Skolka M, Soares AO, Sundheim L, Tarkan AS, Tomov R, Tricarico E, Tsiamis K, Uludağ A, van Valkenburg J, Verreycken H, Vettraino AM, Vilar L, Wiig Ø, Witzell J, Zanetta A, Kenis M (2019) Consistency of impact assessment protocols for non-native species. NeoBiota 44: 1–25. https://doi.org/10.3897/neobiota.44.31650

Gordon DR, Onderdonk DA, Fox AM, Stocker RK (2008) Consistent accuracy of the Australian weed risk assessment system across varied geographies. Diversity and Distributions 14(2): 234–242. https://doi.org/10.1111/j.1472-4642.2007.00460.x

Greiner HG, Kashian DR, Tiegs SD (2012) Impacts of invasive Asian (*Amynthas hilgendorfi*) and European (*Lumbricus rubellus*) earthworms in a North American temperate deciduous forest. Biological Invasions 14(10): 2017–2027. https://doi.org/10.1007/s10530-012-0208-y

Haubrock PJ, Copp GH, Johović I, Balzani P, Inghilesi AF, Nocita A, Tricarico E (2021) North American channel catfish, *Ictalurus punctatus*: A neglected but potentially invasive freshwater fish species. Biological Invasions 23(5): 1563–1576. https://doi.org/10.1007/s10530-021-02459-x

Hill JE, Copp GH, Hardin S, Lawson KM, Lawson Jr LL, Tuckett QM, Vilizzi L, Watson CA (2020) Comparing apples to oranges and other misrepresentations of the risk screening tools FISK and AS-ISK–a rebuttal of Marcot et al. (2019). Management of Biological Invasions 11(2): 325–341. https://doi.org/10.3391/mbi.2020.11.2.10

Hoodless AN, Draycott RAH, Ludiman MN, Robertson PA (2001) Spring foraging behaviour and diet of released pheasants (*Phasianus colchicus*) in the United Kingdom. Game and Wildlife Science 18: 375–386.

Hosmer Jr DW, Lemeshow S, Sturdivant RX (2013) Applied logistic regression. John Wiley & Sons, Hoboken, 511 pp. https://doi.org/10.1002/9781118548387

Hulme PE (2017) Climate change and biological invasions: Evidence, expectations, and response options. Biological Reviews 92(3): 1297–1313. https://doi.org/10.1111/brv.12282

IAVH (2021) El pez basa, panga o pangasius, *Pangasianodon hypophthalmus* (Sauvage, 1878) (Siluriformes: Pangasiidae) en Colombia. Instituto de Investigación de Recurso Biológicos Alexander von Humboldt, Ministerio de Ambiente y Desarrollo Sostenible. Bogotá, D. C., Colombia. http://repository.humboldt.org.co/bitstream/handle/20.500.11761/35863/El%20pez%20pasa%2c%20panga%20o%20pangasius%20en%20Colombia%20112021.pdf?sequence=4&isAllowed=y

Interesova E, Vilizzi L, Copp GH (2020) Risk screening of the potential invasiveness of non-native freshwater fishes in the River Ob basin (West Siberian Plain, Russia). Regional Environmental Change 20(2): e64. https://doi.org/10.1007/s10113-020-01644-3

Keller RP, Cox AN, Van Loon C, Lodge DM, Herborg L-M, Rothlisberger J (2007) From bait shops to the forest floor: Earthworm use and disposal by anglers. American Midland Naturalist 158(2): 321–328. https://doi.org/10.1674/0003-0031(2007)158[321:FBSTTF]2.0.CO;2

Killi N, Tarkan AS, Kozic S, Copp GH, Davison PI, Vilizzi L (2020) Risk screening of the potential invasiveness of non-native jellyfishes in the Mediterranean Sea. Marine Pollution Bulletin 150: e110728. https://doi.org/10.1016/j.marpolbul.2019.110728

Kim Y-N, Robinson B, Boyer S, Zhong H-T, Dickinson N (2015) Interactions of native and introduced earthworms with soils and plant rhizospheres in production landscapes of New Zealand. Applied Soil Ecology 96: 141–150. https://doi.org/10.1016/j.apsoil.2015.07.008

Kopecký O, Bílková A, Hamatová V, Kňazovická D, Konrádová L, Kunzová B, Slaměníková J, Slanina O, Šmídová T, Zemancová T (2019) Potential invasion risk of pet traded lizards, snakes, crocodiles, and tuatara in the EU on the basis of a Risk Assessment Model (RAM) and Aquatic Species Invasiveness Screening Kit (AS-ISK). Diversity (Basel) 11(9): 164. https://doi.org/10.3390/d11090164

Kopp KC, Wolff K, Jokela J (2012) Natural range expansion and human-assisted introduction leave different genetic signatures in a hermaphroditic freshwater snail. Evolutionary Ecology 26(3): 483–498. https://doi.org/10.1007/s10682-011-9504-8

Kumar L, Kumari K, Gogoi P, Manna RK, Madayi RC, Salim SM, Muttanahalli Eregowda V, Raghavan SV, Das BK (2021) Risk analysis of non-native three-spot cichlid, *Amphilophus trimaculatus*, in the River Cauvery (India). Fisheries Management and Ecology 28(2): 158–166. https://doi.org/10.1111/fme.12467

Kuzmin S, Denoël M, Anthony B, Andreone F, Schmidt B, Ogrodowczyk A, Ogielska M, Vogrin M, Cogalniceanu D, Kovács T, Kiss I, Puky M, Vörös J, Tarkhnishvili D, Ananjeva N (2009) *Bombina variegata*. The IUCN Red List of Threatened Species 2009: e.T54451A11148290. https://doi.org/10.2305/IUCN.UK.2009.RLTS.T54451A11148290.en

Lapwong Y, Dejtaradol A, Webb JK (2021) Shifts in thermal tolerance of the invasive Asian house gecko (*Hemidactylus frenatus*) across native and introduced ranges. Biological Invasions 23(4): 989–996. https://doi.org/10.1007/s10530-020-02441-z

Lei J, Booth DT (2014) Temperature, field activity and post-feeding metabolic response in the Asian house gecko, *Hemidactylus frenatus.* Journal of Thermal Biology 45: 175–180. https://doi.org/10.1016/j.jtherbio.2014.09.006

Lemic D, Mikac KM, Ivkosic SA, Bažok R (2015) The Temporal and Spatial Invasion Genetics of the Western Corn Rootworm (Coleoptera: Chrysomelidae) in Southern Europe. PLoS ONE 10(9): e0138796. https://doi.org/10.1371/journal.pone.0138796

Li XJ, Tang WQ, Zhao YH (2021) Risk analysis of fish invasion in Haihe River Basin caused by the central route of the South-to-North Water Diversion Project. Shengwu Duoyangxing 29(10): 1336–1347. https://doi.org/10.17520/biods.2021130

Manole T, Chireceanu C, Teodoru A (2017a) Current status of *Diabrotica virgifera virgifera* LeConte, 1868 (Coleoptera: Chrysomelidae) in Romania. Acta Zoologica Bulgarica (Suppl. 9): 143–148. http://www.acta-zoologica-bulgarica.eu/downloads/acta-zoologica-bulgarica/2017/supplement-9-143-148.pdf

Manole T, Chireceanu C, Teodoru A (2017b) The broadening of distribution of the invasive species *Diabrotica virgifera virgifera* Leconte in the area of Muntenia region under specific climatic and trophic conditions. Scientific Papers - Series A. Agronomy (Basel) 60: 495–499. http://agronomyjournal.usamv.ro/pdf/2017/vol2017.pdf

Marcot BG, Hoff MH, Martin CD, Jewell SD, Givens CE (2019) A decision support system for identifying potentially invasive and injurious freshwater fishes. Management of Biological Invasions 10(2): 200–226. https://doi.org/10.3391/mbi.2019.10.2.01

Matthies M, Giupponi C, Ostendorf B (2007) Environmental decision support systems: Current issues, methods and tools. Environmental Modelling & Software 22(2): 123–127. https://doi.org/10.1016/j.envsoft.2005.09.005

Moghaddas SD, Abdoli A, Kiabi BH, Rahmani H (2020) Risk assessment of the potential invasiveness of *Coptodon zillii* (Gervais, 1848) in Anzali Wetland using AS-ISK. Environmental Sciences 18(2): 255–270. [In Persian] https://doi.org/10.29252/envs.18.2.255

Moghaddas SD, Abdoli A, Kiabi BH, Rahmani H, Vilizzi L, Copp GH (2021) Identifying invasive fish species threats to RAMSAR wetland sites in the Caspian Sea region–A case study of the Anzali Wetland Complex (Iran). Fisheries Management and Ecology 28L(1): 28–39. https://doi.org/10.1111/fme.12453

Mumford JD, Booy O, Baker RHA, Rees M, Copp GH, Black K, Holt J, Leach AW, Hartley M (2010) Non-native species risk assessment in Great Britain. In: Evans A (Ed.) What Makes an Alien Invasive? Risk and Policy Responses. Aspects of Applied Biology, 104, Association of Applied Biologists, 49–54.

Murchie AK, Gordon AW (2013) The impact of the 'New Zealand flatworm', *Arthurdendyus triangulatus*, on earthworm populations in the field. Biological Invasions 15(3): 569–586. https://doi.org/10.1007/s10530-012-0309-7

Peel MC, Finlayson BL, McMahon TA (2007) Updated world map of the Köppen-Geiger climate classification. Hydrology and Earth System Sciences 11(5): 1633–1644. https://doi.org/10.5194/hess-11-1633-2007

Pheloung PC, Williams PA, Halloy SR (1999) A weed risk assessment model for use as a biosecurity tool evaluating plant introductions. Journal of Environmental Management 57(4): 239–251. https://doi.org/10.1006/jema.1999.0297

Piria M, Copp GH, Dick JT, Duplić A, Groom Q, Jelić D, Lucy FE, Roy HE, Sarat E, Simonović P, Tomljanović T, Tricarico E, Weinlander M, Adámek Z, Bedolfe S, Coughlan NE, Davis E, Dobrzycka-Krahel A, Grgić Z, Kırankaya ŞG, Ekmekçi FG, Lajtner J, Lukas JAY, Koutsikos N, Mennen GJ, Mitić B, Pastorino P, Ruokonen TJ, Skóra ME, Smith ERC, Šprem N, Tarkan AS, Treer T, Vardakas L, Vehanen T, Vilizzi L, Zanella D, Caffrey JM (2017) Tackling invasive alien species in Europe II: Threats and opportunities until 2020. Management of Biological Invasions 8(3): 273–286. https://doi.org/10.3391/mbi.2017.8.3.02

Radočaj T, Špelić I, Vilizzi L, Povž M, Piria M (2021) Identifying threats from introduced and translocated non-native freshwater fishes in Croatia and Slovenia under current and future climatic conditions. Global Ecology and Conservation 27: e01520. https://doi.org/10.1016/j.gecco.2021.e01520

Ries C, Schneider N, Vitali F, Weigand A (2021) First records and distribution of the invasive alien hornet *Vespa velutina nigrithorax* du Buysson, 1905 (Hymenoptera: Vespidae) in Luxembourg. Bulletin (Societe des Naturalistes Luxembourgeois) 123: 181–193. https://www.snl.lu/publications/bulletin/SNL_2021_123_181_193.pdf

Robertson PA, Mill AC, Adriaens T, Moore N, Vanderhoeven S, Essl F, Booy O (2021) Risk Management Assessment Improves the Cost-Effectiveness of Invasive Species Prioritisation. Biology (Basel) 10(12): e1320. https://doi.org/10.3390/biology10121320

Rödder D, Solé M, Böhme W (2008) Predicting the potential distributions of two alien invasive Housegeckos (Gekkonidae: *Hemidactylus frenatus*, *Hemidactylus mabouia*). North-Western Journal of Zoology 4: 236–246.

Roy H, Rorke S, Wong LJ, Pagad S (2020) Global Register of Introduced and Invasive Species - Great Britain. Version 1.7. Invasive Species Specialist Group ISSG.

Rubel F, Kottek M (2010) Observed and projected climate shifts 1901–2100 depicted by world maps of the Köppen-Geiger climate classification. Meteorologische Zeitschrift (Berlin) 19(2): 135–141. https://doi.org/10.1127/0941-2948/2010/0430

Ruykys L, Ta KAT, Bui TD, Vilizzi L, Copp GH (2021) Risk screening of the potential invasiveness of non-native aquatic species in Vietnam. Biological Invasions 23(7): 2047–2060. https://doi.org/10.1007/s10530-020-02430-2

Schaffner F, Ries C (2019) First evidence and distribution of the invasive alien mosquito Aedes japonicus (Theobald, 1901) in Luxembourg. Bulletin (Societe des Naturalistes Luxembourgeois) 121: 169–183. https://www.snl.lu/publications/bulletin/SNL_2019_121_169_183.pdf

Singh SK, Ash GJ, Hodda M (2015) Keeping 'one step ahead' of invasive species: Using an integrated framework to screen and target species for detailed biosecurity risk assessment. Biological Invasions 17(4): 1069–1086. https://doi.org/10.1007/s10530-014-0776-0

Singh J, Schädler M, Demetrio W, Brown GG, Eisenhauer N (2019) Climate change effects on earthworms - a review. Soil Organisms 91: 114–138.

Sotiropoulos K (2020) The Amphibians. In: Pafilis P (Ed.) The fauna of Greece. Biology and management of wild fauna. Broken Hills Publishers Ltd, Nicosia, Cyprus, 579–623. [in Greek]

Spencer JL, Hibbard BE, Moeser J, Onstad DW (2009) Behaviour and ecology of the western corn rootworm (*Diabrotica virgifera virgifera* LeConte). Agricultural and Forest Entomology 11(1): 9–27. https://doi.org/10.1111/j.1461-9563.2008.00399.x

Šprem N, Gančević P, Safner T, Jerina K, Cassinello J (2020) Barbary sheep *Ammotragus lervia* (Pallas, 1777). In: Hackländer K, Zachos FE (Eds) Handbook of the Mammals of Europe. Springer, Cham, 1–14. https://doi.org/10.1007/978-3-319-65038-8_35-1

Sturman A, Wanner H (2001) A comparative review of the weather and climate of the Southern Alps of New Zealand and the European Alps. Mountain Research and Development 21(4): 359–369. https://doi.org/10.1659/0276-4741(2001)021[0359:ACROTW]2.0.CO;2

Suresh VR, Ekka A, Biswas DK, Sahu SK, Yousuf A, Das S (2019) Vermiculated sailfin catfish, *Pterygoplichthys disjunctivus* (Actinopterygii: Siluriformes: Loricariidae): invasion, biology, and initial impacts in east Kolkata Wetlands, India. Acta Ichthyologica et Piscatoria 49(3): 221–233. https://doi.org/10.3750/AIEP/02551

Tarkan AS, Tricarico E, Vilizzi L, Bilge G, Ekmekçi FG, Filiz H, Giannetto D, İlhan A, Killi N, Kırankaya ŞG, Koutsikos N, Kozic S, Kurtul I, Lazzaro L, Marchini A, Occhipinti-Ambrogi A, Perdikaris C, Piria M, Pompei L, Sari H, Smeti E, Stasolla G, Top N, Tsiamis K, Vardakas L, Yapici S, Yoğurtçuoğlu B, Copp GH (2021) Risk of invasiveness of non-native aquatic species in the eastern Mediterranean region under current and projected climate conditions. The European Zoological Journal 88(1): 1130–1143. https://doi.org/10.1080/24750263.2021.1980624

Thunnissen NW, de Waart SA, Collas FPL, Jongejans E, Jan Hendriks A, van der Velde G, Leuven RSEW (2022) Risk screening and management of alien terrestrial planarians in The Netherlands. Management of Biological Invasions 13(1): 81–100. https://doi.org/10.3391/mbi.2022.13.1.05

Tinsley NA, Estes RE, Gray ME (2013) Validation of a nested error component model to estimate damage caused by corn rootworm larvae. Journal of Applied Entomology 137(3): 161–169. https://doi.org/10.1111/j.1439-0418.2012.01736.x

Tiunov AV, Hale CM, Holdsworth AR, Vsevolodova-Perel TS (2006) Invasion patterns of Lumbricidae into the previously earthworm-free areas of northeastern Europe and the western Great Lakes region of North America. In: Hendrit PF (Ed.) Biological Invasions Belowground: Earthworms as Invasive Species. Springer, Dordrecht, 22–34. https://doi.org/10.1007/978-1-4020-5429-7_4

Toepfer S, Kuhlmann U (2006) Constructing life-tables for the invasive maize pest *Diabrotica virgifera virgifera* (Col.; Chrysomelidae) in Europe. Journal of Applied Entomology 130(4): 193–205. https://doi.org/10.1111/j.1439-0418.2006.01060.x

Toth S, Szalai M, Kiss J, Toepfer S (2020) Missing temporal effects of soil insecticides and entomopathogenic nematodes in reducing the maize pest *Diabrotica virgifera virgifera.* Journal of Pest Science 93(2): 767–781. https://doi.org/10.1007/s10340-019-01185-7

Uyan U, Oh C-W, Tarkan AS, Top N, Copp GH, Vilizzi L (2020) Risk screening of the potential invasiveness of non-native marine fishes in South Korea. Marine Pollution Bulletin 153: e111018. https://doi.org/10.1016/j.marpolbul.2020.111018

Vilà M, Basnou C, Gollasch S, Josefsson M, Pergl J, Scalera R (2009) One hundred of the most invasive alien species in Europe. Handbook of Alien Species in Europe. Invading Nature - Springer Series in Invasion Ecology, Vol 3. Springer, Dordrecht, 265–268. https://doi.org/10.1007/978-1-4020-8280-1_12

Vilizzi L, Copp GH, Adamovich B, Almeida D, Chan J, Davison PI, Dembski S, Ekmekçi FG, Ferincz A, Forneck SC, Hill JE, Kim J-E, Koutsikos N, Leuven RSEW, Luna SA, Magalhães F, Marr SM, Mendoza R, Mourão CF, Neal JW, Onikura N, Perdikaris C, Piria M, Poulet N, Puntila R, Range IL, Simonović P, Ribeiro F, Tarkan AS, Troca DFA, Vardakas L, Verreycken H, Vintsek L, Weyl OLF, Yeo DCJ, Zeng Y (2019) A global review and meta-analysis of applications of the freshwater Fish Invasiveness Screening Kit. Reviews in Fish Biology and Fisheries 29(3): 529–568. https://doi.org/10.1007/s11160-019-09562-2

Vilizzi L, Copp GH, Hill JE, Adamovich B, Aislabie L, Akin D, Al-Faisal AJ, Almeida D, Azmai MNA, Bakiu R, Bellati A, Bernier R, Bies JM, Bilge G, Branco P, Bui TD, Canning-Clode J, Cardoso Ramos HA, Castellanos-Galindo GA, Castro N, Chaichana R, Chainho P, Chan J, Cunico AM, Curd A, Dangchana P, Dashinov D, Davison PI, de Camargo MP, Dodd JA, Durland Donahou AL, Edsman L, Ekmekçi FG, Elphinstone-Davis J, Erős T, Evangelista C, Fenwick G, Ferincz Á, Ferreira T, Feunteun E, Filiz H, Forneck SC, Gajduchenko HS, Gama Monteiro J, Gestoso I, Giannetto D, Gilles AS, Gizzi Jr F, Glamuzina B, Glamuzina L, Goldsmit J, Gollasch S, Goulletquer P, Grabowska J, Harmer R, Haubrock PJ, He D, Hean JW, Herczeg G, Howland KL, İlhan A, Interesova E, Jakubčinová K, Jelmert A, Johnsen SI, Kakareko T, Kanongdate K, Killi N, Kim J-E, Kırankaya ŞG, Kňazovická D, Kopecký O, Kostov V, Koutsikos N, Kozic S, Kuljanishvili T, Kumar B, Kumar L, Kurita Y, Kurtul I, Lazzaro L, Lee L, Lehtiniemi M, Leonardi G, Leuven RSEW, Li S, Lipinskaya T, Liu F, Lloyd L, Lorenzoni M, Luna SA, Lyons TJ, Magellan K, Malmstrøm M, Marchini A, Marr SM, Masson G, Masson L, McKenzie CH, Memedemin D, Mendoza R, Minchin D, Miossec L, Moghaddas SD, Moshobane MC, Mumladze L, Naddafi R, Najafi-Majd E, Năstase A, Năvodaru I, Neal JW, Nienhuis S, Nimtim M, Nolan ET, Occhipinti-Ambrogi A, Ojaveer H, Olenin S, Olsson K, Onikura N, O'Shaughnessy K, Paganelli D, Parretti P, Patoka J, Pavia RTB, Pellitteri-Rosa Jr D, Pelletier-Rousseau M, Peralta EM, Perdikaris C, Pietraszewski D, Piria M, Pitois S, Pompei L, Poulet N, Preda C, Puntila-Dodd R, Qashqaei AT, Radočaj T, Rahmani H, Raj S, Reeves D, Ristovska M, Rizevsky V, Robertson DR, Robertson P, Ruykys L, Saba AO, Santos JM, Sarı HM, Segurado P, Semenchenko V, Senanan W, Simard N, Simonović P, Skóra ME, Slovák Švolíková K, Smeti E, Šmídová T, Špelić I, Srėbalienė G, Stasolla G, Stebbing P, Števove B, Suresh VR, Szajbert B, Ta KAT, Tarkan AS, Tempesti J, Therriault TW, Tidbury HJ, Top-Karakuş N, Tricarico E, Troca DFA, Tsiamis K, Tuckett QM, Tutman P, Uyan U, Uzunova E, Vardakas L, Velle G, Verreycken H, Vintsek L, Wei H, Weiperth A, Weyl OLF, Winter ER, Włodarczyk R, Wood LE, Yang R, Yapıcı S, Yeo SSB, Yoğurtçuoğlu B, Yunnie ALE, Zhu Y, Zięba G, Žitňanová K, Clarke S (2021) A global-scale screening of non-native aquatic organisms to identify potentially invasive species under current and future climate conditions. Science of the Total Environment 788: e147868. https://doi.org/10.1016/j.scitotenv.2021.147868

Vilizzi L, Hill JE, Piria M, Copp GH (2022) A protocol for screening potentially invasive non-native species using Weed Risk Assessment-type decision-support toolkits. Science of the Total Environment 832: e154966. https://doi.org/10.1016/j.scitotenv.2022.154966

Walkenbach J (2007) Excel 2007 bible. John Wiley and Sons Inc., New York, 912 pp.

Wei H, Chaichana R, Vilizzi L, Daengchana P, Liu F, Nimtim M, Zhu Y, Li S, Hu Y, Copp GH (2021a) Do non-native ornamental fishes pose a similar level of invasion risk in neighbouring regions of similar current and future climate? Implications for conservation and management. Aquatic Conservation 31(8): 2041–2057. https://doi.org/10.1002/aqc.3609

Wei H, Liu C, Hu Y, Wang X, Mu X, Gu D, Xu M, Fang M (2021b) (in press) Invasiveness identification using Aquatic Species Invasiveness Screening Kit of non-native ornamental fish in China: A case study of non-native Loricariidae species. Journal of Ecology and Rural Environment. https://kns.cnki.net/kcms/detail/32.1766.X.20211015.1941.001.html [In Chinese with an English abstract]

Weterings R, Vetter KC (2018) Invasive house geckos (*Hemidactylus* spp.): Their current, potential and future distribution. Current Zoology 64(5): 559–573. https://doi.org/10.1093/cz/zox052

Wironen M, Moore TR (2006) Exotic earthworm invasion increases soil carbon and nitrogen in an old-growth forest in southern Quebec. Canadian Journal of Forest Research 36(4): 845–854. https://doi.org/10.1139/x06-016

Yoğurtçuoğlu B, Bucak T, Ekmekçi FG, Kaya C, Tarkan AS (2021) Mapping the establishment and invasiveness potential of rainbow trout (*Oncorhynchus mykiss*) in Turkey: With special emphasis on the conservation of native salmonids. Frontiers in Ecology and Evolution 8: e599881. https://doi.org/10.3389/fevo.2020.599881

Zając KS, Hatteland BA, Feldmeyer B, Pfenninger M, Filipiak A, Noble LR, Lachowska-Cierlik D (2020) A comprehensive phylogeographic study of *Arion vulgaris* Moquin-Tandon, 1855 (Gastropoda: Pulmonata: Arionidae) in Europe. Organisms, Diversity & Evolution 20(1): 37–50. https://doi.org/10.1007/s13127-019-00417-z

Zemanova MA, Knop E, Heckel G (2016) Phylogeographic past and invasive presence of *Arion* pest slugs in Europe. Molecular Ecology 25(22): 5747–5764. https://doi.org/10.1111/mec.13860

Zemanova MA, Knop E, Heckel G (2017) Introgressive replacement of natives by invading *Arion* pest slugs. Scientific Reports 7(1): e14908. https://doi.org/10.1038/s41598-017-14619-y

Zemanova MA, Broennimann O, Guisan A, Knop E, Heckel G (2018) Slimy invasion: Climatic niche and current and future biogeography of Arion slug invaders. Diversity and Distributions 24(11): 1627–1640. https://doi.org/10.1111/ddi.12789

Zięba G, Vilizzi L, Copp GH (2020) How likely is *Lepomis gibbosus* to become invasive in Poland under conditions of climate warming? Acta Ichthyologica et Piscatoria 50: 37–51. https://doi.org/10.3750/AIEP/02390

Supplementary material 1

Table S1

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Data type: docx file

Explanation note: List of the 55 questions (Qs) making up the Terrestrial Animal Species Invasiveness Screening Kit (TAS-ISK v2.3.1). Sector codes (in parentheses): C = Commercial; E = Environmental; S = Species or population nuisance traits. Changes of questions relative to AS-ISK v2.3.1: G = Guidance; Q = Question (text). For each Q, the corresponding Question (text), Guidance and choice of Response (with coding as displayed in report and score) are indicated.

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Supplementary material 2

Combined TAS-ISK report

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Data type: pdf file

Explanation note: Combined TAS-ISK report including the nine screenings for the sample terrestrial animal species.

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