Invasive alien species records are exponentially rising across the Earth

Roger Paulo Mormul · **Denner Serafim Vieira** · **Dayani Bailly** · **Karina Fidanza** · **Valéria Flávia Batista da Silva** · **Weferson Júnio da Graça** · **Vanessa Pontara** · **Marcelo Leandro Bueno** · **Sidinei Magela Thomaz** · **Renio Santos Mendes**

Abstract Humans are facilitating the introduction and range expansion of invasive alien species (IAS), which have negatively impacted ecological and socio-economic systems worldwide. Understanding the global diffusion of IAS is important for developing environmental policies and management strategies. We estimate the rate of increase and the doubling times of the number of new records of 178 IAS using a global dataset with c. 3.4 million records obtained over c. 100 years. Here, we show that the number of records of IAS have exponentially increased with a mean double time of c. 14 years across the Earth. For the most analysed species, the number of records increased faster in the non-native than native continents, suggesting that such IAS might be exponentially expanding their range size. We also found that each continent has a taxonomic group with a particular increase in IAS records. Governments and scientists should pay attention to these taxonomic groups to implement appropriate control or management actions. Our study provides an indication that the current local, regional and continental efforts to control invasions may be not sufficient at the global scale. This is a concerning situation given the great number of areas available for invasion worldwide.

R. P. Mormul (✉) · D. Bailly · K. Fidanza ·
W. J. da Graça · S. M. Thomaz
Programa de Pós-Graduação em Ecologia de Ambientes Aquáticos Continentais, Núcleo de Pesquisas em Limnologia, Ictiologia e Aquicultura, Universidade Estadual de Maringá, Av. Colombo, 5790, Bloco H90, Jd. Universitário, 87020-900 Maringá, Paraná, Brazil
e-mail: roger@nupelia.uem.br

D. Bailly
e-mail: dayanibailly@gmail.com

K. Fidanza
e-mail: karina.fidanza@gmail.com

W. J. da Graça
e-mail: weferson@nupelia.uem.br

S. M. Thomaz
e-mail: smthomaz@nupelia.uem.br

D. S. Vieira · R. S. Mendes
Programa de Pós-Graduação em Física, Complex Lab, Universidade Estadual de Maringá, Maringá, Paraná, Brazil
e-mail: dennersvieira@gmail.com

R. S. Mendes
e-mail: rsmendes@dfi.uem.br

V. F. B. da Silva · V. Pontara · M. L. Bueno
Universidade Estadual do Mato Grosso do Sul, Dourados, Mato Grosso do Sul, Brazil
e-mail: vfbsilva19@gmail.com

V. Pontara
e-mail: vanessapontara@gmail.com

M. L. Bueno
e-mail: buenotanica@gmail.com

Introduction

The recognition of the impacts caused by invasive alien species (IAS) and the progressive increase in their records worldwide have increased scientific and social concerns regarding invasions (Blackburn et al. 2014; Kumschick et al. 2015). After overcoming geographical barriers through human-facilitated transport, IAS face environmental filters (Elton 1958; Levine and D'Antonio 1999) during the introduction and establishment stages of the invasion process (sensu Blackburn et al. 2011). During these stages, IAS could face unsuitable habitats or biotic interactions (e.g., biotic resistance) which may limit their success (Rahel 2002; Kennedy et al. 2002). Such barriers to the introduction and establishment of IAS may lead to limited development of IAS populations, driving a saturated curve of IAS records at regional scales (Kelly et al. 2014). However, the abiotic and biotic limitations to IAS success are usually tested and applied at local or regional spatial scales.

When local and regional filters are overcome (Theoharides and Dukes 2007), the increase in number of records over landscapes (e.g., continental areas) can occur. In this case, each new IAS record in different locations could be considered a new source population of propagules that can facilitate further spread. Then, increasing source populations may increase propagule pressure and propagule diversity (Foxcroft et al. 2004; Zenni and Simberloff 2013), which, in turn, cause multiple introductions (Foxcroft et al. 2004; Garnas et al. 2016) and may lead to exponential rates of increase in the number of IAS records at larger spatial scales. Global analyses indicate that the number of extra-continental naturalized species is still increasing in most continents (van Kleunen et al. 2015) and that the annual rate of initial IAS records has increased over the last 200 years worldwide (Seebens et al. 2017). Although these results indicate that the number of IAS is increasing worldwide, there is no information on whether different invasive taxa are still rising their records at a global scale and if so, would be crucial to understand how the long-term trends of such rising records are.

More comprehensive analyses of the rising rates of IAS records at the global scale are urgently needed (van Kleunen et al. 2015) because such analyses are essential for understanding the importance of the increasing number of source populations and the emerging patterns that can lead several taxonomic groups to become cosmopolitan. Here, we applied a spatiotemporal model using records of IAS in their non-native and native ranges to evaluate whether the number of new sites occupied by a given IAS tends to stabilize over time. The exponential (Malthusian) behaviour of our model would suggest that there are no constraints in the IAS records at the large geographical scale and that IAS populations are potentially expanding. The logistic (Verhulst) behaviour would suggest that some constraints are limiting the expansion of IAS populations. Such constraints could be related to biotic and abiotic filters from natural environments and to the efforts from many nations to control the advance of or eradicate IAS according to the ninth goal of the Aichi Biodiversity Targets (CBD 2011). In this sense, we suggest that the logistic behaviour found here might also indicate the success of the local and regional efforts in controlling invasions, thus helping to decrease IAS records at the global scale. Our evaluation is based on a global dataset with c. 3.4 million records of 178 IAS obtained from a historical dataset of c. 100 years that includes molluscs, arachnids, crustaceans, insects, fishes, amphibians, reptiles, birds, mammals, pteridophytes and angiosperms.

Materials and methods

Data compilation

To analyse the spatiotemporal pattern of IAS records worldwide, we conducted an extensive search for occurrences of IAS considered ecological or economic threats in any extent of the globe. We used the Global Invasive Species Database (GISD, http://www.iucngisd.org/gisd/) as a basis to compile a list of IAS that threaten native biodiversity and natural ecosystems worldwide. All continental Metazoa and Metaphyta species in the GISD were considered potential units of analysis. We filtered the species list from the GISD to control biases related to scientific names and range uncertainty. The filtering process

consisted of excluding subspecies (e.g., *Trachemys scripta elegans*), information pertaining only to genera (e.g., *Pinus*), inaccurate taxonomic information (e.g., *Nylanderia (=Paratrechina) pubens*) and species names with the abbreviations 'spp.' (e.g., *Pterygoplichthys* spp.) and 'var' (e.g., *Pueraria montana* var. *lobata*). Considering the raw occurrence data, we also disregarded records devoid of geographical coordinates and those referring to non-valid names, trinomials (subgenera, subspecies, varieties and forms), hybrids and taxonomic names with abbreviations, such as 'sp.', 'spp.', 'aff.', 'cf.' and 'gr.'. This filtering process was conducted to avoid false positives. Then, to ensure that only occurrence data of current valid species names were used in the analyses, we checked the validity of all animal and plant species. For the animal species, we used the information available in the Catalogue of Life (http://www.catalogueoflife.org/) and Integrated Taxonomic Information System (https://www.itis.gov), which are general databases that gather knowledge and opinions from expert taxonomists worldwide in a unified and hierarchically ranked system of taxonomy that embraces the diversity of life (Ruggiero et al. 2015). We also consulted specific databases, such as MolluscaBase (http://www.molluscabase.org) for molluscs, WoRMS (http://www.marinespecies.org) for crustaceans and molluscs, Antbase (https://antbase.org) for hymenopteran insects, FishBase (http://www.fishbase.org/search.php) and the Catalog of Fishes (http://www.calacademy.org/scientists/projects/catalog-of-fishes) for fishes, AmphibiaWeb (http://www.amphibiaweb.org/) for amphibians, the Reptile Database (http://www.reptile-database.org/) for reptiles, Avibase (https://avibase.bsc-eoc.org/avibase.jsp?lang=EN) for birds, and Mammal Species of the World (https://www.departments.bucknell.edu/biology/resources/msw3/) for mammals. For the plant species, we used the Taxonomic Name Resolution Service v4.0 (http://tnrs.iplantcollaborative.org/index.html). This filtering process allowed us to eliminate all taxa with taxonomic problems from the analyses.

After creating the list of IAS using the abovementioned criteria, the occurrence data of the species were obtained from the GBIF (Global Biodiversity Information Facility; http://www.gbif.org/), BISON (Biodiversity Information Serving Our Nation; https://bison.usgs.gov/#home) and SpeciesLink (http://splink.cria.org.br/) databases, which provide georeferenced occurrence data based on information from different natural history museums and herbariums worldwide. We also added information related to vertebrate species from VertNet (http://vertnet.org/).

Data processing

The compiled occurrence data were analysed based on location (latitude and longitude) and the event dates on which the observations of a particular species were recorded. In total, 54,162,808 records of 559 IAS were analysed. Because many observations refer to the same location at different times, only the first record of a given pair of geographic coordinates was considered, thus leading to 10,491,969 records. In addition, because our study focused on invasion dynamics, we excluded the occurrences of species in their native continents from the first analysis, resulting in 7,051,085 IAS remaining records. To minimize statistical fluctuations and address deficient sampling records, only species with more than 1,000 records from outside their native range were considered. However, in the taxonomic groups in which no IAS reached 1,000 records (molluscs, arachnids and crustaceans), we considered the species with more than 500 records due to the importance of using distinct higher taxonomic groups and species that inhabit different environments to broaden our understanding of IAS global increasing record rates. However, even with this less rigorous threshold, the taxonomic groups Annelida and Gymnospermae were not analysed because they had fewer than 200 records per species. This filtering process led to 3,443,928 records of 178 IAS that were suitable for the modelling according to our criteria (Fig. 1), from which 28 IAS are among the 100 world's worst invaders according to GISD (Table S1). We filtered the coordinates using the shapely Python package. This module enables us to filter any data point within a given polygon. Each coordinate in our dataset was transformed into a shapely point object and tested if it was in or out of a given continent, a shapely polygon. If we found the coordinate within a given continent, considering the information on native range of species, we flagged it as a native occurrence, otherwise as invasive. We emphasize that there is no perfect solution to perform these tasks and potential uncertainties remain, such as inaccuracies and data information gaps from the databases.
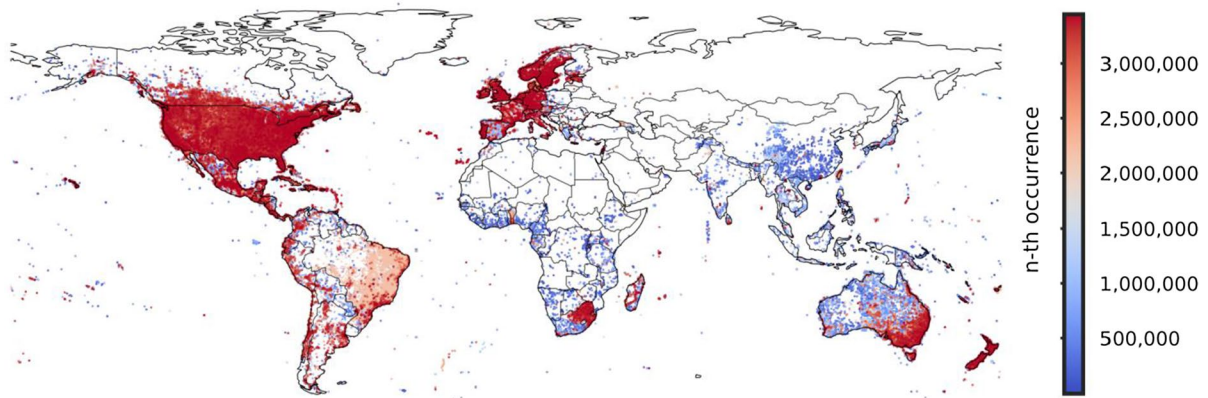
Fig. 1 Global distribution of IAS represented by all records used to construct our record model. Records refer to occurrence information for 178 IAS outside their native range after the year 1900

Despite the tendency to be concentrated in North America and Europe, the records are spread across all continents and latitudes (Fig. 1). Most records were dated after the year 1900 (99.85%, i.e., 3,440,884 records). We also observed that there were years without any occurrences and inconsistencies in the pattern of data of many species. Therefore, to further minimize statistical fluctuations and standardize our analysis, we used the latest 90% of the accumulated annual data available. We emphasize the robustness of our results since we found the same temporal trend even for intervals ranging from 70 to 90% of the available data.

The exclusion of occurrence data from the native continents was performed by delimiting the geographical native range of each species using the information available from the IUCN (International Union for Conservation of Nature) Red List (http://www.iucnredlist.org/technical-documents/categories-and-criteria), which is the most comprehensive and detailed inventory of the global conservation status of animals and plants that also considers the geographical range of species in its assessments (criterion B, specifically). For the species not studied by the IUCN, the native range was obtained from specific databases, such as the Invasive Species Compendium (CABI; www.cabi.org/isc/), Catalog of Fishes from the California Academy of Science (CAS) (http://researcharchive.calacademy.org/research/ichthyology/catalog/fishcatget.asp?genid=9801), Encyclopedia of Life (EOL; http://eol.org/) and Global Invasive Species Database (GISD; http://www.iucngisd.org/gisd/)

(see Table S1). Alien invasive species whose native continents were undefined/unclear were excluded from the analyses.

Notably, within native continents the limits of the native range of a species are not always precisely defined, resulting in uncertainties in the precise native range mapping. The finer the scale at which the distributions are mapped, the smaller the area that a particular taxon is found to occupy since a greater number of areas in which the taxon is unrecorded are revealed. In contrast, fewer unoccupied areas are represented at coarser mapping scales, resulting in range estimates that are more likely to exceed the thresholds (IUCN 2012). Thus, the scale at which ranges are estimated combined with the scarcity of studies in particular regions constitute an important source of uncertainty in terms of the delimitation of the native ranges of IAS. This precludes the precise separation of native and non-native occurrences within continents that encompass native distributions. To address this range uncertainty, and in order to guarantee only the inclusion of true positive non-native records in the analyses of IAS expansion, we excluded the occurrence records falling on continents that encompass native distributions. Thus, our study is strictly focused on the movement of IAS records over time outside their native continents, disregarding translocations events (transfer of species to non-native portions of the native continent).

We highlight that some iconic IAS that are widespread worldwide (e.g., *Pinus pinaster, Leucaena leucocephala*, *Salvinia molesta*, *Caulerpa taxifolia*,

and *Oreochromis niloticus*) were not addressed in our results due to insufficiency in their georeferenced occurrence data in the databases we used. This lack of data may have occurred because these IAS often become so common in certain areas that they do not raise interest in their cataloguing. In addition, many museums and herbariums do not accept deposits of non-native taxa.

Growth model of the IAS records

The rising records of IAS were examined by accumulating their occurrences in chronological order while considering the following three levels: (i) all IAS data (all 178 IAS pooled); (ii) higher taxonomic groups (all IAS belonging to a particular taxonomic group); and (iii) individual species (each particular species). We used a mono-log transformation to observe the linear trends of the number of occurrences as a function of time. With the data transformed, we then performed a simple linear regression to obtain the coefficients. We used the most general definition of the coefficient of determination, R^2, which is based on the division of the residual sum of squares by the total sum of squares. The model we used can represent IAS expansion in both the introduction and/or the spreading stage of the invasion process (e.g., Blackburn et al. 2011) in a coarser scale if the fit assumes exponential behaviour with records rising faster in non-native ranges than in native ones (see details below).

Our IAS record model considers that only a few individuals of a species migrate or are transported by humans from the site they originally invaded (latitude and longitude) to another site after time τ (Fig. 2a). If this movement from a site leads to the invasion of distinct locations, the number of occupied sites doubles after this period. At time 2τ, the duplication process occurs in all invaded locations, and the number of initial sites consequently quadruples (Fig. 2b). In general, the duplication process occurs after each elapsed time τ (Figs. 2a–c). Therefore, after n successive periods of time τ, the number of sites occupied (N) by a particular species equals $N_0 2^n$, where N_0 is the initial number of cells (pair of coordinates) occupied by the individuals of the species. Regarding the total time elapsed, $t = n\tau$, which can be written as

$$N(t) = N_0 2^{t/\tau} \tag{1}$$

or, alternatively

$$N(t) = N_0 e^{\alpha t} \tag{2}$$

where $\alpha = \ln 2/\tau$. We refer to τ as the doubling time of the number of records and α as the intrinsic rate of increase in the number of records (IRR). The doubling time is the number of years required to double the number of occupied sites. As the doubling time increases, the record number slows. The IRR is considered the relative rate at which species occupy new sites outside their native range over time, and potentially, the IAS expansion increases as the IRR increases. Thus, an increase in IRR reduces the doubling time. Notably, $N(t)$ is simply a Malthusian expression for the number of occupied sites but not the number of individuals as usually employed in studies concerning population growth.

The existence of uninhabited sites becomes scarcer over time, consequently, individuals would occupy
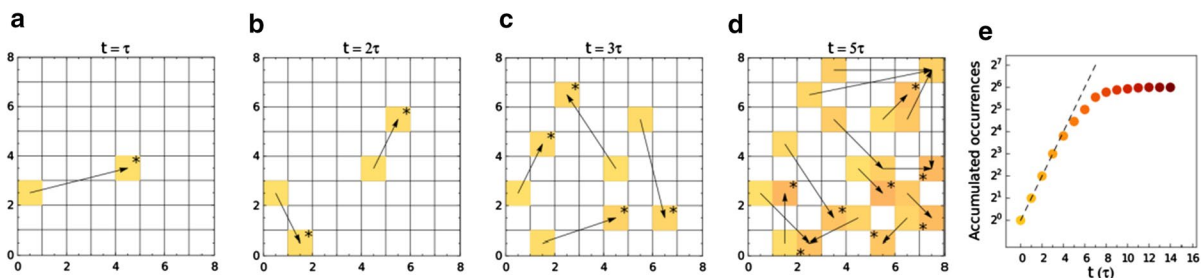

Fig. 2 Record model explanation. Panels 'a', 'b', and 'c' show how the duplicative behaviour holds until a certain number of elapsed doubling times. Panel 'd' illustrates how the choice of new sites starts to saturate. Panel 'e' shows how accumulated occurrences of new occupied sites evolve from a duplicative process to a saturated one, supposing a random migration. The mono-log scale was preferred in 'e' to emphasize the Malthusian growth at the beginning. The arrows point out the movement direction from the original to the new invaded sites, demarcated by asterisks (*)

areas that have already been invaded (Fig. 2d). In consideration of our data filtering process in which only the first record of a given pair of geographic coordinates was retained, saturation in the number of occupied sites may emerge if the number of new records declines over time. In this case, exponential (Malthusian) behaviour no longer represents the pattern of rising records over time and rather resembles a logistic (Verhulst) pattern (Fig. 2e).

Finally, we considered that the number of occurrence records of a species can also increase over time as a result of the sampling effort effect in both non-native and native regions. However, in non-native areas, the records also represent new occurrences of IAS, reflecting punctual events of the introduction and/or spreading processes as aforementioned. Conversely, in native regions, the number of records solely represent the insertion of new occurrence records into online databases (such as GBIF and other platforms), thus purely reflecting a sampling effort effect. Therefore, to compare the rising records of IAS between native and non-native areas, the derivative of the number of occurrence records was calculated for every year on native and non-native ranges. We then used the dynamic time warping (DTW) analysis (Sakoe and Chiba 1978), to measure the similarity between these time series. The DTW finds the optimal nonlinear alignment between two sequences of numerical values, which may have unequal lengths, then their entries are matched through local compression or stretching to minimize the Euclidian distance between them (Giorgino 2009). Thus, by using the DTW it is possible to find patterns between measurements of events with different rhythms (i.e., records in native and non-native ranges). This method has been used to study patterns in animal movement (Pasquaretta et al. 2021), spatiotemporal pollution of rivers (Wang et al. 2019) and bird activity in disturbed landscapes (Ducrettet et al. 2020). Here, higher IRR values in non-native ranges associated with lower DTW values imply that groups are increasing records in non-native ranges faster than expected by the sampling effect, with 68% of the comparisons accepting the null hypothesis that the derivative values come from the same distribution (p value > 0.01, two-sample Kolmogorov–Smirnov) (Corder and Foreman 2009). In general, the lower the DTW values the higher is the increasing record rate in the non-native region, and so, the greater should be

the attention regarding the need for studies evaluating whether the individual species or taxonomic group should be included management planes for the control of invasions. Therefore, we pointed out the three lowest DTW values for each taxonomic group to help defining in which continent each specific taxonomic group needs attention. We performed all analyses using the software Python. All the data and code necessary to retrieve our results, as well as more information regarding how to prepare the data for analyses can be found in the GitHub repository https://github.com/vieirads/alien-species-records.

Results

We found that all higher taxonomic groups and all IAS pooled exhibited a pattern of increase in the number of records consistent with exponential growth outside their native ranges (Fig. 3). There are few signs of saturation in global records among the 178 IAS (Fig. S1). Despite the variation in the rate at which IAS are recorded in new sites over time, the exponential pattern of new records was consistent across all taxonomic groups (Fig. 3). These findings (90% of our fits had an R^2 value greater than 0.927) show that IAS are rapidly been recorded globally with a high IRR. In addition, from the 11 higher taxonomic groups we evaluated, nine had a minimum of 500 records in both the native and non-native ranges, allowing comparisons of the intrinsic rate of increase in the number of records (IRR) in both ranges. Globally, molluscs, crustaceans, insects, amphibians, reptiles, and birds presented a higher IRR in their non-native range than their native ranges, while fishes and mammals presented similar IRRs between the native and non-native ranges (Fig. 3). Only angiosperms presented an IRR higher in the native range than in the non-native range. When all IAS are pooled, we found a slightly higher IRR in their non-native than in their native range with a large increase in invasion records after the late 1990s, which also occurred in fishes, amphibians and mammals.

Despite the great consistency to describe the global rising records of IAS using an exponential function, the IRRs and doubling times of the records number varied among the taxonomic groups (Fig. 4). Among the metazoans, arachnids have the lowest IRR, and mammals and amphibians present
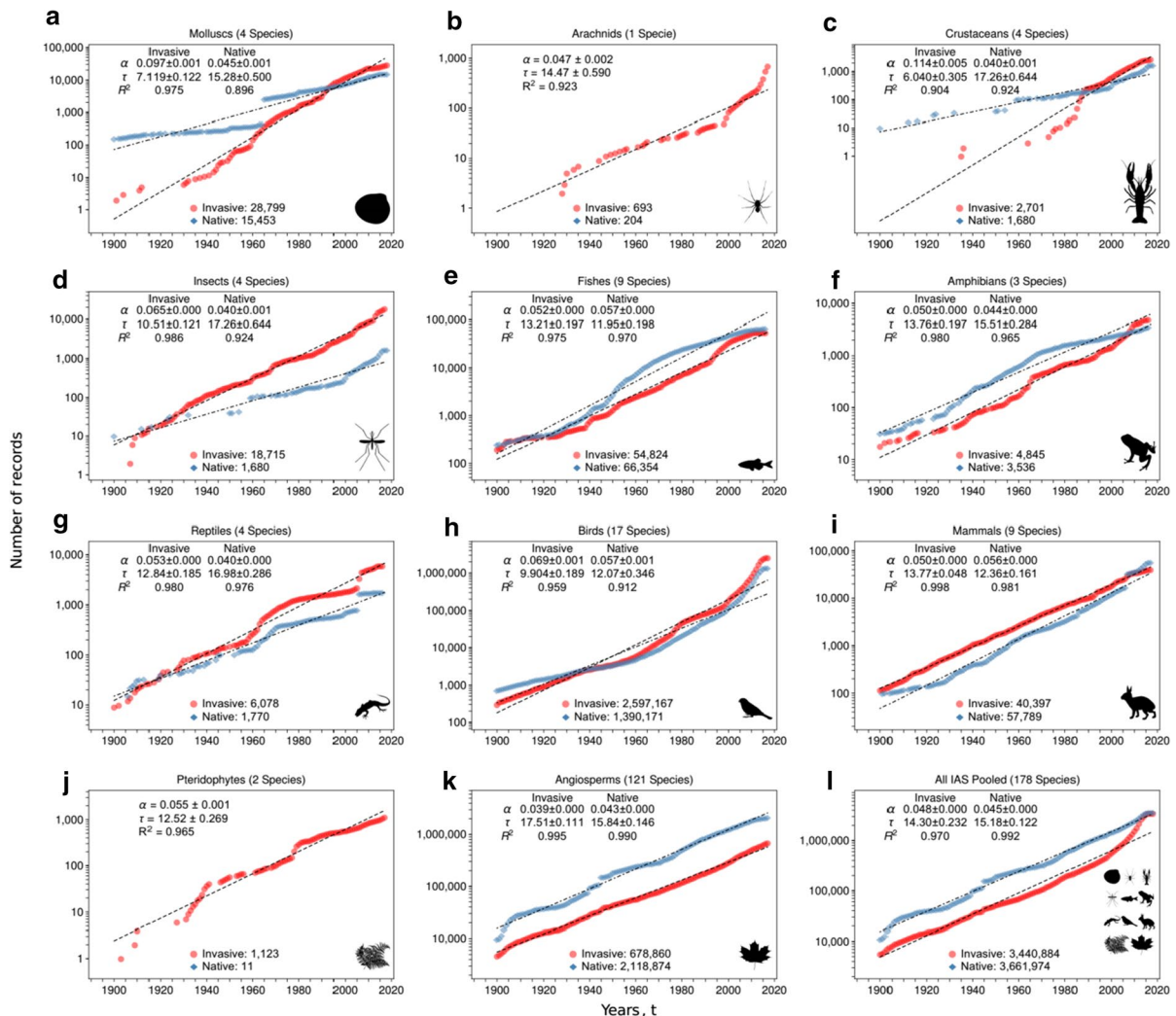
Fig. 3 Relationship between the accumulated occurrences of species and the temporal series on a logarithmic scale. Species are classified into higher taxonomic groups (**a–k**), and all IAS pooled (l). Dashed lines are the exponential model fits; α is the intrinsic record rate (IRR); τ is the doubling time; R^2 is the regression determination coefficient. Non-native range (red) and Native range (blue) symbols are followed by values representing the number of records

similar IRR values. Reptiles and fishes also had similar IRRs, while insects and birds showed minor differences (Fig. 4a). Molluscs and crustaceans had higher IRRs than the other groups, with crustaceans presenting the highest IRR. The doubling time of the number of new records varied from c. six years in crustaceans and molluscs to c. 14 years in the arachnids. Among the metaphytes, we found that angiosperms had the lowest IRRs, while pteridophytes showed the highest values (Figs. 3, 4b and Fig. S1). The doubling time of the number of

metaphyte records varied from c. 12 years in pteridophytes to c. 17 years in angiosperms.

We refined our analysis of the 11 higher taxonomic groups to identify which taxonomic groups should need a closer look to evaluate potential needs for management actions in each continent. The DTW analyses showed that all continents have some taxonomic group that present high IRR values (Table S2 and Fig. S2). North America has more taxonomic groups exhibiting increasing invasion records (10 of 11 groups). Only the rising records of birds seems to
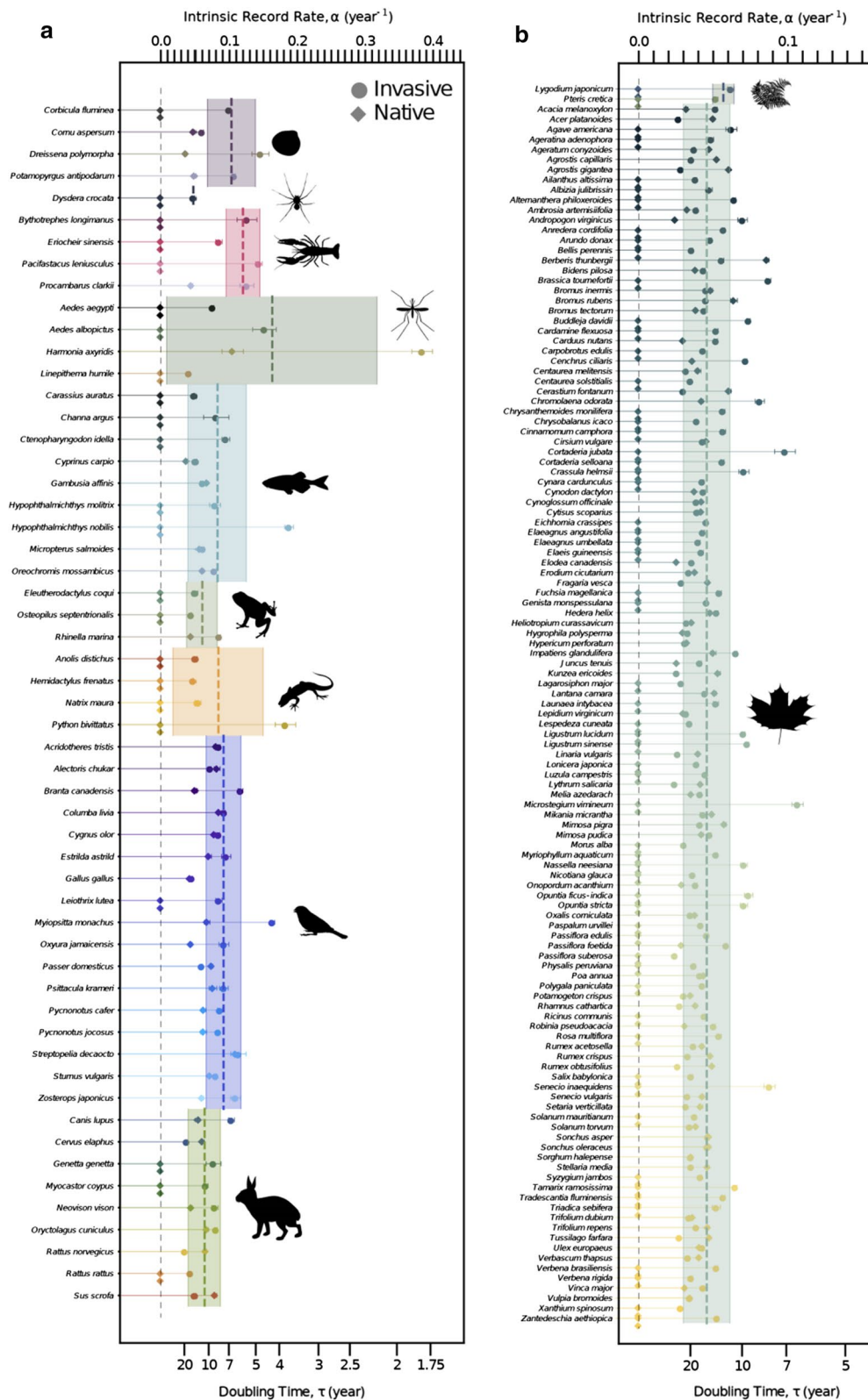
be not a matter of concern in North America when compared with other taxonomic groups and other continents according to DTW analyses (Table S2 and Fig. S2). Europe is the second, with five taxonomic groups that present high IRR values, followed by Asia and South America (three groups), Oceania (two groups) and Africa (one group). Continents also share similar taxonomic groups with high IRR values. Insects, birds, mammals and angiosperms are increasing their records in at least three continents, while molluscs, crustaceans, fishes, amphibians and reptiles are increasing their records in at least two continents. Arachnids and pteridophytes are increasing their records only in North America. Finally, we highlight that the DTW analyses comparing the native and the non-native ranges revealed that crustaceans, insects, fishes, amphibians and birds are particularly rising their records across the globe at faster rates than the expected by sampling effort effect (Fig. 4, Table S2 and Fig. S2).

Discussion

Our findings indicate an exponential rising of IAS records globally with high IRRs in the non-native than in the native range for most taxonomic groups we analysed, suggesting that these IAS can be having an exponential global expansion. We propose that three non-mutually exclusive factors are linked to the rapid and continuous unlimited rising of IAS records at the global scale. First, the number of IAS records can rise with the increased IAS introductions caused by international trade over time, leading to multiple introductions and, in turn, helping IAS succeed (Trakhtenbrot et al. 2005). The spread of IAS may also increase in continents where socio-political changes facilitate people immigration (Roques et al. 2016). This higher human movement, along with the longer history of introductions, and hence higher sampling intensity, could explain why North America and Europe have a higher number of IAS records than Africa and South America. Second, the increased rising record rates may be associated with disturbance regimes caused by habitat degradation because IAS are more successful in disturbed habitats (Havel et al. 2005). This second factor could also indicate that in Africa and South America, human activities may have caused less degradation in the natural environment than in North America and Europe. Third, IAS thrive in habitats with a surplus of resources (Parepa et al. 2013), which has been recognized to be a global threat (e.g., nitrogen in terrestrial ecosystems—Sala et al. 2000; and eutrophication in aquatic ecosystems—Diaz and Rosenberg 2008). Although the second and third factors are related to the introduction and establishment phases of the invasion process, their interaction certainly increases the expansion of IAS in large areas by increasing the number of new populations. In addition, habitat disturbance and increased nutrient availability are special concerns related to the presence of IAS in future environmental scenarios because these two impacts are expected to increase worldwide as a result of human-induced climate change (Sala et al. 2000). Finally, all these mechanisms have been accelerated by the increased propagule pressure caused by anthropogenic transport after the IAS have been established (Donaldson et al. 2014; O'Malia et al. 2018).

The variation found in the IRRs and record doubling times among the taxonomic groups was expected since the potential expansion of IAS is related to human interference, along with different mechanisms of dispersal (Arim et al. 2006) and introduction (Foxcroft et al. 2004) among species. For example, molluscs and crustaceans likely expand at the fastest rates because their larvae are easily transported downstream by water movement, across landscapes and continents in vessels, by aquatic birds and by their fish hosts (Dillon 2000; Galil et al. 2011). However, in particular, large invasive spiders, such as *Dysderacrocata,* use human-mediated transfer (Nedvěd et al. 2011), which may help increase the IRR of these species. Similar to angiosperms and vertebrates (except for birds), the successful expansion of pteridophytes worldwide is likely due to their high dispersal ability (linked to ferns producing numerous small spores that are easily transported over long

distances and remain viable for several years; Tryon 1970; Smith 1972; Wolf et al. 2001), while plants, in general, expand in great part because of horticulture, aquaculture (Reichard and White 2001) and aquarium trade (Peres et al. 2018).

Despite the evidence that IAS are rising their records globally, it is necessary to compare their record rates between their non-native and native ranges to ensure that higher rates are not simply the results of the sampling effort effect. Higher IRR values in native than in non-native ranges or similar IRR values between them can be interpreted as a sampling effort effect, while higher IRR values in non-native ranges indicate sampling effort effect together with new introductions (i.e., introduction stage— sensu Blackburn et al. 2011) and/or IAS population spread (i.e., spreading stage—sensu Blackburn et al. 2011). Thus, a steeper increase of records in the non-native range compared to the native range may indicate the expansion of IAS populations in the former range. Although our data indicate that the records of all higher taxonomic groups analysed and the IAS pooled are rising exponentially, crustaceans, insects, fishes, amphibians, and birds are the concerning groups because they are consistently rising records faster in their non-native ranges than in their native ranges (see Table S2). In this sense, each region of the globe has taxonomic groups with rising records, which call attention to studies that evaluate the need for monitoring, controlling or management actions (see Figure S2) because these records can indicate potential spatial expansion of IAS. For example, arachnids and pteridophytes are rising records only in North America, while birds should receive more attention in South America, Africa and Europe. Also, recent findings are showing increases in records of alien tetrapods in the last two decades in the Iberian Peninsula (Ascensão et al. 2021). In this sense, nations should consider concentrating more efforts on the development of studies assessing the need to implement strategies to containment and management of specific groups of IAS and thus reduce the introduction and spread of new IAS, as is the case in North America and Europe. This perspective highlights the potential of an alternative view in creating international conservation plans between nations that share IAS groups with rising records and considered them as a priority in the investigation regarding the need for continental containment and management

and agreements with common targets between continents. It has been posed that more effective conservation solutions should be achieved by coordinated planning at the continental, rather than local level (Kark et al. 2009; Moilanen et al. 2013; Robertson et al. 2015; Aizen et al. 2019; Pyšek et al. 2020). Such initiatives are particularly challenging because continents are composed of countries with a myriad of political, social, economic, ecological, and cultural systems (Orlikowska et al. 2016), which present different postures in dealing with environmental and conservation issues. Despite these difficulties, studies have argued the integrated approaches through international cooperation are pivotal to avoid and block transnational species invasions with the potential to negatively affect biodiversity, ecosystem services, and human livelihoods (Aizen et al. 2019; Pyšek et al. 2020).

In conclusion, we show that despite the myriad of mechanisms that may hamper the expansion of species (Theoharides and Dukes 2007), IAS overcome these limitations and are increasing their records across the Earth with few signs of saturation. More importantly, our findings show that the number of invaded sites may double within 14 years due to exponential growth considering the analysis of all IAS. From a theoretical perspective, we found a general pattern of IAS rising record at the global scale that can be described by an exponential function, which has a high degree of explanation and can be applied to a wide range of taxonomic groups and individual species, but when the pattern is deconstructed by continents, some Verhulst signs appear, i.e., few signs of suturing behaviour. This consistent pattern was likely found because we analysed records at the large spatial scale, where emergent ecological properties appear out of contingencies (Lawton 1999). We emphasize that we tried minimizing the uneven temporal and spatial sampling effort in the global dataset by using our methods. However, we cannot rule out the possibility that uneven sampling at least partly drives the observed trends due to varying sampling intensities, changing in time, space and among taxonomic groups. From a practical perspective, the pattern we found indicates the need for more efforts on all fronts to control the advance of or to eradicate IAS locally and, that the prevention of new invasions worldwide (i.e., the ninth goal of the Aichi Biodiversity Targets) can still be not sufficient. It is unlikely that nations

become completely resistant against IAS, but it is necessary to introduce comprehensive border-control and IAS population control policies (Early et al. 2016). Transnational coordination could be a good initiative to homogenise preventative measures and to contain invasions across borders (Hulme 2021), and sometimes containment and management of invasions require highly context-specific interventions that can actually be hindered by rigid international and transnational agreements (Oficialdegui et al. 2020; García-Díaz et al. 2021). Therefore, increase the knowledge of the factors controlling expansion rates within large regions and actions to prevent expansion at the continental scale carried out by invasion ecologists and managers are urgently needed to subsidise policy decisions. We argue that governments should increase the efforts to overcome political barriers to implement policies based on scientific knowledge to help identify better strategies to attain the ninth Aichi target on a global scale.

Declarations

Conflict of interest Authors declare no conflicts of interest/competing interests.

Data availability All data is available in the GitHub repository under the link https://github.com/vieirads/alien-species-records.

Code availability All codes are available in the GitHub repository under the link https://github.com/vieirads/alien-species-records.

References

Aizen MA, Smith-Ramírez C, Morales CL et al (2019) Coordinated species importation policies are needed to reduce serious invasions globally: the case of alien bumblebees in South America. J Appl Ecol 56:100–106. https://doi.org/10.1111/1365-2664.13121

Arim M, Abades SR, Neill PE, Lima M, Marquet PA (2006) Spread dynamics of invasive species. Proc Natl Acad Sci USA 103:374–378. https://doi.org/10.1073/pnas.0504272102

Ascensão F, D'Amico M, Martins RC et al (2021) Distribution of alien tetrapods in the Iberian Peninsula. NeoBiota 64:1–21. https://doi.org/10.3897/neobiota.64.55597

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

Blackburn TM, Essl F, Evans T et al (2014) A unified classification of alien species based on the magnitude of their environmental impacts. PLoS Biol 12(5):e1001850. https://doi.org/10.1371/journal.pbio.1001850

CBD (Convention on Biological Diversity) (2011) Strategic Plan for Biodiversity 2011–2020, Including Aichi Biodiversity Targets. https://www.cbd.int/sp/

Corder GW, Foreman D (2009) Nonparametric statistics for non–statisticians: a step–by–step approach. Wiley, New Jersey

Diaz RJ, Rosenberg R (2008) Spreading dead zones and consequences for marine ecosystems. Science 321:926–929. https://doi.org/10.1126/science.1156401

Dillon RT Jr (2000) The Ecology of Freshwater Molluscs. Cambridge University Press, New York

Donaldson JE, Hui C, Richardson CM, Robertson MP, Webber BL, Wilson JRU (2014) Invasion trajectory of alien trees: the role of introduction pathway and planting history. Glob Change Biol 20:1527–1537. https://doi.org/10.1111/gcb.12486

Ducrettet M, Forget P-M, Ulloa JS et al (2020) Monitoring canopy bird activity in disturbed landscapes with automatic recorders: a case study in the tropics. Biol Conserv 245:108574. https://doi.org/10.1016/j.biocon.2020.108574

Early R, Bradley BA, Dukes JS et al (2016) Global threats from invasive alien species in the twenty-first century and national response capacities. Nat Commun 7:12485. https://doi.org/10.1038/ncomms12485

Elton CS (1958) The Ecology of Invasions by Animals and Plants. Springer, Boston. https://doi.org/10.1007/978-1-4899-7214-9

Foxcroft L, Rouget M, Richardson DM, Fadyen SM (2004) Reconstructing 50 years of *Opuntia stricta* invasion in the Kruger National Park, South Africa: environmental determinants and propagule pressure. Divers Distrib 10:427–437. https://doi.org/10.1111/j.1366-9516.2004.00117.x

Galil BS, Clark PF, Carlton JT (2011) In the Wrong Place—Alien Marine Crustaceans: Distribution, Biology and Impacts. Invading Nature—Springer Series in Invasion Ecology, Netherlands. https://doi.org/10.1007/978-94-007-0591-3

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

Garnas JR, Auger-Rozenberg M-A, Roques A, Bertelsmeier C, Wingfield MJ, Saccaggi DL, Roy HE, Slippers B (2016) Complex patterns of global spread in invasive

insects: eco-evolutionary and management consequences. Biol Invasions 18:935–952. https://doi.org/10.1007/s10530-016-1082-9

Giorgino T (2009) Computing and visualizing dynamic time warping alignments in R the dtw package. J Stat Softw 31:1–24. https://doi.org/10.18637/jss.v031.i07

Havel JE, Lee CE, Zanden JMV (2005) Do reservoirs facilitate invasions into landscapes? Bioscience 55:518–525. https://doi.org/10.1641/0006-3568(2005)055[0518:DRFIIL]2.0.CO;2

Hulme PE (2021) Unwelcome exchange: international trade as a direct and indirect driver of biological invasions worldwide. One Earth 4(5):666–679. https://doi.org/10.1016/j.oneear.2021.04.015

International Union for Conservation of Nature—IUCN (2012) Red List Categories and Criteria: Version 3.1. Second edition. Gland, Switzerland and Cambridge, UK

Kark S, Levin N, Grantham HS, Possingham HP (2009) Between-country collaboration and consideration of costs increase conservation planning efficiency in the Mediterranean Basin. Proc Natl Acad Sci USA 106:15368–15373. https://doi.org/10.1073/pnas.0901001106

Kelly R, Lundy MG, Mineur F, Harrod C, Maggs CA, Humphries NE, Sims DW, Reid N (2014) Historical data reveal power-law dispersal patterns of invasive aquatic species. Ecography 37:581–590. https://doi.org/10.1111/j.1600-0587.2013.00296.x

Kennedy TA, Naeem S, Howe KM, Knops JMH, Tilman D, Reich P (2002) Biodiversity as a barrier to ecological invasion. Nature 417:636–638. https://doi.org/10.1038/nature00776

Kumschick S, Gaertner M, Vilà M et al (2015) Ecological impacts of alien species: quantification, scope, caveats, and recommendations. Bioscience 65:55–63. https://doi.org/10.1093/biosci/biu193

Lawton JH (1999) Are there general laws in Ecology? Oikos 84:177–192. https://doi.org/10.2307/3546712

Levine JM, D'Antonio CM (1999) Elton revisited: a review of evidence linking diversity and invasibility. Oikos 87:15–26. https://doi.org/10.2307/3546992

Moilanen A, Anderson BJ, Arponen A, Pouzols FM, Thomas CD (2013) Edge artefacts and lost performance in national versus continental conservation priority areas. Divers Distrib 19:171–183. https://doi.org/10.1111/ddi.12000

Nedvěd O, Pekár S, Bezděčka P, Líznarová E, Řezáč M, Schmitt M, Sentenská L (2011) Ecology of Arachnida alien to Europe. Biocontrol 56:539–550. https://doi.org/10.1007/s10526-011-9385-3

O'Malia EM, Johnson LB, Hoffman JC (2018) Pathways and places associated with nonindigenous aquatic species introductions in the Laurentian Great Lakes. Hydrobiologia 817:23–40. https://doi.org/10.1007/s10750-018-3551-x

Oficialdegui FJ, Delibes-Mateos M, Green AJ, Sánchez MI, Boyero L, Clavero M (2020) Rigid laws and invasive species management. Conserv Biol 34(4):1047–1050. https://doi.org/10.1111/cobi.13481

Orlikowska EH, Roberge JM, Blicharska M, Mikusiński G (2016) Gaps in ecological research on the world's largest internationally coordinated network of protected areas: a review of Natura 2000. Biol Conserv 200:216–227. https://doi.org/10.1016/j.biocon.2016.06.015

Parepa M, Fischer M, Bossdorf O (2013) Environmental variability promotes plant invasion. Nat Commun 4:1604. https://doi.org/10.1038/ncomms2632

Pasquaretta C, Dubois T, Gomez-Moracho T, Delepoulle VP, Le Loc'h G, Heeb P, Lihoreau M (2021) Analysis of temporal patterns in animal movement networks. Methods Ecol Evol 12:101–113. https://doi.org/10.1111/2041-210X.13364

Peres CK, Lambrecht RW, Tavares DA, Castro WAC (2018) Alien express: the threat of aquarium e-commerce introducing invasive aquatic plants in Brazil. Perspect Ecol Conserv 16:221–227. https://doi.org/10.1016/j.pecon.2018.10.001

Pyšek P, Hulme PE, Simberloff D et al (2020) Scientists' warning on invasive alien species. Biol Rev 95:1511–1534. https://doi.org/10.1111/brv.12627

Rahel FJ (2002) Homogenization of freshwater faunas. Annu Rev Ecol Evol Syst 33:291–315. https://doi.org/10.1146/annurev.ecolsys.33.010802.150429

Reichard SH, White P (2001) Horticulture as a Pathway of Invasive Plant Introductions in the United States: most invasive plants have been introduced for horticultural use by nurseries, botanical gardens, and individuals. Bioscience 51:103–113. https://doi.org/10.1641/0006-3568(2001)051[0103:HAAPOI]2.0.CO;2

Robertson PA, Adriaens T, Caizergues A et al (2015) Towards the European eradication of the North American ruddy duck. Biol Invasions 17:9–12. https://doi.org/10.1007/s10530-014-0704-3

Roques A, Auger-Rozenberg M-A, Blackburn TM et al (2016) Temporal and interspecific variation in rates of spread for insect species invading Europe during the last 200 years. Biol Invasions 18:907–920. https://doi.org/10.1007/s10530-016-1080-y

Ruggiero MA, Gordon DP, Orrell TM et al (2015) A higher level classification of all living organisms. PLoS One 10(6):e0130114. https://doi.org/10.1371/journal.pone.0119248

Sakoe H, Chiba S (1978) Dynamic programming algorithm optimization for spoken word recognition. IEEE Trans Acoust Speech Signal Process 26:43–49. https://doi.org/10.1109/TASSP.1978.1163055

Sala OE, Chapin FS III, Armesto JJ et al (2000) Global biodiversity scenarios for the year 2100. Science 287:1770–1774. https://doi.org/10.1126/science.287.5459.1770

Seebens H, Blackburn TM, Dyer EE et al (2017) No saturation in the accumulation of alien species worldwide. Nat Commun 8:14435. https://doi.org/10.1038/ncomms14435

Smith AR (1972) Comparison of fern and flowering plant distributions with some evolutionary interpretations for ferns. Biotropica 4:4–9. https://doi.org/10.2307/2989639

Theoharides KA, Dukes JS (2007) Plant invasion across space and time: factors affecting nonindigenous species success during four stages of invasion. New Phytol 176:256–273. https://doi.org/10.1111/j.1469-8137.2007.02207.x

Trakhtenbrot A, Nathan R, Perry G, Richardson DM (2005) The importance of long-distance dispersal in biodiversity

conservation. Divers Distrib 11:173–181. https://doi.org/10.1111/j.1366-9516.2005.00156.x

Tryon R (1970) Development and evolution of fern floras of oceanic islands. Biotropica 2:76–84. https://doi.org/10.2307/2989765

van Kleunen M, Dawson W, Essl F et al (2015) Global exchange and accumulation of non-native plants. Nature 525:100–106. https://doi.org/10.1038/nature14910

Wang K, Wang P, Zhang R, Lin Z (2019) Determination of spatiotemporal characteristics of agricultural non-point source pollution of river basins using the dynamic time warping distance. J Hydrol 283:124303. https://doi.org/10.1016/j.jhydrol.2019.124303

Wolf PG, Schneide H, Ranker TA (2001) Geographic distributions of homosporous ferns: does dispersal obscure evidence of vicariance? J Biogeogr 28:263–270. https://doi.org/10.1046/j.1365-2699.2001.00531.x

Zenni RD, Simberloff D (2013) Number of source populations as a potential driver of pine invasions in Brazil. Biol Invasions 15:1623–1639. https://doi.org/10.1007/s10530-012-0397-4