



Review Paper

Intentional introduction pathways of alien birds and mammals in Latin America

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ABSTRACT

Assessing the pathways by which alien species are introduced is essential if we are to identify potential risks and evaluate management decisions. Intentional introductions are responsible for the introductions of millions of animals throughout Latin America. We explore: (1) the relative role of several intentional introduction pathways (hunting, feeding, fur, biological control, the pet trade and others) in the general context of introductions of alien species; (2) the relative importance of the intentional pathways across the different taxa; (3) similar patterns as regards the composition of alien species across countries; (4) the underlying factors that drive the richness of alien species in Latin America; and (5) the potential impacts of alien species on the region. According to our results, 69 species of mammals and 62 species of birds were introduced into Latin America by means of intentional pathways, of which the most important taxa were Artiodactyls, Primates, Passeriformes and Psittaciformes. The main introductions pathways were the pet/ornamental trades (70.9%) for birds, and hunting (39.1%) and pet trade/ornamental purposes (37.7%) for mammals. The composition of species differed among countries, with a higher richness of species in those countries with a high percentage of urban populations, with a higher native species biodiversity, with a high % of GDP owing to imports (birds) and in those with a high number of trafficked species (mammals). This review stresses that the pet trade and hunting are important pathways for the introduction of alien species, some of which have had severe impacts on many countries.

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1. Introduction

Biological invasion is one of the main drivers of global environmental change and the loss of native biodiversity (Bellard et al., 2016; Blackburn et al., 2019). In addition, anthropogenic transcontinental movements involve a continually increasing global traffic and the subsequent intentional and unintentional transfer of organisms, and a diverse array of human-mediated

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pathways have appeared to transport numerous species between different eco-regions (Kuhlenkamp and Kind, 2018). Global databases (e.g. GISD, GAVIA, or GRIIS) provide an overview of global geographic patterns of species invasions and/or introductions, origins and introduction pathways (Turbelin et al., 2017).

Mammals and birds were the first organisms to be intentionally introduced around the world for sport hunting, commercialization and domestication as livestock (e.g. feeding, fur, transport or work animals), pets or for pest control (Long, 2003; Lever, 2005; Clout and Russell, 2008; Blackburn et al., 2009). Some other species have been accidentally introduced (e.g. hitchhiker species, such as rats in ships). However, most established mammals originate from intentional releases into natural environments (Blackburn et al., 2011). The establishment of alien species has, therefore, primarily occurred near human populations, after which they have spread into natural and semi-natural areas (Da Rosa et al., 2017; Carpio et al., 2017a).

The impact and risk of biological invasions is particularly high in Latin America owing to its invaluable world biodiversity hotspots (Myers et al., 2000). However, the introduction of alien species has been poorly studied in this region (Speziale et al., 2012) compared to scientific production in the United States, Europe, or Australia. According to the Global Invasive Species Database (GISD), 283 of the 613 invasive species (46%) that are listed in the GISD are present in some Latin America and Caribbean countries (Pauchard et al., 2011), and 20% of the known world-wide mammal introduction events have occurred in Latin America (Novillo and Ojeda, 2008). In addition, at least 41 of the 100 of the world's worst invasive alien species are already established in Latin America (IUCN-ISSG, 2000). However, the role of intentional pathways as a source of alien species in Latin America has received relatively little attention (Speziale et al., 2012; Essl et al., 2015; but see; Buenavista and Palomares, 2018) and the information available is biased towards some specific regions, such as Chile (Jaksic, 1998; Jaksic et al., 2002; Iriarte et al., 2005), the Galapagos Islands (Toral-Granada et al., 2017), Argentina (Lizarralde et al., 2004; Bonino and Soriguer, 2009) or Brazil (Da Rosa et al., 2017). Information about alien species is, on the contrary, often scarce, inaccurate or reported only in grey literature (Pereira-Garbero et al., 2013).

Goods transported as a result of the intensive international trade have been identified as an important factor affecting the occurrence of invasive species (Hulme, 2009). The magnitude of merchandise imports, specially commodities is a significant determinant of the number of species (<https://besjournals.onlinelibrary.wiley.com/doi/10.1111/j.1365-2664.2008.01600.x> Westphal et al., 2008) as well as the rate of new species introductions (<https://besjournals.onlinelibrary.wiley.com/doi/10.1111/j.1365-2664.2008.01600.x> Levine and D'Antonio, 2003) of a wide range of alien taxa. Moreover, a further increase in the number of alien species can be expected when associated with the increase in gross domestic product (GDP) and trade (Pyšek et al., 2008; Hulme, 2009; Seebens et al., 2017), since GDP reflects levels of infrastructure (roads, canals, railways etc.) that can also facilitate invasions (Hulme, 2009). Cardador et al. (2019) demonstrated the magnitude of a trade ban as regards preventing biological invasions. Another important factor is the proportion of urban population (McKinney, 2008), since areas with dense human populations are relevant sources of alien species owing to: (1) the release and escape of alien species kept as pets (Spear et al., 2013), (2) the intensity of tourism, which is directly associated with both intentional and unintentional introduction pathways (Anderson et al., 2015) and (3) the intensity of anthropogenic disturbance, which can be considered as a proxy for the risk of invasion (Hulme, 2009; Spear et al., 2013; Carpio et al., 2017a). Moreover, some studies indicate that areas with a higher biodiversity tend to host a higher richness of alien species on large spatial scales (e.g. Stohlgren et al., 2003, 2006; Fridley et al., 2007; Dyer et al., 2017a), since those areas with a high native diversity have a greater number of microniches (or more spatial heterogeneity), which would, therefore, allow a greater number of alien species to be accommodated (Davies et al., 2005). We accordingly predict that Latin America will possibly achieve the highest richness globally of alien species because of its great variety of niches and biodiversity, along with an increase in trade, GDP and other correlated activities (Speziale and Lambertucci, 2010).

In this paper, we aim to identify cases of alien species of birds and mammals that were introduced into Latin American countries through intentional pathways. Our specific objectives are: (1) to review the relative role of intentional introduction pathways (hunting, feeding, fur, biological control, the pet trade and others) in the context of alien species introductions; (2) to assess the relative importance of the intentional pathways across the different taxa; (3) to identify similar patterns in the composition of alien species across countries, (4) to assess the underlying factors that drive the richness of alien species in Latin America, and (5) to quantify the potential impacts of these species on this region.

2. Material and methods

2.1. Data collection

We reviewed several lists of species introduced into South America, Mesoamerica and Caribbean Islands, which were obtained from scientific papers, books and technical reports (Kairo et al., 2003; Long, 2003; Lever, 2005; Flueck, 2010; Pimentel, 2014; Da Rosa et al., 2017; Dyer et al., 2017b; Buenavista and Palomares, 2018). We used the following search terms: "non-native species" OR "invasive species" OR "non-indigenous species" OR "alien species" OR "exotic species" OR "introduced species" AND "intentional purposes" OR "intentional introduction pathways" OR "hunting" OR "pet" OR "feeding" [or "feed" or "food"] OR "biological control" OR "fur" [or "fur trapping"] OR "acclimatization society" [or "rewilding" or "ecological replacement"] AND "mammal" OR "bird" AND "Latin America" OR "Central America" OR "South America" OR "Mesoamerica" OR "Caribbean Islands". We also used databases, such as the Global Invasive Species Database ("GISD" <http://www.iucngisd.org/gisd/>), and those of the International Union for Conservation of Nature ("IUCN" <https://www.iucnredlist.org/>), The

Convention on International Trade in Endangered Species of Wild Fauna and Flora ("CITES" <https://www.cites.org/>) the Invasive Species Compendium (<https://www.cabi.org/isc/>), Inter-American Biodiversity Information Network (<http://bd.institutohorus.org.br/www/>), Global Avian Invasions Atlas (GAVIA <https://doi.org/10.6084/m9.figshare.4234850>) and the Global Register of Introduced and Invasive Species (GRIIS <http://www.griis.org/>). The sources of information used for this study are shown in List S1a. Scientific papers addressing the introduction of species for intentional purposes were searched using three main web engines: Google Scholar™, ISI Web of Science® and Scopus®. All of this information was then combined to draw up a list of the species intentionally introduced into each country (Fig. S3). Finally, only those species with a documented introduction pathway in any Latin-American country were included in Table S3. The species that are native to one part of Latin America but alien to another were included. Those species for which references concerned another introduction purpose (natural dispersal, accidental, stowaway, etc.) or when no reliable confirming references were found (databases without a reference to confirm it), were, however, rejected. Notwithstanding, several species required more detailed investigation to be considered as intentionally introduced species. Rodents (such as *Rattus rattus*, *Rattus norvegicus*, *Peromyscus fraterculus* and *Mus musculus*) have been introduced into many areas around the world as a result of their having following humans (Atkinson, 1985; Pimentel, 2014) and were therefore not considered as intentional introduction. However, some species, such as *Rattus exulans*, *Cavia porcellus* and *Dasyprocta punctata*, have been introduced as a source of food (Fiedler, 1990) and therefore considered in this study. Another case is that of *Bubulcus ibis*, whose introduction may have occurred as the result of natural range extensions without human assistance (Kairo et al., 2003) and was not included in the study, or *Didelphus marsupialis*, which was introduced into the Lesser Antilles in Trinidad, although the species could be native (Long, 2003), or into Saint Lucia, where it is classified as native by some authors and alien by others (therefore we have considered it as alien), who believe that it was introduced by Amerindians (Caribbean Conservation Association, 1991). Therefore, species that were accidentally introduced have not been included in this analysis (e.g. *Mus musculus*, *Bubulcus ibis* or some species of the *Rattus* genus).

The biogeographic region from which each species was derived (native range according to IUCN and GISD) was also identified (Palearctic, Nearctic, Afrotropics, Indomalaya, Neotropics, Australasia and Oceania). Those widespread species that occupy several regions were classified as either Holarctic, New World or multiregional (Abellán et al., 2016; see Fig. S1). Finally, we evaluated in which country or countries each species occurs in Latin America, according to GISD, GAVIA, GRIIS and CABI (Fig. S3).

2.2. Intentional introduction pathways of alien species in Latin America

The term 'introduction pathways' describes the processes that result in the introduction of alien species from one geographical location to another (Richardson et al., 2011). A species may have different introduction pathways in different countries. The analysis was carried out for a species only when the main reason for its introduction was "intentional" mainly release and escape (Hulme et al., 2008): (hunting, feeding, pet/aesthetic, biological control, fur and others; Kraus, 2003) in at least one Latin America country. The 'others' group includes aspects such as scientific research, working animals or military activities (CBD, 2014). The introduction pathway relevant to each intentionally introduced species is documented in Table S3.

2.3. Relative importance of different taxa in the global introductions

In order to show the distribution of the different taxonomic groups (order or clade in the case of ungulates), the species were grouped as birds and mammals separately, along with their taxonomic group (Ungulates, Rodentia, Primates...). The number of aliens species intentionally introduced were classified according to the introduction pathways, although some species have multiple pathways of introduction and therefore were classified in more than one pathway (Table S3). In addition, we assessed taxonomic biases in the introduction of alien species at order or clade level by comparing the number of species per taxonomic group introduced intentionally with a random expectation generated using the hypergeometric distribution (Van Wilgen et al., 2010) in R v. 3.4.0 (R Core Team, 2018). The hypergeometric distribution is similar to a binomial distribution and describes the probability of a given number of successes given a specified number of draws, without replacement. In this instance, a set number of species are sampled from a pool of orders and clades of known size (species available per clade worldwide). Taxonomic groups outside the 95% confidence intervals were deemed to be either over- or under-represented in the introduction process, compared to expectations based on the size of the order or clade and the total number of species that were introduced intentionally (Van Wilgen et al., 2018).

2.4. Composition of alien species introduced intentionally throughout the countries

Similarities in the composition of alien species introduced intentionally throughout the countries studied were explored by using clustering analyses. A visualisation of the five closest Euclidean distances to each country is shown by means of a network plot. The elements were clustered by employing hierarchical clustering, using the Ward method and Euclidean distances (Fig. S3). The Euclidean distance between two countries is based on a multidimensional imaginary space in which each coordinate is the presence/absence of an alien species (Fig. S4). The distance between two elements (two countries in our case) is calculated using function 1, where n is equal to the number of variables (in our case, alien species), and p and q

take two values according to the presence/absence of each species (1,0) on each one of each couple of countries (p and q) (Carpio et al., 2019; Oteros et al., 2019).

$$\begin{aligned} d(p, q) = d(q, p) &= \sqrt{(q_1 - p_1)^2 + (q_2 - p_2)^2 + \dots + (q_n - p_n)^2} \\ &= \sqrt{\sum_{i=1}^n (q_i - p_i)^2} \end{aligned} \quad (1)$$

The analyses were computed by using R statistical software (R Core Team, 2018). The most relevant packages where from the family tidyverse (Wickham, 2016, 2017), circlize (Gu et al., 2014) and cluster (Maechler et al., 2018).

2.5. Factors that drive intentionally introduced species richness in Latin America

The underlying factors driving the richness of alien species introduced intentionally per country were determined by performing two Generalised Linear Models (GzLM) using the total number of alien bird species introduced into each country (Model 1) and the total number of mammal species introduced into each country (Model 2) as response variables. Country size, percentage of rural population (refers to the population in areas that have a lower population density than urban areas and are spread over a larger area out than urban centres), No. of pieces confiscated/million inhabitants (quantity of specimens reported as imports by the importing country), No. of trafficked species/million inhabitants, % of GDP imports (% of GDP due to imports) and the gross domestic product (GPD) per capita were included as explanatory variables in the models (the data sources are shown in List S2). In addition, a third GzLM was performed with the number of species established in each country (Model 3) as response variable. In this model, number of species of native birds, species of native mammals and total number of intentional introduced alien species were included as explanatory variables. All models were fitted with a gamma distribution and with a log link. The most plausible models were selected by comparing Akaike's information criterion (AIC) in the models (Burnham and Anderson, 2002), following a backward procedure (Zuur et al., 2009). In particular, we compared the Akaike information criteria for small sample sizes (AICc value) in each candidate model and the best model (that with the lowest AICc). Statistical analyses were performed using InfoStat software (Balzarini et al., 2008).

2.6. Impacts of the alien species introduced intentionally

Many introduced species commonly have a high reproductive rate, which is one of the reasons why they are commercially exploited for sport hunting, feeding, or fur resources (Stokes et al., 2006). The impacts of each species were obtained by using the scientific species name and impacts as a search terms in databases such as Google Scholar, ISI Web of Knowledge and databases above mentioned, manually filtering through the sources identified by reading titles and (if applicable) abstracts (List S1c). The impacts were grouped into three categories: environmental, economic, or health impacts (according to Vilà et al., 2010; Keller et al., 2011; Kumschick et al., 2015). Environmental impacts include hybridization, a reduction in native biodiversity, the modification of hydrology/water regulations, purification and quality/soil moisture, the modification of nutrient pools and fluxes, habitat degradation and the modification of successional patterns. Health impacts include: disease transmission and parasitism, while economic impacts include: damage to agriculture/forestry, a reduction in/damage to livestock and products, human nuisance, damage to aquaculture/mariculture/fish, damage to infrastructures, alteration of recreational use and tourism, and other economic impacts (Mack et al., 2000). Therefore, the number of species that has each type of impact were determined, although some species may have more than one impact (Table S4). In addition, we used the Socio-Economic Impact Classification of Alien Taxa (SEICAT) and Environmental Impact Classification of Alien Taxa (EICAT) to determine the magnitude of socio-economic and environmental impacts respectively (Table S3), and to classified the species in six categories (massive: MV, major: MR, moderate: MO, minor: MN, minimal concern: MC and data deficient: DD) according to Bacher et al. (2018) and Blackburn et al. (2014).

3. Results

3.1. General distribution patterns of intentionally introduced species

According to our results, 69 species of mammals and 62 species of birds were intentionally introduced into some Latin American countries. Of the 131 aliens species, 11 (ten mammals and one bird) are included in the list of the 100 of the world's worst invasive alien species (Luque et al., 2013). However, the introduction of these species has not been spatially uniform, and countries such as Argentina, Brazil, Cuba, Mexico or Chile stand out in this respect (30 or more intentionally introduced species). On the contrary, other countries such as Nicaragua, El Salvador, Honduras, Paraguay or French Guiana have a lower incidence of species introductions (Fig. 1). Although this may also be due to the lower scientific production in these countries.

Furthermore, this pattern is not the same for birds and mammals. For example, in the case of birds, countries such as Peru, Mexico, Brazil, Argentina, and islands such as Puerto Rico, Jamaica, the Bahamas or Barbados, stand out in terms of the number of aliens' birds (Fig. 1A). In contrast, in the case of mammals, countries such as Argentina, Brazil, Chile, Cuba, Peru,

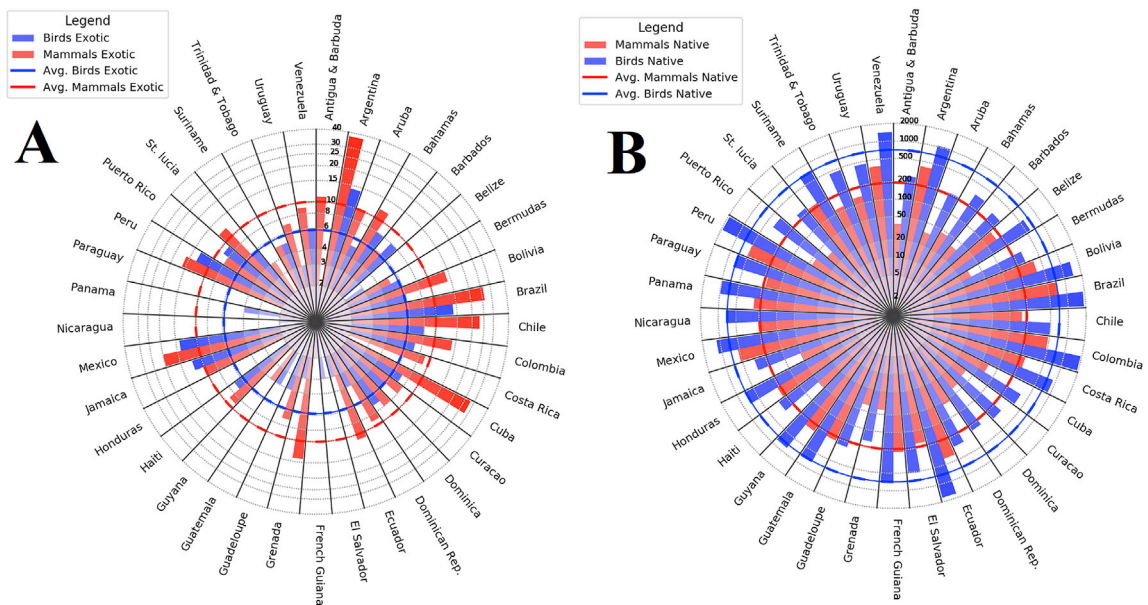


Fig. 1. Number of aliens' species of mammals and birds per country (A), and number of native species of mammals and birds per country (B). The data are shown on a logarithmic scale.

Mexico, Grenada or St. Lucia harbour a large variety of intentionally introduced species (Fig. 1A). However, normalized intentionally introduced alien species values (n° alien species intentionally introduced/land area of the country in km^2 ; Turbelin et al., 2017) show that tropical islands such as Bermuda, Aruba, Grenada or Antigua & Barbuda have highest ratio of alien species per land area of the country (between 0.1 and 0.025 species/ km^2 for these four islands). In contrast, the countries with the lowest ratio of alien species per land are of the country were Brazil, Paraguay, Nicaragua, Guatemala or Venezuela (<0.00001 species/ km^2). The origin of these species is also highly heterogeneous. The Neotropics biogeographic region stands out as the source of the majority of introduced species (34%). It is followed by Indomalaya (18.3%), the Palearctic region (14.5%) and the Afrotropic region (9.9%) (see Fig. S1).

3.2. Role of intentional introductions pathways in the context of alien species introductions

Of the 131 introduced species, 42.6% were introduced as pets or for aesthetic purposes (especially cage birds, 22%), while 23.7% are currently exploited as hunting species (Fig. 2B). We specifically noticed that of the 69 mammals and 62 birds introduced into Latin America (Fig. 2B), 27 mammal species (39%) and 13 bird species (22%) were introduced primarily for hunting purposes, 11.2% were released for feeding purposes, 5.3% for biological control, 4.2% for fur industry and 13% for others intentional purposes (research, military, working animals...) (Fig. 2B). Overall, our results further show that aesthetics and the pet trade were the most important pathways in the case of birds (at least 75% of the introduced species), while hunting was the main reason in the case of the introductions of mammals (39% of introduced species). (Fig. 2B).

3.3. Relative importance of different taxa

According to our results, 69 species of mammals (52.7%) and 62 species of birds (47.3%) were introduced by means of multiple intentional pathways (Fig. 2A). Most of the introduced mammal species were ungulates ($n = 27$) (Fig. 2B). In this respect, ungulates represent 39.1% of all the mammal species introduced into Latin America, where at least 18 species have been introduced as hunting species (Fig. 3A). Another well-represented group of mammals was that of primates, and at least 14 species were introduced for intentional purposes (pet/research/experimental), representing 20.3% of the introduced mammals (Fig. 2B). Rodents are also noteworthy, with 12 species (with different purposes: feeding, hunting, fur), which represented 17.4% of the mammals introduced (Fig. 2B).

Our results further show that the majority of intentionally introduced bird species belong to two orders: Passeriformes and Psittaciformes, accounting for 38.7% and 25.8% of the total number of introduced bird species, respectively (Fig. 2A). The main introduction pathway for these two taxa was the pet trade/aesthetic purposes (90% of species) (Fig. 3B). Galliformes is also a very important taxon, with a total of 9 introduced species (14.5% of the introduced birds). However, Ungulates and Psittaciformes are the most introduced groups, the number of species from these groups present in Latin America countries

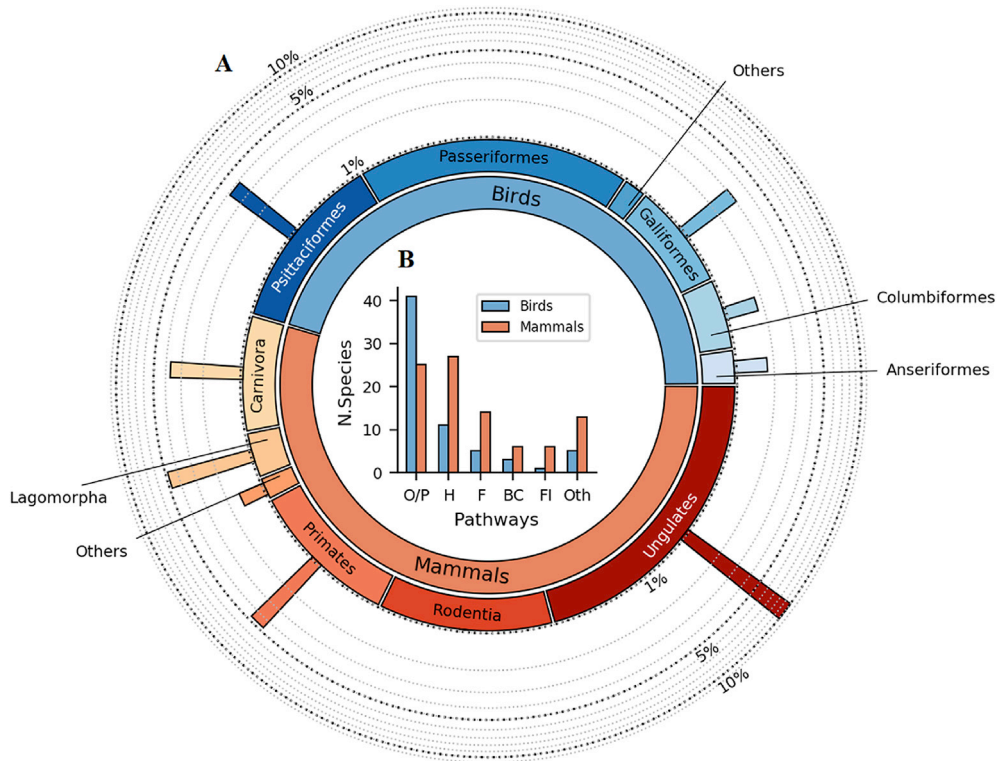


Fig. 2. A) Proportion of species from different taxonomic groups of birds and mammals that were, according to our review, intentionally introduced into Latin America. Vertical bars indicate the proportion of alien species in a group/total number of species in that group * 100 (in logarithmic scale). B) Number of species introduced based on the introduction pathway: O/P=Ornamental/Pet; H=Hunting; F= Feeding; BC=Biological control; FI=Fur industry and Oth = Other pathways, for birds and mammals separately.

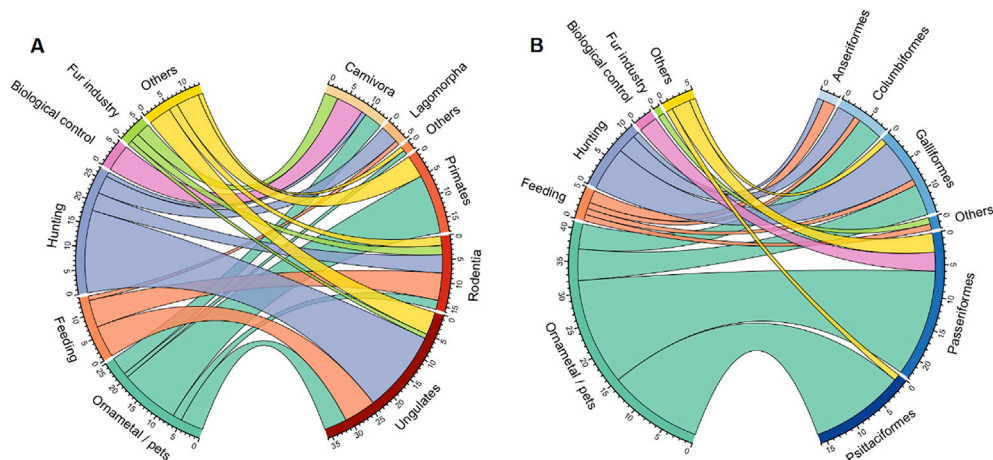


Fig. 3. The number of exotic species intentionally introduced by introduction pathways to a taxonomic group using chord diagrams: A) mammals, B) birds. The sources of information used for this analysis are shown in List S1a.

actually over-represents these groups (Ungulates and Psittaciformes). When comparing the number of introduced species in relation to total available species within such taxa in the world, both groups are overrepresented in the Latin America (10% and 4%) of species from these respective groups occur in at least one of the countries included in this review (Fig. S3). Instead, Rodentia and Passeriformes are the largest mammals and birds' groups and as such, the number of species from these groups were under-represented (only 0.6% and 0.4% of species from these groups were intentionally introduced in at least one country).

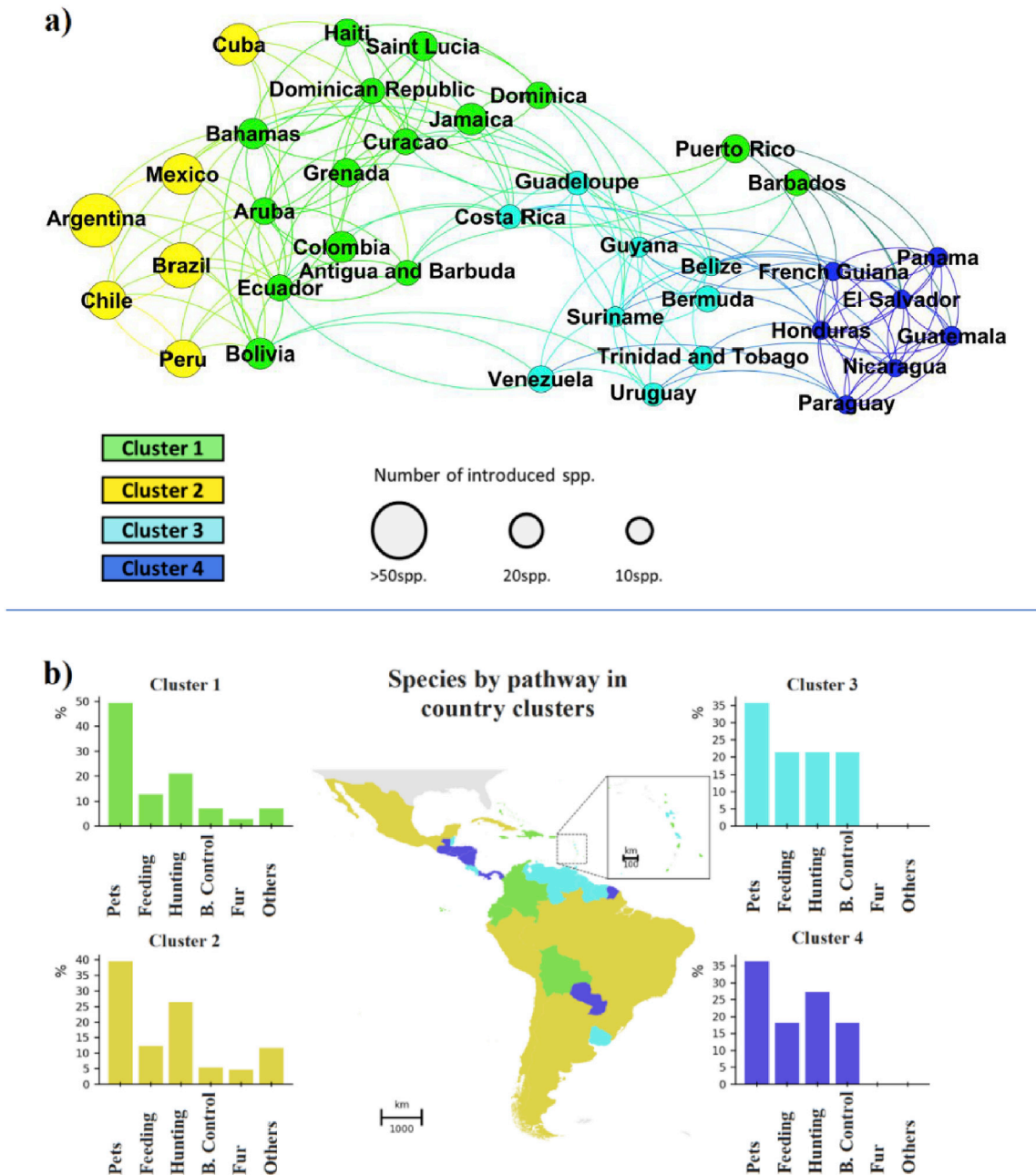


Fig. 4. A) Network plot showing countries distributed according to the composition of intentionally introduced species. The five closest Euclidean distances to each element are represented. The size of the nodes is related to the amount of species (Max: 55 Argentina, Min: 3 Nicaragua), while the longitude of the edges is related to the Euclidean distance between the two connected nodes. The Euclidean position of the elements is used for clustering: Cluster 1 (green), cluster 2 (yellow), cluster 3 (light blue) and cluster 4 (dark blue). B) Percentage of intentionally introduced species by means of different introduction pathways in each cluster. Map representing the clusters of grouped countries. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

3.4. Composition of intentionally introduced species throughout the continent

The clustering analyses carried out grouped different countries according to the similarity of the intentionally introduced species in their territories (Fig. 4A). Countries were grouped in 4 clusters (Figs. S2A and S2B), and some countries had intermediate features: i.e. Puerto Rico and Barbados belong to cluster 1, although the distance between them and the rest of the cluster is greater than that of the other countries (Fig. S2B). The network plot displays those countries between cluster 1 and cluster 4. Costa Rica and Guadeloupe in cluster 3 are, similarly, transitional cases between clusters 3 and 1. C1 is formed of

large countries into which the greatest number of species was introduced (Fig. S3). In Fig. 4A, each cluster is represented by a different colour. All the countries are clustered in four groups. The closer the countries in the figure, the more similar they are in terms of alien species. The farther the countries in the figure, the more different they are. For example, Guatemala and El Salvador are located both countries very close in Cluster 4, meaning that both have similar amount and kind of alien species introduced. Argentina and Chile are also together in Cluster 2. Keeping the same example, the couple of countries Guatemala-El Salvador are very far from the couple Argentina-Chile, meaning that the features of the species invasion are very distant. The main introduction pathway in all the clusters was the pet/ornamental trade, followed by hunting. Feeding and biological control were also an important pathway, especially in C3 and C4. However, other pathways stand out in C2 (Fig. 4B).

3.5. Factors that drive intentionally introduced species richness in Latin America

The factors retained in the best models (Models 1 and 2) employed to assess the effect of countries' characteristics on the total number of alien bird and mammal species intentionally introduced into each country are shown in Table 1. The results show that the % of GDP imports was statistically and positively associated with the total number of introduced bird species per country, while the % of rural population and number of pieces confiscated/million inhabitants were negatively related to this variable (Model 1). Furthermore, the number of mammals' species introduced per country was positively affected by the number of trafficked species/million inhabitants, whereas it was negatively associated with the % of rural population and number of pieces confiscated/million inhabitants (Model 2). Regarding the number of established species (Model 3), the results show that the three variables (number of native bird and mammal species and total number of introduced species) positively affected this variable (Table 1). However, the variable with the greatest effect was the number of species introduced ($F = 108.9$; $p < 0.001$).

3.6. Potential impacts of species intentionally introduced into Latin America

Finally, in this study, the impacts of these species are quantified according to the sources of information consulted (List S1c). Of the 131 species introduced intentionally, the results show 335 potential species-impacts (expressing that each species causes impacts in more than one category; Fig. 5). The greatest number of species-impacts (measured as the number of species in each category) are on the ecosystem (185 species-impacts), followed by economic impacts (97 species-impacts) and health impacts (53 species-impacts). Of the environmental impacts, the reduction in native biodiversity stands out (with 101 species-impacts), while with regard to the impacts on health, impact disease transmission stands out (48 species-impacts) and damage to agriculture/forestry was the main economic impact (44 species-impacts).

In addition, suitable data for socio-economic and environmental impacts was found in literature and database involving 95 species (72.5%) with EIACT and 69 species (52.7%) with SEICAT (List S1c). Most alien mammals and birds had low impacts, categorized as either minimal concern (MC) or minor (MN) (51.04% for EIACT) and (53.62% for SEICAT). However, 28 and 21 species had (MO) impacts (EIACT and SEICAT, respectively), 9 with major (MR) and 9 with massive (MV) and 10 with major (MR) and 1 with massive (MV) environmental and socioeconomic impacts, respectively (Table S3).

4. Discussion

4.1. General distribution patterns of intentionally introduced species

The Convention on Biological Diversity (Aichi target 9) states that 'by 2020, invasive alien species and pathways must be identified and prioritised (UNEP, 2010). The identification of the introduction pathway can inform management strategies

Table 1

Best models explaining the number of intentionally introduced alien bird (Model 1), alien mammal (Model 2) and number of established species (Model 3) in each country.

Variable	Estimate \pm S.E.	F-value	p-value
Number of intentionally introduced species of birds (Model 1)			
Intercept	2.8 \pm 0.24	11.64	<0.001
% GDP imports	0.002 \pm 0.01	19.5	<0.001
No. of pieces confiscated/million inhabitants	-0.005 \pm 0.006	6.65	<0.01
% Rural population	-0.02 \pm 0.01	4.91	<0.05
Number of intentionally introduced species of mammals (Model 2)			
Intercept	3.88 \pm 0.33	11.73	<0.001
% Rural population	-0.07 \pm 0.01	28.2	<0.001
No of trafficked species/million inhabitants	0.06 \pm 0.02	10.92	<0.01
No. of pieces confiscated/million inhabitants	-0.018 \pm 0.005	10.36	<0.01
Number of established species (Model 3)			
Intercept	1.20 \pm 0.14	70.31	<0.001
No of introduced species	0.07 \pm 0.01	108.87	<0.001
No of species of native mammals	0.004 \pm 0.001	9.57	<0.01
No. of species of native birds	0.001 \pm 0.0004	6.17	<0.05

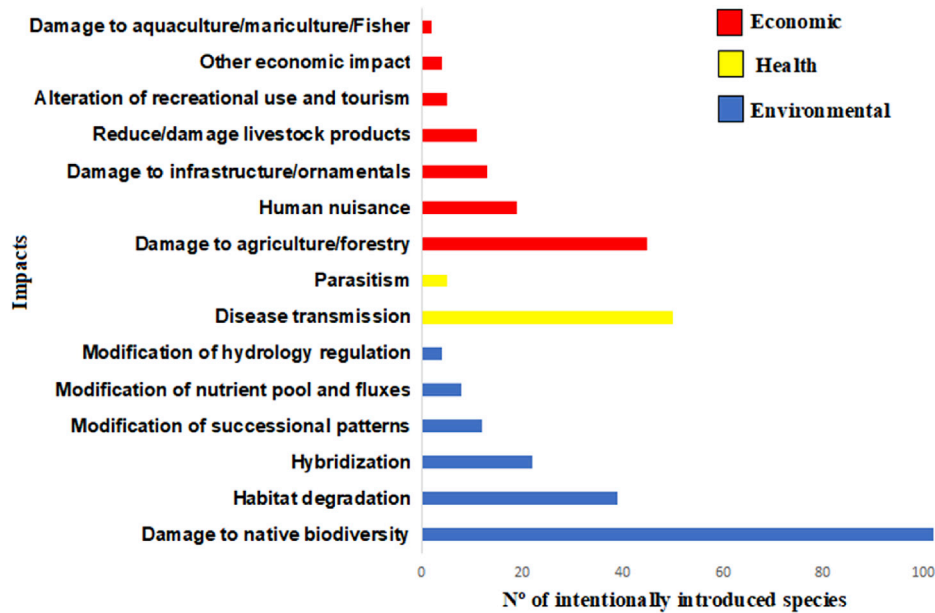


Fig. 5. Number of species intentionally introduced according to the impact generated.

that in turn can reduce the number of alien species (i.e. colonisation pressure) and individuals (i.e. propagule pressure) introduced (Hulme et al., 2008; Pergl et al., 2017). This work provides the first assessment of the introduction pathways for exotic mammals and birds in Latin America, which, together with earlier national contributions (Jaksic, 1998; Jaksic et al., 2002; Kairo et al., 2003; Novillo and Ojeda, 2008; Ballari et al., 2016; Da Rosa et al., 2017; Toral-Granda et al., 2017), broadens our understanding of the biological invasion pathways in the region.

Alien species have only recently (last 25 years) been considered as a problem for biodiversity in Latin America (Rodríguez, 2001; Speziale and Lambertucci, 2010; Ballari et al., 2016; Buenavista and Palomares, 2018). The International Council for Science (ICSU) has recognised that information on invasive species in this region is highly asymmetrical between the countries in terms of what information exists, what is readily available in each country, and research breadth (ICSU, 2009). In addition, Latin America has historically been a region into which many mammals and birds have been introduced since European colonisation (Crosby, 2003). According to our review, 69 mammal and 62 bird alien species were intentionally introduced into Latin America. Of the 869 alien and invasive species that negatively impact biodiversity found in GSID (including all taxa), 498 (57%) are registered as invasive in one or more countries/territories in Latin America (including all taxa), of which 41 appear in the list of 100 of the World's Worst Invaders, and 11 were intentionally introduced (Luque et al., 2013).

4.2. Role of intentional introduction pathways in the context of alien species introductions

According to our results, pet and ornamental traffic and hunting were the two most important introduction vectors for birds (44 and 12 species, respectively) and mammals (27 and 26 species) (Fig. 3A and B). This is of particular concern in Latin America, where pet keeping is popular, and is consequently increasing the consumer community for exotic pets, mainly birds (Alves et al., 2010; Bush et al., 2014). These results also coincide with Kraus (2003) or Carrete and Tella (2008), who showed that hunting and the pet trade were the dominant pathways for bird introduction (76% of the birds introduced), while feeding and hunting are the main reasons for the introductions of mammals (Goss and Cumming, 2013). Similar results were also found by Carpio et al. (2017b) in Europe, where 24% of the mammals and 30% of the birds introduced were released for hunting purposes. With regard to food production, Hulme et al. (2008) pointed out in a review that feeding and hunting were the primary introduction pathways for birds and mammals. In our case, feeding was the third most important reason for the introduction of mammals, with 14 species.

4.3. Relative importance of different taxa

Our review shows that the introduction of these species has been biased towards several wildlife orders, mainly Artiodactyls, Primates, Passeriformes and Psittaciformes (Fig. 2A). Ungulates stand out from the others (39% of the introduced mammal species), probably because of their importance in sport hunting (Spear and Chown, 2009; Flueck, 2010) and for feeding (Jenkins, 1996). The Primates constituted the mammal order with the second most introduced species ($n = 14$),

mainly for the pet trade and research experiments. The most common species of primates introduced for experiments are *Chlorocebus aethiops*, *Macaca mulatta*, and *M. fascicularis* (Carlsson et al., 2004). Bush et al. (2014) showed that the mammals most frequently introduced as pets were primates and carnivores. This study also showed that Parrots (*Psittaciformes*), and songbirds (*Passeriformes*), were the most common avian orders in the pet trade, which is in line with our results, in which *Psittaciformes* and *Passeriformes* represented 41.9% and 25.8% of the total bird species. Similar results are shown in Abellán et al. (2016), in which 70% of bird species introduced belonged to just three orders (*Passeriformes*, *Psittaciformes* and *Anseriformes*), and were introduced primarily as cage birds and ornamental species.

The overrepresentation of some taxa is shown in Fig. 2A. Parrots (pet trade) and Ungulates (hunting purposes) were reported more often than randomly expected (Bush et al., 2014), while other orders such as *Passeriformes* or *Rodentia* are underrepresented, principally owing to the large number of species in these taxa (Van Wilgen et al., 2018).

4.4. Composition of intentionally introduced species throughout the continent

We found well defined clusters of countries as regards the composition of intentionally introduced species (Fig. 4). Cluster 1 is formed mainly of Caribbean countries and the main reason for introducing species into those countries was ornamental/pets (~50% of the cases). This cluster is also characterised by a larger proportion of introduced bird species. Cluster 2 is formed of the largest countries, i.e. Argentina, México and Brazil. Hunting was a key reason for introducing species into these countries (>25% of the cases). Clusters 3 and 4 are quite similar as regards the composition of introduced species. Both clusters are formed of Caribbean countries, although islands (e.g. Guadeloupe, Belize, Bermuda, Trinidad and Tobago, ...) and south American countries (e.g. Guyana, Suriname, Venezuela,...) are more common in cluster 3 and those from central America (e.g. Panama, Guatemala, El Salvador...) are more common in cluster 4. The pathways Feeding and Biological control are very high in both clusters.

4.5. Factors that drive intentionally introduced species richness in Latin America

The country size variable was not retained in any model, therefore the spatial distribution of intentionally introduced species is independent of the size of the country. Larger continental territories often receive similar numbers of introduced species to smaller islands (Van Kleunen et al., 2015; Dawson et al., 2017; ICSU, 2009) because tropical and temperate oceanic islands seem to be especially sensitive to alien species (Loehle and Eschenbach, 2012). However, Argentina (37 species), Brazil or Chile stand out as regards the introduction of mammals (Novillo and Ojeda, 2008; Da Rosa et al., 2017; Jaksic, 1998). According to "extinction-based saturation", which is consistent with Island Biogeography Theory (IBT) the total number of species present in an area could be maintained as a balance between extinction and colonisation. One possible explanation for the lack of relationship between area and number of alien species is that biotic exchange when intentionally promoted by humans has little to do with the size of a country (Weber, 1997; ICSU, 2009).

Interestingly, countries with a higher rural population, such as Guatemala, Belize or Guyana, were characterised by a low number of intentionally introduced species, which reflects an increasing social demand for pets in largely urban societies (Carrete and Tella, 2008), unlike rural societies, which have a wide range of local pets (Paul and Serpell, 1992). Because of the intentional nature of these introduction pathways, those areas in which concentrated anthropogenic activities take place are, therefore, points of entry or release for alien species (Padayachee et al., 2017). This also concurs with the result that the % of GDP owing to imports is related to the number of aliens species, which is directly related to the transportation and movement of different products. Hulme (2009) showed that the exposure of economies to trade is highlighted by the significant role of merchandise imports in biological invasions, particularly in the case of island ecosystems. This result also coincides with those of Westphal et al. (2008) or Marini et al. (2011), who showed that the value of merchandise imports was a strong predictor of the number of exotic species. Our results coincide with those of these authors since country area or GDP per capita were not found to be important determinants of a country's degree of biological invasion. With regard to border controls, our results also show that the highest number of intentionally introduced species (birds and mammals) appears in countries with a lower number of pieces confiscated/million inhabitants, which may be owing to the lack of border controls. Recently, Cardador et al. (2019) demonstrated the effectiveness of a trade ban as regards preventing biological invasions. However, a regional ban can produce geographic redirections in trade, with important consequences for a worldwide invasion risk (Cardador et al., 2017), i.e. a redirection of trade toward developing countries or less regulated countries. Contrary to this, the number of trafficked species/million inhabitants was positively associated with the number of alien species of mammals, which is not surprising since the greater the number of species, the greater the risk of escape or release (Rosen and Smith, 2010). On the other hand, a significant predictor of number of established alien species was the number of native species (both birds and mammals), which had a positive relationship with the number of established alien species. This result coincides with the "the rich get richer" acceptance hypothesis, which predicts a higher number of established alien species in areas in which there is a high diversity of native species (Stohlgren et al., 2003, 2006; Fridley et al., 2007), since those areas with a high native diversity have a greater number of microneches (or more spatial heterogeneity), which would, therefore, allow a greater number of alien species to be accommodated (Davies et al., 2005). In addition, as expected, the number of introduced species was positively related to the number of established species. As the number of releases and/or the number of individuals released increases, propagule pressure also increases (Lockwood et al., 2005).

4.6. Potential impacts of intentionally introduced species in Latin America

Several studies have reported the impacts of alien mammals' species in the region. The European rabbit has impacted on a large part of Chile (Iriarte et al., 2005) or Argentina (Bonino and Soriguer, 2009). Barrios-García and Ballari (2012) showed the impacts of wild boar on the economy (crop damage), health (transmit diseases) and the environment in the form of the predation of some animal communities. Other studies, such as that by Flueck (2010), show the impacts of a certain group (Ungulates), or in a certain region (mammals in Argentina, Chile and Uruguay: Ballari et al., 2016). Information on birds is much more limited in the region, although species such as *Passer domesticus* or *Columbia livia* could potentially displace native passerines through competition for food or transmit parasites and diseases to native avifauna (Valenzuela et al., 2014).

The most common impacts of animal species are through changes in native biodiversity and habitat degradation (Ehrenfeld, 2010). Simberloff (2011) suggested that most invasions produce impacts on ecosystems, although many of these impacts are idiosyncratic, subtle or indirect. This author also proposes that the lag phenomenon in invasions implies that at least some existing alien species that are currently having little or no impact will eventually have much greater ones. It is, therefore, possible to predict that many alien species whose impact has not yet occurred or has not yet been recognised will eventually have impacts on ecosystems, which is very worrying in such a mega-diverse region.

5. Conclusions

Understanding the introduction pathways of alien species implies carrying out risk assessments, management, monitoring, and surveillance (Essl et al., 2015). This is especially important in Latin America, since it is one of the most biodiverse places in the world (Myers et al., 2000). In this respect, our review shows that countries with a higher rural population are characterised by a low number of intentionally introduced species, while there is no relationship between area and number of alien species. The results also show a higher number of alien species in areas with high native species diversity.

We recommend improvements to risk assessment and education in order to prevent escapes and translocation, and prioritised inspection strategies to reduce intentional introductions, since the % of GDP owing to imports was an important predictor of alien species. The traffic reports of CITES species should also be improved or increased, since the number of animals confiscated was related to a lower introduction of alien species (Cardador et al., 2019).

Declaration of competing interest

The authors declare that they have no conflict of interest.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.gecco.2020.e00949>.

References

- Abellán, P., Carrete, M., Anadón, J.D., Cardador, L., Tella, J.L., 2016. Non-random patterns and temporal trends (1912–2012) in the transport, introduction and establishment of exotic birds in Spain and Portugal. *Divers. Distrib.* 22, 263–273. <https://doi.org/10.1111/ddi.12403>.
- Alves, R., Nogueira, E.E.G., Araujo, H.E.P., Brooks, S.E., 2010. Bird-keeping in the Caatinga, NE Brazil. *Hum. Ecol.* 38, 147–156. <https://doi.org/10.1007/s10745-009-9295-5>.
- Anderson, L.G., Rocliffe, S., Haddaway, N.R., Dunn, A.M., 2015. The role of tourism and recreation in the spread of non-native species: a systematic review and meta-analysis. *PLoS One* 10 (10), e0140833. <https://doi.org/10.1371/journal.pone.0140833>.
- Atkinson, I.A.E., 1985. The spread of commensal species of *Rattus* to oceanic islands and their effect on island avifaunas. In: Moors, P.J. (Ed.), *Conservation of Island Birds*. International Council for Bird Preservation, Technical Publication, Cambridge, UK, pp. 35–81.
- Bacher, S., et al., 2018. Socio-economic impact classification of alien taxa (SEICAT). *Methods Ecol. Evol.* 9, 159–168. <https://doi.org/10.1111/2041-210X.12844>.
- Ballari, S.A., Anderson, C.B., Valenzuela, A.E., 2016. Understanding trends in biological invasions by introduced mammals in southern South America: a review of research and management. *Mamm. Rev.* 46, 229–240. <https://doi.org/10.1111/mam.12065>.
- Balzarini, M.G., González, L., Tablada, M., Casanoves, F., Di Rienzo, J.A., Robledo, C.W., 2008. InfoStat: software estadístico: manual del usuario. Córdoba, Argentina se.
- Barrios-García, M.N., Ballari, S.A., 2012. Impact of wild boar (*Sus scrofa*) in its introduced and native range: a review. *Biol. Invasions* 14, 2283–2300. <https://doi.org/10.1007/s10530-012-0229-6>.
- Bellard, C., Cassey, P., Blackburn, T.M., 2016. Alien species as a driver of recent extinctions. *Biol. Lett.* 12. <https://doi.org/10.1098/rsbl.2015.0623>, 20150623.
- Blackburn, T.M., Lockwood, J.L., Cassey, P., 2009. *Avian Invasions: the Ecology and Evolution of Exotic Birds*, vol. 1. Oxford University Press.
- Blackburn, T.M., Pyšek, P., Bacher, S., Carlton, J.T., Duncan, R.P., Jarosšík, V., Wilson, J.R.U., Richardson, D.M., 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, T.M., et al., 2014. A unified classification of alien species based on the magnitude of their environmental impacts. *PLoS Biol.* 12, e1001850. <https://doi.org/10.1371/journal.pbio.1001850>.

- Blackburn, T.M., Bellard, C., Ricciardi, A., 2019. Alien versus native species as drivers of recent extinctions. *Front. Ecol. Environ.* 17, 203–207. <https://doi.org/10.1002/fee.2020>.
- Bonino, N., Soriguer, R., 2009. The invasion of Argentina by the European wild rabbit *Oryctolagus cuniculus*. *Mamm. Rev.* 39, 159–166. <https://doi.org/10.1111/j.1365-2907.2009.00146.x>.
- Buenavista, S., Palomares, F., 2018. The role of exotic mammals in the diet of native carnivores from South America. *Mamm. Rev.* 48, 37–47. <https://doi.org/10.1111/mam.12111>.
- Burnham, K.P., Anderson, D.R., 2002. *Model Selection and Multimodel Inference: a Practical Information-Theoretic Approach*, second ed. Springer-Verlag, New York.
- Bush, E.R., Baker, S.E., Macdonald, D.W., 2014. Global trade in exotic pets 2006–2012. *Conserv. Biol.* 28, 663–676. <https://doi.org/10.1111/cobi.12240>.
- Cardador, L., Lattuada, M., Strubbe, D., Tella, J.L., Reino, L., Figueira, R., Carrete, M., 2017. Regional bans on wild-bird trade modify invasion risks at a global scale. *Conserv. Lett.* 10, 717–725. <https://doi.org/10.1111/conl.12361>.
- Cardador, L., Tella, J.L., Anadón, J.D., Abellán, P., Carrete, M., 2019. The European trade ban on wild birds reduced invasion risks. *Conserv. Lett.* 12, e12631 <https://doi.org/10.1111/conl.12631>.
- Carlsson, H.E., Schapiro, S.J., Farah, I., Hau, J., 2004. Use of primates in research: a global overview. *Am. J. Primatol.* 63, 225–237. <https://doi.org/10.1002/ajp.20054>.
- Carpio, A.J., Barasona, J.A., Guerrero-Casado, J., Oteros, J., Tortosa, F.S., Acevedo, P., 2017a. An assessment of conflict areas between alien and native species richness of terrestrial vertebrates on a macro-ecological scale in a Mediterranean hotspot. *Anim. Conserv.* 20, 433–443. <https://doi.org/10.1111/acv.12330>.
- Carpio, A.J., Guerrero-Casado, J., Barasona, J.A., Tortosa, F.S., Vicente, J., Hillström, L., Delibes-Mateos, M., 2017b. Hunting as a source of alien species: a European review. *Biol. Invasions* 19, 1197–1211. <https://doi.org/10.1007/s10530-016-1313-0>.
- Carpio, A.J., De Miguel, R.J., Oteros, J., Hillström, L., Tortosa, F.S., 2019. Angling as a source of non-native freshwater fish: a European review. *Biol. Invasions* 1–16. <https://doi.org/10.1007/s10530-019-02042-5>.
- Carrete, M., Tella, J., 2008. Wild-bird trade and exotic invasions: a new link of conservation concern? *Front. Ecol. Environ.* 6, 207–211. <https://doi.org/10.1890/070075>.
- CBD, 2014. Pathways of Introduction of Invasive Species, Their Prioritization and Management. <https://www.cbd.int/doc/meetings/sbstta/sbstta-18/official/sbstta-18-09-add1-en.pdf>.
- Clout, M.N., Russell, J.C., 2008. The invasion ecology of mammals: a global perspective. *Wildl. Res.* 35, 180–184. <https://doi.org/10.1071/WR07091>.
- Caribbean Conservation Association, 1991. *St. Lucia Country Environmental Profile*. St. Lucia Country Environmental Profile. Castries, Saint Lucia, p. 335.
- Crosby, A.W., 2003. *The Columbian Exchange: Biological and Cultural Consequences of 1492*, vol. 2. Greenwood Publishing Group.
- Da Rosa, C.A., de Almeida Curi, N.H., Puertas, F., Passamani, M., 2017. Alien terrestrial mammals in Brazil: current status and management. *Biol. Invasions* 19, 2101–2123. <https://doi.org/10.1007/s10530-017-1423-3>.
- Davies, K.F., Chesson, P., Harrison, S., Inouye, B.D., Melbourne, B.A., Rice, K.J., 2005. Spatial heterogeneity explains the scale dependence of the native-exotic diversity relationship. *Ecology* 86, 1602–1610. <https://doi.org/10.1890/04-1196>.
- Dawson, W., et al., 2017. Global hotspots and correlates of alien species richness across taxonomic groups. *Nat. Ecol. Evol.* 1, 0186 <https://doi.org/10.1038/s41559-017-0186>.
- Dyer, E.E., et al., 2017a. The global distribution and drivers of alien bird species richness. *PLoS Biol.* 15, e2000942 <https://doi.org/10.1371/journal.pbio.2000942>.
- Dyer, E.E., Redding, D.W., Blackburn, T.M., 2017b. The global avian invasions atlas, a database of alien bird distributions worldwide. *Sci. Data* 4, 170041. <https://doi.org/10.1038/sdata.2017.41>.
- Ehrlfeld, J.G., 2010. Ecosystem consequences of biological invasions. *Annu. Rev. Ecol. Evol. Syst.* 41, 59–80. <https://doi.org/10.1146/annurev-ecolsys-102209-144650>.
- Essl, F., et al., 2015. Crossing frontiers in tackling pathways of biological invasions. *Bioscience* 65, 769–782. <https://doi.org/10.1093/biosci/biv082>.
- Fiedler, L.A., 1990. Rodents as a food source. In: Davis, L.R., Marsh, R.E. (Eds.), *Proceedings of 14th Vertebrate Pest Conference*. Published at University of California, Davis, pp. 149–155.
- Flueck, W.T., 2010. Exotic deer in southern Latin America: what do we know about impacts on native deer and on ecosystems? *Biol. Invasions* 12. <https://doi.org/10.1007/s10530-009-9618-x>, 1909–1922.
- Fridley, J.D., et al., 2007. The invasion paradox: reconciling pattern and process in species invasions. *Ecology* 88, 3–17. [https://doi.org/10.1890/0012-9658\(2007\)88\[3:TIPRPA\]2.0.CO;2](https://doi.org/10.1890/0012-9658(2007)88[3:TIPRPA]2.0.CO;2).
- Goss, J.R., Cumming, G.S., 2013. Networks of wildlife translocations in developing countries: an emerging conservation issue? *Front. Ecol. Environ.* 11 (5), 243–250. <https://doi.org/10.1890/120213>.
- Gu, Z., Gu, L., Eils, R., Schlesner, M., Brors, B., 2014. Circlize implements and enhances circular visualization in R. *Bioinformatics* 30, 2811–2812. <https://doi.org/10.1093/bioinformatics/btu393>.
- Hulme, P.E., 2009. Trade, transport and trouble: managing invasive species pathways in an era of globalization. *J. Appl. Ecol.* 46, 10–18. <https://doi.org/10.1111/j.1365-2664.2008.01600.x>.
- Hulme, P.E., et al., 2008. Grasping at the routes of biological invasions: a framework for integrating pathways into policy. *J. Appl. Ecol.* 45, 403–414. <https://doi.org/10.1111/j.1365-2664.2007.01442.x>.
- ICSU (International Council for Science), 2009. Biodiversity Knowledge, Research Scope and Priority Areas: an Assessment for Latin America and the Caribbean. International Council for Science. <https://docplayer.net/25895438-Biodiversity-knowledge-research-scope-and-priority-areas-an-assessment-for-latin-america-and-the-caribbean.html>.
- Iriarte, J.A., Lobos, G.A., Jaksic, F.M., 2005. Invasive vertebrate species in Chile and their control and monitoring by governmental agencies. *Rev. Chil. Hist. Nat.* 78, 143–154. <https://doi.org/10.4067/S0716-078X2005000100010>.
- IUCN-ISSG, 2000. 100 of the World's Worst Alien Invasive Species. <http://www.iucngisd.org/gisd/?st=100ss>. (Accessed 31 March 2019).
- Jaksic, F.M., 1998. Vertebrate invaders and their ecological impacts in Chile. *Biodivers. Conserv.* 7, 1427–1445. <https://doi.org/10.1023/A:1008825802448>.
- Jaksic, F.M., Iriarte, J.A., Jiménez, J.E., Martínez, D.R., 2002. Invaders without frontiers: cross-border invasions of exotic mammals. *Biol. Invasions* 4, 157–173. <https://doi.org/10.1023/A:1020576709964>.
- Jenkins, P.T., 1996. Free trade and exotic species introductions. *Conserv. Biol.* 10, 300–302. <https://doi.org/10.1046/j.1523-1739.1996.10010300.x>.
- Kairo, M., Ali, B., Cheesman, O., Haysom, K., Murphy, S., 2003. *Invasive Species Threats in the Caribbean Region*. Report to the Nature Conservancy, Arlington, p. 132 pp. <http://hdl.handle.net/20.500.11822/15481>.
- Keller, R.P., Geist, J., Jeschke, J.M., Kühn, I., 2011. Invasive species in Europe: ecology, status, and policy. *Environ. Sci. Eur.* 23, 23. <https://doi.org/10.1186/2190-4715-23-23>.
- Kraus, F., 2003. *Invasion Pathways for Terrestrial Vertebrates. Invasive Species: Vectors and Management Strategies*. Island Press, Washington, DC, pp. 68–92.
- Kuhlenkamp, R., Kind, B., 2018. Introduction of non-indigenous species. In: *Handbook on Marine Environment Protection*. Springer, Cham, pp. 487–516.
- Kumschick, S., et al., 2015. Comparing impacts of alien plants and animals in Europe using a standard scoring system. *J. Appl. Ecol.* 52, 552–561. <https://doi.org/10.1111/1365-2664.12427>.
- Lever, C., 2005. *Naturalised Birds of the World*. Christopher Helm Ornithology, London, UK.
- Levine, J.M., D'Antonio, C.M., 2003. Forecasting biological invasions with increasing international trade. *Conserv. Biol.* 17, 322–326. <https://doi.org/10.1046/j.1523-1739.2003.02038.x>.

- Lizarralde, M., Escobar, J., Deferrari, G., 2004. Invader species in Argentina: a review about the beaver (*Castor canadensis*) population situation on Tierra del Fuego ecosystem. *Interciencia* 29, 352–356. <http://www.redalyc.org/articulo.oa?id=33909402>.
- Lockwood, J.L., Cassey, P., Blackburn, T., 2005. The role of propagule pressure in explaining species invasions. *Trends Ecol. Evol.* 20 (5), 223–228. <https://doi.org/10.1016/j.tree.2005.02.004>.
- Loehle, C., Eschenbach, W., 2012. Historical bird and terrestrial mammal extinction rates and causes. *Divers. Distrib.* 18, 84–91. <https://doi.org/10.1111/j.1472-4642.2011.00856.x>.
- Long, J.L., 2003. *Introduced Mammals of the World—Their History, Distribution and Influence*. Csiro Publishing, Collingwood.
- Luque, G.M., Bellard, C., Bertelsmeier, C., Bonnaud, E., Genovesi, P., Simberloff, D., Courchamp, F., 2013. Alien species: monster fern makes IUCN invader list. *Nature* 498, 37. <https://doi.org/10.1038/498037a>.
- Mack, R.N., Simberloff, D., Mark Lonsdale, W., Evans, H., Clout, M., Bazzaz, F.A., 2000. Biotic invasions: causes, epidemiology, global consequences, and control. *Ecol. Appl.* 10, 689–710. <https://doi.org/10.2307/2641039>.
- Maechler, M., Rousseeuw, P., Struyf, A., Hubert, M., Hornik, K., 2018. *Cluster: Cluster Analysis Basics and Extensions*. R Package Version 2.0, 7-1.
- Marini, L., Haack, R.A., Rabaglia, R.J., Toffolo, E.P., Battisti, A., Faccoli, M., 2011. Exploring associations between international trade and environmental factors with establishment patterns of exotic Scolytinae. *Biol. Invasions* 13, 2275–2288. <https://doi.org/10.1007/s10530-011-0039-2>.
- McKinney, M.L., 2008. Effects of urbanization on species richness: a review of plants and animals. *Urban Ecosyst.* 11, 161–176. <https://doi.org/10.1007/s11252-007-0045-4>.
- Myers, N., Mittermeier, R.A., Mittermeier, C.G., Da Fonseca, G.A., Kent, J., 2000. Biodiversity hotspots for conservation priorities. *Nature* 403, 853–858. <https://doi.org/10.1038/35002501>.
- Novillo, A., Ojeda, R.A., 2008. The exotic mammals of Argentina. *Biol. Invasions* 10, 1333–1344. <https://doi.org/10.1007/s10530-007-9208-8>.
- Oteros, J., et al., 2019. Building an automatic pollen monitoring network (ePIN): selection of optimal sites by clustering pollen stations. *Sci. Total Environ.* 688, 1263–1274. <https://doi.org/10.1016/j.scitotenv.2019.06.131>.
- Padayachee, A.L., Irlich, U.M., Faulkner, K.T., Gaertner, M., Procheş, Ş., Wilson, J.R., Rouget, M., 2017. How do invasive species travel to and through urban environments? *Biol. Invasions* 19, 3557–3570. <https://doi.org/10.1007/s10530-017-1596-9>.
- Pauchard, A., Quiroz, C.L., García, R., Anderson, C.B., Kalin, M.T., 2011. *Invasiones biológicas en América Latina y El Caribe: tendencias en investigación para la conservación*. In: Simonetti, J.A., Dirzo, R. (Eds.), *Conservación biológica: Perspectivas desde América Latina*. Editorial Universitaria, Santiago, pp. 79–90.
- Paul, E.S., Serpell, J., 1992. Why children keep pets: the influence of child and family characteristics. *Anthrozoös* 5, 231–244. <https://doi.org/10.2752/089279392787011340>.
- Pereira-Garbero, R., Barreneche, J.M., Laufer, G., Achaval, F., Arim, M., 2013. Mamíferos invasores en Uruguay, historia, perspectivas y consecuencias. *Rev. Chil. Hist. Nat.* 86, 403–421. <https://doi.org/10.4067/S0716-078X2013000400003>.
- Pergl, J., et al., 2017. Troubling travellers: are ecologically harmful alien species associated with particular introduction pathways? *Neobiota* 32, 1–20. <https://doi.org/10.3897/neobiota.32.10199>.
- Pimentel, D., 2014. *Biological Invasions: Economic and Environmental Costs of Alien Plant, Animal, and Microbe Species*. CRC Press, Taylor & Francis Group, Baton Rouge.
- Pyšek, P., Richardson, D.M., Pergl, J., Jarošík, V., Sixtova, Z., Weber, E., 2008. Geographical and taxonomic biases in invasion ecology. *Trends Ecol. Evol.* 23, 237–244. <https://doi.org/10.1016/j.tree.2008.02.002>.
- R Core Team, 2018. *R: A Language and Environment for Statistical Computing*. R: a language and environment for statistical computing, Vienna, Austria, 2017. Retrieved from: <https://www.Rproject.org/>.
- Richardson, D.M., Pyšek, P., Carlton, J.T., 2011. A compendium of essential concepts and terminology in biological invasions. In: Richardson, D.M. (Ed.), *Fifty Years of Invasion Ecology: the Legacy of Charles Elton*. Blackwell Publishing, Chichester, pp. 409–420.
- Rodríguez, J.P., 2001. Exotic species introductions into South America: an underestimated threat? *Biodivers. Conserv.* 10, 1013–1023. <https://doi.org/10.1023/A:1013151722557>.
- Rosen, G.E., Smith, K.F., 2010. Summarizing the evidence on the international trade in illegal wildlife. *EcoHealth* 7, 24–32. <https://doi.org/10.1007/s10393-010-0317-y>.
- Seebens, H., et al., 2017. No saturation in the accumulation of alien species worldwide. *Nat. Commun.* 8, 14435. <https://doi.org/10.1038/ncomms14435>.
- Simberloff, D., 2011. How common are invasion-induced ecosystem impacts? *Biol. Invasions* 13, 1255–1268. <https://doi.org/10.1007/s10530-011-9956-3>.
- Spear, D., Chown, S.L., 2009. The extent and impacts of ungulate translocations: South Africa in a global context. *Biol. Conserv.* 142, 353–363. <https://doi.org/10.1016/j.biocon.2008.10.031>.
- Spear, D., Foxcroft, L.C., Bezuidenhout, H., McGeoch, M.A., 2013. Human population density explains alien species richness in protected areas. *Biol. Conserv.* 159, 137–147. <https://doi.org/10.1016/j.biocon.2012.11.022>.
- Speziale, K., Lambertucci, S., 2010. A call for action to curb invasive species in South America. *Nature* 467, 153. <https://doi.org/10.1038/467153c>.
- Speziale, K.L., Lambertucci, S.A., Carrete, M., Tella, J.L., 2012. Dealing with non-native species: what makes the difference in South America? *Biol. Invasions* 14, 1609–1621. <https://doi.org/10.1007/s10530-011-0162-0>.
- Stohlgren, T.J., Barnett, D., Flather, C., Fuller, P., Peterjohn, B., Kartesz, J., Master, L.L., 2006. Species richness and patterns of invasion in plants, birds, and fishes in the United States. *Biol. Invasions* 8, 427–447. <https://doi.org/10.1007/s10530-005-6422-0>.
- Stokes, K.E., Montgomery, W.L., Dick, J.T.A., Maggs, C.A., McDonald, R.A., 2006. The importance of stakeholder engagement in invasive species management: a cross-jurisdictional perspective in Ireland. *Biodivers. Conserv.* 15, 2829–2852. <https://doi.org/10.1007/s10531-005-3137-6>.
- Stohlgren, T.J., Barnett, D.T., Kartesz, J.T., 2003. The rich get richer: patterns of plant invasions in the United States. *Front. Ecol. Environ. Times* 1, 11–14. [https://doi.org/10.1890/1540-9295\(2003\)001\[0011:TRGRPO\]2.0.CO;2](https://doi.org/10.1890/1540-9295(2003)001[0011:TRGRPO]2.0.CO;2).
- Toral-Granda, M.V., et al., 2017. Alien species pathways to the Galapagos islands, Ecuador. *PLoS One* 12, e0184379. <https://doi.org/10.1371/journal.pone.0184379>.
- Turbelin, A.J., Malamud, B.D., Francis, R.A., 2017. Mapping the global state of invasive alien species: patterns of invasion and policy responses. *Global Ecol. Biogeogr.* 26, 78–92. <https://doi.org/10.1111/geb.12517>.
- UNEP, 29 October 2010. *The Strategic Plan for Biodiversity 2011–2020 and the Aichi Biodiversity Targets*. Nagoya, Japan: COP CBD Tenth Meeting UNEP/CBD/COP/DEC/X/2. Report No.: UNEP/CBD/COP/DEC/X/2.
- Valenzuela, A.E., Anderson, C.B., Fasola, L., Cabello, J.L., 2014. Linking invasive exotic vertebrates and their ecosystem impacts in Tierra del Fuego to test theory and determine action. *Acta Oecol.* 54, 110–118. <https://doi.org/10.1016/j.actao.2013.01.010>.
- Van Kleunen, M., et al., 2015. Global exchange and accumulation of non-native plants. *Nature* 525, 100–103. <https://doi.org/10.1038/nature14910>.
- Van Wilgen, N.J., Wilson, J.R.U., Elith, J., Wintle, B.A., Richardson, D.M., 2010. Alien invaders and reptile traders: what drives the live animal trade in South Africa? *Anim. Conserv.* 13, 24–32. <https://doi.org/10.1111/j.1469-1795.2009.00298.x>.
- Van Wilgen, N.J., Gillespie, M.S., Richardson, D.M., Measey, J., 2018. A taxonomically and geographically constrained information base limits non-native reptile and amphibian risk assessment: a systematic review. *PeerJ* 6, e5850. <https://doi.org/10.7717/peerj.5850>.
- Vilà, M., et al., 2010. How well do we understand the impacts of alien species on ecosystem services? A pan-European, cross-taxa assessment. *Front. Ecol. Environ.* 8, 135–144. <https://doi.org/10.1890/080083>.
- Weber, E.F., 1997. The alien flora of Europe: a taxonomic and biogeographic review. *J. Veg. Sci.* 8, 565–572. <https://doi.org/10.2307/3237208>.
- Westphal, M.I., Browne, M., MacKinnon, K., Noble, I., 2008. The link between international trade and the global distribution of invasive alien species. *Biol. Invasions* 10, 391–398. <https://doi.org/10.1007/s10530-007-9138-5>.
- Wickham, H., 2016. *ggplot2: Elegant Graphics for Data Analysis*. Springer.
- Wickham, H., 2017. *The tidyverse*. Available from: <http://tidyverse.org>.
- Zuur, A.F., Ieno, E.N., Walker, N.J., Saveliev, A.A., Smith, G.M., 2009. *Mixed Effects Models and Extensions in Ecology with R*. Springer, New York.