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LETTER

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The European trade ban on wild birds reduced invasion risks

Laura Cardador[1] | José L. Tella[2] | José D. Anadón[3,4] | Pedro Abellán[5] | Martina Carrete[2,6]

[1]Centre for Biodiversity and Environment Research, Department of Genetics, Evolution, and Environment, University College London, London, WC1E 6BT, United Kingdom

[2]Department of Conservation Biology, Estación Biológica de Doñana (CSIC) Américo Vespucio, 41092, Sevilla, Spain

[3]Department of Biology, Queens College, City University of New York, 65-30 Kissena Blvd Flushing New York, NY, 11367, USA

[4]Ecology, Evolution and Behavior Subprogram, The Graduate Center, City University of New York, 365 5th Ave New York, NY, 10016, USA

[5]Department of Zoology, Universidad de Sevilla, Avenida Reina Mercedes 6, 41012 Sevilla, Spain

[6]Department of Physical, Chemical and Natural Systems, Universidad Pablo de Olavide, Ctra. Utrera km 1, 41013 Sevilla, Spain

Correspondence
Laura Cardador, Centre for Biodiversity and Environment Research, Department of Genetics, Evolution, and Environment, University College London, London WC1E 6BT, United Kingdom.
Email: lcardador81@gmail.com

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Abstract

International wildlife trade is a major source of current biological invasions. However, the power of trade regulations to reduce invasion risks at large, continental scales has not been empirically assessed. The European wild bird trade ban was implemented in 2005 to counter the spread of the avian flu. We tested whether the ban reduced invasion risk in two European countries, where 398 nonnative bird species were introduced into the wild from 1912 to 2015. The number of newly introduced species per year increased exponentially until 2005 (in parallel with the volume of wild bird importations), and then sharply decreased in subsequent years. Interestingly, a rapid trade shift from wild-caught birds to captive-bred birds, which have lower invasive potential than wild-caught birds, allowed the maintenance of bird availability in markets. Our results demonstrate the effectiveness of a trade ban for preventing biological invasions without impacting the ability to meet societal demands.

KEYWORDS
biological invasions, nonnative species, pet markets, trade regulations, wild-caught birds

1 | INTRODUCTION

Biological invasions are a significant component of global change through their effects on biodiversity, ecosystems, and human societies (Millennium Ecosystem Assessment 2005; Vilà et al., 2010). Awareness about the impacts of biological invasions and the need for effective management has led to a persistent effort to understand introduction pathways and the factors driving invasion success. Increasing international trade and human transport is now recognized as an important and rapidly growing source of introduction of nonnative species worldwide (Hulme, 2009). Wildlife trade, in particular, has been directly related to the introduction of nonnative vertebrate species (Carrete & Tella 2008). Because established invaders are very difficult and costly to eradicate, it is widely agreed that the regulation of the wildlife trade is an effective strategy to prevent new invasions (Mack et al., 2000; Simberloff et al., 2013).

Trade measures against invaders often take the form of restrictions, such as black and white lists, quarantines, and punitive actions against individuals or companies that do not comply (Keller, Geist, Jeschke, & Kühn, 2011). The number of imported individuals and species is expected to positively correlate with the number of accidentally escaped or released (i.e., introduced) individuals/species (Abellán, Carrete, Anadón, Cardador, & Tella, 2016; Cardador, Carrete, Gallardo & Tella, 2016). Moreover, propagule pressure and colonization pressure (Lockwood, Cassey, & Blackburn, 2009) are known to be major predictors of invasion success and alien species richness (Blackburn, Lockwood, & Cassey, 2015; Lockwood, Cassey, & Blackburn, 2005, 2009). For these reasons, it has been assumed that both the number of introduction events and, in general, invasion risks should decrease after trade restrictions (Simberloff et al., 2013). While studies directly assessing the effects of trade restrictions on invasion risks are absent, the declines in nonnative plants established in New Zealand in the 1990s, coinciding with the application of stringent biosecurity policies (Biosecurity Act adopted in 1993), seems to support this idea (Seebens et al., 2017). Along the same lines, Simberloff et al. (2013) showed that Europe and New Zealand had similar invasion rates of nonnative mammals through the 19th century, but no invasions occurred in New Zealand during the 20th century (contrary to Europe) after public perceptions shifted and biosecurity policies were adopted. Nevertheless, the generalized increase in the numbers of nonnative species worldwide raises doubts about the effectiveness of past regulations (Seebens et al., 2017). One important aspect that may explain this inconsistency is the fact that trade regulations are much more difficult to apply in contiguous countries or regions than on islands because of the absence of geographically distinct borders. Moreover, the responsibility for protection against invasive species lies mostly with national governments. Thus, global patterns in the trends of nonnative species richness may obscure differences in those same patterns among countries or regions due to the disparity in regulations (Cardador et al., 2017). Assessment of the effectiveness of trade regulations requires specific analyses at the regional level where those regulations have been implemented.

The European Wild Bird Trade Ban (EU ban hereafter) prohibits the import of wild-caught birds into the European Union countries and was adopted in October 2005. It was initially a temporal measure aimed at preventing the spread of the avian flu, but it was made permanent in 2007 and its focus was broadened to include the conservation of traded species and animal welfare. Although not directly related to invasion management, this regulation is likely to have affected invasion risks, given the invasive potential of wild-caught traded birds (Abellán, Tella, Carrete, Cardador, & Anadón, 2017; Carrete & Tella, 2008, 2015). Two recent modeling approaches have predicted that the EU ban may have reduced invasion risks in target regions, while legal trade fluxes were redirected to other regions along with predicted risks (Cardador et al., 2017; Reino et al. 2017). Thus far, however, no empirical support for their effectiveness has been provided.

Here, we assessed (i) the effectiveness of the EU ban to reduce invasion risks at an early stage of the invasion process, using yearly data of nonnative birds recorded in the wild in Spain and Portugal for over 100 years. We then explored the underlying mechanisms through which the ban may have affected bird introductions by assessing (ii) the temporal pattern in the legal transport of wild-caught birds and its relationship to introduction numbers and (iii) the changes in bird availability for sale in local pet markets.

2 | METHODS

2.1 | Changes in wild bird introductions

As a measure of invasion risk, we focused on the annual number of new nonnative species recorded in the wild. As invasion is a multistage process, in which introduced species constitute the pool of species from which establishment, and thereafter spread (stage when a species is considered invasive), can take place (Blackburn et al., 2011). Information on bird introductions was obtained from a comprehensive dataset compiling 15,915 records of 398 nonnative species observed in the wild from 1912 to 2015 in Spain and Portugal (updated at the beginning of 2017 from the dataset provided by Abellán et al., 2016). This dataset is based on a systematic review of scientific and gray literature, complemented with our own data and unpublished observations from researchers. Data from 2016 and early 2017 were not considered in analyses, since they were expected to be incomplete due to lags between observations of new introduced species and their reporting/publication. This lag was considered reasonable based on our previous compiled data (Abellán et al., 2016). Other studies have used more conservative approaches. For example, more than 10 years of data were disregarded in a study of established species by Seebens et al. (2017), where lags between introduction and establishment that are usually much larger can also affect data quality.

The year that each species was first recorded in the wild was obtained to estimate the number of new species introduced per year. First record date was available for 327 species (82.16%). The other 71 species were reported for the first time in regional or local publications without exact dates of first record— only the publication year was available (one species published in 2001, six in 2002, 21 in 2003, and 43 in 2006). For these latter species, we considered the year before the publication as the first record date, given the usual lag between observations and their printed publication according to our previous compiled data. Thus, for all of these 71 species first record date was before 2005, so their inclusion in analyses may slightly alter the temporal patterns before the EU ban, but not the ban effect.

We assessed potential differences in the temporal trend of new introductions before and after the EU ban by testing the interaction "year × period" in a generalized linear model (GLM), using the number of new nonnative species observed each year as a response variable (Poisson error distribution; log-link function). Period was a categorical variable with two levels ("before" and "after" the EU ban). To control for variations in reporting patterns over time, we also included the total number of records in each year (i.e., the total number of nonnative bird occurrences in the updated Abellán et al., 2016 database). All continuous variables were standardized before modeling. Then, to predict the number of new nonnative species expected in the absence of the ban, we used a model calibrated only on the data from 1976 to 2005. This model prediction assumes that current exponential trends in introductions (which did not show saturation, Supporting Information Figure S1) might be maintained in the future, without limitations related to, for example, a potential depletion of incoming species pools or regional saturation. To further account for these potential constraints, we considered an alternative, more conservative approach where the number of new introduced species was predicted based on expected bird imports (i.e., number of wild-caught nonnative species or individuals imported, Figure 1A) using Michaelis–Menten models (Seebens et al., 2017). Under this alternative approach, the relationship between introductions and imports was assumed to be nonlinear, saturating at high import values (see Supporting Information Appendix S1 and Figure S2). The Michaelis–Menten models were calibrated using import (see below) and introduction data from 1976 to 2005 and then used for predictions in the period 2006–2015. Import values for the period 2006–2015 were derived from a Weibull

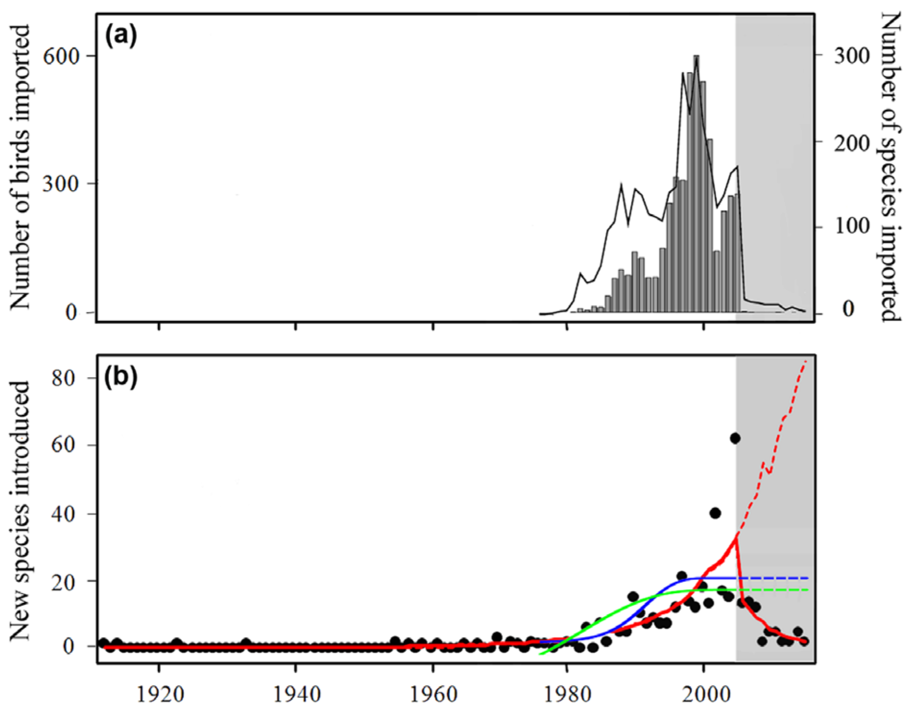

FIGURE 1 Temporal changes in the number of bird introductions and importations before and after the EU ban. Panel (a) shows annual numbers of wild-bird species (black line) and imported individuals in thousands (grey bars). Note that CITES records began in 1976. Panel (b) shows observed values (black points) of new introduced species, fitted values according to a GLM (solid red line), and predicted values (dotted red line) after the ban according to the previous trend. Fitted values according to Michaelis–Menten models (solid lines) based on the number of wild-bird individuals (blue) and species (green) imported and predicted values (dotted lines) after the ban are also shown

distribution accounting for recent declines in the number of individuals and species imported (see Supporting Information Appendix S1 and Figure S3 for more details).

2.2 | The role of the wild-bird trade in bird introductions

We assessed the relationship between the number of new nonnative species observed each year (response variable) and the annual numbers of wild-caught bird individuals and species imported by fitting a GLM (Poisson error distribution; log-link function). To account for the discrepancy between reported first occurrences (somewhere in a given year) and trade (bird import totals for the whole year), we considered the relationship between the number of new nonnative species observed in a given year x and the number of species/birds imported in the year $x - 1$. Preliminary analyses considering longer temporal lags did not improve model accuracy (but note very similar results for some of them, e.g., 4-year lag, Supporting Figure S4). Both linear and quadratic effects of predictors were considered. We also included the total number of nonnative birds recorded in the wild in each year to control for variations in reporting patterns over time. Importations were obtained as the total number of live birds reported by CITES (http://www.cites.org) from 1976 (the first year for which CITES compiled records) to 2015. For a more detailed description of importation data and potential caveats see Supporting Information Appendix S2.

2.3 | Short-term changes in bird markets

We assessed potential changes in bird availability for sale in local pet markets by visiting 19 Spanish pet shops from September 2004 to September 2007 (mean: 7.5 visits per pet shop). In each visit, we counted the number of individuals available for sale from each species and their origin (wild-caught or captive-bred, see Carrete & Tella, 2015 and Supporting Information Appendix S3 for details on their determination). We then compared the abundance of individuals, the richness (i.e., total number of species) and diversity (measured by means of the Shannon–Wiener diversity index) of species (GLM: normal error distribution; identity link function) and the proportion of wild birds (GLM: binomial error distribution; logistic link function) before and after the ban. Since only the wild bird trade was prohibited by the ban, our main expectation was a shift toward captive-bred species.

3 | RESULTS

3.1 | Change in introduction numbers

The numbers of new nonnative bird species annually recorded increased from the beginning of the 20th century at an average annual growth rate of 4.5%, reaching a total of 336 species before the EU ban. The annual increase was even sharper (18%) when excluding the period previous to 1986. However, after the EU ban, the number of new nonnative species drastically decreased (interaction period × year, $Z = 6.21$; $P < 0.001$; Figure 1B) at an average annual growth rate of –29%, dropping to only two species in 2015. Temporal changes in the number of new nonnative species introduced paralleled the annual numbers of wild individuals and species imported in the previous year (67.3% of the variability in introductions explained by these variables, Figure 1 and Supporting Information Table S1). The pattern of decline observed after the ban contrasts with both values predicted from the previous exponential trend (see consistent results when omitting exceptional introduction values in 2002 and 2005, Supporting Information Figure S5) and more constrained Michaelis–Menten models (Figure 1B).

3.2 | Bird availability for sale in local pet markets

We recorded 223 bird species in Spanish pet markets (Supporting Information Table S2). Neither the abundance of individuals nor the richness or diversity of species available at pet markets decreased significantly after the ban (Figure 2 and Supporting Information Figure S6). This was due to a shift in the sources of commercialized birds, from wild to captive-bred (Figure 2), also causing a change in species composition (Supporting Information Table S2).

4 | DISCUSSION

Our results support the often assumed—but scarcely tested—link between international wildlife trade and effective introduction numbers, suggesting that the EU ban reduced invasion risks by limiting potential invaders at early stages of the invasion process. The pattern of decline observed after the ban contrasts with values predicted from the previous exponential trend and more constrained models. As a correlational study, however, we cannot discount that other factors may have resulted in the observed pattern or acted synergistically with the ban. This could be the case of the economic downturn associated with the start of the economic recession in Spain and Portugal in 2008. However, decreases in introduction rates were already observed between 2005 and 2008 and were steeper than those expected simply from changes in Gross Domestic Product (GDP; Supporting Information Figure S7), suggesting that introduction rates were mostly influenced by the ban. Even so, 62 new bird species were recorded during the period 2006–2015. This could be explained by the accumulated number of species already traded before the ban (by 2012, more than 1,000 nonnative

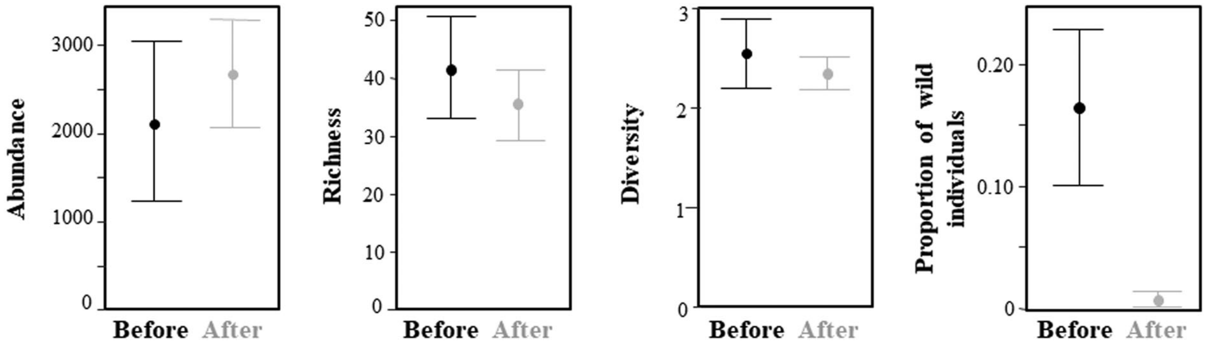

FIGURE 2 Bird availability for sale in local pet markets. Changes in the abundance of individuals (number of individuals), the richness (number of species) and diversity (measured by means of the Shannon–Wiener diversity index) of species, and the proportion of wild birds in the Spanish pet market one year before (before: black lines) and one year and a half after (after: grey lines) the EU ban (average values and 95% CI are shown). Abundance of individuals and the richness and diversity of species (normal error distribution; identity link function) were similar before and after the ban ($F = 1.00$, $P = 0.3235$; $F = 1.58$, $P = 0.2174$; and $F = 1.21$, $P = 0.2787$, respectively) while the proportion of wild birds (binomial error distribution; logistic link function) differed between the two periods ($\chi^2 = 12428$, $P < 0.001$)

bird species were or had been kept in captivity in Spain and Portugal; Abellán et al., 2016) and potential temporal lags between importation and introduction into the wild (Abellán et al., 2016). While our results suggest that overall introduction probability increases soon after increases in import values (1-year lag), variation in temporal patterns may exist across species (Aagaard & Lockwood 2014). It is worth mentioning that the extent of illegal trade, another important concern associated with bans (Rivalan et al., 2007), remains today rather anecdotal in Spain and Portugal compared to legal bird trade before the ban (Figure 1A). According to available data, only 145 illegally traded birds were confiscated in the period 2007–2011 (Mundy-Taylor, 2013). In any case, this illegal trade, which is difficult to quantify, would have only masked the ban effects in the context of our study, making our results more conservative.

The reduction in the number of new introduced species was accompanied by a rapid shift in the sources of commercialized birds, from wild-caught to captive-bred birds. Although some wild individuals were registered in low numbers after the ban (mainly during the first months, see Fig. S6, as there was still a stock of wild-caught birds to be sold), their main origin changed after 2005 (Table S2). Limited data suggests that this shift could have been possible initially by obtaining captive-bred birds from other European countries (mostly The Netherlands and Belgium), with a longer tradition of breeding exotic birds in captivity, and reinforced until now by an increase in the size and number of breeding facilities in Spain and Portugal to satisfy the increasing national demand (authors unpubl. observations). Nowadays, in Spain and Portugal there are tens of legally constituted exotic bird associations, each one bringing together thousands of aviculturists (e.g., the association to which one of the authors belongs (JLT; *Aviornis Ibérica*) comprises > 2,200 Spanish and Portuguese aviculturists).

The implications of the shift from wild caught to captive bred in reducing invasion risks are threefold. First, the intraspecific variability in survival probability of wild-caught birds with different traits from capture in source countries to escape or release in importing countries may select for individuals with phenotypes and genotypes that make them better invaders (Baños-Villalba, 2018; Carrete et al., 2012; Mueller et al., 2017). Second, captive-bred birds have less ability to survive in the wild than wild-caught, due to changes in behavioral and physiological traits (Cabezas, Carrete, Tella, Marchant, & Bortolotti, 2013; Carrete & Tella 2015), and would have lower probabilities of being successfully introduced (e.g., they have lower escaping abilities Carrete & Tella 2015) and subsequently established (e.g., almost all escaped individuals are recaptured or predated, Carrete & Tella, 2008, 2015). As such, breeding origin is one of the main factors influencing invasion success of current non-intentional bird introductions (Abellán et al., 2017). Third, our observation of market prices suggested that captive-bred birds were more expensive than their wild-caught conspecifics at the beginning of the ban. This could make captive bred species more valuable, maybe reducing the probability of voluntary or involuntary releases. However, more accurate temporal data on prices is needed, as a reduction in values similar to those of wild-caught individuals by 2005 seemed to occur as captive breeding increased and the commercialized species changed.

Despite past concerns and heated debates arising from the blanket ban on the wild bird trade in the EU (e.g., Cooney & Jepson 2005; Rivalan et al., 2007), our results suggest that this ban helped reduce the introduction of alien birds, which constitute the pool of species from which establishment, and thereafter spread, can take place (Abellán et al., 2016, 2017). The long-term persistence of the EU ban, or any other trade ban, may, however, depend on its effects on local markets. If there are major economic costs and the societal demand for traded goods is not satisfied, society could press for the restoration of the wildlife trade, especially when the original goal of the ban (in the case of the EU ban, to avoid the spread of the avian flu) loses importance over time. However, our data show that while the EU ban was largely effective in drastically reducing the importation and availability of wild-caught birds in the market, it did not significantly affect the general availability of nonnative birds for sale.

However, to avoid unintended consequences of the EU ban or other regional bans, such as unexpected geographic redirections or taxonomic changes in the international pet trade (Cardador et al., 2017; Reino et al. 2017), more global intercontinental strategies that address biological invasions as a global issue are required. Applying the precautionary principle, blanket bans, such as the EU ban, should be seriously considered at a global level. However, blanket bans are widely debated (Cooney & Jepson 2005; Rivalan et al., 2007; Roe, 2006) and could be difficult to apply. Alternatively, a trade regulation framework similar to that developed by CITES—the main current international instrument available to monitor and control wildlife trade of threatened species—should be created with the aim of creating binding international standards to regulate, monitor, and control the trade of potentially harmful species in both importer and exporter countries. Although international policy responses to combat biological invasions have increased over the last several decades (McGeoch et al., 2010), responsibility for protection against invaders lies mostly on national governments. This has led to important differences in legislation among countries, even for those signatories of the Convention on Biological Diversity, which includes prevention, eradication, and control of invasive species as a commitment (McGeoch et al., 2010). Additionally, when applied, this legislation mostly takes the form of defensive measures (mainly bans and quarantines) to protect particular importer countries or regions against the potentially harmful effects of imported species. This legislation is often underpinned by prioritizated lists of the more risky species for the particular importer countries or regions, as in the recently proposed EU regulation 1143/2014 (Carboneras et al., 2018; Tollington et al., 2017). While these regulations offer an option to reduce the invasion likelihood in countries or regions of implementation, they do not tackle the problem of invasive species as a global issue, as risky species can still be exported to other countries.

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of the article.

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