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Research Article

Checklist and prioritization for management of non-native species of phanerogam plants and terrestrial vertebrates in eight protected areas on the Ecuadorian coast

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Abstract

Protected areas (PAs) are important tools for biodiversity conservation. In developing countries, incomplete information about the presence of non-native species makes it difficult to develop strategies to manage this threat to biodiversity. Although a list of non-native species for continental Ecuador has recently been published, information on the status of these species in PAs in terms of invasiveness and impacts is scarce. This study presents a method that proposes criteria to prioritize the management of non-native phanerogam plants and terrestrial vertebrate species in eight PAs in the coastal region of Ecuador, based on the minimal information available. The study area covers 79.6% of all coastal PAs. For the non-native species inventory, we collated information from global biodiversity databases, research papers, theses and project repositories from local private and public universities, public institutions, and management plans for the selected PAs. To categorize the monitoring priority of the non-native species from the selected PAs, we used a pipeline scheme based on species invasion risk principles. We registered 78 non-native species within the eight PAs, consisting of 64 phanerogam plants and 14 terrestrial vertebrates. The PA with the highest non-native plant species richness is Manglares Churute Ecological Reserve (ER) with 25 species (39.1%; n = 64). For terrestrial vertebrates, the highest non-native species richness was in Isla Santay National Recreation Area (NRA) (71.4%; n = 14). 15.6% of the phanerogam plants and 78.5% of the terrestrial vertebrates were classified in the high priority category. The majority of non-native plants are categorized as "more studies required", reflecting the state of art of invasion ecology in this country. This method allowed us to classify the species based on theoretical and occurrence data. It can potentially be replicated throughout the country and used as a rapid assessment method, complemented with specific invasion/impact studies on PAs in Ecuador.

Key words: conservation areas, conservation strategies, Ecuador, exotic species, risk assessment

Introduction

Protected areas (henceforth PAs) are important tools for biodiversity conservation (Margules and Pressey 2000). The main goal of PAs is to conserve ecosystems and their functions, as well as to function as tourist attractions, thus generating significant revenue for biodiverse countries (Roman and Nahuelhual 2009). Major threats faced by PAs include habitat loss, illegal trade and biological invasions (Pauchard and Villarroel 2002; Laurance et al. 2012; Spear et al. 2013). In developing countries, there is not enough information about the extent of invasion by non-native species in PAs, making it difficult to develop strategies to manage or control these species to ensure conservation objectives (Tu 2009; Nuñez and Pauchard 2010; Speziale et al. 2012).

PAs were conceived to protect a portion of biodiversity. However, the number of non-native species inside PAs is increasing worldwide (Foxcroft et al. 2013). In Africa, the 19 national parks in South Africa reported 663 introduced species, 20% of them with high invasive potential (Spear et al. 2013). For instance, the Kruger National Park has the highest number of non-native species, with a total of 350 introduced plants, followed by Table Mountain with 239 non-native plant species (Spear et al. 2013). In Europe, 450 to 500 non-native plants have been reported in the PAs in the Alps, representing 10% of the area's total flora and 20% of all non-native plant species in the continent (Kueffer 2010).

In the United States, 3756 non-native species have been reported in 216 PAs that cover a total of 7.3 million hectares (Allen et al. 2008). In South America, Brazil has reported 1170 non-native species in 227 PAs and 902 non-native species in 163 fully protected protected areas and 268 non-native species in 64 sustainable use protected areas (Ziller and de Sá Dechoum 2013). In Argentina, Patagonian PAs report 400 introduced species, of which 50% are mammals and fishes and 25% are vascular plants (Sanguinetti et al. 2014). In southern and central Chile, 39 species of plants have been reported in PAs (Pauchard and Alaback 2004).

It is well known that non-native species inside PAs affect biodiversity, mainly through competition and predation on native species. For example, in Laguna Blanca National Park (Argentina) the non-native trout *Oncorhynchus mykiss* (Walbaum, 1792) has negatively affected populations of the native trout *Percichthys trucha* (Valenciennes, 1833) (Ortubay et al. 2006). In Lanín National Park (Argentina), the spread of the non-native deer *Cervus elaphus* Linnaeus, 1759 has modified the structure and composition of vegetation through competition with native herbivores (Vázquez 2002). In Venezuela, in Cerro Saroche National Park, invasion by the non-native succulent *Kalanchoe daigremontiana* Raym.-Hamet & H.Perrier has modified nutrient cycles by mineralizing carbon and nitrogen in the soil (Herrera et al. 2018a), with detrimental effects on native cacti (Herrera et al. 2016).

In Ecuador, the Galapagos Islands have been the target of most of the country's research on invasion biology (e.g., Guézou et al. 2010; Toral-Granda et al. 2017; Torres and Mena 2018; Carlton et al. 2019; Shackleton et al. 2020); however, such studies are almost nonexistent in PAs in continental Ecuador, which cover approximately 20% of the country (Yánez 2016). Although a list of non-native species for continental Ecuador has recently been published (Herrera et al. 2019), information about the invasive status of these species in protected areas is very scarce. In Isla Santay NRA, a protected area on the Ecuadorian coast, invasion by the non-native palm *Roystonea oleracea* O.F. Cook has been documented (Herrera et al. 2017). A further 12 species of non-native plants have also been reported in this PA (Herrera et al. 2018b). Other studies in Sangay National Park, located in the Eastern Cordillera of the Ecuadorian Andes, have recorded the presence of the invasive rat *Rattus rattus* (Linnaeus, 1758), one of the most damaging invasive rodents worldwide (Brito and Ojala-Barbour 2014).

The lack of information on the status of non-native species in PAs of continental Ecuador makes further research imperative in order to devise rapid action plans to prevent, manage and control invasive species (Speziale et al. 2012). This task is urgently required if the 2030 action targets related to reducing threats to biodiversity, established in the Zero Draft of the Post-2020 Global Biodiversity Framework, are to be attained. Governments will have to adopt a new set of biodiversity targets during talks in Kunming, China, in October, to replace the 2020 goals agreed on Aichi, Japan, in 2010 – most of which have been missed.

This study is an attempt to fill these knowledge gaps in relation to non-native species in continental Ecuador. In order to do so, we present an inventory of non-native phanerogam plants and terrestrial vertebrate species within eight PAs in the coastal region of Ecuador. In addition, we present a protocol based on decision trees to prioritize invasive species with presumably higher impacts based on the minimal information available.

Materials and methods

Study area

The study area includes eight PAs located in the Ecuadorian coastal region (Figure 1). This region extends from the Andean foothills to the Pacific Ocean and represents 26% of the country total territory. The composition and structure of the region is influenced by the Tumbes-Choco-Magdalena hotspot and three forest formations; humid tropical evergreen forests in the north, tropical dry forests in the south and mangrove forests along the coast. Of the 21 PAs in the coastal region, we selected eight. The selection was based on the following five criteria: (i) terrestrial protected areas (four marine areas were excluded); (ii) PAs without highly disturbed ecosystems (i.e., without built recreational infrastructure inside the PA, high tourism pressure, urban solid waste, high evidence of habitat fragmentation, or PAs surrounded by urbanized landscape), (iii) at least one PA from each province in the coastal region; (iv) the selected PAs must cover the majority

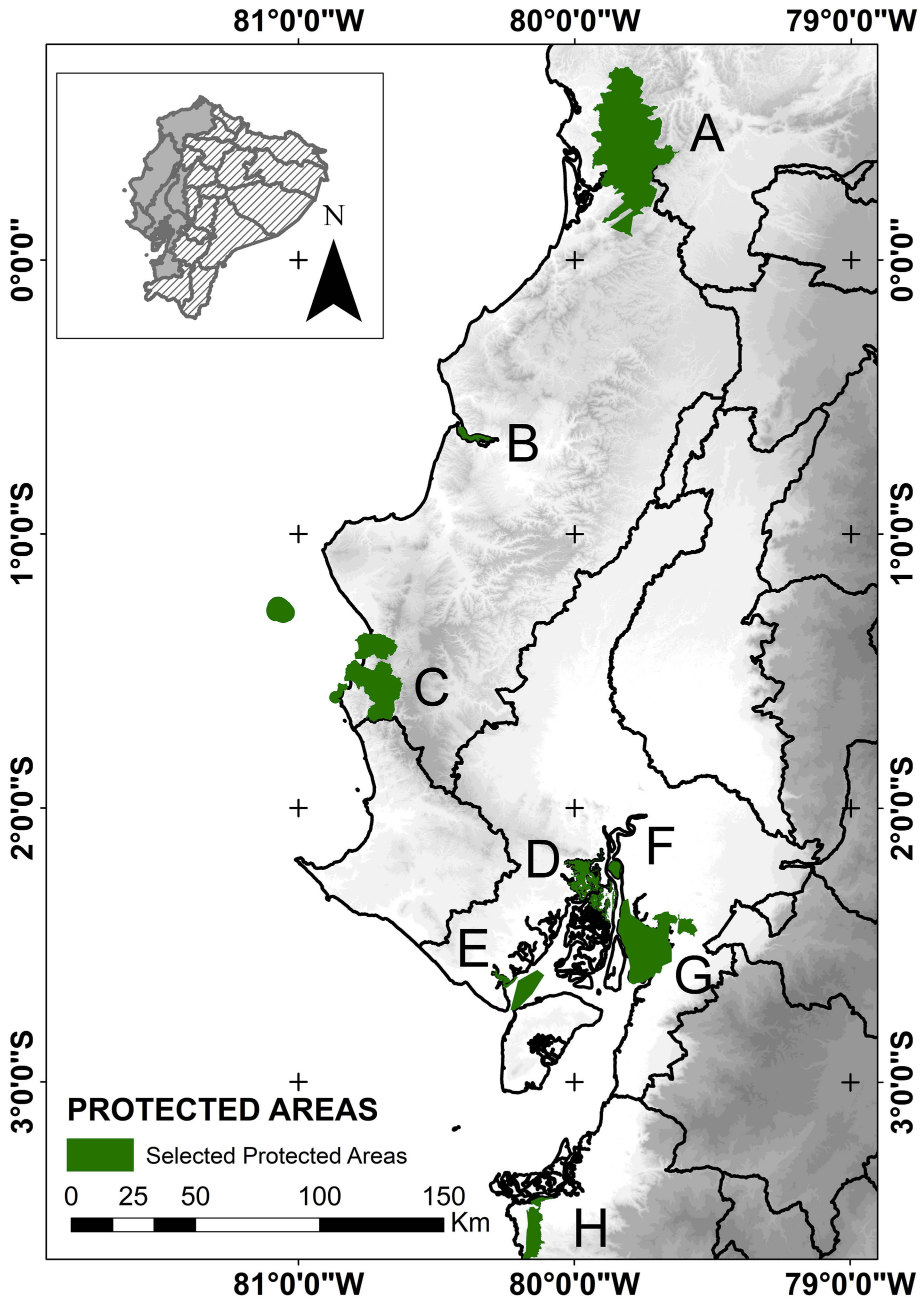

Figure 1. Selected PAs in the coastal region of Ecuador. A: Mache-Chindul ER; B: Islas Corazón y Fragatas WR; C: Machalilla NP; D: Manglares El Morro WR; E: Manglares El Salado FPR; F: Isla Santay NRA; G: Manglares Churute ER; H: Arenillas ER.

of ecosystems and national management categories; (v) PAs with information on non-native species.

The eight selected PAs cover 249,176 ha, representing 79.6% of all coastal terrestrial PAs; they also include 13 of the 15 protected ecosystem types, including all the management categories for terrestrial PAs in the National System of PAs in Ecuador; i.e., NP: National Park, ER: Ecological Reserve, FPR: Fauna Production Reserve, WR: Wildlife Refuge and NRA: National Recreation Area) (Table 1). Table 1 describes the most important characteristics of the eight selected PAs of the coastal region. The table includes PA name, conservation category, year of creation, land area (ha),

Table 1. Characteristics of the eight selected PAs in the coastal region of Ecuador.

Protected area	Management category	Year	Area (ha)	Elevation (m)	Land cover type [a]
Mache-Chindul	Ecological Reserve (ER)	1996	119172	200–800	Evergreen Broadleaf Forest, Deciduous Broadleaf Forest, Mixed Forest, Closed Shrubland, Open Shrubland, Grasslands, Permanent Wetland, Croplands, Cropland/Natural Vegetation Mosaic, and Barren or Sparsely Vegetated
Islas Corazón and Fragatas	Wildlife Refuge (WR)	2002	2811	0–0	Permanent Wetland, Urban and Built-Up, and Barren or Sparsely Vegetated
Machalilla	National Park (NP)	1979	41754	0–840	Water, Evergreen Broadleaf Forest, Deciduous Broadleaf Forest, Mixed Forest, Open Shrubland, Grasslands, Permanent Wetland, Croplands, Urban and Built-Up, Cropland/Natural Vegetation Mosaic, and Barren or Sparsely Vegetated
Manglares El Morro	Wildlife Refuge (WR)	2007	10030	0–28	Permanent Wetland
Manglares El Salado	Fauna Production Reserve (FPR)	2002	10635	0–200	Water, Evergreen Broadleaf Forest, Mixed Forest, Closed Shrubland, Grasslands, Permanent Wetland, Croplands, Urban and Built-Up, and Cropland/Natural Vegetation Mosaic
Isla Santay	National Recreation Area (NRA)	2010	2215	0–10	Water, Evergreen Broadleaf Forest, Mixed Forest, Grasslands, Permanent Wetland, Croplands, and Cropland/Natural Vegetation Mosaic
Manglares Churute	Ecological Reserve (ER)	1979	49389	0–680	Water, Evergreen Broadleaf Forest, Deciduous Needle leaf Forest, Deciduous Broadleaf Forest, Mixed Forest, Closed Shrubland, Open Shrubland, Grasslands, Permanent Wetland, Croplands, Cropland/Natural Vegetation Mosaic, and Barren or Sparsely Vegetated
Arenillas	Ecological Reserve (ER)	2001	13170	0–300	Evergreen Broadleaf Forest, Deciduous Broadleaf Forest, Closed Shrublands, Open Shrublands, Grasslands, Permanent Wetland, Croplands, and Cropland/Natural Vegetation Mosaic

elevation (m) and ecosystems. The largest PA is Mache-Chindul with 119,172 ha, followed by Manglares Churute with 49,389 ha. The smallest PA is Isla Santay with 2,214 ha, followed by Isla Corazón and Fragatas with 2,811 ha. The oldest PAs are Mache Chindul and Manglares Churute, created in 1979. The more recent PA is Isla Santay, created in 2010. The PAs with the greatest diversity of ecosystems are Machalilla and Manglares Churute, and the PA with the lowest diversity of ecosystems is Manglares El Morro.

Non-native species inventory

For the non-native species inventory, we conducted an extensive search using five information sources. First, global and national biodiversity databases; i.e., the Global Register of Introduced and Invasive Species – GRIIS (Herrera et al. 2019), Catalogue of Vascular Plants of Ecuador – W^{3}CEC (Missouri Botanical Garden 2019), the iNaturalist dataset (iNaturalist 2019), an online database of bird distribution and abundance (eBird 2019), and the Global Biodiversity Information Facility (GBIF 2019). Second, research papers from scholarly web browsers and databases (ISI web of knowledge, SCOPUS, Scielo and Google scholar) using the descriptors "protected area AND non-native species AND Ecuador", "protected area AND invasive species AND Ecuador", "protected area AND flora AND Ecuador", and "protected area AND fauna AND Ecuador". Third, thesis and project repositories from local private and public universities. Fourth, information provided by public institutions related to biodiversity in Ecuador (National Biodiversity Institute and Ministry of Environment). Fifth, management plans for the selected PAs retrieved from the environmental information system website (SUIA, by its Spanish acronym, available at http://suia.ambiente.gob.ec/). Additionally, to detect the presence of other unreported non-native species within the PAs, we downloaded the geographic coordinates of all non-native species from the GRIIS checklist of introduced and invasive species – Ecuador (Herrera et al. 2019) and projected them onto the map of Ecuadorian PAs using ArcGIS (version 10.4) and selected the ones with occurrences inside the study area. Finally, for each protected area, we estimated total species richness and weighted richness; i.e., number of species per unit area (1 km^2).

Prioritization

To prioritize the monitoring of non-native species, we used a probability tree based on principles of invasion risk. Risk estimation is made up of two components: invasion likelihood (invasiveness) and potential consequences given that species are invasive (Hulme 2011, 2014).

To discern species invasiveness, we used two criteria; the first criterion was the climate suitability of species in Ecuador compared to its native range and other places where it has been reported (Zalba and Ziller 2007). To estimate similarity, we evaluated climate overlap between the native and invasive area of distribution of the species and continental Ecuador, based on the Köppen-Geiger climate classification. The second criterion was to check whether invasion records were available for each species in the CABI Invasive Species Compendium (CABI 2019), Global Invasive Species Database (ISSG 2015), Global Register of Introduced and Invasive Species, Inter-American Biodiversity Information Network- I3N Brazil (Instituto Hórus 2013), or on CONABIO datasheets in Mexico (CONABIO 2020). Since some of the species categorized as invasive in the CABI Invasive Species Compendium only affect agricultural areas, we filtered the status of the species reported as invasive only by CABI. For these species, we checked whether the report included natural ecosystems.

To assess non-native species impacts, we checked whether they were included in the list of the world's 100 worst invasive alien species (Lowe et al. 2004). In addition, we checked whether the species have generated economic, cultural, health or biodiversity impacts in Colombia; we used this information because Colombia has a similar ecological background (both countries contain a tropical Pacific coast ecosystem) and social background

Table 2. Risk estimation criterion.

Component	Criterion	Description	Source/ Method Basis
Invasion Probability	1 Climate Suitability	Climate correspondence between native area of distribution of the species and regions where the species behaves as invasive	Zalba and Ziller (2007)
	2 Invasion Records	Records of the species behaving as an invasive organism outside its native distribution range	CABI (2019); ISSG (2015)
Impact Probability	3 Detrimental capacity of the species	Check whether the species was included in the 100 most invasive species of the world list, as these have proven detrimental impacts in biodiversity and ecosystems	Lowe et al. (2004)
	4 Reported impacts in similar conditions	Check whether the species has generated impacts in Colombia, the neighboring country with similar ecological and social background as Ecuador	Baptiste et al. (2010)

to Ecuador, as published in the report of the Alexander von Humboldt Research Institute (Baptiste et al. 2010; see Table 2).

The information obtained for each non-native species was used to establish its priority level, passing non-native species through a probability tree. This instrument enables a series of independent events to be represented in a way that dichotomizes species. In our case, the criteria for species separation were climatic match, invasiveness and impact, thus determining which species require high priority actions. To do this, we classified the non-native species within the PAs as: i) high priority (climate match: yes, invasive records: yes; impact records: yes), ii) medium priority (climate match: yes, invasive records: yes; impact records: no), iii) low priority (climate match: no, invasive records: no; impact records: no) or iv) more studies required (climate match: no, invasive records: yes; impact records: no / climate match: yes, invasive records: no; impact records: no) (for more details see Figure 2).

Results

Non-native species inventory

We registered 78 non-native species inside the selected PAs (Supplementary material Table S1), of which 64 species are phanerogams (or flowering plants) (82.1%; n = 78) and 14 are terrestrial vertebrates (17.9%; n = 78). The PA with the highest non-native plant species richness is Manglares Churute ER with 25 (39.1%; n = 64). The highest non-native terrestrial vertebrate species richness was in Isla Santay NRA (71.4%; n = 14) and Machalilla NP (71.4%; n = 14). Manglares El Morro WR registered the lowest species richness for plant species (1.5%; n = 64) and terrestrial vertebrates (14.3%; n = 14) (Figure 3a). Isla Santay NRA presented the highest value for weighted richness by area (1.26 non-native species per km^2) and also for each taxonomic group (phanerogam plants = 0.81 species per km^2; terrestrial vertebrates = 0.45 species per km^2). The lowest weighted richness per unit area was registered in Mache-Chindul ER, again both in total species (0.01 species per km^2) and taxonomic groups (phanerogams = 0.01 species per km^2; terrestrial vertebrates = 0.003 species per km^2) (Figure 3b).

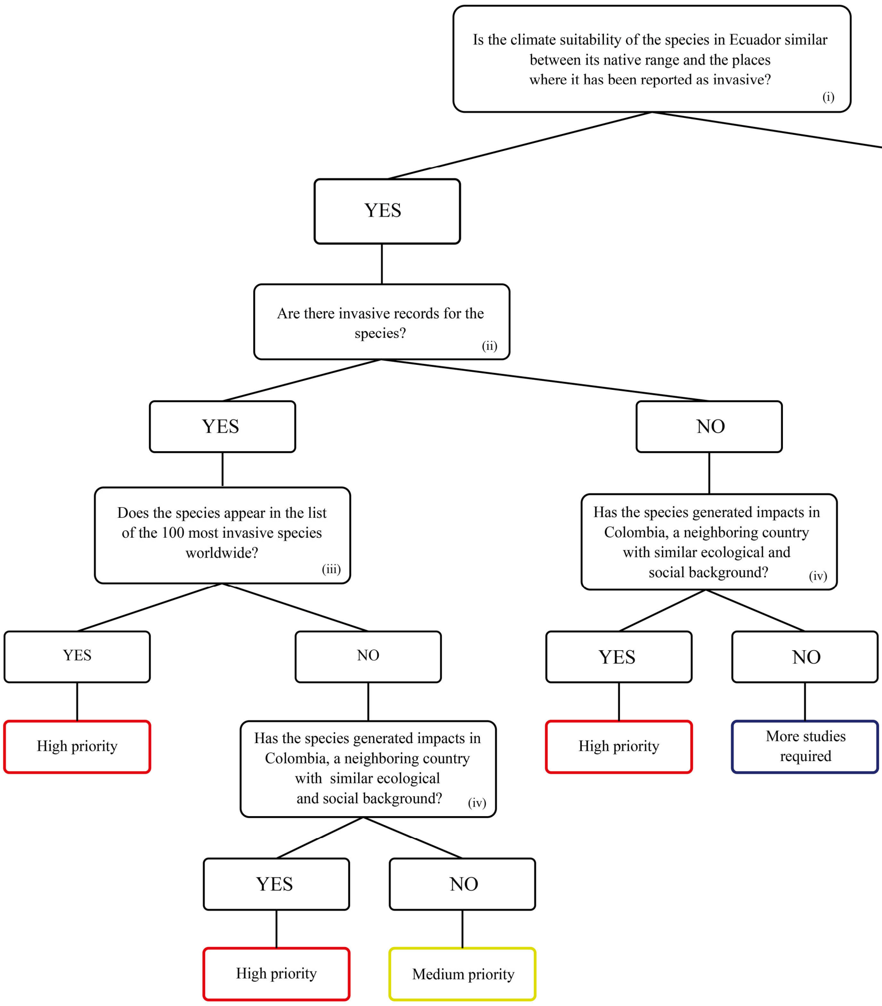

Figure 2. Decision tree determining the pipeline paths for categorization of monitoring priority of the non-native species. The reported species within the study protected areas were classified as: i) High priority, ii) Medium priority, iii) Low priority or iv) More studies required. Four criteria were used for this: i) climate suitability; ii) identifying whether the species had invasion records in the databases (i.e. CABI 2019; ISSG 2015); iii) identifying whether the species is included in the 100 most invasive species worldwide list (Lowe et al. 2004); and iv) identifying whether the species had generated impacts in Colombia (Baptiste et al. 2010).

Non-native plant species represented 31 taxonomic families. Poaceae was the most abundant family with 11 species (17.2%; n = 31), followed by Fabaceae with six species (9.4%; n = 31) (Figure 4). The most common habit was herbaceous with 33 species (61.1%; n = 64), and the least common was vines with four species (7.4%; n = 64) (Table S1). The most common plant species were *Megathyrsus maximus* (Jacq.) B.K. Simon & S.W.L. Jacobs and *Mangifera indica* L., each present in four of the eight PAs (Table S1). A total of 10 orders of non-native terrestrial vertebrates containing 13 taxonomic families were identified. The family Muridae was best represented, with two species (15.4%; n = 14), and only one species for the other families. The most frequent vertebrate species were *Columba livia* (Gmelin, 1789) and *Passer domesticus* (Linnaeus, 1758), present in all PAs, closely followed by *Bubulcus ibis* (Linnaeus, 1758), present in seven PAs (Table S1).

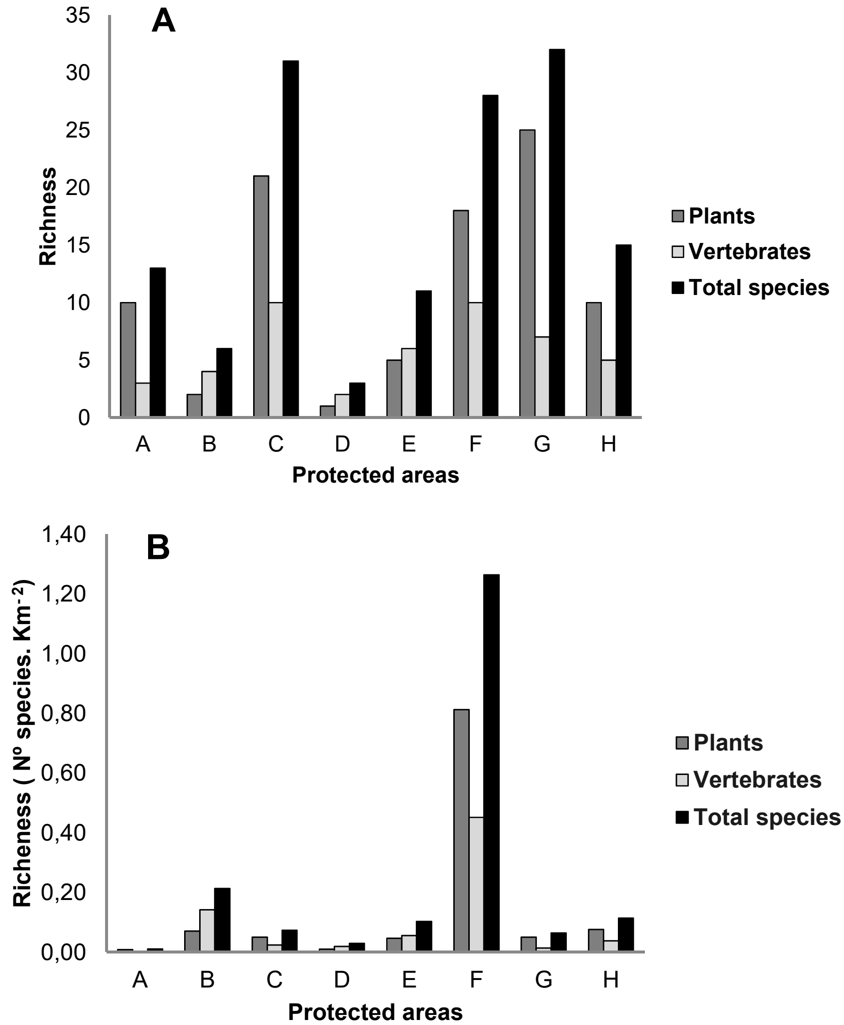

Figure 3. Richness (A) and richness weighted by area (B) of phanerogams and terrestrial vertebrates by protected area. Richness weighted by area was calculated as number of species per square kilometer. A: Mache-Chindul ER; B: Islas Corazón and Fragatas WR; C: Machalilla NP; D: Manglares El Morro WR; E: Manglares El Salado FPR; F: Isla Santay NRA; G: Manglares Churute ER; H: Arenillas ER.

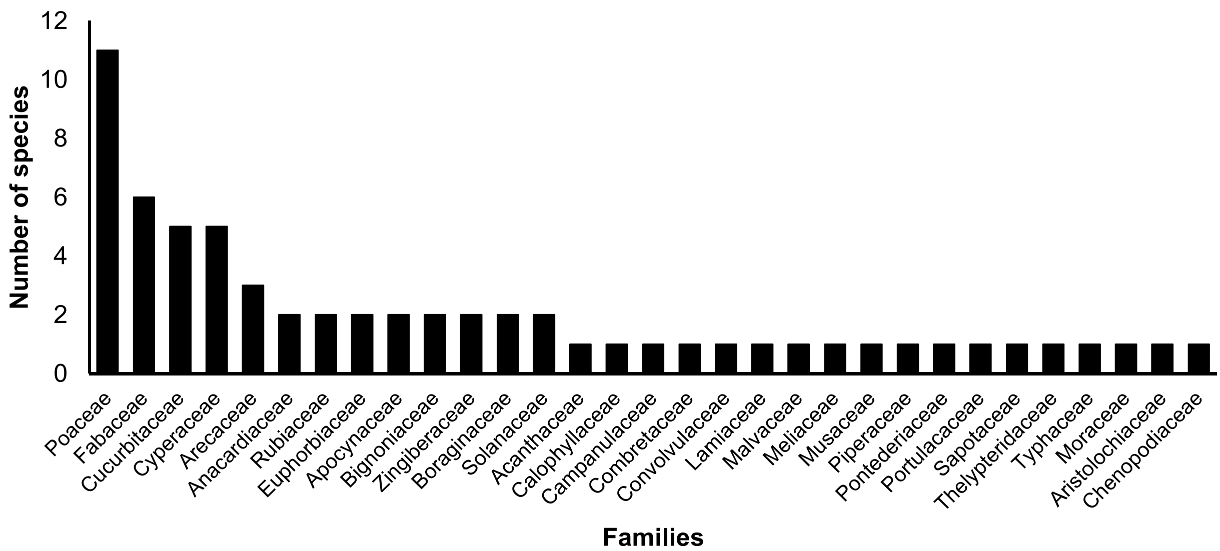

Figure 4. Number of non-native phanerogam species per family.

Prioritization for monitoring

Global analysis - The results of the decision tree revealed that all the PAs had favorable climate conditions for 73 of the species (93.58%; n = 78) (Figure S1). The remaining four species have not been reported as invasive elsewhere, which is why they were categorized as "low priority" and one species was categorized as "more studies required" because it has invasive records but impacts have not been documented (Figure S1). Of the 73 species which matched the Ecuadorean climate, negative impacts have not been reported for 36 growing in similar conditions (i.e., in Colombia; 49.4%; n = 73), so they were categorized as "more studies required" (Figure S1). Of the remaining 37 reported as invasive species, nine appear in the list of 100 worst invasive species in the world, and were consequently categorized as "high priority" (24.3%; n = 37; Figure S1). Of the remaining 28 species, 11 had caused negative impacts in similar climate conditions (i.e., in Colombia), so they were also categorized as "high priority" (39.2%; n = 28; Figure S1). The other 17 species did not cause negative impacts in Colombia so they were categorized as "medium priority" (60.8%; n = 28; Figure S1). The complete prioritization assessment is presented in Table S1.

Assessment by taxa - Sixteen percent of the plant species detected in the PAs assessed were categorized as "high priority" in contrast with more than 78.6% of the vertebrate species in this category (Figure 5A). Some of these species are *Eichhornia crassipes* Solms, *Leucaena leucocephala* (Lam.) de Wit, *M. maximus, C. livia, Felis silvestris catus* (Linnaeus, 1758), *Capra hircus* (Linnaeus, 1758) and *Canis lupus familiaris* (Linnaeus, 1758) (see Table S1). A high percentage (54.7%) of non-native plant species was included in the "more studies required" category (Figure 5A). Out of the total non-native species (n = 78), 46.2% require more studies before a monitoring priority level can be defined (Figure 5A). The prioritization scheme results showed that Isla Santay NRA was the PA with the highest number of species categorized as "high priority" (n = 14), followed by Machalilla NP and Manglares Churute ER with eleven and nine species categorized as "high priority" respectively (Figure 5B). Also, Machalilla NP and Mache-Chindul ER had the highest number of species classified as "medium priority" (n = 9 and n = 5, respectively; Figure 5B). Manglares Churute ER and Mache-Chindul ER were the PAs with the most species classified as "low priority" (Figure 5B). For the category "more studies required," Manglares Churute ER was the PA with the highest number of species (n = 18), followed by Machalilla NP and Isla Santay NRA with eleven and ten species respectively (Figure 5B).

Discussion

In the present study, a method to survey and prioritize non-native species is presented and applied for the first time to PAs in continental Ecuador. Our

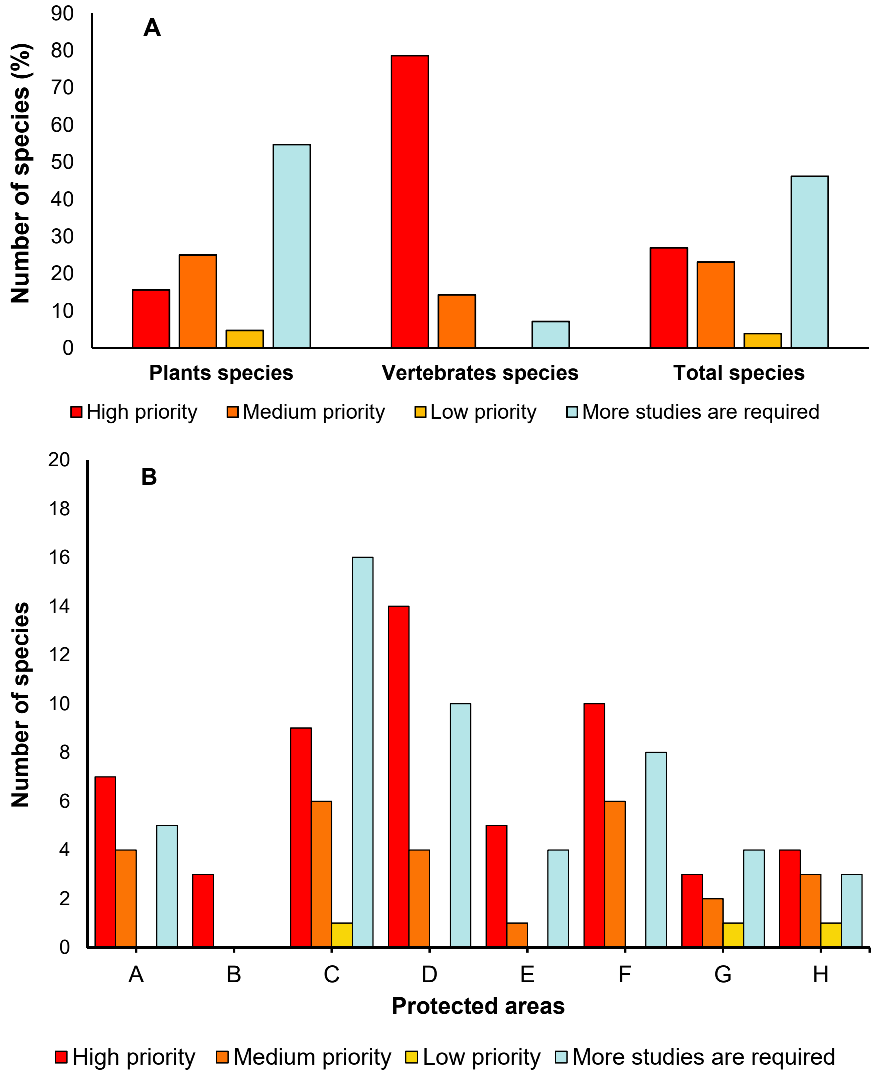

Figure 5. Total priority and uncertainty level (A) and prioritization of non-native species by studied protected areas (B). A: Mache-Chindul ER; B: Islas Corazón and Fragatas WR; C: Machalilla NP; D: Manglares El Morro WR; E: Manglares El Salado FPR; F: Isla Santay NRA; G: Manglares Churute ER; H: Arenillas ER.

study is based on the principle that biotic inventories and prioritization mechanisms for non-native species are necessary to manage biological invasions in ecosystems with high ecological value (Allen et al. 2008; Foxcroft et al. 2017; Gantchoff et al. 2018; Slodowicz et al. 2018). Our results show evidence of the presence of 64 non-native species of phanerogam plants and 14 non-native species of terrestrial vertebrates in eight protected areas assessed on the Ecuadorian coast.

Non-native species in PAs

The inventory of non-native flora indicated that the most common life form is herbaceous, consistent with results obtained from other studies (e.g., Chacón and Saborio 2006; Gantchoff et al. 2018). Nonetheless, Foxcroft et al. (2017) evaluated the life form of 59 non-native plant species in PAs around the world and found that trees were the most representative life form (32%; 19 species), followed by perennial herbs (10 species, 17%) and shrubs (9 species, 15%). In terms of taxonomy, the family with the highest number of species was Poaceae, followed by Fabaceae. A similar pattern of dominance has been repeatedly reported worldwide for protected areas including Denali National Park and Preserve, Wrangell-St. Elias National Park and Preserve, and National Park of American Samoa (Roland 2004; Whistler 1994; Foxcroft et al. 2017). Other studies report Poaceae as the dominant family (e.g., Pyšek 1998; Villarreal et al. 2010). Species of Poaceae have been introduced in several regions of the world due to their importance as food for people and livestock (Chacón and Saborio 2006). Fabaceae have been considered as one of most representative families in the non-native flora around the world (Jiménez et al. 2008), also introduced as an agricultural resource (Myers and Bazely 2003).

At the species level, we identified ten non-native plant species as high priority for management within the PAs studied. Two of these species included in the World's 100 Worst Invasive Alien Species List (Lowe et al. 2004) were found in 25% of the PAs. These species were *E. crassipes* and *L. leucocephala*. Moreover, one of the most frequent non-native plant species in our study area was *M. maximus.* This species has been reported as a high risk invasive in "Risk Analysis and Categorization Proposal of Exotic Species for Colombia" (Baptiste et al. 2010). This plant is a herbaceous African native that has been reported as introduced in 100 countries, and recognized as an invasive species in 49, including Ecuador (other countries include Mexico, Costa Rica, Argentina and Brazil) (Herrera and Cortéz 2009; Ziller and Sá Dechoum 2013; Cabrera et al. 2015; CABI 2019). *Megathyrsus maximus* is considered a successful invader in tropical areas, where it spreads rapidly from seeds and can expand into fields and wastelands (Hitchcock and Chase 1910; Parsons 1972; Savidan et al. 1989). Among aquatic plants, the most frequent non-native species was *E. crassipes* (native to the Amazon basin). It is one of the most invasive aquatic species with demonstrated ecological and socio-economic impacts. Its principal ecological impacts include water quality alteration, reduction of nitrogen and phosphorus pools and a consequent reduction of phytoplankton, invertebrates and ichthyofauna (Villamagna and Murphy 2010). *Leucaena leucocephala*, which is a fast-growing tree native to Mexico and Central America, has proved to be a potential invader in tropical and subtropical ecosystems (Marques et al. 2014). It can modify or affect disturbed habitats by producing toxic chemicals (Kuo 2003), compete with native species (Denslow 2002), and affect species richness (Cronk and Fuller 1995).

Among vertebrates, birds were the most frequent in the PAs studied. At the family level, the Muridae family (with two species: *R. rattus* and *Mus musculus* (Linnaeus, 1758) was the most frequent in the study area, and according to the literature, these species are invasive worldwide (Pereira-Garbero et al. 2013).

The non-native species *C. livia*, *P. domesticus* and *B. ibis* (Class: Aves) were the most frequent at the species level. Similar results have been reported by Liu et al. (2020) in a worldwide PA analysis. These authors showed that *C. livia* and *P. domesticus* were two of the most common animal invaders in PAs around the world. Nevertheless, comparative studies worldwide indicate that the second-highest impact on native organisms by competition is caused by non-native species of the Psittacidaea family (Martin-Albarracin et al. 2015). *Brotogeris versicolurus* (Statius Muller, 1776), a species of psittacid native to the Amazon basin of Peru, Colombia, Brazil and Suriname, is present in four of the studied PAs. The species has been reported as introduced and invasive in Puerto Rico, Florida and California (Falcon and Tremblay 2018). More studies are required to determine population size, distribution and possible impacts caused by *B. versicolurus* in the three coastal PAs where it was detected. Other proven aggressive mammal invaders are present in 25% of the PAs: *F. silvestris catus, C. hircus* and *C. lupus familiaris* (see Clout 2002; Shackleton et al. 2020). For example, Medina et al. (2011) found that *F. silvestris catus* on islands has contributed to 13.9% of global mammal, bird, and reptile extinctions.

Prioritization scheme

The tens rule proposed by Williamson and Fitter (1996) suggests that 1 out of every 100 introduced species will become invasive. This rule was developed on the basis of an examination of plant invaders in the United Kingdom, and seems to be true in plants within a range of variation (around 20%; Caley et al. 2008; Jeschke and Pyšek 2018). We found that 45% of non-native plant species has been reported as invasive in other regions of the world. Following the tens rule proposed by Williamson and Fitter (1996), we proposed that at least 6–13 plant species become invasive in the studied PAs. The tens rule has the limitation that it refers mostly to casual introduction. It does not consider the propagule pressure, climatic matches/mismatches or the role of positive interactions. However, it could be useful for predictive purposes, when similar taxa are considered (Jeschke and Pyšek 2018).

On the other hand, the majority of the plants species (54.7%; 35 species) were classified as "more studies required". This indicates that some studies have established that only a fraction of non-native species naturalize and cause impacts on natural ecosystems (Simberloff 2011), but there is a considerable deficit of information. There are at least 3427 naturalized alien plant species in North America (Qian and Ricklefs 2006), 5789 in Europe (Lambdon et al. 2008) and 4877 in Oceania (Diez et al. 2009), and quantitative assessments of ecological impacts have been undertaken for fewer than 200 alien plants (Hulme et al. 2013), suggesting that the number of plants causing impacts worldwide could be higher.

Jeschke and Strayer (2005) found that the probability of an introduced vertebrate becoming established and invasive exceeds 50%; approximately one of every four introduced species becomes invasive. Lizarralde (2016) supported this pattern, explaining that a cosmopolitan distribution of terrestrial vertebrates improves their establishment and naturalization capabilities, and as a consequence, increases their impact on new ecosystems. Our results were consistent with this pattern; 92.8% of 14 non-native species reported as invasive are found worldwide. Eleven species of vertebrates were classified as "high priority" for management. Invasive mammals alone have been involved in the extinction or endangerment of more than 700 species worldwide (Doherty et al. 2016). The high invasion risk of vertebrates merits attention by environmental decision-makers, because almost all species that are introduced become invasive. Thus, for non-native vertebrate species, prevention measures and control in the early phase of the invasion process must be much stricter.

Although continental Ecuador already had a non-native species inventory list and collected information on the distribution of non-native species (see Herrera et al. 2019), systematic monitoring efforts are lacking. This kind of monitoring can provide an early warning of potential non-native invasive species (Latombe et al. 2017). Moreover, it is important to develop standardized monitoring, examining essential variables: occurrence, non-native species status, invasiveness and impact, to prioritize control and prevention efforts as a key part of effective management (McGeoch et al. 2016; Latombe et al. 2017). The prioritization scheme was designed to identify priority invasive species for monitoring and impact with a simple procedure and minimum information. Such a method may prove a useful tool to keep inventories regularly updated in Ecuadorian protected areas, which is poorly developed. Despite the fact that our prioritization scheme requires little information to categorize priority levels for non-native species monitoring, this scheme did not allow us to categorize more than 54.7% of the plant species, which demonstrates the urgent need to generate basic information on the status of populations of non-native plant species in coastal Ecuadorian PAs.

Vulnerability of PAs

Two of the larger areas (Machalilla NP and Manglares Churute ER and the smallest area (Isla Santay NRA) contained the highest numbers of non-native species. Isla Santay NRA also presented the highest richness of non-native species weighted by area (estimated as richness per km^2) and the highest number of species classified as "high priority." Although this could be attributed to differences in the sampling effort between PAs, Isla Santay NRA does seem to be the PA of highest priority for monitoring of non-native species in the study area. Three characteristics of Isla Santay could contribute to invasibility: i) this PA has the lowest protection level; ii) it is the PA created more recently (2010) and iii) it is immersed in the most populated city (Guayaquil) in continental Ecuador. However, to make an accurate evaluation of overrepresentation and PA invasibility, it would be necessary to reduce the uncertainty level through systematic monitoring of non-native species in the PA system of Ecuador.

Limitations and conclusions

Some limiting factors influenced the number of species registered in the present inventory, such as the scarcity of studies regarding biological invasions in Ecuador, absence of a biological database for native and non-native species inside PAs, variation in the protection budgets of the PAs, variation in the quality of PA management plans and inventories, and misidentification of species. Therefore, a more exhaustive inventory process would increase the number of non-natives recorded inside PAs in the entire country. For this, it is mandatory to plan expeditions to register new non-native species and if possible, to evaluate their impact.

For the present prioritization scheme, we did not gather information in the field. Instead, we conducted the study based on theoretical ideas, occurrence data and published information. It would be desirable to extend this procedure to the whole country, which tends to show a significant increase in the numbers of species at the highest priority level.

Finally, our methodology was limited by the fact that we did not assess introduction pathways to categorize and prioritize species, due to the fact that biological and ecological information on non-native species introductions in continental Ecuador is extremely scarce or nonexistent. The methodology is also limited because it fails to identify species at the level of early detection and rapid response, when eradication is feasible. To address this problem, we suggest that authorities plan exhaustive sampling efforts in customs control at seaports, airports and terrestrial borders to generate estimates of non-native species introductions (propagule pressure). Future studies should focus on the development of basic studies on the invasion ecology of the species we identified as a high priority for management. Once this information is available, the Environmental Impact Classification for Alien Taxa (EICAT; Hawkins et al. 2015) should be developed for these species. We also recommend the development of a curated joint database on invasive alien species that can be constantly updated and synchronized with international biodiversity databases.

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Supplementary material

The following supplementary material is available for this article:

Table S1. Registered non-native species inside the selected PAs and prioritization assessment for each species, showing the list of references where the species records were obtained for each of the protected areas studied.

Table S2. Results of the decision tree to categorize monitoring priority for non-native species.