ENVIRONMENTAL RESEARCH ADVANCES

Invasive Species

Ecology, Impacts, and Potential Uses

Vinícius Londe, PhD
Editor

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INVASIVE SPECIES

ECOLOGY, IMPACTS, AND POTENTIAL USES

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VINÍCIUS LONDE, PHD
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DEDICATION

To my mom, for all her support and faith.

CONTENTS

PREFACE

As is well known, many environmental problems have arisen since the human population began to grow exponentially and dominated the planet. Some examples of environmental problems caused by humans are habitat fragmentation, pollution, global warming, and introduction of exotic species. The last one, coupled with the capacity of some species to rapidly spread and dominate the new environment, is currently one of the main topics investigated in distinct fields of knowledge. However, not all exotic species are invasive. An invasive species is a species that spreads their population and impact the new habitat.

Besides climate change, biological conservation, and ecological restoration, I believe that biological invasion is one of the main themes currently studied. In fact, as the reader will observe in this book, the amount of information on invasive species has grown significantly in the last 20 years. The interest in this theme has increased, not only because invasive species can cause major impacts on ecosystems, but also because it is interesting to understand the dynamics of invasion, the interaction of invasive species with resident species, and how we can manage or use them. All of these aspects inspired me (and my colleagues) to contribute to a new book on invasive species. We worked hard in the last months searching and compiling information on different aspects of invasive species, both animals and plants and from terrestrial and aquatic ecosystems. Surveys carried out on the continent and islands are presented.

I decided to start the book focusing on invasive animals, in particular the New Zealand mud snail (NZMS) and Brazilian rattlesnake. The reader will be informed about the spread of the NZMS in the United States as well as its relation with native species and some functional groups. In Brazil, a rattlesnake species distribution is also spreading due to deforestation. In this case, in addition to ecological consequences, there is also an impact on public health because rattlesnakes are poisonous species and invade urban regions.

The next chapter deals with both invasive animals and plants. The reader will find the advances made in studies on invasive species in the Fernando de Noronha archipelago, Brazil. This archipelago is constituted of 21 islands and islets which are home of many endemic (and threatened) species. Many bird species also use the archipelago as a breeding site. Information about some native species is initially provided. Then, the reader will understand how invasive species are affecting and interacting with native ones. This is an interesting and indispensable opportunity to better understand the relationship between native and invasive species in the light of island ecology.

The following chapters focused on invasive plants. In chapter four, the reader is presented with surveys of exotic species in various fragments of tropical forest inserted in urban matrices. The authors also performed some analyses and discovered which exotic species were causing biological invasion. An interesting issue here is that nearby residents are contributing to propagate invasion.

How does a centenary stand of (exotic and invasive) palm oil tree affects the epiphyte community? In chapter five, the reader will find how an invasive species, brought to Brazil by slaves, affects the epiphyte community configuration. In the next chapter, the reader will discover that the interest on invasive plant species is gradually increasing in recent decades and the main topics investigated have changed over time. The interest in studying invasive plants is also evident when we notice the increasing cooperation between scientists of different nationalities.

On purpose, I decided to end the book by showing the reader that invasive species can be used for distinct objectives. The chemical activation of *Parkinsonia aculeata* wood into activated carbon, for example, has the potential to mitigate water and air pollution. The carbon activation process is described as well as its potential use for removing contaminants from water and air. Phytoremediation of heavy metals is another applicability of invasive plants. In Mexico alone, about half of the invasive species have the potential to accumulate heavy metals from the soil.

Many studies on invasive species were published in recent years but this theme is so broad and complex that more research is still needed. Moreover, new demands and challenges emerge as knowledge is accumulated. This book aims to contribute to the ecology, impacts, and uses of invasive species. I hope the reader enjoys reading it.

Vinícius Londe
Campinas, Brazil
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Chapter 1

AN INVESTIGATION OF INVASION: THE INFLUENCE OF NEW ZEALAND SNAILS IN FRESHWATER AND BRACKISH WATER ECOSYSTEMS

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ABSTRACT

The New Zealand mud snail (*Potamopyrgus antipodarum*; NZMS) is an invasive species present in a variety of ecosystems including brackish estuaries, freshwater rivers, and freshwater lakes. NZMS are of concern because once established they may out-compete native invertebrates, but themselves provide little nutritional value to predators. The objectives of this study were to: 1) determine how populations of NZMS vary across space and time; 2) investigate the relationship between NZMS, benthic macroinvertebrate diversity, and abundance of functional feeding groups (herbivores, detritivores, omnivores, predators), and 3) explore the influence of NZMS on benthic food web structure. Samples of benthic lake, river, and estuarine food webs and NZMS populations were collected between 2006 to 2013 in the state of Oregon, USA. The effects of NZMS on macroinvertebrate diets were assessed with stable isotope analysis of carbon and nitrogen. Stable isotopes were used to estimate trophic ranges of both NZMS and native macroinvertebrates. NZMS densities were found to be dynamic, with population densities fluctuating over time and between locations. NZMS were most abundant in estuaries followed by lakes and rivers. There was a weak negative correlation between NZMS densities and native macroinvertebrate diversity across all systems. The densities of specific functional feeding groups had varying positive (omnivores) and negative (herbivores) correlations with NZMS densities. However, stable isotope analyses indicated that NZMS are likely not causing significant changes in the diets of native macroinvertebrates and thus are not competing for the same food source. Our results suggest that the effects of NZMS on functional feeding groups and food webs may be subtle and dependent upon the composition of the recipient community, necessitating further study across diverse ecosystem types.

Keywords: aquatic invasive species, estuary, functional feeding groups, lakes, macroinvertebrates, New Zealand mud snail, Oregon, rivers, trophic range

INTRODUCTION

Invasive species can compete with native taxa and drive ecological changes such as niche displacement or even extinction (Mooney and Cleland 2001, 5447; Schreiber et al. 2002, 317; Carlsson et al. 2004, 1575).

These ecological changes may include alterations in the distribution, abundance, and diet of native species (Byers 2000, 1225; Carlsson et al. 2004, 1576; Brenneis et al. 2010, 169; Moore et al. 2012, 2435). However, many studies have also shown invasive species can facilitate native species by providing food, shelter, and in some instances, predator release (Schreiber et al. 2002, 317; Beekey et al. 2004, 535; Rodriguez 2006, 927). The establishment and influence of invasive species on communities is highly variable and dependent on numerous factors, including the traits of the invader, the vulnerability of the native community, and properties of the ecosystem (Leung and Mandrak 2007, 2603; Lockwood et al. 2007, 107).

The New Zealand mud snail (NZMS; *Potamopyrgus antipodarum*) is an example of an invasive species whose traits have facilitated a broad expansion. NZMS was first discovered in the United States in the Snake River, Idaho in 1987 (Bowler 1991, 174). Within the first ten years, they colonized more than 640 km of the Snake River and spread across the North American continental divide (Zaranko et al. 1997, 809). More recently, NZMS has spread to over 1600 unique locations in nearly 400 waterbodies across North America (USGS 2019). NZMS can spread actively upstream up to 1 $km \cdot yr^{-1}$ (Loo et al. 2007, 181) and float passively downstream independently or attached to vegetation (Kerans et al. 2005, 123). Additionally, they can be transported by fish species (passing live through the digestive tract of fish; Brenneis et al. 2011) or humans, typically via recreational boating and fishing gear (Alonso and Castro-Díez 2008, 107).

Environmental conditions and interactions with native species can also influence the establishment of invasive species. NZMS are tolerant of a wide range of abiotic conditions (Jacobsen and Forbes 1997, 260; Zaranko et al. 1997, 809; Dybdahl and Kane 2005, 1592). Their densities do not show evidence of declining until temperatures exceed 28°C, and they can reproduce at temperatures > 24°C if food or space is not limiting (Dybdahl and Kane 2005, 1594). Furthermore, the presence of an operculum allows NZMS to resist desiccation, which is beneficial for overland dispersal, but the operculum also allows the typically freshwater NZMS to tolerate and even thrive in brackish environments with salinity levels as high as 15 psu (Alonso and Castro-Díez 2008, 107). Densities have typically been greatest in systems with high primary productivity, constant temperatures, and constant flow rates, with peak densities in the summer (Richards et al. 2001, 375).

NZMS can impact benthic aquatic food webs through multiple pathways, including altering the distributions, abundances, and diets of native species. NZMS is an invasive grazing herbivore and detritivore (i.e., generalist feeder) that utilizes the same food sources as many native macroinvertebrates, and can consume large volumes of algae and detritus (Zaranko et al. 1997, 809; Kerans et al. 2005, 123; Hall et al. 2006, 1121). Managers are concerned about the spread and proliferation of NZMS because once established, they may out-compete native grazers that provide an important food source for salmon and trout species. Although snails are generally important links between primary producers and higher order consumers in aquatic ecosystems (Hawkins and Furnish 1987, 209), NZMS provide little nutritional value to upper trophic levels (McCarter 1986, 551; Hall et al. 2006, 1121). NZMS can reach high densities (>290,000 individuals·m^{-2}; Kerans et al. 2005), which may reduce niche space for native competitors. For instance, benthic invertebrates competing with a high density NZMS population changed their diets from their preferred food source of periphyton to detritus as a result of competitive exclusion, switching from a specialized feeding habit to a more generalized feeding habit (Moore et al. 2012; 2435). However, Brenneis et al. (2010, 177) concluded that even at high densities there were no obvious negative impacts of the NZMS on the densities of native competitors in an estuarine system. A more thorough investigation on the invasion of NZMS on these varying ecosystems is necessary to gain a better understanding of their influence on benthic food webs.

Our overall objective in this chapter was to examine NZMS population densities over time, across different aquatic habitats, and relationships with native macroinvertebrates. Separate studies have examined NZMS population dynamics and their influence on food webs in different freshwater and brackish ecosystems (Hall et al. 2006, 1123; Bersine et al.

2008, 1381; Brenneis et al. 2011, 169; Moore et al. 2012, 2436); however, there exist few comparative studies across multiple ecosystem types (i.e., rivers, lakes, and estuaries). Critically, NZMS may have unique impacts and respond differently to environmental variables in different ecosystems. Further, studying NZMS across these diverse aquatic ecosystems may also broaden the known gradients of densities and environmental conditions, allowing observation of effects that may only be noticeable at the extremes. For instance, Moore et al. (2012) only observed effects of NZMS on native macroinvertebrates when NZMS were abundant. Our specific questions include: (1) How does the density of NZMS vary over time and across space in estuaries, rivers, and lakes? (2) How do NZMS relate to macroinvertebrate community diversity and density of functional feeding groups (i.e., herbivores, omnivores)? (3) Do NZMS alter benthic food webs?

We hypothesized the following: (1) NZMS densities will be highest in systems controlled more by physical and/or chemical disturbance (i.e., rivers, estuaries) compared to systems that are more stable (i.e., lakes) given the ability of NZMS to survive and even thrive in harsh and variable abiotic conditions (e.g., Dybdahl and Kane 2005). Additionally, we predict that NZMS densities will be most temporally stable in estuaries compared to rivers and lakes because of higher ecosystem productivity in estuaries (Dame 2008, 1407). (2) NZMS densities will have a negative relationship with benthic invertebrate diversity, as well as the density of competing functional feeding groups. Systems with available niche space (potentially due to disturbance or harsh environmental conditions) are favorable for invaders compared to less disturbed or more stable systems with limited available niche space (Paavola et al. 2005) and NZMS will out-compete native invertebrates of the same feeding niche. (3) NZMS will fill specialized, narrower trophic ranges (i.e., detritivore or herbivore) when at high densities compared to low densities by outcompeting native invertebrates, thus creating an Eltonian niche shift, i.e., a shift in its functional role (Elton 1927, 1-191; Soberón 2007, 1115; Larson et al. 2010, 2). Consequently, NZMS will also force native competitors into more generalized diets, especially at high densities (e.g., Moore et al. 2012, 2435).

METHODS

Study Sites

To address our objectives, we sampled four rivers, six estuaries, and six lakes between 2006 and 2013 (Figure 1, Table 1). All four coastal freshwater lakes in Oregon with reported NZMS sightings (as of summer 2013; USGS 2019) were selected for this study. Similarly, four brackish estuaries with reported NZMS sightings were sampled in the summer of 2013, one of which had been previously sampled at irregular intervals since 2006. In addition, two freshwater coastal lakes and two estuaries without known NZMS presence were sampled as reference systems (Figure 1). Four rivers were sampled at an upstream and downstream location to capture natural variability; two of the rivers had documented NZMS populations in 2006, and two of the rivers did not have NZMS in 2006. All sites are located primarily in forested areas with moderate levels of agriculture, with the exception of the Deschutes and John Day Rivers, which are located in the arid Columbia Plateau.

At each site, locations were selected adjacent to boat access ramps and within each site, 4–6 sub-sites were sampled, depending on accessibility. When diverse habitats were present (e.g., macrophyte stands, bedrock, cobble,) we sampled across all habitat types to incorporate the maximum amount of diversity present at each site (Table 1). River sub-sites were characterized by silt, cobble, or gravel (Columbia Plateau) or tidal mudflats/sandbars (coastal rivers). Estuarine sub-sites were generally characterized by exposed mudflats or rocky shoreline, with samples collected in least one pelagic sub-site. Lake sub-sites were largely sandy or muddy littoral habitats with one pelagic sub-site.

Table 1. Sampling design for New Zealand mud snails (NZMS) and benthic invertebrate surveys from 2006 to 2013. Number of samples taken in each year is indicated in parentheses

Ecosystem	Ecoregion	Site	Sample Years	Sample Size & Site Description	Location	Year: NZMS?
River	Coastal Rivers	Siuslaw US*	2006, 2012-2013	Rocky substrate (n = 4)	44.031N - 123.858W	2006: No 2013: No
		Siuslaw DS	2006, 2012-2013	Tidally influenced, mudflats (4)	43.967N - 124.102W	2006: No 2013: Yes
		Umpqua US	2006-2008 2012-2013	Rocky substrate, bedrock (4)	43.650N - 123.839W	2006: No 2013: Yes
		Umpqua DS	2006-2013	Tidally influenced, silt/sand (4)	43.710N - 124.095W	2006: Yes 2013: Yes
	Columbia Plateau Rivers	John Day US*	2006-2012	Rocky and gravel substrate (4)	44.916N - 120.470W	2006: No 2012: No
		John Day DS*	2006-2012	Rocky and gravel substrate (4)	45.477N - 120.469W	2006: No 2012: No
		Deschutes US	2006-2012	Gravel and bedrock (5)	45.388N - 120.871W	2006: Yes 2012: Yes
		Deschutes DS	2006-2012	Silt and bedrock (4)	45.633N - 120.913W	2006: Yes 2012: Yes
Estuary	Brackish Estuaries	Columbia: Youngs Bay	2006-2013	Mudflats (3), Rocky shore (2), Pelagic (1)	46.170N - 123.834W	2006: Yes 2012: Yes 2013: Yes
		Tillamook	2013	Mudflats (2), Rocky (3), Pelagic (1)	45.472N - 123.891W	2013: Yes
		Nestucca	2013	Mudflats/Organic (3), Rocky (2), Pelagic (1)	45.207N - 123.961W	2013: Yes
		Yaquina	2013	Mudflats/Organic (5), Pelagic (1)	44.591N - 123.943W	2013: Yes
		Coquille*	2013	Mudflats/Rocky (3), Rocky (2), Pelagic (1)	43.148N - 124.401W	2013: No
		Rogue	2013	Mudflats/Rock (2), Sandy (3), Pelagic (1)	42.420N - 124.423W	2013: Yes
Lake	Coastal Lakes	Coffenbury	2013	Littoral (5): Muddy/ organic; Pelagic (1)	46.173N - 123.963W	2013: Yes
		Cullaby*	2013	Littoral (5): Sandy/ silty; Pelagic (1)	46.087N - 123.906W	2013: No
		Lytle	2013	Littoral (5): Muddy/ organic; Pelagic (1)	45.624N - 123.940W	2013: Yes
		Devils	2013	Littoral (5): Sandy/rock; Pelagic (1)	44.979N - 123.991W	2013: Yes
		Mercer*	2013	Littoral (5): Organic/ sandy; Pelagic (1)	44.048N - 124.075W	2013: No
		Garrison	2013	Littoral (5): Organic/ sandy; Pelagic (1)	42.754N - 124.506W	2013: Yes

Legend: * = reference site, US = upstream, DS = downstream

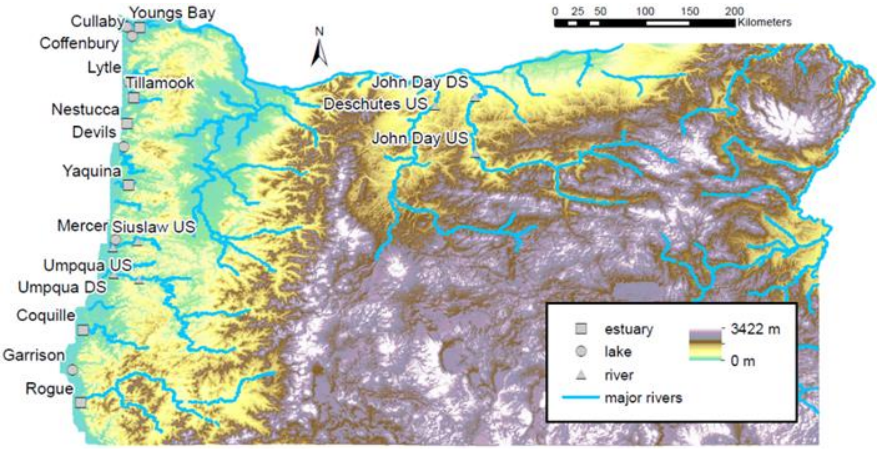

Figure 1. An elevation map of Oregon, USA depicting the New Zealand mud snail sampling locations and the type of waterbody for each location. US = upstream, DS = downstream.

Sampling Methods

The quantitative sampling method used for the lake and river sites was modified from the Oregon Department of Environmental Quality protocols for wadeable streams (ODEQ 2004a). Benthic invertebrates were sampled in lake and riverine sub-sites by disturbing a randomly selected fixed area (0.74 m^2) of substrate with a D-net (250-µm mesh size). When aquatic vegetation was present, the D-net was swept over the grid area several times. The spacing between grids within a sub-site ranged between 5 and 20 m. Sampling of the exposed mudflats and rocky shoreline of the estuarine sites was performed using a circular PVC ring (0.07 m^2 in area, 30.5-cm diameter, depth of 2 cm). The sampling ring was placed at different sub-sites at low tide (0.37 m^2 total area sampled), incorporating cobble and mud substrate at each estuary. The distance between sampling sub-sites was determined using a random number generator between 1 and 10 and pacing out the distance of the number generated.

Any large rocks present were scraped and cleaned for invertebrates and the entire sub-site sample was pooled. All material collected was condensed with a 250-µm sieve and preserved at a final concentration of 70% ethanol. Samples for stable isotope analyses (described below) were collected using the same methods as the quantitative sampling methods described above with the addition of using an Ekman grab to collect a benthic profundal sample for stable isotope analysis at each lake and estuary.

Samples of aquatic vegetation from the lake, river, and estuary sites were collected from each sub-site, placed in Ziploc bags, and frozen for subsequent stable isotope analysis. Periphyton and phytoplankton samples were also collected for stable isotope analysis at each sub-site by brushing periphyton off rocks and other smooth surfaces and using a 3.2-L Van Dorn sampler to collect phytoplankton from the water column. Phytoplankton and the periphyton slurry were filtered on to a pre-combusted Whatman glass microfiber filter (0.7-µm pore size; Whatman International Ltd., Maidstone, England) using a BD 60-mL syringe (BD, Franklin Lakes, NJ). Periphyton and phytoplankton filters were wrapped in tinfoil and flash frozen using dry ice. Zooplankton were sampled at the pelagic site with horizontal tows of an 80-µm plankton net (30-cm diameter) and preserved at a final concentration of 70% ethanol. Baited minnow traps were used at each sub-site to catch secondary invertebrate consumers like crayfish, which were then preserved at a final concentration of 70% ethanol. In the estuaries, minnow traps were used during higher tide and at depths that were still submerged during low tide for collection of secondary consumers.

In situ measurements of abiotic conditions were taken using a YSI ProPlus (Yellow Springs, OH), including measurements at 1-m intervals for specific conductance ($\mu S \cdot cm^{-1}$), dissolved oxygen ($mg \cdot L^{-1}$), pH, and temperature (°C) in lakes, and surface measurements in rivers and estuaries (Table 2). Research was conducted on NOAA Scientific Permit #17879.

Table 2. The median, minimum, maximum, and standard deviation of abiotic variables in the freshwater coastal lakes (n = 6), freshwater rivers (n = 8), and brackish estuaries (n = 6) sampled through the extent of the research

Variable	**Measure**	**Rivers**	**Lakes**	**Estuaries**
Temperature (°C)	Median	13.5	21.6	19.1
	Min	12.2	19.7	17.3
	Max	24.2	22.6	22.2
	Standard Deviation	4.1	1.0	1.5
Dissolved Oxygen (mg·L^{-1})	Median	10.70	8.14	8.21
	Min	7.75	7.54	7.01
	Max	11.40	8.63	9.03
	Standard Deviation	1.30	0.41	0.73
Specific Conductance (µS·cm^{-1})	Median	131.7	119.5	6900.0
	Min	53.5	74.5	2710.0
	Max	449.9	202.0	19650.0
	Standard Deviation	127.1	40.5	6824.0
pH	Median	7.92	7.68	7.57
	Min	7.08	6.70	6.81
	Max	8.40	8.25	8.37
	Standard Deviation	0.46	0.48	0.51

Sample Processing

For quantitative macroinvertebrate samples, the Caton subsampling apparatus was used, which consists of a standardized gridded screen (370-µm opening) and a tray (Caton 1991, 317). Each sample from a sub-site was emptied and washed onto the Caton gridded screen, spreading the sample contents evenly (Blackwood 2007). Macroinvertebrates ≥ 1.3 cm in length that occurred in four or fewer grids were noted and included in the subsample (Blackwood 2007). At least 12.5% of the grids were chosen randomly for subsampling, with the goal to count ≥ 300 and < 500 organisms; if less than 300 specimens were counted, another grid was randomly selected for sampling (Blackwood 2007).

A Leica MI65C microscope and IC80HD camera (Leica Microsystems Inc., Buffalo Grove, IL) was used to identify organisms. Macroinvertebrates were identified to family or genus. Alpha diversity was calculated for the lowest taxonomic designation of macroinvertebrates with the Shannon-Wiener Index. Macroinvertebrates were grouped into functional feeding groups: herbivores, detritivores, omnivores, predators, collector-filterers, and collector-gatherers. The collector-filterer and collector-gatherer categories were used for isotopic baselines (described below). Although there can be variation in feeding modes within families, this was the most practical approach and likely indicates that our results are conservative. Macrophytes and mosses were also categorized into groups primarily by how they photosynthesize (C_3 plants, C_4 plants, CAM plants) and location of photosynthesis (pelagic, littoral, and terrestrial leaf litter). Macroinvertebrate functional feeding groups and primary producer groups were determined using various published keys and reports (references in Cimino 2016).

Stable carbon (C) and nitrogen (N) isotope ratios can be used to provide a better understanding of the effects of invasive species on aquatic food webs (Vander Zanden et al. 1999, 464). The stable carbon isotope ratio ($\delta^{13}C$) can be useful in determining the primary production source responsible for energy flow in the ecosystem, and the stable nitrogen isotope ratio ($\delta^{15}N$) can be used to estimate trophic position (Fry 2006, 40-75). Stable isotope of nitrogen and carbon were analyzed on preserved invertebrate, macrophyte, algal, and terrestrial leaf litter samples. Collected samples were dried at 60°C for 24 to 48 hours until a constant dry weight was achieved and homogenized with a mortar and pestle. A total of 1 mg (± 0.2 SD) of dry weight for individual invertebrate samples and a total of 3–5 mg of dry weight for individual primary producer samples were enclosed in tin capsules, and kept dry in a desiccator (UC Davis Stable Isotope Facility 2013). All samples were then analyzed for $\delta^{15}N$ and $\delta^{13}C$ using a PDZ Europa ANCA-GSL elemental analyzer interfaced to a PDZ Europa 20-20 isotope ratio mass spectrometer (Sercon Ltd., Cheshire, UK) at the UC Davis Stable Isotope Facility. The facility expresses measuring error as the long-term standard deviation of 0.2‰ $\delta^{13}C$ and 0.3‰ $\delta^{15}N$. All benthic invertebrate samples were preserved in 70% ethanol and this preservation process can alter isotopic signatures. To correct for alteration of isotopic signatures, a constant adjustment factor was used, subtracting 0.39‰ from $\delta^{15}N$ and 1.18‰ from $\delta^{13}C$ (Ventura and Jeppesen 2009, 297).

An isotopic baseline was used to estimate trophic position within each ecosystem following Post (2002, 703). Organisms used as baselines for most sites were bivalve collector-filterers, which are typically long-lived primary consumers with tissue that is not very sensitive to short-term seasonal fluctuations in nutrients (Cabana and Rasmussen 1996, 1575; Post 2002, 703). For sites without collector-filterers, other collector-gatherer taxa were used as the baseline organisms. In order to examine interactions between NZMS and native macroinvertebrates, we determined the amount of overlap in the trophic ranges of functional feeding groups (i.e., herbivores, detritivores, omnivores, predators) and NZMS derived from stable isotopes. Convex hulls can be used to represent the total extent of a feeding group's trait space or niche space (Cornwell et al. 2006, 1465; Layman et al. 2007, 42). Convex hulls were created for each feeding group depicting the trophic range of individuals from each sampling location (≥ 3 individuals were required). Separate convex hulls of NZMS trophic range were also created for each site, which were overlaid with feeding group convex hulls (Cimino 2016) to measure % overlap between the trophic ranges. The area of the hull was measured using the program ImageJ.

Statistics

To maximize power, all sites were used to help determine if the presence and density of NZMS influenced diversity. Thus, both upstream and downstream sites of river systems were treated as independent replicates based on spatial distance of sampling sites, elevation differences, and substrate differences (Cimino 2016). Samples taken from 2006–2008 and from 2012–2013 (averaged within time period) were also treated as independent based on the four-year time gap between sampling events. This approach better captures how variable NZMS densities may relate to native macroinvertebrate communities. Lastly, to aid in comparisons between different habitats, the densities of both NZMS and native macroinvertebrates were scaled to the maximum value within each habitat. For example, the lake with the highest NZMS densities, Devils Lake (1612 individuals·m^{-2}), was given a value of 1.0, whereas Lytle Lake (474 individuals·m^{-2}) was given a value of 0.29. This process was repeated for estuarine and river sites.

A one-way ANOVA with Tukey HSD post-hoc test was conducted to test for differences between densities of NZMS between different systems (lake, river, and estuary). Sites with no recorded NZMS were not retained for this analysis. Densities were transformed ($\log_{10}(x) +1$) to achieve normality. The assumption of normality was examined using the Shapiro-Wilk test and histograms and boxplots of densities.

The relationship between NZMS density and alpha diversity (Shannon-Weiner index), as well as NZMS density and the density of specific functional feeding groups (herbivores, detritivores, omnivores, and predators) was examined with a Pearson's correlation for all sites and sampling dates. Herbivores, detritivores, and omnivores were deemed to be the most likely to compete for food sources with NZMS (Zaranko et al. 1997, 809; Kerans et al. 2005, 123) and predators were included because invasive species have been observed to facilitate predators (Rodriguez 2006, 927).

The percent overlap between NZMS trophic range and native macroinvertebrate functional feeding groups were related to NZMS densities at the same sampling location with Pearson's correlation for each feeding group. The % overlap could only be calculated for sites with sufficient number of individuals (≥ 3) within a functional feeding group to construct a convex hull; therefore, sample size was reduced compared to other analyses. Lastly, a one-way ANOVA with Tukey HSD post hoc test was conducted to test for significant differences in the amount of trophic overlap shared between NZMS and the different functional feeding groups. All analyses were performed in R version 3.0.1 (R Development Core Team 2013).

RESULTS

Objective 1: NZMS Temporal and Spatial Dynamics

This study revealed some dramatic changes in NZMS densities over time as well as new detections at previously undetected sites. Notably, NZMS were detected for the first time in the lower Siuslaw River at the Tiernan boat ramp in summer 2012 (Figure 2) and in the Nestucca River estuary in summer 2013.

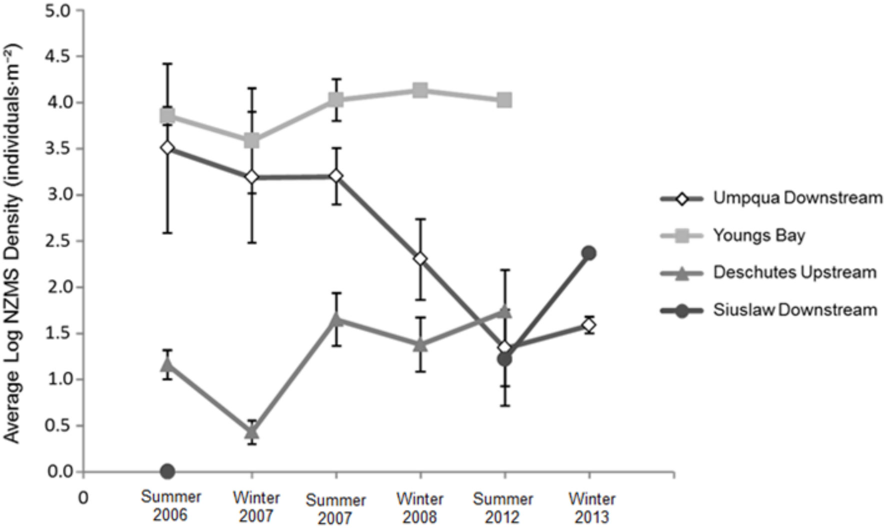

Figure 2. The average $\log_{10}$ New Zealand mud snail (NZMS) density (individuals·m^{-2}) at each of the riverine sites tested from 2006–2013 (±1 standard error). Only the Umpqua downstream and Siuslaw downstream river sites were sampled in the winter of 2013. Deschutes downstream, Umpqua upstream, Siuslaw upstream, and the John Day River upstream and downstream were not included in this graph due to absence or near absence of NZMS at every sampling period.

Conversely, the densities of NZMS at the Umpqua River downstream sites appeared to have declined by over two orders of magnitude during the seven-year sampling period from ~3,200 NZMS·m^{-2} to ~30 NZMS·m^{-2} (Figure 2). The Deschutes River upstream sites had consistently low

NZMS densities with a slight upward trend over the study period (Figure 2). NZMS densities remained high in estuarine Youngs Bay throughout the sampling period, with values averaging around 10,000 NZMS·m^{-2} (Figure 2). There were no or few detections of NZMS at the upstream and downstream reaches of the John Day River, Siuslaw River upstream, Umpqua River upstream, and the Deschutes River downstream (average < 2 NZMS·m^{-2}) sites during every sampling period (see Cimino 2016 for densities).

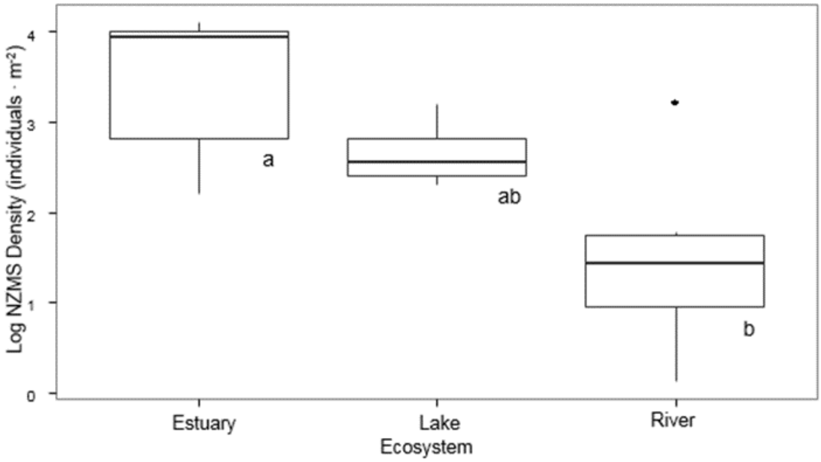

Figure 3. New Zealand mud snail (NZMS) density ($\log_{10}$; individuals·m^{-2}) in estuaries (n = 7), lakes (n = 4), and rivers (n = 7) including all samples and all sampling periods. Only sites with NZMS present were included in the boxplots. The center line of each box represents the median, the lower reach of the box represents the lower quartile, and the upper reach of the box represents the upper quartile. The whiskers of the boxplots represent the 5th and 95th percentile range, with the black symbol representing an outlier). Letters represent the results of Tukey HSD post hoc tests, where different letters are significantly different from each other ($p < 0.05$).

NZMS densities varied greatly across lakes, rivers, and estuaries: estuaries had the greatest densities of NZMS followed by lakes, and then rivers (Figure 3). There was a significant difference in NZMS densities between ecosystems ($F_{2,15} = 9.41$, $p = 0.002$): a Tukey HSD test determined that the river and estuary sites were significantly different ($p = 0.002$), but there were not differences between lake and estuary sites (*p* = 0.342) or river and lake sites (*p* = 0.098).

Objective 2: Relationships between NZMS and Native Macroinvertebrates

Within each system (lakes, rivers, or estuaries), there was no significant correlation between NZMS densities and macroinvertebrate diversity in all sites, including reference sites (lakes *r* = -0.422, *p* = 0.404, *n* = 6; estuaries *r* = -0.396, *p* = 0.379, *n* = 7; rivers *r* = -0.273, *p* = 0.307, *n* = 16; Figure 4). However, across all systems, there was a significant negative correlation between NZMS density and the diversity of the benthic invertebrate community (*r* = -0.476, *p* = 0.009, *n* = 29; Figure 4).

There was a modest negative correlation between NZMS densities and the densities of herbivores across all sites, including reference sites (*r* = -0.366, *p* = 0.051, *n* = 29) (Figure 5a). Conversely, there was a significant positive correlation between the densities of NZMS and omnivores (*r* = 0.393, *p* = 0.035, *n* = 29; Figure 5d). Detritivore (*r* = 0.205, *p* = 0.286, *n* = 29) and predator (*r* = 0.279, *p* = 0.144, *n* = 29) densities were not correlated with NZMS densities (Figure 5b, c).

Objective 3: Effect of NZMS on Native Food Webs

The degree of feeding similarity between NZMS and its competitors was evaluated using the amount of overlap between trophic niches as observed with stable isotope analyses. There was no difference in feeding similarities between NZMS and any functional feeding groups ($F_{3,26}$ = 1.711, *p* = 0.182; Figure 6).

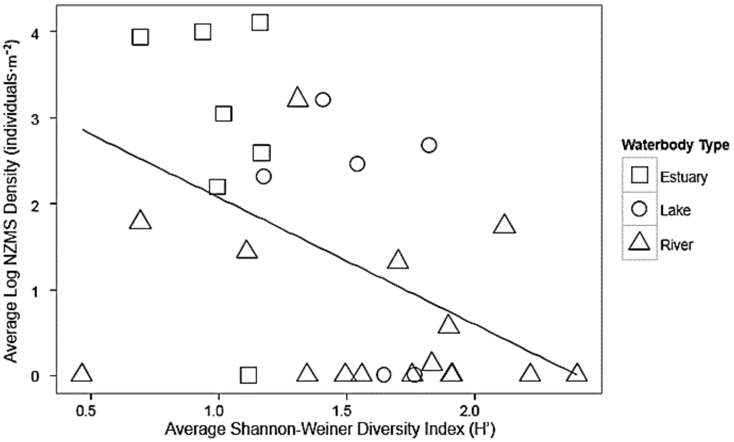

Figure 4. Average $\log_{10}$-transformed New Zealand mud snail (NZMS) density (individuals·m^{-2}) and average Shannon-Weiner diversity index (H'). Rivers (n=16) are represented by triangles, estuaries (n = 7) are represented by squares, and lakes (n=6) are represented by circles. All sites were included in this analysis.

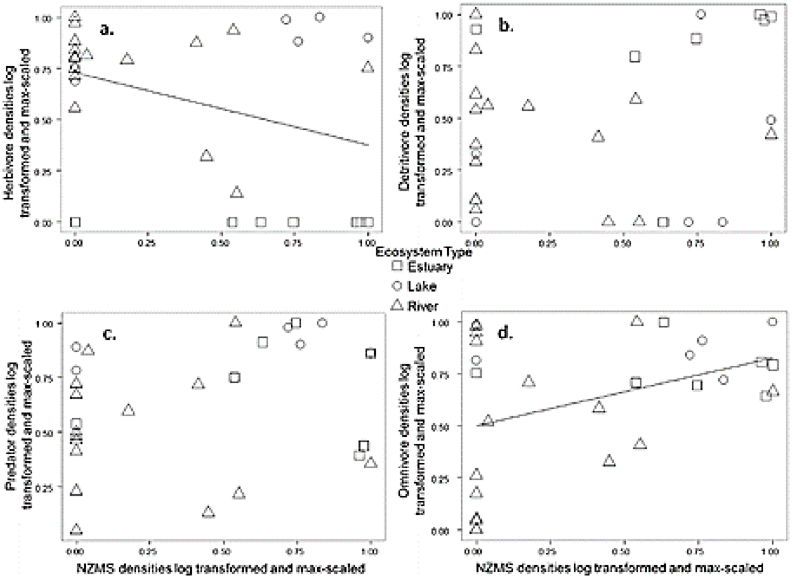

Figure 5. New Zealand mud snail (NZMS) densities compared to (a) herbivore, (b) detritivore, (c) predator, and (d) omnivore densities (all $\log_{10}$-transformed and scaled to the maximum value within each ecosystem (max-scaled)). All sites were included in this analysis. Sample size as in Figure 4.

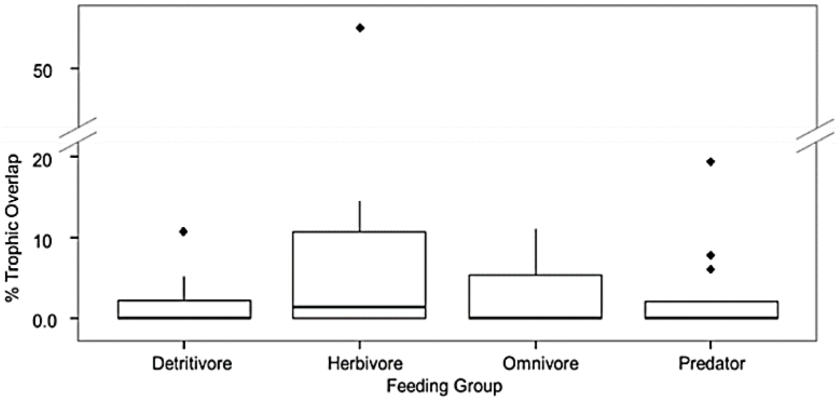

Figure 6. The percentage of trophic overlap between the trophic range of New Zealand mud snail and functional feeding groups. There were no significant differences between groups and how much trophic overlap they had with New Zealand mud snails.

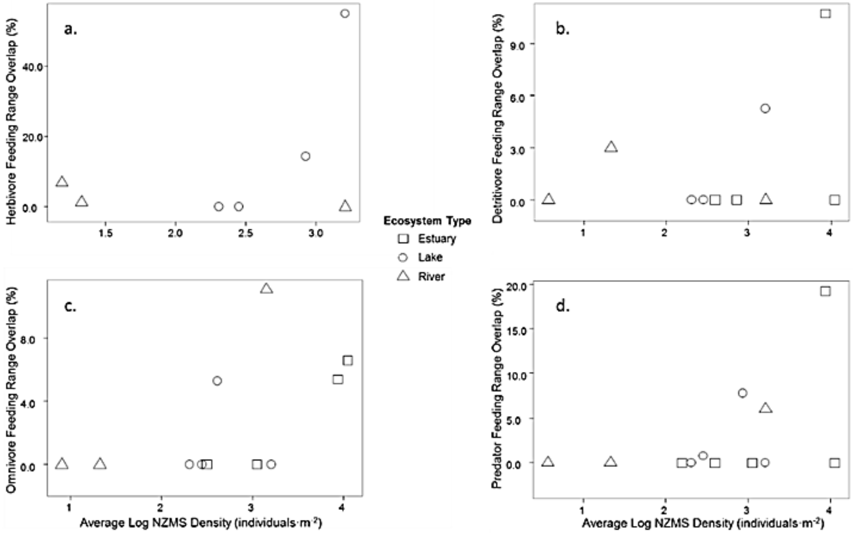

Figure 7. Correlations between trophic overlap (% of NZMS range overlapping other feeding group's range) of (a) herbivores, (b) detritivores, (c) omnivores, and (d) predators and log-transformed New Zealand mud snails (NZMS) densities (individuals·m^{-2}). Ecosystem symbols as in Figure 4. Note that % overlap could only be calculated for sites with sufficient number of individuals within a functional feeding group to construct a convex hull; therefore, sample size was reduced compared to other analyses (i.e., Figure 5). No herbivores were sampled from the estuaries.

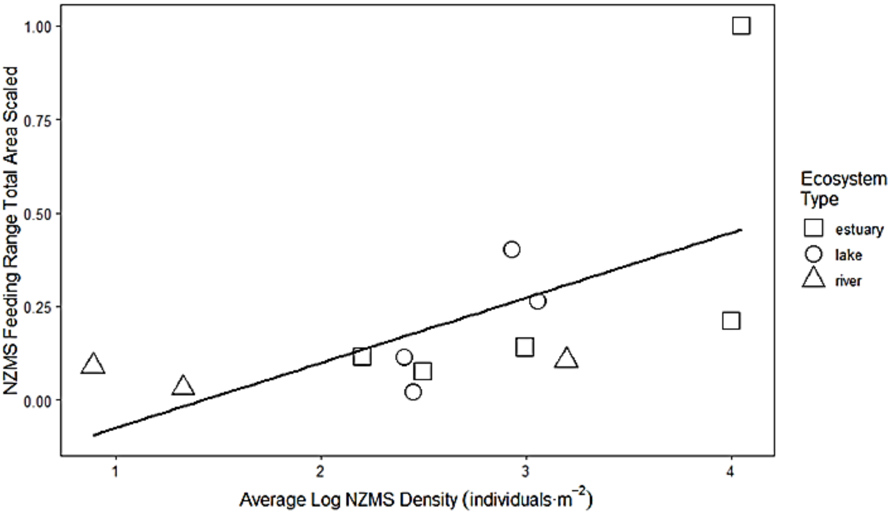

Figure 8. Correlation between trophic range (total area of convex hulls, max-scaled) of New Zealand mud snails (NZMS) and $\log_{10}$-transformed NZMS densities (individuals$\cdot$m^{-2}). Each point represents an individual convex hull area from a sampling site. Ecosystem symbols as in Figure 4. Note that feeding niche was only calculated for sites appearing in Figure 7, i.e., with sufficient number of individuals to construct a convex hull.

Additionally, though there was a tendency for high density NZMS sites to show a greater degree of trophic overlap between NZMS and functional feeding groups, none of the correlations were significant (herbivores $r = 0.438$, $p = 0.326$, $n = 7$; detritivores $r = 0.366$, $p = 0.298$, $n = 10$; omnivores $r = -0.552$, $p = 0.078$, $n = 11$; predators $r = -0.472$, $p = 0.121$, $n = 12$; Figure 7). There was a significant positive correlation between the size of the NZMS trophic range and NZMS density ($r = 0.604$, $p = 0.037$, $n = 12$; Figure 8) suggesting that as NZMS densities increased their trophic range also increased.

DISCUSSION

New Zealand mud snails are unique in their ability to successfully establish in a variety of very different aquatic environments, and range widely in the densities at which they occur. Our study is unique in that it observes NZMS populations and their influence on native food webs simultaneously across multiple ecosystem types (rivers, lakes, and estuaries), in contrast to previous studies that examined NZMS population dynamics and their influence on specific food webs in either freshwater or brackish ecosystems (Hall et al. 2006, 1121; Bersine et al. 2008, 1381; Moore et al. 2012, 2435). As expected, we observed varying densities of NZMS across all ecosystems, though estuarine Youngs Bay appeared to be the most stable over time (Figure 2). Additionally, we observed different patterns in densities of functional feeding groups across gradients of NZMS densities, which suggest that the effects of NZMS on invaded communities may be subtle. Below, we discuss some of the mechanisms that could have driven changes in NZMS densities as well as their influence on benthic communities, as well as implications for NZMS ecology and management.

NZMS Temporal and Spatial Dynamics

Although the three ecosystem types surveyed are quite distinct from each other, NZMS densities were only significantly different in estuaries compared to rivers, with estuaries having ~ 92.5% larger population sizes than rivers, on average (Figure 3). We had hypothesized that estuaries and rivers would have the highest densities because these systems are driven by disturbance and harsh conditions. Although sites with higher specific conductance (mainly estuaries) tended to have higher NZMS densities, there was no significant trend observed ($R^2 = 0.132$, $F_{1,11} = 1.68$, $p = 0.222$). We also hypothesized that greater ecosystem productivity in estuaries compared to rivers would increase the temporal stability of estuarine vs. riverine sites, which we observed in our study (Figure 2). Thus, population density and stability of NZMS may be driven by factors such as disturbance and resource availability, with extreme environmental conditions having less of an effect (e.g., Zaranko et al. 1997, 809; Dybdahl and Kane 2005, 1592). It is well known that invasions can be promoted by disturbance through the creation of available niche space (Hobbs and Huenneke 1992, 324; Mack and D'Antonio 1998, 195), thus, NZMS may be exploiting empty niche space in estuaries, created by variable and harsh abiotic conditions. These conditions could include low dissolved oxygen (freshwater sites) and high and variable salinity (estuarine sites), which have been observed to increase vulnerability to NZMS establishment (Herbst et al. 2008, 324). In contrast, systems with very low disturbance like spring-fed freshwater streams with constant temperatures and flow rates are also conducive to NZMS establishment and can promote high densities (Hall et al. 2003, 1121; Kerans et al. 2005, 123; Alonso and Castro-Díez 2008, 107).

Relationships between NZMS and Native Macroinvertebrates and Food Webs

We observed a negative relationship between NZMS density and native macroinvertebrate diversity when comparing across all ecosystems (Figure 4), which supported our hypothesis. There are two possible interpretations of this result: (1) NZMS may be having a negative effect on the biodiversity of the invaded communities via competition; and (2) NZMS have better establishment success in systems with already low native diversity. NZMS densities thriving at low disturbance (other studies) and high disturbance (this study) sites may be explained by the intermediate disturbance hypothesis. In this hypothesis, diversity forms a bell-shaped curve in relation to disturbance, where low disturbance and high disturbance both result in lower community diversity (Connell 1978, 1302; Townsend et al. 1997, 938). Biodiversity in estuaries is typically controlled more by organisms' ability to tolerate disturbance rather than by competition (Kittelson and Boyd 1997, 770; Chabrerie et al. 2001, 1088), whereas environmental conditions, like salinity, tend to be comparatively less variable in rivers, resulting in greater diversity (Attrill 2002, 262). The negative correlation we observed between native macroinvertebrate diversity and NZMS densities suggests that more disturbed systems may be creating available niche space for NZMS due to a lack of native macroinvertebrate diversity. Other studies have found this same pattern and determined that communities with low species richness and diversity have fewer species to compete with invasive species and therefore contain a wider range of unoccupied niches (Wolff 1998, 393; Leppäkoski and Olenin 2000, 151; Paavola et al. 2005, 738).

The relationship between NZMS and macroinvertebrate functional feeding groups provides insight into the mechanism by which invasive species may affect native communities. NZMS and herbivore densities exhibited a weak negative correlation (Figure 5a), supporting our hypothesis and other studies' findings of competition at high invasive herbivore densities (Byers 2000, 1225; Mooney and Cleland 2001, 5446; Hall et al. 2006, 1121). Additionally, herbivore communities exhibited some overlap with the trophic niche of NZMS, ranging from 0–55% (Figure 7). However, these results may be affected by the absence of herbivores in estuarine systems, which had high NZMS densities. These estuaries may not actually be absent of herbivores: collector-gather type organisms like oligochaetes (classified as detritivores) could be functioning as herbivores in estuaries to fill the empty niche. In contrast, omnivores were abundant at all sites and were positively correlated to NZMS densities (Figure 5d), which may provide evidence of facilitation or predation. Omnivorous crayfish were present in several of our study systems and are known consumers of NZMS (Brenneis et al. 2011, 169; Twardochleb et al. 2012, 1162); we hypothesize that crayfish may be benefitting from NZMS as an additional food source. This hypothesis is supported by our use of stable isotopes to analyze the trophic ecology of NZMS in their invaded habitats: the $\delta^{15}N$ of crayfish was typically higher than that of NZMS, indicative of potential crayfish predation on NZMS (Cimino 2016). Another potential explanation is that the systems and conditions that are suitable for NZMS are also suitable for omnivores. Studies have observed that factors promoting native species richness and density can also promote the establishment and population density of invasive species (Levine 2000, 852; Schreiber et al. 2002, 317; Brenneis et al. 2010, 1755). For example, NZMS have the ability to break down organic matter and can also provide high nutrient waste, potentially facilitating native taxa (Schreiber et al. 2002, 317). Yet, detritivore densities were not correlated with NZMS densities in our study. Thus, using a combination of organismal trait data on feeding and stable isotopes may provide useful information to understand the impacts of invasive species on native taxa, but should be interpreted cautiously given known variations in intra- and inter-specific feeding.

We observed very little overlap between NZMS feeding range and that of benthic competitors' and predators' feeding ranges (average overlap across all functional feeding groups = 5.3%), which suggests that NZMS are not currently competing with these other functional feeding groups for food. A possible explanation for this pattern could be that NZMS at high densities have already forced competitors to change their diets (e.g., Hall et al. 2006, Moore et al. 2012). However, we also observed an intriguing pattern whereby the feeding range of NZMS was larger at sites with higher NZMS densities than at sites with low NZMS densities (Figure 8). This did not support our hypothesis that the NZMS feeding range would become more specialized at higher densities (i.e., an Eltonian niche shift). A larger trophic range would indicate a more generalized diet and thus would not be likely to force competing functional feeding groups to change their diets. Future experimental studies could elucidate this pattern by manipulating both native biodiversity and food availability.

CONCLUSION

We found that estuarine systems were the most conducive to high densities of NZMS possibly due to the lack of herbivore competitors and availability of niche space in a disturbance-controlled estuary, as opposed to the lack of niche space in the more competition-controlled rivers and lakes. We contend that NZMS likely thrive in systems with low biodiversity and ample niche space rather than outcompeting native competitors, which is supported by the lack of feeding range overlap of NZMS and other functional feeding groups, as well as the negative correlation between NZMS densities and native invertebrate diversity. Even at high NZMS densities in estuaries and some of the freshwater systems, our results do not support the conventional wisdom that NZMS will out-compete native taxa or markedly influence competitors' diets (Hall et al. 2006, Moore et al. 2012). In fact, some competitors may even be facilitated by NZMS (e.g., omnivores). Our study echoes the findings of multiple other studies, which show that the influence of invasive species is often subtle and dependent on the composition of the recipient community (Schreiber et al. 2002, 317; Rodriguez 2006, 927; Brenneis et al. 2010, 1755). Densities of NZMS in our study were moderate compared to other studies (e.g., Kerans et al. 2005, 123), thus it is possible that systems with higher NZMS densities will observe greater influences on native competitors. Thus, management should focus on proactive approaches to minimize NZMS densities and keep NZMS from spreading to systems with available niche space, particularly in systems that may be controlled by disturbance rather than competition where NZMS populations can reach extremely high densities.

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Chapter 2

BRAZILIAN RATTLESNAKE (*CROTALUS DURISSUS LINNAEUS* 1758) IN THE ATLANTIC FOREST OF RIO DE JANEIRO STATE: A CASE OF BIOLOGICAL INVASION OR A RED ALERT TO PUBLIC HEALTH?

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ABSTRACT

Crotalus durissus is the snake species within its genus which covers the largest geographic distribution, reaching almost the entire South American territory. This species is widely dispersed and adaptable to open and anthropized environments. *C. durissus* is increasingly entering areas which once belonged to the Atlantic Forest biome and were covered by Ombrophilous (broadleaf) Forest. The main reason this species is establishing in this new environment is due to Atlantic Forest deforestation; initially to establish coffee and sugarcane plantations, and later to establish pastures for livestock. The new environment has become similar to the *Cerrado* biome, which was the original habitat of *C. durissus* in southeastern Brazil. *Cerrado* is one of the most representative biomes in the State of Minas Gerais, and from which several *C. durissus* individuals have migrated from towards the State of Rio de Janeiro. With this new panorama, the species has started to cause sanitary and possibly ecological problems such as an increase in number of local residents injured by snakebites (affecting the distribution of snake anti-venom in health posts, and predation and competition with native species). In this chapter, we aim to review and analyze the past and present view of *C. durissus* distribution in the State of Rio de Janeiro through scientific literature review and consultations in the biological collections of the Rio de Janeiro National Museum and Vital Brazil Institute. We also correlated its distribution to the time it took to advance into the State.

Thus, we found *C. durissus* records in 13 municipalities and 11 of them were new records for the State. There was no prior confirmation about the presence of the species in these sites, evidencing that it is increasing its geographical distribution mainly toward degraded areas in the State of Rio de Janeiro. The new *C. durissus* records indicate that it is spreading into Rio de Janeiro through three routes: *Paraíba do Sul* River Valley, mountainous region, and *Serra da Bocaina* in the municipality of *Paraty*. The new rattlesnake records were associated to the Atlantic Forest degradation in these locations. The expansion in the distribution area of this poisonous snake into densely populated areas is expected to increase the number of snakebite accidents, which will require greater production of snake anti-venom. In addition, its expansion is expected to have ecological impacts on the native fauna. More attention should be given to this issue in the State of Rio de Janeiro in order to prevent the species from advancing into new areas.

Keywords: bioinvasion, environmental degradation, geographic distribution, rattlesnake

INTRODUCTION

The planet's Neotropical Region shelters invaluable and great biological diversity due to vicarious effects. Within them there are three main effects which can be drawn: the emergence of the Andes; the Panama isthmus; and the constant climatic changes during and after the Pleistocene epoch, which occurred approximately 1.8 million years ago (e.g., Wing and Sues, 1992, 327-416; Mittermeier et al., 2005; Wüster et al., 2005, 1095-1108).

This last event is the most known and controversial Atlantic Rainforest speciation model known as the Pleistocene Refuge hypothesis (Haffer 1969, 131-137; Vanzolini and Williams 1981, 251-255), during which the Amazon had its dimensions reduced due to climatic fluctuations, specifically at the end of the Pleistocene and Pliocene epoch (Quijada-Mascareñas et al., 2007, 1296-1312). This fact resulted in intense fragmentation of the biome and created geographical barriers which favored allopatric biological diversity formation (Haffer 1969, 131-137; Haffer and Prance 2001, 579-607).

Commonly known as rattlesnakes, serpents of the *Crotalus* genus had their speciation well evidenced with the occurrence of only one species on the south portion of the American continent *Crotalus durissus* (Linnaeus 1758). However, this species has the greatest geographical distribution among the genus, possessing 11 described subspecies (Campbell and Lamar 2004, 539-547), among which six can be encountered in Brazilian territory (Costa and Bérnils 2018, 57). These Brazilian subspecies are: *Crotalus durissus durissus* (Linnaeus 1758), described in the State of Amapá; *Crotalus durissus ruruima* (Hoge 1966), described in the State of Roraima; *Crotalus durissus cascavella* (Wagler in Spix 1824), registered in the States of Maranhão, Piauí, Ceará, Rio Grande do Norte, Paraíba, Pernambuco, Alagoas, Sergipe, Bahia, and Minas Gerais; *Crotalus durissus marajoensis* (Hoge 1966), recorded at *Marajó* Island (Pará); *Crotalus durissus collilineatus* (Amaral 1926), found in Tocantins, Mato Grosso, Mato Grosso do Sul, Goiás, Distrito Federal, Minas Gerais, and São Paulo; and lastly, *Crotalus durissus terrificus* (Laurenti 1768), found in Tocantins, Mato Grosso, Minas Gerais, Rio de Janeiro, São Paulo, Santa Catarina, Paraná, and Rio Grande do Sul (Hoge and Romano 1978/79, 491; Campbell and Lamar 2004, 539-547; Dos-Santos et al., 2005, 958-961; Costa and Bérnils 2018, 29). The species inhabits almost all of the Brazilian biomes, especially those with open and dry vegetation such as the Brazilian *Cerrado* (Brazilian Savannah), but they do not inhabit humid tropical forests such as the Amazon and the Atlantic Rainforest. *Crotalus d. terrificus* is the only subspecies with recent records in the State of Rio de Janeiro in the Atlantic Forest domain (e.g., Campbell and Lamar 2004, 539-547; Bastos et al., 2005, 812-815; Costa and Bérnils 2018, 29) (Figures 1 and 2).

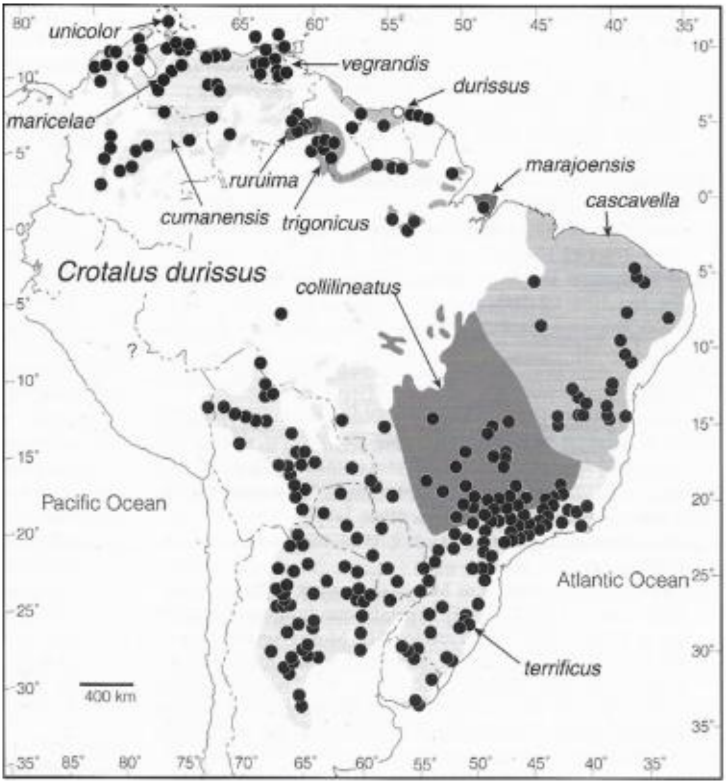

Figure 1. Geographical distribution of *Crotalus durissus* (Linnaeus 1758) in South America (black dots), with locality type (white dot). Map by Campbell and Lamar (2004).

Figure 2. *Crotalus durissus terrificus* (Laurenti, 1768) ♀ in an area of Atlantic Forest, Paraty/Cunha (Rio de Janeiro/São Paulo), Brazil. Photography by Jorge Pontes 2016.

C. durissus is commonly found in areas below 1,000 m altitude, but there is a recorded registry of 2,300 m at Sandia, Peru (Mijares-Urrutia and Arends 2000, 1-30). The species presents variable dorsal coloration in different regions in Brazil (Campbell and Lamar 2004, 539-547). It has the ability to colonize insular areas such as Aruba, and the climate changes and vegetal cover modifications during the Pliocene and Pleistocene epochs had an effect on the species distribution in South America, reaching pastures and coastal open vegetation (Mijares-Urrutia and Arends 2000, 1-30; Campbell and Lamar 2004, 539-547).

Exotic and invasive species are currently one of the greatest causes of native species extinction around the globe (e.g., Gurevich and Padilla 2004, 470-474; Pimentel 2011, 1-7). In competition for resources or by predation of native species, invasive exotic fauna grow rapidly in association with human occupation and other actions (e.g., Pimentel 2011, 1-7; Rocha and Bergallo 2012, 1-2). Several serpent species have become environmental, economic and public health problems in some places around the world (e.g., Jorge and Ribeiro 1992, 347-354; Duarte and Eterovic 2003, 126-135; Warrel 2004, 709-761; Bergstrom et al., 2009, 73-81). Public health problems in Brazil are aggravated by the reduction in the availability of snake anti-venom in health posts, including Rio de Janeiro State (Cardoso and Wen 2003, 3-5; Citeli et al., 2018, 123-124).

The species was not found in the State of Rio de Janeiro before 1940 because this region was not part of its natural geographic distribution and the forest matrix was a barrier for the rattlesnake to enter into the State (Melgarejo 2003, 52-53; Melgarejo 2013, 59; present study). However, *C. durissus* is becoming an invasive species in the Atlantic Rainforest biome, mostly due to deforestation and its aptness to adapt to open surroundings and anthropized environments such as man-made fields and pastures (Marques, Eterovic and Sazima. 2019, 44; Bastos et al., 2005, 812-815; Wüster et al., 2005, 1095-1108; present study). Its oldest record in the institutional collection in Rio de Janeiro State dates from 1988 in the municipality of *Itatiaia* (present study).

Ever since this time, *Crotalus durissus* has been taking advantage of its new environment, which (despite anthropic influences) resembles its original habitat, the *Cerrado*. Once established in its new habitat, the species begins to not only affect the native fauna, but also the local human population, thereby causing new snakebite accidents being recorded in places in which they did not previously occur. Therefore, this becomes a problem in the Health Ministry's snake anti-venom distribution directive for the incident regions, as *Crotalus durissus* is responsible for 7.7% of the snakebite accidents in Brazil (Duarte and Menezes 2013, 1-4; Melgarejo 2013, 59).

Although this percentage seems relatively low, the mortality rate is 1.84% (Pinho and Pereira 2001, 24-29). This is higher than deaths caused by the *Bothrops* genus (0.45% mortality rate) which is responsible for 90.6% of snakebite incidents in Brazil (Luna and Silva 2011, 130). The remaining incidents of medical importance are attributed to serpents of the *Lachesis* genus (known in Brazil as "*sucurucu*," "*pico-de-jaca*," and "*surucutinga*"), accountable for 1.4% of incidents, and the *Micrurus* genus ("*corais-verdadeiras*"), responsible for approximately 0.4% of the incidents (Pinho and Pereira 2001, 24-29).

In view of the above, in this chapter we aimed to compile the most recent records of the occurrence of *Crotalus durissus* in Rio de Janeiro

State, as well as to know the situations in which the species advance in its area of occurrence. More specifically, we aimed to clarify when the species began to spread to new municipalities in Rio de Janeiro State, particularly in the most populous regions, which might become a leading factor for an increase in snakebite accidents.

METHODS

Literature Review

A bibliographic revision was performed in order to identify articles and further publications regarding biogeography, ecology and registry of the *Crotalus durissus* species in the municipalities of the Rio de Janeiro and neighboring States. The criteria used to choose these articles were the keywords regarding the *Crotalus* genus, invasion specimens, snakebites in Rio de Janeiro State, and seasonal activity. Papers used in this study were mainly compiled in academic articles which comprehend *Crotalus durissus* distribution inside or outside of Rio de Janeiro State. Google Scholar, Scielo, ScienceDirect and Web of Science databases were used to search for the articles.

Collection Review

We performed a review on the herpetological collection of the Rio de Janeiro National Museum (MNRJ) and the Vital Brazil Institute, Niterói (IVB) to compile rattlesnake records in the State of Rio de Janeiro and vicinity between 1980 and 2019. The inventory books of the visited institutions contain capture data, local, rostrum-cloacal length (RCL), gender, acronym (record number) and other additional observations. The review at the Vital Brazil Institute was done via digital inventory books using a filter to solely compile the State of Rio de Janeiro and vicinity, as well as individuals which had been run over by vehicles. It was not possible to identify the gender of the animals or the RCL for data collected at the Vital Brazil Institute because the Biological Collection Division was relocated and disorganized at the time of consultations.

A physical inventory book was rearranged at the National Museum using the same filters as in the aforementioned review. A great deal of the deposited specimens in the Museum were collected from roads and their physical condition presented damage to such an extent that the RCL had to be taken by attaching body parts (head, body, and tail) with the use of metric tape. Specimen gender discrimination was performed by visualizing the reproductive organs (when possible) or by consulting the inventory books. We have also included the date and place of collection in the review, and other relevant information observed in the field.

The data gathered in the institutions were processed using Microsoft Excel® software, from which graphics and lists were elaborated using municipality's occurrence as criteria.

Maps

We mapped the invasion of *Crotalus durissus* in municipalities of the State of Rio de Janeiro from 1988 to 2019 through bibliographic data and collection records. The map was generated via ArcGIS software, entering the geographic coordinates (UTM, SIRGAS 2000) and indicating the species occurrences. A breakthrough in the species distribution in the State of Rio de Janeiro was created by organizing occurrences by decades. Some records, more specifically the roads sampled (n = 10), did not have additional information about the exact collection point. Thus, we chose to perform a coordinate approximation according to the indicated locations. When the registry only mentioned the municipality name in the inventory books, we used the central point (coordination) of the city as reference.

RESULTS

We found 77 records of *C. durissus* between 1988 and 2019 for the State of Rio de Janeiro: 58 records were found in the Vital Brazil Institute, 15 records in the National Museum (Figure 3), and four other casual encounters: two in the municipality of Paraty, one in Cunha, and another in Miguel Pereira. The records found in the Vital Brazil and National Museum institutions comprised 13 municipalities, in which eleven were new occurrences (*Nova Friburgo*, *Miguel Pereira*, *Paraty*, *Petrópolis*, *Vassouras*, *Itatiaia*, *Conservatória*, *Levy Gasparian*, *Paraíba do Sul*, *Três Rios*, and *Volta Redonda*). Of the 77 records, 54.5% (n = 42) were found in the municipality of *Valença*. An other 9.0% (n = 7) were registered in *Três Rios*, 9.0% (n = 7) in *Rio das Flores*, 7.8% (n = 6) in *Paraíba do Sul*, 3.9% (n = 3) in *Levy Gasparian*, 2.6% (n = 2) in *Paraty*, 2.6% (n = 2) from the BR 040 highway, and 1.3% (n = 1) in *Conservatória*, *Volta Redonda*, *Itatiaia*, *Nova Friburgo*, *Petrópolis*, *Miguel Pereira*, *Cunha*, SP/*Paraty*, RJ (border municipalities) and *Vassouras*.

Among the examined specimens, seven were identified at the subspecies level (*Crotalus durissus terrificus*). All of them were found in the municipality of *Rio das Flores*. The number of *C. durissus* collected in the State of Rio de Janeiro and deposited in the institutional collections varied over a period of 31 years (between 1988 and 2019) (arithmetic average 2.3 ± 3.2 individuals per year). Most specimens were collected in the early 1990s and this may be related to collection activities carried out at the *Sítio Divina Providência*. The site is in the municipality of *Valença* and is bordered by the State of Minas Gerais.

A total of 10 *C. durissus* individuals were registered at the BR-040 highway nearby the cities of *Matias Barbosa*, MG (n = 4), *Levy Gasparian*, RJ (n = 1), *Juiz de Fora*, MG (n = 2), *Simão Pereira*, MG (n = 2) and *Lima Duarte*, MG (n = 1). The oldest record was from 1988 at the *Represa do Funil* in the municipality of *Itatiaia*, albeit the most recent records were found between 2018 and 2019 in *Miguel Pereira*, *Nova Friburgo*, *Petrópolis* and *Três Rios*.

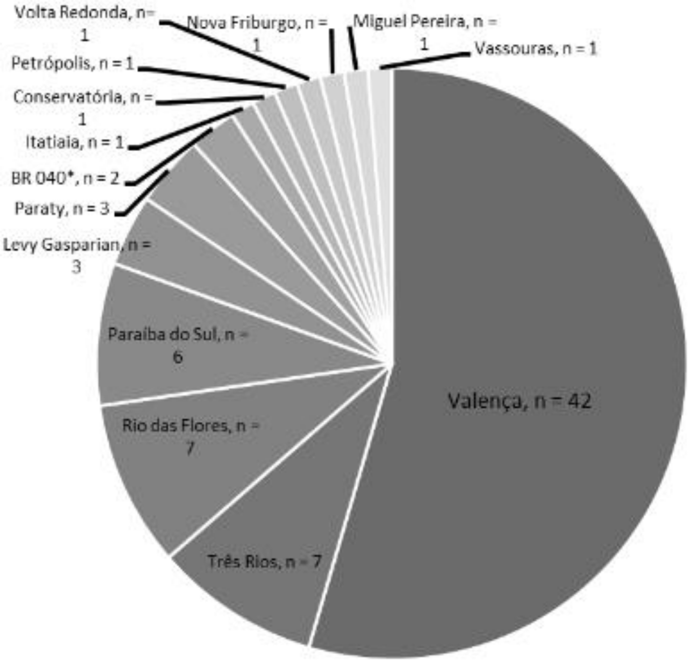

Figure 3. *Crotalus durissus* specimens (n) collected in the State of Rio de Janeiro between 1988 and 2019 and deposited at the Rio de Janeiro National Museum (Rio de Janeiro City) and Vital Brazil Institute collection (Niterói City).

We observed that another *C. durissus* was captured in 2006 in *Rio das Flores* after a 10-year gap. There were six new registries found in 2007. Furthermore, there was a considerable number of these serpents encountered at the roadside (n = 10) in the 2010 decade, usually run over by vehicles.

Although *Paraty* has been indicated as a new viable route for southern invasion into the State of Rio de Janeiro, all recent occurrences of *C. durissus* have been recorded in the Rio de Janeiro Metropolitan Region. In addition, an individual was registered in 2019 in the *Miguel Pereira* municipality by Murilo Vicente (personal communication). The taxonomic identity of this specimen was confirmed through photographic evidences (Figure 4).

Figure 4. A *Crotalus durissus* (Linnaeus 1758) individual was registered in the municipality of *Miguel Pereira*, Rio de Janeiro State. Photograph by Murilo Vicente on March 04, 2019.

Figure 5. *Crotalus durissus* (Linnaeus 1758) specimen being captured in the SP–171 highway, border of the municipality of *Cunha* (São Paulo State) and *Paraty* (Rio de Janeiro State), near *Serra da Bocaina* National Park. Photography by José Vitor Maciel, 2016.

In 2016, during a study by researchers from the Rio de Janeiro State University in *Paraty*, three *C. durissus* were registered at the *Serra da Bocaina* National Park. One of those individuals was run over on the border of the States of Rio de Janeiro and São Paulo, in the municipality of *Cunha* (Figure 5).

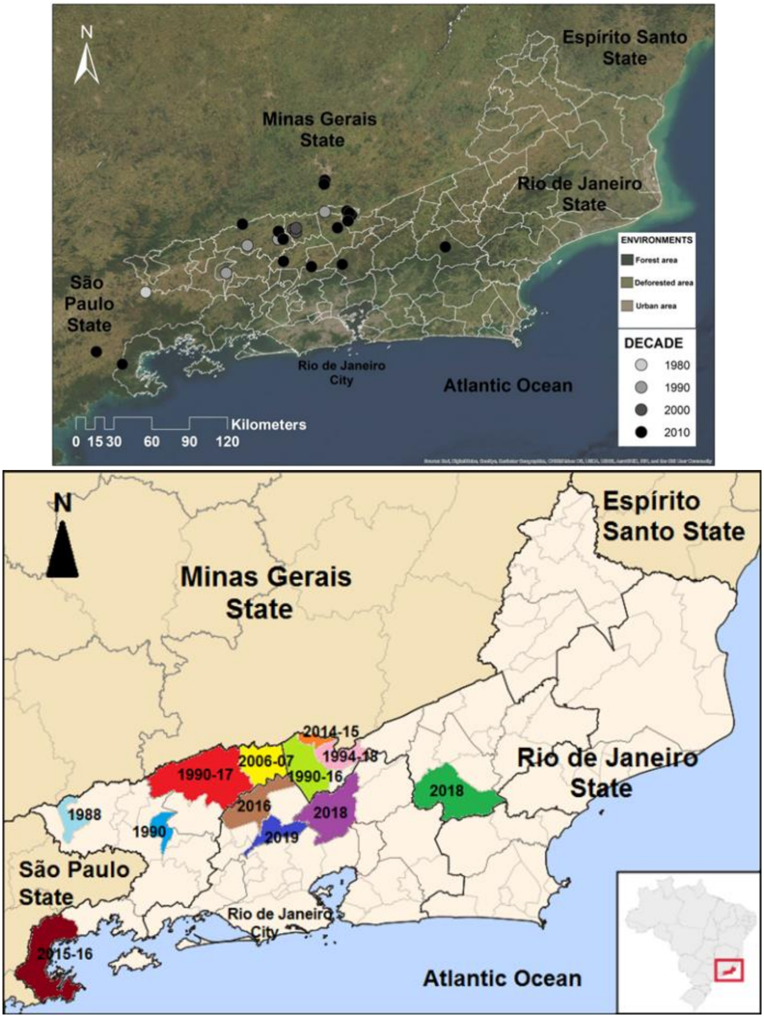

Figure 6. Forest cover map indicating *Crotalus durissus* (Linnaeus 1758) registries by decade between 1988 and 2019 (a). The occurrence records were extracted from the Rio de Janeiro National Museum (Rio de Janeiro city) and Vital Brazil Institute (Niterói city) Collections. Advance in space and time of *C. durissus* records in the municipalities of Rio de Janeiro State (b).

We observed that the area of occurrence of the species increased between 1988 and 2019, and it is advancing towards the central region of the State of Rio de Janeiro. Advancement through municipalities and all records overlaps deforested and anthropized areas, mainly pasture and secondary vegetation in the early stages of regeneration (Figures 6 and 7).

Figure 7. Aspect of *Serra da Bocaina* on the border of Paraty (Rio de Janeiro State) and *Cunha* (São Paulo State) municipalities. Forest fragmentation and anthropic environments facilitate the entrance of *Crotalus durissus* (Linnaeus 1758) into *Paraty*, Rio de Janeiro State. Photography by Jorge Pontes, 2011.

DISCUSSION

The advancing area of *Crotalus durissus* occurrence was directly influenced by anthropic actions in the State of Rio de Janeiro, mainly due to deforestation which gave space to coffee and sugarcane plantations, and culminated in a reduction of the Dense Ombrophilous Forest to sparse forest fragments, making way for the serpent species incursion. Almost 80% of the Atlantic Rainforest domain has already been lost (Bastos et al., 2005, 812-815; Castro and Lardosa 2015, 6-27), and pasture areas for cattle ranching have been established in its place. According to Bastos et al., (2005), the presence of *Crotalus durissus* in these regions may be attributed to the great floods in the *Paraíba do Sul* River Basin which occurred in the 1950s and 60s. This argument seems questionable due to the fact that the species indicates its advance regardless of natural disasters, and it is believed that the advance is caused by occupancy of anthropic environments similar to the *Cerrado* biome, which were found at the Vale do Paraíba region due to farming exploitation (Costa and Clemente 2009, 57-65; Melgarejo 2013, 59).

Our study adds eleven new legal municipalities with the occurrence of *C. durissus* to the study by Bastos et al., (2005), where only *Valença, Rio das Flores* and *Resende* have some officially registered specimens, with the others being informal records. The records for the studied period (1988-2019) indicate that the species has been advancing to deforested areas and areas replaced by pasture or anthropic fields. This fact is also pointed out by Melgarejo (2003).

This advance form with a high concentration of records in the north of the State may be related to the high rates of land occupation by pasture, livestock and sugarcane plantations in established farms (Rocha et al., 2003, 3-32; Costa and Clemente 2009, 57-65). The advance of *C. durissus* may also be related to deforestation and the fire propagation cycle which lead to secondary vegetation (Tanizaki-Fonseca and Bohrer 2009, 81-90), and creates habitats and passageways for the entry and dispersion of the species.

In addition to the spatial records, the generated maps enable visualizing the expansion of the species to areas which were previously free from *C. durissus* invasion, such as *Miguel Pereira, Nova Friburgo* and *Petrópolis*. Interestingly, the aforementioned localities do not present proper climatic patterns for the natural habitat of serpent species (Campbell and Lamar 2004, 539-547; Melgarejo 2003, 52-53), indicating that rattlesnakes are expanding their distribution to colder and damper environments.

There was an increase in the number of rattlesnakes in the 2010s, especially in the highways linking Minas Gerais and Rio de Janeiro. However, another possible invasion route seems to have been established through *Paraty* because the region is vastly occupied by pasture and anthropic fields (Rocha et al., 2003, 3-32; Mittemeier et al., 2005; Santos et al., 2009, 41-56).

Population expansion of *C. durissus* seems evident in the interior of Rio de Janeiro State and throughout the central and coastal areas. Its occupation coincides with more populous areas where some municipalities will be considerably more affected than others. The species is in second place regarding snakebite accidents in Brazil (Araújo, Santa Lucia and Cabral 2003, 7) and the reduction in snakebite anti-venom production, health posts and the lack of organization regarding available information (Bochner and Struchiner 2002, 741-744; Citeli et al., 2018, 123-124; Ministério da Saúde 2019) could mean an increase in the number of complications due to accidents and even more deaths.

The situation regarding the advance of an exotic species from another Brazilian region which was able to adapt to the Ombrophilous Forest domain is not only creating a concerning panorama for environmental conservation regarding the implications of biological invasion, but also to the healthcare system as a whole in Rio de Janeiro State.

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Chapter 3

TERRESTRIAL INVASIVE SPECIES ON FERNANDO DE NORONHA ARCHIPELAGO: WHAT WE KNOW AND THE WAY FORWARD

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ABSTRACT

Fernando de Noronha (FN) is an oceanic archipelago in the northeast coast of Brazil, approximately 360 km away from the continent. It is composed of 21 islands and the biggest (also named Fernando de Noronha) has 16.4 km² and is inhabited since the 17th century. The archipelago is an important breeding site for several marine birds, including regionally threatened species such as *Puffinus lherminieri*, *Sula sula*, *Phaethon aethereus,* and *Phaethon lepturus*. It also contains critical habitat for endemic species such as the mabuia lizard (*Trachylepis atlantica*), Ridley's worm lizard (*Amphisbaena ridleyi*), and birds like the noronha elaenia (*Elaenia ridleyana*) and the noronha vireo (*Vireo gracilirostris*), all endangered. Since its discovery and colonization, FN's native species have been impacted by the arrival of several exotic and invasive species, especially domestic cats (*Felis silvestris catus*), dogs (*Canis lupus familiaris*), brown and black rats (*Rattus norvegicus* and *R. rattus*), house mice (*Mus musculus*), cururu-toad (*Rhinella jimi*), tegu lizard (*Salvator merianae*), cattle egret (*Bubulcus ibis*), little fire ant (*Wasmannia auropunctata*), rock cavy (*Kerodon rupestris*), and the river tamarind (*Leucaena leucocephala*). Moreover, several zoonotic diseases, like salmonellosis and toxoplasmosis, have been linked to some of these invasive species. Until recently, the geographic isolation of FN from research centers, in combination to elevated operational costs in conducting research in this remote location, have prevented studies on FN's invasive species. However, recently established partnerships among governmental executive bodies, NGO's, research institutes and universities have promoted the development of research and management of invasive species on the archipelago. In this chapter, we describe and discuss the scientific advances of the last decade assessing the population demographics of FN's invasive species, their impacts, knowledge gaps, and required species management to conserve the native biodiversity of this unique Atlantic archipelago.

Keywords: threatened species, native biodiversity, island ecology, biodiversity loss

INTRODUCTION

Fernando de Noronha is an oceanic archipelago in the Northeast of Brazil, approximately 360 km away from the continent (Figure 1). Of volcanic origin, the archipelago was formed from 11 up to 2 million years ago and discovered in 1503 by Amerigo Vespucci. It constitutes 21 islands and islets with the largest one (also named Fernando de Noronha) being inhabited since the XVII century. Today the official resident population is around 3,100 inhabitants (IBGE 2019) — although with temporary residents and tourists included it is reported by local authorities that this number may be almost double. At least four commercial flights arrive every day on the island, with tourists from all over the world, but Brazilian tourists predominate. The main island is 16.4 km^2 and the highest point is 321 m (Almeida 2002, 361). Approximately 70% of the terrestrial extent of the archipelago is included in the Fernando de Noronha Marine National Park (PARNAMAR). The whole archipelago is an important breeding site for several marine birds, including regionally threatened species (e.g., *Puffinus lherminieri* Lesson & R 1839, *Sula sula* Linnaeus 1766, *Phaethon aethereus* Linnaeus 1758, and *Phaethon lepturus* Daudin 1802) (Ministerio do Meio Ambiente 2014) and habitat for endemic species such as the *mabuya* lizard (*Trachylepis atlantica* Schmidt 1945), Ridley's worm lizard (*Amphisbaena ridleyi* Boulenger 1890), and birds like the *noronha elaenia* (*Elaenia ridleyana* Sharpe 1888) and the *noronha* vireo (*Vireo gracilirostris* Sharpe 1890).

Introductions of alien species to oceanic Islands such as Fernando de Noronha are frequent, despite scientific discouragement (Usher 1988, 119). Negative effects of alien introductions to islands include deep structural changes to natural environments, increase in predation of native species and spread of diseases through the local domestic and wild animal populations (Clavero and Garcia-Berthou, 110; Donlan et al., 2004, 267). Seven major inadvertent and intentional introductions have occurred since the archipelago's colonization: the domestic cat (*Felis silvestris catus* Linnaeus 1758), the brown and the black rats (*Rattus norvergicus* Berkenhout 1769 and *R. rattus* Linnaeus 1758), the *cururu* toad (*Rhinella jimi* Stevaux 2002), the tegu lizard (*Salvator merianae* Duméril & Bibron 1839), the electric ant (*Wasmannia auropunctata* Roger 1863) and the river tamarind (*Leucaena leucocephala* (Lam.) de Wit). All these species are considered alien and invasive in the archipelago and are a major concern of local authorities, despite the reptiles and ant being native to continental Brazil. Another introduced species in the archipelago, the rock cavy (*Kerodon rupestris* Wied-Neuwied 1820), is also native and endemic to the continental Brazilian Caatinga (drylands), classified as an exotic species but not invasive on the archipelago (Micheletti, Mangini, and Gasparotto in press). Apart from the direct impact on native fauna, several zoonotic diseases, like salmonellosis and toxoplasmosis, have been linked to some of these vertebrate invasive species.

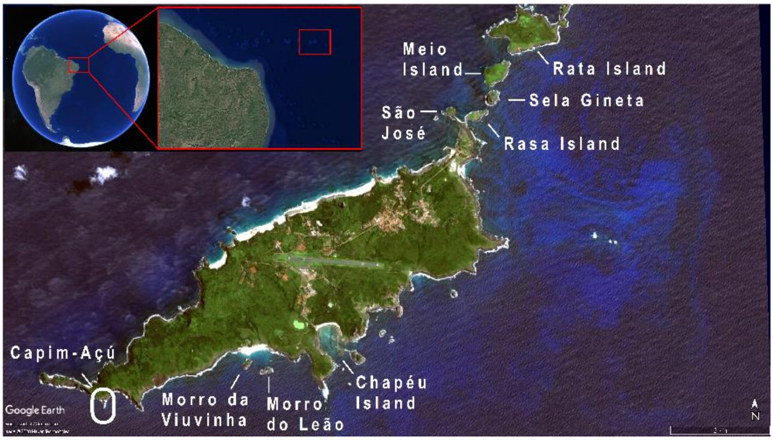


Figure 1. Location of the Fernando de Noronha archipelago, Pernambuco state, Brazil.

We start the present chapter by discussing the scientific advances of the last decade in describing native biota of the island and assessing the impacts of Fernando de Noronha's invasive species on these. We then move on to debating the current information available on invasive species population demographics, knowledge gaps, *in situ* actions, and required species management to conserve the native biodiversity of this unique Atlantic archipelago. Impacts of different invasive species vary in both intensity and extent, and there is no panacea for dealing with all invasive species at once. The information in the present chapter is two-fold: (1) it calls for urgent measures due to the impacts presented on native and endemic species, and (2) discloses foundational information such as population sizes, distribution on the island and potential risks realized in other places, on each individual invasive species to support the development of more efficient and long-term effective management plans.

MABUYA

The *noronha*-skink, or *mabuya*, is the only species of the genus *Trachylepis* naturally found in the Americas. All other species are distributed only in the Old World (Eurasia and Africa). The *mabuya's* distribution is restricted to the Fernando de Noronha archipelago and its radiation from an African ancestor occurred during the Miocene, which suggests these might have been one of the first inhabitants of the archipelago (Mausfeld et al. 2002, 281). The mabuya is considered a pollinating reptile, seed disperser, and has fundamental ecological importance in the archipelago ecosystem, also acting as environmental decomposer. Although its abundance is relatively high compared to other continental lizard species, locals report a reduction in the number of individuals over the years (V.P.O. Gasparoto, personal observation). The estimated density of individuals per hectare on the main (inhabited by humans) island is half that of the estimated density on the secondary (uninhabited) islands, suggesting the population abundance on the main island is at least 50% below carrying capacity (V.P.O. Gasparotto, pers. obs.). Significant morphometric differences — smaller biometric parameters such as length — have been observed for individuals present on the main island in comparison to the individuals present on the secondary islands. This might suggest predator preference for bigger sized lizards, or lower life expectancy on the main island (V.P.O. Gasparotto, pers. obs.).

Anthropogenic environmental changes and the introduction of invasive species are considerable threats to Fernando de Noronha's mabuya, which in 2017, was classified as threatened by Pernambuco state, to which Fernando de Noronha belongs. Cats, followed by rats and egrets, are responsible for most of its population decrease, mainly by predation (V.P.O. Gasparotto, pers. obs.). In fact, local residents even keep cats in their homes for the specific purpose of controlling the mabuya in their properties, contributing to its population suppression. Systematic and continuous monitoring of the population is a very important action for the conservation of this species.

SEABIRDS

About 38% of the seabird species recognized globally occur in Brazil as breeders, migrants or vagrants (Piacentini et al. 2015, 291). In Brazil, 14 seabird species breed at the four offshore islands and one atoll: Fernando de Noronha, and São Pedro and São Paulo (SPSPA) archipelagos; Trindade Island together with Martin Vaz, the *Abrolhos* Archipelago over the continental shelf and *Atol das Rocas* (Vooren and Brusque 1999). The most important breeding area in terms of the number of seabird species and abundance is the Fernando de Noronha archipelago (Mancini, Serafini, and Bugoni 2016, 94). Nevertheless, until recently, knowledge of the seabirds of Fernando de Noronha was limited to a list of species and occasional observations upon them. The uncertainty in population estimates was highlighted in the National Plan of Action for the Conservation of Albatross and Petrel (Neves et al. 2006) and other reports. The discovery of at least one additional breeding species, the Audubon's shearwater (Soto and Filippini 2003, 330), and increased interest in the conservation status of the archipelago in general, led the Brazil's National Center for Bird Conservation and Research (ICMBio/CEMAVE) to start in 2009 to survey seabird population numbers more consistently (P. Serafini pers. obs.). Follow-up surveys have been occurring since 2010, including studies on tropicbirds (Leal et al. 2016, 559) and the shearwater (Silva and Olmos 2010, 139). Mancini et al. (2016, 94) presented population size estimates for the eleven seabird species that breed on Fernando de Noronha, and continuous monitoring of the colonies using standardized census techniques has started since then (P. Serafini unpublished data).

Although not yet extensively documented, occasional observations showed that alien predators such as cats and tegus in Fernando de Noronha play an important role on seabird, egg and chick predation (Russell et al. 2018, 193). On Fernando de Noronha most of the remaining seabird colonies are generally located in uninhabited secondary islands and islets of the archipelago, cliffs or headlands, all safeguarded from invasive species. The main concern for invasive species impacts on seabirds are related to the regionally threatened species (*Sula sula*, *Phaethon aethereus, Phaethon lepturus* and *Puffinus lherminieri*) (Ministerio do Meio Ambiente 2014). Currently, in Brazil, the red-footed Booby (*Sula sula*) breeds only on Fernando de Noronha (Fonseca-Neto 2004, 119) and its arboreal breeding habits seems to protect the species from terrestrial predators such as tegu lizard (*Salvator merianae*) and feral cats (*Felis catus*) (Barbosa-Filho et al. 2010, 101) . Other species do not have the same luck: a small colony of no more than 20 individuals of masked boobies (*Sula dactylatra* Lesson & R 1831) persist in trying to breed on the main island at *Ponta Capim-Açu*. Unfortunately, this species is more susceptible to predation as it nests on the ground. During the past two years of population monitoring no more than one or two successful nests have been recorded at this site (P. Serafini, pers. obs.).

Red-billed tropicbird (*Phaethon aethereus*) breeds in small numbers on Fernando de Noronha and fewer than ten individuals have been recorded in the last few years (P. Serafini, unpubl. data). Given its larger size precluding nesting on sheer cliffs, the species would have been more vulnerable to introduced mammal predation. The red-billed tropicbird now remains breeding only at *Ponta das Caracas*, on the main island. Another species, the white-tailed tropicbird (*Phaethon lepturus)*, has remained relatively stable in numbers since the first count in 1982. The population is estimated around 200 individuals from breeding pairs counts (Mancini, Serafini, and Bugoni 2016). The white-tailed tropicbird nests on cliffs of the main island and also on secondary islets and islands such as *Chapeu*, *Meio*, *Rasa* and *Rata*. Historically, white-tailed tropicbirds were once considered common on this island (Oren 1984, 19), but are now restricted to less than 100 nests. The white-tailed tropic birds' sub-species found on Fernando de Noronha is considered to be the same as on Ascension Island, but hatching and fledging success observed in Fernando de Noronha was much lower than Puerto Rico, Ascension Island and Cousin Island (Leal et al. 2016, 559).

Exotic species such as *Rattus rattus*, *R. norvegicus* and *Felis catus* are known to prey on this seabird species' eggs and chicks elsewhere and might be one of the reasons for the low fledging success and low number of nests on Fernando de Noronha archipelago (Leal et al. 2016, 601). Repeated events of white-tailed tropicbird predation by cats were observed at different times, between 2014 and 2016, by researchers of *Tríade* Institute (Brazilian Institute for Conservation Medicine), who have been working with invasive species on the archipelago for more than a decade.

These predation events were observed mainly in nesting areas of the *P. lepturus* species inside of the PARNAMAR on the main island of the archipelago.

Audubon's shearwater (*Puffinus lherminieri*) is known to be restricted to 15 individuals on Fernando de Noronha, located only on the two neighboring small rugged southern offshore islets of *Morro de Leão* and *Morro da Viuvinha* (Mestre, Roos, and do Nascimento 2009, 1), which are believed to be free of rats and other invasive fauna. No more than one or two nests of Audubon´s shearwater have been recorded in the past two breeding seasons, which occur from May to November (P. Serafini, pers. obs). Nevertheless, Fernando de Noronha is the breeding site with the highest number of records of this species in the South Atlantic, making crucial the importance of keeping invasive species off this site.

LANDBIRDS

Only four species of landbirds occur on Fernando de Noronha archipelago: the *cucuruta* or *noronha elaenia* (*Elaenia ridleyana*), *sebito* or *noronha* vireo (*Vireo glacilirostris*), eared dove (*Zenaida auriculate* Des Murs 1847), and the exotic sparrow (*Passer domesticus* Linnaeus 1758) (Silva 2008). Both the *noronha elaenia* and the vireo are endemic bird species of the archipelago (IUCN 2019). *Elaenia ridleyana*, a grayish brown tyranid restricted to the Fernando de Noronha archipelago, occurs on the main island and also on *Rata* Island (P. Serafini, pers. obs.), and is the least common of the three resident landbird species. Estimates indicate a population ranging from 100 to 1000 individuals, with an average of 500 mature individuals on the main island (IUCN 2019; R. Krul pers. obs.). The population trend is considered stable and its small occurrence range is estimated at 20 km^2. According to the IUCN Red List (2019), *noronha elaenia*'s conservation status is 'vulnerable to extinction', explained by the existing small population. Noronha's *elaenia* can be found in shrubs, trees and even in gardens. Its diet is composed of insects and small fruits, particularly those of *Ficus longifolia* (Schott), an endemic tree of the archipelago. Reproduction occurs mainly between February and May, the period corresponding to the rainy period of the archipelago (Albano Schulz-Neto 2004). Although we lack historical population estimates, this species is believed to have been more abundant before the clearing of large trees after human occupation in the archipelago and introduction of cats (P. Serafini pers. obs.). It is common to observe cat predation on this bird species across the island (P. Serafini, R. Krul, R. Dias, P. Mangini, and T. Micheletti, pers. obs.; Oren 1984, 20; Ridgely and Tudor 1994, 1; IUCN 2019). Still, the biggest current threat to *noronha's elaenia* is the expansion of tourism on the island, burning and clearing of native vegetation, and the introduction of invasive plant species that can cause great damage to its habitat (Oren 1984, 20; Ridgely and Tudor 1994; IUCN 2019).

Sebito or *noronha's* vireo (*Vireo glacilirostris*) is also an endemic species of the Fernando de Noronha archipelago and considered as 'near threatened' (IUCN 2019). The Noronha vireo presents modified wing shape and associated musculature in comparison to its sister species *Vireo olivaceus* (Linnaeus 1766), suggesting an association with forest habitats. Although *noronha's* vireo occasionally occurs in the urbanized areas of the main island, its population density is higher in the most preserved areas with trees and shrubs (Schulz-Neto 2004, P. Serafini and R. Krul pers. obs.). The vireo's habit is restricted to the main island of the archipelago, feeding on insects, arthropods, small fruits and nectar, foraging in all strata of vegetation. It nests in trees and shrubs, building its nests with leaves and soft plant fibers, trapping them in the forks of the branches (Schulz-Neto 2004; Olson 2006, 178). Population estimates are around 670 mature individuals and the population trend is unknown (IUCN 2019).

Also known locally as "*avoante*", *Zenaida auriculata*, the eared dove, occurs from the Antilles to Tierra del Fuego with disjoint distribution throughout Brazil (Souza et al. 2010, 28). This species occurs in open vegetation areas such as fields, Cerrado (Brazilian savanna) and Caatinga, as well as areas of agricultural culture and urban centers (Souza et al. 2010, 28). The species has gregarious habits, forming flocks in sleeping areas and feeding on seeds and fruit. It is considered an important disperser of native plant species on the archipelago (R. Krul, P. Serafini, T. Mello, pers. obs.). As the eared dove builds fragile nests in bushes, palm trees or even on the ground, the loss of eggs and young by wind, rain or predation is very common (R. Krul and P. Serafini, pers. obs.). In Fernando de Noronha, it is considered as an endemic subspecies (*Z. auriculata noronha* Chubb & C 1919, Silva 2008). Even though mortality rates have not yet been studied, this species apparently undergoes predation by introduced small mammals such as cats (T. Mello, pers. obs.). Since 2016, landbird point counts have been conducted on the main island, and on some secondary islets, focusing on the evaluation of population estimates and trends and the results should become available in the near future.

NATIVE FLORA

The vegetation on the Fernando de Noronha archipelago is xeromorphic and seasonally deciduous, with herbaceous, shrubby and forest physiognomies. Currently, 211 native plant species are known for the archipelago, and six of them are endemic: *Cereus insularis* (Hemsl.), *Jacquemontia euricola* (Ridl.), *Combretum rupicola* (Ridl.), *Ceratosanthes rupicola* (Ridl.), *Spermacoce noronhensis* ((Sucre) Govaerts), and *Ficus longifolia* (Schott) (Freitas 2007). Some of the native species found on the archipelago have important relationships with the native and endemic fauna. One example is the *mulungu* (*Erythrina velutina* Willd.), which appears to have a mutualistic relationship with the mabuya providing food resources for the endemic lizard, and receiving pollination services (Sazima, Sazima, and Sazima 2009, 26). With human colonization in the 17th century, the archipelago's vegetation suffered with various forms of anthropic disturbance, such as burning, deforestation, logging, introduction of cattle and the expansion of open environments (fields, dams, roads, yards). Currently the native vegetation is reduced to fragments located in the PARNAMAR national park (R. J. V. Alves 2006).

CATS

Cats (*Felis catus*) have been listed as one of the most threatening introduced species worldwide, especially on insular ecosystems (Maeda et al. 2019, 1). There are no specific records, but domestic cats were probably introduced to Fernando de Noronha during its colonization in the 17th century, when cats were commonly used to control rats on European ships crossing the Atlantic. Free-ranging domestic cats in Noronha are classified in three main groups: outdoor pet cats, strays — cats that are in close proximity to humans and settlements but do not have an identifiable owner —, and feral cats — cats that are not subsidized by humans in any way, and are in general found inside the PARNAMAR national park. All free ranging cats have substantial negative impacts on the native and endemic fauna (Woolley et al. 2019, 354), and the pet and stray group's close association with humans favors the spreading and maintenance of zoonotic diseases in the environment. Recent research supports the hypothesis that direct or indirect food provisioned by humans favors cat population growth and therefore increases the predation pressure on native species (Maeda et al. 2019, 2).

Cat density has been monitored on Fernando de Noronha and was estimated to be 2 individuals/ha in 2017, totaling 1,287 individuals on the main island of the archipelago (Dias et al. 2017, 2339). Specifically, the feral cat population was estimated at 311 animals in 2015 (Dias et al. 2017, 2339) and re-estimated at 439 individuals in 2019 (F. Fonseca, unpubl. data.), which might indicate a population increase of approximately 41% in four years. In 2019, data collected in a survey counted 605 pet cats supervised by owner. Considering that most of the population of pet cats has free access to the outdoors environment, both pet and feral cat populations have significant impact on native species. In Fernando de Noronha, cats have already been reported feeding on endemic and endangered species such as the mabuya and landbirds (Russell et al. 2018, 193). The high cat density leads to a cascade of ecological imbalance that may be associated with environmental and public health issues, including exacerbated predation of native species and transmission of zoonosis, such as toxoplasmosis (Algar, Burbidge, and Angus 2002).

Among some strategies that have been applied to control cat population growth, neutering and spaying are considered efficient only when applied to the entire population (Leo et al. 2018, 57). Currently, there is an active neutering program on the island, as part of an action plan for cat population control on Fernando de Noronha. Since 2004, intensive neutering campaigns have been sporadically conducted on the archipelago thanks to many organizations, such as *Quintal de São Francisco*, Brazilian Institute for Conservation Medicine (*Tríade*), Pernambuco Federal Rural University (UFRPE), ICMBio and ATDE-Fernando de Noronha. Between 2004 and 2010, 450 cats were sterilized. In 2019, a sterilization campaign, which was part of an action plan for cat management in Fernando de Noronha (Instituto Chico Mendes de Biodiversidade 2019), resulted in the sterilization of 605 cats in 20 days (Ricardo Araújo, pers. obs.). However, those cats continue to negatively impact wildlife directly through predation (Greenwell, Calver, and Loneragan 2019, 445), and other strategies must be implemented in order to achieve biodiversity conservation goals. An additional part of this action plan aims to capture and euthanize feral cats inside PARNAMAR national park areas.

Cat management strategies such as poison and kill traps have been implemented on some islands around the world, but almost exclusively on uninhabited ones. Eradication has also only been achieved in some cases. The most successful management strategies rely on combining different methods such as capture, poisoning and hunting using rifles and dogs (Parkes et al. 2014, 311). In Brazil, lethal methods are not culturally accepted. Still, it is important to highlight that a recent effort (5,376 trap-hours) for live trapping feral cats inside the PARNAMAR national park resulted in zero captures (F. Fonseca, unpubl. data.), which emphasizes the importance of questioning the choice between accepting the need of other strategies for cat control and eradication (i.e., poison, culling) or gambling with the extinction of endemic species. Independently of the selected strategy for control or eradication of cats on Fernando de Noronha, one fact remains: working closely with the local community and selected stakeholders will be extremely important for successfully achieving the proposed management goal.

RATS

Effects of invasive rats on islands flora and fauna can be sufficiently extensive to affect ecosystem structure and functioning. Previous studies have shown rat’s capacity to suppress forest plants, and to induce extinctions or declines of flightless invertebrates, ground-dwelling reptiles, land birds, and burrowing seabirds (Towns, Atkinson, and Daugherty 2006, 863). The introduction of rats to Fernando de Noronha archipelago likely occurred in the 16th century (Carleton, Olson, and Vespucci 1999), following the first European visitors of the archipelago. Rats’ omnivorous diet leads to high ecological plasticity in different habitats, and their commensal behavior with humans allows them to colonize even distant locations (Drake and Hunt 2009, 1483). Those characteristics, combined with their high reproductive rates, make rats one of the most successful invasive species on oceanic islands. On Fernando de Noronha archipelago, *Rattus rattus* and *Mus musculus* (Linnaeus 1758) are widely spread, both on main and secondary islands. These species are well established even in remote places of the archipelago, where there are no human communities. On the other hand, *Rattus novergicus* is an invasive species that has been registered only in association with human habitations.

Among the rodent species mentioned above, *R. rattus* is the most relevant on Fernando de Noronha in terms of negative impacts. Its current presence and abundance on the archipelago may be an important obstacle to the maintenance and survival of endemic passerine bird populations, such as *Elaenia ridleyana* and *Vireo gracilirostris*. Apart from birds, *R. rattus* were also recently registered preying upon endemic mabuya (V. P. O. Gasparoto and P. R. Mangini, pers. obs.). Furthermore, there is much scientific evidence of rats as reservoirs of important infectious diseases that may affect the health status of human, livestock, and several native species’ populations (Strand and Lundkvist 2019).

No estimates of *R. rattus* population size and distribution are available for the entire archipelago, and there are evidences that the densities vary both among islands and habitats (P. Mangini, pers. obs.). The species has been observed on islands of the northern chain (*São José*, *Rasa*, *do Meio*, and *Rata*), but seems to be absent from some close islands from the southern coast (e.g., *do Chapéu*, *da Viúva*, *do Leão*). This has been reported by Soto (2009) and, more recently, by some of this chapter's authors. Considering the need to establish management strategies to eradicate rat population from important marine bird breeding sites, there is still an important lack of information in key sites such as steep *Sela Ginete* island, a potential stepping stone in the northern island chain between the main island and *Rata* Island.

Russell et al. (2018, 193) estimated *R. rattus* density as fluctuating from 29 to 49 rats per hectare in a specific location of the main island called *Quixaba*. Rat spatial distribution and density in other regions of the main island were not assessed. In a prospective evaluation of rat distribution on Rata island, the second largest island of the archipelago and currently uninhabited, researchers observed a distinct difference in *R. rattus* density according to different habitats sampled (e.g., shrub, arboreal or herbaceous vegetation, archaeological sites) (P. R. Mangini, pers. obs.). On *Meio* island, which is less than 17 hectares and was never inhabited, a preliminary study in 2016 indicated a very high density of 322 to 590 rats per hectare, without significant variation according habitat types. Researchers have anecdotally reported that during visits to the *Meio* Island for seabird census, especially during the night, rat infestation was so high, these would jump over their shoes and boots while they walked around the island.

In 2017, a project aiming to protect and improve the breeding status of seabird colonies successfully eradicated *R. rattus* from *Meio* island. The methods included the use of the rodenticide Brodifacoum in pellets and paraffin blocks deposited in 360 feeding stations placed 20 meters from each other in a series of five applications in intervals of one and a half to three months (P. R. Mangini, pers. obs.). Before and during rodenticide application, the abundance of native species populations was monitored.

Three months after complete rat population suppression, the local population of *E. ridleyana* and *Sula leucogaster* (Boddaert 1738) were already showing positive responses in number of individuals breeding, which could be associated with rat absence (P. R. Mangini, pers. obs.).

Regarding rat management strategies, there seems to be no genetic flow between Rata population and the main island, the two islands located in the extremes of the northern island chain (Gatto-Almeida et al. in review). Genetic analyses support differentiation between the *Rata* population and two populations on the main island (Harbor and *Quixaba*). This suggests the potential for eradication sustainability on Rata Island, as re-infestations are unlikely within a short period of time. Currently, there is a lack of information regarding the magnitude of *R. rattus* impact on *mabuyas*, marine and endemic birds, and other native species (such as turtles). Also, more studies are needed to determine efficient eradication strategies for rodents in the archipelago, and to estimate their real costs and benefits.

TEGU LIZARD

The tegu lizard (*Salvator merianae*) is the largest lizard in South America and as an omnivorous opportunist predator, tegu feed on fruits, insects and small vertebrates, but can also be observed scavenging on carcasses and human waste. Tegu will chase and hunt any smaller prey such as the mabuya, the threatened terrestrial crab *Johngarthia lagostoma* (H. Milne Edwards 1837), or the Noronha worm lizard. Birds eggs and hatchlings are also common items in the tegu diet (Bovendorp, Alvarez, and Galetti 2008, 9; Muscat, Olmos, and Rotenberg 2016, 36), and predation of green turtle (*Chelonia mydas* Linnaeus 1758) hatchlings has also been recorded on Fernando de Noronha (Figure 2), as tegus actively dig sea turtles' nests (Bellini and Sanches 1996, 12; Silva 2008).

Figure 2. Predation of the native green turtle (*Chelonia mydas*) hatchings by exotic/invasive tegu lizard (*Savator merianae*) on Fernando de Noronha archipelago, Brazil. Photography: Eliseu Souza Junior.

The first reliable record of tegu on Fernando de Noronha was in 1950, when *Alfredo Tito dos Santos* mentioned tegu as a hunting item in the island (Santos 1950, 1). Therefore, introduction of tegu occurred before 1950 but after 1888, when the naturalist Henry Ridley (1890) listed all animal species found on Fernando de Noronha, and tegu was not in the list. The reasons why and how tegu was introduced to Fernando de Noronha remains unclear, although there are anecdotes of its tentative use for controlling rats, and as a source of animal protein in case ships bringing food couldn't harbor on the island for longer periods of time. An interesting feature recorded for tegu lizards on Fernando de Noronha that may be associated with their abundance and distribution on the archipelago is their ability to voluntarily swim and even dive to short depths. Divers and local fisherman report events of tegu swimming both near and far from shoreline (Luiza Sampaio pers. comm.). The reasons for this behavior remain speculative. Presence of tegu on *Rata* Island has been recorded from indirect sign, and the presence of tegu on other islands of the archipelago is considered possible, but has not been confirmed, and might also be transient. *Rata* Island has been uninhabited since 1986. Nonetheless, even occasional tegu visits could be a threat to bird colonies nesting on the ground, such as both endangered tropicbird species.

Another aspect that deserves attention from stakeholders on the island is that tegu can be reservoirs of *Salmonella,* a bacterium with zoonotic potential that may represent a risk to children, elders and immunosuppressed people. *Salmonella* was detected in more than half of the anal swab samples collected from tegu of the main island, and in more than two thirds of the locations where tegu lizards were sampled, showing that the bacteria is well spread in the tegu population on the island. Further evaluations regarding *Salmonella* isolation in tegu samples will be available in the near future (C. Abrahão, unpubl. data).

A recent study on tegu density carried out on the main island of Fernando de Noronha estimated densities of three to five individuals per hectare in inhabited areas and from 10 to18 animals per hectare in uninhabited areas. Extrapolating this estimative to the whole island, it is possible that there is a tegu population of 7,000 to 12,000 individuals, which is the most abundant population ever recorded for this species. Densities in other studies are 0.2 animals/ha in continental Brazil and 0.83 animals/ha on *Anchieta* Island, off the eastern coast of Brazil (Bovendorp, Alvarez, and Galetti 2008, 9). A monthly census performed on Fernando de Noronha also revealed a decrease in tegu activity in the months of July and August, likely due to lower movement during the rainy (i.e., winter) season. A recent study describing general information about the tegu population on Fernando de Noronha, including potential control methods and predictions on population dynamics, is available (Abrahão 2019). This information is crucial for control programs to be applied to protect sensitive areas of the archipelago.

CURURU TOAD

The *cururu* toad (*Rhinella jimi*) (Figure 3A), was introduced to Fernando de Noronha about 100 years ago. There is no exact date of its introduction, but Forti et al. (2017) note the introduction occurred between 1888 (the last survey in the archipelago when toads were not encountered Ridley 1890) and 1973 (the first confirmed record of the species in the archipelago (Forti et al. 2017). In continental Brazil, the *cururu* toad has a generalist diet, ranging from insects to vertebrates (Oliveira et al. 2015, 19). On Fernando de Noronha, it has been shown that the toad's diet include endemic and endangered species from the archipelago, such as the gastropod *Hyperaulax ridleyi* (Smith 1890) (Freitas et al. 2020, 65, Figure 3B) and the *mabuya* (Figure 3C) both endemic to Fernando de Noronha, and the crustaceans *Ocypode quadrata* (Fabricius 1787) and *Johngarthia lagostoma* (Figure 3D) (M. S. Miranda and F. D. Passos, pers. comm.), the last endangered with extinction. Still, more specific studies that could identify the whole spectrum of the toad's diet, as well as quantify the proportion of each item, could greatly improve our understanding of the impacts of this introduced toad to the endemic and threatened native fauna.

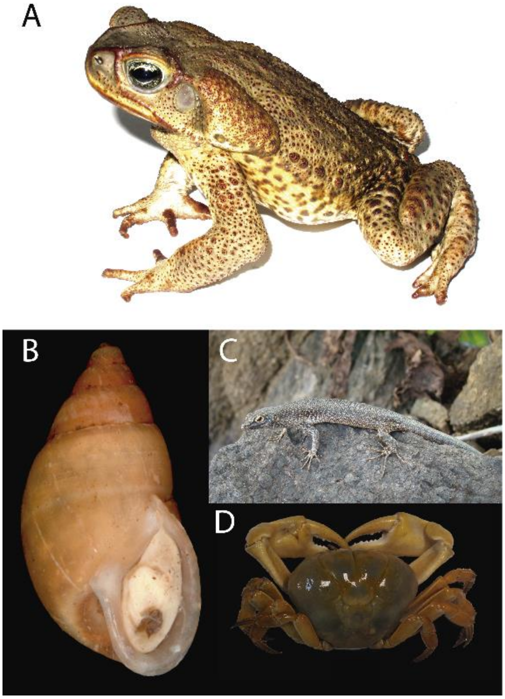


Figure 3. Adult male of the invasive *cururu* toad *Rhinella jimi* (a) and some of its (native) food items: the endemic snail *Hyperaulax ridleyi* (b), the endemic lizard *Trachylepsis atlantica* (c), and the endangered crab *Johangartia lagostoma* (d).

Besides the direct impact on local species by predation, the *cururu* toad can transmit parasites and pathogens to other animals, or even to humans. Among these pathogens, an important iridovirus that might be present in the island's population is the *Ranavirus*. This virus has recently been reported to occur in wild amphibians and fish in the Atlantic forest (Ruggeri et al. 2019, 897). Presumably, the presence of this virus in the *cururu* toad population could facilitate infections of Fernando de Noronha endemic species such as *Trachylepis atlantica* (Mausfeld et al. 2002, 281; Rocha et al. 2009, 450) and *Amphisbaena ridley* (Gans 1963, 102), and potentially be transferred to the marine environment.

A pathogen that was confirmed in Fernando de Noronha toad's population is a *Leptospira* sp., the causative agent of leptospirosis, which has also been seen in other species of *Rhinella* (Gravekamp et al. 1991, 403). In an analysis of 14 randomly sampled adult individuals from the main island of Fernando de Noronha, four were diagnosed with two serovars of *Leptospira* sp. (*Autumnalis* and *Bratislava* varieties) (G. C. P. da Silva, F. Gaviolli, pers. comm.). The infection of toads by *Leptospira* sp. can threaten the local human population by spillover infection (Hayman et al. 2013, 2; Jobbins, Sanderson, and Alexander 2014, 113). Currently, it is not possible to confirm if the population of *R. jimi* from Fernando de Noronha is a reservoir or a host population, but it breeds in areas where rats (*R. rattus* and *R. norvegicus*) and cats (*Felis catus*), possible hosts of this pathogen (J. C. R. Silva et al. 2017, 220), are present. As water reservoirs where *cururu* toads breed in Fernando de Noronha are used by human settlements across the main island, further analyses should be performed to verify the whole spectrum of pathogens that infect *cururu* toads, such as the zoonotic *Batrachochytrium dendrobatidis* (Longcore, Pessier & D.K. Nichols 1999) and *Ranavirus*, and the potentially zoonotic *Mycobacterium* sp. Studies that can explain the transmission of such viruses and bacteria from toads and water bodies to other species should also be performed.

An intriguing fact regarding the introduced population of *R. jimi* in Fernando de Noronha is the high incidence of malformations. About one in every two toads present morphological abnormalities of some sort (Toledo and Ribeiro 2009, 351; Tolledo and Toledo 2015, 167), and about 20% of the individuals present partial or total blindness (Tolledo and Toledo 2015, 351). Although Fernando de Noronha is among the sites with the highest prevalence of amphibian malformation on Earth (Toledo and Ribeiro 2009, 351), its causes remain unrevealed. Two main hypotheses have been suggested to explain the phenomenon: (i) inbreeding, a process related to small and isolated populations that reduces genetic variation, and/or (ii) environmental pollution, which could impact *cururu* toads' growth and body development (Toledo and Ribeiro 2009, 351). Although the inbreeding hypothesis was never tested for anurans, in Bermuda the high malformation prevalence in *Rhinella marina* (Linnaeus 1758) was associated with environmental pollution, especially from pesticides and heavy metals (Linzey et al. 2003, 125). Therefore, we suggest the need for morphological studies to improve the information on malformation cause; while genetic studies can provide information about the historical introduction of the population to Fernando de Noronha and test the inbreeding hypothesis, biochemical studies should test the environmental contamination as a possible link to toad's abnormalities. Knowledge acquired from such studies could improve local human, animal and environmental health.

EGRET

The cattle egret (*Bubulcus ibis* Linnaeus 1758) is a species with great capacity to invade and occupy new areas in different regions of the world (Nunes et al. 2010, 315). On the Fernando de Noronha archipelago, the cattle egret was first reported in the 1980s, a colonization apparently natural, with individuals arriving from the American continent by flight (Nunes et al. 2010, 316). Since its arrival, the cattle egret population has had a noticeable increase in size, which in the long-term, causes (i) ecological, (ii) microbial and (iii) social risks. Firstly, the cattle egret contributes to increase the risk of extinction of several native and endemic species due to competition for spatial resources (i.e., other bird species) and predation (i.e., mabuya) (Nunes et al. 2010, 316). This species also increases the risk of *Salmonella* infection, which has been indicated as a threat to wildlife conservation in general terms (Silva et al. 2018, 559), as well as toxoplasmosis (Abrahão 2019; Dubey and Jones 2008, 1257), both by maintaining the pathogens in the environment. Finally, they contribute to an increase in risk of human deaths on Fernando de Noronha due to collisions with aircrafts that depart and arrive every day on the island (Dolbeer, Wright, and Cleary 2000, 372; Sodhi 2002, 587). Monitoring and managing the cattle egret population would improve human safety and environmental health.

A recent article (Nunes et al. 2010, 317) presented historical data for the cattle egret population from 1986 until 2008. The study indicates that the population went from 655 individuals in 2005 to 298 individuals in 2008. Authors used the estimated annual increment rate (i.e., the percentage of increase in the cattle egret population) as the main indicator to evaluate the cattle egret population trend. These results suggest that this species' population is very sensitive to management measures. When no management was implemented (from 1986 to 2005), there was an increase of 35.2% per year in the population size. In the subsequent period (from 2005 to 2008) cattle egret individuals were captured and euthanized, and the population size decreased 41.5% per year. From 2008 until the current year, 2019, no management has been performed and the population abundance and trends are unknown.

As for future perspectives, a quantitative ecological and microbial risk assessment for the cattle egret on Fernando de Noronha, using the methodology proposed by Duarte et al. (2019, 10), may provide scientific support for management decisions. Such assessments are based on models that can describe the cattle egret population dynamics in the next 100 years under varying scenarios of control measures and human impact, generating useful information regarding its population dynamics for decision makers. These models are probabilistic by nature being able to account for variability in parameters. Also, it allows for the assessment of the risk of human deaths caused by airplane accidents as a function of cattle egret population abundance, which is useful to determine a threshold for the cattle egret population at which the risk is acceptable to aviation.

ROCK CAVY

The rock cavy (*Kerodon rupestris*) is a native continental Brazilian rodent that naturally inhabits the Caatinga (Brazilian dryland). It was intentionally introduced to the main island of Fernando de Noronha by the military to be used as a food resource (Schulz-Neto 1995). According to Oren (1984) and Alves and Leite (1992), between 1967 and 1969, four individuals (two males and two females) were captured on the continent and released on the island. Since then, the introduced rock cavy population grew on the island, benefitting from the presence of rocky outcrops — the species' natural habitat — widespread in the local landscape. Impacts as changes in endemic flora, dispersal of alien seed species, and the health risk of zoonotic and epizootic pathogens' transmission are allegedly attributed to this species (Alves and Leite 1992), even though no scientific studies have been performed or published to support that.

The rock cavy is a species with demonstrated hierarchical colonial structure, extremely adapted to and dependent on its environment. These colonies are restricted to quarries and natural rocky environments, depending highly on preexistent fractures in these rocky outcrops to guarantee its escape from predators (Adrian and Sachser 2011, 39; Souza Portella and Vieira 2016, 279; Lacher 1981). Most of the absence of information on the rock cavy population on Fernando de Noronha is likely due to considerably low capture rates for capture-recapture studies. On the continent, trapping success has been cited as far less than 1% (Lacher 1979, 67). The difficulties of capturing rock cavies reported in the past are still corroborated by more recent studies (Freitas, Rocha, and Simões-Lopes 2005, 119; Sousa and Menezes 2006, 443; Xavier et al. 2007, 119; Zappes, Portella, and Lessa 2014, 251; Delciellos 2016, 1916).

Rock cavy population size has been estimated as 5,473 individuals with 95% confidence interval ranging from 3,114 to 9,622 when using a

Poisson-log normal mark-resight zero truncated model (McClintock et al. 2009, 237) applied to one rock cavy colony and extrapolated to the whole island of Fernando de Noronha (Micheletti, Mangini, and Gasparotto in press). Using a concurrent marking-observation hierarchical integrated population model, however, the population size was calculated as 6,652 ± 1,587 individuals (T. Micheletti, unpubl. data). Micheletti et al. (unpubl. data), also investigated possible management options for the rock cavy on the island, but concluded that none of the management options tested — (i) removal, resulting from trapping and euthanasia or translocation for both sexes, as well as just the (ii) removal of females, and the (iii) removal of males; ovariohysterectomy or bilateral orchiectomy would be performed in females and males, respectively in (iv) spaying and (v) castration scenarios, and a combined scenario investigating the effects of (vi) neutering both males and females — were efficient to eradicate the rock cavy from the island. Management interventions used a harvest effort of 120 traps per day, per year, lasting 10 years. Therefore, it is recommended that an impact assessment of the rock cavy is performed, with a follow-up cost benefit analysis of management interventions.

INTRODUCED ANTS

Very little is known about the ants of Fernando de Noronha, but as well as native species from the genus *Pseudomyrmex* a number of tramp ant species are known from the island, including *Pheidole megacephala* (Fabricius 1793), *Paratrechina longicornis* (Latreille 1802), *Tetramorium simillimum* (Smith 1851), *Tetramorium bicarinatum* (Nylander 1846), and *Tapinoma melanocephalum* (Fabricius 1703) (Benoit Guénard pers. comm.). Throughout the PARNAMAR national park area a recently introduced biting ant has become highly abundant and is commonly known by locals as "*cafifa*". In 2017 the species was identified as electric or little fire ant (*Wasmannia auropunctata*) (Benoit Guénard pers. comm.). Locals describe it as a recent introduction, possibly introduced in nursery stock from continental Brazil where it is native. The sting of the ant is a recurrent human nuisance but the electric ant is likely to be having strong negative impacts on all native animal species due to its aggressive and swarming behavior (Yitbarek, Perfecto, and Vandermeer 2017, 4). Vulnerable species include nesting land and sea birds and their chicks, nesting turtles and their hatchlings, small reptiles and invertebrates. Management of the electric ant on Fernando de Noronha will be very difficult and likely only possible over very small areas of high biodiversity value. Further research is needed to assess the distribution and abundance of the electric ant across the main and secondary islands of Fernando de Noronha, and determination of impacts by comparing ant invaded and ant free areas, or across ant density gradients. Further surveys of ant biodiversity on Fernando de Noronha are also required, as additional native and introduced species likely remain undocumented, and some introduced species may behave more invasively in future. Prevention through robust border biosecurity of further ant and other invertebrate introductions from continental Brazil should be implemented.

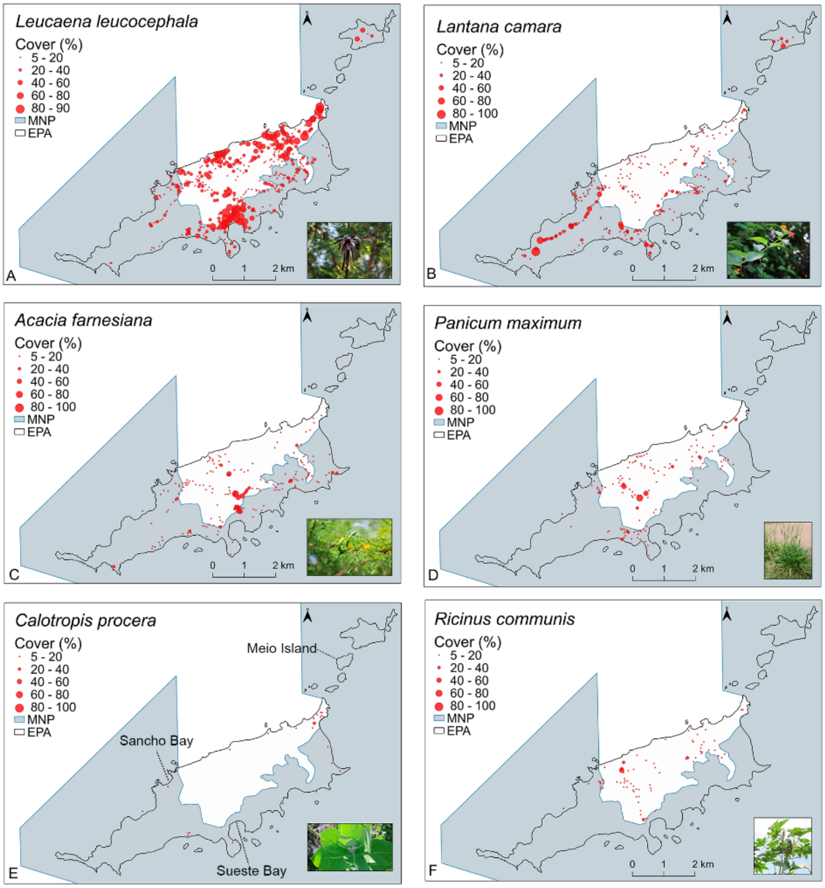
INVASIVE FLORA

Most of the main invasive plant species on Fernando de Noronha were introduced intentionally, for various purposes such as human food use (e.g., *Terminalia catappa* L.*)*, animal forage production (e.g., *Leucaena leucocephala),* shading (e.g., *Azadirachta indica* A. Juss.) and ornamental use (e.g., *Calotropis procera* (Aiton Dryand)). The first vegetation survey dates back to the 19th century (Ridley 1890a, 12), and already reported the large numbers of weeds that were introduced by humans to the archipelago. The main negative consequence of plant invasion is the reduction of available habitats for native species, which is especially problematic on isolated islands with a small area like Fernando de Noronha. The invaded areas on Fernando de Noronha tend to become homogenized, with species richness reducing significantly (Mello and Oliveira 2016). Therefore, despite the current protection status of the archipelago, vegetation conservation and regeneration are hindered by the presence of invasive alien species (Mello and Oliveira 2016). Few studies have investigated the mechanism behind these impacts (Chou and Kuo 1986, 1431; Hierro and Callaway 2003, 29; Rizvi et al. 1999, 773; Prasad and Subhashini 1994, 1689), but it has been experimentally shown that the exotic legume *L. leucocephala* can affect the balance of interactions between native plants (Mello and Oliveira 2016), making natural regeneration even more difficult.

A comparison between vegetation surveys conducted by Ridley (1890) and Freitas (2007) demonstrated the disappearance of species, probably due to habitat loss and competition with invasive species. One example is the liana *Combretum rupicola*, which is endemic to Fernando de Noronha (Ridley 1890a, 16) and was last found represented by a single individual collected at *Morro do Francês* in 2007 (AFreitas 2007). In 2018 an expedition was carried out to try to find the specimen, but the rocky region where it was previously recorded was covered with vines and clusters of the invasive alien species *L. leucocephala* and *Lantana camara* (L.) (M. G. Freitas, pers. obs.). The archipelago also harbors the only oceanic mangrove habitat in the South Atlantic region (Claudino-Sales 2019, 217), with just over one hectare in area in the *Sueste* Bay. This ecosystem is highly invaded by exotic species, especially *L. leucocephala*.

Lianas and vines cover large areas on the archipelago and seem to be more common on the edges of fragments. They pose a significant threat to the ecosystem, as they overgrow trees, killing them by suffocation or breaking the branches due to weight (T. J. Mello, pers. obs.). One of these liana species, *Cissus verticillate* ((L.) Nicolson & C. E. Jarvis), covers important native trees like *Erythrina velutina*, commonly used for seabird nesting, possibly resulting in reduced availability of suitable branches for these species to nest. Despite the potential negative effects of the invasive flora on both native flora and fauna, the impacts of the invasive plants remain unknown, and management strategies for invasive lianas and vines are still required.

Currently, 17 invasive exotic plant species have been recorded on Fernando de Noronha (Table 1). However, research on this subject is still rare on the archipelago, especially for the secondary islands and herbaceous vegetation, lianas and vines. In a survey conducted on the archipelago in 2018, 14 invasive alien species were sampled at 586 points, including trees, shrubs, herbs and grasses (Freitas and Mello 2018)). The most abundant invasive alien plant on Fernando de Noronha is *L. leucocephala* (Figure 4A), followed by *L. camara* (Figure 4B), and *Acacia farnesiana* ((L.) Willd.) (Figure 4C).



Source: Freitas and Mello (2018).

Figure 4. Estimated surface cover (%) by exotic invasive plant species on Fernando de Noronha Archipelago, Brazil: *Leucaena leucocephala* (a); *Lantana camara* (b); *Acacia farnesiana* (c), *Panicum maximum* (d); *Calotropis procera* (e), and *Ricinus communis* (f). MNP = Marine National Park; EPA = Environmental Protected Area.

Table 1. Invasive alien species recorded in Fernando de Noronha, PE, Brazil (M.G.R. Freitas, personal observation)

Species	**Family**	**Life form**
Acacia farnesiana (L.) Willd.	Fabaceae	Shrub
Arundo donax L.	Poaceae	Grass
Azadirachta indica A. Juss.	Meliaceae	Tree
Calotropis procera (Aiton) R. Br.	Apocynaceae	Shrub
Cissus verticillata (L.) Nicolson & C.E. Jarvis	Vitaceae	Liana
Crotalaria retusa L.	Fabaceae	Shrub
Euphorbia tirucalli L.	Euphorbiaceae	Shrub
Jatropha gossypiifolia L.	Euphorbiaceae	Shrub
Lantana camara L.	Verbenaceae	Shrub
Leucaena leucocephala (Lam.) de Wit	Fabaceae	Tree
Melia azedarach L.	Meliaceae	Tree
Neomarica candida (Hassl.) Sprague	Iridaceae	Shrub
Panicum maximum Jacq.	Poaceae	Grass
Pennisetum setaceum (Forssk.) Chiov.	Poaceae	Grass
Ricinus communis L.	Euphorbiaceae	Shrub
Terminalia catappa L.	Combretaceae	Tree
Tithonia diversifolia (Hemsl.) A. Gray	Asteraceae	Shrub

Leucaena leucocephala is an aggressive invader listed as one of the 100 World's Worst Invasive Alien Species (GISD 2019b). With rapid growth, nitrogen assimilation, drought and salinity tolerance, it forms dense stands with a continuous canopy that can be monospecific or contain only a few other species (Mello 2013). On the main island of Fernando de Noronha the species is widely distributed, densely covering most of the places where it occurs (Mello and Oliveira 2016). Chemical and mechanical control techniques were locally tested, and trunk cutting with chainsaw combined with triclopyr herbicide application was the most efficient technique, considering mortality rate and cost (i.e., time and labor) (Raimundo-Junior, Freitas, and Mello 2018, 118; Silva et al. 2018, 559). Cutting the trees without the application of herbicide was ineffective as trees would vigorously resprout. It was concluded that all the root mass must be removed in order to successfully eliminate the specimen. Uprooting is suggested for small areas at the beginning of the invasion, but it is unfeasible on a large scale as the effort needed is around five times higher than cutting and adding herbicide.

Lantana camara was introduced in Fernando de Noronha for ornamental use in domestic and urban gardens. It has high occupation and dominance on the main island, and is often found near trails and areas formerly occupied by agriculture and livestock. In addition, it is found on the secondary *Meio* Island in large numbers (Figure 4B). It forms extensive, dense and impenetrable stands. The species is dominant in the understory and appears to hinder natural regeneration and alter the fire regime (GISD 2019a). However, no research was conducted to evaluate the impacts and how to control this species on the archipelago.

Although not very abundant, some of the invasive exotic species may be in a latent or initial phase of invasion and become problematic if not managed promptly. This is the case of the grass *Panicum maximum* (Figure 4D) and the bush *Calotropis procera* (Figure 4E), which have their negative impact described worldwide. *Calotropis procera* is an invader of arid and semi-arid regions of Africa and Asia that has a negative impact described worldwide. The species is found on Fernando de Noronha mainly on coastal and steep rocky areas, in small populations. It can compromise the local environment, disrupting vegetation and turtle nesting grounds (M. G. Freitas, pers. obs.). Mechanical management is efficient for the control of the species only with the total removal of the individuals (Raimundo-Junior, Freitas, and Mello 2018, 118). Considering its initial phase and high invasion potential, this is a priority species for control. In 2018 PARNAMAR managers conducted a control campaign and all the 120 individuals previously mapped were uprooted and their fruits were collected and destroyed (Silva, 2018). To prevent the establishment of new individuals it is important to monitor the areas susceptible to invasion.

Disturbed areas are usually more prone to invasion (Connell and Slatyer 1977, 1119; Lockwood, Cassey, and Blackburn 2009, 904). Native insular species, in contrast, tend to grow slowly and produce fewer seeds. One species that seems to benefit from the high disturbance level on the archipelago is *Ricinus communis* (L.) (Figure 4F). It is mainly found in the Environmental Protected Area (the part of the archipelago where human use and settlement is permitted), near waste land and in association with watercourses, such as sewage outlets and rainfall runoff areas. Invasion in the PARNAMAR is in an initial phase, and is currently happening in the Sancho Bay region, where young isolated individuals are found by an intermittent watercourse, which probably carries seeds from upstream invaded areas. Another example is the herbaceous *Crotalaria retusa* (L.). It has the potential to displace native vegetation and alter soil chemical conditions due to its nitrogen-fixing nature (Database 2019). On Fernando de Noronha the species is found invading coastal zones on sandbanks and near rocks, surrounding the main island. It is also found along trails even in the most remote areas of the PARNAMAR, such as the *Capim Açu* region, where it probably arrived carried in boots or in lawn mowers used for the maintenance of the trails.

CONCLUSION

Fernando de Noronha contains a curious combination of typical invasive species (e.g., cats and rats) as well as species native to continental Brazil but introduced and behaving invasively on the archipelago (e.g., tegu, rock cavy and electric ant) Until recently, the geographic isolation of Fernando de Noronha from research centers, in combination with elevated operational costs in conducting research in this remote location, have prevented study of Fernando de Noronha’s invasive species. However, established partnerships among governmental executive bodies, NGO’s, research institutes and universities over the last decade or so have been promoting the development of research and management of invasive species on the archipelago.

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Chapter 4

BIOLOGICAL INVASION BY EXOTIC PLANTS IN URBAN FOREST FRAGMENTS OF NITERÓI, RIO DE JANEIRO, SOUTHEASTERN BRAZIL

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ABSTRACT

Biological invasion is the process of introducing and adapting species which are not naturally part of an ecosystem but which produce

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viable offspring, propagating at significant distances from the mother plant and causing changes in the environment. In this chapter, an inventory of exotic species in forest fragments of the municipality of *Niterói*, Rio de Janeiro state, and an evaluation of the biological invasion process by exotic plants are presented. Six forest remnants located in protected areas were used as the study objects. These urban forest remnants belong to the Atlantic Forest domain. We used the walking method to identify species, in which we traversed the trails and streets around the areas and then listed the exotic species present. The biological invasion process by exotic plants was evaluated through the line intersection method. The general survey revealed a total of 143 exotic species, distributed in 121 genera and 55 families. The richest families were Leguminosae (13 spp.), Asparagaceae (10 spp.), Poaceae (9 spp.), Acanthaceae, Araceae and Lamiaceae (6 spp. each), Apocynaceae, Euphorbiaceae, Malvaceae and Moraceae (5 spp. each), which corresponded to 48.9% of the inventoried material. Regarding the biological invasion evaluation in conservation units, eight species were found in *Morro das Andorinhas* and 25 in *Vale do Córrego dos Colibris*. Considering the biological invasion stages, three exotic species were indicated as established species in *Morro das Andorinhas* and presented invasive characteristics (*Furcraea foetida*, *Megathyrsus maximus,* and *Phyllostachys aurea*), while four species stood out as invasive in *Vale do Córrego dos Colibris*, (*Dieffenbachia seguine, Epipremnum pinnatum, Syngonium podophyllum* and *Tradescantia zebrina*). As exotic species are the main cause of biological diversity loss in conservation units, their presence in the study areas and other areas around the world is incompatible with conservation purposes. Monitoring and permanent control of these species need to be included as priority actions in management plans of protected areas. The results also revealed the historical use of exotic species by human population in the surrounding areas. Residents have grown exotic plants close to protected forests and such species are invading the protected areas. Therefore, this study also highlights the importance of implementing formal and non-formal educational information and awareness activities for populations surrounding protected areas worldwide.

Keywords: Atlantic forest, biological invasion, exotic plants, protected areas

INTRODUCTION

Exotic plant species are species introduced into an environment which does not correspond to their original distribution. Depending on where exotic plants are allocated and the environmental conditions, they can reproduce spontaneously, maintain a viable population and disperse to areas far from their original point of introduction, establishing locally (Pivello 2011, 1). From there they can cause biological invasion, which is the process of a species introducing and adapting itself where they are not naturally part of the ecosystem, but where they can become naturalized, thereby causing changes in ecosystem functioning (Ziller and Galvão 2002, 41). This process is verified both between different countries and between heterogeneous areas within a country, with the potential to become invasive (Ziller and Dechoum 2007, 356).

The introduction of a species is motivated by social, economic and environmental issues (Leão et al. 2011, 10). Exotic plant species can be used in squares, gardens, urban centers, agriculture, biological control, food and urban afforestation. However, in addition to the biological diversity devaluation in the municipalities, this maneuver acts as a radiant nucleus for biological invasions and is considered one of the main causes of homogenization of vegetation on a global scale (Mckinney 2006, 247-248; Dehnen-Schmutz et al. 2007, 527-528).

Established species go through several stages until they can be considered invasive. In the introduction phase, species leave their natural range and reach a new environment; in the establishment stage, species adapt to the new environment and begin to reproduce, ensuring the sustainability of their populations and begin the invasion process when dispersal mechanisms allow them to expand beyond the point of introduction and to dominate native species (Williamson 1996, 28-52). Based on these steps, Pyšek et al. (2004, 135-136) proposed a nomenclature with clear definitions for the bioinvasion field. This terminology was then taken up by Moro et al. (2012, 993-995), recognizing the importance of adjusting the classification used in Brazil with the international literature and warned of how these species are treated in taxonomic, floristic and phytosociological studies. However, this classification is not purely nomenclatural and must be carefully elaborated for each local reality, as many exotic species can be invasive in some regions and have little or no negative impact on others, even within the same country. From the perspective of managing nature conservation units, this careful assessment is necessary to optimize resource allocation and focus the efforts on those species which have the greatest invasiveness and potential for modifying natural ecosystems (Durigan et al. 2013, 25-27).

Biological invasion should be more actively addressed in conservation units, where the introduction of exotic species is prohibited. These areas are legally instituted by the government to ensure that they are properly conserved within their territorial limits, as provided for the Federal Law no. 9.985/2000, which established the National System of Conservation Units in Brazil (Brasil 2000). Unlike other forms of environmental degradation, biological invasions tend to grow indefinitely over time, mainly due to increased propagule pressure in these areas, as well as fragmentation and anthropization around conservation units (Leão et al. 2011, 15).

A reduction in species diversity, hybrid generation, native species substitution, changes in habitat, competition, predation, parasitism, and consequent changes in the food chain and nutrient cycle are among the various impacts of bioinvasions (Pimentel 2011, 1-2; Sampaio and Schmidt 2013, 35). Petenon (2006) emphasize that such impacts are facilitated by some characteristics which favor the biological invasion process in plants such as accelerated reproduction due to the absence of predators; competition with native species for local resources; optimal colonization, especially in disturbed environments; a short juvenile period and interval between large seed output; as well as benefiting from the pollinators and dispersers of the invaded sites.

Exotic species are one of the world’s major environmental problems (Pimentel 2011, 1-2) and constitute the second leading cause of biological diversity loss in Brazil, ranking first in protected areas and island environments (Leão et al. 2011, 15-18). The presence of this threat is incompatible with the conservation of natural resources (Leão et al. 2011, 15). Surveillance to prevent biological invasion, eradication of exotic species out of their natural environments and their control are priorities which are reinforced in the goals set by the Global Plant Conservation Strategy (GSPC) (Sharrock et al. 2018, 1502-1503). Goal 10 of this document states that effective management plans should be implemented in protected areas (PA) by 2020 to prevent further biological invasions and manage the invaded ones (Dechoum et al. 2018, 1568).

Biological invasion in Brazil has been seriously faced, but much greater efforts are needed to identify such invasions. This theme has particularly been addressed in the state of Rio de Janeiro by several authors in the last decade (e.g., Siqueira (2005; 2006); Gomes (2007); Ribeiro and Zaú (2007); Raíces et al. (2008; 2017); Barroso (2009); Abreu; Rodrigues (2010); Ribas et al. (2010); Moura (2011); Cruz et al. (2013); Ferreira (2013); Nascimento et al. (2013); Santos (2013); Zucaratto and Pires (2014); Freitas et al. (2015; 2017); Magalhães et al. (2015); Mello et al. (2015); Bergallo et al. (2016); Faria (2016); Zimmermann et al. (2016; 2017) and Solórzano et al. (2018)).

In this context, the objectives of this chapter are to list the exotic plant species present in forest fragments in the municipality of *Niterói* with an indication of their forms, geographical origin and uses of the species, and to structurally analyze if exotic species in two areas of the *Serra da Tiririca* State Park (Rio de Janeiro state) are causing biological invasion. In addition, we aimed to know if part of the inventoried exotic flora was used in urban afforestation in different locations in Rio de Janeiro.

METHODS

Study Sites

The municipality of *Niterói* is located in the state of Rio de Janeiro (22º53’ – 23º30’ S, 43º07’ – 42º20’ W), southeastern Brazil. *Niterói* has an area of 133,919 km², and is about 13 km from the state capital city of Rio de Janeiro. Niterói is the fifth most populous municipality in the state and the 39th in the country, with about 487,562 inhabitants (IBGE 2010). About 33% of its territory is protected in the form of conservation units (*Prefeitura Municipal de Niterói* 2018, 4), and the main remaining forests are concentrated in six places. Three of them represent the largest stretches of continuous forest, namely *Morro da Viração* Complex, the Darcy Ribeiro Ecological Reserve, and *Serra da Tiririca*. Another three areas are isolated in the urban matrix at the *Mata do Ministro*, the inselbergues of *Morro da Peça* and *Morro das Andorinhas* (Figure 1).

These forest remnants are inserted in the Atlantic Forest domain and represent the following phytophysiognomies: Submontane Dense Ombrophilous (Broadleaf) Forest, Lowlands, and rocky outcrop vegetation, according to Veloso et al. (1991, 63-67), and updated by the IBGE (2012, 64-66). These forest fragments present rocks which are around 600 million years old, comprised in the facoid gneiss and Cassorotiba geological units (Penha 1999, 4-6), with the presence of migmatites, granitoid gneiss and basaltic dikes (Cunha and Francisco 1981, 16-18).

The main soils found in these areas are yellow argisol, red-yellow argisol and litholic neosol. *Serra da Tiririca* also registered chromic alisol, hypochromic luvisol and organosols (Barros 2008, 27). In the *Morro da Viração* Complex, lithosols (hillside areas), red-yellow, hydromorphic (lowland), podzol-hydromorphic, and turfo-gleized near river or canal mouths are indicated (Barroso et al. 1994, 189).

The climate of the region is influenced by the presence of Continental Equatorial air masses in summer and Tropical Atlantic during the remainder of the year, as well as the frequent passage of polar fronts, especially during the spring. The climate is framed in the climactic zone of coastal plains and massifs with average rainfall between 1,000 and 1,500 mm/year (Barbiére and Coe-Neto 1999, 50). The area of the present study is part of the rainfall transition between the coastal plain and coastal massif. The climate of the region is *Aw* according to the Köppen-Geiger classification, viz, hot and humid, with a rainy season in the summer and dry in the winter (May and June). The average temperature is around 23 ºC, with January and February being the warmest months, and June the coldest month. The predominant wind is from the northeast (Bernardes, 1952, 61-75).

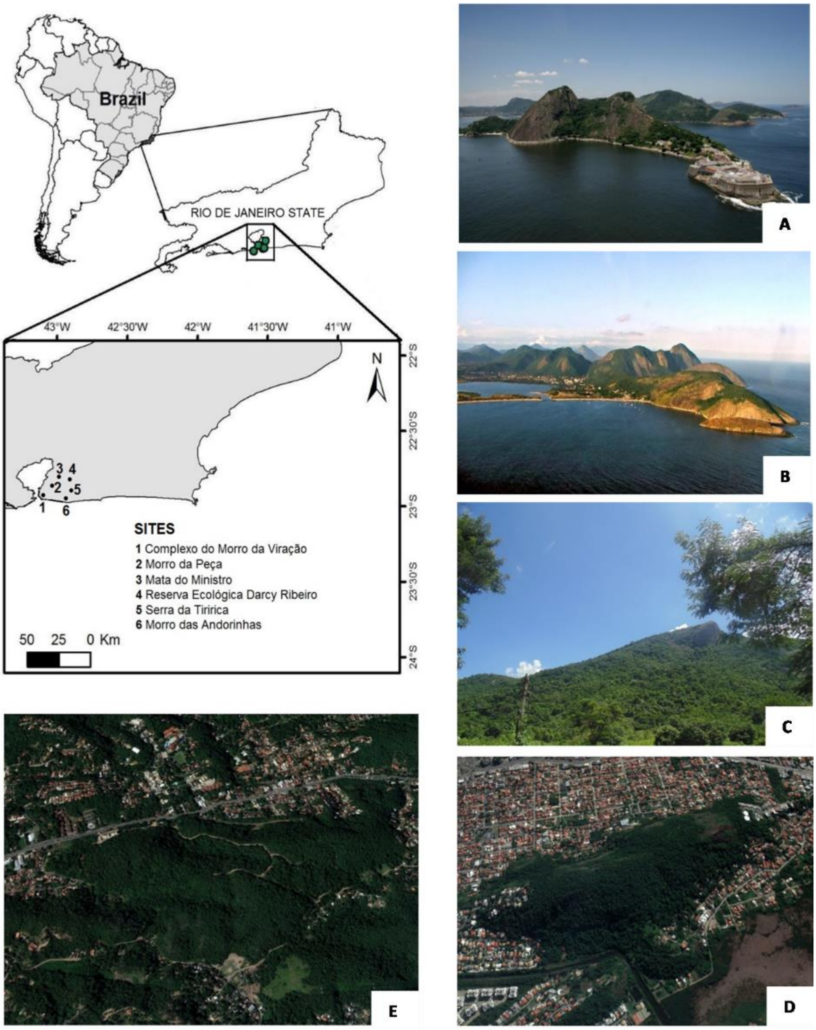

Figure 1. Location of the six areas of continuous forest in the municipality of *Niterói*, Rio de Janeiro state, Brazil. (a) *Morro da Viração* Complex, at the entrance to *Guanabara* Bay; (b) *Morro das Andorinhas* (in the foreground) and *Serra da Tiririca*, seen from *Itaipu* Beach; (c) *Darcy Ribeiro* Ecological Reserve, on the slope of the *Serra do Cantagalo*; (d) Aerial view of the *Morro da Piece*; and (e) *Mata do Ministro*. Source: (a) Rio Media Center (2017); (b) *PESET* (2016); (c) D.N.S. Machado (2015); (d, e) Google Earth (2018).

Characterization of the Six Forest Remnants

The *Viração* Complex (Figure 1a) is a set formed by the *Parque da Cidade* (256 m) and *Imbuí* (129 m), *Pico* (232 m), *Preventório* (267 m), *Sapezal* (303 m), *Viração* (319 m), and *Santo Inácio* (348 m) hills, whose slopes face the *Niterói* neighborhoods of *Jurujuba*, *São Francisco*, *Cachoeira*, *Charitas*, *Maceió*, *Cafubá*, and *Piratininga*. The "*Viração*" denomination comes from the observation of the wind currents which reach this hill made by fishermen indicating the "turn of time" (Pacheco et al. 2016, 229). This area is covered by municipal Protected Areas: the *Morro do Morcego*, *Santa Cruz* Fortress, and *Pico* and *Rio Branco* Forts Environmental Protection Area (APA), and the *Niterói* Municipal Natural Park (PARNIT) (Prefeitura Municipal de Niterói 2018, 39-40). Historic fortifications were installed in the Imbuí and Pico hills to guard the entrance of *Guanabara* Bay against possible foreign invasions from the 16th to the 19th centuries, being integrated by the Santa Cruz Fortress and *Barão do Rio Branco*, *Imbuí*, *Pico*, *São Luís*, and *Tamandaré da Laje* Forts (Barros et al. 2016, 883).

Serra da Tiririca (Figure 1b) is formed by a set of seven hills: *Penha* (128 m), *Costa de Itacoatiara* (217 m), *Cordovil* (256 m), *Serrinha* (277 m), *Catumbi* (344 m), *Telégrafo* (387 m), and *Alto Mourão* (412 m). The *Serra da Tiririca* also covers part of the municipality of *Maricá*, but most of its area is in *Niterói*, specifically in the *Engenho do Mato*, *Itaipu*, *Itacoatiara*, and *Várzea das Moças* neighborhoods. The name of this mountain (*Serra da Tiririca*) is related to the passage of troops who crossed it on donkeys towards the Lakes Region through a path full of herbs of the *Cyperus* genus (Cyperaceae), popularly known in Brazil as "*tiririca*" (Pontes 1987, 90; Barros and Pimentel 2010, 799). *Serra da Tiririca* is part of the *Serra da Tiririca* State Park (*PESET*), which was created by State Law no. 1901/1991 (Rio de Janeiro 1991). *PESET* underwent three extensions (Rio de Janeiro 2007; 2008; 2012), and is divided in four sectors: Darcy Ribeiro, *Insular*, *Lagunar*, and *Serra da Tiririca*. *PESET* was established in the publication of the management plan (INEA 2015a, 9).

Morro das Andorinhas (196 m) (Figure 1b) is isolated from the *Serra da Tiririca* and represents a natural border between the Itaipu and *Itacoatiara* neighborhoods. Its name refers to a rocky area situated where swallows seek shelter (Leite 2003, 8). Like other land and island elevations of *Niterói*, *Morro das Andorinhas* is an inselberg (insel = island; berg = mountain); rocky outcrops in normally monolithic granite or gneiss mountains originating from the Precambrian period (Porembski et al., 2007, 579-580). They ecologically function as terrestrial islands with exposed rock surrounded by shrubby vegetation (Conceição et al. 2007, 335-336). *Morro das Andorinhas* also integrates *PESET* through State Law no. 5079/2007 (Rio de Janeiro 2007), composing the *Lagunar* sector (*INEA* 2015a, 9).

The *Córrego dos Colibris* is a valley in *Morro do Telégrafo,* located in the *Serra da Tiririca* sector of *PESET*. *Córrego dos Colibris* comprises the Lowland and Submontane Dense Ombrophilous Forest and features rocky outcrops. It is an area which is very close to urban occupation, where there are easily accessible trails and is very popular for ecotourism activities, climbing, environmental education, leisure, and a sacred space for Brazilian religions of African origin (INEA 2015b, 168-181).

The old *Darcy Ribeiro* Ecological Reserve (Figure 1c) encompasses the *Serra do Jacaré* (231 m), *Malheiro* (264 m), Grande (384 m), and *Cantagalo* (407 m) areas. It covers the neighborhoods of *Itaipu*, *Jacaré*, *Piratininga*, *Rio do Ouro*, and *Várzea das Moças*. The Reserve was created in 1997 by the Municipal Law no. 1,566 (Niterói 1997), and its name honors the Brazilian anthropologist Darcy Ribeiro, who died in the same year as the Reserve creation. Darcy Ribeiro Ecological Reserve represents an important region, as it has water sources in the Oceanic hydrographic macrobasin and some of the main rivers of *Niterói* are born there (*Jacaré* river, *Arrozal*, *Santo Antônio* stream, *Malheiro* stream, Muriqui, *Sapê*, and *Pendotiba*) (INEA 2015a, 102). The Ecological Reserve has been an integral part of *PESET* (Rio de Janeiro 2012) since October 2012, currently composing the namesake sector (*INEA* 2015a, 9).

Morro da Peça is also an inselberg isolated from the three presented forest sections with an altitude of 117 m. It is located between the *Santo Antônio* and *Itaipu* neighborhoods (Figure 1d). The toponymy of this area is uncertain, although some versions mention the presence of a piece of artillery (cannon), which existed on this elevation summit in colonial Brazil (INEA 2015b, 248). Morro da Peça was included with the Darcy Ribeiro Ecological Reserve in *PESET* and expanded the area of the *PESET* by 38 ha (Rio de Janeiro 2012), currently composing the Lagunar sector (INEA 2015a, 9).

Mata do Ministro is a forested area which is also isolated from the three forest sections presented, with an altitude of 112 m and is located in the Matapaca neighborhood (Figure 1e). This forest fragment belonged to the former *Maria Paula* Farm, which existed in the Pendotiba Region. It is inserted in the Environmental Protection Area of *Niterói* Lagoon and Forests, and in the Area of Special Environmental Interest *Fazendinha* (Soares et al. 2019, 1).

Data Collection

Inventory of Exotic Plants in Forest Remnants

Exotic plant species were inventoried in field trips carried out from 2011 to 2019 in the areas of *Morro da Viração* Complex, *Morro da Peça*, *Mata do Ministro*, Darcy Ribeiro Ecological Reserve, *Serra da Tiririca*, and *Morro das Andorinhas*. The walking method proposed by Filgueiras et al. (1994) was used for the floristic inventory. Fertile material was collected and herborized according to usual botanical techniques (Peixoto and Maia 2013). The species were identified through specialized bibliography and consultations in herbariums of the *Faculdade de Formação de Professores (RFFP)* [Faculty Teacher Training] from the *Universidade do Estado do Rio de Janeiro (UERJ)* [Rio de Janeiro State University] and the Rio de Janeiro Botanical Garden (RB).

Table 1. Studies carried out in protected areas in the state of Rio de Janeiro used for data comparison in this chapter

Location	**Protected Area**	**City***	**Species richness (n)**	**Reference**
PUC/RJ *campus*	*Tijuca* National Park surroundings	2	23	Siqueira (2006)
Morro do Forte do Pico	*Morro do Morcego*, the Santa Cruz Fortress and the *Pico* and *Rio Branco* Forts Environmental Protection Area	1	10	Paes (2013)
Serra da Tiririca	*Serra da Tiririca* State Park	1	9	Pontes (1998); Barros (2008); Andreata et al. (2008).
Morro das Andorinhas	*Serra da Tiririca* State Park	1	79	Machado et al. (2020)
Pedra da Gávea	*Tijuca* National Park	2	9	Carauta and Valente (1974)
Cara de Cão, Pão de Açúcar, and *Urca*	Municipal Natural Park of the *Carioca* Landscape	2	5	Carauta and Oliveira (1984)
Cagarras Islands	*Cagarras* Islands Natural Monument	2	2	Bovini et al. (2014)
Estrada da Barragem	*Serra dos Órgãos* National Park	3	2	Barroso (2009)

* Cities: 1 = Niterói; 2 = Rio de Janeiro; and 3 = Teresópolis.

The floristic list was organized according to APG IV (2016) for Angiosperms and Judd et al. (2008) for Gymnosperms, except for Leguminosae which followed the LPWG proposal (2017). The spelling of species names was confirmed through the International Plant Names Index (IPNI 2019). The control material was incorporated into the *RFFP* herbarium, with duplicates in the RB, which were also consulted to complement the species list. Data of geographical origin as well as form of use were based on literature (Lorenzi and Souza 2001; Lorenzi et al. 2003; Lorenzi; Matos 2008).

The species were classified according to their life forms as herbaceous, shrubby, arboreal, climbing, hemiepiphytic and epiphytic (Lima and Guedes-Bruni 1997, 315). The forms of use were grouped into eight categories: 1) Food: when part of the plant is used for human consumption; 2) Foraging/Pasture: when some part of the plant is intended for feeding livestock; 3) Hedgerow: when used to delimit spaces; 4) Ornamental: when the plant is grown for ornamentation; 5) Medicinal: when some part of the plant has medicinal use; 6) Ritualistic: when it has liturgical use in some religion; 7) Shading: whose crown provides shade; 8) Recompositing: plants used in the restoration of pastures, degraded and/or eroded areas.

The results were compared with other studies which inventoried exotic plants in CUs in the state of Rio de Janeiro and also focused on urban afforestation (Table 1). Thus, it was sought to understand whether part of this inventoried species richness was common to the species richness traditionally used in afforestation/landscaping projects.

Structural Analysis of Exotic Plants

The structural analysis of the biological invasion was carried out in two locations of the *PESET*, namely *Morro das Andorinhas* and *Vale do Córrego dos Colibris* (*Morro do Telégrafo*). These locations were chosen because they are inserted in urban areas and have different records of anthropic interventions over the years. The biological invasion in *Morro das Andorinhas* was evaluated by the transect-point and intercept-line techniques. The transect point has been used on open trails and enables measuring exotic species over long distances (Ribas et al. 2010, 657; Bergallo et al. 2016, 192). The line intercept method was used to assess which exotic plants were expanding from the main trails and how much area they were occupying in the forest (Richardson et al. 2000, 98). These two techniques are complementary and enable measuring the sampled individuals, regardless of their form.

The *Trilha das Andorinhas* (main trail) was traveled with the aid of a Garmim Map 60CSX GPS, and the presence of exotic plants was recorded at horizontal intervals of 100 m in two circular portions of five meters on either side of the trail. This trail is 2.6 km long, and 19 sample points were marked (Figure 2). The sinuous trail and its altimetric variation had an influence on the number of points sampled, since GPS marking does not follow these contour variations. Each species coverage values were estimated using the Braun-Blanquet cover-abundance scales (Mueller-Dombois and Ellenberg 1974, 62-63), adapted for the present study. Five classes of coverage were considered: C1 (10%), C2 (25%), C3 (50%), C4 (75%) and C5 (100%). The absolute frequency and coverage were calculated in percentage from the recorded data.

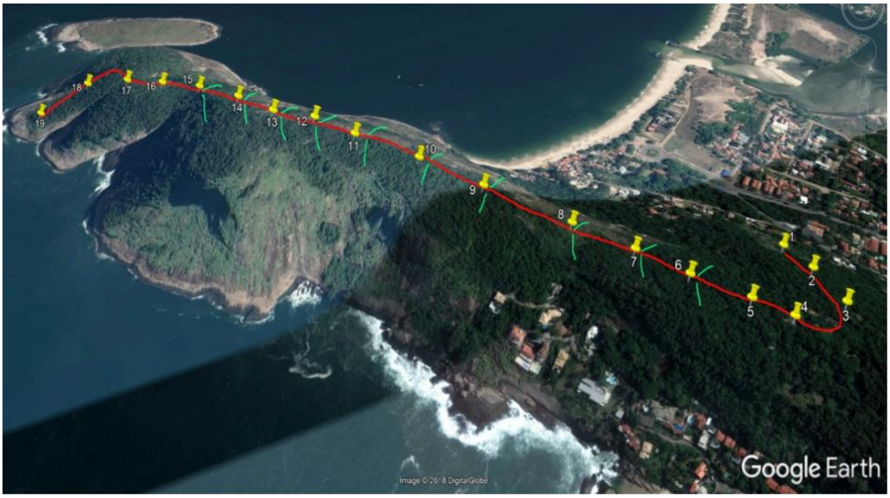

Figure 2. *Morro das Andorinhas* trail (red continuous line) in the municipality of *Niterói*, Rio de Janeiro, Brazil. The locations where exotic plant species were measured are indicated by the numbered yellow points. Lines perpendicular to the points were used to assess colonization towards the rocky outcrop. Source: Google Earth (2018), modified by the authors.

Trails adjacent to main *Andorinhas* trail were also investigated in relation to the exotic plants presence to assess their colonization towards the rocky area (Figure 2). First, 20 transects were established at the points where the highest concentration of exotic plants was observed, ten on each side of the main trail for the evaluation of exotic plants by the intercept-line method (Canfield 1941, 388-394). The sizes of these transects ranged from 20 to 50 m, as there were adjacent trails ending in a cliff (Figure 3). The clump size was measured to estimate the area occupied by herbaceous species and the arboreal crown length to measure arboreal-shrub area.

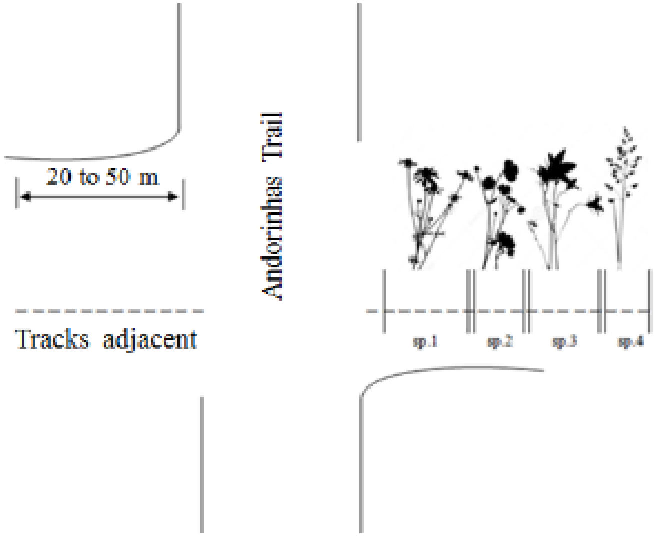

Figure 3. Scheme of the design used to assess exotic plant species in the tracks adjacent to the main *Andorinhas* trail in the municipality of *Niterói,* Rio de Janeiro state, Brazil.

The biological invasion in *Vale do Córrego dos Colibris* was investigated using the intercept-line method (Canfield 1941, 388-394). Fifteen points were chosen from three distinct areas where transects ranging in size from 50 m to 250 m were established, with a distance of 20 m between them (Figure 4). This procedure was necessary because the valley forms a “V” inwards and some of the lines eventually meet the rocky shore, not completing the expected 250 m. Area I departed from *Scyllia Souza Ribeiro* street into the forest, which is the closest to the main trail, where the lines ranged from 150 to 250 m.

Areas II and III are places with the most waste disposal from gardening by the surrounding residents because they are not part of the valley visitation itinerary. These areas correspond to the lines which were stretched from a college wall and *São João* street, respectively, all 50 m in length. These areas were chosen because the college can be a possible source of propagules for the forest. The 50 m transect was adopted because the presence of exotic species was not observed beyond this extension.

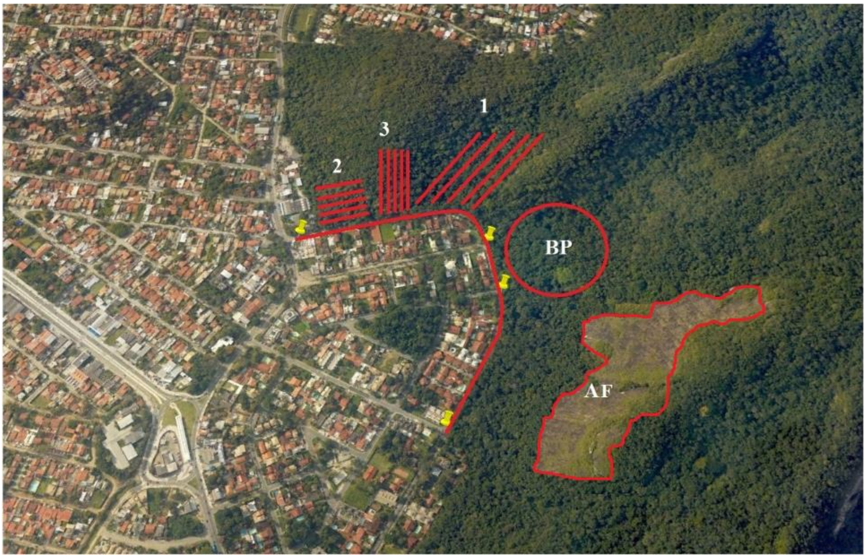

Figure 4. Region of the *Vale do Córrego dos Colibris* study area in the municipality of *Niterói*, Rio de Janeiro state, Brazil. The line surrounding the city identifies the forest edge and the yellow markers indicate the streets used to evaluate the exotic plant species. Parallel lines (with number 1, 2 and 3) represent the transects used to assess the colonization by exotic species from the streets towards the forest. The general inventory of this area additionally included the exotic species present around the rocky outcrop (AF) and *Brejo das Pacas* (BP), a wetland. Source: Google Earth (2018), modified by the authors.

The phytosociological parameters calculated for the two communities evaluated (*Morro das Andorinhas* and *Vale do Córrego dos Colibris*) were: Relative Specific Density (DE), Frequency (F), Relative Frequency (RF), Linear Coverage Index (CI) and Importance Value (IV) (Vuono 2002, 55-58). Microsoft Excel software (2010) was used for these calculations. The Importance Value (IV) was obtained by adding up the relative frequency with the relative specific density. Rarefaction curves based on individuals with 95% confidence intervals were generated using the R software (R Development Core Team 2017) and the "rarecurve" function of the Vegan package (Oksanen et al. 2019) to compare species richness. Only exotic species were considered in the analyzes.

RESULTS

Inventory of Exotic Plants in Forest Remnants

The inventory of exotic species in the forest remnants of the municipality of *Niterói* revealed the presence of 143 species, distributed in 121 genera and 55 families (Table 2). Of the 143 species, two were Gymnosperms (two genera and families). The families of Angiosperms with the largest number of species were: Leguminosae (13 spp.); Asparagaceae (10 spp.); Poaceae (9 spp.); Acanthaceae, Araceae and Lamiaceae (6 spp. each); and Apocynaceae, Euphorbiaceae, Malvaceae and Moraceae (5 spp. each) (Figure 5).

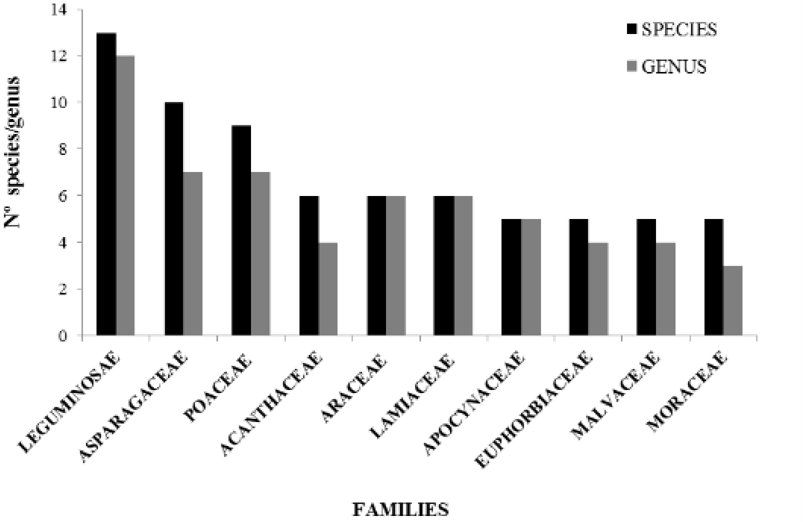

Figure 5. The most representative botanical families of exotic plants collected in forest fragments in the municipality of *Niterói*, Rio de Janeiro state, Brazil.

Table 2. Exotic plant species registered in six urban forest fragments in the municipality of *Niterói*, Rio de Janeiro state, Brazil

Family (Nº Genera/Species)	**Species**	**Habit**	**Origin**	**Use**	**Forest Remnants**					
Angiosperms					1	2	3	4	5	6
Acanthaceae (4/6)	*Brillantaisia lamium* (Nees) Benth.	herbaceous	Africa	Or				X	X	
	Ruellia simplex C. Wright	herbaceous	Brazil	Or					X	
	Sanchezia oblonga Ruiz & Pav.	herbaceous	Brazil	Or					X	
	Thunbergia alata Bojer ex Sims	climbing	Africa	Or/Ri	X	X	X	X	X	X
	Thunbergia erecta (Benth.) T. Anderson	herbaceous	Africa	Or	X				X	X
	Thunbergia grandiflora Roxb.	climbing	Asia	Or					X	X
Aizoaceae (1/1)	*Aptenia cordifolia* (L. f.) N.E. Br	herbaceous	Africa	Or						X
Amaranthaceae (1/1)	*Alternanthera sessilis* (L.) R.Br.	herbaceous	Asia	Or					X	
Anacardiaceae (2/2)	*Anacardium occidentale* L.	arboreal	Brazil	Al/Me	X			X	X	X
	Mangifera indica L.	arboreal	Asia	Al	X		X	X	X	X
Apiaceae (1/1)	*Centella asiatica* (L.) Urb.	herbaceous	Asia	Me	X					
Apocynaceae (5/5)	*Allamanda cathartica* L.	shrubby	Brazil	Or						X
	Catharanthus roseus (L.) G. Don	herbaceous	Brazil	Or						X
	Ceropegia linearis E. Mey	herbaceous	Africa	Or					X	
	Nerium oleander L.	arboreal	Mediterranean*	Or						X
	Plumeria rubra L.	arboreal	America	Or					X	
Araceae (6/6)	*Alocasia macrorrhizos* (L.) G. Don	herbaceous	Asia	Or					X	
	Anthurium andraeanum Linden ex André	herbaceous	America	Or						X

Table 2. (Continued)

Family (Nº Genera/Species)	**Species**	**Habit**	**Origin**	**Use**	**Forest Remnants**					
	Dieffenbachia seguine (Jacq.) Schott	herbaceous	Brazil	Or/Ri	X			X	X	X
	Epipremnum pinnatum (L.) Engl.	hemiepiphytic	Oceania	Or	X		X	X	X	X
	Philodendron hederaceum (Jacq.) Schott	hemiepiphytic	Brazil	Or					X	
	Syngonium podophyllum Schott	hemiepiphytic	America	Or	X	X	X	X	X	X
Araliaceae (2/3)	*Polyscias guilfoylei* (W. Bull.) L. H. Bailey	arboreal	Oceania	Or	X				X	X
	Schefflera actinophylla (Endl.) Harms	arboreal	Oceania	Or						X
	Schefflera arboricola (Hayata) Merr.	shrubby	Asia	Or					X	X
Arecaceae (3/3)	*Caryota urens* L.	arboreal	Asia	Or					X	
	Cocos nucifera L.	arboreal	Asia	Al/Or	X					
	Dypsis lutescens (H.Wendl.) Beentje & J.Dransf.	arboreal	Africa	Or			X		X	X
Asparagaceae (7/10)	*Agave americana* L.	herbaceous	America	Or	X					X
	Cordyline fruticosa (L.) A. Chev.	shrubby	Asia	Or	X				X	X
	Dracaena fragrans (L.) Ker-Gawl	shrubby	Africa	Or/Ri	X			X	X	X
	Dracaena marginata Hort.	shrubby	America	Or			X		X	X
	Furcraea foetida (L.) Haw.	herbaceous	America	Or	X	X		X	X	X
	Ophiopogon japonicus (L. F.) Ker Gawl	herbaceous	Asia	Or					X	
	Sansevieria cylindrica Bojer ex Hook.	herbaceous	Africa	Or/Ri					X	X
	Sansevieria trifasciata Prain	herbaceous	Africa	Or/Ri	X	X			X	X
	Sansevieria trifasciata var. *laurentii* (De. Wild.) N.C. Br.	herbaceous	Africa	Or/Ri					X	

Family (Nº Genera/Species)	**Species**	**Habit**	**Origin**	**Use**	**Forest Remnants**					
	Yucca gigantea Lem.	shrubby	America	Or	X					X
Asteraceae (3/3)	*Pseudogynoxys chenopodioides* (Kunth) Cabrera	herbaceous	America	Or					X	
	Tithonia rotundifolia (Mill.) S.F. Blake	herbaceous	America	Or					X	
	Unxia kubitzkii H. Rob.	herbaceous	Brazil	Or					X	
Balsaminaceae (1/1)	*Impatiens walleriana* Hook. f.	herbaceous	Africa	Or	X		X		X	X
Bignoniaceae (2/2)	*Spathodea campanulata* P. Beauv.	arboreal	Africa	Or/So				X		X
	Tecoma stans (L.) Juss. ex Kunth	shrubby	America	Or			X			X
Boraginaceae (1/1)	*Cordia abyssinica* R.Br.	arboreal	Africa/ Asia	Or					X	
Bromeliaceae (1/1)	*Ananas comosus* (L.) Merr.	herbaceous	Brazil	Al					X	
Cactaceae (2/2)	*Cereus* x *hildemannianus* K. Schum	herbaceous	Brazil	Or						X
	Nopalea cochenillifera (L.) Salm-Dyck	herbaceous	America	Or					X	X
Caricaceae (1/1)	*Carica papaya* L.	arboreal	America	Al	X		X		X	
Casuarinaceae (1/1)	*Casuarina equisetifolia* L.	arboreal	Oceania	So	X					
Combretaceae (2/2)	*Combretum indicum* (L.) Jongkind	shrubby	Asia	Or	X					
	Terminalia catappa L.	arboreal	Africa/ Asia	Or/Ri/ So	X		X			X
Commelinaceae (1/1)	*Tradescantia zebrina* Bosse	herbaceous	America	Or	X	X	X	X	X	X
Convolvulaceae (2/4)	*Distimake tuberosus* (L.) A.R. Simões & Staples	climbing	America	Or	X					
	Ipomoea batatas (L.) Lam.	herbaceous	America	Al					X	
	Ipomoea carnea Jacq.	shrubby	Brazil	Or					X	
	Ipomoea horsfalliae Hook.	climbing	Asia	Or			X			

Table 2. (Continued)

Family (Nº Genera/Species)	Species	Habit	Origin	Use	Forest Remnants					
Crassulaceae (1/2)	*Kalanchoe blossfeldiana* Poelln.	herbaceous	Africa	Or			X			X
	Kalanchoe delagoensis Eckl. & Zeyh.	herbaceous	Africa	Or						X
Cucurbitaceae (2/2)	*Cucurbita pepo* L.	herbaceous	America	Al					X	
	Luffa cylindrica (L.) M. Roem.	climbing	Asia	Me					X	
Cyperaceae (1/1)	*Cyperus papyrus* L.	herbaceous	Africa	Or						X
Euphorbiaceae (4/5)	*Codiaeum variegatum* (L.) Rumph. ex A. Juss.	shrubby	Asia	Or					X	
	Euphorbia mili Des Moul	herbaceous	Africa	Or						X
	Euphorbia umbellata (Pax) Bruyns	shrubby	Africa	Or						X
	Manihot esculenta Crantz	shrubby	America	Al					X	X
	Ricinus communis L.	shrubby	Africa	Fo					X	
Lamiaceae (6/6)	*Clerodendrum thomsoniae* Balf. f.	climbing	Africa	Or						X
	Leonotis nepetifolia (L.) R. Br.	herbaceous	Africa	Or			X	X		
	Leucas martinicensis (Jacq.) R.Br.	herbaceous	America	Me					X	
	Mentha x *piperita* L.	herbaceous	Europe	Me					X	
	Ocimum gratissimum L.	herbaceous	Asia	Me	X		X	X		
	Plectranthus barbatus Andrews	shrubby	Asia	Me			X		X	X
Lauraceae (1/1)	*Persea americana* Mill.	arboreal	America	Al						X
Leguminosae (12/13)	*Acacia seyal* Delile	arboreal	Africa	Or						X
	Albizia lebbeck (L.) Benth.	arboreal	America	Fo/Re	X			X	X	X
	Alysicarpus vaginalis (L.) DC.	herbaceous	Asia	Fo/Re	X					
	Bauhinia variegata L.	arboreal	Asia	Or					X	X

Family (Nº Genera/Species)	**Species**	**Habit**	**Origin**	**Use**	**Forest Remnants**			
	Caesalpinia pulcherrima (L.) Sw.	shrubby	America	Or				X
	Cassia fistula L.	arboreal	Asia	Or/So		X		X
	Cenostigma pluviosum var. *peltophoroides* (Benth.) Gagnon & Lewis	arboreal	Brazil	Or/So				X
	Clitoria fairchildiana R.A. Howard	arboreal	Brazil	Or/So				X
	Delonix regia (Hook) Raf.	arboreal	Africa	Or/So		X	X	X
	Erythrina variegata L.	arboreal	Asia	Or			X	X
	Erythrina velutina Willd.	arboreal	Brazil	Or			X	X
	Leucaena leucocephala (Lam.) de Wit	arboreal	America	Fo/Re			X	X
	Tamarindus indica L.	arboreal	Africa/ Asia	Al/Or				X
Lythraceae (1/1)	*Punica granatum* L.	arboreal	Asia	Me				X
Malpighiaceae (1/1)	*Malpighia glabra* L.	shrubby	America	Al/Me		X	X	X
Malvaceae (4/5)	*Dombeya wallichii* (Lindley) K. Schum.	shrubby	Africa	Or		X		
	Hibiscus rosa-sinensis L.	shrubby	Americas	Or			X	X
	Hibiscus schizopetalus (Dyer) Hook.f.	shrubby	Africa	Or			X	
	Malvaviscus arboreus Cav.	shrubby	America	Or	X		X	X
	Pachira aquatica Aubl.	arboreal	Brazil	Or				X
Marantaceae (3/3)	*Calathea ornata* (Lindley) Körn.	herbaceous	America	Or			X	
	Ctenanthe setosa (Roscoe) Eichler	herbaceous	Brazil	Or				X
	Goeppertia majestica (Linden) Borchs. & S. Suárez	herbaceous	Brazil	Or			X	
Monimiaceae (1/1)	*Peumus boldus* Molina	herbaceous	America	Or			X	

Table 2. (Continued)

Family (Nº Genera/Species)	**Species**	**Habit**	**Origin**	**Use**	**Forest Remnants**					
Moraceae (3/5)	*Artocarpus heterophyllus* Lam.	arboreal	Asia	Al	X		X	X	X	X
	Ficus carica L.	arboreal	Mediterranean*	Al						X
	Ficus elastica Roxb.	arboreal	Asia	Or/Ri	X					X
	Ficus pumila L.	hemiepiphytic	Asia/ Oceania	Or						X
	Morus nigra L.	arboreal	Asia	Al/Me					X	X
Musaceae (1/2)	*Musa acuminata* Colla	herbaceous	Asia	Or					X	
	Musa x *paradisiaca* L.	herbaceous	Asia	Al/Me	X			X	X	X
Myrtaceae (3/4)	*Eucalyptus citriodora* Hook.	arboreal	Oceania	Or	X		X			
	Psidium guajava L.	arboreal	Brazil	Al/Me			X		X	
	Syzygium cumini (L.) Skeels	arboreal	Asia	Al					X	
	Syzygium jambos (L.) Alston	arboreal	Asia	Al				X		
Orchidaceae (2/2)	*Oeceoclades maculata* (Lindley) Lindley	herbaceous	Africa	Or	X			X	X	X
	Renanthera coccinea Lour.	epiphytic	Asia	Or			X			X
Piperaceae (1/1)	*Peperomia pellucida* (L.) Kunth	herbaceous	Brazil	Me					X	
Plantaginaceae (1/1)	*Russelia equisetiformis* Schltdl. & Cham.	herbaceous	America	Or					X	
Plumbaginaceae (1/1)	*Plumbago auriculata* Lam.	herbaceous	Africa	Or				X	X	
Poaceae (7/9)	*Bambusa vulgaris* Schrad. ex Wendl fo. *vulgaris*	herbaceous	Asia	Cv	X	X	X	X	X	X
	Bambusa vulgaris fo. *vittata* (Rivière & Rivière) McClure	herbaceous	Asia	Cv						X

Family (Nº Genera/Species)	**Species**	**Habit**	**Origin**	**Use**	**Forest Remnants**					
	Cymbopogon citratus (DC.) Stapf	herbaceous	Asia	Me					X	
	Eleusine indica (L.) Gaertn.	herbaceous	Africa	Fo						X
	Megathyrsus maximus (Jacq.) Simon & Jacobs	herbaceous	Africa	Fo	X	X	X	X	X	X
	Melinis minutiflora P.Beauv.	herbaceous	Africa	Fo	X	X		X	X	X
	Melinis repens (Willd.) Zizka	herbaceous	Africa	Fo	X	X	X	X	X	X
	Pennisetum purpureum Schum.	herbaceous	Africa	Fo	X					
	Phyllostachys aurea Rivière & C. Riviere	herbaceous	Asia	Or	X				X	X
Polygonaceae (1/1)	*Antigonon leptopus* Hook. & Arn.	climbing	America	Or	X		X		X	
Portulacaceae (1/1)	*Portulaca oleracea* L.	herbaceous	Europe	Or						X
Rosaceae (1/1)	*Eriobotrya japonica* (Thunb.) Lindley	arboreal	Asia	Al/Me			X		X	
Rubiaceae (2/2)	*Coffea arabica* L.	shrubby	Asia	Al			X	X	X	
	Ixora chinensis Lam.	herbaceous	Asia	Or						X
Rutaceae (2/4)	*Citrus limon* (L.) Osbeck	shrubby	Asia	Al/Me					X	
	Citrus reticulata Blanco	shrubby	Asia	Al			X		X	
	Citrus sinensis (L.) Osbeck	shrubby	Asia	Al/Me					X	
	Murraya paniculata (L.) Jack	shrubby	Asia	Or			X		X	X
Sapotaceae (1/1)	*Manilkara zapota* (L.) P. Royen	shrubby	America	Me/Or			X			
Solanaceae (2/3)	*Brugmansia suaveolens* (Willd.) Bercht. & J. Presl.	shrubby	America	Or			X		X	X
	Solanum lycopersicon L.	herbaceous	America	Al					X	
	Solanum seaforthianum Andrews	climbing	America	Or			X	X		
Strelitziaceae (1/1)	*Strelitzia reginae* Banks	herbaceous	Africa	Or						X

Table 2. (Continued)

Family (Nº Genera/Species)	**Species**	**Habit**	**Origin**	**Use**	**Forest Remnants**					
Verbenaceae (2/2)	*Duranta erecta* L.	herbaceous	America	Or					X	X
	Petrea volubilis L.	herbaceous	Brazil	Or			X			X
Zingiberaceae (1/2)	*Alpinia purpurata* (Vieill.) K. Schum.	herbaceous	Asia	Or					X	X
	Alpinia zerumbet (Pers.) B.L. Burtt & R.M. Sm.	herbaceous	Asia	Me/Or						X
Gimnosperms										
Araucariaceae (1/1)	*Araucaria columnaris* (Forst.) Hook.	arboreal	Oceania	Or			X			X
Cycadaceae (1/1)	*Cycas revoluta* Thunb.	herbaceous	Asia	Or					X	
Total					39	9	37	25	88	84

Forest remnants: 1 (*Morro da Viração* Complex), 2 (*Morro da Peça*), 3 (*Mata do Ministro*), 4 (Darcy Ribeiro Biological Reserve), 5 (*Serra da Tiririca*), and 6 (*Morro das Andorinhas*). Use: Al = Feeding, Fo = Pasture or Forage, Cv = Hedge or Hedgerow, Or = Ornamental, Me = Medicinal, Ri = Ritualistic, So = Shading, Re = restoration. *Mediterranean Region: Mediterranean Sea region (between Europe, Asia and Africa).

These families corresponded to 48.9% of the inventoried material and another 35 families were represented by only one or two species and corresponded to 32.17% of the total. Herbaceous was the most representative form (65 spp.), followed by arboreal (38 spp.), shrubby (27 spp.), climbing (8 spp.), hemiepiphytic (4 spp.) and epiphytic (1 spp.).

The bibliographic survey on exotic plants in conservation units in the state of Rio de Janeiro reached seven studies, two of which only focused on exotic species (Table 1). The other studies were inventories which contained exotic species in their lists. In comparing the exotic species found in the present study with those of literature survey, it was observed that many species were common (Souza and Cintra (2007); Santos et al. (2010); Faria (2013); Freitas et al. (2015)). The largest number of exotic species was listed for *Serra da Tiririca* (88 spp.) and *Morro das Andorinhas* (84 spp.), followed by *Mata do Ministro* (37 spp.), *Morro da Viração* Complex (37 spp.), Darcy Ribeiro Ecological Reserve (25 spp.) and *Morro da Peça* (9 spp.) (Table 2).

The predominant form of use was ornamental (Figure 6), of which seven were also used for rituals (*Dieffenbachia seguine*, *Dracaena fragrans*, *Ficus elastica*, *Sansevieria cylindrica*, *Sansevieria trifasciata*, *S. trifasciata* var. *lamentii*, and *Thunbergia alata*). Five others were used for shading (*Cassia fistula*, *Cenostigma pluviosum* var. *peltophoroides*, *Clitoria fairchildiana*, *Delonix regia*, and *Spathodea campanulata*), and two for medicinal use (*Alpinia zerumbet* and *Manilkara zapota*). *Terminalia catappa* (almond) was used as ornamental, ritualistic and for shading.

In terms of food, 24 species stood out and eight were also medicinal (*Anacardium occidentale*, *Citrus limon*, *Citrus sinensis*, *Eriobotrya japonica*, *Malpighia glabra*, *Morus nigra*, *Musa* x *paradisiaca* and *Psidium guajava*), and two were also ornamental (*Cocos nucifera* and *Tamarindus indicates*). Other minor uses were those for medicinal purposes (*Centella asiatica*, *Cymbopogon citratus*, *Leuca martinicensis*, *Luffa cylindrica*, *Mentha* x *piperita*, *Ocimum gratissimum*, *Peperomia pellucida*, *Plectranthus barbatus* and *Punica granatum*), hedges (*Bambusa vulgaris* fo. *vittata* and *Bambusa vulgaris* fo. *vulgaris*), foraging

(*Megathyrsus maximus*, *Melinis minutiflora*, *M. repens*, *Pennisetum purpureum* and *Ricinus communis*), and shading (*Casuarina equisetifolia*). Four other species were traditionally used for restoring pastures, and degraded or eroded areas (*Albizia lebbeck*, *Alysicarpus vaginalis*, *Eleusine indica* and *Leucaena leucocephala*).

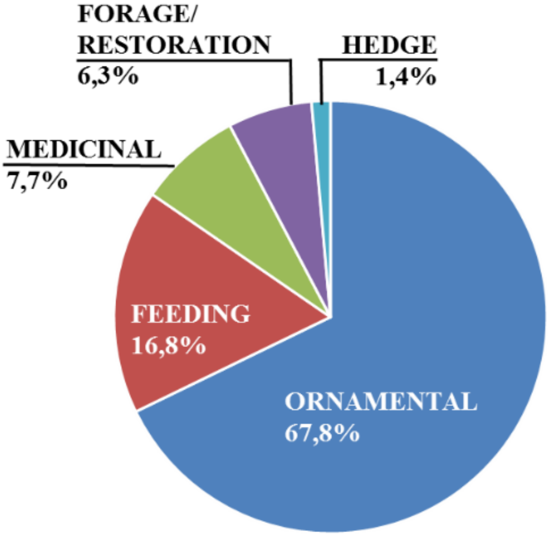

Figure 6. Main usage forms of exotic plant species collected in six forest fragments in the municipality of *Niterói*, Rio de Janeiro state, Brazil.

With regard to the origin, there was a predominance of species from Asia, followed by the Americas, Africa, Brazil, Oceania, and Europe (Figure 7). *Ficus pumila* has been reported in the literature as originating from two continents (Asia and Oceania), as well as *Cordia abyssinica*, *Tamarindus indica* and *Terminalia catappa* (Africa and Asia). *Ficus carica* and *Nerium oleander* had its place of origin in the literature as being the Mediterranean Sea Region.

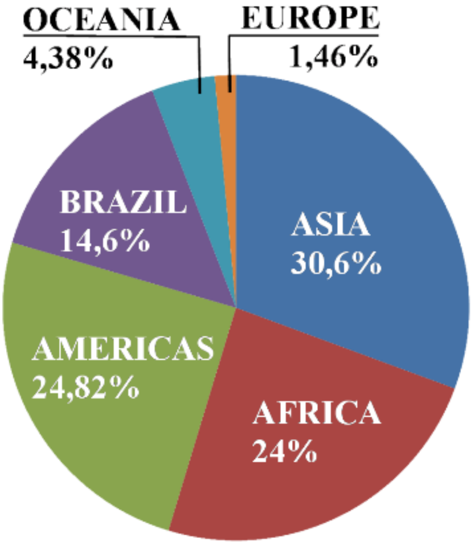

Figure 7. Origin of exotic plant species found in forest fragments in the municipality of *Niterói*, Rio de Janeiro state, Brazil.

Structural Analysis of Exotic Plants

Morro das Andorinhas

We found 15 exotic species on the *Andorinhas* trail and they belonged to 13 families. Herbaceous was the most representative form (8 spp.), followed by shrubby (5 spp.), arboreal (2 spp.), and epiphytic (1 spp.). The predominant form of use was ornamental (11 spp.), of which two also have ritual use (*Dieffenbachia seguine* and *Dracaena fragans*). In addition to ornamental, *Polyscias balfouriana* was also used for hedges or hedgerows. Another form of use was for food (2 spp.), of which *Musa paradisiaca* was also used as medicinal. One species, *Megathyrsus maximus,* was used for forage. Most species were from Africa and Asia (n = 5), Americas (n = 3), Brazil and Europe (n = 1). *Ctenanthe setosa* has its origin in Brazil, but did not naturally occur in the area.

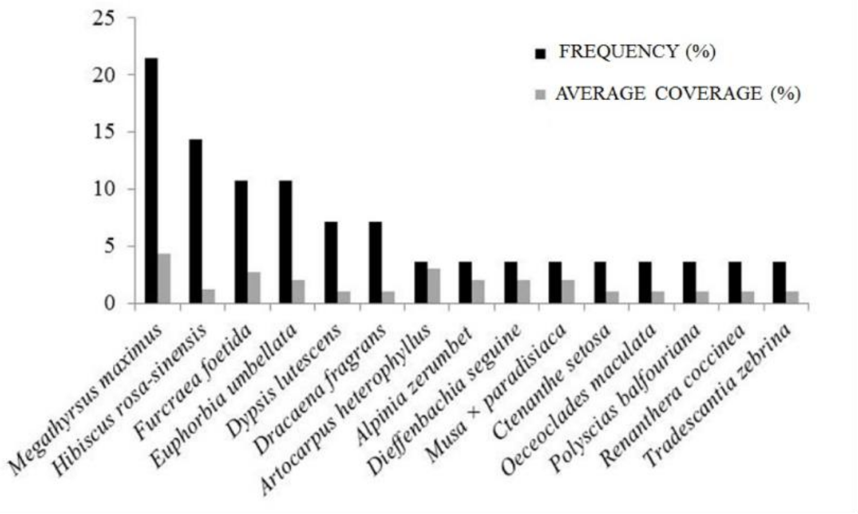

Figure 8. Frequency and average coverage of exotic plant species registered at the *Andorinhas* trail, municipality of *Niterói*, Rio de Janeiro state, Brazil.

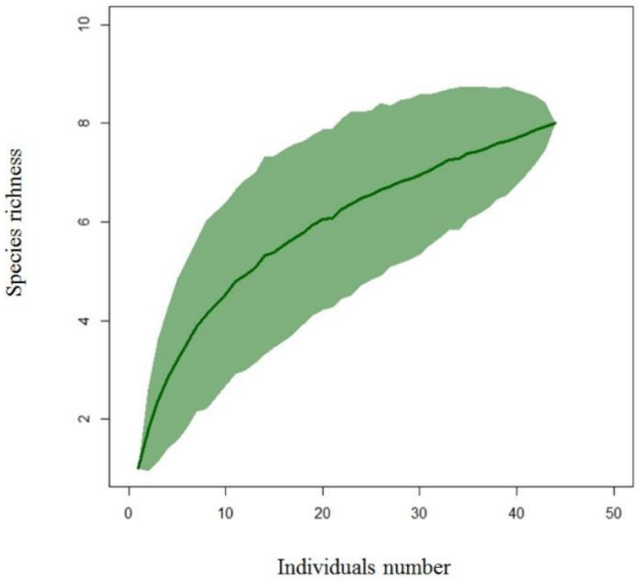

Figure 9. Rarefaction curve for exotic species sampled along 20 transects adjacent to the *Andorinhas* trail, municipality of *Niterói*, Rio de Janeiro state, Brazil.

Table 3. Phytosociological analysis of exotic species recorded in transects on adjacent tracks in the study area of *Morro das Andorinhas*, Niterói, Rio de Janeiro state, Brazil

Species	F	RF	LCI	RDD	IVV
Furcraea foetida (L.) Haw.	11	25,0	2,86	22,08	47,08
Phyllostachys aurea Rivière & C. Riviere	1	2,3	5,79	44,79	47,06
Oeceoclades maculata (Lindl.) Lindl.	18	40,9	0,24	1,89	42,80
Megathyrsus maximus (Jacq.) Simon & Jacobs	3	6,8	2,92	22,61	29,43
Tradescantia zebrina Bosse	6	13,6	0,05	0,37	14,01
Melinis repens (Willd.) Zizka	1	2,3	0,90	6,93	9,20
Plectranthus barbatus Andrews	3	6,8	0,10	0,80	7,62
Dracaena marginata Hort.	1	2,3	0,07	0,53	2,81

Legend: F = frequency, RF = relative frequency (%), LCI = linear coverage index (%), RDD = relative dominance (%), IVV= value of importance (%).

The most frequent species were *Megathyrsus maximus* (colonist grass), *Hibiscus rosa-sinensis* (hibiscus), *Furcraea foetida* (pita), and *Euphorbia umbellata* (dairy) (Figure 8). Eight exotic species were sampled using the intercept-line technique and belonged to five families (Table 3). The most important species were *F. foetida*, *Phyllostachys aurea* (garden bamboo), *Oeceoclades maculata* (ground orchid), *M. maximus*, and *Tradescantia zebrina* (zebrina) (Table 3). The rarefaction curve based on species richness and abundance did not reach stability (Figure 9).

Vale do Córrego dos Colibris

We recorded 26 exotic species in the *Vale do Córrego dos Colibris* study area and they belonged to 16 families. Araceae and Heliconiaceae families had the highest species richness (4 spp.), which corresponded to 22% of the inventoried material. Herbaceous was the most representative form (15 spp.), followed by shrubby (5 spp.), arboreal (3 spp.) and hemiepiphytic (3 spp.).

Table 4. Phytosociological analysis of exotic species recorded in transects on tracks in the study area of *Vale do Córrego dos Colibris*, Niterói, Rio de Janeiro state, Brazil

Species	**F**	**RF**	**RDD**	**IVV**
Tradescantia zebrina Bosse	13	7.26	19.66	26.92
Epipremnum pinnatum (L.) Engl.	27	15.08	7.08	22.16
Artocarpus heterophyllus Lam.	22	12.29	9.41	21.70
Syngonium podophyllum Schott	16	8.93	8.24	17.17
Heliconia episcopalis Vell.	11	6.14	9.82	15.96
Dieffenbachia seguine (Jacq.) Schott	12	6.70	6.76	13.46
Mangifera indica L.	10	5.58	7.77	13.35
Heliconia spathocircinata Aristeg.	11	6.14	6.49	12.63
Musa paradisiaca L.	14	7.82	2.96	10.78
Oeceoclades maculata (Lindl.) Lindl.	14	7.82	2.32	10.14
Sansevieria trifasciata Prain.	6	3.35	5.42	8.77
Coffea arabica L.	9	5.02	0.95	5.97
Tithonia rotundifolia (Mill.) S. F. Blake	2	1.11	3.49	4.60
Heliconia psitacorum L. F.	1	0.55	3.42	3.97
Dracaena fragans (L.) Ker. Gawl.	6	3.35	0.47	3.82
Syzygium cumini (L.) Skeels	2	1.11	1.43	2.54
Phylodendron hederaceum (Jacq.) Schott	2	1.11	1.23	2.34
Malvaviscus arboreus Cav.	2	1.11	1.00	2.11
Megathyrsus maximus (Jacq.) Simon & Jacobs	1	0.55	1.09	1.64
Dracaena marginata Hort.	2	1.11	0.09	1.20
Costus spicatus (Jacq.) Sw.	2	1.11	0.08	1.19
Heliconia velloziana L. Emygdio	1	0.55	0.47	1.02
Unxia kubitzkii H. Rob.	1	0.55	0.10	0.65
Goeppertia majestica (Lindley) Borchs & S. Suárez	1	0.55	0.06	0.61
Murraya paniculata (L.) Jack	1	0.55	0.01	0.56

Legend: F = frequency, RF = relative frequency (%), RDD = relative dominance (%), IVV= value of importance (%).

The predominant form of use was ornamental (19 spp.), of which seven also had ritual use (*Dieffenbachia seguine*, *Dracaena fragans*, *Heliconia episcopalis*, *H. psitacorum*, *H. spathocircinata*, *H. velloziana*, and *Sansevieria trifasciata*). In addition to ornamental, *Costus spicatus* was also used for medicinal and ritualistic purposes. Six species were used for food, of which *Musa* x *paradisiaca* was also used as medicinal. One species, *Megathyrsus maximus*, was used for forage. With regard to the origin, seven species were from Asia, six from the Americas, four from Africa, one from Oceania. Eight species have their origin in Brazil, but did not naturally occur in the studied area.

The ten species with the highest importance value in *Vale do Córrego dos Colibris* were *Tradescantia zebrina* (*zebrina*), *Epipremnum pinnatum* (*boa*), *Artocarpus heterophyllus* (jackfruit), *Syngonium podophyllum* (*singonio*), *Heliconia episcopalis* (heliconia), *Dieffenbachia seguine* (dumb cane), *Mangifera indica* (mango tree), *Heliconia spathocircinata* (heliconia), *Musa* x *paradisiaca* (banana tree), and *Oeceoclades maculata* (ground orchid) (Table 4). The rarefaction curve based on species richness and abundance also did not achieve stability (Figure 10).

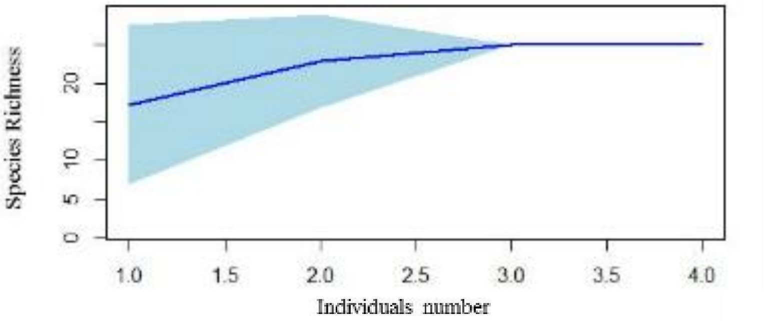

Figure 10. Rarefaction curve for the species sampled along the 15 transects in the *Vale do Córrego dos Colibris*, municipality of *Niterói*, Rio de Janeiro state, Brazil.

DISCUSSION

Inventory of Exotic Plants in Forest Remnants

Studies addressing exotic plant species in the state of Rio de Janeiro can be divided into two groups: those focusing on the occupation of an exotic species in a given area (e.g., Gomes (2007); Ribeiro and Zaú (2007); Raíces et al. (2008; 2017); Abreu; Rodrigues (2010); Moura (2011); Ferreira (2013); Nascimento et al. (2013); Santos (2013); Zucaratto and

Pires (2014); Magalhães et al. (2015); Mello et al. (2015); Bergallo et al. (2016); Faria (2016); Zimmermann et al. (2016; 2017) and Freitas et al. (2017)), and those aiming to characterize the floristic and/or structure of the entire (exotic) community in one or more areas, as well as its influence on native plants (e.g., Siqueira 2005; 2006; Barroso 2009; Ribas et al. 2010). This study is inserted in the second group, and shows a high richness of exotic species compared to other studies with similar designs.

Forest remnants in the municipality of *Niterói* are immersed in an urbanized matrix, where anthropogenic pressure culminates in their deterioration (Fernandez et al. 2018, 30). In view of this, illegal occupation and public urbanization stands out, which are increasingly expanding to protected areas, permanent preservation areas or areas of special environmental interest. Emphasis is also given to the lack of trained technicians to conduct surveys and inspections in forests which are still preserved. Another factor is urban violence which limits prevention and management actions related to biological invasion by plants in these areas.

Exotic plants are being cultivated in this difficult and conflicting scenario, and may spread to the surrounding forest remnants. This is a serious situation because most of these remnants are integral protection areas. In comparing this study with others which performed floristic and structural analyzes in the state of Rio de Janeiro, we observe that species composition was basically the same in both studies. Their current presence in such places maintains the memory of the old uses to which these sites have been subjected in the past (Oliveira and Silva 2011, 71).

The species richness found in this study is high when comparing the data of the present study to those inventoried in other conservation units in Rio de Janeiro (Table 1). This result reflects a sampling effort performed in the forest remnants of the municipality of *Niterói* to know the diversity of exotic species. However, this study differs from most floristic studies because exotic species in other inventories are omitted or only cited in the checklists. Moro et al. (2012, 998) argue that exotic species should be presented in the studies using the classification in the literature to say which are exotic, invasive or simply ruderal, and thus apply efforts to eradicate those considered more aggressive. When comparing the results with other inventories of species used in the afforestation of urban areas in the state of Rio de Janeiro, it is similarly noticed that most exotic arboreal species are shared between them. According to Santos et al. (2010, 210) and Souza et al. (2015, 15), exotic species are predominant in the afforestation of large Brazilian cities, and in some cases they correspond to 80% of the specimens used.

Most of the total listed exotic species are of African and Asian origin (Figure 6), showing the historical distribution of plants with economic and ornamental interest due to introductions in the old Portuguese trading posts in Brazil. According to Dean (2001, 1), these translocations represent a critical aspect in the "cultural conquest", in which cultural elements were selected by the colonizers to maintain and control the colonial society. This process was initiated in several parts of the world in the European expansion era in order to create a familiarity with the European landscape (Santos et al. 2010, 210). These translocations were subsequently intended to meet agricultural, forestry and other direct use needs (Ziller 2001, 78).

Between the late 18th and early 20th century, it is clear that three other issues drove the transport of exotic species to Brazil. Primarily, two royal decrees, issued in July 1809 and 1810, offered prizes and tax exemptions to anyone willing to introduce economically valuable plants in the country (Dean 1996, 142). On the other hand, the landscaping movement in Brazil, which began in the late 18th century, took part as a great way to translocate species. European plants were introduced to compose gardens or even forest large cities. Naturalist *Auguste François Marie Glaziou* was one of the first to use these plants for landscape projects, especially in the city of Rio de Janeiro, when he was appointed as Director of the Public Gardens, Afforestation and Forests in 1893. The introduction of species for this purpose have subsequently been minimized but not completely abandoned, as noted in the designs of architect and landscape architect *Roberto Burle Marx*. Although he started prioritizing the use of native Brazilian species, he still employed many others of foreign origin (Santos et al. 2010, 210-211).

The translocations of species to the capital Rio de Janeiro had an influence on the cultivated flora in the city of *Niterói*, with most of the plants being used in exotic public afforestation. However, it was observed that many of these species introduced as ornamentals came to compete with native species, mainly because of their high reproductive and dispersal capacity (Pivello 2011, 2). The wide use of exotic species is linked to their easy adaptation to different local environmental factors, quickly occupying the spaces in which they are cultivated, mainly for economic and ornamental purposes (Cupertino and Eisenlohr 2013, 745-746).

It should be noted that many of the streets, squares, and gardens that surrounds the forest remnants may be propagule sources of exotic species (McKinney 2006, 247-260; Dehnen-Schmutz et al. 2007, 527-534; Santos et al. 2010, 209-222). In this sense, attention is needed to the issue of propagule pressure that adjacent areas may subject to those which are still preserved. This hypothesis is related to the abundance of individuals released in an introductory event and the number of events, which means that the greater the insistence on adapting a species to a new environment, or the lesser supervision to avoid introductions, the greater the propagule pressure (Lockwood et al. 2005, 223). Factors such as plant form, reproductive system and dispersal mode may also favor biological invasion (Pyšekand Richardson 2006, 2). The concern regarding national protected areas was pointed out by Leão et al. (2011), Sampaio and Schmidt (2013) and Ziller and Dechoum (2013), because the higher the propagule pressure, the greater the chances of invasion (Lockwood et al. 2005, 223).

The species used to compose pasture and foraging areas are also noteworthy, as well as those traditionally used for restoration degraded or eroded areas, which are present in the six areas investigated. *Albizia lebbeck* and *Leucaena leucocephala* have a history of use in agroforestry systems and recovery of degraded areas because they are nitrogen-fixing legume trees with symbiotic associations with undulating bacteria (Dias et al. 2007, 119-120). On the other hand, the introduced *Poaceae* species are not only present around forests, but also in rocky outcrop areas, competing with native species and occupying a type of environment where endemic and endangered plants are found (Barros 2008, 150).

Further noticeable in the floristic inventory are allochthonous species (e.g., *Anacardium occidentale*, *Cereus hildmannianus*, *Cenostigma pluviosum* var. *peltophoroides*, *Clitoria fairchildiana*, *Erythrina velutina*, *Pachira aquatica, Ctenanthe setosa* and *Petrea volubilis*) which, although native to Brazil, do not belong to these forest remnants. They were probably brought from other parts of Rio de Janeiro or other states and grown for ornamental purposes. Of the cited species, *C. setosa* is endemic to Brazil (Flora do Brasil 2020), but was cultivated in the area by a local resident, and *C. pluviosum* var. *peltophoroides* occurs naturally in the Lakes Region in an area of Semi-decidual Seasonal Forest (Ribeiro and Lima 2009, 117).

Structural Analysis of Exotic Plants

Morro das Andorinhas

The floristic inventory in the main trail resulted in 15 exotic species, all common to the total of 84 species inventoried in the total area of *Morro das Andorinhas*. Most of the species were used as ornamental (e.g., *Ctenanthe setosa, Dypsis lutescens, Euphorbia umbellata*, *Hibiscus rosa-sinensis, Oeceoclades maculata*, *Polyscias balfouriana*, *Renanthera coccinea,* and *Tradescantia zebrina*) and usually found at the beginning of the sampling points because these points were close to the residences. Proximity to residences is an important source for introduction of exotic species and requires government attention because exotic species can become an environmental problem. In turn, the adjacent trails established on the exposed rocky outcrop resulted in eight exotic species, four of which were common to the *Trilha das Andorinhas* and the general inventory (e.g., *F. foetida*, *O. maculata, M. maximus* and *T. zebrina*).

Richardson et al. (2000, 98) proposed a distance for considering an exotic species as invasive as a reference, being the species ability to disperse without human aid at distances greater than 100 m from the place of introduction in less than 50 years. For plants with vegetative reproduction (e.g., root growth, rhizome, stolon, or stem), distances greater than 6 m in 3 years. In this case, exotic species are also called invasive, and their eradication becomes quite difficult after reaching such potential (Pitelli 2007, 412). Considering all stages of the biological invasion proposed by Williamson (1996, 28-52), 3.9% of the exotic species from Morro das Andorinhas can be considered established and with invasive disposition. These species were *F. foetida*, *M. maximus,* and *Phyllostachys aurea*. These are herbaceous species with vegetative reproduction and are expanding in the area, as predicted by Richardson et al. (2000, 98).

Most exotic species found have no invasive character (96.1%), such as *O. maculata*, which had the third highest importance value (42.80%), but occupied very specific sections in the adjacent trails. These results corroborate those of Williamson and Fitter (2006) and Horowitz et al. (2013), which found that most of the introduced species do not go beyond the introduction or stabilization stages, and therefore do not become invasive. However, their presence in rocky environments needs to be investigated and clarified to the surrounding population, drawing attention to the problems caused by exotic plants.

M. maximus stood out on the main trail (frequency 21.4%/coverage 4.3%), and was the fourth most important species on adjacent trails (29.43%). This species was introduced in Brazil via propagules being accidentally carried on African slaves’ clothes brought for forced labor in sugarcane and coffee monocultures and for commercial foraging purposes (Oliveira and Silva 2011, 71). *M. maximus* is considered one of the most aggressive exotic species, and its eradication is difficult because it has a highly resistant underground rhizome-like stem system (Faria 2016, 15-16). Some attempts were made between 1995 and 2000 to recompose the vegetation in some patches occupied by *M. maximum* in *Morro das Andorinhas*. The grass clumps were manually removed and seedlings of (tree) native species were planted. This initiative was successful due to the constant maintenance of the recovered areas (Barros and Garcez 2003, 211). However, the grass expanded again as maintenance activities declined. Aximoff et al. (2016, 157) argued that the presence of *M. maximum* and *F. foetida* in rocky outcrops may be related to the ecological changes produced by the spread of fire, where less resistant native species are replaced by more adapted ones. The same process was observed by Carauta and Valente (1974), Meirelles et al. (1999) and Barros and Garcez (2003).

F. foetida presented the third largest invasive potential in the main trail (frequency 10.7%/coverage 2.7%) and the highest importance value (47.08%) in the adjacent outcrop trails. There are reports about the presence of this species in the Brazilian coastal environments (Barbosa 2017, 45-47) and in South Africa (Crouch and Smith 2011, 196). *F. foetida* produces numerous propagules in the fruiting period and they can spread to a large extent and generate numerous seedlings. *F. foetida* was also found in the *PESET* in the areas of *Alto Mourão*, *Enseada do Bananal*, and *Morro do Telégrafo*. *F. foetida* is sometimes confused with *Agave americana*, another species of the Asparagaceae which also occurs in *Morro das Andorinhas*.

P. aurea had the second highest importance value (47.06%), occupying a significant area in the rocky outcrop facing *Itaipu* Beach. This species was planted by a local resident. It reproduces well in a vegetative way and has currently expanded without control and competing with native species. A similar fact was also recorded in *Alto da Serra de Paranapiacaba* Biological Reserve, *Santo André* city (São Paulo state), and *Brasília* (Federal District), where it was introduced to contain the erosive processes, but then caused environmental problems (Filgueiras 1988, Pastore et al., 2012, 25).

Vale do Córrego dos Colibris

The inventory in *Vale do Córrego dos Colibris* featured 25 exotic species, all common to those inventoried in the surroundings. The region belonged to an old farm, on which some of the exotic plants were grown as fruit trees. Agricultural activities in this region have decreased over the years, and the Atlantic Forest vegetation has returned. However, many of the species remained in the middle of the forest area as a material mark of previous land use. The surroundings of the forest became crowded from the 1970s onwards with an intensification in the occupation of the oceanic region of *Niterói*, and many houses which were built used various ornamental plants in the gardens.

However, the material disposed of cleaning the gardens is commonly thrown into the forest in front of the houses, even though it is within an integral conservation unit legally established by the government. Some residents even made small gardens in the *PESET* to "improve the ornamental value" of the street. Species grown in the conservation unit as well as those disposed of as "garden waste" can propagate and cause environmental impairment, hindering the regeneration of native plants. The forest is now encompassed by the urban network by the anthropic expansion of rural areas into the municipality of *Niterói*, being surrounded by houses which constantly increase the propagule pressure with discarded ornamental plants in the evaluated areas of *PESET*. As a result, the process of biological invasion represents a serious problem here. Even though residents have a regular garbage collection available by CLIN (the *Niterói* Cleaning Company) three times a week, garden garbage is commonly observed in the forest edge.

Considering all stages of biological invasion proposed by Williamson (1996), 6.1% of the exotic species in *Vale do Córrego dos Colibris* can be recognized as established and invasive species, namely *Dieffenbachia seguine, Epipremnum pinnatum, Syngonium podophyllum,* and *Tradescantia zebrina* (ornamental and ritualistic species). *T. zebrina* is a Mexican reptant herb which grows very well in shady and humid areas, easily multiplying by cuttings (Lorenzi and Souza 2001, 384). *T. zebrina* was the species with the largest importance value in *Vale do Córrego dos Colibris* (review Table 4). This species was observed as having hemiepiphytic form in climbing some trees and rocks. It may prevent the development of native forest species in the herbaceous stratum. Furthermore, it is difficult to handle, with it being necessary to manually remove them. Due to the ease of vegetative propagation, *T. zebrina* can spread rapidly as small fragments of the plant become propagules with high regenerative efficiency (Siqueira 2006, 325).

D. seguine, E. pinnatum and *S. podophyllum* are species of the Araceae, one of the families which stood out in the inventory. Araceae is economically important due to its ornamental, feeding, and medicinal purposes, and some species have mystical symbolism (Assis and Sakuragui 2005, 520). While *D. seguine* has an herbaceous form, *E. pinnatum* and *S. podophyllum* are secondary hemiepiphytics. They initially grow as reptant terrestrial herbaceous and then take root in the arboreal trunks as hemiepiphytic, characterizing this form secondarily when they lose their connection with the soil (Gonçalves and Lorenzi 2011, 77).

E. pinnatum is an Asian species and has great propagating potential because it is easily multiplied by cuttings, and given its rapid growth, it leans on arboreal forms and may suffocate them. This species had the second greatest importance value in *Vale do Córrego dos Colibris* (review Table 4), being widely used for gardens ornamentation, and forming a cover of flower beds (Lorenzi and Souza 2001, 203). Its presence on tree and palm species generates moisture, which induces early death in trunks and branches consequent of the individual used as support (Siqueira 2006, 325; Valadares 2011, 28). It produces larger leaves as it achieves higher heights on its support, fixing roots in the trunks during its growth. This feature may be related to a strategy to increase the shade for young individuals still on the ground, or even to ensure greater use of incidence to perform photosynthetic processes (Kupas 2016, 37).

D. seguine is originally from Central America and is one of the species which stands out due to its lush leaves which are widely used for ornamentation (Lorenzi and Souza 2001, 202) and for mystical purposes. Its presence shows a mark in the landscape that concerns its non-material dimension, meaning the culture associated with the territory managed by man (Oliveira and Silva 2011, 71). *D. seguine* was observed in woods close to old residences, and occupied large areas when having favorable conditions. It develops very well in humid places, displacing native species by competition and covering the undergrowth (Siqueira 2006, 325). *D. seguine* had the sixth greatest importance value and was observed more than 200 m from the street.

S. podophyllum is also originally from Central America and is widely used for ornamentation (Kupas 2016, 36). It is characterized by foliar dimorphism, with young leaves stalked or sagittal with white stripes and fully green pedatissect leaves (Chong et al. 2010, 1-3). Young individuals form “rugs” in *Vale do Córrego dos Colibris* and cover the herbaceous stratum of the forest, while adults progressively ascend the tree trunks. *S. podophyllum* is among the 106 species listed as exotic found in Brazilian federal conservation units (Sampaio and Schmidt 2013, 40) and is reported to be exotic in countries such as the United States, Seychelles, and Singapore (Matyot 1999, 70-71; Chong et al. 2010, 5).

Philodendron hederaceum did not stand out as a species with a high importance value, but it had the same occupation characteristics as *E. pinnatum*. It can occupy the canopy forcefully and small pieces which separate from the plant can take root and grow rapidly. Its presence was restricted to the forest edge where it was introduced. However, *P. hederaceum* may expand its cover as has been observed in other areas of *PESET*. No references were found in the literature about the biological invasion caused by this species, but it is necessary to be aware of it. It must be removed from the area in which it was observed (near the school wall), before it spreads and becomes difficult to handle.

Finally, four exotic species of the *Heliconia* genus stood out in the sample, namely *H. episcopalis, H. psitacorum, H. spathocircinata,* and *H. velloziana*. *Heliconia* species produce very showy inflorescence, being used as ornamental and ritualistic in some religions (Maioli-Azevedo and Fonseca-Kruel 2007, 268; Santos 2009, 35). The propagation of these species in *Vale do Córrego dos Colibris* has been facilitated by local environmental conditions favorable to these plants.

CONCLUSION

The use of exotic plants for urban afforestation and/or ornamentation in the city of *Niterói* acts as an important center for biological invasion into forest remnants. This chapter presented a diagnosis on biological invasion processes by exotic plant species in rocky outcrops and lowland areas near urban centers. The exotic plants of *Morro das Andorinhas* presented greater concentration near the dwellings, advancing along the trails and also significantly occurring on the rocky outcrop. *Furcraea foetida*, *Megathyrsus maximus,* and *Phyllostachys aurea* were some exotic herbaceous species with vegetative reproduction which have been expanding. The cultivation and dissemination of propagules of ornamental species from surrounding residences in *Vale do Córrego dos Colibris* were an important source for exotic species introduction. *Dieffenbachia seguine*, *Epipremnum pinnatum*, *Syngonium podophyllum,* and *Tradescantia zebrina* were in the process of biological invasion in this area. Although presenting high importance value, the *Artocarpus heterophyllus*, *Musa* x *paradisiaca*, *Mangifera indica,* and *Heliconia* spp. species were not behaving as invasive species in *Vale do Córrego dos Colibris*. Along with the other many exotic species found in the six urban forest fragments of *Niterói*, the aforementioned species represent the negative effect of human actions on "natural" ecosystems. Many other ecosystems worldwide are similarly being affected by exotic and invasive species causing serious damage on resident species. Studies such as this one are important to know which exotic species are behaving as invasive in order to plan management actions to control them.

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Chapter 5

INVASIVE EXOTIC OIL PALM TREES MODIFY THE STRUCTURE OF VASCULAR EPIPHYTE COMMUNITY ON AN ATLANTIC FOREST ISLAND

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ABSTRACT

Demand for agricultural areas has suppressed forests throughout the Brazilian coast. Most of these areas have been used for monoculture cultivation, which has distinct microclimatic and structural conditions compared to native forests. Exotic monocultures contribute to the extinction of epiphyte populations and simplify the community. The *Elaies guineensis* oil palm was introduced in Brazil by African slaves in the 16th century and is considered to be a facilitator of ecological succession, providing shade and organic matter, in addition to attracting seed-dispersing fauna. However, this species behaves as invasive in native environments in different parts of the country, changing the biodiversity and ecosystem services. Herein, we present an evaluation of species composition and community structure of vascular epiphytes in a *E. guineensis* sub-spontaneous stand (*SSEG*) in Ilha da Marambaia, Rio de Janeiro state, Brazil. Our goal was to evaluate whether the structure of the vascular epiphyte community has been modified in terms of composition, richness, spatial distribution and functional characteristics. The species richness (S = 23) in the *SSEG* can be considered low compared to native forests. The functional groups differed from what was expected; the number of characteristic and optional holoepiphytes was lower than expected (30.4% and 17.5% respectively), while the accidental group was higher than expected (30.4%). Among the accidental groups, the vast majority were zoochoric species. Clumped distribution was the most common distribution pattern between epiphyte species, probably because the environmental conditions were more limiting (e.g., low availability of microhabitats) in the *SSEG*. Phorophyte morphometry generally influenced species richness and epiphyte abundance in the *SSEG*. These changes in the epiphyte community structure indicate that the low heterogeneity of the stand reduces epiphyte species richness and leads to unrepresentative species (in mature forests) becoming dominant. Although we did not evaluate the contribution of organic matter, these changes in the epiphyte community structure are possibly due to the accumulation of organic matter and moisture in the permanent sheaths of oil palm leaves, which creates conditions for accidental and zoochoric species to colonize this environment. We concluded that the sum of oil palm tree effects is leading to a reduction in the epiphyte diversity in the area. Thus, it is necessary that oil palm monoculture managers monitor and manage these systems, especially those which have been abandoned due to low productivity, in order to control the invasion of this species into natural systems and the consequent diversity loss of vascular epiphytes and other biological groups which are sensitive to invasion.

Keywords: biodiversity, community change, epiphyte conservation, exotic tree monoculture, insular Atlantic Forest, oil palm tree

INTRODUCTION

Forest suppression is a result of the demand for agricultural areas (Lugo 1997). Most parts of deforested areas in the tropics are used for monoculture cultivation, which depletes and exhausts the soil. Other consequences of traditional monocultures are the loss of biological diversity (including endangered species), changes in ecosystem services and an increase in atmospheric gas concentration levels (Lugo 1997).

Deforested areas may recover spontaneously when abandoned and reach an advanced degree of recovery in approximately 20 years if degrading factors stop (Brown and Lugo 1990; Lugo 1997). Different techniques and methodologies have been used for re-establishing vegetation (e.g., seedling planting and unassisted natural regeneration) (Brown and Lugo 1994), restoring ecosystems and their ecological functions, accelerating ecological succession and allowing the return of the lost biodiversity (Lugo 1997; Martins et al. 2009, 19-51). Planting native species can accelerate the regeneration of degraded areas, even without the removal of exotic cultivated species (Lugo 1997; Martins et al. 2009, 19-51).

Tree monocultures such as *Acacia* or *Pinus* plantations (Lamb et al. 2005) can be used in some situations as a starting point for forest rehabilitation. The cultivated species cover the area, thereby modifying the microclimate by their shade and enabling re-establishment of native species (Khemnark 1994; Lugo 1997). Recent studies have demonstrated that monocultures of tropical tree species enable re-establishing a large number of native species in the lower stratum, accelerating the regeneration process in a degraded area (Lugo 1988a, 58-70, and b; 1992; Guariguata et al. 1995; Lugo 1997).

However, the use of monocultures for forest restoration is controversial. Most studies suggest that monocultures may have low stability, little resource efficiency and low biological diversity, in addition to modifying soil physicochemical qualities and water availability (Lugo 1997; Höfer et al. 2001; Kourtev et al. 2002; Siemann and Rogers, 2006). Nevertheless, the discussion on the use of monocultures to accelerate forest regeneration has enabled a constant reassessment of this tool's goals, indicating that long-term research in these areas is necessary (Lugo 1992; 1997).

One plant species which has undergone much expansion in its planted area in recent decades is the oil palm (*Elaeis guineensis* Jacq. - Arecaceae), an African palm species (Meijaard et al. 2018, 01-17). It is planted for different purposes, including biofuel production (Meijaard et al. 2018, 01-17), however oil palm monocultures have been the target of intense criticism by the conservationist community because areas of natural forests, especially in the tropics, have been suppressed for implanting large areas with this species. In addition to the direct loss of biodiversity caused by the suppression of native vegetation for its implantation, different biological groups respond negatively to oil palm plantations, leading to more biodiversity loss, such as birds (López-Ricaurte et al. 2017), butterflies (Koh, 2008), aquatic insects (Luiza-Andrade et al. 2017), and terrestrial arthropods (Ghazali et al. 2016) etc. On the other hand, oil palm also presents invasive behavior in natural areas. Its fruits are attractive to a large number of disperser species and seeds have high germination and establishment rates (Butler and Laurance, 2009).

The cultivation of oil palm trees was initiated in the 16th century in Brazil when African slaves brought seeds and used them as an agricultural intervention in the Atlantic Forest (Veiga et al. 2000, 23). *E. guineensis* settled in some areas after cultivation was abandoned through sub-spontaneous stands, possibly due to the hot and humid climate and the high and well-distributed rainfall throughout the year (Veiga et al. 2000, 23). These climatic characteristics are typical of Dense Ombrophilous Forest (one of the Atlantic Forest types) and its associated formations (IBGE 1992, 65-67). There are few studies about the oil palm in Brazil and the minority are directed to understand the ecological function of this exotic species in the environment (Moura et al. 2008).

One of the environmental changes caused by human action in *Ilha da Marambaia* (municipality of *Mangaratiba*, Rio de Janeiro state) is the presence of a sub-spontaneous stand of *E. guineensis* at *Praia da Armação*. This area has been abandoned for about 100 years, and the African oil palm population is expanding, especially due to local environmental characteristics. Preliminary field observations indicate that some typical elements of the Atlantic Forest can be found in this area, despite the oil palm dominance (Ribeiro and Nunes-Freitas, unpublished data). For example, typical species of the understory are found such as those of the *Psychotria* and *Miconia* genera, and trees of different regeneration stages such as *Guarea guidonea* and *Cupania oblongifolia*. However, because the *E. guineensis* sub-spontaneous stand was abandoned and therefore was not managed or controlled, seed dispersal occurred to nearby areas and have been massively colonized by the species. This has caused changes in the structure of the Dense Ombrophilous Forest and *Restinga* Forest communities because the understory is quickly colonized by seedlings and young *E. guineensis* individuals (Ribeiro and Nunes-Freitas, unpublished data).

In this chapter, we evaluate the species richness, species composition and structure of the epiphytic guild in an *E. guineensis* sub-spontaneous stand in *Praia da Armação, Ilha da Marambaia*. Our hypothesis was that the oil palm can create conditions which hinder the colonization of epiphyte species on its trunk due to its morphological characteristics, thereby modifying the community structure of this ecological group. As oil palm is an invasive species in different Atlantic Forest locations which can have a negative impact on the epiphytic flora, since it can lead to the loss of species in this group. Therefore, our goal was to answer the following questions: (i) What is the composition and species richness of the epiphyte community in this stand? (ii) What is the formation and dispersal syndrome of vascular epiphyte species in the area? (iii) What is the horizontal distribution pattern of epiphytes in the *E. guineensis* sub-spontaneous stand? (iv) What is the vertical occupancy pattern of epiphytes in *E. guineensis* individuals? (v) Does the size of the phorophyte affect the species richness of vascular epiphytes?

METHODS

Study Area

We conducted the study in an *Elaeis guineensis* Jacq. sub-spontaneous stand (*SSEG*) located in *Praia da Armação* (23º 0.3’ 17.37” S and 43º 57’ 18.99” W), *Ilha da Marambaia*, a district of *Mangaratiba*, on the southern coast of the state of Rio de Janeiro. The island (Ilha da Marambaia) is connected to the continent by a sand zone of about 40 km dominated by *Restinga* vegetation, which is medium-sized trees and shrubs adapted to drier and nutrient-poor substrate (sand) (Roncarati and Menezes 2005, 15-38). The climate in the region is tropical wet (Aw by the Koppen Classification), with dry winters (Mattos 2005, 55-66). The precipitation is concentrated between November and March, the average annual rainfall is 1239.7 mm (Mattos 2005, 55-66) and the average temperature oscillates between 16.8ºC (minimum) and 32.8ºC (maximum) (Lourenço et al. 2010).

The *SSEG* occupies an area of approximately 30 ha located in an ecotone between *Restinga* and Sub-montane Dense Ombrophilous Forest in *Praia da Armação*. Oil palm was introduced in the 16th century in an area of about 15 ha by African slaves as a subsistence culture, and it was abandoned about 100 years ago. The species was able to sub-spontaneously establish and dominate the area without management. There are *E. guineensis* individuals of different growth stages. The canopy has an average height of 12 meters. Some emerging trees (up to 30 m high) can be observed in the oil palm stand, generally belonging to the *Ocotea* genus. The understory is dense, mainly composed of young *E. guineensis* individuals and *Psychotria* specimens (personal observation).

Data Collection

We established five transects at the *SSEG*, 20 m apart. A total of 50 plots of 100 m² (10 x 10 m) were demarcated in the transects, equidistant at 10 m. All *E. guineensis* individuals and other phorophyte specimens with diameter at breast height (DBH at 1.30 m) equal or greater than 10 cm were measured and inspected for the occurrence of epiphytes in each plot. Each phorophyte had its CBH (circumference at breast height, measured with the support of a tape measure with 1 mm precision), total height (TH), bole height (BH) and the major and minor diameters of the canopy measured. Field trips were carried out between 2008 and 2009. Thus, we calculated the DBH (in cm), height (m), canopy volume (m^3) and the bole surface (in m^2) of each phorophyte based on these data.

The bole of the *E. guineensis* trees was divided into height classes of 1 m from the ground level (we used a 15-meter telescopic tree pruner, with demarcations at each meter as a height model). All epiphyte species and abundance were registered in each height class. Epiphyte species were identified in the field or in the laboratory with the help of identification keys and specialists. All species had botanical material collected for later deposit in the Herbarium of the Botany Department (RBR), Federal Rural University of Rio de Janeiro.

Data Analysis

The richness of epiphytes species (S) for the *SSEG* was obtained and the rarefaction curve calculated through Jackknife 1 and Chao 1 estimators (Magurran 2003, 78-84) in the EstimateS 9.1.0 software. The sampled epiphytes were included in one of four ecological categories, according to their relation with the phorophyte: characteristic holoepiphyte, facultative holoepiphyte, accidental holoepiphyte or hemiepiphyte (Benzing 1990, 31-34). The species were also classified according to their dispersal syndrome as zoochoric, anemochoric or sporochoric (autochoric).

The following phytosociological parameters were calculated for the epiphytic guild and for each epiphyte species individually in order to evaluate possible changes in the epiphyte community structure (modified from Giongo and Waechter 2004): absolute frequency (Afr), relative frequency (Rfr (%)); absolute frequency of occurrence (Afp), relative frequency of occurrence (Rfp (%));absolute phorophyte frequency (Paf); and relative phorophyte frequency (Prf (%)) (Waechter 1998).

The frequency of epiphyte species was measured at each height class and individually by phorophyte. This method enabled us to determine the pattern of vertical occupancy of epiphytes in the phorophytes. The non-parametric Kruskal-Wallis test was used (Zar 1999, 195-197) to evaluate whether the epiphyte species were distributed differently among the height classes of phorophytes. The Systat 11.0 software program was used for this analysis.

The epiphyte distribution pattern in each sampling unit (phorophyte) was measured through the Morisita Dispersion Index (DI) (where: $DI = 1$, random distribution; $DI < 1$, uniform distribution; $DI > 1$, clumped distribution) (Brower et al. 1997, 156-157). The F-test was used to verify the significance of the Morisita Dispersion Index ($DI \neq 1$) (Poole 1974, 116-117). Simple regression analysis was used to test whether the phorophyte size (in terms of height, DBH and bole surface) has an effect on the abundance and epiphyte species richness (Zar 1999, 324-358).

RESULTS

A total of 490 trees (178 *E. guineensis* individuals, 291 from other species, and 21 dead trees) were surveyed in the *SSEG*. Epiphytes were found in 49 oil palm and 303 individuals from other tree species from the total of sampled trees. We found 1264 epiphyte individuals belonging to 24 species, 22 genera and 15 families (Table 1). Among the specimens found, only one (4.2% of the total) was identified at the genus level (*Microgramma* sp.). Polypodiaceae and Araceae were the richest families, with four (22.2%) and three species (67.7% of total individuals), respectively (Table 1).

Table 1. Ecological category, dispersal syndrome and species code name of vascular epiphyte sampled in an *Elaeis guineensis* sub-spontaneous stand in *Ilha da Marambaia*, state of Rio de Janeiro, Brazil

Family/species	**Species code name**	**Ecological category**	**Dispersal syndrome**
Anacardiaceae			
Tapirira guianensis Aulb.	TAPGUI	AHO	Zoochory
Araceae			
Anthurium harrisii (Kunth) Engl.	ANTHAR	HEM	Zoochory
Monstera adansonii var. *adansonii* (Schott) Madison	MONADA	HEM	Zoochory
Philodendron cordatum Kunth	PHICOR	HEM	Zoochory
Arecaceae			
Elaeis guineensis Jacq.	ELAGUI	AHO	Zoochory
Bromeliaceae			
Tillandsia gardineri Lindley	TILGAR	CHO	Anemochory
Cactaceae			
Lepismium cf. *crussiforme* (Vellozo) Miquel.	LEPCRU	HEM	Zoochory
Cecropiaceae			
Cecropia pachystachya Trécul.	CECPAC	AHO	Zoochory
Commelinaceae			
Dichorisandra thyrsiflora J. C. Mikan	DICTHY	AHO	Zoochory
Dryopteridaceae			
Rumohra adiantiformis (Forst.) Ching.	RUMADI	FHO	Sporocory
Tectaria incisa Cav.	TECINC	AHO	Sporocory
Lomariopsidaceae			
Nephrolepis biserrata (S. W.) Schott.	NEPBIS	FHO	Sporocory
Melastomataceae			
Miconia prasina (S. W.) D.C.	MICPRA	AHO	Zoochory
Moraceae			
Ficus adhatodifolia Schott.	FICADH	HEM	Zoochory
Ficus sp	FICsp	HEM	Zoochory
Orchidaceae			
Encyclia oncidioides (Lindl.) Schltr.	ENCONC	CHO	Anemochory
Polypodiaceae			
Microgramma sp.	MICGEM	CHO	Sporocory
Pleopeltis pleopeltifolia (Raddi) Alston.	PLEPLE	CHO	Sporocory
Serpocaulon triseriale (S. W.) A. R. Sm.	SERTRI	CHO	Sporocory
Pteridaceae			
Adiantum latifolium Lam.	ADILAT	FHO	Sporocory
Hemionitis tomentosa (Lam.) Raddi	HEMTOM	FHO	Sporocory

Table 1. (Continued)

Family/species	**Species code name**	**Ecological category**	**Dispersal syndrome**
Vittaria lineata (L.) Sm.	VITLIN	CHO	Sporocory
Solanaceae			
Solanum argentums Duna ex. Poir.	SOLARG	AHO	Zoochory

Legend: AHO = accidental holoepiphyte; CHO = characteristic holoepiphyte; FHO = facultative holopiphyte; HEM = hemiepiphyte.

The rarefaction curves obtained through Chao 1 and Jackknife 1 estimators suggested an average species richness of 29.0 ± 5.1 and 29.1 ± 2.7, respectively, indicating that the number of species registered corresponds to approximately 80% of the expected, and that the applied methodology was effective to sample the largest number of estimated species for the study area.

With regard the ecological groups of epiphytes, characteristic and accidental holoepiphytes represented 30.4% each (S = 7), while hemiepiphytes represented 21.7% (S = 5) and facultative epiphytes represented 14.5% (S = 4) (Table 1). Among the dispersal syndromes, the largest number of species were zoochoric (S = 12; 52.2%), followed by spore dispersal (S = 9; 39.1%), and anemochoric (S = 2; 8.7%) (Table 1).

The *Monstera adansonii* and *Microgramma* sp. species had the highest absolute (Afr) and relative frequency (Rfr) values (Afr = 858 and 218, Rfr = 67.9% and 17.2%, respectively). The frequency data enabled dividing the species into four groups: dominant species (Rfr > 50%), species with high abundance (5 < Rfr < 50%), species with low abundance (1 < Rfr <5%) and rare species (Rfr < 1%) (Table 2).

Thirteen species (56.5% of total) had clumped dispersion (DI > 1), and the most abundant ones were more clumped (Table 2). The occurrence of individuals varied in more than half of the plots (56%); from 21 to 50 epiphytic individuals (an average of 24.4 ± 14.5 epiphytes/plot). The density of phorophytes ranged between one and 23 individuals, with an average of 9.2 ± 5.4 phorophytes/plot. There was a significant positive relation between the abundance of epiphytes and phorophytes in the plots ($F_{2, 48} = 37.621$, $p < 0.001$, $r^2 = 0.663$, N = 50) (Figure 1).

Table 2. Phytosociological analysis of epiphyte species from an *Elaeis guineensis* sub-spontaneous stand in *Ilha da Marambaia*, state of Rio de Janeiro, Brazil. Species are ordered according the absolute frequency

Species	**Afr**	**Rfr (%)**	**Afp**	**Rfr (%)**	**Paf**	**Prf (%)**	**DI**	**F**	**AC**
Monstera adansonii	858	67.9	47	94	402	82.0	1.34	6.93	C
Microgramma sp.	218	17.2	35	70	94	19.2	2.8	8.87	C
Philodendron cordatum	42	3.3	7	14	9	1.8	7.14	2.62	C
Miconia prasina	39	3.1	17	34	22	4.5	3.37	2.74	C
Nephrolepis biserrata	20	1.6	9	18	12	2.4	22.72	5.87	C
Serpocaulum triseriale	16	1.3	4	8	4	0.8	13.75	4.9	C
Pleopeltis pleopeltifolia	14	1.1	6	12	7	1.4	10.43	3.5	C
Solanum argentum	12	0.9	7	14	9	1.8	5.3	1.96	C
Hemionitis tomentosa	9	0.7	2	4	2	0.4	22.22	4.46	C
Lepismium crussiforme	7	0.6	2	4	3	0.6	26.19	4.08	C
Vittaria lineata	5	0.4	2	4	3	0.6	21.42	3.5	C
Adiantum latifolium	4	0.3	4	8	4	0.8	0	0.93	UN
Dichorisandra thysriflora	4	0.3	3	6	4	0.8	8.33	1.44	C
Ficus sp.	3	0.2	3	6	3	0.6	0	0.95	UN
Tapirira guianensis	3	0.2	2	4	2	0.4	16	1.63	C
Tectaria incisa	2	0.2	2	4	2	0.4	--	--	--
Rumohra adiantiformis	2	0.2	2	4	2	0.4	--	--	--
Anthurium harrisii	1	0.1	1	2	1	0.2	--	--	--
Cecropia pachystachya	1	0.1	1	2	1	0.2	--	--	--
Elaeis guineensis	1	0.1	1	2	1	0.2	--	--	--
Encyclia oncidioides	1	0.1	1	2	1	0.2	--	--	--
Ficus adhatodifolia	1	0.1	1	2	1	0.2	--	--	--
Tillandsia gardinerii	1	0.1	1	2	1	0.2	--	--	--

Legend: Afr = absolute frequency; Rfr = relative frequency; Afp = absolute frequency in quadrat; Rfp = relative frequency in quadrat; Paf = absolute frequency in phorophytes; Prf = relative frequency in phorophytes; DI = Morisita index; F = F test; AC = aggregation category (UN = uniform. C= clumped distribution).

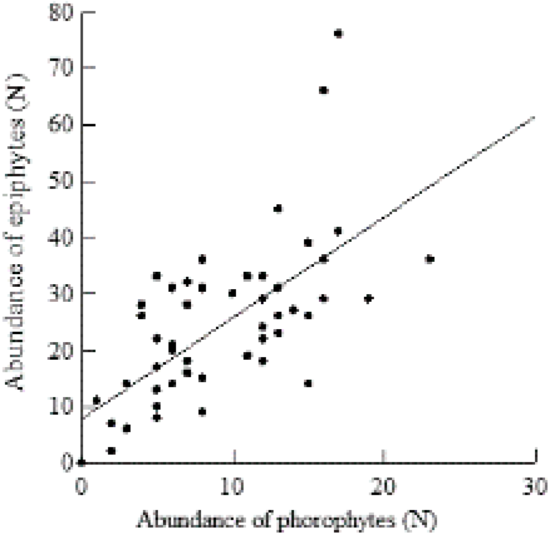

Figure 1. Relation between phorophytes and epiphytes in an *Elaeis guineensis* sub-spontaneous stand (an exotic palm species) in Ilha da Marambaia, Rio de Janeiro State, Brazil.

Epiphyte species preferably occupied the lower strata of phorophytes, especially below 5 m in height (Figure 2). *Microgramma* sp. and *Monstera adansonii* occurred in all height classes. However, *Microgramma* sp. mainly occupied the upper layers, and *M. adansonii* occupied the entire vertical gradient (Figure 2). Epiphyte species differed in the vertical distribution pattern (Kruskal-Wallis = 37.547; $p < 0.001$), with some species differing significantly from each other, such as *Microgramma* sp. and *Pleopeltis pleopeltifolia* (Dunn test, $p < 0.05$) (Figure 2; Table 3).

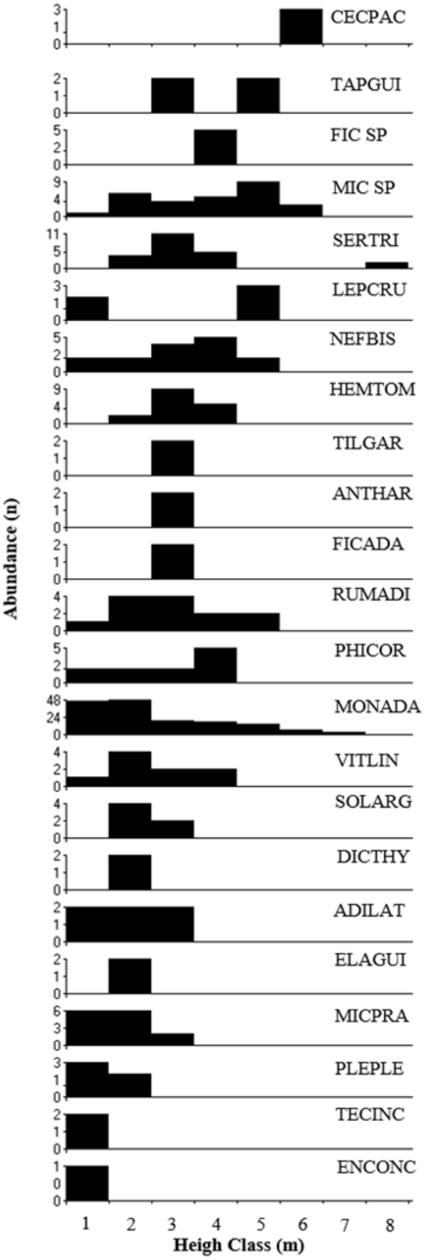

Figure 2. Vertical distribution pattern of epiphyte species on the oil palm stem in a sub-spontaneous stand in *Ilha da Marambaia*, Rio de Janeiro State, Brazil. Reference for species code names can be found in Table 1.

Table 3. Results of the Dunn pair test to evaluate the vertical distribution of vascular epiphyte species over the palm stem in a sub-spontaneous stand in *Ilha da Marambaia*, state of Rio de Janeiro, Brazil. In the upper matrix, p values are found, while in the lower matrix the results of the Dunn test are presented pair by pair. Reference for species code names can be found in Table 1

	NEPBIS	**PLEPLE**	**MICSP**	**RUMADI**	**SOLARG**	**TAPGUI**	**PHICOR**	**MONADA**	**LEPCRU**	**SERTRI**	**HEMTOM**	**VITLIN**	**ADILAT**	**MICPRA**	**DICTHY**
NEPBIS		0,137	0,329	0,151	0,137	0,065	0,758	1	0,305	0,281	0,305	0,218	0,166	0,137	0,058
PLEPLE	14,38		0,040	0,959	0,979	0,837	0,329	0,058	0,918	0,817	0,898	0,878	0,959	1	0,837
MIC	34,64	4,22		0,040	0,040	0,007	0,218	0,383	0,118	0,061	0,096	0,054	0,040	0,017	0,007
RUMADI	15,86	100,70	4,22		0,959	0,878	0,356	0,051	0,877	0,877	0,877	0,878	0,959	0,959	0,817
SOLARG	14,38	102,90	4,22	100,70		0,837	0,329	0,058	0,918	0,817	0,898	0,878	0,959	1	0,837
TAPGUI	6,81	87,93	0,74	92,16	87,93		0,199	0,016	0,663	0,626	0,626	0,590	0,878	0,798	0,979
PHICOR	79,62	34,64	22,93	37,38	34,64	20,98		0,555	0,573	0,505	0,505	0,505	0,383	0,305	0,182
MONADA	105,00	6,07	40,25	5,39	6,07	1,67	58,31		0,137	0,124	0,137	0,112	0,058	0,040	0,014
LEPCRU	32,03	96,42	12,37	92,16	96,42	69,61	60,13	14,38		0,979	0,918	0,858	0,918	0,798	0,626
SERTRI	29,56	85,84	6,43	92,16	85,84	65,74	53,02	13,01	102,90		0,959	0,979	0,837	0,798	0,626
HEMTOM	32,03	94,29	10,04	92,16	94,29	65,74	53,02	14,38	96,42	100,70		0,979	0,878	0,798	0,626
VITLIN	22,93	92,16	5,72	92,16	92,16	61,98	53,02	11,75	90,04	102,90	102,90		0,878	0,837	0,626
ADILAT	17,45	100,70	4,22	100,70	100,70	92,16	40,25	6,07	96,42	87,93	92,16	92,16		0,959	0,837
MICPRA	14,38	105,00	1,79	100,70	105,00	83,75	32,03	4,22	83,75	83,75	83,75	87,93	100,70		0,817
DICTHY	6,07	87,93	0,74	85,84	87,93	102,9	19,15	1,45	65,74	65,74	65,74	65,74	87,93	85,84	

Epiphytic species tended to occupy a wide range of phorophyte sizes, although there were differences in occupation between some species (Height: $F_{2,\ 23} = 3.839$, $r^2 = 0.363$, $p < 0.001$; DBH: $F_{2,\ 23} = 6.290$, $r^2 = 0.449$, $p < 0.001$; Surface: $F_{2,\ 23} = 6.280$, $r^2 = 0.449$, $p < 0.001$) (Figure 3).

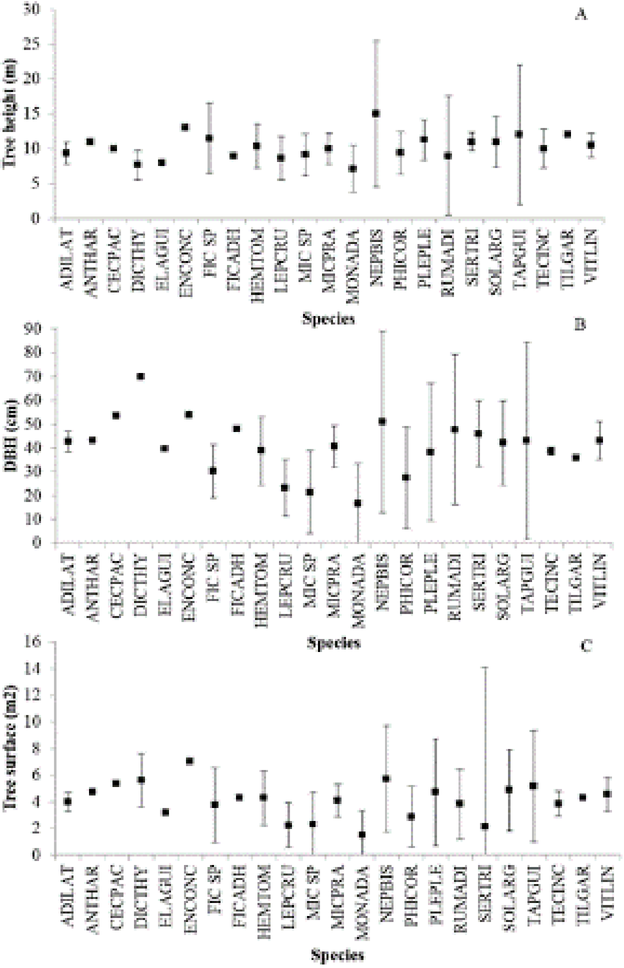

Figure 3. Occupation pattern of epiphytic species according to average phorophyte height (m) (± SD) (AA, DBH (diameter at breast height, cm) (B), and average phorophyte surface (m^2) (C) in a sub-spontaneous stand of the *Elaeis guineensis* exotic species in *Ilha da Marambaia*, Rio de Janeiro State, Brazil. ■ = average; ┬ = standard deviation. Reference for species code names can be found in Table 1.

The epiphyte species (S) richness in the *SSEG* responded significant and positively to the total height of the phorophyte ($F_{2,\ 48} = 24.016$, $p < 0.001$, $r^2 = 0.594$, $N = 46$) and bole surface ($F_{2, 48} = 6.691$, $p = 0.014$, $r^2 = 0.383$, $N = 46$) (Figure 4a and b), but did not respond to DBH ($p = 0.483$). Likewise, abundance (N) significantly responded to the same variables: total height of the phorophyte ($F_{2,\ 48} = 30.132$, $p < 0.001$, $r^2 = 0.638$, $N = 46$) and bole surface ($F_{2, 48} = 14.373$, $p = 0.001$, $r^2 = 0.519$, $N = 46$) (Figure 4c and d), but did not respond to DBH ($p = 0.830$).

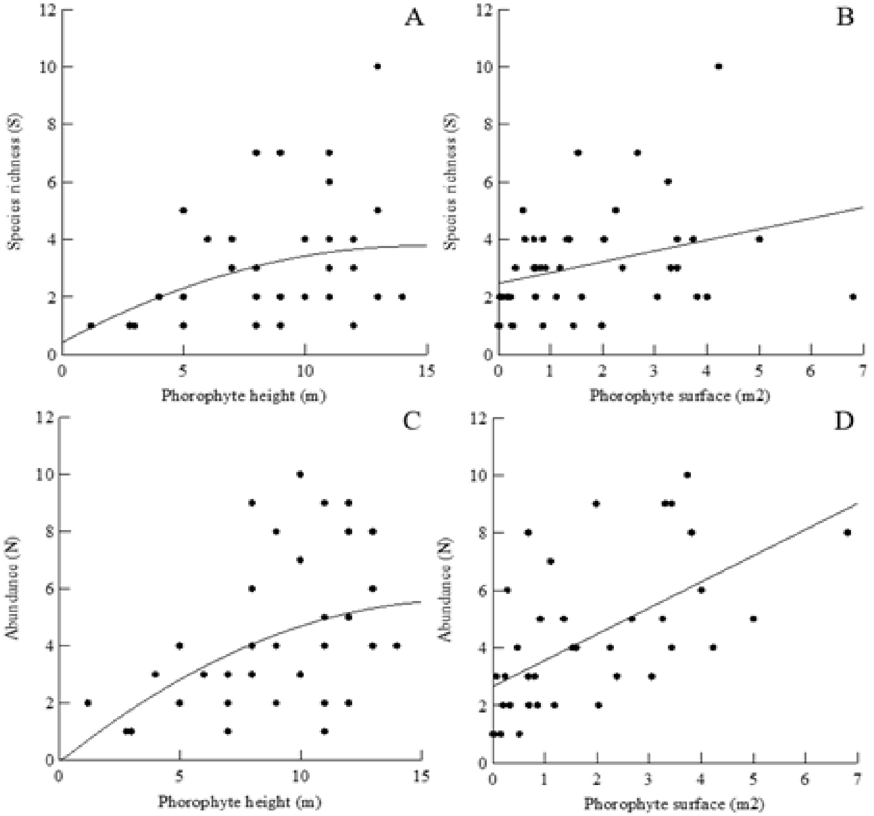

Figure 4. Relation between the species richness of epiphyte (S) and phorophyte height (m) (A); species richness of epiphyte (S) and phorophyte surface (m^2) (B); abundance of epiphyte (N) and phorophyte height (m) (C); and abundance of epiphyte (N) and phorophytes surface (m^2) (D) in an *Elaeis guineensis* sub-spontaneous stand (exotic palm species) in *Ilha da Marambaia*, Rio de Janeiro State, Brazil.

DISCUSSION

The epiphyte species richness in the *SSEG* corresponded to only 18.2% of the total epiphyte species in Ilha da Marambaia (Ribeiro et al. unpublished data). This can be considered low compared to other studies in areas of *Restinga* or ecotones between *Restinga* and Submontane Dense Ombrophilous Forest (Waechter 1986; Dislich and Mantovani 1998; Borgo and Silva 2003) because the latter are not monoculture areas. Structural characteristics of the habitat may alter the structure and regeneration of epiphytes (Benzing 1990, 224-229; Zotz and Vollrath 2003). The dominance of *E. guineensis* in the stand and the impact caused by falling leaves may alter the structure and regeneration of epiphyte species. This unstable environment may cause a direct effect on the epiphyte community, where only species resistant to the physical impact of sand carried by the wind and salinity are able to occupy the *E. guineensis* trunk, thus leading to the low species richness observed.

The characteristic and facultative holoepiphytes represented most of the species found (60.8%) among the sampled species. This was an expected result, even for a non-native stand (Benzing 1990, 224-229). Most of the species found are anemochoric or spore dispersed (autochoric) (Pteridophyta). These species are extremely dependent on humidity and water for reproduction, which may be an impediment for its population maintenance in the *Restinga*. On the other hand, epiphyte propagules find a nutrient-rich and humid substrate which favors germination when they reach the leaf sheath of *E. guineensis*. This may explain why epiphytes are common in the exotic stand. Therefore, the registered epiphyte species can be considered pioneer species (sensu Kersten and Kunyoshi 2009; Bataghin et al. 2010), because they are able to occupy isolated trees, open areas or high insolation areas, such as those sites dominated by *E. guineensis* (Foster et al. 2011). As a result, shade tolerant species (such as some Orchidaceae, Bromeliaceae, and Gesneriaceae) tend to only colonize these areas when lighting conditions are more favorable, being the last ones to occupy a new environment (Barthlott et al. 2001; Benavides et al. 2006; Wolf and Flamenco-S 2006, 375-391).

The hemiepiphytes comprised a quarter of the sampled species. This proportion is considered high, as they are normally observed in younger systems (Benavides et al. 2006). These plants only pass part of their life cycle as epiphytes, as they grow in the soil by germinating seedlings (Araceae, Moraceae and Pteridophyta) or by transporting viable body fragments (Araceae and Pteridophyta) (Benavides et al. 2006). Therefore, these species produce viable populations in a shorter period of time due to this kind of establishment pattern, which may also explain the high proportion of this group in the *SSEG*. Moreover, hemiepiphyte propagules can also find favorable germination and establishment conditions in the sheaths of senescent palm leaves (Marinho Filho 1999), which may increase the chances of survival and colonization of hemiepiphytes in an environment dominated by palms such as in the *SSEG*.

Nevertheless, one third of the registered species is composed of terrestrial plants, herein classified as accidental epiphytes. This result differs from the vast majority of studies with vascular epiphytes, which may be explained by the accumulation of organic material and moisture in the permanent leaf sheaths of oil palm. Most of the accidental epiphytes are terrestrial and typically zoochoric, as *E. guineensis* itself (Matuzak et al. 2008), *Tapirira guianensis* (Pizo et al. 2002), *Miconia prasina* (Alves et al. 2008a) and *Cecropia pachystachya* (Mello et al. 2004; Galetti et al. 2011). Seeds of these accidental epiphytes are carried by dispersers and fall into the overlapping and perennial *E. guineensis* leaf sheath. These seeds will germinate if they find favorable conditions for germination (nutrient and water).

Araceae and Polypodiaceae were the most representative families in the study area. Although these botanical families have a large number of epiphyte species (Benzing 1990, 27-29), they are not commonly the most representative families in floristic studies (Benzing 1990, 27-29; Waechter 1998; Nieder et al. 2000; Kersten and Silva 2001; Fontoura et al. 2009). Furthermore, Araceae and Polypodiaceae accounted for almost 90% of the individuals in the *SSEG,* mainly due to the dominance of *Monstera adansonii* (Araceae) and *Microgramma* sp. (Polypodiaceae). The pattern of few species being represented by many individuals (and several species being represented by few individuals) seems to be common for epiphytes (Kersten and Silva 2001; Gonçalves and Waechter 2002).

M. adansonii is commonly a generalist and inhabits forests and open areas (Temponi et al. 2006). *Monstera adansonii* can be considered a pioneer species occupying the stages of initial regeneration and open environments. These same ecological characteristics occur with the *Microgramma* genus, which is composed by pioneer species commonly found in different formations (mainly in open and urban areas) (Alves et al. 2008b; Bataghin et al. 2010). These ecological characteristics indicate that these species rapidly occupy the area when they find favorable environmental conditions. They can saturate the phorophyte microenvironment (Flores-Palacios and García-Franco 2006; Benavides et al. 2006), dominate the area and possibly reduce the chances of other species colonizing the trunks.

Monstera adansonii and *Micrograma* sp. had a clumped dispersion index (DI > 1), such as most of the epiphytes registered in this study (86% of total). This result was corroborated by the significant relation between epiphyte and phorophyte abundances, and constituting a pattern which has already been reported for many other species (Nieder et al. 2000). These relationships suggest that epiphytes tend to present clumped populations (Nieder et al. 2000). Such a pattern is related to two different strategies: pre-establishment (dispersion) and post-establishment (investment in clonal growth).

Dispersion at close range is common in epiphytes (Zotz and Vollrath 2003). In this case, propagules are placed in trees close to their parents and consequently in places (phorophytes) with similar and favorable microclimatic conditions, which increase the chances of germination, establishment and maintenance of small clumped populations (Hietz-Seifert et al. 1996; Nieder et al. 2000; Zotz and Vollrath 2003). After dispersal and establishment in favorable sites, most holoepiphytes species, like those sampled in this study (e.g., Bromeliaceae and Pteridophyta) invest in recruitment through clonal growth. They send a large number of shoots which can stay connected together or not (Nunes-Freitas and Rocha 2007). This type of growth found in different groups of epiphytes (Benzing

1990, 27-29; Mondragón et al. 2004) enables the species to quickly occupy the substrate, especially when environmental conditions are not favorable for investment in sexual reproduction (Mondragón et al. 2004). Thus, clonal reproduction increases the chances of obtaining and sharing resources such as light and water (Mondragón et al. 2004), maintaining dense populations in certain forest locations.

The epiphytes sampled in the *SSEG* had different vertical occupancy patterns; one group occupied the lower stratum (up to 5 m) and another group occupied the higher stratum. This distribution pattern and organization model of epiphytes is generally a response to light and vertical humidity gradients (Steege and Cornelissen 1989; Freiberg 1996; Kersten and Silva 2001; Nunes-Freitas and Rocha 2007). However, contrary to what has been observed in other studies, the lower stratum was richer. This pattern was only previously observed in a coastal forest in southern Brazil (Kersten and Silva 2001). This finding in the *SSEG* may result from the large accumulation of organic matter in *E. guianensis* leaf sheaths. As the lower stratum is older, it may have more organic matter accumulated in the leaf sheaths. These old leaf sheaths are exposed longer and may increase the chances of occupation by epiphytes.

Most species occupied phorophytes of similar size. The occupation of substrates with specific characteristics such as similar sized trees is related to the eco-physiological needs of epiphytes (Nieder et al. 1999). This suggests that the spatial organization and community structure of epiphytes follows the ecological model of niche differentiation (Futuyma and Moreno 1988; Kassen 2002). However, we suggest that only pioneer and generalist epiphytes, such as those which were most abundant in this study (e.g., *M. adansonii*, *Micrograma* sp.), would more widely occupy the environment in systems with low spatial heterogeneity, such as young forests or monocultures of exotic species, with a large variation of phorophyte sizes (Benavides et al. 2006). In addition, there would be reduced competition between different niches under unfavorable microclimatic conditions observed in the *SSEG* (i.e., low spatial heterogeneity and high light intensity), thereby allowing generalist species to broaden their space occupation (Hietz-Seifert et al. 1996; Barthlott et al.

2001; Padmawathe et al. 2004; Solis-Monteiro et al. 2005; Haro-Carrión et al. 2009; Boelter et al. 2011).

The relations found between *E. guineensis* morphometric parameters and species richness and abundance of epiphytes indicate that larger palms tend to have more complex and richer epiphytic flora. Similar results were observed in other forest types (Zotz and Vollrath 2003; Flores-Palacios and Garcia-Franco 2006; Poltz and Zotz 2011). As expected, larger phorophytes are older and they have been exposed to a propagule supply for a longer time, thus they present a higher number of niches and microhabitats (Benavides et al. 2006).

Although palm oil creates microclimatic conditions which enable the establishment of epiphyte species present in the *Restinga* and Atlantic Forest areas which limit the *SSEG*, this community undergoes considerable structural and functional changes. This is observed in the high number of species of accidental epiphytes, in the number of zoochoric species, and in the dominance of a small number of species whose ecological characteristics classify them as generalist pioneers. Despite the small number of studies carried out with interactions between epiphytes and palm trees (Marinho Filho, 1992; Martins, 2017; Barbarena et al. 2019), especially with oil palm (Adeleye et al. 2017), the results demonstrate that the presence of palm trees with permanent sheaths, especially exotic invaders, leads to changes in the structure of the vascular epiphyte community (Cruz et al. unpublished data). In the case of oil palm, which in Brazil has an invasive behavior (e.g., Zenni and Ziller, 2011), microclimate changes in the invaded environments can lead to a reduced loss in species populations, since the vast majority have specific ecophysiological requirements (Benzing, 1990). Thus, it is essential that oil palm plantations in use or abandoned be monitored and managed in order to reduce the chances of invasion of nearby preserved areas and the modification of these habitats and the loss of epiphyte species.

CONCLUSION

Our results show that *E. guineensis* monocultures can modify the structure and function of epiphyte communities by causing species loss (mainly those classified as characteristic holoepiphytes) and allowing only colonization by pioneer and generalist species. At the same time, species with greater ecophysiological requirements, even though dispersed for this habitat, are unable to establish themselves. On the other hand, oil palm creates conditions (organic matter and moisture accumulated in its permanent sheaths) for a greater number of terrestrial species to occupy the epiphytic pattern.

However, the number of studies analyzing the interaction of epiphytes with oil palm is small, which prevents further generalizations. Thus, it is essential that long-term studies are carried out on active and abandoned *E. guineensis* monocultures, seeking to understand the effects of this species on the vascular epiphyte community. In addition, areas invaded by oil palm also need to be managed and monitored so that we can understand the effects of the invasion of this species on different Brazilian forest formations.

In several parts of the world, including different Brazilian regions, oil palm monocultures have occupied extensive areas for different purposes, especially for the production of biofuels. If the managers of these areas do not elaborate plans for the control and management of oil palm invasion, they may colonize adjacent native areas, leading to changes in the structure and functional role of biological communities.

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Chapter 6

THE HUGE INCREASED INTEREST IN INVASIVE PLANTS AND CHANGES IN THE MAIN TOPICS STUDIED: AN ANALYSIS OF THE LAST TWO DECADES

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ABSTRACT

Humans have broken geographical barriers and have been dispersing species across the planet for a long time. Species, which are introduced into a new environment and are able to increase their population, can cause biological invasion. In turn, biological invasions are currently one of the main causes of biodiversity loss. The amount of information on invasion ecology has gradually increased since its establishment. However, there is a lack of knowledge on how many studies on invasive plant species were conducted in the last decades, as well as the main topics investigated and the countries where such studies were developed. Thus, I performed a literature review on invasive plant species to fill this gap, and used a novel analytical tool to verify if the number of articles published between 2000 and 2019 has increased. I also examined which disciplines and countries have contributed the most to this subject, and

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checked if terms and keywords have changed between decades. I found 14,113 articles published in that period with an increase of 219% from 2000–2009 to 2010–2019. Most articles (51%) were published in ecology and plant sciences journals. The occurrence of terms and keywords changed between decades: most topics from 2000 to 2009 were related to understanding and the impacts of invasive plants, while from 2010 to 2019 most topics focused on the control and management of such species. The United States and Australia published the most articles in the first decade, and then China, Germany, Spain (and the United States) contributed more from 2010 to 2019. In addition, the number of links between countries substantially increased from one decade to another. The results clearly indicate that research on invasive plant species gained much attention in the last 20 years and the interests in the subject have changed over time. Moreover, there was more countries involved and more cooperation between researchers and scientists to address this issue. Identifying such aspects is useful to know how the knowledge on invasive plant species developed and what topics are currently being investigated.

Keywords: biological invasion, invasive plant species, literature review, network data

INTRODUCTION

With the human population increasing as well as increasing exploitation of virtually all environments on the planet, biogeographical barriers have been broken and previously isolated populations are now expanding their range (Mooney and Cleland 2001, 5446). Many species taken by humans to new locations can become invasive and cause biological invasions, i.e., species are introduced outside of their native range and then successfully spread in the new environment (Levine 2008, 57). Scientists divide invasion into four different stages to better understand the invasion process, namely: introduction, establishment, spread, and impact (Levine 2008, 58).

Scientists have been warning about the introduction of new species, their impacts on native residents, and the homogenization of earth's biota for many years (Lodge 1993, 133). Today, despite some criticism (e.g., Gurevitch and Padilla 2004, 470), it has been proven that invasive species really impact their new environment, causing extinctions and altering the evolutionary pathway of native species (Mooney and Cleland 2001, 5446; Clavero and García-Berthou 2005, 110). For plants in particular, it was discovered that the establishment of invaders depends on their fitness advantage or niche differentiation in relation to the resident species (MacDougall, Gilbert, and Levine 2009, 609).

Information of this nature (and much more) are only possible because many studies were developed since the establishment of invasion ecology in the late 1950s (Elton 1958). However, the amount of information published on this issue is still little explored. A study conducted more than 10 years ago revealed that the number of articles on biological invasions were increasing rapidly, and new themes were arising (Richardson and Pyšek 2008, 162). As many years have passed, a new assessment on biological invasion is necessary, especially because new analytical tools are now available and they can provide a broader view on the subject. In addition to the number of articles published, it is also important to account for changes over time and the origin of these articles, i.e., where the studies were carried out.

In this chapter, I have revised the literature published on invasive plant species in the last 20 years to achieve the following objectives: (1) to verify if the interest in this subject has increased over time; (2) to examine which scientific disciplines have added more knowledge on invasive plants; (3) to check if the main topics investigated on invasive plants have changed between decades; and (4) to verify which countries have contributed the most to research on invasive plants. I predict that the number of published papers has increased in the last decades because invasive species became a serious problem in the Anthropocene. Moreover, as ecology deals with interactions, I suppose that it should be the main discipline investigating plant invasion issues. The interest in invasive plant species must have changed over time because new concerns and solutions have arisen in recent years.

METHODS

Data Survey

The data used in this study were obtained from the *Web of Science*® database of *Clarivate Analytics* (https://webofknowledge.com). I performed a basic search in the *Web of Science Core Collection* database using the terms “invasive AND plant AND species” within the category “topic.” I performed this search for two time periods: 2000–2009 and 2010–2019. Only *articles* were included in the refined results. The data about each article were exported as “Other file formats” (Tab-delimited (Win)). The number of articles published each year and for the 10 categories/disciplines with most articles were also recorded.

Data Analysis

The data were submitted to simple linear regression to verify if the interest in invasive plants has increased over time. The independent variable was the year, and the number of articles published was the dependent variable. The assumption of normality was assessed thorough the Kolmogorov-Smirnov and Shapiro-Wilk tests, and equality of variances through the Levene test. I performed these analyzes in SPSS version 23 software. I also plotted the number of papers published in the last 20 years for a visual analysis to check which disciplines have published more articles in the field of plant invasion.

Finally, I performed analyzes for the co-occurrence of terms (terms extracted from titles and abstracts), co-occurrence of keywords (all keywords), and co-authorship (countries) in the *VOSviewer* 1.6.13 software program (van Eck and Waltman 2010) to know and compare the main topics used in plant invasion and to verify which countries have contributed the most to this subject between 2000–2009 and 2010–2019. The *VOSviewer* is a useful tool for creating maps based on network data and for visualizing and exploring these maps (van Eck and Waltman 2019).

Bibliographic database files from *Web of Science* and *Scopus*, for example, can be used to construct networks of scientific publications, scientific journals, researchers, research organizations, countries, keywords, or terms (van Eck and Waltman 2019). Maps are constructed based on a co-occurrence matrix through three steps: a similarity matrix, VOS mapping technique, and translation, rotation and reflection (van Eck and Waltman 2010). I used the data downloaded from *Web of Science* to create the maps. The default settings for normalization (association strength method) and clustering were used in these analyzes. After creating the maps, I then compared the main terms and keywords used in both decades, as well as the main countries where the studies on plant invasion were performed.

RESULTS

There was a total of 14,113 articles retrieved in the data survey, in which 3,367 articles were published on invasive plant species from 2000 to 2009, and 10,746 articles were published on the same subject from 2010 to 2019. These values represent a 219% increase in articles published on invasive plant species from one decade to another. As expected, the number of articles published had a strong positive relationship with time ($F_{1,\ 18} = 3364.39$; $p < 0.001$, $R^2 = 99$) (Figure 1). Only 76 papers were published on invasive plants in the year 2000, but this number rose to 1395 in 2019.

Considering the scientific disciplines (or categories in the *Web of Science*), ecology and plant sciences contributed to 51% of the articles published between 2000 and 2019 (Figure 2). Disciplines such as agronomy, evolutionary biology, and marine freshwater biology contributed less articles on invasive plant species (Figure 2).

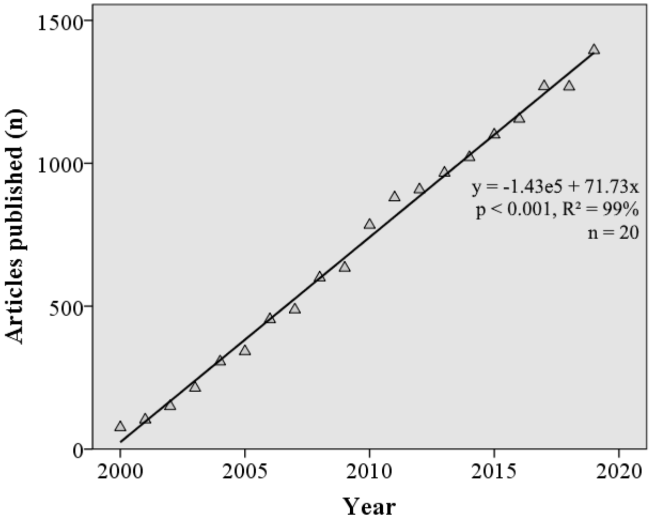

Figure 1. The number of articles published on invasive plant species greatly increased in the last two decades. Data retrieved from the *Web of Science* database.

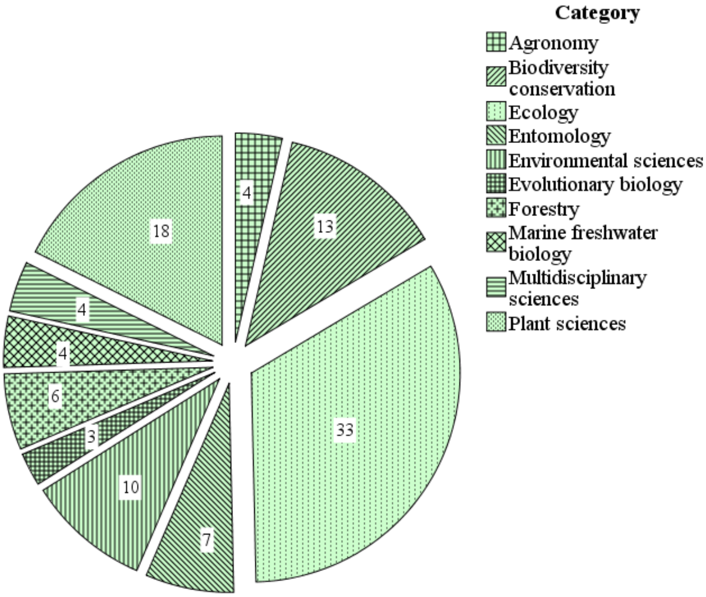

Figure 2. Percentage of articles published on invasive plant species between 2000 and 2019 according to the 10 most relevant categories/disciplines (categories with most publications).

The data survey for 2000–2009 amounted to 1,168 terms with six clusters being formed (Figure 3). Some of the most frequently occurring terms between 2000 and 2009 were “America” (347 occurrences), “biomass” (344), “cover” (300), “concentration” (200), “risk” (151), and “biological control” (138) (Figure 3). There were 3,489 terms found between 2010 and 2019, and six clusters were formed. In general, the terms used in that period differed from those of the previous decade (Figure 4). Some of the most common terms used between 2010 and 2019 were “cover” (718 occurrences), “Africa” (570), “identification” (314), “herbicide” (257), “map” (216) and “policy” (147) (Figure 4).

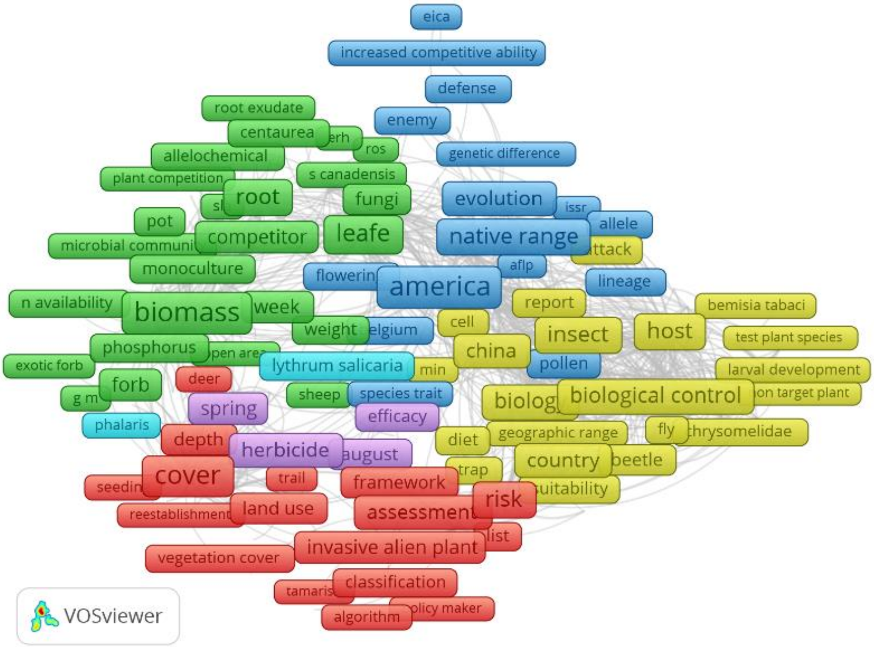

Figure 3. Network map of co-occurrence of terms for invasive plant species articles published between 2000 and 2009. The size of the label and frame of an item reflects the weight of the item. Larger frames have greater weight. Closer terms are more closely related (they are cited together).

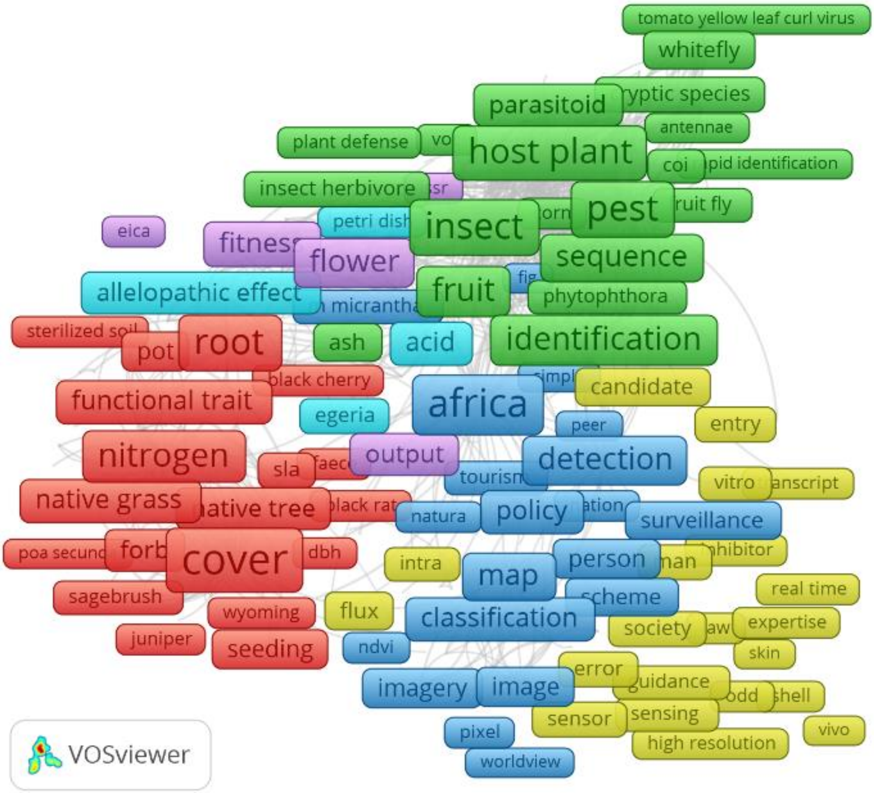

Figure 4. Network map of co-occurrence of terms for invasive plant species articles published between 2010 and 2019. The size of the label and frame of an item reflects the weight of the item. Larger frames have greater weight. Closer terms are more closely related (they are cited together).

Some of the main keywords used between 2000 and 2009 were "growth" (368 occurrences), "competition" (245), "biodiversity" (226), "evolution" (181), "ecology" (177), and "biological-control" (150) (Figure 5). Other keywords with a focus on management have emerged in the most recent decade (2010–2019). Topics such as "diversity" (1215), "management" (831), "impacts" (705), "restoration" (560), "biological control" (378), "resistance" (284), and "species distribution models" (127) were more common in recent years (Figure 6).

Figure 5. Network map of co-occurrence of keywords for invasive plant species articles published between 2000 and 2009. Keywords such as "invasive species," "invasive plants," "plants," and "biological invasions" were removed from the map because these terms were intrinsic in the data survey. The size of the label and frame of an item reflects the weight of the item. Larger frames have greater weight. Closer terms are more closely related (they are cited together).

Figure 6. Network map of co-occurrence of keywords for invasive plant species articles published between 2010 and 2019. Note that management terms were more common in this decade than in the previous one. Keywords such as "invasive species," "invasive plants," "plants," and "biological invasions" were removed from the map because these terms were intrinsic in the data survey. The size of the label and frame of an item reflects the weight of the item. Larger frames have greater weight. Closer terms are more closely related (they are cited together).

With regard to the countries where the studies were conducted or the authors were affiliated, the vast majority of the documents between 2000 and 2009 were from the United States (n = 1908) (Figure 7). Other relevant countries were Australia (255 documents), France (173), South Africa (161), and Canada (160). US researchers also maintained a strong bond with other countries, especially China (Figure 7, thicker yellow line).

US was also the main country publishing articles on invasive plant species between 2010 and 2019 (Figure 8). However, unlike the previous decade, many other countries significantly contributed to information on invasive plant species in that period. China (990 documents), Germany (580), Spain (483), Brazil (411), and Italy (401) were some examples of countries which contributed on this subject. Another important finding was that the co-authoring network between countries has greatly increased (Figure 8, lines connecting countries).

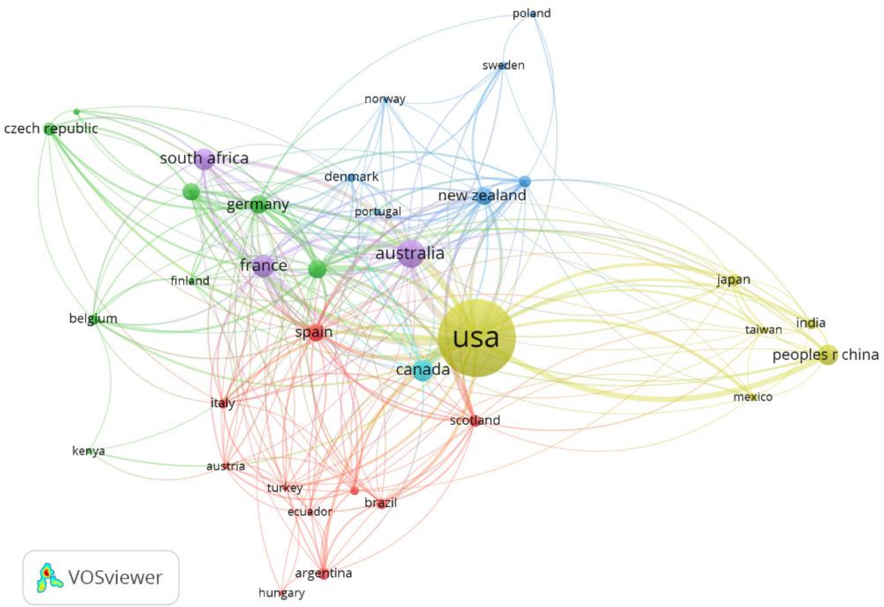

Figure 7. Network map of co-authorship (countries) for invasive plant species articles published from 2000 to 2009. The size of the label and circle of an item reflects the weight of the item. Larger circles have greater weight. Lines indicate the number of links (co-authorship) between countries.

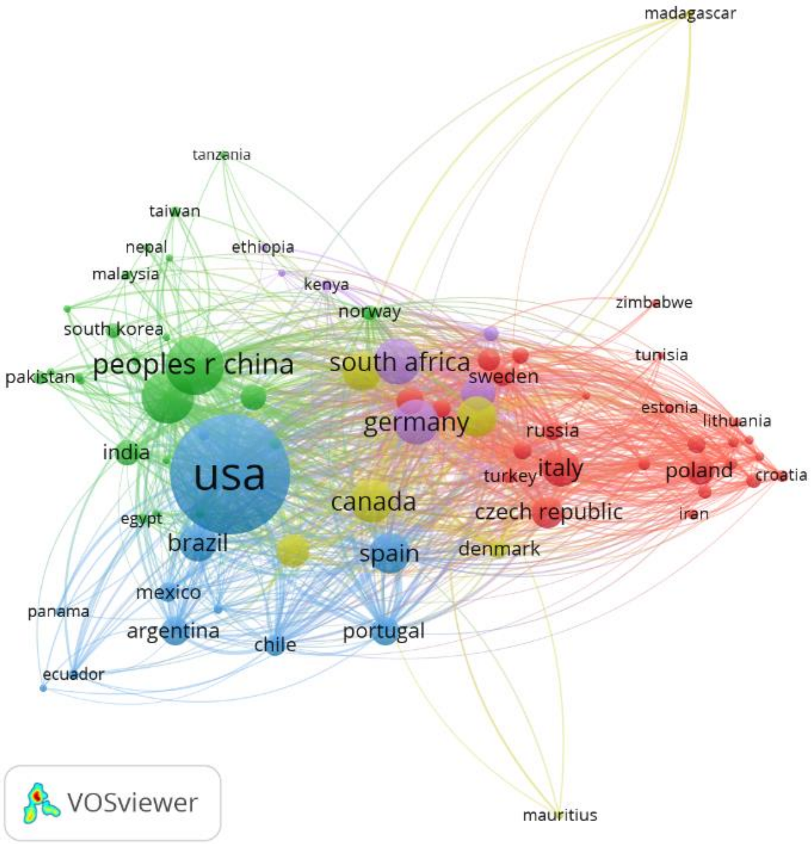

Figure 8. Network map of co-authorship (countries) for invasive plant species articles published from 2010 to 2019. The size of the label and circle of an item reflects the weight of the item. Larger circles have greater weight. Lines indicate the number of links (co-authorship) between countries. Note that the number of links has increased greatly.

DISCUSSION

This research quantitatively demonstrated the great increase of studies on invasive plant species in the last decades and the strong relationship between the number of articles published and time. This great increase reveals the interest of academics and managers on this subject, especially because invasive species have serious consequences on biodiversity and ecosystem services worldwide (Pyšek and Richardson 2010, 25). The social perception of invasive species has also been gaining attention more recently (Luna, Edelaar, and Shwartz 2019, 71; Shackleton et al. 2019, 10). This huge number of studies has brought new information and techniques to assess biological invasion (Kumschick and Richardson 2013, 317), and the scientific community today can confirm that biological invasions cause major impacts and biodiversity loss (Courchamp et al. 2017, 13).

An interesting result to notice was the massive increase in articles published between 2000–2009 and 2010–2019, indicating that the interest in invasive plants abruptly increased in recent years. More people are becoming concerned with the subject and trying to understand the mechanisms, impacts and how to manage invasive plant species in different parts of the world. This statement is also corroborated by the increase in number of links between countries from one decade to another. Invasive species now occur on a global scale and humans have to coexist with them (managing and controlling), as the costs are so high that their complete elimination is unfeasible (Head et al. 2015, 318).

As predicted, most studies on invasive plants were published in ecology, but also in plant sciences categories. As ecology deals with plants, the plant ecology field, which investigates the relationships between plants and their physical and biotic environment (Hull, Neufeld, and Gilliam 2019, 528), I already expected invasive plants to be studied a lot in this category. The same is valid for plant sciences (or botany), which studies all aspects of living plants. In addition, it was interesting to notice that other disciplines such as agronomy and forestry have been publishing articles related to invasive species, certainly because invasive plants can harm agronomic and forestry activities. For example, in the United States non-indigenous plants reduce about 12% in crop yields and billions of dollars are lost every year because of them (Pimentel 2008). Moreover, some plant species used for agriculture and forestry can also become invasive, especially exotic ones (e.g., *Pinus* spp.) (Lavi et al. 2005, 85; Reichard and White 2001, 113).

With regard to the terms used in invasive plant species over time, we could see that the terms between 2000 and 2009 were more related to understanding biological invasion. For example, the terms cover, land use, root, biomass, competitor, allelochemicals, spring, August, native range, and risk may be interpreted as attempts to know the performance and interaction of invasive plants with non-invasive ones. On the other hand, the main terms used between 2010 and 2019 (e.g., imagery, map, policy, surveillance, detection, identification, and sequence) may be attributed to the identification and inspection of invasive plants.

The same pattern can also be viewed when observing the common keywords over time. The keywords in the 2000s were more related to the characterization of invasive processes. Evolution, competition, biodiversity, urbanization, climate, allelopathy, spread, and residence time may be related to the understanding of how invasive plants behave and affect other plants. In fact, it was recognized that a growing number of studies addressed connections between ecology and evolution on biological invasions up to 2010 (Vanderhoeven et al. 2010, 193). Then in the 2010s, the keywords became related to management; for example, restoration, management, species distribution models, biological control, resistance, tolerance, and grazing. Invasive species management is an important tool to prevent or mitigate the various problems which arise from introducing non-native species (Crowley, Hinchliffe, and McDonald 2017b, 133). Economic, social, and ecological dimensions of invasive species management have gained attention in recent years because management has been increasing in scope and scale (Crowley, Hinchliffe, and McDonald 2017a, 351; Holden, Nyrop, and Ellner 2016, 712).

A term or keyword that I expected to be common in the recent decade (2010–2019) but was not is climate change. Actually, more papers linking invasive species and climate change were published in the 2000s. My expectations regarding these terms (invasive plants and climate) occurred because both climate change and invasive species are “hot” terms and affect biodiversity worldwide (Willis and Bhagwat 2009, 806; Clavero and García-Berthou 2005, 110). Moreover, climate change not only affects native plant species, but also invasive ones of important agronomic value with consequences for human food supply (Ziska et al. 2011, 13). On the other hand, keywords which have arisen in the recent decade were those related to statistical modeling (e.g., model, models, and species distribution models). Such models are useful tools which are employed to predict the potential invasive range and to optimize strategies for controlling non-native species (Barbet-Massin et al. 2018). The use of models is increasingly common.

Changes also occurred in the number of (and which) countries where the studies were carried out, indicating that more people worldwide are becoming interested in invasive plants. The considerable increase in co-authors of different nationalities was also a relevant finding because the invasive plant species subject is of general interest. Many experiments were performed in different ecosystems worldwide due to these collaborations, and a lot of information is becoming available. For example, we now know that research based on invasive-plant traits is important and the distinction between native and alien species should be explicit in studies (van Kleunen, Dawson, and Dostal 2011, 317). Moreover, the dynamics of plant invasions through all stages of invasion are now better understood, as well as the various impacts caused by these species (Henderson, Dawson, and Whittaker 2006, 25). Biological invasion science increased greatly in the last 50 years (Richardson and Pyšek 2008, 161), but new challenges are arising. Rapid warming and intensified anthropic activities in the Artic, as well as genomic modifications and expanding transportation networks are some examples of new challenges in plant invasion (Ricciardi et al. 2017, 464). Collaboration between different institutions and countries must continue, so that biological plant invasion may provide answers for these challenges.

CONCLUSION

In this chapter, I found that the number of articles published on invasive plant species increased greatly between 2000 and 2019. This information was not yet explicit, and shows how invasive plants (their understanding, impacts, and management) have gained attention in recent decades. I also showed that changes occurred in the main terms and keywords used in the published articles. The number of co-authors from different countries has also increased, and studies on invasive plants are now published in different parts of the world. This information enabled identifying changes in the main interests on invasive plants over time, as well as the evolution of knowledge in this field. I suggest that similar investigations should be carried out for other organisms (i.e., animals), so that we can have a broader view on studies of invasive species in general.

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Chapter 7

INVASIVE WOOD DERIVED NANOPOROUS CARBONS FOR MITIGATION OF WATER AND AIR POLLUTION

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ABSTRACT

Activated carbons (ACs) are well-known adsorbents widely used in a variety of applications due to large specific surface areas arising from their highly developed porous structures. Continuous growth of activated carbons consumption is especially linked to their use in mitigation of environmental pollution. Novel precursors as well as sustainable, efficient activation processes are being steadily investigated to fulfill the forecasted increasing demand. In particular, invasive wood species as non-conventional precursors for activated carbons development may

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represent an interesting alternative, that has been scarcely explored. The present chapter reports chemical activation of invasive *Parkinsonia aculeata* wood into activated carbons using solutions of potassium carbonate and potassium hydroxide as main activating reagents at pre-established experimental conditions. Potentialities of the resulting ACs for mitigation of water and air pollution were evaluated. A commercial activated carbon was also used for the sake of comparison. Physicochemical characteristics indicate that the activated carbons obtained with both reagents show similar BET surface areas of ~770–780 m^2/g and predominance of pores smaller than 2 nm, even though contribution of these pores to the total pore volume was greater for those obtained with KOH. Effectiveness of the carbons in adsorption of phenol, as representative model of priority water contaminants, was investigated. The effects of the adsorbent's dose and solution pH were first examined to establish optimal conditions. Kinetics of phenol adsorption onto the derived activated carbons was determined and properly described by a pseudo-second order model. Adsorption isotherms were also obtained and well represented by conventional models. High adsorption capacities point to possible use of the ACs for the removal of phenolic compounds usually found in industrial wastewater. Their performance exceeds that of the commercial sample possessing a higher BET surface area but a lower volume of pores smaller than 2 nm. On the other hand, the capacity of the samples developed with K_2CO_3 solution in adsorption of CO_2, the largest anthropogenic greenhouse gas on the planet contributing to the global warming, was investigated. CO_2 adsorption capacity from gaseous streams approaching post-combustion conditions was evaluated by gravimetric adsorption tests and breakthrough curves determined in a fixed-bed column at 30°C and 50°C and atmospheric pressure. The amounts of CO_2 adsorbed as determined by both methodologies were around the order of those reported in the literature for similar conditions, and would be associated to the nanoporous structure of the ACs. Accordingly, the nanoporous carbons derived from invasive *P. aculeata* wood demonstrate ability for their potential use in the removal of contaminants from water and air.

Keywords: greenhouse gases, invasive wood-based activated carbons, *Parkinsonia aculeata*, phenolic compounds removal, post-combustion CO_2 capture, wastewater

INTRODUCTION

Invasive species represent a significant threat to ecosystems, having serious environmental, economic and social impacts. Understanding, predicting and controlling the spread of invasive species is crucial to protect biodiversity and natural ecosystems, as well as to contribute to mitigate related deleterious effects (De la Fuente and Beck 2018, 2557-80).

Invasive woody plant species are of particular concern because their rapid expansion may hinder land use, restrict food production, replace natural ecosystems as barriers for many human pathogens facilitating their spread, among other serious repercussions (Sikorska et al. 2019, 1-16). Management of invasive plant species is complex and requires a lot of effort for eradication (Nourani et al. 2018, 29). In the last years, control through utilization has been proposed as an approach to improve management, and some strategies have been reported for transformation of invasive tree species into useful, added-value products (O'Donnell et al. 2019, 142-9). They include possible uses as building material, wood for charcoal, as well as applications as biofuel and biofilter medium for water (Dominguez et al. 2014, 301-8; Ellsworth et al. 2018, 577-85). In order to fulfill the increasing demand promoted by global sustainability trends for water and air treatment as well as greener process alternatives, some studies have also explored the feasibility of employing wood from invasive species as non-conventional precursor for the production of activated carbons and/or other sorbent materials (De Celis et al. 2009, 217-23; Nabarlatz et al, 2012, 109-15; Cukierman et al. 2012, 1-46; Gurevich-Messina et al. 2016, 551-7; Saravanakumar et al. 2019, 1-14). Activated carbons are known to play a key role in liquid and gas phase applications throughout chemical process industries, ranging from conventional applications, such as municipal water purification and flue gas treatment, to cutting-edge applications, such as adsorbed natural gas storage and double-layer capacitors (Cukierman 2013, 1-31; Koehlert 2017).

Activated carbons from a given raw material used as a precursor may be tailored for specific applications by proper selection of the process and operating conditions employed for their preparation. In particular, our research group has been earlier concerned with the conversion of invasive *Parkinsonia aculeata* L. wood into activated carbons by phosphoric acid activation (Cukierman et al. 2012, 1-46; Nunell et al. 2016, 347-56). *P. aculeata* belongs to the Fabaceae family. This woody plant species is native from America but it is also found in several regions all over the world. It forms dense thickets, preventing access for humans, native animals and livestock to waterways, grows easily on most types of soils, and withstands a great deal of heat, surviving in dry areas and for long prolonged dry spells. Besides, seeds remain viable for many years; the seedpods float, and the plant spreads by dropping pods into water, or pods are washed downstream by seasonal flooding (Cochard and Jackes 2005, 13-31; Cukierman et al. 2012, 1-46). Control of *P. aculeata* is still difficult. Therefore, accounting for the need for novel precursors for the manufacture of activated carbons as well as for sustainable activation processes in terms of intended applications, it is worth continuing to explore the possibility of converting invasive *Parkinsonia aculeata* wood into activated carbons for environmental applications as well as to examine their behavior for this purpose in terms of main physicochemical-textural characteristics.

In this direction, the present chapter deals with chemical activation of invasive *Parkinsonia aculeata* wood into activated carbons using solutions of potassium carbonate and potassium hydroxide as main activating reagents at pre-established experimental conditions. Potentialities of the resulting activated carbons for water mitigation is examined by evaluating the effectiveness of the developed carbons in the removal of phenol from water, as representative model of priority water contaminants. Besides, the capability of carbon samples to capture CO_2 from gaseous streams under post-combustion conditions was also investigated. This is of outmost importance since CO_2 is the greenhouse gas with the largest impact on climate change. At present, CO_2 represents 76% of the total source of greenhouse gases, mainly arising from huge amounts released into the atmosphere due to combustion of fossil fuels in power plants, with detrimental impacts on both environmental and ecosystem conditions (Mukherjee et al. 2019, 46-63). Development of cost-effective technologies for the separation and capture of CO_2 are crucial for reducing CO_2 emissions on an industrial scale. Currently, most of commercial CO_2 capture plants use amine-based processes and wet scrubbing systems, that are energy intensive due to the large amount of water needed in these systems. Other drawback related to these processes is amine degradation by oxidation leading to corrosion of process equipments (Ding and Liu 2020, 116382). Adsorption is considered as one of the potential options because of the low energy requirement, cost advantage, and ease of applicability over a relatively wide range of temperatures and pressures, even though this approach requires easily regenerable and durable adsorbents with high CO_2 adsorption capacity and selectivity.

METHODS

Preparation of the Invasive Wood-Based Activated Carbons

A piece of trunk (0.30 m length, 0.15 m cross-section diameter) of *P. aculeata* was collected in a field of Buenos Aires province, Argentina. It was debarked, air dried, crushed, milled, and sieved. Fractions of 1-2 mm average particle diameter were selected for preparation of the activated carbons (ACs). They were washed and dried at 60°C until constant weight was attained (Nunell et al. 2015a, 278). A schematic representation of the procedure applied for ACs preparation is displayed in Figure 1.

Briefly, the first stage consisted in carbonization of the wood sawdust used as precursor in a horizontal fixed-bed stainless steel tubular reactor, externally heated by an electric furnace. The sample was heated under flowing air (150 mL/min) at 10ºC/min up to 350ºC keeping this temperature for 1 h (Figure 1). The carbonized sample was then impregnated either with solution of potassium carbonate (K_2CO_3) (50 wt%) or potassium hydroxide (KOH) (50 wt%) with a chemical reagent/carbonized precursor weight ratio of 1. The mixture was dried at 80ºC in an oven. The impregnated samples were further thermally treated in the same reactor abovementioned for the carbonization stage. They were heated from room temperature to 300ºC at 10ºC/min, and held for 2 h. Subsequently, the temperature was raised to 800ºC at the same heating rate and held for 2 h before cooling down (Figure 1). The whole activation process was carried out under a nitrogen flow rate of 150 mL/min. The resulting material was washed with 0.1 N HCl acid solution and with distilled water until absence of chloride ions in the washing water. The washed samples were then dried in an oven at 100ºC to constant weight. The activated carbons obtained with the K_2CO_3 and KOH solutions are labeled as ACC and ACH, respectively. Yields in both cases were obtained from weight differences.

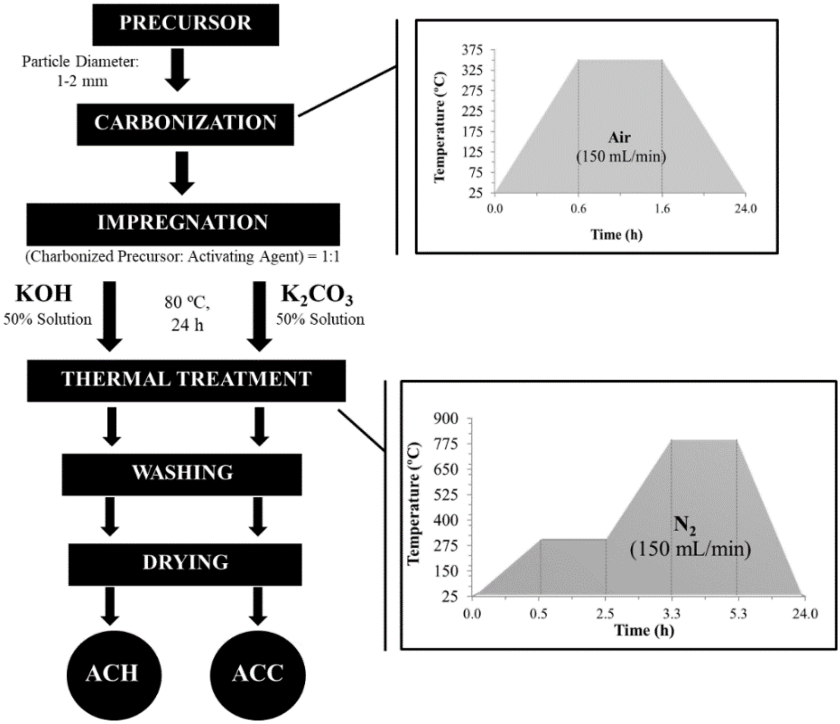

Figure 1. Schematic diagram of the chemical activation process applied for conversion of the invasive wood sawdust into activated carbons. ACH and ACC indicate the activated carbons obtained with solutions of potassium hydroxide and potassium carbonate, respectively.

Chemical and Textural-Morphological Characterization

Ash content and elemental composition of the precursor, the ACs and the commercial sample used as a reference (CAC) were assessed. The former was performed according to the American Society of Testing and Materials (ASTM) standards with a thermal analyzer TA instrument SDT Q-600. Elemental compositions were determined with an automatic instrument (Carlo Erba EA 1108 CHNS-O).

Contents of the total and individual surface acidic/polar oxygen functionalities (carbonyls, phenols, lactones, and carboxyl groups) for the ACs and the commercial sample were determined following a procedure earlier described in detail (Nunell et al. 2012, 87-95). Briefly, 0.5 g of each sample was mixed with 50 mL of a 0.05 N solution of sodium ethoxide, sodium hydroxide, or sodium bicarbonate in closed flasks and agitated for 24 h at room temperature. Then, the suspensions were filtered and 10 mL aliquot of each resulting filtrate was added to 15 mL of a 0.05 N HCl acid solution. The resulting solutions were back-titrated with 0.05 N NaOH. The total content of basic surface groups was also assessed. For this purpose, 0.5 g of each sample was added to 50 mL of 0.05 M HCl solutions. The slurries were stirred for 24 h and afterward filtered. A 10 mL aliquot of the resulting solutions was subsequently titrated with 0.05 N NaOH. Values are expressed as milliequivalents per gram of sample.

To get information about the nature of the carbons surface, pH and the point of zero charge (pHzc) were determined. The pH was measured according to the following procedure: 1 g of carbon was contacted with 100 mL of distilled water, and the mixture was boiled for 5 min. The suspension was cooled down to ambient temperature and the pH was measured to the nearest 0.1 pH unit. The pHzc of each sample was estimated from the pH of a concentrated dispersion as detailed elsewhere (Nunell et al. 2015b, 102-8). Briefly, 0.6 g of each activated carbon was added to 10 mL of a 0.1 M $NaNO_3$ solution. The suspensions were kept in a shaker thermostatically controlled with an external circulating bath at 25°C (Lauda Ecoline E200), until equilibrium was attained. Afterward, they were filtered with 0.45 μm Nylon membranes and the equilibrium pH was measured with a pHmeter (Consort C860).

N_2 adsorption–desorption isotherms at (-196ºC) for the derived ACs, and the commercial carbon were determined with an automatic volumetric sorption analyzer (Micromeritics ASAP-2020 HV). Prior to gas adsorption measurements, the samples of 0.1–0.15 g were outgassed at 120ºC overnight under flowing He employing a degassing station (Micromeritics Flowprep 060). Textural properties were assessed from the isotherms, according to conventional procedures earlier depicted in detail in own previous studies (Bonelli and Cukierman 2015, 141-67). The Brunauer–Emmett–Teller (BET) surface area (S_{BET}) was determined by the standard BET procedure. Total pore volumes (V_t) were estimated from the amount of N_2 adsorbed at the relative pressure of 0.95 ($P/P_o = 0.95$). The mean pore widths (W) were calculated from $W = 4\ V_t/S_{BET}$. Pore size distributions (PSDs) of the ACs were also calculated from N_2 adsorption isotherm data using the DFT Plus Software (Micromeritics Instrument Corporation), based on the non-local density functional theory, that assumes slit pore shape. In addition, adsorption isotherm of CO_2 at 0°C was determined for the ACs obtained with K_2CO_3 using the abovementioned instrument. The Dubinin–Radushkevich (DR) equation was applied to calculate the volume of pores smaller than 2 nm (V_{DR}) (Ramos et al. 2011, 475-508).

The surface morphology of the ACs was examined by scanning electron microscopy (SEM) analysis using a scanning electron microscope (Zeiss Supra 40®). The images were taken with an in-lens detector and 3 kV acceleration voltages. The samples were placed on an aluminum holder, supported on conductive carbon tape and sputter coated.

Phenol Adsorption Experiments

To examine the effectiveness of the ACs developed with both reagents in phenol adsorption, batch experiments at room temperature were performed using model dilute solutions. A stock solution (22 mmol/L) was first prepared by dissolving the necessary amount of analytical grade phenol in distilled water. This solution was then diluted to obtain standard solutions of concentrations in the range between 0.5 and 11 mmol/L. Concentrations of phenol in solution were determined by UV–visible spectrophotometry employing a Shimadzu UVmini-1240 model instrument (Shimadzu Corp., Kyoto, Japan) at $\lambda = 270$ nm (Valente-Nabais et al. 2009, 904-10).

The effect of the sample's dose on the adsorption of phenol was first investigated by contacting different weighed amounts of each sample (0.05–1.5 g) with 50 mL of 1.6 mmol/L phenol solutions up to equilibrium. The pH was adjusted to pH = 6 by dilute NaOH or HCl solutions. The pH value was selected on the basis of earlier studies concerned with phenol adsorption onto other activated carbons (Soto et al. 2011, 1-27). Capped glass flasks containing the slurries were kept in a shaker at 300 rpm and thermostatically controlled at 25 º C, with an external circulating bath, until equilibrium was attained. To check for possible pH drift during the assays, the pH was recorded at the start and at the end of each experiment. No variations were determined with pH values maintained within 0.2 units. Once equilibrium was attained, the slurries were filtered through 0.45 μm membranes and phenol concentrations in solution were determined as already detailed. Percentages of phenol removal were calculated according to:

$$Removal\left[\%\right] = 100\left(\frac{C_o - C_e}{C_o}\right) \tag{1}$$

where C_o and C_e are the initial and equilibrium phenol concentrations in solution, respectively.

Adsorption isotherms of phenol onto the ACs were determined by contacting 0.5 g of each sample with 50 mL of phenol solution of varying initial concentrations (0.5–11 mmol/L). The general procedure depicted above was followed, keeping the same experimental conditions (T = 25ºC, pH = 6, 300 rpm). The amount of phenol adsorbed at equilibrium per mass of activated carbon, q_e (mmol/g), was calculated according to:

$$q_e = \frac{(C_o - C_e)}{D} \tag{2}$$

where D is the dose of sample used (g/mL). In all cases, equilibrium conditions were verified from preliminary assays for prolonged times.

All the experiments were performed at least twice. Differences between replicates were less than 3% in all cases. Average values are reported. Furthermore, solute and adsorbent blanks were simultaneously conducted for control in all the assays under the corresponding conditions.

CO_2 Adsorption Assays

CO_2 adsorption assays for the ACs developed with K_2CO_3 were carried out by thermogravimetric analysis. 10 mg of the sample (particle diameter ≤44 µm) was placed in a thermobalance (TA Instrument SDT Q-600), and heated at 120ºC under a stream of N_2 (100 mL/min, 99.9% v/v), up to constant weight. Once the temperature was reached, the samples were cooled to 30ºC; then, the N_2 flow was changed to CO_2 (100 mL/min, 99.9% v/v) and the increment in weight, representing the CO_2 adsorption capacity, was registered.

Furthermore, dynamic experiments were conducted in a fixed-bed column (7 cm length and 2.2 cm internal diameter) packed with 2.5 g of the ACC of 1-2 mm particle diameter. The concentration of CO_2 in the feed stream was 20% v/v (balanced with N_2), and two different gas flow rates (50 and 70 mL/min) were employed to examine the effect of the total inlet gas flow. The composition of the inlet gaseous stream was established by regulating control valves and capillary flowmeters. The bed was maintained at a temperature of 30ºC by means of a tubular furnace commanded by an electronic controller, based on a microprocessor (NOVUS N2 2000). Gas samples were collected in hermetic vials from output streamline at different times. CO_2 concentrations were determined with a gas chromatograph (Shimadzu GC 8A) equipped with a Porapak N column and a thermal conductivity detector which used Ar as the carrier gas. Prior to each run, desorption of the ACs was conducted by circulating N_2 at 150ºC for 30 min until CO_2 absence in the output stream.

RESULTS AND DISCUSSION

Yields and Physicochemical Characteristics of the Activated Carbons

Overall yields for the ACs developed with K_2CO_3 and KOH solutions were ~20–21% in weight. These values were comparable to some others reported in the literature using other precursors and similar activation processes (Valizadeh et al. 2016, 1-13). Low yields may be attributed to appreciable volatilization that takes place as a result of the thermal decomposition of the precursor at low temperatures, before chemical reactions with the activating agent that are thermodynamically feasible at high temperatures proceed (Lillo-Ródenas et al. 2004, 1371-75).

Ash content and elemental compositions of the invasive wood used as precursor, the derived ACs, and the commercial sample are reported in Table 1. pH and pH_{ZC} values are also listed in the table.

Carbonization of *P. aculeata* wood and further activation led to noticeable changes in chemical characteristics of the resulting activated carbons, depending on the chemical reagent used. The standard error of values reported in Table 1 did not exceed 5%. The results in the table indicate that the ACs developed with both reagents possessed relatively higher contents of ash, C, and N, and lower contents of H and O than the precursor (PA), owing to release of volatile matter during the thermal treatment stage involved in the activation process. Percentages of C and H of the samples obtained with the KOH solution were lower than those developed with K_2CO_3 and the commercial sample. Differences in the reaction systems involved in the activation process depending on the chemical reagent employed may explain the different characteristics of the resulting ACs (Lillo-Rodenas et al. 2004, 1371-75).

Table 1. Ash content, elemental composition for *Parkinsonia aculeata* wood (PA) precursor and the ACs developed with solutions of KOH (ACH) and K_2CO_3 (ACC), and for the commercial activated carbon (CAC). Values of pH and pH_{ZC} for the ACs

Sample		**PA**	**ACH**	**ACC**	**CAC**
Ash (wt%, dry basis)		1.4	3.6	3.1	5.8
Elemental analysis (wt%, dry, ash-free basis)	C	48.0	74.7	80.6	86.3
	H	5.7	0.5	1.6	2.1
	N	0.2	0.5	0.4	0.6
	O[a]	46.1	24.3	17.4	11.0
pH			7.0	7.2	3.0
pH_{ZC}			6.8	7.5	4.1

[a] Estimated by difference.

Neutral values of pH were determined for the ACs developed from the invasive wood. Instead, the commercial sample showed a strong acid character, suggesting that its manufacture took place through an acid activation process. Similarly, pHzc values for the invasive wood-based ACs were higher than the one corresponding to the commercial sample.

Chemical characteristics of activated carbons are assigned to surface chemical heterogeneity, which is related to the presence of heteroatoms in the carbon structure, such as oxygen, nitrogen, hydrogen, sulfur, and phosphorus. The type and quantity of these elements depend either on the nature of the precursor or on their introduction during the activation process. Surface functional groups derived from these heteroatoms and the delocalized electrons of the carbon structure are responsible for the acidic or basic character of the activated carbon surface. The acidic character of the activated carbon surface is closely related to oxygen containing surface groups, which are mainly present on the outer surface or edges of the basal plane (Bandosz and Ania 2006, 159-229). Functional groups postulated as the sources of surface acidity include carboxylic acid or carboxylic anhydride, lactone, phenols, and carbonyls. In turn, the basic character of the activated carbons surface is ascribed to resonating π-electrons of carbon aromatic rings that attract protons, and basic surface functionalities (e.g., nitrogen containing groups) that are capable of binding with protons. Certain oxygen containing surface functionalities, such as chromene, ketone, and pyrone can also contribute to carbon basicity, although the basic character of activated carbons results primarily from delocalized π-electrons of graphene layers, acting as Lewis bases (Shafeeyan et al. 2010, 143-51).

In Figure 2 individual contents of acid surface functional groups and total contents of basic functionalities are displayed. The presence of different oxygen-containing functional groups on the surface of activated carbons is a relevant specific characteristic because it has great impact on their adsorption capabilities (Cukierman 2013, 1-31).

Activation of the carbonized wood with both reagents induced the formation of acid and basic functionalities (Figure 2). Both ACs showed similar quantities of carbonyls (0.6 meq/g), phenols and lactones (0.6-0.7 meq/g), but only the one developed with the K_2CO_3 solution possessed carboxyl groups even though in minor proportion (0.1 meq/g). Reportedly, decomposition of strong acidic functionalities, carboxylic, anhydrides and lactones, takes place at lower temperatures, while the weaker acidic functionalities, such as carbonyl, phenol and quinone, decompose at higher temperatures (Shafeeyan et al. 2010, 143-51). The sample obtained with the carbonate solution also possessed a slightly greater content of basic groups (0.9 meq/g) than the AC prepared with KOH (0.7 meq/g), reflecting differences in the reaction systems involved in each case. Carbonization of the precursors before impregnation and subsequent thermal treatment could lead to enhance the development of surface functionalities of basic character in the resulting ACs (Lorenc-Garabowska et al. 2013, 235-43). Instead, the commercial activated carbon showed complete absence of basic functionalities, a noticeably greater proportion of carboxyls and smaller number of carbonyls than the developed ACs. Differences in the surface chemistry of the ACs and the commercial sample highlighted the strong influence of the precursor and/or activation process employed for their development. pHzc values were in line with functionalities present on the surface of the samples.

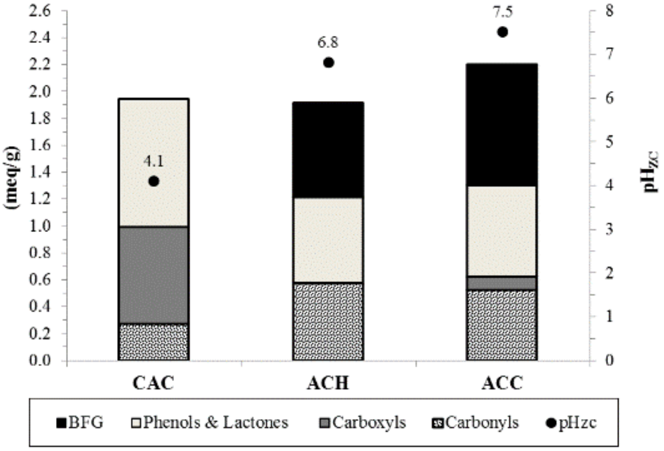

Figure 2. Contents of individual acid surface functional groups and total content of basic functionalities (BFG) of the activated carbons developed with solutions of KOH (ACH) and K_2CO_3 (ACC), and the commercial activated carbon (CAC), along with pHzc values.

Figure 3 displays N_2 adsorption isotherms for the developed ACs and the commercial sample. Textural parameters, as evaluated from these isotherms, are reported in Table 2.

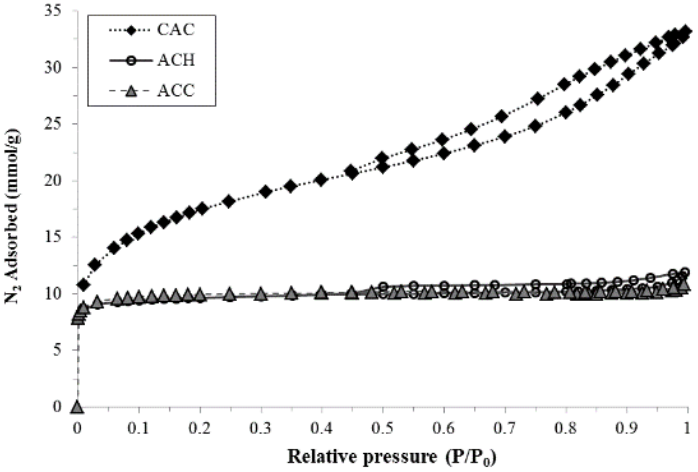

Figure 3. N_2 adsorption isotherms (-196°C) for the activated carbons developed from the invasive wood with solutions of KOH (ACH) and K_2CO_3 (ACC), and the commercial activated carbon (CAC).

The isotherms in Figure 3 represent the amount of N_2 adsorbed per unit mass of sample as a function of the relative pressure (P/P_o), being P, the equilibrium pressure, and P_o, the saturation pressure of the adsorbate at the working temperature (-196°C). The isotherms for the ACs developed with both reagents showed typical type I shape, according to IUPAC classification, pointing to predominance of pores <2 nm in the porous structures (Rouquerol et al. 2014, 18-114). The results in Table 2 indicate similar BET areas and total pore volumes for the ACs developed with both reagents. They showed a predominant presence of nanopores <2 nm, although it was more pronounced for those obtained by KOH activation (73% vs 60% of contribution to the total pore volume). Greater development of nanopores in KOH activation has been attributed to collaborative effects of chemical activation that entails redox reaction of KOH to carbonate, *in situ* physical activation that involves carbon gasification with CO_2, and expansion of carbon lattices due to intercalation of the potassium compound (Rashidi and Yusup 2017, 277-90).

Table 2. Textural characteristics of the ACs developed with both chemical reagents and the commercial sample calculated from N_2 adsorption isotherms

Sample	**S_{BET}[a] (m^2/g)**	**V_t[b] (cm^3/g)**	**V_n[c] (%)**	**V_m[d] (%)**	**W[e] (nm)**
ACH	768	0.37	73.2	26.8	1.9
ACC	777	0.35	60.0	40.0	1.8
CAC	1424	1.10	26.3	73.7	3.0

[a] BET surface area; [b] Total pore volume; [c] Volume of pores smaller than 2 nm calculated by the DFT method; [d] Volume of mesopores calculated by difference, [e] Average pore diameter.

Instead, the isotherm for the commercial sample presented a noticeably different shape, with features resembling those characteristics of isotherms of Type I and II, according to IUPAC classification, and a hysteresis loop associated to slit-shaped pores (Fernandez et al. 2014, 437-45). The commercial carbon adsorbed larger volumes of N_2 than the developed adsorbents overall the relative pressure range, indicating a more pronounced porous structure development. The BET surface area and total pore volume of the commercial carbon were significantly higher than the values estimated for the invasive-wood based activated carbons, with predominant presence of mesopores (>2 nm). The average pore diameter of this sample was also larger than that for the wood-based carbons (Table 2).

CO_2 (0°C) adsorption isotherms onto the ACs developed with both activating reagents are illustrated in Figure 4. Textural parameters calculated from these isotherms are listed in Table 3. Besides, Figure 5 shows pore size distributions obtained from the CO_2 isotherms.

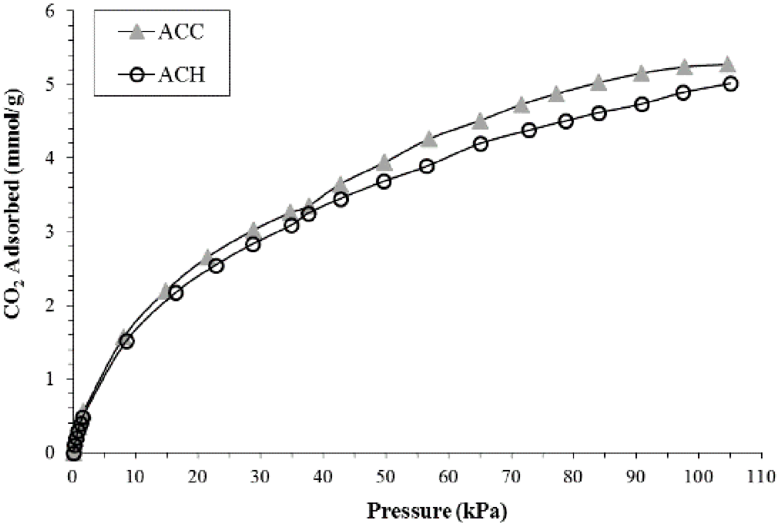

Figure 4. CO_2 adsorption (0°C) for the activated carbons obtained by activation with solutions of KOH (ACH) and K_2CO_3 (ACC).

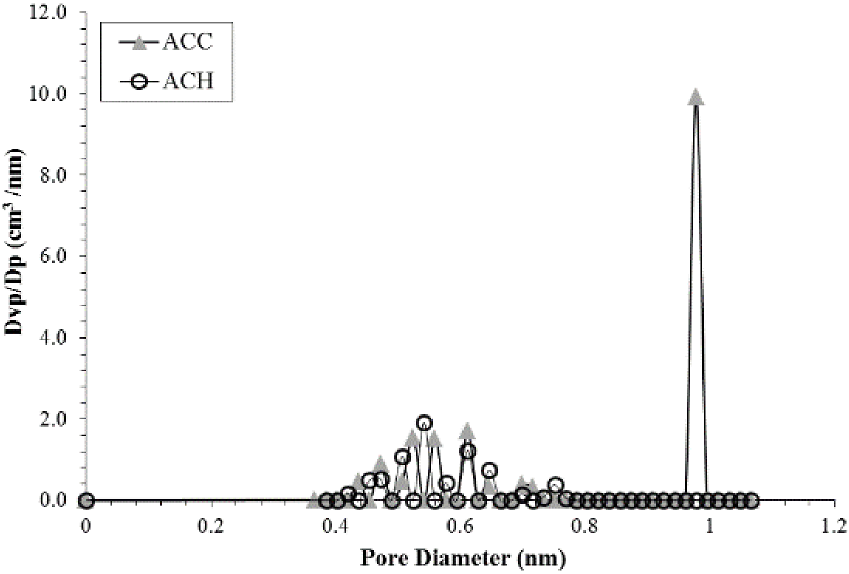

Figure 5. Pore size distributions calculated by the density functional theory to CO_2 isotherms (0°C) for the ACs obtained with the solutions of K_2CO_3 (ACC) and KOH (ACH).

CO_2 adsorption isotherms for both ACs were quite similar (Figure 4). Pore size distributions evaluated from these isotherms showed nanopores smaller than 1 nm, with the samples obtained with KOH possessing all nanopores in the narrow range 0.4-0.8 nm (Figure 5). Parameters in Table 3 indicate similar volumes of nanopores (V_{DR}), as evaluated by the DR equation to CO_2 adsorption data. Likewise, total amounts of CO_2 adsorbed at the highest pressure (100 kPa), that represent the maximum adsorption capacity at the investigated temperature, were also similar (5.0-5.3 mmol/g). Values of CO_2 adsorbed were among the highest ones reported for other activated carbons, ranging between 1.3 and 6.9 mmol/g for the same conditions (0°C, 100 kPa) (Gomez-Delgado et al. 2019).

Table 3. Parameters calculated from CO_2 (0°C) adsorption isotherms for the ACs developed with the solutions of K_2CO_3 (ACC) and KOH (ACH)

Parameter	**ACC**	**ACH**
V_{DR} (cm^3/g)	0.38	0.35
CO_2 Adsorbed (mmol/g STP)	5.3	5.0

Different morphological characteristics can be noticed between the samples depending on the chemical reagent employed and its reactivity (Figure 6). The ACH displays a spongiform-like structure with rough, not defined endings presumably because of the greater reactivity of KOH. For the ACC, the directional pattern of the wood original vascular structure seems well preserved. The cell structure shows smooth walls of considerable thickness, with sharp, well-defined edges. Aggregates of particles of different size and shape on the surface and inside the porous structure of both ACs may be observed. They could arise from thermal decomposition and/or fragmentation of the cell walls of the precursor owing to the high temperature employed during the thermal treatment as well as to possible condensation of carbonaceous material (Cukierman and Bonelli 2017, 1-30).

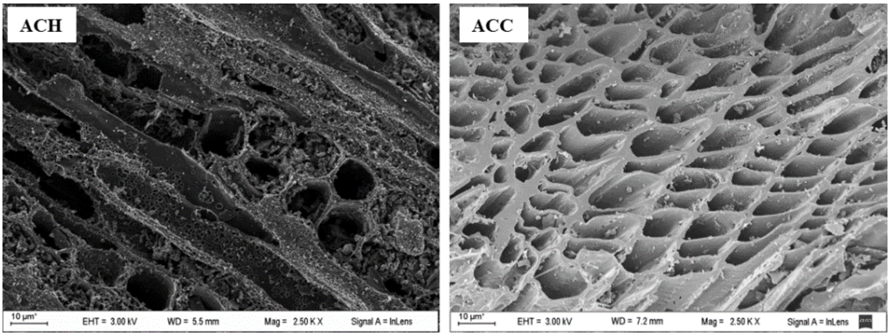

Figure 6. SEM images of the ACs obtained by activation with solutions of KOH (ACH) and K_2CO_3 (ACC). Magnification: X2500.

EFFECTIVENESS OF THE DERIVED ACTIVATED CARBONS IN PHENOL ADSORPTION

The effect of the sample's dose on the removal of phenol from solutions of 1.6 mmol/L initial concentration, at equilibrium, is illustrated in Figure 7. As expected, increasing the dose of the ACs led to enhancements in phenol adsorption for all the samples because of the greater availability of adsorption sites. Main differences among the ACs developed with both reagents and the commercial sample corresponded to doses smaller than 0.2 g/50 mL. As seen in the figure, phenol removal higher than ~95% was achieved for the developed ACs with doses of 0.1 g/50 mL, while the removal for the commercial sample was considerably lower. Accordingly, the results indicate that invasive wood-based ACs were more effective for phenol removal than the commercial carbon, attaining high levels of removal with low doses.

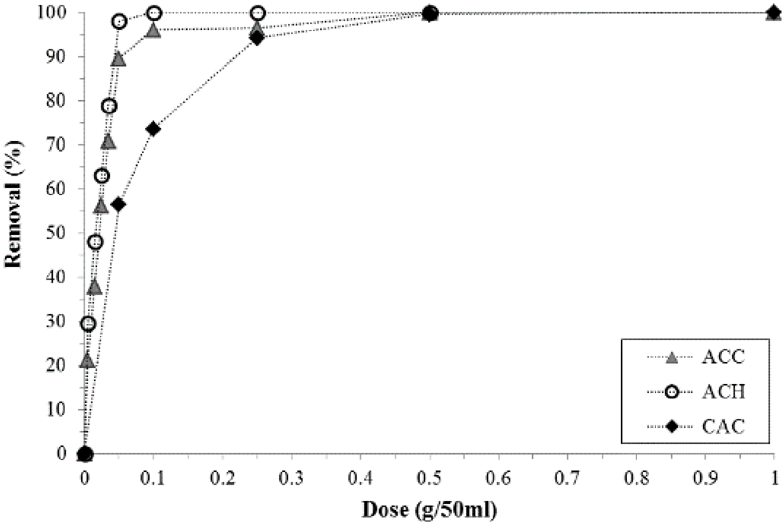

Figure 7. Effect of the adsorbent's dose on phenol adsorption at equilibrium conditions for the ACs obtained by activation with solutions of KOH (ACH) and K_2CO_3 (ACC) and the commercial sample (CAC). Experimental conditions: Co = 1.6 mmol/L; pH = 6; t = 4 h; T = 25°C. Lines are included only to guide the eyes.

In Figure 8, the influence of the solution pH on phenol adsorption onto the ACs is displayed. As seen, phenol adsorption onto the developed ACs remained almost complete and invariable for a wide range of solution pH, almost up to 7–8, thus indicating that it is favored at acidic pH values. At pH higher than 9, phenol adsorption decreased sharply. The trend may be explained taking into account changes that undergone the surface charge of the activated carbons with variations in the solution pH, as a consequence of the ionization state of the surface functional groups of the samples and phenol present in solution. Since pKa of phenol at 25°C is 9.89, adsorption was mainly favored at pHs smaller than 9. At higher pH, phenol should be dissociated and negatively charged. As the pHzc for the developed ACs was lower than 7.6, the ACs might be positively charged at solution pHs below this value, thus favoring interactions of electron donor/acceptor type between the aromatic ring of phenol and functional groups on the ACs surface. As the pH increased, the ACs surface should turn progressively negatively charged and, therefore, phenol adsorption decreased. Reduction in phenol adsorption at pH higher than 9 might be due to repulsion interactions between phenol and the negatively charged surface of the ACs. The behavior agreed with other results reported in the literature (Soto et al. 2011, 1-27).

Kinetic data for phenol adsorption onto the developed ACs and the commercial sample are shown in Figure 9. Instantaneous quantities of phenol adsorbed per mass unit of sample (q_t) are represented as a function of contact time (t). A pseudo-second order kinetic model was found to best represent the experimental results. The model is described by the following expression (Ho 2006, 681-9):

$$q_t = \frac{{q_{e_H}}^2 K_H t}{1 + q_{e_H} K_H t} \tag{3}$$

where q_{eH} are the amounts of phenol adsorbed at equilibrium per unit mass of sample (mmol/g), and K_H, the adsorption rate constant (g/mmol min). Predictions of the pseudo-second order kinetic model are also displayed in Figure 9. Model characteristic parameters are summarized in Table 4. They were estimated by nonlinear regression analysis for a 5% significant level by minimizing the sum of the squared differences between the observed and predicted values of the dependent variable.

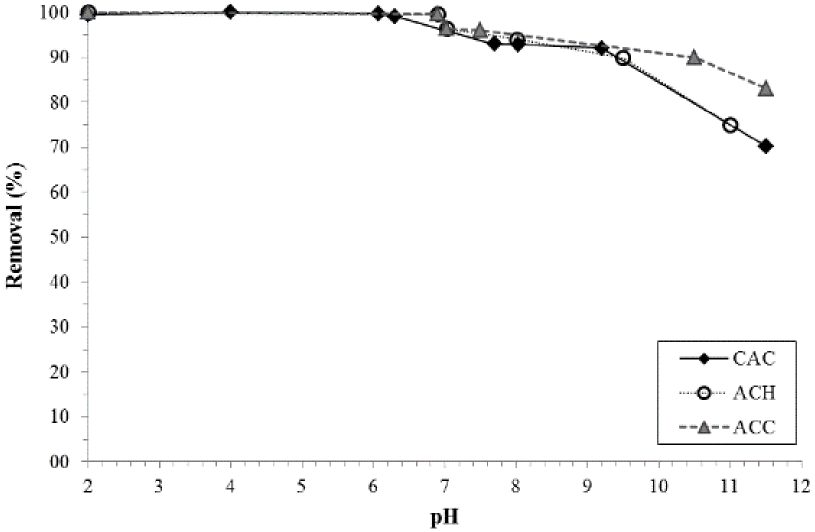

Figure 8. Effect of initial solution pH on phenol adsorption at equilibrium for the ACs developed from invasive wood by activation with solutions of KOH (ACH) and K_2CO_3 (ACC) and the commercial sample (CAC). Experimental conditions: Co = 1.6 mmol/L; T = 25°C; dose = 0.5 g/50 mL.

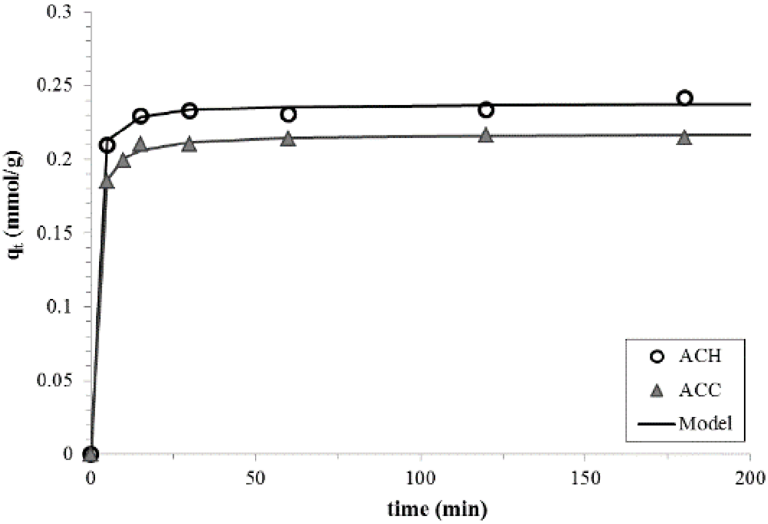

Figure 9. Kinetics for phenol adsorption onto the activated carbons developed with solutions of KOH (ACH) and K_2CO_3 (ACC). Comparison between the experimental data and predictions of the pseudo-second order kinetic model. Experimental conditions: Co = 1.6 mmol/L; pH = 6; T = 25°C; dose = 0.05 g/ 50 mL.

Table 4. Kinetic model characteristic parameters evaluated for phenol adsorption onto the invasive-wood based ACs

Sample	q_{eH} **(mmol/g)**	K_H **(g/mmol min)**	R^2
ACH	0.24	7.0	0.998
ACC	0.22	5.3	0.999

As seen in Figure 9, phenol adsorption onto both ACs increased sharply, approaching rapidly equilibrium. The model enabled to describe properly the experimental kinetic curves. The adsorption rate constant for the sample developed with KOH was higher than the one estimated for the ACC, thus indicating higher adsorption rate.

Adsorption isotherms are helpful in describing the interaction of adsorbates with adsorbents, and the equilibrium distribution of adsorbate molecules between the liquid and solid phases. They are relevant for the optimal utilization of adsorbents (Njoku et al. 2014, 198-204). Figure 10 shows phenol adsorption isotherms determined for the invasive-wood derived ACs and the commercial activated carbon. Amounts of phenol adsorbed at equilibrium (q_e) are represented as a function of the equilibrium concentration (C_e). Conventional two-parameter adsorption models were applied in order to represent the experimental isotherms for the three samples. The Freundlich's model, which assumes adsorption onto heterogeneous solid surface and adsorption energy sites of exponential type, led to the best fit of the adsorption data, as may be appreciated in Figure 10.

The Freundlich model is represented by the following equation (Freundlich 1906, 385-470):

$$q_e = K_F C_e^{\,n_F} \tag{4}$$

where K_F is the Freundlich isotherm constant, which represents the adsorption capacity of the system, whereas n_F is another constant indicating the adsorption intensity. Characteristic parameters of the model are presented in Table 5. They were estimated by non-linear regression analysis for a 5% significance level.

The results in Table 5 indicate that phenol adsorption capacity of the carbons developed from the invasive wood with both reagents were similar and almost duplicated the value estimated for the commercial sample, even though the former ones had a BET surface area lower than the latter. Accordingly, the porous network should not exert a predominant effect on phenol adsorption. Instead, surface functional groups of the developed activated carbons, likely basic functionalities, that were absent on the surface of the commercial sample, could play a key role. On the other hand, values of n_F pointed to similar adsorption intensity between phenol and each adsorbent.

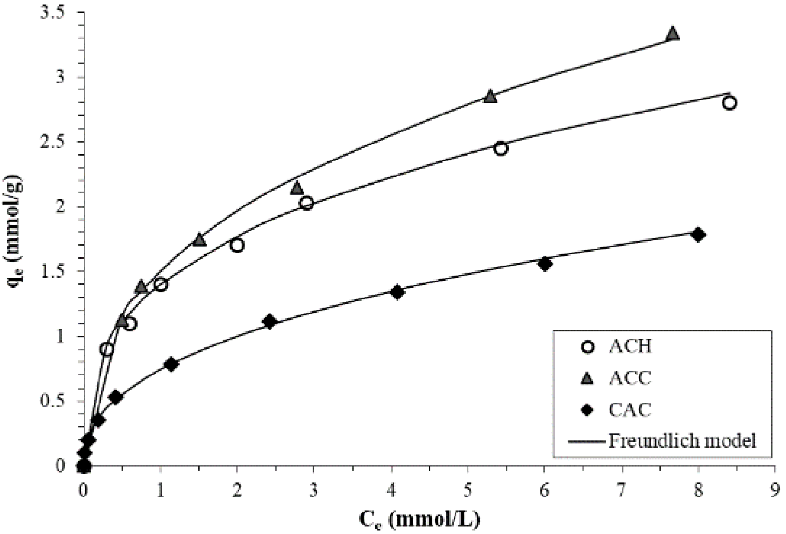

Figure 10. Phenol adsorption isotherms for the ACs obtained by activation with solutions of KOH (ACH) and K_2CO_3 (ACC) and for the commercial activated carbon (CAC). Comparison between the experimental data and predictions of the Freundlich model. Experimental conditions: Co = 0.5-11 mmol/L; pH =6; T = 25°C; D = 0.05 g/ 50 mL.

Table 5. Characteristic parameters of the Freundlich's model estimated for phenol adsorption onto the ACs developed with solutions of KOH (ACH) and K_2CO_3 (ACC), and the commercial sample (CAC)

Sample	n_F	K_F [(mmol/g) (L/mmol) ^{nF}]	R^2
ACH	0.34	1.40	0.997
ACC	0.38	1.50	0.999
CAC	0.43	0.74	0.999

EFFECTIVENESS OF THE ACTIVATED CARBONS IN CO_2 ADSORPTION AT POST-COMBUSTION CONDITIONS

The present section provides information of key data for techno-economic analysis and design aspects for adsorption units targeted at post-combustion CO_2 capture involving the activated carbons developed with K_2CO_3 solution (ACC). Figure 11 shows amounts of CO_2 adsorbed onto the ACC as a function of time. As detailed in the experimental section, they were obtained from CO_2 adsorption thermogravimetric experiments at 30°C and 50°C.

In order to represent kinetic data, they were fitted to two kinetics models: the Linear Driving Force (LDF) model and the Pseudo-Second-Order (PSO) model, by nonlinear regression. The LDF model assumes that the difference in concentration between the bulk phase and the adsorbed phase is the driving force for adsorption. All mass transfer resistances, namely external, intraparticle, and intramicropore, are lumped together as a single barrier to diffusion. The LDF model is described according to the following equation (Landaverde-Alvarado et al. 2017, 40-8):

$$q_m = q_e\left(1 - e^{-k_f t}\right) \tag{5}$$

where k_f is the lumped mass transfer coefficient; q_m is the average concentration of adsorbed phase over the entire carbon surface at time t, and q_e, the equilibrium concentration.

In turn, the PSO model assumes that surface interactions mainly control the entire adsorption process, and that the adsorption rate is related to the square of the number of available adsorption sites (Loganathan et al. 2014, 1-8). The model equation is given by:

$$q_m = \frac{q_e{}^2 k_p t}{1 + q_e k_p t} \tag{6}$$

where k_p is the adsorption rate constant.

The characteristic parameters for the LDF and PSO models were estimated by non-linear regression analysis for a 5% significance level. Models characteristic parameters are listed in Table 6. Predictions of both models (Eqs 5 and 6) are represented in Figure 11. The results showed that the LDF model led to a better fit than PSO model, thus indicating that CO_2

adsorption onto the invasive wood-based activated carbons took mainly place by physisorption. As expected, the CO_2 adsorption capacity decreased notably at the higher adsorption temperature. This trend also pointed to a physisorption mechanism, since at higher temperature Van der Waals forces are weaker, reducing attraction forces between the adsorbent's surface and the molecules of the adsorbate.

Table 6. Estimated characteristic parameters for the Linear Driving Force (LDF) and the Pseudo-Second-Order (PSO) kinetic models

T (ºC)	**Model Parameters**						
	LDF				PSO		
	q_{exp} (mmol/g)	k_f (l/min)	q_e (mmol/g)	R^2	k_p (g/mmol min)	q_e (mmol/g)	R^2
30	0.36	0.58	0.36	0.99	0.015	0.53	0.99
50	0.14	1.72	0.13	0.99	0.637	0.14	0.96

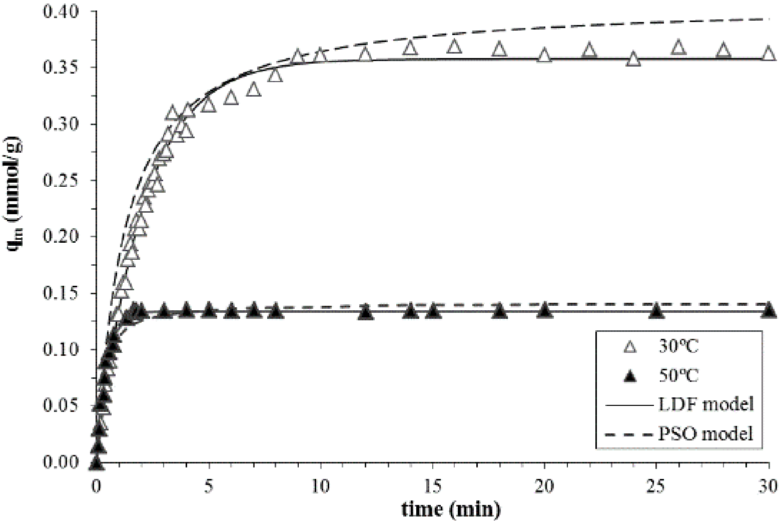

Figure 11. Kinetics of CO_2 adsorption at 30°C and 50°C for the ACs obtained from the invasive wood by K_2CO_3 activation. Comparison between the experimental results and predictions of the Linear Driving Force (LDF) model and the Pseudo-Second-Order (PSO) model.

Figure 12 displays the breakthrough curves determined in the column packed with the samples developed from the invasive wood with K_2CO_3 solution for a gaseous stream containing 20% (v/v) of CO_2 under two different gas flow rates. The breakthrough curves are expressed in terms of normalized concentrations, defined as the ratios of effluent to influent gas concentrations (C_{ef}/C_{in}), versus the operating time (t). They provide information on the efficiency of the ACs in the capture of CO_2, and, consequently, they are useful for the design and operation of fixed bed adsorption column towards the rational scale up of an adsorption process from a practical viewpoint (Gomez- Delgado et al. 2019).

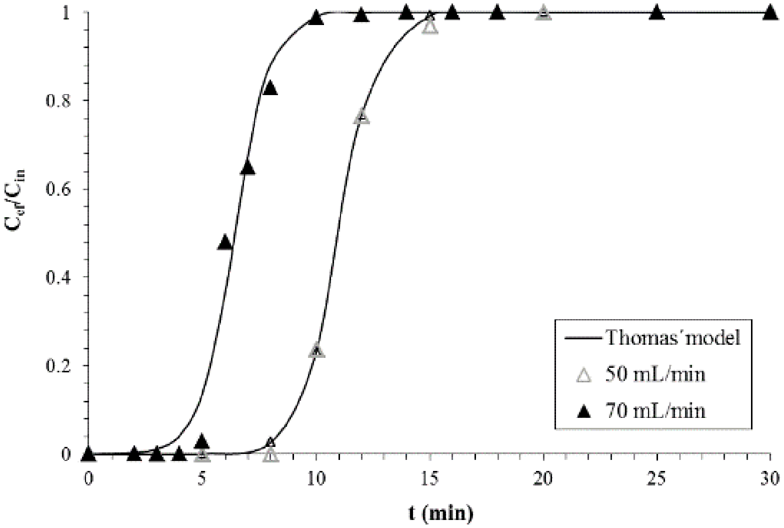

Figure 12. Breakthough curves for CO_2 adsorption onto the ACs developed from the invasive wood by K_2CO_3 activation at 30°C for two different gas flow rates. Comparison of experimental data with Thomas' model predictions.

In order to contribute to a better description of the breakthrough curves, performance parameters were calculated (Fernandez et al. 2015, 177-83; Silas et al. 2018, 150-65). The breakthrough time (t_r), namely the time at which C_{ef}/C_{in} increases 2%, allows to know the time of the adsorber after which the adsorbent must be replaced. Another important parameter is the saturation time (t_s), i.e., the time at which C_{ef}/C_{in} reaches 99%. The amount of CO_2 adsorbed per gram of adsorbent at the breakthrough time and saturation time (q_{tb} and q_{ts}, respectively) results from:

$$q_{tb} = \frac{V_r}{w} \tag{7}$$

$$q_{ts} = \frac{V_s}{w} \tag{8}$$

where V_r and V_s are the amounts of CO_2 removed up to the breakthrough and saturation times, respectively, and w, the mass of the activated carbon used in the fixed- bed. V_r and V_s are given by:

$$V_r = t_r q_{CO_2} \tag{9}$$

$$V_s = t_s q_{CO_2} \tag{10}$$

where t_r is the breakthrough time, t_s, the saturation time, and q_{CO2}, the inlet flow of CO_2.

To represent the experimental breakthrough curves, the Thomas' model was applied. This model assumes that the adsorption process is limited by gas-solid interfacial mass transfer and that adsorption at equilibrium follows Langmuir isotherm (Gomez-Delgado et al. 2019). It is described by the following equation:

$$\frac{C_{ef}}{C_{in}} = \left[1 + exp\left(\frac{K_{th}}{F}\left(Q_o w - C_{in} V_o\right)\right)\right]^{-1} \tag{11}$$

where K_{th} is the Thomas' rate constant (mL mmol/min), Q_o, the equilibrium adsorption capacity (mmol/g), w, the amount of adsorbent in the column (g); C_{in} and C_{ef} (mmol/mL) are the CO_2 concentrations in the influent and effluent at time *t* (min), respectively. *F* is the total flow rate (mL/min), and V_o, the effluent volume (mL). The values of K_{th} and Q_o were determined by non-linear regression. Predictions of the Thomas' model are presented in Figure 12. The estimated characteristic parameters of this model together with performance parameters are reported in Table 7.

Table 7. Performance and Thomas' model parameters estimated for dynamic adsorption of CO_2 (20% v/v) onto the activated carbons developed by K_2CO_3 activation for two flow rates

Parameters	**Flow rate (mL/min)**	
	50	70
t_b (min)	8.2	4.7
t_s (min)	18.4	10.7
q_{tb} (mmol CO_2 /g)	1.3	1.0
q_{ts} (mmol CO_2 /g)	1.6	1.2
K_{th} [mL /(min mmol)]	148.6	159.1
Q_o (mmol/g)	1.8	1.5
R^2	0.99	0.99

The breakthrough curves obtained showed typical S-shape. As expected, the breakthrough and saturation occurred earlier when increasing the flow rate, as clearly reflected in shorter times values in Table 7. Figure 12 also shows that Thomas' model described appropriately the breakthrough curves for CO_2 adsorption onto the ACC with high regression coefficients (Table 7).

CONCLUSION

Sawdust from invasive *Parkinsonia aculeata* wood was successfully converted into nanoporous activated carbons by chemical activation using comparatively solutions of KOH and K_2CO_3 as activating agents. Activation with both reagents led to activated carbons with low ash content, almost neutral values of pH and pHzc, and formation of acid and basic surface functional groups. Both samples exhibited similar quantities of carbonyls, phenols and lactones, but only the one obtained with K_2CO_3 presented a minor proportion of carboxyl groups. They also showed development of surface functionalities of basic character, with the sample prepared by K_2CO_3 activation possessing a slightly greater content of basic groups. Instead, the commercial activated carbon used for comparison showed a complete absence of basic functionalities, a noticeably greater proportion of carboxyls and smaller number of carbonyls than the developed activated carbons. Differences in the surface chemistry of the ACs and the commercial sample reflected the strong influence of the precursor and/or activation process employed for their development. pHzc values were in line with surface functionalities contents of the three samples.

The invasive wood-based activated carbons showed well-developed porous structures characterized by BET surface areas of ~770-780 m^2/g, total pore volume of ~0.35 cm^3/g, and predominance of nanopores (≥60%). These features differed from those determined for the commercial activated carbon. It presented higher BET surface area and total pore volume (1400 m^2/g; 1.10 cm^3/g, respectively) than the developed carbons, but a noticeably smaller contribution of nanopores (~23%). CO_2 adsorption isotherms determined at 0ºC for the invasive wood-based carbons indicated similar volumes of nanopores, as calculated by applying the DR equation. They also enabled to evaluate pore size distributions, that showed nanopores smaller than 1 nm for both samples, although those obtained by KOH activation presented all nanopores in the narrow range 0.4-0.8 nm. Likewise, maximum amounts of CO_2 adsorbed at 0ºC and 100 kPa were similar for the samples developed with both reagents, attaining values (~5 mmol/g) that were among the highest ones reported in the literature for other activated carbons, for the same conditions.

The activated carbons developed from *P. aculeata* wood by activation with both reagents were more effective in the adsorption of phenol, employed as representative model of priority water contaminants, than the commercial sample, in spite of their less porous development. The adsorption capacity of the developed carbons, as estimated from the experimental adsorption isotherms represented by the Freundlich's model, almost duplicated the value obtained for the commercial sample. The results thus indicated that textural characteristics should not exert a predominant effect. Instead, surface functional groups, likely functionalities of basic character which were absent on the commercial carbon surface, seemed to govern phenol adsorption. Accordingly, adsorption capacities of the developed ACs pointed to their possible use for the removal of phenolic compounds usually found in industrial wastewater. On the other hand, the samples developed by activation with K_2CO_3 solution also showed potentialities for adsorption of CO_2 from gaseous streams approaching post-combustion conditions. Overall, the nanoporous activated carbons derived from invasive *P. aculeata* wood demonstrated ability for their potential use in the removal of contaminants from water and air.

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Chapter 8

HALF OF MEXICO'S INVASIVE PLANT SPECIES MAY BE USEFUL FOR HEAVY METAL PHYTOREMEDIATION

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ABSTRACT

The exploitation and utilization of metallurgical resources promote the production of dangerous residues, known as mine tailings. These residues contain potentially toxic elements (PTE) that produce alterations and adverse effects in the environment affecting the surrounding biota. Heavy metals (HM) may induce different alterations in organisms (e.g., genetic, metabolic, physiological, and morphological). Plants are possibly the biological group most affected by the environmental presence of HM because they directly depend on the substrate for their limited mobility. However, several plants have different mechanisms that help them to tolerate, neutralize, or counteract the toxic effects of HM, suggesting that some plant species can establish themselves in contaminated sites without suffering modifications that affect their development. Within these species, the HM hyperaccumulator plants have been used in diverse phytoremediation processes with promising results. Worldwide about 450 HM hyperaccumulator plant species have been reported, many of which have an invasive species status. In particular, in Mexico, 215 invasive plant species were reported, of which approximately 51% were documented in heavy metals bioaccumulation studies. Although invasive species have been considered a significant cause of biodiversity loss, causing untoward effects on resident communities and ecosystem functioning, these plants have great potential as an alternative for phytoremediation of mine tailings containing complex mixtures of bioavailable HM. Phytoremediation offers owners and managers of metal-contaminated sites an innovative and cost-effective option to address environmental contaminants. The use of plants to restore or stabilize contaminated sites, collectively known as phytoremediation, takes advantage of the natural abilities of plants to take up, accumulate, store, or degrade organic and inorganic substances. This chapter presents an updated list of invasive plant species that are considered bioaccumulators and hyperaccumulators of HM. We found that 110 invasive species (51%) were documented to contain the bioaccumulation of HM, principally in leaves, shoots and stems. Of this total, 29% have been proposed as hyperaccumulators. Pontederiaceae, Asteraceae and Fabaceae were the plant families most representatives by the number of species that bioaccumulate HM. Also, we point out which of these species present the highest values for HM extraction, also, if some of these species show an affinity for specific metals. Additionally, we document invasive species that have the potential for their use as biosensors in HM contaminated sites.

Keywords: contamination, invasive species, heavy metals; phytoremediation

INTRODUCTION

The introduction of invasive species (IS) is considered one of the greatest threats to biodiversity in terms of loss of genes, species, and ecosystems. These species have a wide geographic distribution caused by their exceptional ability to become established outside their natural distribution areas (Simberloff 2013, 107). Also, these species are considered biological elements that promote changes in the resources and conditions in the areas where they are established. Invasive species can also alter ecosystem dynamics and subsequent modification in the structure of communities (Davis 2010, 106). In particular, they may cause an unrecoverable loss of native species. Disturbing events facilitate the introduction of IS into a biological system (Prieur-Richard and Lavorel 2000, 3). Specifically, the disturbances related to anthropogenic activities alter the natural dynamics of ecosystems, causing adverse effects over availability of the resources and conditions needed for native species (Davis 2010, 57–63; Simberfolff 2013, 25). By contrast, these disturbances promote advantageous conditions for invasion by exotic species.

The increase in the frequency, duration, and intensity of the environmental disturbances has exacerbated the adverse effects caused by IS by promoting novel habitats for their establishment and subsequent invasion of the native ecosystems for the local flora (Mack et al. 2000, 698). In low or moderate-intensity of anthropogenic disturbances, the resilience of forests allows their recovery. However, when the disturbances are severe new conditions are promoted that hinder succession processes. In disturbed sites, plant diversity is usually dominated by invasive species, which compete with native species for resources and conditions (Gioria and Osborne 2014, 3).

In particular, mining activities during ore deposits exploitation cause environmental disturbances, due to significant magnitude of excavation techniques, such as open-pit mining, bare exports, and quarries. During mining processes, vegetation cover is removed, and large amounts of rocks and soil with or without little economic value are abandoned outdoors at the end of the excavation. These wastes are denominated mining tailings (Alekseenko et al. 2018, 261). These material deposits can be widely spread over areas of several square kilometers and rise to 500 m in height (Montenegro et al. 2009, 427; Valles 2012, 22). In Mexico, the mining industry has driven economic development since its inception during the pre-Hispanic era (Gutiérrez and Moreno 1995, 38). Nowadays mining is still an important economic activity in the country (Gómez-Bernal et al. 2010, 131).

However, according to the Mexican Geological Service (SGM 2014, 19), around 30 million hectares are currently under concession for the mining sector, which corresponds to 15% of the national territory. Although there is currently not enough information to appreciate the cost of the environmental impact caused by mining throughout the country, it is recognized that it is significantly higher than that of other industries, mainly because it produces 65% of total industrial wastes that are generated in Mexico (Delfín-Alcalá and Durán-de-Bazúa 2008, 2).

The primary concern about mine tailings is the flow of potentially toxic elements (PTEs). These elements include heavy metals (HM) which, due to their oxidation, can cause acid drainage, facilitating their mobility and subsequent absorption by the surrounding biota (Talavera et al. 2005, 50; Mussali-Galante et al. 2013, 7666). HM contamination is one of the most severe problems with important implications for human and environmental health. These elements have high persistence in biota, low solubility and are classified as carcinogenic and mutagenic, which causes effects on all levels of biodiversity, resulting in impacts even at the evolutionary level (Nagajyoti et al. 2010, 199; Mussali-Galante et al. 2013, 7666).

Therefore, sanitation of HM contaminated sites is a priority. Techniques such as bioremediation have been established, in which a variety of organisms (e.g., plants, fungi, bacteria, among others) are used to remove (extract), degrade (biodegrade) or transform (biotransform) toxic organic compounds into less toxic or harmless metabolic products (Velasco and Volke 2003, 41). The phytoremediation is an example of bioremediation, in which plants and agronomic techniques are used together to eliminate, retain, or reduce the toxicity of soil contaminants (Carpena and Bernal 2007, 1). Two of the most successful techniques of phytoremediation are 1) phyto-extraction (or phytoaccumulation) — based on the ability of some plants to accumulate contaminants in their roots, stems or foliage —, and 2) phyto-stabilization — where plants immobilize pollutants through root accumulation. This last process reduces the mobility of the pollutant and prevents its migration to water or air and reduces the bioavailability of the pollutant (Carpena and Bernal 2007, 1).

Several studies have documented that there are plant species that have managed to establish themselves in soils rich in HM and can absorb and accumulate them (Lin and Aarts 2012, 3188). Some of these species can tolerate high concentrations of HM in the soil by restricting their absorption and/or translocation towards the leaves; this strategy is known as exclusion. On the other hand, other species actively absorb and accumulate HM in their aerial biomass, which requires highly specialized physiology (Diez-Lázaro et al. 2002, 314). In particular, plants that accumulate high concentrations of HM in the aerial organs, above the levels found in most species and without presenting significant phytotoxic effects, are called hyperaccumulators (Rascio and Narvio-Izzo 2011, 170). Worldwide about 450 HM hyperaccumulator plant species have been reported, many of which have an invasive species status.

In this study, we documented plant species that have invasive status in Mexico and described which ones have been studied for their ability to bioaccumulate HM in tissues. Likewise, we performed an analysis of the botanical families that contain the largest number of bioaccumulative species, the type of HM bioaccumulated, and the plant structure involved. Moreover, the proportion of *ex situ* or *in situ* studies, the HM most frequently analyzed or detected, as well as the bioaccumulation concentrations are also presented. This work provides a guide to select vegetal species for remediation purposes, because offers specific information on the HM that absorb and accumulate in plant tissues that grow in polluted areas.

INVASIVE SPECIES IN MEXICO

Mexico is positioned as the 4th country with the greatest mega-diversity worldwide, containing a wide variety of terrestrial and aquatic ecosystems (Espinosa et al. 2008, 34). However, these systems have been affected by an increase in natural and anthropogenic disturbances causing habitat alterations. These alterations have promoted the proliferation of species resistant to disturbances as well as the introduction and establishment of exotic organisms (Alexander et al. 2015, 478).

Currently, it has been officially documented that Mexico has 215 plant species with the status of "invasive species" (CONABIO 2014; DOF 2016; Estrada-Arellano et al. 2018). These species were distributed in 63 plant families. In general, the most important families by the number of species that included were: Poaceae (13.9%)> Lythraceae (13.5%)> Asparagaceae (9.8%)> Fabaceae (8.8%)> Brassicaceae (3.7%) = Asteraceae (3.7%). Table 1 presents the taxonomic list of plant species recognized as invasive for Mexico and the taxonomic family they belong to.

Table 1. Plant species officially recognized as invasive in Mexico

Group	**Family**	**Species**	**Group**	**Family**	**Species**
Sphenophyta				Malpighiaceae	*Hiptage benghalensis* (L.) Kurz
	Equisetaceae	*Equisetum arvense* L.		Malvaceae	*Abutilon theophrasti* Medik.
Pteridophyta				Melastomataceae	*Clidemia hirta* (L.) D. Don
	Salviniaceae	*Azolla pinnata* R. Br. subsp. *africana* (Desv.) R. M. K. Saunders & K. Fowler			*Miconia calvescens* DC.
		Salvinia molesta D. S. Mitch.		Myricaceae	*Myrica faya* Dryand.

Group	**Family**	**Species**	**Group**	**Family**	**Species**
Pinophyta				Myrtaceae	*Callistemon citrinus* (Curtis) Skeels
	Pinaceae	*Pinus cembroides* Zucc.			*Melaleuca quinquenervia* (Cav.) S. T. Blake
		Pinus pinaster Aiton			*Psidium cattleianum* Afzel. ex Sabine
Magnoliophyta				Nitrariaceae	*Peganum harmala* L.
Dicotyledons				Oleaceae	*Ligustrum lucidum* W. T. Aiton
	Acanthaceae	*Thunbergia alata* Bojer ex Sims			*Ligustrum robustum* (Roxb.) Blume
	Aizoacea	*Mesembryanthemum crystallinum* L.		Oxalidaceae	*Oxalis megalorrhiza* Jacq.
		Carpobrotus edulis (L.) N.E. Br.			*Oxalis perdicaria* (Molina) Bertero
	Amaranthaceae	*Atriplex semibaccata* R. Br.			*Oxalis pes-caprae* L.
		Bassia scoparia (L.) A.J. Scott		Papaveraceae	*Argemone mexicana* L.
		Salsola kali L. subsp. *tragus* (L.) Čelak.		Passifloraceae	*Passiflora mollissima* (Kunth) L. H. Bailey
		Salsola vermiculata L.		Plantaginaceae	*Digitalis purpurea* L.
		Searsia lancea (L. f.) F.A. Barkley			*Linaria vulgaris* Mill.
		Schinus terebinthifolia Raddi			*Veronica arvensis* L.
	Apiaceae	*Apium graveolens* L.		Polygonaceae	*Reynoutria japonica* Houtt.
		Centella asiatica (L.) Urb.			*Persicaria nepalensis* (Meisn.) H. Gross
		Conium maculatum L.			*Rumex crispus* L.
		Fallopia convolvulus (L.) Á. Löve		Primulaceae	*Ardisia elliptica* Thunb.
		Heracleum mantegazzianum Sommier & Levier		Rosaceae	*Rubus ellipticus* Sm.
		Foeniculum vulgare Mill.		Rubiaceae	*Cinchona pubescens* Vahl
	Apocynaceae	*Asclepias syriaca* L.		Rubiaceae	*Galium spurium* L.
		Catharanthus roseus (L. G. Don			*Richardia brasiliensis* Gomes
		Cryptostegia grandiflora R. Br.		Ruppiaceae	*Ruppia maritima* L.

Table 1. (Continued)

Group	Family	Species	Group	Family	Species
	Araliaceae	*Hedera helix* L.		Solanaceae	*Solanum marginatum* L. f.
	Asclepiadaceae	*Calotropis procera* (Aiton) W. T. Aiton		Tamaricaceae	*Tamarix aphylla* (L.) H. Karst.
	Asteraceae	*Cynara cardunculus* L.			*Tamarix ramosissima* Ledeb.
		Acanthospermum hispidum DC.		Urticaceae	*Cecropia peltata* L.
		Ambrosia camphorata (Greene) W. W. Payne		Verbenaceae	*Lantana camara* L.
		Ambrosia cordifolia (A. Gray) W. W. Payne	Monocotyledonous		
		Anaphalis margaritacea (L.) Benth. & Hook. f.		Arecaceae	*Cocos nucifera* L.
		Anthemis cotula L.		Araceae	*Pistia stratiotes* L.
		Baccharis sarothroides A. Gray			*Zantedeschia aethiopica* (L.) Spreng.
		Carthamus lanatus L.		Asparagaceae	*Sansevieria hyacinthoides* (L.) Druce
		Carthamus oxyacantha M. Bieb.		Asphodelaceae	*Asphodelus fistulosus* L.
		Centaurea calcitrapa L.		Cannaceae	*Canna indica* L.
		Centaurea melitensis L.		Commelinaceae	*Commelina benghalensis* L.
		Centaurea solstitialis L.			*Tradescantia fluminensis* Vell.
		Chromolaena odorata (L.) R. M. King & H. Rob.		Cyperaceae	*Cyperus alternifolius* L.
		Chrysanthemum coronarium L.			*Cyperus papyrus* L.
		Cirsium vulgare (Savi) TEN.			*Cyperus rotundus* L.
		Cotula australis (Sieber ex Spreng.) Hook. f.		Hydrocharitaceae	*Hydrilla verticillata* (L. f.) Royle
		Crupina vulgaris Pers. ex Cass.		Orchidaceae	*Oeceoclades maculata* (Lindl.) Lindl.
		Emilia fosbergii Nicolson		Poaceae	*Aegilops cylindrica* Host
		Emilia sonchifolia (L.) DC ex DC.			*Agrostis gigantea* Roth

Group	Family	Species	Group	Family	Species
		Guizotia abyssinica (L. f.) Cass.			*Agrostis stolonifera* L.
		Helminthotheca echioides (L.) Holub.			*Aira elegantissima* Scrur
		Hypochaeris radicata L.			*Alopecurus myosuroides* Huds.
		Mikania micrantha Kunth			*Andropogon gayanus* Kunth
		Senecio inaequidens DC.			*Anthoxanthum aristatum* Boiss.
		Senecio madagascariensis Poir.			*Anthoxanthum odoratum* L.
		Soliva sessilis Ruiz & Pav.			*Apera spica-venti* (L.) P. Beauv.
		Tripleurospermum inodorum (L.) Sch. Bip.			*Arthraxon hispidus* (Thunb.) Makino
		Xanthium strumarium L.			*Arundo donax* L.
	Balsaminaceae	*Impatiens walleriana* Hook. f.			*Bothriochloa pertusa* (L.) A. Camus
	Bignoniaceae	*Spathodea campanulata* P. Beauv.			*Bromus madritensis* L.
	Boraginaceae	*Cynoglossum amabile* Stapf & J. R. Drumm.			*Bromus rubens* L.
	Brassicaceae	*Brassica rapa* L.			*Bromus tectorum* L.
		Brassica tournefortii Couan			*Cenchrus ciliaris* L.
		Camelina sativa (L.) Crantz			*Chloris gayana* Kunth
		Cardamine hirsuta L.			*Chrysopogon aciculatus* (Retz.) Trin.
		Hirschfeldia incana (L.) Lagr.-Fossat			*Coix lacryma-jobi* L.
		Lepidium draba L.			*Cortaderia selloana* (Schult. & Schult. f.) Asch. & Graebn.
		Lepidium latifolium L.			*Cynodon dactylon* (L.) Pers.
		Raphanus raphanistrum L.			*Dactylis glomerata* L.
	Cactaceae	*Nopalea cochenillifera* (L.) Salm-Dyck			*Dactyloctenium aegyptium* (L.) Willd.
		Opuntia stricta (Haw.) Haw.			*Digitaria abyssinica* (Hochst. ex A. Rich.) Stapf

Table 1. (Continued)

Group	Family	Species	Group	Family	Species
	Caryophyllaceae	*Agrostemma githago* L.			*Digitaria sanguinalis* (L.) Scop.
		Spergularia rubra (L.) J. Presl & C. Presl			*Digitaria velutina* (Forssk.) P. Beauv.
		Vaccaria hispanica (Mill.) Rauschert			*Echinochloa colona* (L.) Link
	Casuarinaceae	*Casuarina equisetifolia* L.			*Echinochloa crus-galli* (L.) P. Beauv.
	Crassulaceae	*Bryophyllum pinnatum* (Lam.) Oken			*Echinochloa pyramidalis* (Lam.) Hitchc. & Chase
		Bryophyllum delagoense (Eckl. & Zeyh.) Schinz			*Eleusine indica* (L.) Gaertn. var. *brachystachya* Trin.
	Cucurbitaceae	*Cucumis dipsaceus* Ehrenb. ex Spach			*Elymus repens* (L.) Gould
		Cucumis melo L			*Eragrostis ciliaris* (L.) R. Br.
	Elaeagnaceae	*Elaeagnus angustifolia* L.			*Eragrostis curvula* (Schrad.) Nees
	Euphorbiaceae	*Euphorbia esula* L.			*Eragrostis lehmanniana* Nees
		Euphorbia terracina L.			*Holcus mollis* L.
		Mercurialis annua L.			*Hordeum murinum* L. subsp. *glaucum* (Steud.) Tzvelev
		Ricinus communis L.			*Hyparrhenia hirta* (L.) Stapf
	Fabaceae	*Acacia mearnsii* De Wild.			*Hyparrhenia rufa* (Nees) Stapf
		Alhagi maurorum Medik.			*Imperata cylindrica* (L.) Raeusch.
		Alhagi pseudalhagi (M. Bieb.) Desv. ex B. Keller & Shap.			*Melinis minutiflora* P. Beauv.
		Cajanus cajan (L.) Huth			*Melinis repens* (Willd.) Zizka
		Crotalaria retusa L.			*Miscanthus sinensis* Andersson
		Leucaena leucocephala (Lam.) de Wit			*Panicum maximum* Jacq.
		Mimosa casta L.			*Panicum muticum* Forssk.

Group	Family	Species	Group	Family	Species
		Mimosa pigra L.			*Panicum purpurascens* Raddi
		Mucuna pruriens (L.) DC. var. *utilis* (Wall. ex Wight) Baker ex Burck			*Panicum repens* L.
		Prosopis glandulosa Torr.			*Pennisetum clandestinum* Hochst. ex Chiov.
		Prosopis velutina Wooton			*Pennisetum purpureum* Schumach.
		Pueraria montana (Lour.) *Merr*. var. *lobata* (Willd.) Maesen & S. M. Almeida ex Sanjappa & Predeep			*Pennisetum setaceum* (Forssk.) Chiov.
		Pueraria phaseoloides (Roxb.) Benth.			*Poa annua* L.
		Securigera varia (L.) Lassen			*Poa pratensis* L.
		Senna siamea (Lam.) H. S. Irwin & Barneby			*Polypogon monspeliensis* (L.) Desf.
		Sesbania grandiflora (L.) Poir.			*Rottboellia cochinchinensis* (Lour.) Clayton
		Sesbania sesban (L.) Merr.			*Schismus arabicus* Nees
		Spartium junceum L.			*Schismus barbatus* (L.) Thell.
		Ulex europaeus L.			*Sorghum halepense* (L.) Pers.
	Geraniaceae	*Geranium dissectum* L.			*Stenotaphrum secundatum* (Walter) Kuntze
		Erodium cicutarium (L.) L'Hér. ex Aiton			*Zea diploperennis* Iltis, Doebley & R. Guzmán
		Erodium malacoides (L.) L'Hér.		Pontederiaceae	*Eichhornia crassipes* (Mart.) Solms
	Haloragaceae	*Myriophyllum spicatum* L.			*Pontederia vaginalis* Burm.f.
	Lamiaceae	*Galeopsis tetrahit* L.		Typhaceae	*Typha domingensis* Pers.
		Leonotis nepetifolia (L.) R. Br.		Zingiberaceae	*Hedychium coronarium* J. Koenig

Table 1. (Continued)

Group	Family	Species	Group	Family	Species
	Lythraceae	*Lythrum salicaria* L.			*Hedychium gardnerianum* Sheppard ex Ker Gawl.

Source: (CONABIO 2014; DOF 2016; Estrada-Arellano et al. 2018). The nomenclature of all plant species was reviewed in the International Plant Names Index (IPNI) database.

INVASIVE SPECIES CAPABLE OF BIOACCUMULATING HEAVY METALS

For literature review, we searched articles published between 1979 and 2019 which reported HM bioaccumulation in plant species previously recognized as invasive species in Mexico. Only one paper for species was included. Mexico has 215 invasive species and we estimated that 110 of them (51%) were documented to contain the bioaccumulation of HM in their tissues. The most representative plant families by the number of species that bioaccumulate HM were: Pontederiaceae (23)> Asteraceae (14)> Fabaceae (12)> Brassicaceae (6)> Poaceae (4) = Amaranthaceae (4) >Apiaceae (3). The remaining number of the species (44) were grouped into 36 plant families (Table 2).

In total, the bioaccumulation of 22 HM [Chromium (Cr), Mercury (Hg), Molybdenum (Mo), Lead (Pb), Strontium (Sr), Aluminum (Al), Calcium (Ca), Cadmium (Cd), Cobalt (Co), Copper (Cu), Iron (Fe), Potassium (K), Lithium (Li), Magnesium (Mg), Manganese (Mn), Sodium (Na), Nickel (Ni), Thallium (Tl), Vanadium (V), Zinc (Zn), Barium (Ba), and Tin (Sn)] and the metalloid Arsenic (As) were registered in invasive species. Likewise, the bioaccumulation of these HM were evaluated in nine different plants tissues, according to the number of studies carried out, and their order of importance was as follows: roots (48)> leaves (34)> shoots (32)> stems (16)> whole plant (10)> aerial part (9)> seeds (3)> branches (2)> rhizome (1).

In order of importance, the 10 HM and metalloids most evaluated in invasive species were: Pb> Zn> Cu> Cd> Ni> Cr> Fe> Mn> Co> As. In particular, the plant structures in which the most HM bioaccumulation studies were carried out are: leaves (21 HM) > roots (20 HM) > steam (19 HM) = shoots (19 HM) > rhizome (12 HM) > seeds (11 HM) > whole plants (9 HM) > aerial part (8 HM) > branches (7 HM). The more frequently documented HM in each plant structure were: leaves (Pb > Cu = Zn > Cd > Ni > Cr > Fe) > roots (Pb > Zn > Cu > Cd > Ni > Cr > Fe) > steam (Cd > Pb = Zn > Cu = Ni > Cr = Fe) = shoots (Pb > Zn > Cu > Cd > Cr = Ni > Fe) > rhizome (Pb = Fe > As = Cr = Hg = Al = Cd) > seeds (Pb > As = Cr = Hg = Al = Fe) > whole plants (Pb > Cu > Cd = Zn > Cr > Fe = Ni = Hg) > aerial part (Pb = Cu > Zn > Ni > Cd > Cr = Fe = Mn) > branches (Pb = Cr = Cu = Co = Fe = Ni = Zn = Mn).

We found 110 studies of invasive species that bioaccumulate heavy metals of which the 82% were carried out in the field and the remaining 18% in the laboratory through experimental design and under controlled conditions. In order of importance, the most evaluated sources of contamination were: mining 46.3% > standard solutions 13.4% > sewage water 12.2% > industry 9.8% > natural origin 7.3% > petroleum derivatives 6.1% > crops 3.7% > garbage dump 1.2%. In addition, of the total number of invasive species where bioaccumulation of heavy metals was documented, 29% have been proposed as hyperaccumulators, highlighting the following families by the number of species they group: Poaceae > Amaranthaceae = Chenopodiaceae = Fabaceae = Polygonaceae. Finally, inductively coupled plasma spectrometry and its experimental variations were the most commonly used techniques to evaluate the concentration of metals in plant structures.

Table 2. Invasive plant species (n = 110) that bioaccumulate heavy metals distributed in Mexico, including scientific and common names, type of metal and concentration bioaccumulate by vegetal structure, technique, study conditions and comments

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Aizoaceae					
Mesembryanthemum crystallinum L. (crystalline iceplant)	units: 100 (µM Ni2+) shoot: 468 (µg Ni2+/g DW) roots: 1,898 (µg Ni2+/g DW)	Inductively coupled plasma mass spectrometry	Laboratory: pollution source. Experimental design with Ni2+	M. crystallinum would be suitable for metal phytostabilization of moderately polluted sites.	Amari et al. 2014
Carpobrotus edulis (L.) N. E. Br. (hottentot-fig)	units: mg/g structure analyzed: stems and leaves Pb: 175 Cd: 28	Atomic absorption spectrophotometetry	Laboratory: experiment for the removal of toxic pollutants from the wastewaters	C. edulis was found to be efficient in removing Cd and Pb from aqueous solution as compared to other adsorbents already used for the removal of these ions.	Erdogan et al. 2011
Amaranthaceae					
Atriplex semibaccata R. Br. (creeping brunette)	units: mg/Kg total plant: Cu: 283, Fe: 767, Ni: 661, Cd: 277	Atomic absorption spectrophotometry	Field study: terrestrial Abandoned copper mining	A. semibaccata can tentatively be considered as a hyperaccumulator of Cu and Cd, but it needs further investigation for its potential in phytoremediation.	Baycu et al. 2015
Bassia scoparia (L.) A. J. Scott (burning bush)	units: mg/Kg leaves: As: >3000	Atomic absorption spectroscopy (GFAAS)	Field study: terrestrial Fertilizer plants that produced phosphate fertilizers with metals	B. scoparia had the highest values for As accumulation.	Gisbert et al. 2008

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Salsola kali L. subsp. tragus (L.) Čelak. (prickly Russian thistle)	units: mg/Kg roots: Cd: 2696 stem: Cd:2075 leaves: Cd: 2016	Inductively coupled plasma optical emission spectrometer	Laboratory: experimental design with Cd	Tumbleweed could be considered as potential Cd hyperaccumulating species.	de la Rosa et al. 2004
Salsola vermiculata L. (mediterranean saltwort)	units: mg/g leaves: Cu: 45-238.0, Ni: 2.4-144.0	Shimadzu spectrophotometer	Laboratory: experimental design with Ni (II) and Cu	The Freundlich and Langmuir models provide a better description of the adsorption of Cu (II) and Ni (II) respectively.	Benderdouche et al. 2003
Apiaceae					
Centella asiatica (L.) Urb. (philippine Aquibuse)	units: total plant: Cu: 36.74, Pb: 134.88 Zn: 57.86	Atomic absorption spectroscopy	Field study: terrestrial paper mill wastes	C. asiatica Assess the concentrations of Pb, Cu and Zn in some selected herbaceous species as well as in their dwelling soils contaminated with paper mill wastes.	Bahnika and Baruah 2014
Conium maculatum L. (poison parsley)	Units: mg/Kg shoot: Cu: 3.28-13.59, Zn: 42.98-92.64, Ni: 2.59-5.95, Cd: 0.34-0.81, Cr: 1.37-7.83, Pb: 4.17-18.82 roots: Cu: 23.15-64.02, Zn: 143.84-314.84, Ni: 9.43-24.07, Cd: 0.55-0.82, Cr: 16.93-97.08, Pb: 3.55-17.81	Inductively coupled plasma mass spectrometry	Field study: sewage water	Results showed that addition of elemental S is an appropriate technique to remove the heavy metals from sewage sludge	Dede and Ozdemir, 2016

Table 2. (Continued)

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Fallopia convolvulus (L.) Á. Löve (black bindweed)	units: (mg/Kg) shoots: Cu: 105 roots: Cu: 800	Atomic absorption spectrometry	Laboratory: experimental design	Excessive Cu was accumulated and adsorbed mainly by the fine roots.	Pedersen et al. 2000
Apocynaceae					
Asclepias syriaca L. (silkweed)	units: mg/Kg structure analyzed: Cd: 0.10-3.31, Pb: 0.20-21.10, Zn: 60-398	Inductively coupled plasma mass spectrometry	Field study: terrestrial mining, waste material	severe magnitude of heavy metal exposure to Tribal members who gather and consume traditional plants within the Grand Lake watershed	Garvin et al. 2017
Catharanthus roseus (L.) G. Don (madagascar periwinkle)	units: µg/g leaves: Cd: 40-150	Atomic absorption spectrometry	Laboratory: experimental garden	Toxic effects of Cd and lead is maximum during germination and the plant gradually becomes more resistant.	Pandey et al. 2007
Araceae					
Pistia stratiotes L. (water cabbage)	units: mg/Kg roots: Ca: 15.8-31.5, K: 36.7-39.1, Mg; 4.98-6.92, Na;12.3-13.4, Al; 2750-3400, Cd; 0.611-0.614, Co; 0.395-0.909, Cr; 17.6-20.9, Cu; 30.0-39.4, Fe; 5295-6126, Mn; 230-760, Ni; 6.1-8.0, Pb; 6.02-6.94, Zn; 40.6-184	Inductively coupled plasma optical emission spectrometry	Field study: aquatic wastewater and stormwater polluted by heavy metals	Hyperaccumulator for Cr, Cu, Fe, Mn, Ni, Pb, and Zn	Lu et al. 2011

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
	shoots: Ca; 33.9-40.1, K; 29.0-32.0, Mg; 6.9-3.8, Na; 0.8-11.3, Al; 623-796, Cd; 0.263, Co; 0.125, Cr; 2.20-3.12, Cu; 2.93-4.17, Fe; 243-354, Mn; 145-196, Ni; 0.78-0.95, Pb; 0.93, Zn; 38.3- 71				
Araliaceae					
Hedera helix L. (English ivy)	units: µg/g Pb: 1.25 ± 0.07, As: 1.30 ± 0.06, Cd: 0.41 ± 0.02, Hg:0.26 ± 0.02, Zn: 0.49 ± 0.08	Atomic absorption spectrometry	Field study: terrestrial medicinal plants	The presence of heavy metals was in very low quantities	Rehman et al. 2019
Arecaceae					
Cocos nucifera L. (coconut)	0.5 (mg/L): Pb2+: 0.309, Cu2+: 0.355, Cd2+: 0.046, As3+: 0.090 1.0 (mg/l): Pb2+: 0.819, Cu2+: 0.428, Cd2+: 0.046, As3+: 0.246 2.0 (mg/l): Pb2+: 1.786, Cu2+: 1.076, Cd2+: 0.444, As3+: 0.722	Atomic absorption spectrometry	Laboratory: experimental design	Coconut shell can be used as a low cost adsorbent for the removal of heavy metals (Pb2+, Cu2+, Cd2+ and As3+) in aqueous solution	Okafor et al. 2012

Table 2. (Continued)

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
	3.0 (mg/l): Pb2+: 2.730, Cu2+: 1.946, Cd2+: 2.571, As3+: 1.324 4.0 (mg/L): Pb2+: 3.757, Cu2+: 2.924, Cd2+: 3.578, As3+: 1.815 5.0 (mg/L): Pb2+: 4.720, Cu2+: 2.924, Cd2+: 4.566, As3+: 2.019				
Asclepiadaceae					
Calotropis procera (Aiton) W. T. Aiton (apple of sodom)	units: mg/Kg (mean ± s.e.) shoots: Co; 1.2 ± 0.1, Cr; 1.5 ± 0.9, Cu; 6.3 ± 2.3, Fe; 58 ± 36.7, Mn; 52 ± 36.7, Ni; 4.4 ± 1.4, Pb; 7.5 ± 5.6, Zn; 22.0 ± 17.2	Atomic emisson spectrophotometer	Field study: terrestrial industrialization and urbanization with oil, gas, mining, and metallurgical.	No comment	Alyazouri et al. 2010.
Asteraceae					
Acanthospermum hispidum DC. (bristly starbur)	units: ppm leaves: Zn: 4760 stem: Zn: 3338	Atomic absorption spectrometry (CZ AAS-I).	Field study: terrestrial metal deposit	Esta especie se ha considerado como pseudo-metalofita y/o metaloresistente	Tiagi and Aery 1982
Anaphalis margaritacea (L.) Benth. & Hook. f. (pearly everlasting)	units: ppm root: As: 95 shoot: As: 13	Inductively coupled argon plasma spectrometry	Field study: terrestrial copper mine tailing	Adequate plant biomass and shoot tissue exclusion of high concentrations of Al and Zn when grown in copper mine tailings.	Mol 2016

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Anthemis cotula L. (stinking chamomile)	units: mg/Kg aerial parts: Cu: 203	atomic absorption spectroscopy	Field study: terrestrial mining deposits	There is a decrease in the concentration of Cu in aerial biomass as the season progresses	González et al. 2008
Baccharis sarothroides A. Gray (broom baccharis)	units: mg/Kg roots: K: 89736, Al: 5420.9, Fe: 6069.3, Mg: 3939.8, Ca: 87548, Cu: 818.3, Pb:151.9, Mn: 344.5, Mo: 73.9, V: 39.5, Cr: 57.1, Zn: 40.1, As: 44.6, Ni: 96.8, Co: 26.7, shoot: K: 284004, Al: 1469.8, Fe: 3354.9, Mg: 3470.2, Ca: 377981, Cu: 1214.1, Pb: 107.3, Mn: 2112.6, Mo: 105.8, V: 84.1, Cr: 105.5, Zn: 55.2, As: 36.9, Ni: 30.9, Co: 10.9	ICP/OES	Field study: terrestrial copper mine tailings	Considering the translocation factor, enrichment coefficient, and the accumulation factor, desert broom could be a potential hyperaccumulator of Cu, Pb, Cr, Zn, As, and Ni.	Haque et al. 2008
Carthamus lanatus L. (woolly distaff thistle)	units: mg/Kg root: Cd: 1.182, Cr: 2.456, Cu: 2.456, Fe: 2159.48, Pb: 0.273, Zn: 101.956, P: 1405.06. shoot: Cd: 0.888, Cr: 0.940, Cu: 20.628, Fe: 892.077, Pb: 1.044, Zn: 72.956, P: 1405.06.	Inductively coupled plasma atomic emission spectroscopy (ICP-AES)	Field study: terrestrial mining activities	C. lanatus is good candidate for phytostabilization. The soil-plant transfer factor (TF) higher was for Cd: 8.40.	Nouri et al. 2013

Table 2. (Continued)

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Carthamus oxyacantha M. Bieb. (Kantiari)	units: mg/Kg root: Cu: 680.2, Fe: 34696.7, Pb: 43.6, Zn: 445.9 shoot: Cu: 551.7, Fe: 48116, Pb: 21.2, Zn:481.6	Flame atomic absorption method	Field study: terrestrial mining activities	C. oxyacantha is the best Fe and Pb accumulator in aerial parts.	Nematian and Kazemeini 2013
Chromolaena odorata (L.) R.M. King & H. Rob. (Siam weed)	units: µmol/g (mean at day 30 ± s.e.) roots: Al: 619.2 ± 21, Cd: 5.32 ± 1.5, Fe: 754.69 ± 2.8, Hg: 2.14 ± 0.05, Cr: 102 ± 1.39, Cu: 52.4 ± 0.85, Pb: 927.9 ± 23.6, Ni: 6.8 ± 0.12, Zn: 5.68 ± 0.07 leaves: Al: 157.2 ± 2.8, Cd: 4.48 ± 0.18, Fe: 160.88 ± 1.7, Hg: 0.36 ± 0.009, Cr: 10.4 ± 0.04, Cu :0.256 ± 0.01, Pb: 1.2 ± 0.03, Ni: 1.92 ± 0.04, Zn: 1.6 ± 0.04	Atomic absorption spectrophotometry	Laboratory. experimental metals were measured at day 1, 15 and 30.	C. odorata root system shows maximum amount of all metals in the roots in general and Al, Fe, Cu and Pb in particular. Notwithstanding, considerable quantities of Fe, Cu, Ni and Zn in the root tissues, and are found to be translocated to stem and leaf because they are essential nutrients for plants.	Swapna et al. 2014
Chrysanthemum coronarium L. (Glebionis coronaria)	units: mg/Kg (mean ± s.d.) shoot: Cu: 585 ± 43, Pb: 385 ± 41, Zn: 586 ± 78, Cd: 1.3 ± 0.2	Inductively coupled plasma atomic emission spectroscopy	Laboratory: experiments were conducted in a glasshouse.	C. coronarium was the most efficient at accumulating Cu and Pb in its shoots with the application of EDTA.	Luo et al. 2006

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Cirsium vulgare (Savi) TEN. (common thistle)	units: mg/Kg Na: 400, K: 3786, Ca: 7405, Mg: 3273, Fe: 459, Mn: 134, Cr: 131, Ni: 166, Co: 36, Cu: 66, Pb: 13, Zn: 21, Cd: 0.	atomic absorption spectrometry	Field: study: terrestrial mining and chromite deposits	The concentration of Cr was the lower in C. vulgare	Muhammad et al. 2013
Cynara cardunculus L. (artichoke)	units: mg/Kg shoot: Pb; EDTA: 1,332 ± 244.3, EDDS: 310.2 ± 100.7 root: Pb; EDTA: 6,695.2 ± 711.2 EDDS: 4,165.8 ± 607.9	Atomic absorption spectrometer	Field study: terrestrial experimental design	Hyperaccumulator plant species for phytoextraction of Pb	Epelde et al. 2008
Guizotia abyssinica (L. f.) Cass. (niger)	units: µg/g K: 5594–8203, Mg: 2404–4656, Ca: 340–680, Na: 159–736, Fe: 31.6–370, Zn: 23.4–46.2, Pb: 15.5–19.3, Mn: 13.4–34.3, Ni: 13.0–32.4, Cu: 9.5–61.2, Co: 4.9–27.3, Cr: 4.0–16.8	Atomic absorption spectrometry	Field study: terrestrial standard solution of heavy metals	Is a good source of essential metals and free from toxic metals and hence safe for daily human consumption	Syume and Chandravanshi 2015
Mikania micrantha Kunth (bitter vine)	units: mg/Kg total plant: Cu: 17.68, Pb: 157.52, Zn: 54.14	Atomic absorption spectroscopy	Field study: terrestrial paper mill wastes	The concentrations of Pb, Cu and Zn in some selected herbaceous species as well as in their dwelling soils contaminated with paper mill wastes.	Bahnika and Baruah 2014

Table 2. (Continued)

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Tripleurospermum inodorum (L.) Sch. Bip. (scentless mayweed)	units: mg/Kg roots: Al: 19.5-28.3, Ba: 0.4-0.7, Ca: 0.2 ± 0.02, Cr: 0.2-0.4, Cu: 0.5-0.6, Fe: 17.3-32.6, K: 0.7-0.9, Mg: 0.1; Mn: 1.5-3.1, Na: 0.1-0.2, Ni: 0.1-0.2, Pb: 0.1-0.2, Sr: 1.6 - 1.7, Zn: 1.4-2.9 stems: Al; 6.6 -1.3, Ba: 0.2-0.3, Ca: 0.1-0.2, Cr: 0.1-0.2, Cu: 0.3-0.7, Fe: 5.8-7.0, K: 0.6-0.7, Mg: 0.03-0.04, Mn: 1.4-3.8, Na: 0.03-0.1, Ni: 0.1-0.1, Pb: 0.04-0.1, Sr: 1.2-1.4, Zn: 1.4-2.7, leaves: Al; 6.8-16.2, Ca: 0.6-0.8, Fe: 9.2-60.4, K: 1.5-1.8, Mg: 0.2-0.3, Mn: 3.6-9.6, Na: 0.1-0, Pb: 0.1-0.03, Sr: 1.5-1.9, Zn: 1.7-2.9	Microwave plasma atomic emission spectrometry	Field study: terrestrial sewage sedimentation ponds	T. inodorum is able to indicate and accumulate various metals of soil and, therefore, have good potential in the early stages of phytoremediation.	Tőzsér et al. 2019.

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Xanthium strumarium L. (rough cocklebur)	units: mg/Kg shoot: Pb: 43.7, Cr: 21.1, Cd: 5.1, Zn: 31.6, Fe: 169.6, Ni: 10.7, As: 56.6 root: Pb: 25.0, Cr: 31.9, Cd: 6.1, Zn: 65.2, Fe: 160, Ni: 6.0	Atomic absorption spectrophotometer	Field study: aquatic sewage water of different industries such as ghee, fertilizers, chemicals and textiles	The results showed higher content of Cd in the roots	Irshad et al. 2015
Balsaminaceae					
Impatiens walleriana Hook. f. (buzy Lizzie)	units: mg/Kg shoot: Cd [20] 282 ± 34, [40] 342 ± 103, [80] 1,168 ± 140 roots: Cd [20] 128 ± 21, [40] 196 ± 67, [80] 462 ± 17	Flame atomic absorption spectrophotometer	Field study: terrestrial contaminated sites with Cd	I. walleriana was reported as hyperaccumulator plant for phytoextraction of Cd.	Wei et al. 2012
Brassicaceae					
Brassica rapa L. (turnip rape)	units: mg/Kg total plant: Cd: 0.055 ± 0.006, Cr: 4.28 ± 0.58, Pb: 4.85 ± 0.61, Ni: 3.83 ± 0.45	Atomic Absorption Spectrophotometry	Field study: terrestrial heavy metal natural presence in soil	The study was conducted to explore the heavy metal profile of wild plants to evaluate their nutritional status and safety.	Khan et al. 2016
Camelina sativa (L.) Crantz (gold-of-pleasure)	units: mg/Kg root: Cd: 500, Pb: 17000, Zn: 8000, Co:1900 shoot: Cd: 350, Pb: 240, Zn: 1800, Co:350	Coupled plasma mass spectroscopy	Laboratory experiment	The transgenic lines of this plant accumulated more heavy metals than wild types.	Park et al. 2014

Table 2. (Continued)

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Cardamine hirsuta L. (hairy bittercress)	units: mg/Kg leaves: Li; 819-4221	Flame emission spectroscopy	Laboratory: screened greenhouse	C. hirsuta accumulated high concentrations of Li, the size of the plant could never be used successfully in any agro-mining project which require substantial amounts of biomass.	Kavanagh et al. 2018
Hirschfeldia incana (L.) Lagr.-Fossat (shortpod mustard)	units: mg/Kg roots at the vegetative stage, Pb: 106. roots at the floral stage, Pb: 77. rosette leaves, Pb: 0.95. siliques, Pb: 0.65	Inductively coupled plasma-atomic emission spectrometry (ICP-AES)	Field study: terrestrial metalliferous mine spoils	H. incana accumulates Pb, but an important translocation of Pb can be observed from the roots to the leaves and even to the siliques	Auguy et al. 2013
Lepidium draba L. (whitetop)	units: mg/Kg root: Zn: 19.84 ± 10.92, Cd: 0.13 ± 0.07, Cu: 3.59 ± 0.86, Ni: 18.51 ± 41.56, Fe: 354.25 ± 235.34 stem: Zn: 22.71 ± 21.43, Cd: 0.20 ± 0.07, Cu: 2.74 ± 0.68, Ni: 10.36 ± 20.75, Fe: 271 ± 270.67 leaves: Zn: 36.14 ± 21.96, Cd: 0.20 ± 0.11, Cu: 4.62 ± 1.49, Ni: 64.62 ± 119.65, Fe: 1037.45 ± 1149.0	Atomic absorption spectrometry (Agilent AAS 55A and Agilent 240Z).	Field study: terrestrial dust from ferro-nickel smelter plant	Maximum heavy metal bioaccumulation was observed in leaves.	Micevska et al. 2019

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Lepidium latifolium L. (perennial pepperweed)	units: mg/Kg total plant: Cd: 0.046 ± 0.005, Cr: 5.39 ± 0.65, Pb: 5.84 ± 0.59, Ni: 4.24 ± 0.43	Atomic absorption spectrophotometer (Perkin-Elmer AAS-5000)	Field study: terrestrial heavy metal natural presence in soil	The study was conducted to explore the heavy metal profile of wild plants.	Khan et al. 2016
Cactaceae					
Nopalea cochenillifera (L.) Salm-Dyck (cochineal plant)	units: mg/Kg root: Cr(VI): 25,263.396 ± 1,722.672 shoots: Cr(VI): 705.714 ± 32.324	atomic absorption spectroscopy	Laboratory: experimental design	Plant hyper accumulates Cr (IV), mainly in roots	Adki et al. 2013
Cannaceae					
Canna indica L. (sierra leone arrowroot)	units: mg/Kg leaves: 12.16 ± 0.926 steam: 14.23 ± 3.125 rhizome: 327.77 ± 99.875 root: 2480.077 ± 374.995	Inductively coupled plasma atomic emission spectroscopy (ICP-OES)	Laboratory: six replicates	C. indica is a very good candidate for rhizofiltration of water contaminated with Pb.	Cule et al. 2016
Caryophyllaceae					
Agrostemma githago L. (corn-cockle)	units: mg/Kg aerial parts: Co: 1.1, Cr: 1.3, Cu: 2.5, Fe: 181, Mn: 35.8, Ni: 16.7, Pb: 1.9, Zn: 8.3	atomic absorption spectrophotometry	Field study: terrestrial abandoned mining	Analysis of metal accumulation in the plant tissues revealed that a number of plants are able to tolerate high concentrations of toxic trace elements	Freitas et al. 2004
Spergularia rubra (L.) J. Presl & C. Presl (red sandspurry)	units: µg/g leaves: Cr; 5.5, Cu; 109.0, Fe; 911.0, Mn; 225.0, Ni; 18.5, Pb; 14.0, Ti; 145.0, Zn: 502.0	Atomic absorption spectrophotometer	Field study: aquatic drainage of a mining and industrial field	S. rubra accumulated low concentrations of Cr, Ni and Pb.	Luque et al. 1999.

Table 2. (Continued)

Scientific and common name	Type of metal and concentration. Structure analyzed	Technique	Study conditions and source of pollution	Comments	Reference
Cactaceae					
Opuntia stricta (Haw.) Haw. (prickly pear)	units: mg/Kg Eastern zone stem: K (3805.0 ± 16.2), Ca (1159.7 ± 7.7), Mg (1044.3 ± 10.7), Zn (13.9 ± 2.3), Mn (8.4 ± 0.07), Cu (2.9 ± 0.4), Ni (2.4 ± 0.3) Southeastern zone stem: K (4640.9 ± 10.3) > Ca (1221.8 ± 7.7) > Mg (1103.6 ± 9.2). Zn (18.4 ± 0.6) > Mn (7.4 ± 0.2) > Cu (3.4 ± 0.4) > Ni (2.5 ± 0.5).	Flame atomic absorption spectrophotometry	Field study: terrestrial cultivated cactus plants	The concentrations of K, Ca and Zn are significantly higher in this specie	Berhe and Chetan 2016
Casuarinaceae					
Casuarina equisetifolia L. (coast sheoak)	units: ppm leaves (needles): Zn: 141-440, Pb: 23.73- 67.5, Cu: 16.3- 37.02, Ni: 11.7-21.07	Flame atomic absorption spectrophotometry	Field stud: air heavy metal air pollution.	C. equisetifolia needles can be used as biomonitors of heavy metal pollution in urban areas.	Aissa and Kéloufi 2012
Commelinaceae					

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Commelina benghalensis L. (Benghal dayflower)	units: mg/Kg total plant: dry season Zn: 4.48 mg/g ± 2.19, Cd: 1.35 ± 1.19 total plant: wet season Zn: 4.48 ± 2.19, Cu: 1.93 ± 0.90	Atomic absorption spectroscopy	Field: Aquatic-land the mobil oil terminal operation base, Southeast Niger Delta, Nigeria	Zn was found to have the highest concentration in stem while Cu showed a remarkable concentration in leaves.	Umoh et al. 2014
Crassulaceae					
Bryophyllum pinnatum (Lam.) Oken (goethe plant)	units: mg/Kg leaves: Pb: 399.90 ± 4.32, Zn: 150.51 ± 0.33, Cu: 81.01 ± 2.3, Cr: 32.48 ± 3.21, Ni: 11.91 ± 2.32, V: 5.81 ± 0.08, Cd: 3.12 ± 1.03	Atomic absorption spectrophometry	Laboratory: greenhouse experiment	The amount of each heavy metal absorbed by B. pinnatum increased as the period of exposure to the metals increased.	Ekwumemgbo et al. 2013
Cupressaceae					
Cupressus sempervirens L. (mediterranean cypress)	units: mg/Kg root: Fe: 1,074.92 ± 770.8, Zn: 28.19 ± 2.8, Mn: 21.57 ± 2.2, Cu: 42.30 ± 2.6, Cr: 30.93 ± 1.7, Pb:	ICP-instrument (ICP-JY Ultima)	Field study: terrestrial irrigated soil with treated water	C. sempervirens has been suggested for their potential for phytoremediation.	El-Hamid et al. 2017

Table 2. (Continued)

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
	0.67 ± 0.06, Ni: 16.64 ± 1.4, Co: 5.44 ± 0.8 branches: Fe: 906.35 ± 160.3, Zn: 23.77 ± 1.6, Mn: 39.02 ± 6.6, Cu: 36.51 ± 1.0, Cr: 15.33 ± 1.5, Pb: 0.36 ± 0.05, Ni: 11.20 ± 1.1, Co: 3.24 ± 0.8 leaves: Fe: 1,085.19 ± 136.6, Zn: 34.88 ± 2.4, Mn: 42.86 ± 4.4, Cu: 20.82 ± 1.4, Cr: 11.00 ± 2.0, Pb: 0.53 ± 0.04, Ni: 7.65 ± 1.0, Co: 3.19 ± 0.6				
Equisetaceae					
Equisetum arvense L. (horse tail)	units: µg/mL total plant (Aboveground): Cu: 3-74, Zn: 30-1017, Cd: 0.1-2.4, Pb: 1.9-3.4. (Underground) Cu: 9-348, Zn: 41-1618, Cd: 0.3-7.3, Pb: 2.0-7.3	Not showed	Field study: aquatic-soil mining area	Suggested as a suitable biological monitor for certain heavy metal pollution	Ray and White 1979

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Euphorbiaceae					
Mercurialis annua L. (annual mercury)	units: mg/Kg Pb: 37, Zn: 205, Cu: 24, Cd: 1.2, As: 6	Flame atomic absorption spectrophotometry	Field study: aquatic and soil pollution. pyritic sludge	In this work was collected and studied plant species on contaminated soils with potential to be used in the remediation	Del Río et al. 2002
Ricinus communis L. (castor bean)	units: mg/Kg root: Pb: 400-25000	Inductively coupled plasma optical emission spectrometry	Laboratory: soil different concentrations of lead	R. communis is a hyperaccumulator species for Pb and presents tolerance properties in lead light concentration.	Romeiro et al. 2006.
Fabaceae					
Alhagi maurorum Medik. (camelthorn)	units: ppm total plant: Cd: 0.596 ± 0.009, Pb: 0.787 ± 0.007	UV-vis spectrophotometry, Fourier transform infrared spectroscopy FT-IR spectra, X ray diffraction (XRD)	Laboratory: soil experimental tests	Bismuth film-modified Au-GN-SeCys presented an advantageous and high-performance platform for the sensing of Cd (II) and Pb (II) in groundwater, soil and Alhagi maurorum plants.	Al-Hossainy et al. 2015
Cajanus cajan (L.) Huth (tropical green pea)	units: µg/g leaves: Cd: 44.62-76.89 stem: 282.56-11 281.76 root: Cd: 129.59-142.35 nodule: Cd: 23.37-19.26	Atomic absorption spectrophotometry	Laboratory: greenhouse experiment	Maximum concentration of Cd2+ by plant organ was: roots > stem > leaves > seeds > pod walls.	Meena Deswal and Laura 2014
Crotalaria retusa L. (devil bean)	units: mg/Kg root: Pb: 8.5, Zn: 170.15, Cd: 5.0 shoot: Pb: 34.5, Zn: 360.24, Cd: 8.2	Method of AOAC,2005	Field study: terrestrial mining (Zinc, Lead and Cadmium)	C. retusa can be used to stabilize zinc. Also, showed phytoextraction capability.	Ogbonna et al. 2015

Table 2. (Continued)

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Leucaena leucocephala (Lam.) de Wit (leadtree)	units: mg/Kg structure analyzed: Cu: 50.0 ± 13.5, Co: 26.7 ± 4.4, Ni: 7.9 ± 2.2, Pb: 1.56 ± 0.18	Atomic absorption spectrophotometer	Field study: terrestrial mine tailing	L. leucocephala has a higher potential for phytostabilisation of pyrite and copper	Ssenku et al. 2014
Mucuna pruriens (L.) DC. var. utilis (Wall. ex Wight) Baker ex Burck (bengal velvet bean)	units: ppm seeds: (white): As: 0.68, Hg: 0.14, Cd: ND, Lead: 0.58; (black): As: 0.54, Hg: 0.23, Cd: ND, Lead: 1.08	Inductively coupled plasma optical emission spectrometry (ICP-OES)	Field study: terrestrial	Evaluation of heavy metals in the seeds of both varieties of M. pruriens was carried with control quality purposes in the extraction of Levodopa (3, 4 dihydroxyphenylalanine)	Kaur et al. 2017
Prosopis glandulosa Torr. (honey mesquite)	units: µg/g DW leaves: Cd: 0.4-22, Pb: 0.8-3.3, Cu: 0.3-10, Ni: 40-60, Zn: 45-45, Cr: 15-48, Co: 30–45, As: 600–1700, Fe:400-2800	Atomic absorption spectrophotometer	Laboratory: terrestrial sewage sludge	P. glandulosa has potential for the phytoremediation of sewage sludges and sewage-amended soils (Cd, Zn, Ni, Co and As)	Abbas et al. 2019
Pueraria montana (Lour.) Merr. var. lobata (Willd.) Maesen & S.M. Almeida ex Sanjappa & Predeep (kudzu)	units: mg/g roots: Pb; 4.45-9.70 shoots: Pb; 0.40-0.48	Atomic absorption spectrophotometer	Laboratory: terrestrial Lead (PbNO3)	P. montana can be considered a hyper-accumulator of lead.	Schwarzauer-Rockett et al. 2013

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Pueraria phaseoloides (Roxb.) Benth. (tropical kudzu) [creeper]	units: mg/Kg seeds: Cr; 3500, Pb; 4000	Not specified	Laboratory: terrestrial heavy metal contamination	Heavy metal accumulation of P. phaseoloides increased with increasing of heavy metal concentration in the soil.	Rathnaweera and De Silva 2012
Securigera varia (L.) Lassen (purple crown vetch)	units: mg/Kg seeds (aboveground) Zn: 56.8, Pb: 67.2, Ni: 37.5, Cu: 33.2, (belowground) Zn: 17.7, Pb: 17.1 Ni: 14.7, Cu: 14.0	Spectrophotometer	Laboratory: aquatic different concentrations of lead	The seedlings of the species survive relatively high concentrations of Cd.	Bae et al. 2016
Senna siamea (Lam.) H.S. Irwin & Barneby (kassod tree)	units: mg/Kg structure analyzed: Cu: 70.0 ± 13.5, Co: 22.9 ± 4.4, Ni: 10.7 ± 3.0, Pb: 1.76 ± 0.18	Atomic absorption spectrophotometer	Field study: terrestrial mine tailing	Have a higher potential for phytostabilisation of pyrite and copper	Ssenku et al. 2014
Sesbania grandiflora (L.) Poir. (hummingbird)	units: mg/Kg roots: Pb: 118 shoots: Pb: 3	Not specified	Laboratory: soil Different concentrations of Mg and lead	*S. grandiflora* has potential for phytoextracting Pb by rhizofiltration	Malar et al. 2014
Spartium junceum L. (Spanish broom)	units: ppm fibers: Cu; 78.0-455.15, Pb; 78.0-455.15	Inductively coupled plasma-atomic emission spectrometry	Field study: terrestrial aqueous solutions of CuSO4.5H2O and Pb(NO3)2	These fibers can be used as a bioadsorbent for adsorption of heavy metal ions such as Cu (II) and Pb (II) from aqueous solutions.	Cerchiara et al. 2016

Table 2. (Continued)

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Geraniacea					
Erodium malacoides (L.) L'Hér. (mediterranean stork's bill)	units: mg/Kg structure analyzed: Aboveground Cu: 12.8, Fe: 763, Ni: 1.80, Cd: 70.0, Pb: 0.045	Atomic absorption spectrophotometer	Field study: terrestrial copper mining area	According to the results this plant has accumulation potential of Fe	Baycu et al. 2015
Lythraceae					
Lythrum salicaria L. (Purple loosestrife)	units: (mg/Kg) root: Ni (II) 3,737.8 leaves: Ni (II) 418.4	Atomic absorption spectrometer	Laboratory study experimental design	The roots are the most effective biomaterial for biosorption and phytoremediation studies for the removal of Ni (II).	Bingöl et al. 2017
Malvaceae					
**Abutilon theophrasti* Medik. (velvetleaf)	units: mg/Kg roots: Pb: 38.7, Zn: 56.1 Cu: 32.5 Cd: 2.1 shoot: Pb: 61.4, Zn: 158.2, Cu: 37.2Cd: 5.4	inductively-coupled plasma-atomic emission spectrometry (ICP-AES).	Field study: terrestrial metal smelter	Exhibited strong accumulative ability to four heavy metals. believed to be potential hyperaccumulators.	Cui et al. 2007

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Melastomataceae					
Clidemia hirta (L.) D. Don (koster's curse)	units: µg/Kg total plant: As: 44.3 ± 47.0, Cd: 67 ± 32, Co: 75.0 ± 30.0, Cr: 487.6 ± 62.0, Cu: 1.54 ± 1.97, Fe: 36.8 ± 32.1, Mn: 1.9 ± 1.6, Ni: 643.3 ± 780, Pb: 141.5 ± 170.0, Zn: 1.64 ± 1.27	Inductively coupled plasma optical emission spectrometry (ICP-OES)	Field study: Aquatic and soil copper mine	This plant was found along the Mamut riverbank that received effluent and runoff from a copper mine upstream. C. hirta is used for their medicinal properties.	Ali et al. 2006
Myrtaceae					
Callistemon citrinus (Curtis) Skeels (lemon bottlebrush)	units: mg/Kg total plant: Pb; 225.925, Ni; 192.60	Atomic absorption spectrophotometry.	Field study: terrestrial urban wastes	This species can absorb and tolerate Pb and Ni and their absorption increase through exposure time.	Almeida and Lima 2014
Eucalyptus camaldulensis Dehnh. (red eucalyptus)	units: mg/Kg roots: Fe: 348.48 ± 22.0, Zn: 18.91 ± 2.8, Mn: 160.77 ± 6.6, Cu: 33.71 ± 3.8, Cr: 9.23 ± 0.1, Pb: 2.80 ± 0.4, Ni: 7.08 ± 0.3, Co: 3.54 ± 0.6 branches: Fe: 1,614.2 ± 278.2, Zn: 18.6 ± 3.0, Mn: 47.2 ± 1.8, Cu: 26.8 ± 1.8), Cr: 16.4 ± 2.3, Pb: 0.12 ± 0.04, Ni: 10.7 ± 1.5, Co: 5.7 ± 1.1	ICP-instrument (ICP-JY Ultima)	Field study: terrestrial irrigated soil with treated water	This tree has been suggested for their potential for phytoremediation because tended to accumulate in roots, branches and leaves.	El-Hamid et al. 2017

Table 2. (Continued)

Scientific and common name	Type of metal and concentration. Structure analyzed	Technique	Study conditions and source of pollution	Comments	Reference
	leaves: Fe: 990.3 ± 104.3, Zn: 32.2 ± 4.1, Mn: 316.615.3, Cu: 14.8 ± 1.0, Cr: 14.5 ± 3.0, Pb: 0.05 ± 0.01, Ni: 9.6 ± 1.1, Co: 5.0 ± 0.7				
Oleaceae					
Ligustrum lucidum W. T. Aiton (chinese privet)	units: PM10 leaves (low air polluted): Pb; 0.25 ± 0.02, Cd; 0.06 ± 0.004, Cr; 1.5 ± 0.07, Cu; 5.41 ± 0.49, Zn; 36.48 ± 1.62, Mn; 33.69 ± 1.39, Fe; 65.72 ± 5.64. leaves: (high air polluted) Pb; 0.23 ± 0.03, Cd; 0.05 ± 0.01, Cr; 1.41 ± 0.07, Cu; 5.17 ± 0.86, Zn; 26.44 ± 6.85, Mn; 48.51 ± 4.7, Fe; 101.46 ± 6.72	Atomic absorption spectrophotometer (AAS)	Field study: terrestrial heavy metal in air (PM10)	Mn and Fe concentrations were higher in leaves from the highly PM10 pollution site.	García-Sánchez et al. 2019
Oxalidaceae					
Oxalis pes-caprae L. (sourgrass)	units: mg/Kg Cu: 68.1, Fe: 697, Ni: 4.69, Cd: 27.9, Pb: 0.010	Atomic absorption spectrophotometer	Field study: terrestrial copper mining area	This plant accumulated high concentrations of heavy metals	Baycu et al. 2015

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Papaveraceae					
**Argemone mexicana* L. (mexican prickly poppy) [grass]	units: mg/Kg leaves: Fe; 77.78 ± 1.581, Zn; 48.36 ± 0.152, Cu; 38.78 ± 0.62, Mn; 50.7 ± 0.1, Ni; 1.42 ± 0.00, Pb; 4.16 ± 0.00 shoot: Fe; 6.42 ± 0.13, Zn; 38.78 ± 0.015, Cu; 22.20 ± 0.1, Mn; 16.15 ± 0.001, Ni; 2.09 ± 0.001, Pb; 4.76 ± 0.045 roots: Fe; 18.34 ± 0.08, Zn; 1.77 ± 0.046, Cu; 2.42 ± 0.005, Mn; 2.47 ± 0.018, Ni; 1.01, Pb; 0.50	Gas chromatography-mass spectrometry (GC–MS)	Field study: terrestrial Pollution origin: disposed distillery sludge	study given a strong evidence for application of these weeds and grasses as tools for in-situ phytoremediation and eco-restoration of polluted sites	Chandra et al. 2018
Plantaginaceae					
Veronica arvensis L. (wall speedwell)	units: µg/g leaves: Cu; 9.41, Pb; 15.31, Zn; 96.55	Inductively coupled plasma-atomic emission spectrometry	Field study: terrestrial industrial	No comment	Rudin et al. 2017.
Poaceae					
Cenchrus ciliaris L. (African foxtail grass)	units: µg/g roots: Cr: 11, Ni: 22.8, Cu: 10, Zn: 17, Cd: 0.5 shoot: Cr: 6.8, Ni: 8.5, Cu: 6, Zn: 15, Cd: 0.5	Inductively coupled plasma (ICP)	Field study: terrestrial Industrial waste water	This species is suitable for phytoextraction but is not hyperaccumulator. This plant may be more suitable for soil stabilization.	Ghoneim et al. 2014

Table 2. (Continued)

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Chrysopogon aciculatus (Retz.) Trin. (golden false beardgrass)	units: mg/Kg Shoot: Ni: 19.80 ± 4.05 Root: Ni: 23.91 ± 4.05	Atomic absorption spectrophotometry	Laboratory: experiment was conducted in the screen house	C. aciculatus can be used as a good candidate for remediating Ni contaminated soils.	Oyedeji et al. 2017
Coix lacryma-jobi L. (Job´s tears)	units: mg/Kg roots: As; 29.17 shoot: As; 8.09	Atomic absorption spectroscopy	Field study: terrestrial cultured in soils spiked by arsenic	Is helpful to eco-remediate lightly or moderately As contaminated soils	Guo et al. 2009
Cynodon dactylon (L.) Pers. (Dhoob)	units: mg/Kg 7 days: Pb: 3,300, Co: 200. 14 days: Pb: 1,300, Co: 100	Atomic Absorption Spectrometer	Laboratory conditions adding heavy metals Pb, Co and Ni	C. dactylon accumulated lead and cobalt.	Rathi et al. 2011
Boraginaceae					
Dactylis glomerata L. (orchard grass)	units: g/Kg roots: Pb; 71.34 ± 22.49, Ni; 55.58 ± 93.91, Cr; 36.51 ± 4.22, Cd; 6.96 ± 5.43; Zn; 122.89 ± 48.33; Cu; 116.67 ± 39.95. leaves Cd; 1.40 ± 2.37, Cr; 21.27 ± 8.32, Pb; 26.48 ± 0.6, Ni; 22.68 ± 4.55, Zn; 75.45 ± 26.99, Cu; 23.11 ± 7.10	Inductively coupled plasma and atomic emission spectrometry	Field study: terrestrial sewage sludge	The sludge was mixed with the soil in proportions of 100, 200, 300, 400, 600 y 800 g/kg	Ortiz and Alcaniz 2006

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Dactyloctenium aegyptium (L.) Willd. (egyptian zacate)	units: mg/Kg shoot: Pb: 23.0, Cr: 15.8, Cd: 8.2, Zn: 92.1, Fe: 108.7, Ni: 8.2, As: 103.9 Root: Pb: 39.3, Cr: 42.6, Cd: 8.9, Zn: 135.9, Fe: 100.0, Ni: 18.5	Atomic absorption spectrophotometer	Field study: Aquatic sewage water of different industries such as ghee, fertilizers, chemicals and textiles	Higher accumulation of Pb was found in the roots	Irshad et al. 2015
Digitaria sanguinalis (L.) Scop. (chicken leg)	units: mg/Kg leaf and stem Cd: 0.049 ± 0.007, Cr: 3.75 ± 0.68, Pb: 5.28 ± 0.69, Ni 4.23 ± 0.45	Atomic absorption spectrophotometer (Perkin-Elmer AAS-5000)	Field study: terrestrial heavy metal natural presence in soil	The study was conducted to explore the heavy metal profile of wild plants.	Khan et al. 2016
Digitaria velutina (Forssk.) P. Beauv. (Annual couchgrass)	units: mg/Kg roots; Fe: 1,275, Cu: 14, Zn: 1, Cd: 9, Cr: 6 stems; Fe: 148, Cu: 11, Zn: 1, Cd: 5, Cr: 5 leaves; Fe: 808, Cu: 11, Zn: 2, Cd: 7, Cr: 5	Atomic absorption spectrometer	Field study: aquatic swamp	Lead was not detected	Bizuru et al. 2015
Pontederiaceae					
Eichhornia crassipes (Mart.) Solms (water lily)	units: mg/Kg total plant Cr: 70	Inductively coupled plasma mass spectrometry (ICP-MS)	Field study: aquatic-soil experimental design	The study shows abnormally high values of Cr concentrations in water samples, soil samples and plants of E. crassipes.	Mangabeira et al. 2004
Elymus repens (L.) Gould (couch grass)	units: mg/Kg Zn; 51.2 ± 24.7	Plasma emission spectrophotometry	Field study: terrestrial mine	The results revealed a tolerance to Zn in this specie	Pastor and Hernández 2009

Table 2. (Continued)

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Hyparrhenia hirta (L.) Stapf (thtching grass)	units: mg/Kg Fresh flowers: Cu: 9.77 ± 0.61; Zn: 0.70 ± 0.20; Fe: 2.11 ± 0.20; Mn: 1.15 ± 0.08; Pb: 3.15 ± 0.10 Dry flowers: Cu: 5.80 ± 1.00; Zn: 0.71 ± 0.20; Fe: 4.12 ± 0.41; Mn: 1.11 ± 0.05; Pb: 1.50 ± 0.08 Fresh leaves: Cu: 8.94 ± 1.41; Zn: 0.57 ± 0.06; Fe: 2.46 ± 0.30; Mn: 1.11 ± 0.00; Pb: 2.41 ± 0.30 Dry leaves: Cu: 4.30 ± 1.26; Zn: 0.77 ± 0.31; Fe: 3.05 ± 0.20; Mn: 1.19 ± 0.08; Pb: 1.83 ± 0.03	Flame atomic absorption spectrophotometer	Field study: terrestrial standard solutions of Pb, Mn, Fe, Zn and Cu	Concentration of heavy metals follows the next order: Cu > Fe > Pb > Mn > Zn in the flowers of H. hirta	Tembeni et al. 2016
Hyparrhenia rufa (Nees) Stapf (jaragua grass)	units: mg/Kg leaves: Fe: 5,800	Atomic absorption spectrophotometry	Laboratory: greenhouse experimental design	Accumulation of Fe in leaves of H. rufa showed a severe oxidative damage as well as limitations in photosynthetic capacity	Oliveira Rios et al. 2017

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Imperata cylindrica (L.) Raeusch. (cogongrass)	units: mg/Kg roots: Pb: 78.0, Zn: 5.7, Cd: 2.55 shoot: Pb: 39.0, Zn: 179.76, Cd: 1.9	Method of AOAC, 2005	Field study: terrestrial Mining (Zinc, Lead and Cadmium)	I. cylindrica has the highest translocation factor of Zn. However, not could be considered a hyperaccumulator of any of the metals	Ogbonna et al. 2015
Melinis repens (Willd.) Zizka (Natal grass)	units: µg/g Sediments roots: Cd 0.105, Cu 95, Pb 2.3, Ni 4.04, Zn 135. stem: Cd 0.037, Cu 4.5, Pb 2.65 Ni 1.05, Zn 726.5 Highway roundabout sediments roots: Cd; 0.2, Cu; 54, Pb; 6.7, Ni; 2.1, Zn; 254 stem: Cd; 0.16, Cu; 5.2, Pb; 0.45, Ni; 1.24,	ICP-MS	Field study: terrestrial roadside soils and sediments	Plants growing on roadside sediments accumulated Zn mainly in stem.	Pratt and Lottermoser 2007
Panicum repens L. (torpedograss)	units: mg/Kg aerial biomass: Cu: 12-28, Mn: 60-110, Zn: 30-50, Pb 50-110	Atomic absorption spectrophotometer	Field study: terrestrial Different concentrations of lead.	Pollutants on the road (automobile derivatives), industrialization and urban growth.	Shaltout et al. 2013.

Table 2. (Continued)

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Stenotaphrum secundatum (Walter) Kuntze (buffalo grass)	units: µg/g Aerial tissues: Pb; 22.0 ± 2.6 Underground tissues: Pb; 14.0 ± 1.5	Atomic absorption spectrophotometer	Field study: terrestrial Lead solutions: Pb(N03)2 90 mM.	The highest concentration of lead was located in the aerial tissues	Barros et al. 2005.
Agrostis gigantea Roth (black bent)	units: mmol m-3 roots: Co: 8–40, Cu: 127->170, Ni: 78–170, Zn: 612 -520	Stock solutions and using clones	Field study: terrestrial mine waste	The toxic effects of heavy metals depend on their concentration	Hogan and Rauser 1979
Agrostis stolonifera L. (creeping bentgrass)	units: mg/Kg leaves: Pb: ≈1, Zn: ≈8, Mg: ≈5.5, Mn: ≈ 130, Cr: ≈1.1	Atomic absorption spectrometry, and X-ray fluorescence spectrometry	Field study: terrestrial magnesite dust contamination	The values that measured significantly above the set limit show contamination that can be considered harmful and toxic.	Fazekaš et al. 2018
Alopecurus myosuroides Huds. (Slender meadow foxtail)	units: ppm total plant: S: 3387 ± 1195, Fe: 229.9 ± 83.1, Zn: 498.7 ± 85.4, Cu: 6.1 ± 0.6	Atomic absorption spectrophotometer	Field study: terrestrial main traffic roads	Highest sulphur content was found in A. myosuroides and can be used in elemental air pollution monitoring studies in urban industrial conurbations.	Huseyinova et al. 2009
Andropogon gayanus Kunth (creeping bentgrass)	units: mg/Kg total plant: Pb: 2-8.1, Cd: 0.25-3.5, Cu: 1.6-9.1, Zn: 4-31.7	Flame atomic absorption spectrophotometry	Field study: terrestrial: crude-oil spills	The concentrations of Pb, Cu, Cd and Zn in the topsoil exceeded international standards.	Fatoba et al. 2015

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Anthoxanthum odoratum L. (sweet vernal grass)	units: mg/Kg root and stem Pb ≈ 25, Cu: ≈ 60, Zn: ≈ 75, Pb: ≈ 35, Cu: ≈ 30, Zn: ≈ 85	Varian atomic absorption spectrophotometry	Field study: terrestrial mining deposits	The uptake was independent of the tolerance characteristics	Qureshi et al. 1985
Apera spica-venti (L.) P. Beauv. (windgrass)	units: mg/Kg aerial parts: Fe: 1056.5, Ni: 63, Cr: 7.2, Mn: 77.6, Co: 3.34, Cu: 33.65, Pb: 7, Zn: 32.35	Spectrometric analysis technique	Field study: terrestrial serpentine waste dump	Concentrations of all the examined metals were higher in underground parts than in aboveground parts.	Koszelnik-Leszek 2012
Arthraxon hispidus (Thunb.) Makino (Small carpetgrass)	units: mg/Kg aerial parts: Pb; 8.10 ± 0.19, Cr; 34.93 ± 0.68, Cu; 12.73 ± 2.28, Zn 56.00 ± 8.80, Cd 0.54 ± 0.05 root; Pb;13.46 ± 2.7, Cr; 27.98 ± 6.43, Cu; 16.33 ± 3.73, Zn 51.73 ± 14.48, Cd 1.4 ± 0.38	Inductively coupled plasma-atomic emission spectroscopy	Field study: terrestrial coal mining	A. hispidus seemed suitable for phytostabilization. The results could contribute to the phytoremediation of the similar sites	
Arundo donax L. (common cane)	units: mg/L total plant: Cr: 8.07-141.99, Cd: 4.50-17.90, Pb: 8.68-62.90	Atomic absorption spectrophotometer at AAEPA	Field study: terrestrial high industrial and urban waste	Plant species had a good accumulation capacity of Cr, Pb and Cd for the phytoremediation activity.	Amberber and Kifle 2016

Table 2. (Continued)

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Bromus madritensis L. (compact brome)	units: mg/Kg mining waste: Zn: 720, Cu: 17.4 urban-industrial wastes: Zn: 98, Cu: 10	Plasma emission spectroscopy	Field study: terrestrial urban, industrial and mining wastes	This species demonstrated to grow well in metal contaminated soil.	Pastor and Hernández 2009
Bromus rubens L. (red brome)	units: mg/Kg roots: Cd: 1.271, Cr: 1. 064, Cu: 36.799, Fe: 3036.247, Pb: 30.802, Zn: 118.937, P: 2451.017 shoot: Cd: 0.866, Cr: 4.583, Cu: 15.987, Fe: 357.570, Pb: 1.629, Zn: 58.661, P: 2251.894	Inductively coupled plasma atomic emission spectroscopy	Field study: terrestrial mining activities	This species demonstrated to grow well in metal contaminated soil taking up only low concentrations of metals. B. rubens is good candidate for phytostabilization.	Nouri et al. 2013
Cenchrus ciliaris L. (Buffel grass)	units: mg/Kg roots: Cr: 11, Ni: 22.5, Cu: 10, Zn: 17.5, Cd: 0.5 shoot: Cr: 7, Ni: 8, Cu: 6, Zn: 15, Cd: 0.5	Inductively coupled plasma	Field study: terrestrial Industrial waste water and mining activity	Plants with low shoot accumulation should be used in order to stabilize the heavy metals and reducing the metals dispersion through grazing animals.	Ghoneim et al. 2014
Pennisetum clandestinum Hochst. ex Chiov. (kikuyu grass)	units: mg Cd: 2 mg, 5mg and 10 mg	Inductively coupled plasma-optical emission spectroscopy	Laboratory: soil nitrate [Cd (NO3) 2 4H2O] with different concentrations.	KAR1 treatments and SW used in the species improve phytoextraction potential Cd (602 and 575 mg kg-1 respectively)	Okem et al. 2015

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Pennisetum purpureum Schumach. (elephant grass)	units: mg/Kg roots: Cr; 452.1-1623.1, Cu; 48-65.8, Zn; 260-475, Pb; 14-23.7 shoots: Cr; 1241.6-629.7, Cu; 2.4-65.8, Zn; 306.1-505.9, Pb; 14-15.3 leaves: Cr; 185.6-366.4, Cu; 14.5-39.3, Zn; 290.5-296.5, Pb; 22.4-26.3	Atomic absorption spectrophotometer	Laboratory: soil tannery sludge	P. purpureum has a good Cr accumulation capacity.	Juel et al. 2018
Poa pratensis L. (Zacate azul de las praderas, Kentucky bluegrass)	units: mg/Kg roots: Mn: 6,650.00, Cd: 8.20, Pb: 226.20 leaves: Mn: 1,557.50, Cd: 2.75, Pb: 87.68	Flame atomic absorption spectrometer	Field study: terrestrial mine	This plant was dominant on the Mn mine tailings and had the ability to tolerate high concentrations of Mn, Pb, and Cd.	Liu et al. 2006
Sorghum halepense (L.) Pers. (johnson grass)	units: mg/Kg leaves: Cu: 13.7-50.6, Fe: 2446-6390, Mn: 56.7-99.3, Zn: 167-284, As: 25.1-86.6, Cd: 0.76-1.50, Ni: 2.49-3.74, Pb: 47.4-196, Tl: 0.29	Inductively coupled plasma spectrophotometry	Field study: terrestrial pyrite mine tailings	The species has a high concentration of As, Cd, Cu, Fe and Pb above toxic levels.	Madejón et al. 2002

Table 2. (Continued)

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Polygonaceae					
Reynoutria japonica Houtt. (japanese knotweed)	units: mg/Kg leaves: Al; 140-280, Fe; 191-312, Ca; 1800-19620, K; 6270-15460, Na; 75-2719, Mg; 2670-6350, S; 1350 -2380 stems: Al; 50-570, Fe; 592-6268, Ca; 3300-9820, K; 5420-19150, Na; 98-2272, Mg; 360-8245, S; 280-730 rhizome: Al; 770, Fe; 517, Ca; 7570, K; 5270, Na; 164, Mg; 1250, P; 1600, S; 1180 seeds: Al; 20, Fe; 62, Ca; 1900, K; 11750, Na; 49, Mg; 2650, S; 2370	Inductively coupled plasma optical emission spectrometry	Field study: terrestrial urban parks.	R. japonica can accumulate Zn, Cd and Pb. The stems contain more Zn than leaves, seeds and rhizome.	Rahmonov et al. 2019
Apiaceae					
Foeniculum vulgare Mill. (fennel)	units: µg/Kg total plant: Cd: 63 ± 1.9, Pb: 351 ± 4.9, Hg: 8 ± 0.002, Sn:35.5 ± 2.9	ICP-AES	Field study: terrestrial natural exposure to soil heavy metals	Plants accumulated Pb at higher concentrations	Ebrahim et al. 2012

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Poaceae					
Polypogon monspeliensis (L.) Desf. (annual rabbitsfoot grass)	units: mg/g dead biomass: Cd (II): 10.875	Infrared spectroscopy	Laboratory: soil industry: Cd released through aquatic route.	Waste biomass was found to be very effective in Cd (II) removal from aqueous solution	Ansari et al. 2009
Ruppiaceae					
Ruppia maritima L. (beaked tasselweed)	units: cpm/g total plant: Cr; 1500-2500	Not specified	Laboratory: aquatic solutions of 51Cr (III and IV).	In the species the absorption of Cr (VI) increases with time, because this species is more resistant to the hexavalent form and useful for any salinity as well.	Araujo et al. 2017.
Tamaricaceae					
Tamarix ramosissima Ledeb. (salt cedar)	units: mg/Kg roots:Cu: 51.28, Fe: 205.78, Pb: 12.29, Zn: 88.18 shoot:Cu: 65.68, Fe: 388.9, Pb: 20.9, Zn: 57.43	Flame atomic absorption method	Field study: terrestrial mining activities	This species can grow on soil with concentrations of a particular element that are toxic to most other plants.	Nematian and Kazemeini 2013
Urticaceae					
Cecropia peltata L. (trumpet tree)	units: ng/g roots: Hg; 3714.15 stem: Hg; 176. 9 leaves: Hg; 329.5	Atomic absorption spectrophometry	Laboratory: greenhouse mining activities	C. peltata has the capacity to be used in phytoremediation of soils contaminated with Hg.	Vidal et al. 2010

Table 2. (Continued)

Scientific and common name	**Type of metal and concentration. Structure analyzed**	**Technique**	**Study conditions and source of pollution**	**Comments**	**Reference**
Verbenaceae					
Lantana camara L. (five bold)	units: mg/Kg roots: Pb; 4.43-18.57 shoots: Pb; 2-13.51	Inductively coupled plasma (ICP)	Field study: terrestrial standard solution suprapur grade of Pb	L. camara exhibits a phytoextraction phenomenon of Pb.	Alaribe and Periatamby 2017
Asphodelaceae					
Asphodelus fistulosus L. (nionweed)	units: µg/g stem: Cd: 0.071-0.118, Cu: 5.281-15.359, Pb: 0.038-0.827, Zn: 15.274-42.210 roots: Cd: 0.064-0.281, Cu: 3.618-9.571, Pb: 0.038-0.199, Zn: 8.094-20.115	Inductively coupled plasma-mass spectrometry	Field study: terrestrial traffic	The measured concentrations were much less than those reported by other local and international studies	Al-fawwaz et al. 2017
Zingiberaceae					
Hedychium coronarium J. Koenig (butterfly lily)	units: µg/g shoot: Pb: 0.710, Hg: ND, Cd: 0.041, As: ND, Fe: 2.33, Cu: 0.176, Cr: 0.042 rhizome: Pb: 0.792, Hg:0.001, Cd: 0.057, As: ND, Fe: 0.799, Cu: 0.040, Cr: 0.096	Atomic absorption spectrophotometer	Field study: aquatic wetland plants, wastewater fertilizers and pesticides	Fe was recorded highest in plant tissues followed by Pb	Meitei 2016

CONCLUSION

This review denotes the high capacity of invasive species to colonize habitats altered by anthropogenic disturbances that promote exposure to heavy metals. Although only one-third of the HM bioaccumulator species has been proposed as hyperaccumulators, and therefore useful in phytoremediation processes, there is also a lack of studies that strengthen the importance of using invasive species in bioremediation processes of contaminated habitats. The order of plant families with the highest number of species that bioaccumulate HM may be biased by the number of specialists researching per plant family.

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