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Invasões Biológicas no Brasil: áreas de conservação e áreas urbanas

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Introdução

A globalização permite que pessoas e produtos sejam transportados pelo mundo de forma fácil e rápida. Associado à essa globalização, muitas espécies tiveram suas áreas de ocorrência expandidas por auxílio antrópico. Espécies exóticas são aquelas que, devido a uma introdução intencional ou acidental feita com auxílio humano, se estabelecem em locais fora de sua área de ocorrência natural. Algumas destas espécies exóticas, denominadas invasoras, tem a habilidade de se propagar na natureza em locais distantes do local de introdução (Richardson et al 2008). Espécies exóticas invasoras podem alterar o funcionamento do um ecossistema ao alterarem os padrões de diversidade locais. Processos de invasão biológica têm sido responsáveis por significativas mudanças na composição e estrutura da vegetação nativa de diversos locais do mundo, além de alterar o funcionamento dos ecossistemas (D'Antonio and Meyerson 2002). A biodiversidade tem papel fundamental na extensão deste impacto (Zavaleta and Hulvey 2004). Locais com baixa diversidade funcional de espécies se apresentam mais suscetíveis à invasão biológica, portanto locais previamente invadidos, tendo apresentado por consequência uma queda na biodiversidade nativa, se tornam mais suscetíveis à novas invasões (Maron and Marler 2007). O Brasil é um país que apresenta uma alta diversidade que está sendo ameaçada por muitos fatores, incluindo invasão biológica. Por estas razões o estudo dos impactos causados por espécies invasoras no Brasil é importante para a criação de mecanismos para a conservação da biodiversidade e de serviços ecossistêmicos. Esta dissertação está divida em dois capítulos sobre invasão biológica, o primeiro com foco em espécies invasoras em áreas urbanas e o segundo em um estudo de caso de invasão de pinheiros em uma área de Cerrado.

O crescimento econômico e a urbanização aumentam o movimento de pessoas e bens, consequentemente aumentando as pressões ambientais sobre os habitats naturais e a diversidade local. A transformação do habitat, resultado da expansão urbana, pode influenciar os ecossistemas e possivelmente permitir a colonização e invasão por espécies exóticas. Desta forma, a urbanização e as invasões biológicas quando combinadas representam uma grande ameaça para a biodiversidade nativa. O Brasil é um país megadiverso e com áreas urbanas em expansão, por este motivo é importante agrupar o conhecimento existente sobre invasões urbanas para dar origem a atividades de conservação nas cidades brasileiras. No primeiro capítulo da dissertação, realizamos uma revisão sistemática de estudos publicados sobre invasões biológicas em áreas urbanas no Brasil. Encontramos 24 publicações que se encaixavam em nosso critério (mencionar claramente áreas urbanas e invasões), 93 casos de invasão biológica em urbanas distribuídas em 11 Estados brasileiros. Dos 103 municípios com relatos de invasões, somente 13 registraram mais de uma espécie invadindo áreas urbanas. As espécies terrestres foram o grupo mais frequente de invasores, principalmente espécies de plantas. A maioria dos estudos que encontramos não abordou aspectos mais amplos da biologia da invasão urbana, portanto nossos resultados possuem um baixo poder de inferência e generalização nas consequências da invasão biológica na biodiversidade urbana. Duas das espécies invasoras mais relatadas nas áreas urbanas brasileiras, *Achatina fulica* e *Aedes aegypti*, são vetores bem conhecidos de doenças infecciosas e são atualmente uma grande ameaça para a saúde pública nas áreas urbanas. O número de casos de Dengue, Zika e Chikungunya aumentou no Brasil em 2016, apesar da despesa do Brasil de R $ 1,25 bilhão (cerca de US $ 402 milhões) em 2015 pelo controle dos mosquitos invasores. O conhecimento sobre espécies invasoras pode ocasionar uma redução no gasto de dinheiro público. Nossa revisão mostra que há grandes lacunas no conhecimento sobre espécies invasoras, especialmente em áreas urbanas. É preciso incentivar o estudo de invasões biológicas para que possamos criar meios para a conscientização da população e a conservação da diversidade nativa urbana. Espécies invasoras são uma ameaça potencial tanto à biodiversidade urbana e ecossistemas naturais, quanto para a saúde humana nas áreas urbanas.

As plantas são transportadas ao redor do globo por muitos anos, e por razões diversas. Uma vez introduzidas em uma nova área, elas podem estabelecer, espalhar e invadir. As plantas invasoras têm o potencial de modificar os padrões da comunidade ecológica local e sua função

(Simberloff et al. 2010). O Cerrado é um bioma brasileiro considerado, por sua alta riqueza endêmica de plantas e animais, para ser um *hotspot* para a biodiversidade. Apesar de ser um hotspot, apenas 7,5% de sua área está protegida. As plantas invasoras estão entre as muitas ameaças à diversidade do Cerrado. Os pinheiros foram plantados em todo o mundo por diversas razões, especialmente sociais e econômicas. Espécies de *Pinus* foram registradas como invasoras em biomas brasileiros, e muitas estão presentes em áreas naturais protegidas. Nosso objetivo com o trabalho apresentado no segundo capítulo da dissertação foi entender de que forma a invasão de *Pinus* afeta padrões de diversidade de plantas no Cerrado. Para isso foi utilizada uma área invadida por pinheiros no Jardim Botânico de Brasília (JBB), uma reserva natural que teve em 1976 espécies de pinheiros plantadas como parte de um experimento florestal. Atualmente o JBB conta com dois pinheiros exóticos invasores, *Pinus caribaea* e *Pinus oocarpa*. Nós selecionamos 30 parcelas circulares de 200 m^2, separadas por pelo menos 50 m, e em cada nós medimos a área basal e identificamos todas as espécies de árvores com altura superior ou igual a 1,3 m. A fitofisionomia das parcelas era cerrado denso. Utilizamos testes-t para verificar se a presença de pinheiros afetou a riqueza nativa de árvores, a diversidade (Índice de Diversidade de Shannon - H), a Evenness (E), a Probabilidade de Encontros Interspecíficos (PIE), a densidade e a área basal de espécies nativas e modelos lineares generalizados para testar como a área basal e a densidade do pinheiro afetaram as mesmas variáveis da comunidade nativa. Para as diferenças na composição das espécies, utilizamos uma análise de escalonamento multidimensional não-métrico (NMDS). 19 das 30 parcelas foram encontradas com pinheiros invasores. Encontramos uma menor densidade de espécies nativas em parcelas invadidas ($p<0,05$), e uma relação significativa negativa entre densidade de espécies nativas e densidade e diâmetro basal de pinheiros. Não houve diferença na composição de espécies. Nossos resultados mostram que os pinheiros exóticos têm impactos na comunidade nativa, e que estes devem ser manejados para conservar as áreas protegidas.

Cada vez mais se mostra urgente a conservação da biodiversidade para a manutenção dos processos ecológicos no mundo. O primeiro passo para a conservação é o conhecimento das ameaças. O estudo de invasões biológicas cresce no mundo, e o Brasil, apesar de em um ritmo ainda bastante inferior quando comparado à tendência mundial, acompanha este crescimento. Estudar esses processos é importante do ponto de vista social, econômico e ecológico. Espécies exóticas invasoras no Brasil já foram relacionadas com muitos prejuízos econômicos. A disseminação de doenças por espécies, além de serem um problema de saúde pública, também pode ser considerada um problema econômico devido a quantidade de recursos investidos na contenção destas espécies e no tratamento da população afetada. Muitos são os prejuízos ecológicos associados à invasões biológicas e estes impactos devem ser estudados com cuidado. A quantificação dos impactos deve ser feita para que políticas públicas sejam feitas e a população possa repensar alguns comportamentos ainda bastante comuns no país. Nesta dissertação agrupamos dados que podem servir de alerta para os perigos associados a invasão biológica no Brasil.

Referências

D'Antonio CM, Meyerson L a (2002) Exotic plant species as problems and solutions in ecological restoration. Restor Ecol 10:703–713. doi: 10.1046/j.1526-100X.2002.01051.x

Elton CS (1958) the Ecology of Invasions By Animals and Plants.

Maron J, Marler M (2007) Native plant diversity resists invation at both low and high resource levels. 88:2651–2661. doi: 10.1890/06-1993.1

Simberloff D, Nuñez M a., Ledgard NJ, et al (2010) Spread and impact of introduced conifers in South America: Lessons from other southern hemisphere regions. Austral Ecol 35:489–504. doi: 10.1111/j.1442-9993.2009.02058.x

Zavaleta ES, Hulvey KB (2004) Realistic species losses disproportionately reduce grassland resistance to biological invaders. Science 306:1175–1177. doi: 10.1126/science.1102643

Urban invasions in Brazil and their consequences to the environment and human health: patterns and perspectives

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Abstract

Economic growth and urbanization increases the movement of people and goods, increasing environmental pressures on natural habitats and local diversity. Habitat transformation resulted from urban expansion can influence ecosystems and possibly enable the colonization and invasion of exotic species. Thus, urbanization and biological invasions combined pose great threat to native biodiversity. Brazil is a megadiverse country with expanding urban areas, so it is important to synthesize the existing knowledge on urban invasions in order to support conservation initiatives in Brazilian urban areas. We conducted a systematic review of exotic species invading in urban areas in Brazil and found a relation between human population density and number of exotic species invasions. Terrestrial species, mostly plants, were the most frequent group of invaders. We found 23 studies and 93 invasive species in Brazilian urban areas. Most studies we found did not address broader aspects of urban invasion biology, having low power of inference and generalization on the consequences of biological invasions in urban ecosystems. However, the two most frequently reported invasive species in Brazilian urban areas, *Achatina fulica* (giant African snail) and *Aedes aegypti* (yellow fever mosquito), are well-known vectors of infectious diseases and are currently a major threat to public health in urban areas. In 2016 alone, Brazil registered about 1.5 million cases of Dengue fever, 211,770 cases of Zika fever, and 263,598 cases of Chikungunya. *Aedes aegypti* is the main vector for all those diseases. In conclusion, we argue that invasive species are a potential threat to urban ecosystems and human health in urban centers, as much as they are for natural ecosystems.

Keywords: Biodiversity, Biological invasions, Brazilian urban areas, dengue fever, public health, Urbanization

Introduction

Urbanization can be considered one of the most prevailing causes of habitat transformation worldwide, showing an intense level of human activity (McKinney 2002). About 50% of the human population currently reside in cities, with this proportion rapidly increasing in the past few decades (Ellis and Ramankutty 2008). Urban expansion events can influence both terrestrial and aquatic ecosystems and change environments (Parmesan et al. 2000). It is expected that urban expansion will encroach native ecosystems, presenting a unique scenario for colonization of transformed environments by species that exhibit superior competitive ability, as many exotics do (Pauchard and Barbosa 2013; Pyšek et al. 2004). Invasive species are known to impact the environment by altering ecological dynamics between native species, possibly leading to changes in the structure of communities (Hooper et al. 2005; Frehse et al. 2016). Altered species composition may lead to considerable changes on the functioning of ecosystems (D'Antonio et al. 2005).

The result of large number of exotic species across ecosystems is biotic homogenization, a known consequence of biological invasions that can lead to losses of global biodiversity (Kühn et al. 2003; McKinney 2006; Olden 2008). Like biological invasions, urbanization has also been associated with biotic homogenization (Kühn et al. 2003; Kühn and Klotz 2006). Thus, when combined, urbanization and biological invasions may pose great threat to biodiversity. Urban areas hold characteristics that can help maintain biodiversity, in spite of the environmental pressures it produces. The presence of patches of natural habitats within urban areas, such as small parks, can serve as biodiversity islands, functioning as recolonization sources (Kühn and Klotz 2006), green corridors for native species and can help enhance some species local genetic diversity (Savard et al. 2000). Preserving the urban biodiversity can be a crucial approach for biodiversity conservation.

One of the main reasons for the increase in the movement of exotic species throughout the globe is the movement of people and the global transportation networks (Sharma et al. 2010). These movements occur largely from one urban area to another. It has been shown that economic growth, human development index (HDI) and Gross Domestic Production (GDP) coincides with the increase in number of introduced invasive species (Lin et al. 2007; Sharma et al. 2010; Zenni et al. 2017a). Developed countries have more exotic species than developing countries, and this can be explained by the comparatively higher importation rates (Vila and Pujadas 2001). Those patterns can change with the constant increase in the demand for imported products, which increases the possibility of intentional and unintentional introductions through importation processes. Habitat changes due to urban growth cause environmental disturbances that can both prevent the establishment of native species and enhance the rate of establishment and naturalization of exotic species (Mooney et al. 2001; McKinney 2006). Disturbed and anthropogenically-induced transformed environments are invaded by exotic species more often than natural areas (Sharma et al. 2010), but there is a major gap of knowledge regarding biological invasions in urban ecosystems (Kuebbing et al. 2013). Biological invasions in urban areas are also related to the spread of various diseases, thus posing a great threat to public health. Historically, many human deaths can be attributed to alarming plague episodes, such as smallpox, typhus and bubonic plague. The virulence of infectious diseases is dependent of the density of infected and susceptible populations. Thus, populous urban areas can possibly enhance public chances of disease infections (Delfino and Simmons 2000). Invasions by pathogens can affect not only humans, but also plants and other animals, possibly leading to changes in community structure (Anderson et al. 1986; Zenni et al. 2017b). Knowing the invasion biology of the species that causes the diseases and their vectors is an important tool for conserving native diversity and public health.

Brazil has a long history of introduction of exotic species (Zenni 2014; Zenni 2015), but little is known about invasive species in human-altered ecosystems in Brazil (Zenni and Ziller 2011; Zenni 2015). Urban areas increase environmental vulnerability to invasions as it affects ecosystem functions (Pauchard and Barbosa 2013), and can act as source of invasive species to natural areas as result of intentional and unintentional introduction (Mclean et al. in press). Here, we review records of invasions in Brazil, looking for cases of biological invasion in urban and peri-urban environments in order to evaluate the ecological statuses of Brazilian cities in terms of invasions by exotic species and to review known impacts. Human population density, because of their known impact on the environment (Goudie and Viles 2013), and the constant trade of goods, influence the possibility of an introduced species becoming established or invasive (Essl et al. 2011). Populous Brazilian cities are known to host more naturalized exotic species (Zenni 2015). Our goal here was to see if there is evidence that more populated urban areas, like São Paulo and Rio de Janeiro, present larger numbers of invasions by exotic species as a consequence of intense and constant flow of people and goods.

Material and methods

We surveyed the literature available on biological invasions in Brazil for records of occurrence of urban biological invasions. First, we checked the list of studies provided by Zenni et al. (2016) that reviewed data regarding biological invasions in Brazil, then searched for more literature using personal libraries and online search engines, such as Google Scholar and Web of Science, using urban areas, biological invasions and invasive species as search terms. We made an effort to check the widest array possible of available primary literature reporting urban invasion records. For the present study, we only considered data that explicitly mentioned biological invasions (not presence-only or naturalized-only exotic species *sensu* Zenni et al. 2015) in urban and peri-urban environments. To include a study in our survey, it had to be conducted in Brazil and at least one species had to be exotic with an actual record of invasion in urban or peri-urban area (both referred as "urban area" from this point forward). For each record found, we took note of the following: year of publication, locality, terrestrial biome (Atlantic Forest, Amazon, Caatinga, Cerrado, Pampa, and Pantanal), urban area, State, geographic coordinates of the studied area, and total number of invasive species. For each urban area with data on exotic species invasion in, we collected data on date of foundation, size of the urban area, number of inhabitants, number of urban parks, gross domestic production (GDP), and human development index (HDI). Human development index is calculated using data on mean years of education, population longevity and gross domestic production. Municipal data were compiled from the Brazilian Institute of Geography and Statistics (IBGE 2015). When papers reported more than one case of invasion, or more than one species invading, each case was compiled separately.

We used generalized linear models (GLM) with Poisson family distribution and log link function to test the effects of urban areas' population size, latitude, number of urban park, years since foundation, GDP, and human development index on the number of known invasive non-native species in the urban areas. We originally had plant and animal species analyzed separately. However, given the low number of animal species listed for most municipalities, and the low number of urban areas with data available for plant species, we decided to analyze the total number of species without separating plants and animals. We also removed clear outliers from the analyses because we were unsure on the reliability of reports of very high numbers of invasive species for very small urban areas and of very low number of invasive species for very large urban area (Table 1). These outliers could lead to a false prediction of invasions in urban areas using social-economic patterns. Goodness-of-fit tests for the statistically significant GLM were performed using the McFadden's pseudo-r^2.

Results

We found data on 93 exotic species invading urban areas (Table 1) for 103 municipalities in Brazil gathered from 24 publications (Table 2). These urban areas were distributed across 11 States, of a total of 26 Brazilian States and the Federal District (Fig. 1). However, the overall number of exotic species reported invading per urban area was generally very low (mean = 2), and 90 urban areas had only one invasion recorded. Crato, Ceará, was the urban area with the greatest number of invasions recorded (n=37), followed by Caruaru, Pernambuco (n=36), Maringá, Paraná (n=16), and Curitiba, Paraná (n=13). Of the 105 urban areas recorded, 72% were in the state of São Paulo, 8% in the State of Santa Catarina, 6% in Rio Grande do Sul, and 3% in Rio de Janeiro (Fig. 4). We found records of 92 different urban invasions, of which 90.2% were invasions by plants and 8.7% were by animals.

The total number of habitants in an urban area in Brazil, the more invasion occurrences were recorded ($z=5.2$, $p<0.001$, pseudo-$r^2=0.2$;). However, the rate of accumulation of invasions was very low compared to the increase of the human population. There were no association of latitude with number of invasions in an urban area ($p>0.05$). There were also no association between time of existence and number of biological invasions in an urban area ($p>0.05$). There were no significant association between number of invasions and GDP. After removing two outliers (small cities with high number of invasions), the urban areas with the highest number of invasion records were the ones with the highest Gross Domestic Production (GDP). Higher Human Development Indexes were associated to lower numbers of invasion records ($t=-2.2$, $p=0.03$; Fig 2).

Discussion

Based on the census made by the Brazilian Institute of Geography and Statistics (IBGE 2015), there are a total of 5,565 urban areas in Brazil, but only 105 (≈2%) had records of urban invasion. This result is probably an underestimation because most regions in Brazil were seldom or never sampled or studied regarding exotic species (Frehse et al. 2016; Zenni 2015). Also, several widespread invasive species (i.e. *Achatina fulica*, *Apis melifera* and *Aedes spp*.), although well-known by researchers and authorities, are often not reported as invaders. Even considering the lack of data, approximately half of the States have at least one record of urban invasion. Of the 24 publications gathered for this study, only seven recorded more than two exotic species invading. South America presents a wide latitudinal gradient that encompasses a great diversity of climates and habitats. However, our results showed no significant correlation between number of invasive species in urban areas and latitude. There was also no significant correlation between an urban area time of existence and the number of biological invasions. We found an association between number of invasions and quantity of economical trades (Fig. 2), so the economic status and human population density possibly influence more than the time of existence (Lin et al. 2007).

The intense movement of people and goods is probably one of the main drivers of biological invasions (Dalmazzone et al. 2000). In Brazil, more populated and more deforested biomes host more naturalized species (Zenni 2015), which is an intermediate stage towards successful invasions (Blackburn et al. 2011). Even though our results showed only a weak association between population size and the number of invasions, the urban areas with more records were Crato and Caruaru. These results were due to detailed inventories of species performed at those two urban areas (Dos Santos et al. 2014), which were not performed in any other Brazilian urban area. Detailed inventories can increase manifold the number of species recorded in an urban area (Mclean et al. in press). Even though these urban areas are probably the only ones with accurate numbers of exotic species invasions, they may be considered outliers for our study. After removing these outliers, urban areas with the highest number of exotic species invasions were the ones with the highest Gross Domestic Production (GDP). A similar pattern was found in China, where economic development was found to be a predictor of biological invasion (Lin et al. 2007). However, this relation may be explained by funding availability for research, surveys, and cataloguing of biological invasions (Sharma et al. 2010). In Brazil, the two urban areas with the largest GDP, São Paulo and Rio de Janeiro (IBGE 2015), were not the urban areas with most records of urban invasions. Also, our results showed a negative association between Human Development Index (HDI) and number of invasive species, contrary to the results found by Sharma et al. (2010).

High human population density can be associated with higher exposure to pathogens, so invasions by exotic species that can act as vectors of diseases pose great threat to human health

(Delfino and Simmons 2000). In our review, we found the giant African snail (*Achatina fulica*) was recorded invading 74 different Brazilian urban areas. The African snail has been related with transmission of eosinophilic meningitis in Brazil and Colombia (Nogueira et al. 1999) and is a well-known biological invader (PAHO 1995). *Aedes aegypti,* and *A. albopictus* were also reported invading many urban areas and are likely invading many more areas considering the number and distribution of Dengue fever cases (Health Ministry of Brazil 2016). *Aedes* spp. are well-known invasive species associated with sanitary problems in urban areas (Nogueira et al. 1999; Cardoso and Câmara 2015), transmitting viral diseases like Dengue fever, yellow fever, Zika, and Chikungunya (Frehse et al. 2016; Mondini 2015). Both *Aedes* species were found in urban areas, but *A. aegypti* is more frequently found in areas with elevated human density and *A. albopictus* less dense areas (Cardoso and Câmara 2015). In 2016 alone, Brazil registered 1,487,924 cases of dengue fever, 211,770 cases of Zika fever, and 263,598 cases of Chikungunya (Health Ministry of Brazil 2016). The number of cases of Dengue fever, Zika and Chikungunya virus increased in Brazil in 2016 (LIRAa - Quick Survey of Infestation Index by *Aedes aegypti*) despite an expense of R$1.25 billion (*ca.* USD 402 million) in 2015 for controlling the invasive mosquitos (Health Ministry of Brazil 2016). The increase in the *Aedes* populations is possibly a result of changes in temperature amplitudes, as a direct consequence of climate change (Kraemer 2015). Dengue, Zika, and Chikungunya fevers are largely associated with urban and peri-urban areas, and their occurrence closely matches to the geographic distribution of the invasive vector (Pauchard et al. 2011). Besides Brazil, Dengue and Chikungunya have become a global threat to public health, putting about half of the world's population at risk (Kraemer 2015). Economic development and modern life style may enhance the spread of diseases as it increases the trades of goods and movement of people, and the production of solid waste that can serve as reproduction site to many disease vectors (Cardoso and Câmara 2015). Controlling urban invasive species can be an important tool for controlling the spread of infectious diseases. Therefore, it should be seen as, in some cases, a benign form of protecting public health.

Aedes aegypti population has been successfully controlled in the past in Brazil, but the ever-growing human population density and the economic recession made it difficult to maintain the control program (Wermelinger and Carvalho 2016). Recently, the water-supply crisis in São Paulo made people improvise for storing more water, creating even more breeding ground for *Aedes* mosquitoes. Also, the increased demand in transporting water can enhance the distribution of contaminated water to other locations (Marcondes and Ximenes 2016). Rainwater storages, household deposits and garbage are the most common breeding sites in Brazilian urban areas (Boechat 2015). Only with a combined effort of population and government will it be possible to repeat the results obtained in Brazil once before (Valle 2016).

Invasive species can generate economic losses and exotic and invasive species have a significant economic toll throughout the globe (Pimentel et al. 2001). Pimentel et al. (2001) assessed that more than US$ 336 billion per year are spent as a consequence to invasion by exotic species. This amount, according to the authors, was an underestimation because precise economic costs associated with invasive species are not available and they argue that if losses in biodiversity could be financially quantified, this estimation would be much higher. Pimentel et al. (2001) suggested that preventing future introduction of potentially harmful exotic species could avoid the waste of billions of dollars in losses to agriculture, forestry, and other aspects of natural and managed environment worldwide. Thus, biological invasion could even be a step back to the progress of developing countries such as Brazil, that have great biodiversity and an emergent economy.

There was only one record of aquatic urban invader (*Poecilia reticulata*) in the 24 publications found (Junqueira et al. 2016), this result is consistent with the literature on invasions available for other Latin American countries (Fearnside 2005). The lack of information about aquatic ecosystems does not mean lack of threats by invasive freshwater species. Aquatic ecosystems in urban areas are specially threatened because of the high number of vectors (ship transportation of people and goods) and because of the aquatic environments physical characteristics (Maron and Marler 2007). Leprieur et al. (2008) reported the global patterns of freshwater fish invasion in river basins covered more than 80% of Earth's continental surface. Despite the importance of aquatic ecosystems, most of the studies for urban areas were done in the terrestrial environments of the Atlantic Rainforest, also consistent with Brazil's scientific ecological literature (Frehse et al. 2016; Zenni et al. 2016). Those regions hold 56% of Brazilian population and 69% of all de Ph.D. researchers (Lodge et al. 1998), so it is expected that these ecoregions retain the majority of the scientific publications (Lodge et al. 1998).

The management of biological invasions in general requires some level of public awareness and support (Novoa et al. this issue), but we currently know very little about invasions in Brazilian urban areas. The lack of awareness of the potential impacts combined with the economic value attributed to many exotic species possibly promotes their expansion in urban and peri-urban areas and increases the threats to native diversity. Educating the urban population about the environmental and economic importance of native species can be part of the solution for this problem. The negative impacts of biological invasions are not a subject in the curricula of South American schools' systems (Olden et al. 2004), therefore the population's awareness cannot be expected. Without a global strategy involving education, research, and policies, people will continue using and protecting exotic species over native ones, endangering diversity in the most biodiverse regions of the world (Olden et al. 2004).

There is a growing number of databases of exotic plants and animals in Brazil (Zenni et al. 2016), but reliable and balanced data for the unbiased comparison regarding levels of invasion are likely to be still difficult to obtain (Zenni 2015). Brazil's current databases are selective because they are not aimed at providing a broader overview of global invaders, but rather at the impact of single species or taxa at local scales. Therefore, our work reflects what is recorded in scientific literature rather than the actual state of biological invasions in urban areas.

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References

Anderson RM, May RM, Joysey K, Mollison D, Conway GR, Cartwell R, Thompson HV, Dixon B (1986) The Invasion, persistence and spread of infectious diseases within animal and plant communities [and discussion]. Philos Trans R Soc B Biological Sciences 314:533–570

Bellard C, Jeschke JM (2016). A spatial mismatch between invader impacts and research publications. Conserv Biol 30(1):230-232

Blackburn TM, Pyšek P, Bacher S, Carlton JT, Duncan RP, Jarošík V, Wilson JRU, Richardson DM (2011) A proposed unified framework for biological invasions. Trends Ecol Evol 26:333–339

Boechat B (2015) Ministério da Saúde divulga principais criadouros de dengue por região. Portal Extra Globo (Accessed 2016 Feb 7th). Available at: http://extra.globo.com/noticias/saude-e-ciencia/dengue/ ministerio-da-saude-divulga-principais-criadouros-de-dengue-por- regiao-11921245.html

D'antonio C, Meyerson LA (2002) Exotic plant species as problems and solutions in ecological restoration: a synthesis. Restor Ecol 10:703–713

Dalmazzone, S. (2000). Economic factors affecting vulnerability to biological invasions. The economics of biological invasions, 17-30

Delfino D, Simmons P (2000) Infectious diseases as invasive in human populations. The economics of biological invasions. EE Publishing, Cheltenham, UK, 31-55

Ellis EC, Ramankutty N (2008) Putting people in the map: anthropogenic biomes of the world. Front Ecol Environ 6:439–447

Elton CS (1958) the Ecology of Invasions By Animals and Plants.

Essl F, Dullinger S, Rabitsch W, Hulme PE, Hülber K, Jarošík V, Kleinbauer I, Krausmann F, Kühn I, Nentwig W, Vilà M, Genovesi P, Gherardi F, Desprez-Loustau ML, Roques A, Pyšek P (2010) Socioeconomic legacy yields an invasion debt. Proc Natl Acad Sci USA 108:203–207

Fearnside PM (2005) Deforestation in Brazilian Amazonia: History, Rates, and Consequences. Conserv Biol 19:680–688

Frehse FDA, Braga RR, Nocera GA, Vitule JRS (2016) Non-native species and invasion biology in a megadiverse country: scientometric analysis and ecological interactions in Brazil. Biol Invasions doi: 10.1007/s10530-016-1260-9

Fridley JD, Stachowicz JJ, Naeem S, Sax DF, Seabloom EW, Smith MD, Stohlgren TJ, Tilman D, Holle BV (2007) The invasion paradox: reconciling pattern and process in species invasions. Ecology 88:3–17

Garcia ACL, Valiati VH, Gottschalk MS, Rohde C, Valente VLS (2008) Two decades of colonization of the urban environment of Porto Alegre, southern Brazil, by *Drosophila paulistorum* (Diptera, Drosophilidae). Iheringia Série Zoo 98(3):329-338

Honório NA, Câmara DCP, Calvet GA, Brasil P (2015) Chikungunya: an arbovirus infection in the process of establishment and expansion in Brazil. Cad Saúde Pública 31:906-908

Hooper DU, Chapin FS, Ewel JJ, Hector A, Inchausti P, Lavorel S, Lawton JH, Lodge DM, Loreau M, Naeem S, Shmid B, Setälä H, Symstad AJ, Vandermeer J, Wardle DA (2005). Effects of biodiversity on ecosystem functioning: a consensus of current knowledge. Ecol Monogr 75(1):3-35

Mondini, A. (2015) Zika: another public health threat transmitted by *Aedes aegypti*. Veterinária e Zootecnia, 23(3), 27-28

Junqueira, N. T., Macedo, D. R., Souza, R. C. R. D., Hughes, R. M., Callisto, M., & Pompeu, P. S. (2016) Influence of environmental variables on stream fish fauna at multiple spatial scales. Neotrop Ichthyol, 14(3)

Kraemer MUG, Sinka ME, Duda KA, et al. (2015) The global distribution of the arbovirus vectors *Aedes aegypti* and *A. Albopictus*. eLife 4

Kuebbing SE, Nuñez MA, Simberloff D (2013) Current mismatch between research and conservation efforts: the need to study co-occurring invasive plant species. Biol Conserv 160:121–129

Kühn I, Brandl R, May R, Klotz S (2003) Plant distribution patterns in Germany– will aliens match natives? Feddes Repert 114:559–573

Kühn I, Klotz S (2006) Urbanization and homogenization – comparing the floras of urban and rural areas in Germany. Biol Conserv 127:292–300

Lambertucci SA, Speziale KL (2011) Protecting invaders for profit. Science 332:35–35

Lin W, Zhou G, Cheng X, Xu R (2007) Fast economic development accelerates biological invasions in China. PLoS One 2(11):e1208

Lodge DM, Stein RA, Brown KM, Covich AP, Brönmark C, Garvey JE, Klosiewskt SP (1998) Predicting impact of freshwater exotic species on native biodiversity: challenges in spatial scaling. Aust J Ecol 23(1):53-67

Lowe S, Browne M, Boudjelas S, Boudjelas S, De Poorter M (2001) 100 of the world's worst invasive alien species. Invasive Species Specialist Group, IUCN, Gland

Lowry E, Rollinson EJ, Laybourn AJ, Scott TE, Aiello-Lammens ME, Gray SM, Mickley J, Gurevitch (2013) Biological invasions: a field synopsis, systematic review, and database of the literature. Ecol Evol 3(1):182-196

Marcondes CB, and Ximenes MdeFFdeM (2016) Zika virus in Brazil and the danger of infestation by *Aedes* (Stegomyia) mosquitoes. Rev Soc Bras Med Trop 49, 4-10

Maron J, Marler M (2007) Native plant diversity resists invasion at both low and high resource levels. Ecology, 88(10): 2651-2661

McGeoch MA, Butchart SHM, Spear D, Marais E, Kleynhans EJ, Symes A, Chanson J, Hoffmann M (2010) Global indicators of biological invasion: species numbers, biodiversity impact and policy responses. Divers Distrib 16(1):95-108

McKinney ML (2002) Urbanization, biodiversity, and conservation: the impacts of urbanization on native species are poorly studied, but educating a highly urbanized human population about these impacts can greatly improve species conservation in all ecosystems. Bioscience 52(10):883-890

McKinney ML (2006) Urbanization as a major cause of biotic homogenization. Biol Conser 127(3):247-260

McLean P, Gallien L, Wilson JRU, Gaertner M, Richardson DM (in press) Small urban centres as launching sites for plant invasions in natural areas: insights from South Africa. Biol Inv

Mooney A, Elsa E (2001) "The evolutionary impact of invasive species." Proc Natl Acad Sci USA 98(10):5446-5451

Myers N, Mittermeier RA, Mittermeier CG, da Fonseca GAB, Kent J (2000) Biodiversity hotspots for conservation priorities. Nature 403(6772):853-858

Nogueira RMR, Miagostovich MP, Schatzmayr HG, dos Santos FB, de Araújo ESM, de Filippis AMB, de Souza RV, Zagne SMO, Nicolai C, Baran M, Filho GT (1999). Dengue in the state of Rio de Janeiro, Brazil, 1986-1998. Memó Inst Oswaldo Cruz 94(3):297-304

Novoa A, Dehnen-Schmutz K, Fried J, Vimercati G (in press) Does public awareness increase support for invasive species management? Promising evidence across taxa and landscape types. Biol Inv

Olden JD (2008). Biotic homogenization. In: Encyclopedia of Life Sciences (ELS). John Wiley & Sons, Ltd: Chichester. doi: 10.1002/9780470015902.a0020471

Olden JD, Poff NL, Douglas MR, Douglas ME, Fausch KD (2004) Ecological and evolutionary consequences of biotic homogenization. Trends Ecol Evol 19(1):18-24.

Organizacion Panamericana de la Salud, Pan American Sanitary Bureau, & Pan American Sanitary Bureau Staff. (1995) Dengue y dengue hemorrágico en las Américas: guías para su prevención y control (No. 548) Organización Panamericana de la Salud, Oficina Sanitaria Panamericana, Oficina Regional de la Organización Mundial de la Salud.

Parmesan C, Root TL, Willig MR (2000) Impacts of extreme weather and climate on terrestrial biota. Bull Amer Met Soc 81(3):443-450

Pauchard A, Alaback PB (2004) Influence of elevation, land use, and landscape context on patterns of alien plant invasions along roadsides in protected areas of South-Central Chile. Conser Biol 18(1):238-248

Pauchard A, Barbosa O (2013) Regional assessment of Latin America: rapid urban development and social economic inequity threaten biodiversity hotspots. In *Urbanization, biodiversity and ecosystem services: Challenges and opportunities* (pp. 589-608). Springer Netherlands.

Pauchard A, Quiroz CL, Garcia R, Anderson CB, Kalin MT (2011) Invasiones biológicas en América Latina y el Caribe: Tendencias en la investigación para la conservación. In: Simonetti JA, Dirzo R (eds) Conservación biológica: Perspectivas desde América Latina. Ed Uni, Santiago, pp 79–90

Pimentel D, McNair S, Kanecka J, Wightman J, Simmonds C, O'Connell C, Wong E, Russel L, Zern J, Aquino T, Tsomondo T (2001) Economic and environmental threats of alien plant, animal, and microbe invasions. Agric Ecosyst Environ 84(1):1-20

Pyšek P, Richardson DM, Pergl J, Jarošík V, Sixtová Z, Weber E (2008) Geographical and taxonomic biases in invasion ecology. Trends Ecol Evol 23(5):237-244

Pyšek P, Richardson DM, Rejmánek M, Webster GL, Williamson M, Kirschner J (2004) Alien plants in checklists and floras: towards better communication between taxonomists and ecologists. Taxon 53(1):131-143

Savard J-PL, Clergeau P, Mennechez G (2000) Biodiversity concepts and urban ecosystems. Landsc Urban Plan 48(3):131-142

Sax D (2008) Latitudinal gradients and geographic ranges of exotic species: implications for biogeography. J Biogeogr 28(1):139-150

Schüttler E, Rozzi R, Jax K (2011) Towards a societal discourse on invasive species management: a case study of public perceptions of mink and beavers in Cape Horn. J J Nat Conserv 19(3):175-184

Sharma GP, Esler KJ, Blignaut JN (2010) Determining the relationship between invasive alien species density and a country's socio-economic status. S Afr J Sci 106(3-4):1-6

Simberloff D, Vitule JRS (2013) A call for an end to calls for the end of invasion biology. Oikos 123(4):408-413

Speziale K, Lambertucci S (2010) A call for action to curb invasive species in South America. Nature 467(7312):153-153

Speziale KL, Lambertucci SA, Carrete M, Tella JL (2012) Dealing with non-native species: what makes the difference in South America? Biol Invasions 14:1609–1621

Tilman D, Reich PB, Knops J, Wedin D, Mielke T, Lehman C (2001) Diversity and productivity in a long-term grassland experiment. Science 294(5543):843-845

Valle D (2016) No magic bullet: citizenship and social participation in the control of *Aedes aegypti*. Epidemiol Serv Saude 25(3):629-632

Vilà M, Pujadas J (2001) Land-use and socio-economic correlates of plant invasions in European and North African countries. Biol Conserv 100:397–401

Wermelinger ED and Carvalho RWde (2016) Métodos e procedimentos usados no controle do *Aedes aegypti* na bem-sucedida campanha de profilaxia da febre amarela de 1928 e 1929 no Rio de Janeiro. Epidemiol Serv Saude 25(4):837-844

Zenni RD (2014) Analysis of introduction history of invasive plants in Brazil reveals patterns of association between biogeographical origin and reason for introduction. Austral Ecol 39(4):401-407

Zenni RD (2015) The naturalized flora of Brazil: a step towards identifying future invasive non-native species. Rodriguésia 66(4):1137-1144

Zenni RD, Dechoum MDS, Ziller SR (2016) Dez anos do informe brasileiro sobre espécies exóticas invasoras: avanços, lacunas e direções futuras. Biotemas 29(1):133-153

Zenni RD, Ziller SR, Pauchard A, Rodriguez-cabal M, Nuñez MA (2017a) Invasion science in the developing world: a response to Ricciardi et al. Trends Ecol Evol in press

Zenni RD, Dickie IA, Wingfield MJ, Hirsch H, Crous CJ, Meyerson LA, Burgess TI, Zimmermann TG, Klock MM, Siemann E, Erfmeier A, Aragon R, Montti L, Le Roux JJ (2017b) Evolutionary dynamics of tree invasions: complementing the unified framework for biological invasions. AoB Plants 10.1093/aobpla/plw085

Ziller SR, Reaser JK, Neville LE, Brand K (2005) Invasive alien species in South America: national reports and directory of resources. Global Invasive Species Program, Cape Town, South Africa

Table and Figure Legends

Table 1 List of the invasive species reported in 24 publications about urban areas in Brazil

Figure 1 Number of invasions recorded in the seven most invaded Brazilian urban areas

Appendix

Table 2 Description of each city as to total area (km^2), number of inhabitants and number of invasive species (flora or fauna) found in urban areas

Table 1

Species	Origen	Invader on non-urban areas	Invader elsewhere	Also urban areas	Impacts	Data Source
Acanthospermun hispidum	Unkown	Unkown	Unkown	Unkown	Unkown	
Achatina fulica	East Africa	Yes	The United States (Hawaii and Florida where it was eradicated), China and several other Asian countries, Australia (Christmas islands), France (New Caledonia), Martinique, Maldives, Philippines	Yes	Predator of plants, competitor, agricultural pest, vector of diseases	I3N
Aedes aegypti	African continent	Unkown	Argentina and United States	Yes	Human health (disease transmitter)	13N
Aedes albopictus	Asia	Yes	United States, Caribbean, Countries of South America and Europe	Yes	Competition with native species, use of chemical products for containment, disease transmitters	I3N
Ageratum conyzoides	Unkown	Yes	Unkown	Unkown	Used as a natural medicine	GRIIS
Amaranthus spinosus	Unkown	Unkown	Argentina, Bhutan, Bulgaria, Canada, China, Colombia, Czech Republic, Ecuador, France, Germany, Ghana, Greece, Haiti, Lao People's Democratic Republic, Madagascar, Malaysia, Mauritius, Norway, Papua New Guinea, Philippines, Republic of Korea, Republic of Moldova, Singapore, Suriname, Sweden, The former Yugoslav Republic of Macedonia, Venezuela, Algeria, Australia, Belgium, Burundi, Cambodia, Cameroon, Chad, Croatia, Cuba, Cyprus, Denmark, Fiji, India, Israel, Japan, Kiribati, Latvia, Marshall Islands, Micronesia (Federated States of), Myanmar, Nauru, Nepal, Pakistan, Palau, Portugal, Romania, Russian Federation, Samoa, Solomon Islands, South Africa, Taiwan, Vanuuatu, Viet Nam, Yemen, Zimbabwe.	Unkown	Unkown	GRIIS
Anemopaegma laeve	Unkown	Unkown	Unkown	Unkown	Unkown	
Anthephora hermaphrodita	American continent	Unkown	Unkown	Unkown	Unkown	ITIS Report
Archontophoenix cunninghamiana	Eastern Australia	Yes	New Zeland, France	Yes	Shade out native species, displace native palm species	I3N; GISD; ARC 2008; Christianini 2006; Williams 2008

Artocarpus heterophyllus	India (mountains of the Western Ghats) and Peninsula of Malaysia	Yes	French Polynesian	Unkown	Habitat change, inhibits the growth of other species, modification of successional patterns, ecosystem changes, reduction of natural biodiversity	I3N
Azadirachta indica	Asia	Yes	Australia, the Dominican Republic, Central America, India, Ghana, Gambia and the Sahel region in Africa, as well as other West African countries	Yes	High control costs due to unpredictability of dissemination, inhibits the growth of other species	13N
Bidens bipinata	Asia and North America	Unkown	Unkown	Unkown	Competition with native species	Available at: https://www.invasiveplantatlas.org/subject.html?sub=5184
Blainvillea acmella	Unkown	Unkown	Unkown	Unkown	Unkown	
Brachiaria decumbens	South Africa and East Africa	Yes	Ecuador (Galapagos Islands)	Yes	Dominance on the environment, competitive exclusion of other species	I3N
Canis lupus	Unkown	Yes	Unkown	Yes	Competition, ecosystem changes, virus transmitter	GISD
Cenchrus echinatus	American continent	Yes	New Zealand, Samoa Micronesia, Hawaii	Yes	Displacement of native grasses, weeds in cultivating areas	I3N
Centratherum punctatum	Unkown	Unkown	Unkown	Unkown	Unkown	
Chamaecrista pilosa	North America	Yes	Venezuela, Australia	Yes	Unkown	Available at: http://florida.plantatlas.usf.edu
Chamaecrista rotundifolia	Unkown	Unkown	Unkown	Unkown	Unkown	Unkown
Citrus aurantium	Unkown	Unkown	United States of America	Unkown	Unkown	Available at: https://www.invasiveplantatlas.org
Citrus limon	Asia	Yes	United States, Mexico, Chile, Argentina, Italy, Spain, Greece, Turkey, Lebanon, South Africa, Australia, Fiji, New Caledonia and Ecuador	Yes	Human health	I3N
Cyperus distans	North America	Yes	Fiji Islands, Papua New Guinea, Singapore	Yes	Unkown	Available at: https://plants.usda.gov
Cyperus uncinulatus	Unkown	Unkown	Unkown	Unkown	Unkown	
Dactyloctenium aegyptium	American continent	Yes	Unkown	No	Quickly colonize areas	CABI.org
Datura stramonium	Unkown	Unkown	Unkown	Unkown	Unkown	
Digitaria insularis	American continent	Yes	Philippines, Hawaii, USA, Papua New Guinea, Paraquay,	Unkown	Unkown	Available at:

			Bolivia, Cuba, Oceania			http://www.cabi.org/isc/datasheet/109596
Diodella teres	Unkown	Unkown	Unkown	Unkown	Unkown	
Drosophila paulistorum	Unkown	Unkown	Unkown	Unkown	Unkown	
Drosophila Yesulans	Unkown	Unkown	Unkown	Unkown	Unkown	
Eleusine indica	Japan, Korea, Oman, Yemen, Africa	Yes	Europe, Asia, Central and South America, the Caribbean and on many islands in the Pacific Ocean	Unkown	Unkown	Available at: http://www.cabi.org/isc/datasheet/20675
Eragrostis sp.	Unkown	Unkown	Unkown	Unkown	Unkown	
Eriobotrya japonica	Ásia	No	Havaí (Estados Uunidos), França, Argentina, África do Sul, Índia, Tonga, Austrália, Nova Zelândia.	No	Ocupa o espaço da vegetação nativa.	I3N
Eucalyptus grandis	Unkown	Unkown	Unkown	Unkown	Unkown	
Eucalyptus robusta	Australia, Tasmania	Yes	Caribbean Islands, United States (Hawaii), France (Reunion Islands)	Yes	Conversion of open ecosystems into forest ecosystems, with loss of biodiversity by shading, soil exposure, erosion and possible silting of watercourses, reduction of pastoral area, reduction in water availability, the more severe the lower the index	I3N
Eucalyptus sp.	South Pacific	Yes	United States (Hawaii, California), Mexico, South Africa, West Coast Australia	Unkown	Unkown	13N
Eucalyptus sp.	Oceania	Yes	United States, Mexico, South Africa	Yes	Competition with native species	I3N
Eucalyptus viminalis	Unkown	Unkown	Unkown	Unkown	Unkown	
Felis catus	Middle East	Yes	Australia, New Zealand, Ecuador, Bahamas, Mexico, United States, Spain, France (Seychelles Islands), South Africa	Yes	Extinction of bird species, vectors of diseases, high cost of handling	I3N
Hedychium coronarium	Unkown	Yes	United States of America (Hawaii), Samoa, Swaziland, Cook Islands, Ecuador (Federated States of Micronesia), Fiji, French Polynesia, Japan, New Caledonia, Palau, Tonga, Australia	Yes	Obstructs pipes in hydroelectric dams	13N
Hovenia dulcis	Asia	No	Tanzania, Paraguay and Argentina	No	Competition with native species	I3N
Impatiens walleriana	African	Yes	Ecuador (Galapagos Archipelago), United States (Hawaii), New Caledonia, Australia, France (Reunion Islands)	Yes	It dominates the lower strata of shaded areas, especially humid environments, displacing native understory plants in the case of forest environments, compromising ecological succession.	I3N
Lantana camara	Central America.	Yes	American Samoa, Australia, Bahamas, Barbados,	Yes	Succession modification	GISD

	South America		Bermuda, British Indian Ocean Territory, Burundi, Cambodia, China, Cook Islands, Dominican Republic, Ecuador, Fiji, French Polynesia, Ghana, Gibraltar, Guam, Haiti, Hong Kong, India, Indonesia, Kenya, Kiribati, Liberia, Madagascar, Malaysia, Marshall Islands, Mauritius, Mayotte, Micronesia, Federated States Of Nauru, New Caledonia, New Zealand, Niue, Norfolk Island, Northern Mariana Islands, Palau, Papua New Guinea, Philippines, Pitcairn, Reunion, Rwanda, Saint Helena, Samoa, Solomon Islands, South Africa, Sri Lanka, Swaziland, Thailand, Tonga, Turkey, Turks And Caicos Islands, Tuvalu, Uganda, United Arab Emirates, United States, United States Minor Outlying Islands, Vanuatu, Wallis And Futuna, Zambia, Zimbabwe			
Lepidaploa remotiflora	Unkown	Unkown	Unkown	Unkown	Unkown	
Lepidium ruderale	North America	Unkown	Unkown	Unkown	Unkown	Available at: https://www.invasivespeciesinfo.gov/plants/databases.shtml#invpl
Leucaena leucocephala	Central America and Mexico	Yes	Barbados, Benin, Bermuda, Bolivia, Botswana, Brunei Darussalam, Burkina Faso, Burundi, Cambodia, Cameroon, Cape Verde, Cayman Islands, Angola, Anguilla, Antigua and Barbuda, Central African Republic, Chad, China, Cocos (Keeling) Islands, Colombia, Congo, Cook Islands, Costa Rica, Cote d`Ivoire, Cuba, Djibouti, Dominican Republic, East Timor, Ecuador, Egypt, El Salvador, Equatorial Guinea, Eritrea Guinea-Bissau, Guyana, Haiti, Honduras, India, Indonesia, Jamaica, Kenya, the former Yugoslav Republic of Macedonia, Ethiopia, Federated States of Micronesia, Fiji, French Guiana, Gabon, Gambia, Ghana, Kiribati, Lao People's Democratic Republic, Liberia, Madagascar, Malawi, Mali, Mali, Martinique, Mauritius, Montserrat, Morocco,	Yes	It forms dense clusters, dominating the environment and preventing the establishment of native plants	I3N

			Mozambique, Myanmar (Burma), Nepal, Finnish Antilles, New Calcedonia, Niger, Nigeria, Pakistan, Panama, Papua New Guinea New Guinea, Paraguay, Peru, Filipin Saint Kitts and Nevis, Saint Vincent and the Grenadines, Samoa, Senegal, Seychelles, Sierra Leone, Singapore, Solomon Islands, Somalia, South Africa, Sri Lanka, Sudan Suriname, Swaziland, Tanzania, Thailand, Togo, Tonga, Trinidad and Tobago, Tunisia, Turks and Caicos Islands, Uganda, United States, Uruguay, Vanuatu, Venezuela, Vietnam, Virgin Islands, Zaire, Zambia and Zimbabwe			
Ligustrum lucidum	China and Korea	Yes	United States (Florida, Texas and North Carolina, Hawaii), New Zealand, Australia, South Africa, and Argentina	Yes	Competition with native species, toxic fruits	I3N
Lithobates catesbeianus	North America	Yes	United States of America (Western Region and Hawaii), Canada (Southwest), Mexico, Venezuela, Guatemala, Philippines, Japan Belgium, Cuba, France, Indonesia, Italy, Japan, Korea, Malaya, Netherlands, Spain, Puerto Rico, Singapore, Thailand and Taiwan	Yes	Transmitter of chytridiomycosis, disease caused by the fungus Batrachochytrium dendrobatidis	13N
Lutzomyia longipalpis	Unkown	Unkown	Unkown	Unkown	Unkown	Unkown
Mangifera indica	India	Yes	Mexico, Australia, China, the United States (Hawaii), Ecuador (Galapagos Islands), Fiji, French Polynesia, Guam, Japan, Nauru, New Caledonia, Niue, France (Réunion Islands), Mauritius, Tonga and Pakistan.	Yes	Alteration of the pH of the water due to the rotting of the leaves and fruits in large quantity, impact on the dispersion of native zoocoric species	I3N
Melia azedarach	East Asia (Japan, India, Burma, China, Persia)	Yes	South Africa, United States (Hawaii, Florida, Texas, Mariana Islands), Australia, Chile, New Zealand (Cook Islands, Niue), Ecuador (Galapagos Islands), Federated States of Micronesia, Fiji, French Polynesia, New Caledonia, Palau	Yes	Competition with native species, toxic fruits	I3N
Melinis repens	South Africa	Yes	United States (Hawaii, Florida, Guam Island), Cook Islands, Fiji, France (French Polynesia), Kiribati, Marshall Islands, Nauru, New Caledonia, Solomon Islands and Australia	Yes	Competition, reduction of natural biodiversity	13N
Mimosa quadrivalvis	Caribbean Islands and American Continent	Unkown	Unkown	Unkown	Unkown	Available at: https://www.itis.gov/s

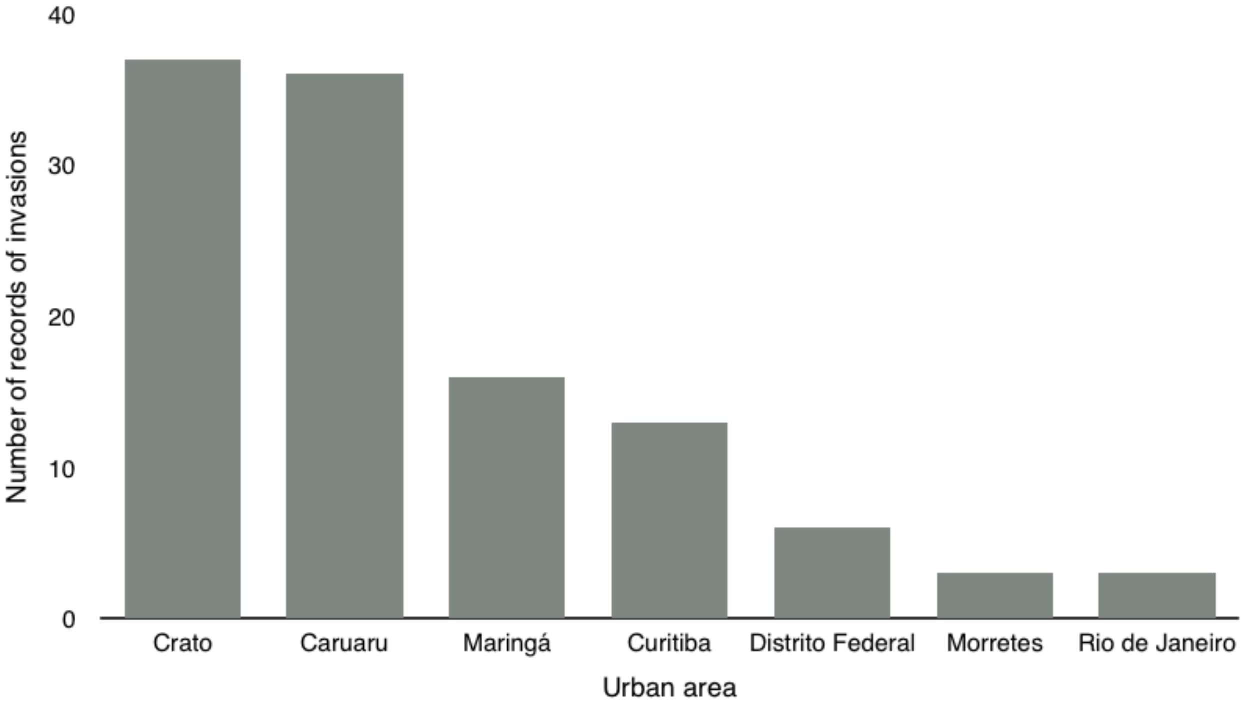
						ervlet/SingleRpt]
Mimosa sensitiva	Unkown	Unkown	Unkown	Unkown	Unkown	Unkown
Mimosa sp.	Australian Northern Territory, Central America, South America		Hong Kong, Indonesia, Java, Sumatra, Malaysia, Peninsular Malaysia, Myanmar, Singapore, Sri Lanka, Thailand, Vietnam, Kenya, Rwanda, South Africa, Tanzania, Zambia, Florida, Cuba, Jamaica, Puerto Rico, Saint Lucia, Galapagos Islands, Australia, Queensland, Papua New Guinea, Cambodia,Christmas Island (Indian Ocean), East Timor, Malaysia, Taiwan, Guinea, Nigeria, Comoros, Gambia, Ghana, Madagascar, Zimbabwe, Antigua and Barbuda, Cuba, Jamaica, Trinidad, Tobago	Unkown	Accumulation of sediments affecting irrigation, invasion of plantations, alteration of swamp floodplains, monopolization of resources, high risk of fire, avoidance of species regeneration, presence of thorns. Pest of crops and pastures	Available at: Cabi.org
Mimosa ursina	Unkown	Unkown	Unkown	Unkown	Unkown	
Morus nigra	Asia	No	United States and Caribbean Islands	No	Displacement of native species	I3N
Panicum maximum	Unkown	Unkown	Unkown	Unkown	Unkown	
Pappophorum pappiferum	Argentina, Puerto Rico	Yes	Unkown	Unkown	Unkown	Available at: http://www.floraargentina.edu.ar // https://biotaxa.org // http://regionalconservation.org
Passiflora cincinnata	Unkown	Unkown	Unkown	Unkown	Unkown	
Pavonia cancellata	Unkown	Unkown	Unkown	Unkown	Unkown	
Pinus elliottii	United States	Yes	South Africa, Australia, Argentina, Uruguay, Paraguay		Replacement of native vegetation by dominance and shadowing of open ecosystems and degraded forest areas, increase the acidity of the soil, alteration of water regime in open ecosystems, where it replaces small vegetation, deposition of litter of slow decomposition hinders the germination of native species, reduction in amphibian richness in areas invaded in Rio Grande do Sul (Machado et al 2012)	I3N
Pinus sp.	Northern Hemisphere, North America, Europe, Asia	No	Canada, Argentina, Uruguay, Chile, Australia, New Caledonia, New Zealand, Madagascar, Malawi, South Africa	Yes	Competition / predation, economic losses, inhibits the growth of other species, reduction of natural biodiversity, human health (allergies)	I3N
Pinus sp.	North America	Yes	Canada, Argentina, Uruguay,	Yes	Replacement of native	13N

			Chile, Australia, New Caledonia, New Zealand, Madagascar, Malawi, South Africa		vegetation, ejected by shading, reduction of water availability	
Pittosporum undulatum	Oceania	Yes	Australia, New Zealand, Mauritius, Bermuda, Jamaica, United States, Cuba, Bolivia, Mexico, Chile, Colombia, India, Israel, France, Portugal, Spain, United Kingdom, China, South Africa	No	Formation and clusters hinder the growth of other species, allelopathy, loss of diversity	I3N
Poecilia reticulata	Venezuela, French Guiana, Guyana, Suriname and northern Brazil (States of Pará And Amapá)	Yes	Africa: Durban (rivers in the south), Namibia (Kuruman and Otijkoto lakes) Japan Australia (Queensland and north)	Unkown	Competition, predation of native species, reduction of natural biodiversity	I3N
Portulaca oleracea	Unkown	Unkown	Dominican Republic	Unkown	Aggression to agricultural crops, reservoir for other pests	Available at: http://www.cabi.org
Psidium guajava	South America	Yes	Argentina, Peru, Ecuador, Germany, Spain, New Zealand, Australia, Papua New Guinea, Reunion Island, Philippines, Indonesia, Malaysia, China, Japan, Nepal	Yes	Loss and abandonment of lands due to control difficulties (invasion with Tecoma stans in northern Paraná, Brazil)	13N
Rhaphiodon echinus	Unkown	Unkown	Unkown	Unkown	Unkown	
Rhodnius neglectus	Unkown	Unkown	Unkown	Unkown	Unkown	
Ricinus communis	Afghanistan, Algeria, Angola, Benin, Botswana, Egypt, Ethiopia, Hungary, Iran, Islamic Republic Of Israel, Jordan, Kenya, Lesotho, Morocco, Morocco, Pakistan, South Africa, Swaziland, Syrian Arab Republic, Turkey	Yes	Argentina, Chile, Colombia, Paraguay, Peru, Uruguay, Guatemala, Costa Rica, Cuba, Ecuador, El Salvador, Jamaica, French Guiana, Panama, Mexico, Martinique, Puerto Rico, Saint Barthélemy, United States, Anguilla, Bahamas, Guadeloupe, Bermuda, Australia, New Zealand, Cayman Islands, Cook Islands, Federated States of Micronesia, Fiji, French Polynesia, Indonesia, Kiribati, Madagascar, Marshall Islands, Nauru, Niue, Norfolk Islands, Northern Mariana Islands, Palau, Netherlands Antilles, New Caledonia, Pitcairn Islands, Reunion Island, Saint Helena, Samoa, Solomon Islands, Tonga Islands, Vanuatu, Wallis and Futuna, Taiwan	Yes	Loss of biodiversity, loss of agricultural areas and pastures, toxic leaves	I3N;GISD
Schizolobium parahyba	South America	Yes	Unkown	Unkown	Establishment in degraded or regenerating remnants	13N
Senegalia langsdorffii	Unkown	Unkown	Unkown	Unkown	Unkown	
Senna obtusifolia	Unkown	Unkown	Unkown	Unkown	Unkown	
Sertania glabrata	Unkown	Unkown	Unkown	Unkown	Unkown	

Setaria parviflora	Unkown	Unkown	Unkown	Unkown	Unkown	
Setaria sp. 1	Unkown	Unkown	Unkown	Unkown	Unkown	
Setaria sp. 2	Unkown	Unkown	Unkown	Unkown	Unkown	
Sida sp.	American continent	Unkown	Chagos Archipelago, China, Christmas Island (Indian Ocean), Cocos Islands, Indonesia, Java, Nusa Tenggara, Japan, Malaysia, Nepal, Singapore, Taiwan, Ghana, Kenya, Mayotte, Mozambique, Nigeria, Seychelles, South Africa, Canary Islands, Uganda, Oceania	Unkown	Damage to pastures and crops, intoxication in cattle, formation of dense populations	Available at: http://www.cabi.org
Sida spinosa	Unkown	Unkown	Unkown	Unkown	Unkown	Unkown
Sida urens	Caribbean, Oceania, Brazil	Unkown	Unkown	Unkown	Unkown	Available at: http://floradobrasil.jbrj.gov.br/reflora/listaBrasil/ConsultaPublicaUC
Solanum americanum	Unkown	Unkown	Unkown	Unkown	Unkown	
Solanum baturitense	Unkown	Yes	Unkown	Unkown	Unkown	Available at: http://memoria.bn.br
Solanum grandiflorum	Unkown	Unkown	Unkown	Unkown	Unkown	
Spermacoce capitata	Unkown	Unkown	Mexico, Bolivia, Argentina, Caribbean	Yes	Unkown	Available at: https://sites.google.com/site/florasbs/r/agriaozinho-tapete
Spermacoce verticillata	South America	Unkown	Niue, Saint Helena, United States	Unkown	Unkown	GISD
Stylosanthes guianensis	Central and South America	Yes	Southern Cook Islands, Fiji, French Polynesia, Austral (Tubuai) Islands,Hawaii ,New Caledonia ,Niue, Palau,Wallis and Futuna ,Australia ,China,Taiwan.	Unkown	Shading of seedlings, modification of natural landscape	Available at: http://www.hear.org // http://www.scielo.br
Stylosanthes scabra	Unkown	Unkown	Unkown	Unkown	Unkown	
Syzygium cumini	Ásia	Yes	África do Sul, Nova Zelândia (ilhas Cook), ilhas Fiji, Polinésia Francesa, Estados Unidos (Guam, Havaí, Florida), França (Nova Caledônia), Niue, Palau, Tonga, China, Indonésia, Malásia, Austrália.	No	Incômodo humano, dificulta o processo de regeneração e, consequentemente, interfere na sucessão vegetal.	I3N, GISD
Tecoma stans	Unkown	Unkown	Bermuda, Cayman Islands, French Polynesia, Mayotte, New Caledonia, Reunion,	Unkown	Competition with native species	GISD

			Saint Helena, Saint Lucia			
Tridax procumbens	Central America	Unkown	Unkown	Unkown	Host of harvest pests	Available at: http://www.cabi.org
Turnera subulata	Unkown	Unkown	Unkown	Unkown	Unkown	
Waltheria rotundifolia	Unkown	Unkown	Unkown	Unkown	Elevado crescimento e elevada produção de flores.	Available at: https://ainfo.cnptia.embrapa.br/digital/bitstream/item/69223/1/Ferreira.pdf
Zaprionus indianus	Unkown	Unkown	Unkown	Unkown	Unkown	

Figure 1


40
30
20
10
0
Number of records of invasions
Crato
Caruaru
Maringá
Curitiba
Distrito Federal
Morretes
Rio de Janeiro
Urban area

Appendix

Table 2

Reference	Brazil's State	City	City's area (km^2)	Number of inhabitants	Invasive flora	Invasive Fauna	Total of invasive species
Both et al. (2011)	Rio Grande do Sul	Agudo	536.117	16729	0	1	1
Ohlweiller et al. (2010)	São Paulo	Americana	133.930	226970	0	1	1
Lessa & Bergallo (2012)	Rio de Janeiro	Angra dos Reis	800.430	181486	0	1	1
Ohlweiller et al. (2010)	São Paulo	Aparecida	121.076	36217	0	1	1
Area & State (2014)	São Paulo	Araçatuba	1167.402	192757	1	0	1
Ohlweiller et al. (2010)	São Paulo	Arapeí	155.707	2493	0	1	1
Ohlweiller et al. (2010)	São Paulo	Areias	306.566	3693	0	1	1
Ohlweiller et al. (2010)	São Paulo	Atibaia	478.101	141654	0	1	1
Ohlweiller et al. (2010)	São Paulo	Bananal	616.320	10728	0	1	1
Both et al. (2011); Both & Grant (2012)	Santa Catarina	Blumenau	519.837	334002	0	1	1
Ohlweiller et al. (2010)	São Paulo	Bom Jesus dos Perdões	108.513	19703	0	1	1
Ohlweiller et al. (2010)	São Paulo	Botucatu	1482.874	137899	0	1	1
Ohlweiller et al. (2010)	São Paulo	Bragança Paulista	513.589	158856	0	1	1
Ferreira e al. (2004)	Distrito Federal	Brasília	5801.937	2914830	0	2	2
Ohlweiller et al. (2010)	São Paulo	Caçapava	369.907	84844	0	1	1
Ohlweiller et al. (2010)	São Paulo	Cachoeira Paulista	287.837	30099	0	1	1
Ohlweiller et al. (2010)	São Paulo	Campinas	794.433	238,3	0	1	1
Brazil (2013)	Mato Grosso do Sul	Campo Grande	8096.051	853622	0	1	1
Ohlweiller et al. (2010)	São Paulo	Caraguatatuba	485.097	111524	0	1	1
dos Santos et al. (2014)	Pernambuco	Caruaru	920.611	351686	36	0	36
Both et al. (2011)	Santa Catarina	Chapecó	626.060	205795	0	1	1

Reference	Brazil's State	City	City's area (km^2)	Number of inhabitants	Invasive flora	Invasive Fauna	Total of invasive species
dos Santos et al. (2014)	Ceará	Crato	1009.202	128680	37	0	37
Hochmüller et al. (2010)	Rio Grande do Sul	Cruz Alta	1360.37	63946	0	2	2
Ohlweiller et al. (2010)	São Paulo	Cruzeiro	304.572	77000	0	1	1
Ohlweiller et al. (2010)	São Paulo	Cunha	1407.25	22086	0	1	1
Bionde & Muller (2013)	Paraná	Curitiba	435.036	1879355	13	0	13
Maeda et al. (2012)	Distrito Federal	Distrito Federal	5787.784	2563963	0	6	6
Moro et al. (2013)	Ceará	Fortaleza	314.93	2591188	1	0	1
Fisher et al. (2005)	Paraná	Guaraqueçaba	2018.906	7988	0	1	1
Ohlweiller et al. (2010)	São Paulo	Guarujá	142.589	311230	0	1	1
Both et al. (2011)	Santa Catarina	Guatambu	204.757	4674	0	1	1
Ohlweiller et al. (2010)	São Paulo	Igaratá	293.322	8825	0	1	1
Ohlweiller et al. (2010)	São Paulo	Ilhabela	347.5	32197	0	1	1
Both et al. (2011)	Santa Catarina	Indaial	430.534	61968	0	1	1
Ohlweiller et al. (2010)	São Paulo	Iporanga	1160.293	4351	0	1	1
Ohlweiller et al. (2010)	São Paulo	Itanhaém	599.017	94977	0	1	1
Ohlweiller et al. (2010)	São Paulo	Itariri	272.277	16602	0	1	1
Both et al. (2011)	Rio Grande do Sul	Ivorá	122.887	2156	0	1	1
Both et tal. (2014)	Rio Grande do Sul	Ivorá	122.887	2156	0	1	1
Ohlweiller et al. (2010)	São Paulo	Jacareí	464.272	226539	0	1	1
Braks et al. (2004)	Rio de Janeiro	Jacarepaguá	75.79	157326	0	2	2
Ohlweiller et al. (2010)	São Paulo	Jacupiranga	706.382	17196	0	1	1
Ohlweiller et al. (2010)	São Paulo	Jambeiro	183.758	5350	0	1	1
Ohlweiller et al. (2010)	São Paulo	Jarinu	207.671	23827	0	1	1
Both et al. (2011)	Santa Catarina	Joinville	1126.106	562151	0	1	1
Both et tal. (2014)	Santa Catarina	Joinville	1126.106	562151	0	1	1
Ohlweiller et al. (2010)	São Paulo	Jundiaí	431.207	401896	0	1	1

Reference	Brazil's State	City	City's area (km^2)	Number of inhabitants	Invasive flora	Invasive Fauna	Total of invasive species
Ohlweiller et al. (2010)	São Paulo	Lagoinha	255.924	4960	0	1	1
Ohlweiller et al. (2010)	São Paulo	Lavrinhas	166.860	6586	0	1	1
Ohlweiller et al. (2010)	São Paulo	Lençóis Paulista	803.860	70331	0	1	1
Ohlweiller et al. (2010)	São Paulo	Limeira	580.963	294128	0	1	1
Ohlweiller et al. (2010)	São Paulo	Lorena	413.776	86764	0	1	1
Ohlweiller et al. (2010)	São Paulo	Mairiporã	320.697	93981	0	1	1
Marsden et al. (1983)	Goiás	Mambaí	859.555	18000	0	2	2
Ríos-Velásquez et al. (2007)	Amazonas	Manaus	11401.092	2057711	0	1	1
Blum et al. (2008)	Paraná	Maringá	487.930	397437	16	0	16
Ohlweiller et al. (2010)	São Paulo	Mongaguá	143.171	51580	0	1	1
Ohlweiller et al. (2010)	São Paulo	Monteiro Lobato	332.740	4123	0	1	1
Fisher et al. (2006)	Paraná	Morretes	684.58	15718	2	1	3
Ohlweiller et al. (2010)	São Paulo	Natividade da Serra	832.606	6678	0	1	1
Ohlweiller et al. (2010)	São Paulo	Nazaré Paulista	326.542	16413	0	1	1
Both et al. (2011)	Santa Catarina	Nova Erechim	644.00	4275	0	1	1
Both et al. (2011)	Rio Grande do Sul	Nova Palma	313.506	6345	0	1	1
Ohlweiller et al. (2010)	São Paulo	Panorama	353.137	14603	0	1	1
Ohlweiller et al. (2010)	São Paulo	Paraibuna	809.794	17384	0	1	1
Ohlweiller et al. (2010)	São Paulo	Paulicéia	373.891	6342	0	1	1
Ohlweiller et al. (2010)	São Paulo	Paulínia	139.332	95221	0	1	1
Ohlweiller et al. (2010)	São Paulo	Pedreiras	109.710	45052	0	1	1
Ohlweiller et al. (2010)	São Paulo	Peruíbe	326.214	64531	0	1	1
Ohlweiller et al. (2010)	São Paulo	Pindamonhangaba	729.9	160614	0	1	1
Both et al. (2011)	Santa Catarina	Pinhalzinho	128.298	18284	0	1	1
Ohlweiller et al. (2010)	São Paulo	Piracaia	384.729	25139	0	1	1

Reference	Brazil's State	City	City's area (km^2)	Number of inhabitants	Invasive flora	Invasive Fauna	Total of invasive species
Ohlweiller et al. (2010)	São Paulo	Piracicaba	1378.069	391449	0	1	1
da Silva et al. (2005); Castro & Vera (2001); Garcia et al. (2008)	Rio Grande do Sul	Porto Alegre	496.682	1476867	0	2	2
Ohlweiller et al. (2010)	São Paulo	Potim	44.651	21984	0	1	1
Ohlweiller et al. (2010)	São Paulo	Praia Grande	147.065	293695	0	1	1
Ohlweiller et al. (2010)	São Paulo	Presidente Prudente	562.794	220599	0	1	1
Ohlweiller et al. (2010)	São Paulo	Queluz	249.826	11309	0	1	1
Ohlweiller et al. (2010)	São Paulo	Redenção da Serra	309.111	3879	0	1	1
Ohlweiller et al. (2010)	São Paulo	Registro	722.411	56203	0	1	1
Ohlweiller et al. (2010)	São Paulo	Ribeirão Preto	650.916	666323	0	1	1
Ohlweiller et al. (2010)	São Paulo	Rio Claro	496.422	28,35	0	1	1
Lourenço-de-Oliveira et al. (2004); Rangel & Neiva (2013); Honório et al. (2009)	Rio de Janeiro	Rio de Janeiro	1200.3	6453682	0	3	3
Ohlweiller et al. (2010)	São Paulo	Roseira	130.190	9606	0	1	1
Ohlweiller et al. (2010)	São Paulo	Santa Branca	275.004	13770	0	1	1
Ohlweiller et al. (2010)	São Paulo	Santo André	174.840	707613	0	1	1
Ohlweiller et al. (2010)	São Paulo	Santo Antônio do Pinhal	132.886	6516	0	1	1
Ohlweiller et al. (2010)	São Paulo	Santos	280.674	433965	0	1	1
Ohlweiller et al. (2010)	São Paulo	São Bento do Sapucaí	252.200	10462	0	1	1
Ohlweiller et al. (2010)	São Paulo	São José do Barreiro	570.629	4097	0	1	1
Ohlweiller et al. (2010)	São Paulo	São José dos Campos	1099.77	688597	0	1	1
Ohlweiller et al. (2010)	São Paulo	São Luiz do Paraitinga	617.148	10397	0	1	1

Reference	Brazil's State	City	City's area (km^2)	Number of inhabitants	Invasive flora	Invasive Fauna	Total of invasive species
Christianini (2006); Dislich et al. (2001); Mengardo & Pivello (2014); Dislich et al. (2002); Ohlweiller et al. (2010)	São Paulo	São Paulo	1522.986	11967825	1	1	2
Ohlweiller et al. (2010)	São Paulo	São Sebastião	399.679	81716	0	1	1
Ohlweiller et al. (2010)	São Paulo	São Vicente	148.424	353040	0	1	1
Ohlweiller et al. (2010)	São Paulo	Silveiras	414.698	5792	0	1	1
Ohlweiller et al. (2010)	São Paulo	Sorocaba	450.382	644919	0	1	1
Ohlweiller et al. (2010)	São Paulo	Sumaré	153.033	236358	0	1	1
Ohlweiller et al. (2010)	São Paulo	Taubaté	625.003	302331	0	1	1
Ohlweiller et al. (2010)	São Paulo	Tietê	392.509	40154	0	1	1
Ohlweiller et al. (2010)	São Paulo	Tremembé	192.416	40985	0	1	1
Ohlweiller et al. (2010)	São Paulo	Ubatuba	723.829	85399	0	1	1
Ohlweiller et al. (2010)	São Paulo	Votuporanga	424.1	91278	0	1	1

Effects of pine invasion in native community structure in an area of Cerrado

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Abstract

Plants have been transported around the globe for many years, and for numerous reasons. Once introduced to a new area, they may stablish, spread, and invade. Invasive plants have the potential to modify native community structure and function. Cerrado is a Brazilian biome considered a hotspot for biodiversity. However, much of Cerrado has already been deforested, and only 7.5% of its area is protected. Invasive non-native plants are among the many threats to Cerrado's biodiversity. Among those invasive species are several *Pinus* species. Our work aimed at understanding ways in which *Pinus* invasion may be affecting plant diversity patterns in the Cerrado. The research was conducted at the Brasília Botanical Garden, a nature reserve that have record of two invasive pines, *Pinus caribaea* and *Pinus oocarpa*. We measured basal area and identified all tree species higher than 1.3 m in 30 plots (200 m^2) separated by at least 50 m from each other. We used t-tests to check if the of presence of pine trees affected native tree richness, diversity (Shannon Diversity Index – H), Evenness (E), Probability of Interspecific Encounters (PIE), density, and basal area of native species and generalized linear models with Gamma family distribution and log link function (GLMs) to test how pine basal area and density affected the same variables of the native community. For differences in species composition we used non-metric multidimensional scaling (NMDS) in all plots were with invasive pines. We found a negative relationship between invaded plots and native tree density. Our results show that pines have impacts on native community, and that pines should be managed in order to conserve protected areas.
Keywords: biological invasions, *Pinus sp.*, woodland Cerrado, exotic pines, diversity patterns, native trees

Introduction

Plants have been transported around the globe for many years for reasons including forestry, and horticulture (Zenni 2014). Once introduced to areas outside their natural occurrences, they may spread and invade locations beyond their introduction sites (Richardson et al 2008). Invasive plant species have the potential to modify native community dynamics by altering species interactions, nutrient, and water cycling (Levine et al. 2014). Given sufficient propagule pressure by some exotic species, few communities are likely to remain clear of plant invasions (Levine et al. 2003, Simberloff 2009, Zenni and Simberloff 2013). Invasive plants commonly physically transform the structure of communities with few trees, such as prairies, marshes and savannas (Simberloff et al 2010).

The Cerrado is the second largest biome in Brazil, occupying 21% of the country's territory, and is recognized as one of the global conservation hotspots (Borlaug 2002, Silva and Bates 2002). The biome holds around 4800 endemic plant and animal species (Strassburg et al. 2017). However, around 2 million km^2 have already been deforested for human use, mostly pasture and agriculture (Borlaug 2002, Klink and Machado 2005). The spread of exotic species, like African grasses, is one of the major threats to Cerrado's biodiversity. Many exotic grasses were planted for pasture, such as *Brachiaria brizantha* and *Andropogon gayanus* (Klink and Machado 2005). These species threaten ecosystem functioning by increasing flammable biomass, consequently altering fire regime (Gorgons-Barbosa et al. 2014). Although we have many records of invasive grasses, they are not the only threats to Cerrado's ecosystems.

Diversity helps to maintain ecosystems services through high levels of productivity, resilience and the capacity to recover from environmental change (Pimentel et al. 2001, Speziale and Lambertucci 2010). The conservation of hotspot's diversity is an important mean of curbing the danger that threatens the environment. Nature preserves are an important tool for biodiversity conservation, and the maintenance of ecosystems functions (Dobson et al. 1997). Despite being a conservation hotspot, only 7.5% of the total extent of the Cerrado is protected in reserves (Strassburg et al. 2017). Protecting the existing nature reserves is one way of conserving Cerrado's diversity and ecosystem services. However, numerous non-native plant species have been spotted invading Brazilian protected areas (Ziller e Dechoum 2013, Sampaio e Schmidt 2013), and their presence can threaten native communities, for instance, through competition and production of allelochemicals (Vilà and Weiner 2004, Lankau et al. 2009).

Pine trees have been planted worldwide for various reasons, mostly commercial and for experimental forestry. These species have spread and are acknowledged as one of the major threats to native biodiversity (Lowe et al. 2000), recently becoming a problem in the Southern Hemisphere (Richardson and Rejmánek 2004, Higgings and Richardson 1998). In Brazil, pines have been present since the end of the nineteenth century (Zenni and Simberloff 2013; Shimizu 2006; Richardson et al. 2008). There are records of pines invading in many regions (Nuñez and Medley 2011, Buckley et al. 2005, Simberloff et al. 2010), including Brazil (Zenni e Simberloff 2013, Stevens e Beckage 2009, Falleiros et al. 2011). Although we have numerous records of invasions by these plants in Brazil, the impacts of pines invasions on native diversity are yet to be widely quantified in Brazil. Among the impacts documented around the globe by *Pinus*, there are changes in local hydrology, soil and nutrient availability, as well as impacts in plant and animal community (Simberloff et al 2010). *Pinus* species have been found in many protected areas in Brazil (Ziller and Dechoum 2013, Sampaio and Schmidt 2013). Our work aimed at understanding how *Pinus* invasions can affect plant communities in the Cerrado. Understanding the impacts of biological invasion can provide important insights for Applied Ecology. Owing to the various impacts that the presence of pine trees can have on native diversity (Abreu and Durigan 2011, Falleiros et al. 2011), we hypothesized that the presence, abundance, and the extent of pine trees negatively impact native patterns of plant diversity, density, and community composition.

Methods

Study site

The Botanical Garden of Brasília (JBB), created in 1985, has a total area of approximately 5,000 ha. 10% of the area of JBB is used for recreation and the 90% remaining is a designated protected area (Estação Ecológica do Jardim Botânico de Brasília). Unlike most existing botanical gardens, JBB is mostly covered by native ecosystems, being more similar to a national park than a tradition botanical garden. The native vegetation of the reserve is Cerrado (Neotropical Savannah), and the climate consists of two well-defined seasons, dry (May - September) and rainy (October - April), with annual precipitation varying from 600 mm to 2,000 mm (Lima and Silva 2008).

In 1976, prior to the JBB, and as part of a forestry experimentation program lead by the Brazilian Institute of Sustainable Development (IBDF) and the Brazilian Company of Agricultural Research (Embrapa), 15 Pine species were planted in a 10 ha stand that now sits in the center of the JBB. Out of the 15 species originally planted, three species (*Pinus caribaea*, *Pinus oocarpa*, and *Pinus patula*) remain, and two species (*P. caribaea* and *P. oocarpa*) are now invading (Braga and Zenni 2014).

Sampling design

We used line transects starting at a minimum of 50 m from the border of the *Pinus* stand and extending for 200 m to establish thirty circular plots (200 m^2) at the vicinity of the pine plantation. All transects and plots were separated by at least fifty meters from each other. At each plot, we counted and measured the trunk circumference at ground level of all trees taller than 1.3 meters and identified them at the species level. We used trunk circumference to calculate basal area, which is the sum of the individual cross-section area of all trees from a species in each plot. Basal area provides a good estimate of the degree to which an area is occupied by each tree species. Sampling occurred between January and May of 2016 (rainy season).

Statistical analysis

First, we separated plots with and without invading pines and used t-tests to check if the presence of pine trees affected native tree richness, diversity (Shannon Diversity Index – H), Evenness (E), Probability of Interspecific Encounters (PIE), density, and basal area of native species. Second, we used generalized linear models with Gamma family distribution and log link function (GLMs) to test how pine basal area and density affected the same variables of the native community. For the GLMs we used only data from plots where pines were present (n=19). We used non-metric multidimensional scaling (NMDS) to compare species composition in invaded and non-invaded sites.

Results

The Cerrado physiognomy of our study site was characterized as woodland Cerrado, consisting mostly of trees and shrubs with few herbaceous species (personal observation). We found 118 species, belonging to 49 botanical families (Table 1). Native species richness varied from 24 to 49 species per plot (Fig. 1). Native species density varied between 141 and 280 individuals per plot. The maximum basal diameter of an individual tree reached 59 cm, whereas the minimum was 1 cm (Fig. 2). We found a total 72 pines with height equal/greater than 1.3 m, distributed in 19 out of the 30 plots sampled. Pine maximum density was 10 plants per plot and maximum basal diameter reached 73 cm^2 (Fig. 3). The furthest invading pine stood approximately 400 meters away from the edge of the plantation. *Pinus oocarpa* and *Pinus caribaea* were the two invasive species found on the site. However, of all the pine tree sampled (n=72), only one was identified as *P. oocarpa*.

Native tree density was higher in the absence of *Pinus* (t=2.1; p<0.05; Fig. 4), but presence of pine plants did not affect other native community variables (Table 2). Native species density was negatively affected by the density of pine plants ($t=2.7$; $p<0.02$; Fig. 5), as well as by Pinus basal area ($t=2.3$; $p<0.03$; Fig. 6). Shannon Diversity Index (H), Evenness (E), Probability of Interspecific Encounters (PIE) were not affected by pine density or pine basal diameter (Table 3, Table 4). NMDS analysis also showed no difference in native species composition for invaded and non-invaded plots (Fig. 7).

Discussion

Pine species are among the most invasive plants across the Southern Hemisphere, presenting attributes associated with invasion success, such as large propagule pressure, numerous and small seed production, and wind seed dispersion (Rejmánek and Richardson 1996, Essl et al. 2010, Lowe et al. 2000, Richardson 2006). Impacts of pine invasion, such as changes in species composition, diversity and density, have been recorded in many places (Abreu and Durigan 2011; Falleiros et al. 2011, Zenni and Simberloff 2013). Our hypothesis that presence and abundance of pines negatively affects native density was supported by our results (Fig. 1). But the hypothesis that the invasion by *Pinus* changes native diversity and species composition in the Cerrado was refuted, as the presence and density of pines showed no significant relation with community diversity variables.

Our results showed no significant relation between pines and native species community metrics. Similarities among species could result in strong competition, possibly causing local competitive exclusion (Webb et al. 2002). Competition with native plants, and the possible release of allelochemicals by *P. caribaea* (Nissanka et al 2005), the predominant pine species found on site, could be explanations for these results and for the lower densities found on plots with higher densities of *Pinus* (Fig. 1, Fig. 2). The accumulation of pine needles in the soil can also be a factor that alters native seedling recruitment, endangering future ecosystem dynamics. As the pine invasion in JBB is still in the early stages, these patterns may change with time.

Our results are in conformity with Abreu and Durigan 2011, that found lower native density in pine invaded plots, but they also higher richness and diversity of native woody species in invaded sites in an area of woodland Cerrado. Zenni and Simberloff 2013 found lower diversity in areas invaded by *Pinus taeda*, *Pinus glabra* and *Pinus eliiottii* in southern Brazil. Ecosystem stability depends on community densities (McCann 2000), therefore, changes in native density can threaten local ecosystem function. High native diversity has already been associated with resistance to biological invasions (Elton 1958, Fargione and Tilman 2005, Ricotta et al. 2010), but as we found no difference in native richness and diversity indexes, it could probably mean that diversity is not playing a resistance part against pine invasion in this area.

Presence of invasive species have been shown to alter native species composition in natural areas (Richardson 1998). Abreu and Durigan 2011 found that species composition showed a tendency for difference in pine invaded and non-invaded plots in woodland Cerrado. For grassland savannah, species composition differed significantly between non-invaded sites and sites invaded by *Pinus elliottii*. In that case, invaded plots consisted of trees and shrubs and non-invaded consisted mostly of shade-intolerant species (Abreu et al. 2013). These results showed that shade-intolerant species were excluded from pine invaded sites, probably because of the larger canopy pines have. We found, for woodland Cerrado, no difference in species composition, which can probably be explained by the presence of many shade-tolerant species on our plots (Pinheiro and Durigan 2012).

P. oocarpa, one of the two pine species found on site, have been reported for JBB to spread in a rate of 12.72 m per year since its introduction in 1976, spreading faster than any other invasive pines recorded in Brazil (Braga et al 2014, Zenni and Simberloff 2013). Those results, combined with the data shown here, provide enough information to draw attention from decision makers on the potential impacts the presence of pines can have on the Cerrado community of the JBB. Although still in the early stages of invasion (Braga et al 2014), pine invasions at JBB already have detectable negative impact on native plant communities. Taken together, these results suggest pines could and should be eradicated from JBB in order to prevent further ecological impacts on native Cerrado. Managing pines can be relatively simple, when compared to other plant invaders, as pines take years before producing viable seeds (7 to 15 years) and the seedlings are easily mechanic controlled (Nuñez et al 2017, Falleiros et al. 2011). The difficulty of controlling pines in South America may be the great extent of area already occupied by pines, and the economic and social value associated with them, as observed for other countries (McConnachie et al. 2015, Dickie et al. 2014, Woodford et al. 2016). Mechanical control and the use of fire are two of the managing strategies against pine invasion used in New Zealand and South Africa (Ledgard 2009, van Wilgen et al. 2016, Ledgard 2001).

Several positive social and economic outcomes have also been associated with pines (e.g., wood production, pulp for paper, reforestation programs, ornamental, and recreational reasons) (Pauchard et al. 2016, Essl et al. 2010). Thus, increasing public awareness of the risks associated with planted pines is one step towards containing and mitigating pine invasion, but unless we present alternatives for the services provided by pines this goal will be difficult to achieve. Presenting the population with native tree species that can provide similar services could be a way of slowing down intentional pine introductions. Planting equivalent exotic non-invasive species could also be a solution. However, this suggestion should be considered with caution, as changes in the introduced environment could change the invasion status of the exotic species. Even in experienced countries such as South Africa, there are still challenges with pine control for the existing conflict of interests (van Wilgen and Richardson 2012). Thus, without a strategy that combines education and research and an intense investment in awareness programs, pines will continue to be planted for those purposes.

Invasion by pines in the Botanical Garden of Brasilia have shown a negative impact on native plant density even at its early stages, and this result combined with others for other areas, should be regarded as a warning about the dangers that exotic pine invasions can have on the local flora. Here, we provided further evidence that the presence of invasive pines has impacts on native plant community. The removal of pine trees in conservation areas can lead to an increase in native vegetation cover (Faleiros et al. 2011). Therefore, we encourage the removal of pines on native landscapes as a conservation management mechanism.

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References

Abreu RCR de, de Assis GB, Frison S, et al (2011) Can native vegetation recover after slash pine cultivation in the Brazilian Savanna? For Ecol Manage 262:1452–1459. doi: 10.1016/j.foreco.2011.06.046

Abreu RCR De, Durigan G (2013) Changes in the plant community of a Brazilian grassland savannah after 22 years of invasion by Pinus elliottii Engelm Changes in the plant community of a Brazilian grassland savannah after 22 years of invasion by. 37–41. doi: 10.1080/17550874.2011.594101

Borlaug NE (1997) Feeding a World of 10 Billion People: The Miracle Ahead. Biotechnol Biotechnol Equip 11:3–13. doi: 10.1080/13102818.1997.10818934

Braga EP, Zenni RD, Hay JD (2014) A new invasive species in South America : Pinus oocarpa Schiede ex Schltdl . BioInvasions Rec 3:6.

Buckley YM, Brockerhoff E, Langer L, et al (2005) Slowing down a pine invasion despite uncertainty in demography and dispersal. J Appl Ecol 42:1020–1030. doi: 10.1111/j.1365-2664.2005.01100.x

Dantas VDL, Pausas JG (2013) The lanky and the corky: Fire-escape strategies in savanna woody species. J Ecol 101:1265–1272. doi: 10.1111/1365-2745.12118

Dobson AP, Bradshaw AD, Baker AJM (1997) Hopes for the Future: Restoration Ecology and Conservation Biology. Science (80-) 277:515 LP-522.

Dickie IA, Bennett BM, Burrows LE, et al (2014) Conflicting values: Ecosystem services and invasive tree management. Biol Invasions 16:705–719. doi: 10.1007/s10530-013-0609-6

Durigan G, Siqueira MF De, Antonio G, Correa D (2007) THREATS TO THE CERRADO REMNANTS OF THE STATE OF SÃO PAULO , BRAZIL. 355–363.

Elton CS (1958) the Ecology of Invasions By Animals and Plants.

Essl F, Moser D, Dullinger S, et al (2010) Selection for commercial forestry determines global patterns of alien conifer invasions. Divers Distrib 16:911–921. doi: 10.1111/j.1472-4642.2010.00705.x

Falleiros RM, Zenni RD, Ziller SR (2011) Invasão e manejo de Pinus taeda em campos de altitude do parque estadual do pico paraná, paraná, brasil. Floresta 41:123–134.

Falleiros RM, Zenni RD, Ziller SR (2011) Invasão e manejo de pinus taeda em campos de altitude do parque estadual do pico paraná, paraná, brasil. Floresta 41:123–134.

Fargione JE, Tilman D (2005) Diversity decreases invasion via both sampling and complementarity effects. Ecol Lett 8:604–611. doi: 10.1111/j.1461-0248.2005.00753.x

Gorgone-Barbosa E, Pivello VR, Bautista S, et al (2014) How can an invasive grass affect fire behavior in a tropical savanna? A community and individual plant level approach. Biol Invasions 423–431. doi: 10.1007/s10530-014-0740-z

Harvey PH, Rambaut A, Trans P, et al (2000) Comparative analyses for adaptive radiations Comparative analyses for adaptive radiations. 1599–1605. doi: 10.1098/rstb.2000.0721

Higgins SI, Richardson DM (1998) Pine invasions in the southern hemisphere: modeling interactions between organism, enviroment and disturbance. Plant Ecol 135:79–93. doi: 10.1023/a:1009760512895

Klink CA, Machado RB (2005) Conservation of the Brazilian Cerrado. Conserv Biol 19:707–713. doi: 10.1111/j.1523-1739.2005.00702.x

Lankau R, Nuzzo V, Spyreas G, Davis A (2010) Evolutionary limits ameliorate the negative impact of an invasive plant. Proc Natl Acad Sci 107:1253–1253. doi: 10.1073/pnas.0913493107

Ledgard N (2001) The spread of lodgepole pine (Pinus contorta, Dougl.) in New Zealand. For Ecol Manage 141:43–57. doi: 10.1016/S0378-1127(00)00488-6

Ledgard NJ (2009) Wilding control guidelines for farmers and land managers. New Zeal Plant Prot 62:380–386.

Levine JM, Vila M, Antonio CMD, Dukes JS, Grigulis K, Lavorel S (2003) Mechanisms underlying the impacts of exotic plant invasions. Proc R Soc B Biol Sci 270:775–781. doi: 10.1098/rspb.2003.2327

Levine JM, Vilà M, D’Antonio CM, et al (2003) Mechanisms underlying the impacts of exotic plant invasions. Proc Biol Sci 270:775–781. doi: 10.1098/rspb.2003.2327

Lima JEFW, Silva EM 2008 Cerrado: Ecologia e Flora. Embrapa Cerrados - Brasília, DF: Embrapa Informação Tecnológica. 1st Edition, p. 93.

Lowe S, Browne M, Boudjelas S, De Poorter M (2000) 100 of the World's Worst Invasive Alien Species. World Published by The Invasive Species Specialist Group (ISSG) a specialist group of the Species Survival Commission (SSC) of the World Conservation Union (IUCN), 12pp.

McCann KS (2000) The diversity–stability debate. Nature 405:228–233. doi: 10.1038/35012234

Mcconnachie MM, van Wilgen BW, Richardson DM, et al (2015) Estimating the effect of plantations on pine invasions in protected areas: A case study from South Africa. J Appl Ecol 52:110–118. doi: 10.1111/1365-2664.12366

Mooney HA, Cleland EE (2001) Mooney and Cleleland 2001 The evolutionary impact of invasive species. Proc Natl Acad Sci 98:5446–5451.

Nissanka SP, Mohotti KM, Wijetunga ASTB (2005) Alleopathic influences of Pinus caribea on vegetation regeneration and soil biodiversity. In: Fourth World Congress on Allelopathy, Austrália.

Nuñez MA, Chiuffo MC, Torres A, et al (2017) Ecology and management of invasive Pinaceae around the world: progress and challenges. Biol Invasions. doi: 10.1007/s10530-017-1483-4

Nuñez MA., Medley KA (2011) Pine invasions: climate predicts invasion success; something else predicts failure. Divers Distrib 17:703–713. doi: 10.1111/j.1472-4642.2011.00772.x

Pauchard A, Cavieres L, Bustamente R, Rapoport E (2004) Increasing the understanding of plant invasion in southern South America: first symposium on alien plant Invasion in Chile. Biol Invasions 6:255–257. doi: 10.1023/B:BINV.0000022137.61633.09

Pauchard A, García R, Zalba S, Sarasota M, Zenni R, Ziller S, Nuñez MA (2016) Pine Invasions in South America : Reducing Their Ecological Impacts Trough Active Management

Perry Jr JP (1991) The pines of Mexico and Central America. Timber Press, Inc.

Pimentel D, McNair S, Kanecka j, Wightman J, Simmonds C, O'Connell C, Wong E, Russel L, Zern J, Aquino T, Tsomondo T (2001). Economic and environmental threats of alien plant, animal, and microbe invasions. Agric Ecosyst Environ, 84(1), 1-20.

Rejmanek M, Richardson DM (1996) What Attributes Make Some Plant Species More Invasive? Ecology 77:1655–1661. doi: 10.2307/2265768

Richardson DM (2006) Pinus: A model group for unlocking the secrets of alien plant invasions? Preslia 78:375–388.

Richardson DM, Pysek P, Rejmanek M, et al (2008) Naturalization and Invasion of Alien Plants : Concepts and Definitions Published by : Blackwell Publishing Stable URL : http://www.jstor.org/stable/2673320. 6:93–107.

Richardson DM, Rejmánek M (2004) Invasive conifers: a global survey and predictive framework. Divers Distrib 10:321–331. doi: 10.1111/j.1366-9516.2004.00096.x

Richardson DM, Van Wilgen BW, Nuñez M a. (2008) Alien conifer invasions in South America: Short fuse burning? Biol Invasions 10:573–577. doi: 10.1007/s10530-007-9140-y

Ricotta C, Godefroid S, Rocchini D (2009) Patterns of native and exotic species richness in the urban flora of Brussels: Rejecting the "rich get richer" model. Biol Invasions 12:233–240. doi: 10.1007/s10530-009-9445-0

Rodriguez-Trejo DA, Fulé PZ (2003) Fire ecology of Mexican pines and a fire management proposal. Int J Wildl Fire 12:23–37. doi: 10.1071/WF02040

Sampaio AB, Schmidt IB (2013) Espécies Exóticas Invasoras em Unidades de Conservação Federais do Brasil. Biodiversidade Bras 3:32–49.

Sampaio AB, Schmidt IB (2013) Espécies Exóticas Invasoras em Unidades de Conservação Federais do Brasil. Biodiversidade Bras 3:32–49.

Shimizu JY (2006) Pínus na silvicultura brasileira. Revista da Madeira, 16(99): 4-14

Silva JMC, Bates JM (2002) Biogeographic Patterns and Conservation in the South American Cerrado: A Tropical Savanna Hotspot. Bioscience 52:225. doi: 10.1641/0006-3568(2002)052[0225:BPACIT]2.0.CO;2

Simberloff D (2009) The Role of Propagule Pressure in Biological Invasions. Annu Rev Ecol Evol Syst 40:81–102. doi: 10.1146/annurev.ecolsys.110308.120304

Simberloff D, Nuñez M a., Ledgard NJ, et al (2010) Spread and impact of introduced conifers in South America: Lessons from other southern hemisphere regions. Austral Ecol 35:489–504. doi: 10.1111/j.1442-9993.2009.02058.x

Speziale K, Lambertucci S (2010) A call for action to curb invasive species in South America. Nature 467:153–153. doi: 10.1038/467153c

Stevens JT, Beckage B (2009) Fire feedbacks facilitate invasion of pine savannas by Brazilian pepper (Schinus terebinthifolius). New Phytol 184:365–375. doi: 10.1111/j.1469-8137.2009.02965.x

Stohlgren TJ, Barnett DT, Kartesz JT (2003) The rich get richer: patterns of plant invasions in the United States. Front Ecol Environ 1:11–14. doi: 10.1890/1540-9295(2003)001[0011:TRGRPO]2.0.CO;2

Strassburg BBN, Brooks T, Feltran-Barbieri R, et al (2017) Moment of truth for the Cerrado hotspot. Nat Ecol Evol 1:99. doi: 10.1038/s41559-017-0099

van Wilgen BW, Fill JM, Baard J, et al (2016) Historical costs and projected future scenarios for the management of invasive alien plants in protected areas in the Cape Floristic Region. Biol Conserv 200:168–177. doi: 10.1016/j.biocon.2016.06.008

Vila M, Weiner J (2004) Are invasive plant species better competitors than native plant species?–evidence from pair-wise experiments. Oikos 105:229–238. doi: 10.1111/j.0030-1299.2004.12682.x

Webb CO, Ackerly DD, McPeek MA, Donoghue MJ (2002) Phylogenies and Community Ecology. Annu Rev Ecol Syst 33:475–505. doi: 10.1146/annurev.ecolsys.33.010802.150448

Woodford DJ, Richardson DM, MacIsaac HJ, et al (2016) Confronting the wicked problem of managing biological invasions. NeoBiota 31:63–86. doi: 10.3897/neobiota.31.10038

Yongyi W, Kunrong C (1996) Ana adaptability study on drought resistance of caribbean pine. Journal of Southwest Forestry College, 3, 000

Zenni RD (2014) Analysis of introduction history of invasive plants in Brazil reveals patterns of association between biogeographical origin and reason for introduction. Austral Ecol 39:401–407. doi: 10.1111/aec.12097

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

Ziller SR, Dechoum MDS (2013) Plantas e Vertebrados Exóticos Invasores em Unidades de Conservação no Brasil.

Table and Figure Legends

Table 1 List of native plant species, number of plots in which they were found, density of plants per square meter, and mean diameter found in thirty plots in an area of woodland Cerrado.

Table 2 Results for t-tests of the comparing native species diversity (Shannon Diversity Index – H), Evenness (E), Probability of Interspecific Encounters (PIE), density (D) and richness (R) in the absence and presence of *Pinus sp.*

Table 3 GLM analyses of the relation between *Pinus sp.* basal area and native plant diversity (Shannon Diversity Index – H), Evenness (E), Probability of Interspecific Encounters (PIE), density (D) and richness (R)

Table 4 GLM analysis of the relation between diversity (Shannon Diversity Index – H), Evenness (E), Probability of Interspecific Encounters (PIE), native density (D) and native richness (R) with *Pinus sp.* density.

Figure 1 Histogram of native tree species richness in thirty plots in an area of woodland Cerrado at the Botanical Garden or Brasilia.

Figure 2 Histogram of native tree species basal area in thirty plots in an area of woodland Cerrado at the Botanical Garden or Brasilia.

Figure 3 Histogram of pine tree basal diameter in nineteen invaded plots in an area of woodland Cerrado at the Botanical Garden or Brasilia.

Figure 4 Box plot of the difference in native tree density in plots non-invaded and invaded by *Pinus sp*. Bold lines represents the median, boxes represents standard deviation and the fine lines represent upper and lower limits.

Figure 5 GLM relation of the effect of native plant density on the density of invasive pines. The line represents the trend and the shaded part represents the standard deviation. Only plots with presence of pines were used.

Figure 6 GLM relation of the effect of native plant density on the basal area of invasive pines. The line represents the trend and the shaded part represents the standard deviation. Only plots with presence of pines were used.

Figure 7 NMDS differences of species composition of invaded and non-invaded by *Pinus sp.* plots. Lines represent basal area of pines.

Table 1

Family	**Species**	**Number of plots**	**Density (ind.m-2)**	**Mean plant diameter**
Anacardiaceae	*Anacardium occidentale*	*2*	*3*	*7.60*
Annonaceae	*Annona coriacea*	*4*	*1*	*13.40*
Annonaceae	*Annona crassiflora*	*2*	*6*	*5.48*
Annonaceae	*Cardiopetalum calophyllum*	*1*	*5*	*2.80*
Annonaceae	*Guatteria sellowiana*	*6*	*17*	*2.98*
Annonaceae	*Xylopia aromatica*	*2*	*4*	*3.38*
Annonaceae	*Xylopia sericea*	*27*	*121*	*2.66*
Apocynaceae	*Aspidormerma tomentosum*	*23*	*76*	*4.45*
Apocynaceae	*Aspidosperma macrocarpon*	*9*	*13*	*6.25*
Apocynaceae	*Hancornia speciosa*	*2*	*4*	*7.40*
Aquifoliaceae	*Ilex affinis*	*8*	*8*	*2.78*
Araliaceae	*Schefflera macrocarpa*	*20*	*43*	*8.45*
Asteraceae	*Eremanthus glomerulatus*	*5*	*9*	*5.69*
Asteraceae	*Piotocarpha rotundifolia*	*4*	*9*	*7.56*
Bignoniaceae	*Cybistax antisyphilitica*	*2*	*2*	*15.65*
Bignoniaceae	*Cybistax sp.*	*1*	*1*	*10.80*
Bignoniaceae	*Handroanthus ochraceus*	*7*	*12*	*5.45*
Bignoniaceae	*Handroanthus serratifolius*	*2*	*1*	*14.30*
Bignoniaceae	*Jacaranda puberula*	*12*	*15*	*3.21*
Bignoniaceae	*Zeyheria montana*	*5*	*6*	*6.25*
Burseraceae	*Protium spruceanum*	*24*	*116*	*2.70*
Calophyllaceae	*Kilmeyera coriacea*	*15*	*34*	*5.80*
Calophyllaceae	*Kilmeyera speciosa*	*18*	*42*	*5.33*
Caryocaraceae	*Caryocar brasiliense*	*14*	*15*	*7.77*
Celastraceae	*Maytenus floribunda*	*6*	*25*	*5.59*
Celastraceae	*Plenckia populnea*	*5*	*5*	*2.58*
Celastraceae	*Salacia crassifolia*	*11*	*16*	*5.71*
Chrysobalanaceae	*Couepia grandiflora*	*4*	*2*	*6.85*
Combretaceae	*Terminalia fagifolia*	*14*	*31*	*6.91*
Connaraceae	*Connarus suberosus*	*14*	*20*	*4.79*
Connaraceae	*Rourea induta*	*1*	*1*	*5.10*
Dichapetalaceae	*Tapura amazonica*	*27*	*179*	*4.98*
Dilleniaceae	*Davilla elliptica*	*1*	*1*	*3.80*
Ebenaceae	*Diospyros hispida*	*4*	*16*	*6.89*
Erythroxylaceae	*Erythroxylum deciduum*	*4*	*5*	*5.64*
Euphorbiaceae	*Maprounea guianensis*	*18*	*64*	*3.43*
Fabaceae - Caes	*Chamaecrista orbiculata*	*3*	*3*	*3.23*
Fabaceae - Caes	*Copaifera langsdorffii*	*8*	*33*	*4.58*
Fabaceae - Caes	*Dimorphandra mollis*	*11*	*9*	*5.66*
Fabaceae - Caes	*Hymenae stigonocarpa*	*16*	*22*	*5.78*
Fabaceae - Caes	*Tachiagali subvenlutina*	*1*	*1*	*22.30*
Fabaceae - Caesalpinioideae	*Tachigali aurea*	*1*	*1*	*4.60*
Fabaceae - Cerciideae	*Bauhinia rufa*	*1*	*1*	*1.40*
Fabaceae - Mimo.	*Enterolobium gummiferum*	*5*	*4*	*6.43*

Fabaceae - Mimosoideae	*Plathymenia reticulada*	*1*	*1*	*6.70*
Fabaceae - Papilionoideae	*Andira vermifuga*	*3*	*4*	*6.58*
Fabaceae - Papilionoideae	*Bowdichia virgilioides*	*6*	*24*	*14.47*
Fabaceae - Papilionoideae	*Dalbergia miscolobium*	*21*	*63*	*12.24*
Fabaceae - Papilionoideae	*Hymenolobium heringerianum*	*1*	*1*	*8.60*
Fabaceae - Papilionoideae	*Leptolobium dasycarpum*	*19*	*27*	*5.09*
Fabaceae - Papilionoideae	*Machaerium acutifolium*	*1*	*5*	*2.24*
Fabaceae - Papilionoideae	*Machaerium hirtum*	*1*	*1*	*11.50*
Fabaceae - Papilionoideae	*Machaerium opacum*	*10*	*11*	*3.62*
Fabaceae - Papilionoideae	*Vatairea macrocarpa*	*1*	*4*	*6.58*
Hypericaceae	*Vismia guianensis*	*1*	*2*	*9.85*
Icacinaceae	*Emmotum nitens*	*17*	*79*	*7.53*
Lamiaceae	*Aegiphila lhotskiana*	*5*	*5*	*2.28*
Lamiaceae	*Aegiphylla sellowiana*	*4*	*3*	*13.47*
Lauraceae	*Neea theifera*	*1*	*7*	*5.01*
Lauraceae	*Ocotea spixiana*	*15*	*58*	*2.50*
Loganiaceae	*Strychnos pseudoquina*	*8*	*7*	*5.86*
Malpighiaceae	*Byrsonima coccolobifolia*	*6*	*4*	*6.80*
Malpighiaceae	*Byrsonima laxiflora*	*5*	*11*	*3.07*
Malpighiaceae	*Byrsonima pachyphylla*	*15*	*27*	*5.11*
Malvaceae	*Eriotheca pubescens*	*7*	*3*	*8.40*
Marcgraviaceae	*Morantea adamantium*	*1*	*2*	*2.50*
Marcgraviaceae	*Ocotea corymbosa*	*7*	*14*	*6.21*
Melastomataceae	*Miconia burchellii*	*26*	*94*	*6.53*
Melastomataceae	*Miconia cuspida*	*21*	*60*	*4.28*
Melastomataceae	*Miconia dodecandra*	*24*	*200*	*4.06*
Melastomataceae	*Miconia ferruginata*	*13*	*27*	*5.17*
Melastomataceae	*Miconia leucoparpa*	*2*	*18*	*2.48*
Melastomataceae	*Miconia sellowiana*	*1*	*1*	*4.10*
Melastomataceae	*Miconia sp.*	*1*	*2*	*2.30*
Moraceae	*Brosimum gaudichaudii*	*18*	*39*	*3.77*
Myristicaceae	*Virola sebifera*	*7*	*66*	*5.24*
Myristicaceae	*Virola urbaniana*	*1*	*1*	*1.60*
Myrsinaceae	*Rapanea parviflora*	*8*	*10*	*4.64*
Myrtaceae	*Blepharocalyx salicifolius*	*30*	*276*	*5.97*
Myrtaceae	*Campomanesia velutina*	*1*	*1*	*5.40*
Myrtaceae	*Eugenia dysenterica*	*9*	*25*	*5.86*
Myrtaceae	*Eugenia florida*	*4*	*41*	*3.13*
Myrtaceae	*Gomidesia lindeniana*	*6*	*7*	*3.26*
Myrtaceae	*Myrcia laruotteana*	*4*	*6*	*6.20*
Myrtaceae	*Myrcia magnoliifolia*	*1*	*1*	*1.10*
Myrtaceae	*Myrcia tomentosa*	*8*	*9*	*2.32*
Myrtaceae	*Psidium laruotteanum*	*6*	*11*	*5.95*
Myrtaceae	*Psidium myrsinites*	*1*	*5*	*1.82*
Myrtaceae	*Siphoneugena densiflora*	*18*	*76*	*3.76*
Nyctaginacae	*Guapira graciflilora*	*24*	*65*	*4.48*
Nyctaginacae	*Guapira noxia*	*20*	*35*	*7.55*
Nyctaginacae	*Guapira sp.*	*2*	*4*	*4.45*
Nyctaginaceae	*Norantea guianensis*	*1*	*7*	*4.87*

Ochnaceae	*Ouratea hexasperma*	*28*	*104*	*5.63*
Opiliaceae	*Agonandra brasiliensis*	*1*	*2*	*7.35*
Peraceae	*Pera glabrata*	*14*	*34*	*2.37*
Proteacea	*Roupala montana*	*23*	*50*	*6.22*
Rubiaceae	*Coussarea hydrandeifolia*	*1*	*3*	*10.00*
Rubiaceae	*Palicourea rigida*	*9*	*6*	*5.08*
Rutaceae	*Zanthoxylum rhoifolium*	*4*	*5*	*2.78*
Salicaceae	*Caseria grandiflora*	*2*	*1*	*2.20*
Salicaceae	*Casearia sylvestris*	*2*	*1*	*2.30*
Salicaceae	*Caseria grandiflora*	*1*	*2*	*8.50*
Siparunaceae	*Siparuna guianensis*	*1*	*3*	*2.73*
Solanaceae	*Solanum lycocarpum*	*2*	*1*	*1.90*
Styracaceae	*Styrax camporum*	*4*	*19*	*3.93*
Styracaceae	*Styrax ferrugineus*	*11*	*22*	*2.61*
Styracaceae	*Styrax oblongus*	*1*	*2*	*7.25*
Vochysiaceae	*Qualea dichotoma*	*4*	*3*	*7.43*
Vochysiaceae	*Qualea grandiflora*	*23*	*71*	*8.30*
Vochysiaceae	*Qualea parviflora*	*24*	*150*	*6.47*
Vochysiaceae	*Qualea parviflora*	*9*	*18*	*7.58*
Vochysiaceae	*Qualea sp.*	*3*	*18*	*3.78*
Vochysiaceae	*Vochysia elliptica*	*1*	*1*	*5.30*
Vochysiaceae	*Vochysia pyramidalis*	*2*	*9*	*3.02*
Vochysiaceae	*Vochysia thyrsoidea*	*11*	*12*	*8.16*
Vochysiaceae	*Vochysia tucanorum*	*3*	*17*	*3.40*

Table 2

	t	df	p
Shannon Diversity Index	-0.19857	19.203	0.8447
Evenness	0.89155	19.278	0.3836
Probability of Interspecific Encounters	-0.37728	16.797	0.7107
Density	2.1166	26.65	0.04378
Richness	0.45326	21.811	0.6548

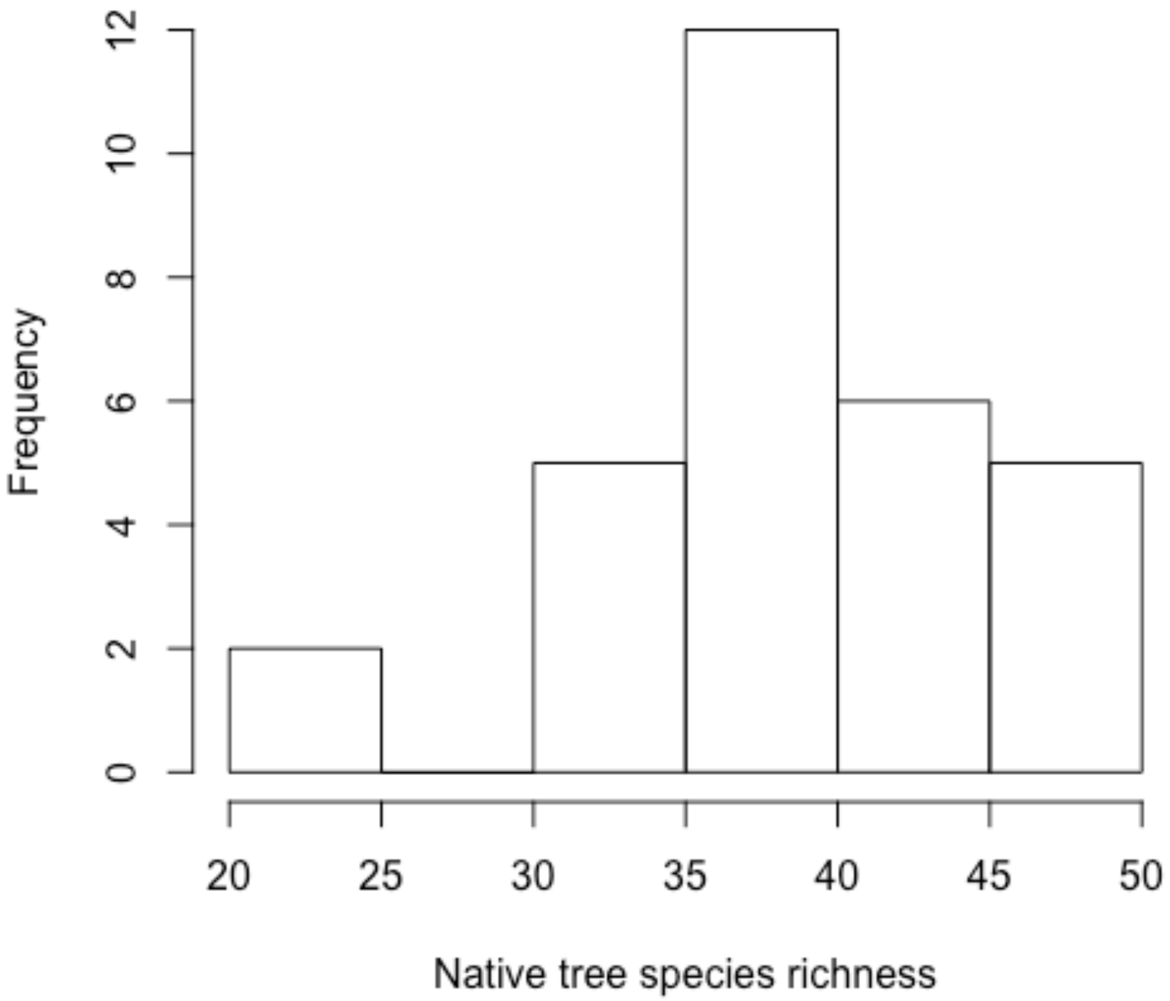
Table 3

	t	p
Shannon Diversity Index	-0.657	0.516
Evenness	0.124	0.902
Probability of Interspecific Encounters	-0.6	0.553
Density	2.053	0.0495*
Richness	-0.332	0.742

Table 4

	t value	p value
Shannon Diversity Index	-0.12	0.906
Evenness	0.374	0.711
Probability of Interspecific Encounters	-0.321	0.75
Density	2.476	0.0196*
Richness	0.121	0.905

Figure 1



Frequency
0
2
4
6
8
10
12
20
25
30
35
40
45
50
Native tree species richness

Figure 2

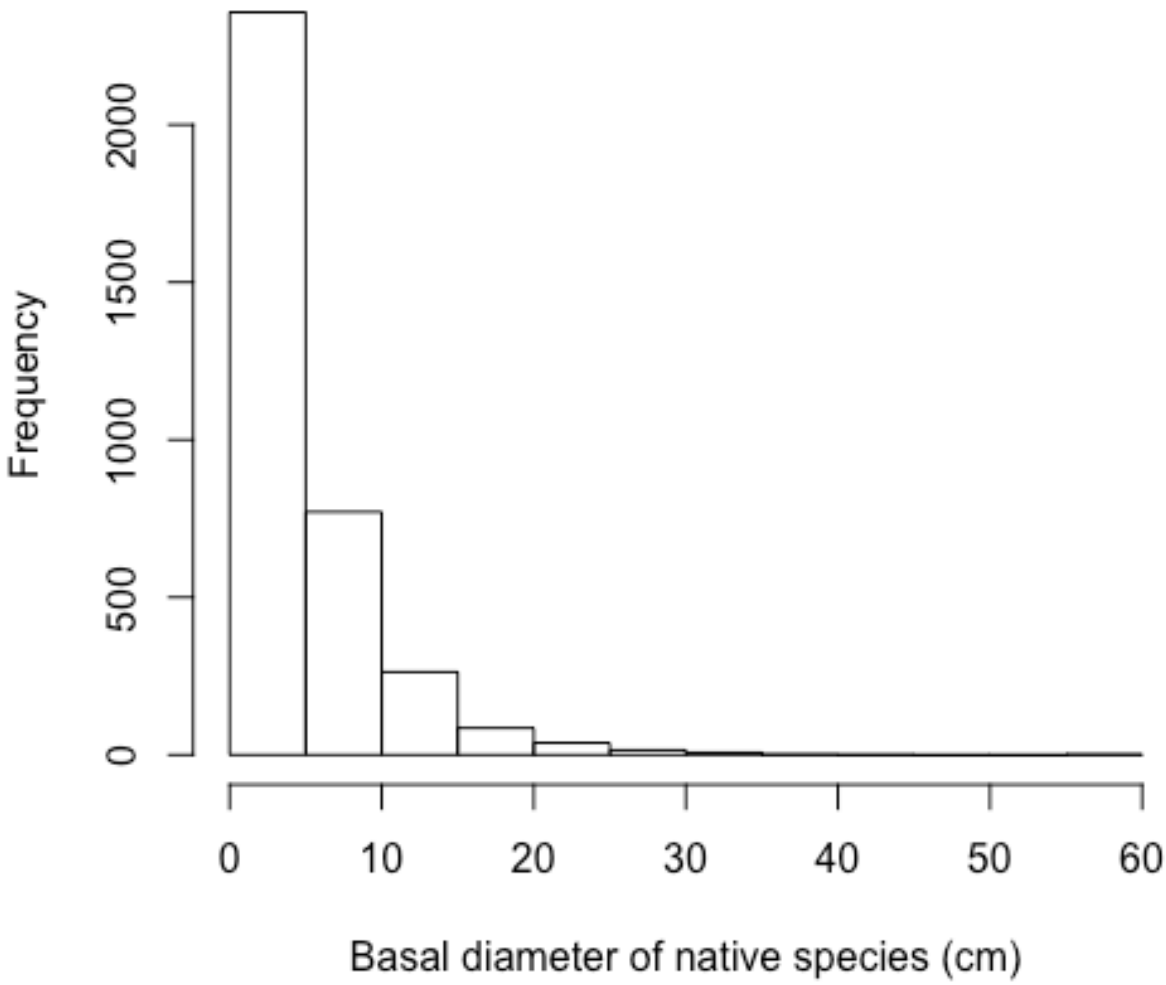
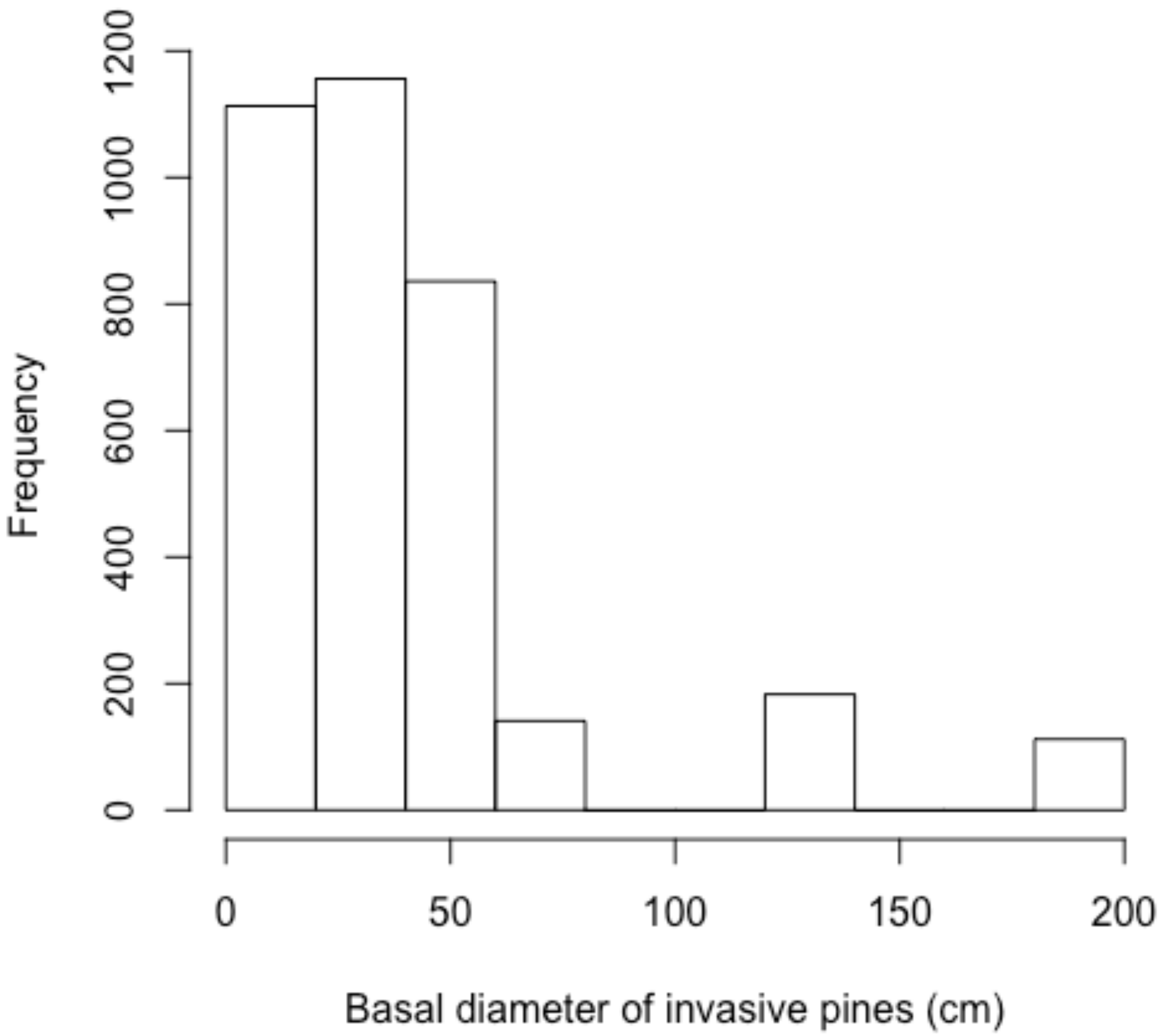

Frequency
0
500
1000
1500
2000
0
10
20
30
40
50
60
Basal diameter of native species (cm)

Figure 3



Frequency
0
200
400
600
800
1000
1200
0
50
100
150
200
Basal diameter of invasive pines (cm)

Figure 4

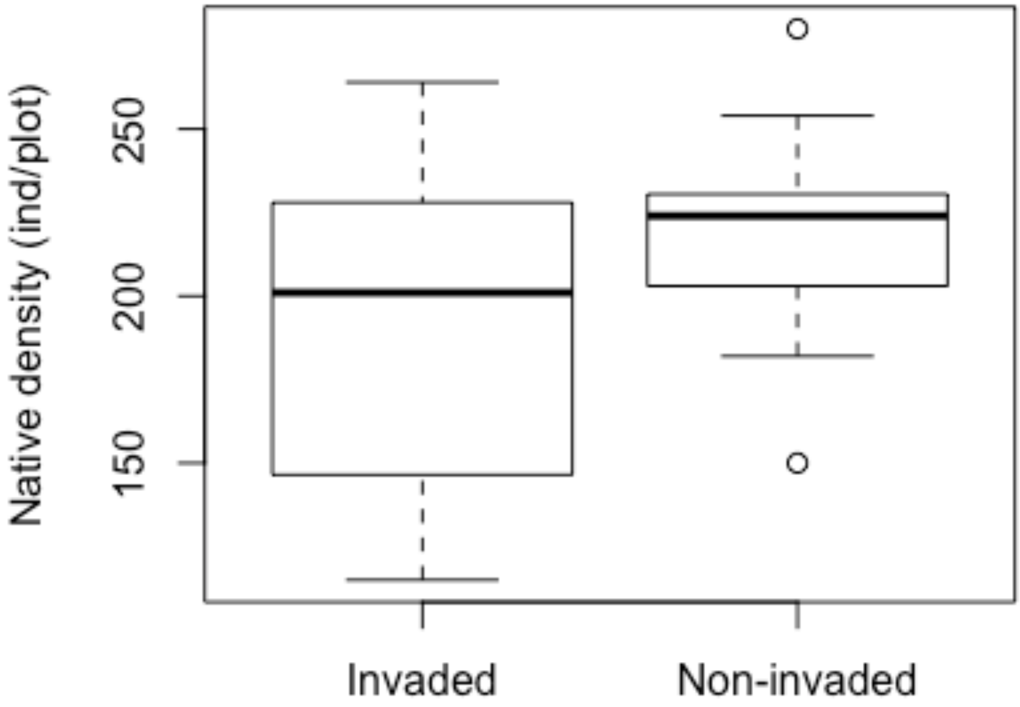
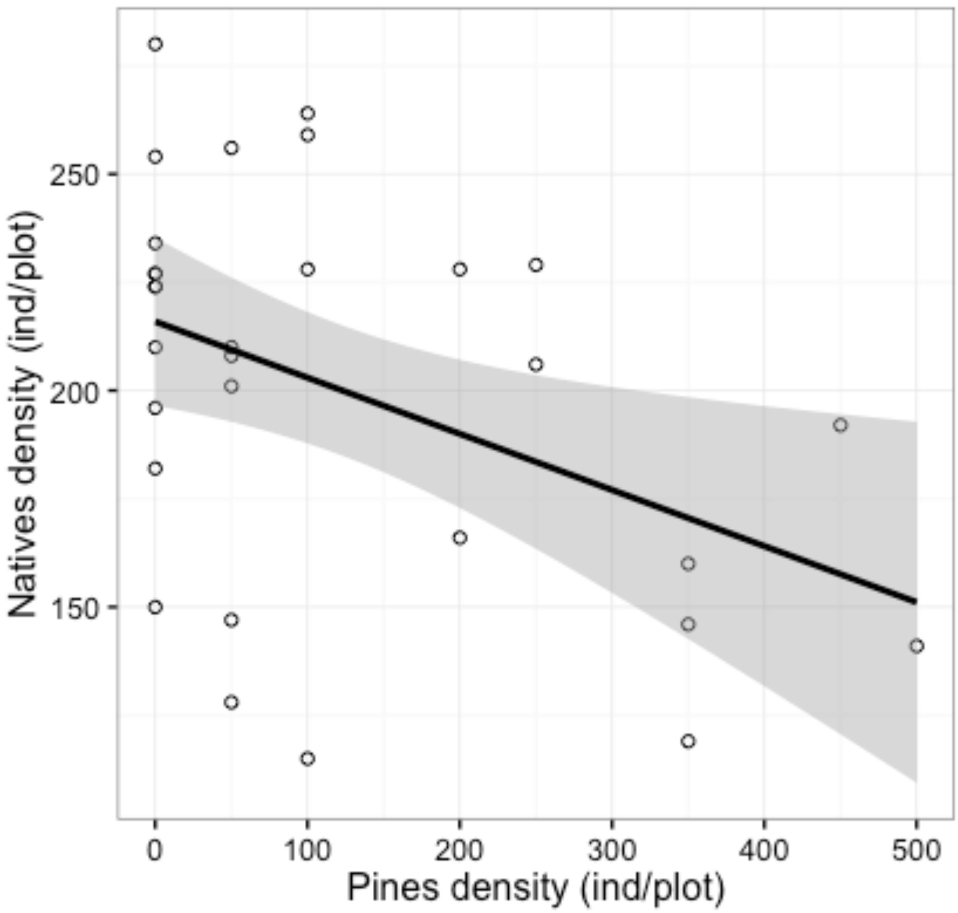
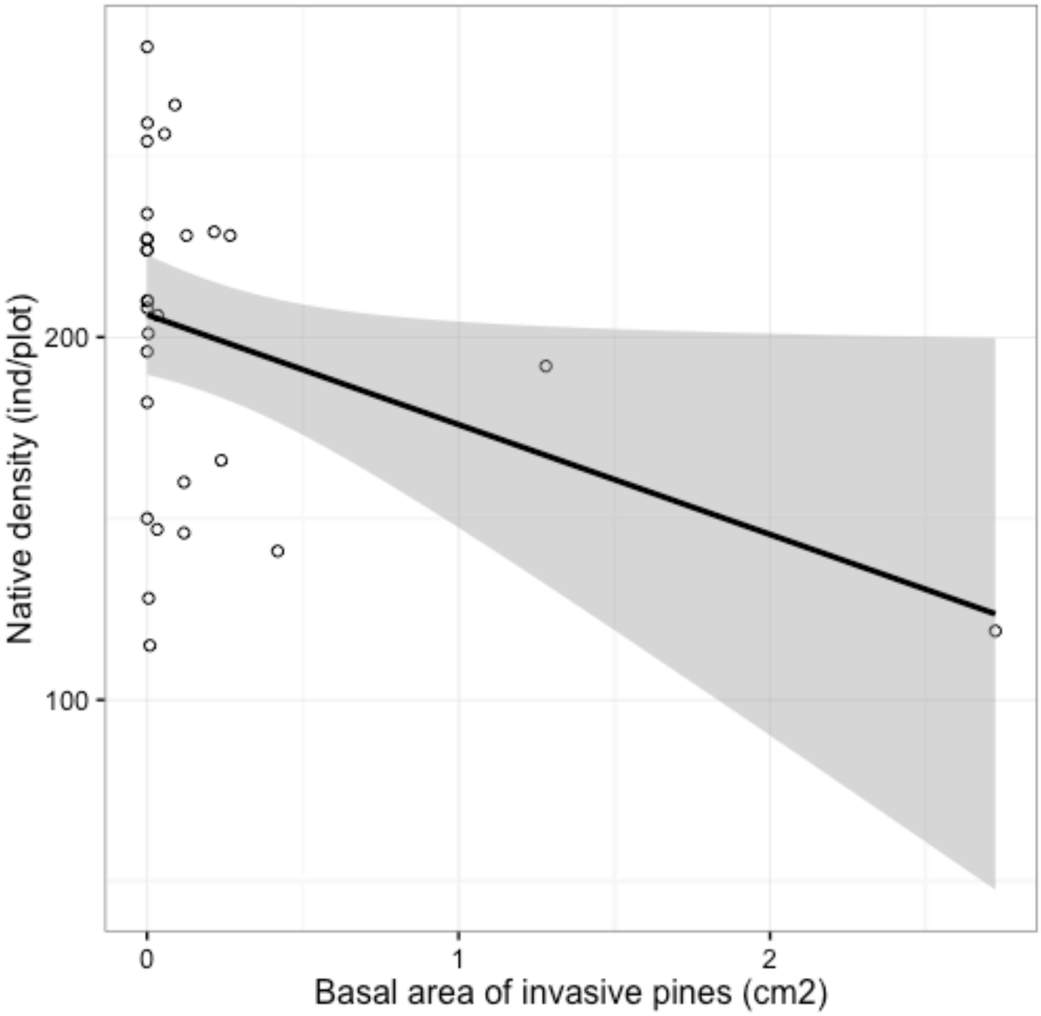


Figure 5



250
200
150
Natives density (ind/plot)
0
100
200
300
400
500
Pines density (ind/plot)

Figure 6



Native density (ind/plot)
200
100
0
1
2
Basal area of invasive pines (cm2)

Figure 7