



Changes to anuran diversity following forest replacement by tree plantations in the southern Atlantic forest of Argentina



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ABSTRACT

The replacement of native forests by high-density tree plantations affects the richness and composition of animal communities through the modification of the resource availability, hydrological regimes, nutrient cycles and soil structure. Previous studies have been conducted mainly on birds and mammals, whereas few have explored the response of anurans, a taxon considered at risk and in steep decline. With this study we aimed to evaluate changes in anuran diversity associated with native forest replacement by pine plantations (*Pinus taeda*) in the southern Atlantic forest of Argentina, a highly diverse and threatened ecosystem. Additionally, we intended to explore the role of the micro-habitat (terrestrial, aquatic) in the response. We characterized vegetation and habitat structure and sampled anurans in terrestrial and aquatic micro-habitats in the native forest and pine plantations monthly over two consecutive years (2012–2014), using three complementary techniques (pitfall traps, audio strip transects and larval sampling). A total of 964 individuals of 21 species were captured: *Physalaemus cuvieri*, *Odontophrynus americanus* and *Elachistocleis bicolor* were the most abundant species. Replacement of the structurally complex native forest by extremely simplified monoculture tree plantations influenced the patterns of anuran alpha and beta diversity. However, these changes were micro-habitat-dependent: changes in anuran diversity in the terrestrial habitat were explained by species loss, while those in the aquatic habitat were explained by both species loss and turnover. The arboreal species of the family Hylidae (eight) were found absent from tree plantations. Both the hydroperiod instability of the water bodies and the simplified vegetation structure of the tree plantations are probably limiting the suitability for both reproduction and larval development of some specialist species. The native vegetation surrounding the water bodies in the tree plantations constitutes a central element to maintain anuran diversity, through increasing the hydroperiod and providing a habitat for species reproduction in the southern Atlantic forest and to facilitate the movement of individuals among native forest remnants.

1. Introduction

The replacement of tropical and subtropical forests by fast-growing tree plantations (mainly *Pinus* and *Eucalyptus*) has become an emergent global activity (FAO, 2007). After South Africa and Australia, South America presents the highest rate of development of this activity (Baldi et al., 2008), being Argentina, Brazil and Chile the countries with the largest extensions of this land use (Simberloff et al., 2010). Native forest replacement by high-density tree plantations affects the richness and composition of animal communities through the modification of the resource availability (Richardson et al. 1994; Richardson and Higgins, 1998), hydrological regimes (Le Maitre et al., 2000), nutrient cycles (Vitousek, 1990; Jackson et al., 2002) and soil structure (Schmitz et al.,

1997). At large scales, tree plantations reduce landscape heterogeneity and the number and diversity of habitats available for species (Lindenmayer and Franklin, 2002; Magura et al., 2000; Raman, 2006). However, in contrast to other intensive land uses (such as livestock, annual crops, and urbanization), tree plantations provide a suitable habitat for many animal species through the preservation of certain components of the native forest, such as canopy cover, microclimate conditions and some trophic resources (Barbaro et al., 2005; Carnus et al., 2006; Klomp and Grabham, 2002; Pawson et al., 2008; Lantschner et al., 2008; Zurita and Bellocq, 2012).

Anurans are considered the most threatened group of vertebrates globally (Wake and Vredenburg, 2008). For the Atlantic Forest anuran community, 3.4% of species are categorized as at risk of extinction

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(Silvano and Segalla, 2005), mainly due to habitat loss and degradation (Blaustein et al., 1994; Eterovick et al., 2005; Collins and Storfer, 2003; Beebee and Griffiths, 2005) and 20% are listed as Data Deficient (IUCN, 2017). Also, because of their ecology and physiology, anurans are considered indicators of disturbance in both terrestrial and aquatic ecosystems (US EPA, 2002). Although forest plantations strongly modify habitat structure and hydrological regimes for anurans, and thus potentially their reproduction and survival, globally only few studies have focused on the effects of forest replacement by tree plantations on this taxon (Pineda et al., 2005; Urbina-Cardona and Londoño-Murcia, 2003; Faruk et al., 2013; Behm et al., 2013; Ernst and Rodel, 2005; Ernst et al., 2006). Specifically, the only two studies performed with *Pinus spp.* in Australia (Parris, 2004) and Brazil (Machado et al., 2012) have shown that monoculture tree plantations reduce anuran richness.

The Semideciduous Atlantic forest of Brazil, Argentina and Paraguay is one of the most diverse areas with the highest number of endemic species worldwide (ICPB, 1992; Laclau, 1994; Stotz et al., 1996). The high levels of humidity and micro-habitat diversity in this ecoregion promote the evolution of specialized reproductive modes in some anuran species (Brown and Brown, 1992; Haddad and Prado, 2005). However, despite the high anuran diversity and specialization and the extent of monoculture tree plantations in the southern Atlantic forest, no studies have evaluated the suitability of tree plantations for this taxon. Previous studies with vertebrates have focused mainly on birds and mammals (Marsden et al., 2001; Pardini et al., 2005; Cockle et al., 2005; Faria et al., 2006; Zurita et al., 2006; Pardini et al., 2009). In this context, our main objective was to quantify the consequences of native forest replacement by monoculture plantations (*Pinus taeda*) on the anuran diversity (both richness and the composition of species) of the southern Atlantic forest of Argentina and study the role of the micro-habitat (terrestrial, aquatic) in these changes.

2. Materials and methods

2.1. Study area and experimental design

The study was performed in the Semideciduous Atlantic forest of Argentina, which retains one of the most continuous and least degraded

remnants of Atlantic forest (Giraudo et al., 2005) (Fig. 1). The climate is subtropical without dry season; the average annual air temperature is 21 °C with monthly means of 25 °C in summer and 15 °C in winter, the annual precipitation ranges between 1600 and 2100 mm, evenly distributed throughout the year (Ligier, 2000). Landscapes in the study area are composed of large continuous native forest remnants in protected areas (Iguazú National Park and Uruguá-í Provincial Park in Misiones province), forest fragments of variable size, and a productive matrix of tree plantations, mostly of *Pinus taeda* (Zurita and Bellocq, 2010).

Within the study area, we selected three areas of continuous native forest (control, NF hereafter) and three of *Pinus taeda* monoculture tree plantations (replicates, TP hereafter). The selected TP were between 13 and 16 years old and in the third harvest cycle. Each sampling site (NF and TP) included two micro-habitats: a temporary water body (aquatic habitat) and the surrounding terrestrial habitat. Aquatic habitat referred to lentic water bodies, in all cases including TP sites, surrounded by native vegetation composed of trees in different strata and a diverse understory with grasses and lianas (*Cyperus sp.* and *Hiraea fagifolia*). All water bodies were filled through rainfall and have similar size when full (NF water bodies area $\bar{X} = 6.352 \text{ m}^2$ (SD = 8.095)/TP water bodies area $\bar{X} = 11.007 \text{ m}^2$ (SD = 9.114)). Sampling sites were located at an average distance of 25.8 km to avoid spatial autocorrelation. Anuran samplings were carried out monthly in every site for two consecutive years (from August 2012 to August 2014).

2.2. Anuran sampling

To obtain a more complete measurement of anuran diversity (both richness and composition) in both the aquatic and the terrestrial micro-habitats, we used three complementary methodologies:

1. Pitfall traps (Corn, 1994; Ribeiro-Junior et al., 2008): in each sampling site (NF and TP), we placed 16 traps, which were active for 72 h each month and checked daily (1728 sampling hours). In each site, they were grouped in four groups of four traps near the water bodies, forming a Y and connected by a shading mesh to maximize capture efficiency (2 habitats \times 3 replicates \times 4 groups \times 4 traps = 96 traps). Groups of traps were separated between 50 and

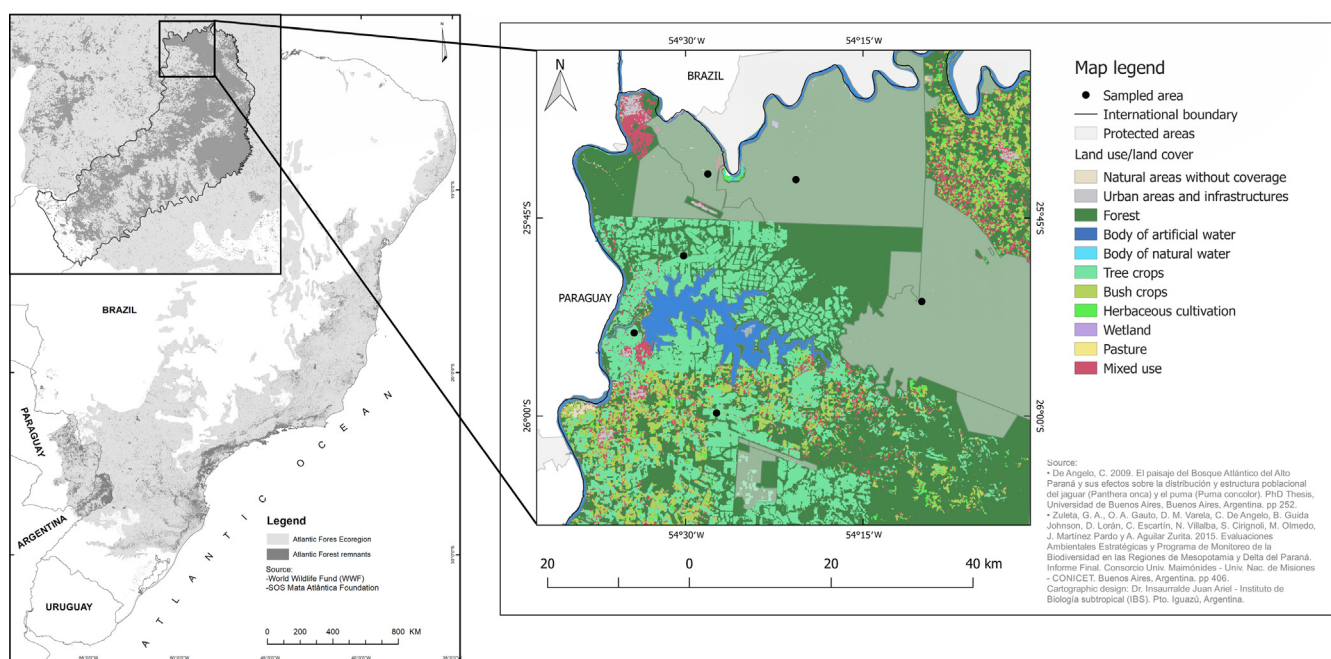


Fig. 1. Study area in the southern Atlantic forest of Argentina. Black circles represent sampling locations.

100 m. All collected individuals were identified to species level (Faivovich et al., 2005; Frost et al., 2006, Frost, 2018), sexed and marked through toe clipping to avoid double counting bias (Donnelly et al., 1994). Afterwards, individuals were released in the same site of capture.

2. Audio strip transects (Zimmerman, 1994): this complementary methodology records species usually not collected in pitfall traps (e.g. arboreal Hylids, others) (Heyer et al., 1994). In this case, we established three transects per replicate following the water body margins. Each transect was 50 m long and 25 wide to ensure the correct identification of species calls (Angulo et al., 2006). Monthly we walked each transect for 10 min between 7:00 and 8:00p.m., recording all calling species. Species were identified through the comparison with sound databases (Straneck et al., 1993; Kwet and Márquez, 2010). Only individuals calling in water margins were considered.
3. Larval sampling (Lips et al., 2001): we established transects along the water body margins and used a random sampling methodology stratified by depth (Heyer et al., 1994). The sampling effort (number of net sweeps) was similar for each site in each month that samples could be taken; in some cases (dry water bodies) samples were not taken. Captured tadpoles were fixed in 10% formalin and later identified in the laboratory (Rossa-Feres and Nomura, 2006; Lynch and Suarez-Mayorga, 2011).

2.3. Terrestrial and aquatic habitat structure

In each replicate, we established three 100-m-long transects to characterize the vegetation structure. Every 10 m of each transect (198 sampling points), we recorded: (1) the canopy cover, through hemispheric photographs (Cook et al., 1995; Nuttle, 1997), (2) the understorey cover, using a 1.5-meter vertical bar to record vegetation intersection regarding their life forms (herbaceous, shrubs, ferns, lianas, reeds, epiphytes, etc.) (Matteucci and Colma, 1982), and (3) the depth of the litter layer above the mineral soil. Also, along each transect, we recorded the Diameter at Breast Height (DBH) of every tree > 10 cm in a 2.5-m area.

To characterize the temporary water bodies, at every replicate we recorded: (1) the hydroperiod, measured as the number of sampling events (months) with water surface (ranges between 1 and 24 events); (2) depth, measured in the center of the water body, and (3) water pH and conductivity, using a portable pH/mV meter (HANNA HI9813-5).

2.4. Data analysis

In relation to anuran community, we analyzed data from (1) pit fall traps (abundance data) and (2) larval sampling and audio strip transects (incidence data) separately, since the former refers to the terrestrial habitat and the latter to the aquatic habitat. We estimated sample coverage as a function of the number of individuals (abundance data) or sample size (incidence data) with the coverage index, following Chao and Jost (2012). We calculated Hill's numbers (q0, q1 and q2) with 95% confidence intervals based on a resampling method with 1000 replicates (Felsenstein, 1985), to compare alpha diversity between NF and TP. We performed rank abundance curves using unconditional averages (Chao et al., 2015), to characterize the structure of anuran communities. Statistical analyses were performed with the iNEXT package (iNterpolation/EXTrapolation) (Hsieh et al., 2016) in R (R Development Core Team, 2011).

We performed a nonmetric multidimensional scaling (NMDS) analysis to explore changes in species composition between NF and TP, using the Bray-Curtis dissimilarity index (Kruskal, 1964; Faith et al., 1987). In this analysis, one of the plantation sites was discarded (n = 11) for having a single record of a single species (one individual). Then, we used the dissimilarity distance matrix to compare community composition between micro-habitats (aquatic and terrestrial) and

habitats (NF and TP), using a permutation multivariate analysis of variance with two factors and the interaction (PERMANOVA) (Anderson, 2001). Analyses were performed using the Adonis function of the vegan package (Oksanen et al., 2011) in R (R Development Core Team, 2011).

We analyzed beta-diversity (β_{sor} – Sorensen dissimilarity) between NF and TP, and its partition, according to Baselga (2010), into two components: one considering the dissimilarity derived exclusively from species turnover (β_{sim} – Simpson dissimilarity) and the other considering the dissimilarity derived from nestedness (β_{nes} – Nestedness resultant dissimilarity). We used a reference forest community calculated as an average of the three forest sites, to estimate the dissimilarity between NF and TP. For both abundance (terrestrial) and incidence (aquatic) data, we used the Betapart package (Baselga et al., 2017) developed for R (R Development Core Team, 2011). Finally, the values of beta diversity from species turnover and nestedness components were compared through a paired t-test (Zar, 1984).

To quantify differences in the aquatic and the terrestrial habitats between NF and TP, we compared: canopy cover, understorey cover, litter depth, and DBH, and hydroperiod, depth, pH and conductivity of water bodies, using a non-parametric Kruskal-Wallis (H) analysis (Di Rienzo et al., 2013).

3. Results

We registered a total of 18 species of adult anurans with pitfall traps and audio strip transects, and 13 species of tadpoles through larval sampling (Table 1).

Pit fall traps collected 963 individuals, mainly of the family Leptodactylidae (n = 685), whereas audio strip transects and larval sampling (incidence data) registered mainly species of the family

Table 1
Species richness and abundance of adult anurans and tadpoles captured with three techniques (pit fall traps, larval sampling and audio strip transects) respectively in tree plantations and the native forest in the southern Atlantic forest of Argentina.

Cod		Native forest (NF)			Tree plantations (TP)		
		Adults	Tadpoles	P/A	Adults	Tadpoles	P/A
Ro	<i>Rhinella ornata</i>	15	1	0	0	0	0
Rs	<i>Rhinella schneideri</i>	2	0	0	0	0	0
Oa	<i>Odontophrynus americanus</i>	129	0	1	24	1	1
Pa	<i>Proceratophrys avelinoi</i>	1	0	0	0	1	1
Ap	<i>Aplastodiscus perviridis</i>	0	0	0	0	0	0
Dm	<i>Dendropsophus minutus</i>	0	1	1	0	0	0
Dn	<i>Dendropsophus nanus</i>	0	1	1	0	0	0
Bc	<i>Boana caingua</i>	0	1	1	0	0	0
Bf	<i>Boana faber</i>	0	0	1	0	0	0
Il	<i>Itapotihyla langsdorffii</i>	0	1	0	0	0	0
Ob	<i>Ollolygon berthae</i>	0	1	0	0	1	1
Scr	<i>Scinax clado ruber</i>	0	1	0	0	0	0
Sf	<i>Scinax fuscovarius</i>	0	1	1	0	0	0
Ss	<i>Scinax squalirostris</i>	0	0	1	0	0	0
Tt	<i>Trachycephalus typhonius</i>	0	1	0	0	0	0
Lg	<i>Leptodactylus gracilis</i>	0	0	0	0	0	0
Le	<i>Leptodactylus elenae</i>	6	0	1	9	0	0
Lf	<i>Leptodactylus fuscus</i>	1	0	0	0	0	0
Ll	<i>Leptodactylus latrans</i>	37	0	0	3	0	0
Lm	<i>Leptodactylus mystacinus</i>	28	0	1	11	0	0
Pc	<i>Physalaemus cuvieri</i>	525	1	1	65	0	0
Eb	<i>Elachistocleis bicolor</i>	22	0	0	85	1	1
	Richness	10	10	10	6	4	4
	Abundance	766			197		

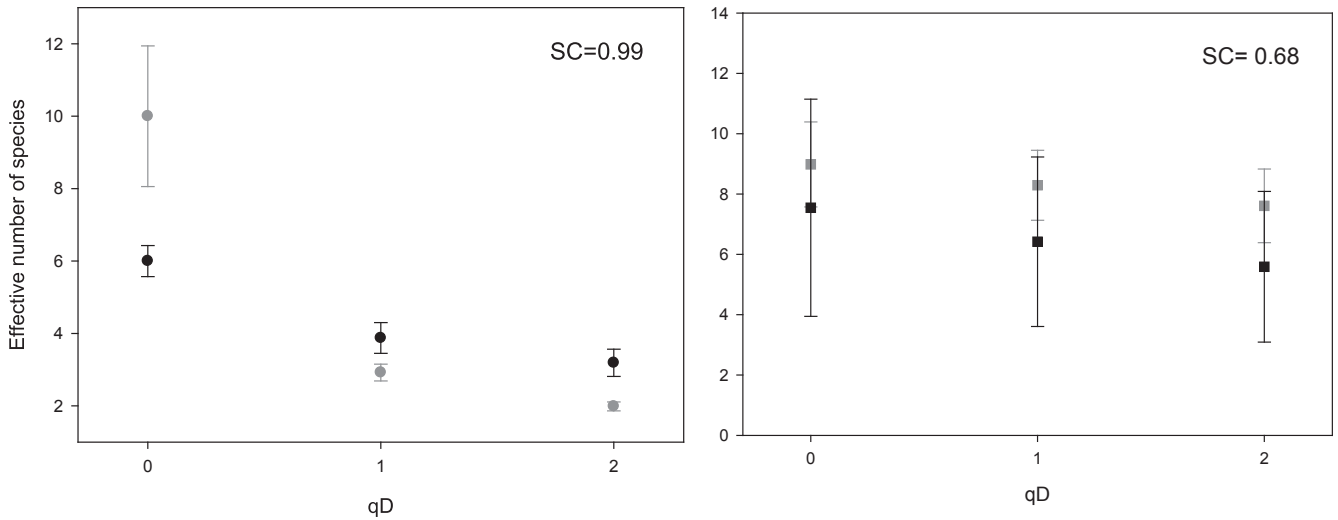


Fig. 2. Alpha diversity (q0, q1, q2) in anuran communities in terrestrial (left figure) and aquatic habitats (right figure) of tree plantations (black figures) and the native forest (gray figures) in the southern Atlantic forest of Argentina. SC: sample coverage, 95% confidence intervals.

Hylidae. Abundance data (pitfall traps, terrestrial habitat) reached 99% of sample coverage for both NF and TP, whereas incidence data (audio strips and larval sampling, aquatic habitat) reached 94% of sample coverage for NF and 68% for TP (Chao et al., 2015). Consequently, we interpolated incidence data to the lowest sample coverage level (68%), to perform a correct interpretation and comparison of alpha diversity between NF and TP.

Alpha diversity comparison showed that richness (q0) was higher while equitability and dominance (q1 and q2) were lower in NF than in TP in the terrestrial habitat (abundance data) (Fig. 2); the aquatic habitat showed a similar pattern, although differences were not significant due to large confident intervals (incidence data, rarified to 68% sample coverage). Rank abundance curves showed similar results: the anuran community in NF was dominated by a single species (*Physalaemus cuvieri* in the terrestrial habitat and *Dendropsophus minutus* in the aquatic habitat), whereas we did not observe a clear dominance of a single species (neither for abundance nor incidence data) in TP (Fig. 3).

The NMDS analysis showed three distinctive groups based on the composition of species: terrestrial habitat of NF and TP formed a single group, while NF aquatic habitat and TP aquatic habitat formed separated groups (Fig. 4). Consistent with this result, the PERMANOVA showed a significant effect of the micro-habitat (aquatic/terrestrial) (R^2

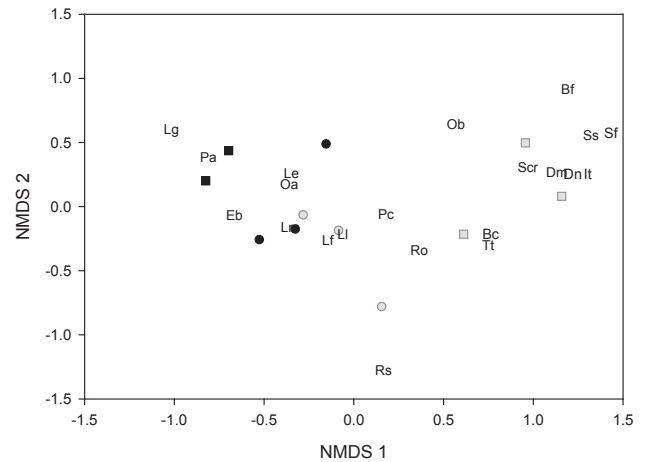


Fig. 4. Non-metric multidimensional (NMDS) analysis of anuran communities in the native forest (gray figures) and tree plantations (black figures) in the southern Atlantic forest of Argentina. Circles: terrestrial habitat, Squares: aquatic habitat. For species names, see Table 1.

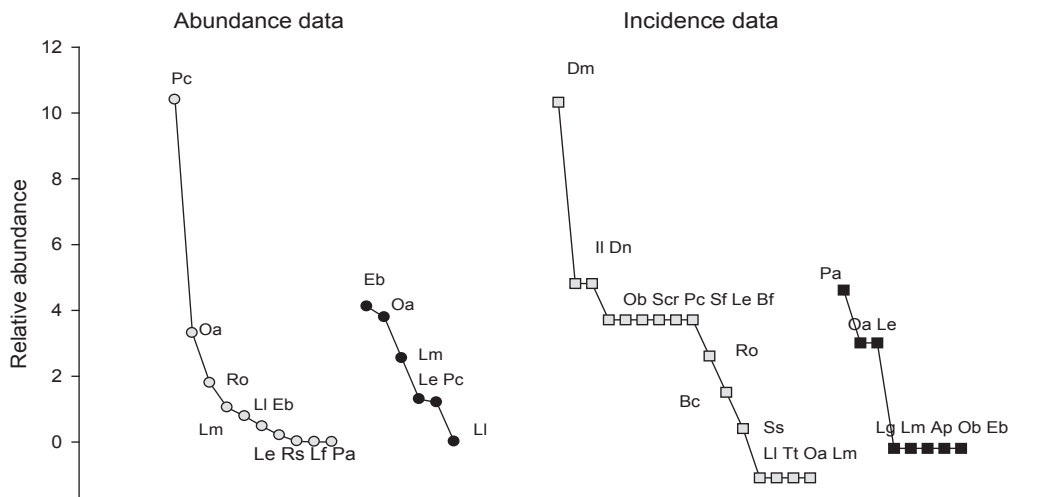


Fig. 3. Rank abundance distributions of anuran communities in terrestrial (abundance data) and aquatic habitats (incidence data) in the native forest (gray circles) and tree plantations (black circles) in the southern Atlantic forest of Argentina. For species names, see Table 1.

Table 2

Anuran dissimilarity values between NF and TP (β SOR), partitioned in turnover (β SIM) and nestedness (β SNE), for aquatic and terrestrial habitats in the southern Atlantic forest of Argentina.

	β SOR	β SNE	β SIM	T	P
Terrestrial NF- TP	0.45	0.38	0.07	-4.14	0.053
Aquatic NF- TP	0.77	0.27	0.5	1.49	0.274

= 0.217, $P = 0.010$), the habitat (NF/TP) ($R^2 = 0.322$, $P = 0.010$) and the interaction between factors ($R^2 = 0.204$, $P = 0.008$) on species composition. Specifically, species of the family Hylidae (*Ololygon berthae*, *S. fuscovarius*, *S. squalirostris*, *Trachycephalus typhonius*, *Dendropsophus nanus*, *D. minutus*, *Boana caingua*, *B. faber* and *Itapotihyla langsdorffii*) were associated with the aquatic habitat of NF but absent in the aquatic habitat of TP.

Beta-diversity (β SOR – Sorensen dissimilarity) partitioning between NF and TP showed that differences in species composition in the aquatic habitat were due to both nestedness and turnover ($t_4 = 1.49$, $P = 0.274$), whereas for the terrestrial habitat differences in species composition were mainly due to nestedness ($t_4 = -4.14$, $P = 0.05$) (Table 2).

Habitat structure comparisons between NF and TP showed that canopy cover, understory cover, hydroperiod and depth of water bodies were higher in NF, while tree DBH was higher in TP (Table 3). Litter depth and water pH and conductivity showed no significant differences between NF and TP (Table 3).

4. Discussion

Like that observed in previous studies in tropical and subtropical regions (Pineda et al., 2005; Urbina-Cardona and Londoño-Murcia, 2003; Behm et al., 2013; Ernst et al., 2006; Parris and Lindenmayer, 2004; Gutiérrez-Lamus et al., 2004), in the present study we observed that the replacement of NF by monoculture TP altered both the richness and composition of the southern Atlantic forest of Argentina anuran communities. However, for communities studied these changes were micro-habitat-dependent: changes in anuran diversity in the terrestrial habitat of TP were explained by species loss, while those in the aquatic habitat were explained by both species loss and turnover.

Environmental heterogeneity has been identified as one of the main determinants of anuran species richness and composition (Parris and McCarthy, 1999; Parris, 2004; Afonso and Eterovick, 2007; Bastazini et al., 2007). The environmental homogenization following the replacement of NF by TP probably explains the loss of species in the terrestrial micro-habitat observed in this and previous studies (Pineda

Table 3

Terrestrial and aquatic habitat structure (median and first and third quartiles) for anurans in the native forest (NF) and tree plantations (TP) in the southern Atlantic forest of Argentina.

Variable	NF	TP	H
Litter depth (cm)	2.00 (1.00–3.00)	2.00 (1.50–2.00)	0.00
Canopy cover (%)	0.97 (0.94–0.98)	0.93 (0.89–0.96)	24.21**
DBH (cm)	27.9 (20.50–46.82)	60.00 (42.75–75.00)	29.25**
Understory cover (intersections)	2.00 (1.00–4.00)	2.00 (0.00–3.00)	7.97**
Hydroperiod (%)	0.92 (0.70–1.00)	0.33 (0.11–0.54)	12.74*
Depth (cm)	15.50 (0.00–20.90)	0.00 (0.00–0.00)	23.44**
pH	6.40 (6.07–6.68)	6.84 (6.32–6.98)	1.55
Conductivity (mV)	58.00 (43.00–82.00)	36.00 (28.00–65.00)	1.83

H = Kruskal-Wallis statistic.

* $P < 0.05$.

** $P < 0.01$.

et al., 2005; Ernst et al., 2006; Behm et al., 2013). Although the terrestrial habitats of NF and TP share a similar general structure, composed of litter, understory and canopy, NF exhibits greater life forms and coverage heterogeneity (such as shrubs, herbs and ferns) within each stratum. Despite NF and TP showed similar litter depth, Zaninovich et al. (2016) showed that necromass composition between NF and TP strongly differ; NF exhibit higher proportion of coarse woody debris (> 2 mm) compared to TP. Coarse woody debris provides a relatively stable source of moisture compared to the litter layer or the mineral soil (Fraver et al., 2002; Harmon and Sexton, 1995), as a consequence necromass moisture content retained in NF was almost four times higher than TP (Zaninovich et al., 2016). This means that in the NF the water storage and sponge-like structure of well-decayed logs makes them an ideal habitat for moisture-sensitive species like anurans (Petranka et al., 1994; Fraver et al., 2002; Yan et al., 2007). The higher understory and canopy coverage and greater complexity of the NF prevent insolation and desiccation and provide refuge and nesting places to anuran adults, as well as protection to eggs and tadpoles. Unlike that observed in NF, the water bodies in TP are exposed to direct sun, which generates high water temperatures, evaporation and different water balance (Cristiano et al., 2015), and thus causes a reduction in both the water depth and the hydroperiod (64%). These differences increase risk of mortality by desiccation for both adults and tadpoles, particularly for sensitive native forest species. The higher environmental homogenization and the lower quality of water bodies probably explain the absence of rare species and the higher equitability observed in TP for both micro-habitats. Previous studies with anurans in degraded habitats have also shown a non-random pattern of species loss, being rare species more affected than common species (Ernst and Rodel, 2005; Ernst et al., 2006; Hillers et al., 2008).

The water bodies in NF were used by eight species of the family Hylidae (*Dendropsophus nanus*, *D. minutus*, *Boana caingua*, *B. faber*, *Itapotihyla langsdorffii*, *Scinax fuscovarius*, *S. squalirostris*, and *Trachycephalus typhonius*), which were not found in the terrestrial or aquatic habitats of TP. Hylids are susceptible to water volume variations, since they deposit their eggs on the water surface or submerged in gelatinous masses (Duellman and Trueb, 1994; Arzabe et al., 1998); also larval developmental time is longer than species of the family Leptodactylidae, which were abundant in TP. On the other hand, habitats with higher vegetation stratification offer a vertical distribution of vocalization sites, benefiting arboreal species such as Hylids, whereas monostratified habitats offer greater horizontal distribution of calling sites benefiting Leptodactylids (Cardoso et al., 1989). Therefore, the two most diverse anuran families of the Atlantic Forest (Hylidae and Leptodactylidae, comprising 64.3% of total species) have substantially different habitat requirements for reproduction. In TP, the hydroperiod instability and simplified vegetation structure strongly reduced the suitability for both reproduction and larval development for most Hylidae species.

While previous studies analyzed changes in anuran species composition between forests and monoculture land uses (beta diversity) (Pineda et al., 2005; Pineda and Halfpeter, 2004; Faria et al., 2006), this is the first study exploring the role of species loss or replacement to explain these changes. As previously mentioned, in the terrestrial habitat, anuran changes in species composition between NF and TP were due exclusively to species loss (particularly of those of the family Hylidae) (Hazell et al., 2001; Guerry and Hunter, 2002), persisting only those more resistant to environmental changes (*Leptodactylus mystacinus*, *L. elenae*, *Odontophrynus americanus* and *Elachistocleis bicolor*). Conversely, species turnover was an important component to explain differences between NF and TP in water bodies; the particular combination of local and environmental factors of each water body combined with the selective colonization of species may explain these differences. In fact, anuran species distribution in water bodies can be very volatile, and species turnover might be the main component of regional diversity (Skelly et al., 1999, Adams, 2000).

Replacement of the structurally complex NF with high habitat heterogeneity by extremely simplified monoculture TP has strong and different effects on anuran diversity of both the terrestrial and aquatic micro-habitats. While several anuran species use the temporary water bodies of tree plantations for reproduction (high species richness at larval stage), many of these species were absent in the terrestrial habitats. This might indicate that species use water bodies to reproduce but (1) do not pass the larval stage, (2) migrate to other habitats in response to the instability of the water bodies in the tree plantations. Moreover, the connectivity between ponds (metapopulation dynamics) can play an important role in anuran communities, and can contribute to the species turnover among water bodies (Burne and Griffin, 2005; Vitt and Caldwell, 2009).

Maintenance of the native vegetation around the periphery of water bodies will probably reduce water evaporation and infiltration, increasing the hydroperiod and enabling many anuran species (mainly Hylids) to complete their life cycle. At the same time, the native vegetation provides shelters and breeding sites to adults. Although national and provincial legislation related to water courses and springs forbids deforestation and determines a minimum of 10 m of native forest in each margin (Misiones provincial law XVI-105, 2010), there is no legislation related to wetlands and temporary water bodies. In contrast, the protection of native vegetation around wetlands in monoculture TP is contemplated in international environmental certifications (Forest Stewardship Council Principles 6.2, 6.5 and 10.6, 2002). Native vegetation around the water bodies in tree plantations probably constitutes a fundamental element to maintain anuran diversity in the southern Atlantic forest of Argentina and to facilitate the movement of individuals among native forest remnants. The results of this study support the need for legislation to create buffers of native vegetation around temporary water sources that are currently being contemplated by international FSC principles.

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References

- Adams, M.J., 2000. Pond permanence and the effects of exotic vertebrates on anurans. *Ecol. Appl.* 10, 559–568.
- Afonso, L.G., Eterovick, P.C., 2007. Spatial and temporal distribution of breeding anurans in streams in southeastern Brazil. *J. Nat. Hist.* 41 (13–16), 949–963.
- Anderson, M.J., 2001. A new method for non-parametric multivariate analysis of variance. *Aust. Ecol.* 26, 32–46. <http://dx.doi.org/10.1111/j.1442-9993.2001.01070.pp.x>.
- Angulo, A., Rueda-Almonacid, J.V., Rodríguez-Mahecha, J.V., La Marca, E., 2006. Técnicas de Inventario y Monitoreo para los anfibios de la Región Tropical Andina. Conservación Internacional, Serie Manuales de Campo. Panamericana Formas e Impresos, Bogotá, Colombia.
- Arzabe, C., Carvalho, C.X.Y., Costa, M.A.G., 1998. Anuran assemblages in crasto forest ponds (Sergipe State, Brazil): comparative structure and calling activity patterns. *Herpetology* 8, 111–113.
- Baldi, G., Noretto, M.D., Jobbágy, E.G., 2008. El efecto de las plantaciones forestales sobre el funcionamiento de los ecosistemas sudamericanos. *Ambiência* 4, 23–34.
- Barbaro, L., Pontcharraud, L., Vetillard, F., Guyon, D., Jactel, H., 2005. Comparative responses of bird, carabid, and spider assemblages to stand and landscape diversity in maritime pine plantation forests. *Ecoscience* 12, 110–121. <http://dx.doi.org/10.2980/11195-6860-12-1-110.1>.
- Baselga, A., 2010. Partitioning the turnover and nestedness components of beta diversity. *Global Ecol. Biogeogr.* 19, 134–143.
- Baselga, A., Orme, D., Villeger, S., De Bortoli, J.Y., Leprieur, F., 2017. Betapart: Partitioning Beta Diversity into Turnover and Nestedness Components. R package version 1.4-1. <https://CRAN.R-project.org/package=betapart>.
- Bastazini, C.V., Munduruca, J.F.V., Rocha, P.L., Napoli, M.F., 2007. Which environmental variables better explain changes in anuran community composition? A case study in the restinga of Mata de Sao Joao, Bahia, Brazil. *Herpetologica* 63 (4), 459–471.
- Beebee, T.J.C., Griffiths, R.A., 2005. The amphibian decline crisis: a watershed for conservation biology? *Biol. Conservat.* 125, 271–285.
- Behm, J.E., Yang, X., Chen, J., 2013. Slipping through the cracks: rubber plantation is unsuitable breeding habitat for frogs in Xishuangbanna, China. *Plos ONE* 8 (9), e73688.
- Blaustein, A.R., Wake, D., Sousa, W.P., 1994. Amphibians declines: judging stability, persistence, and susceptibility of populations to local and global extinctions. *Conserv Biol* 8, 60–71.
- Brown, K.S.Y., Brown, G.G., 1992. Habitat alteration and species loss in Brazilian forests. In: Whitmore, T.C., Sayer, J.A. (Eds.), *Tropical Deforestation and Species Extinction*. Chapman and Hall, London, pp. 119–142.
- Burne, M.R., Griffin, C.R., 2005. Habitat associations of pool-breeding amphibians in eastern Massachusetts, USA. *Wetl. Ecol. Manage.* 13 (3), 247–259.
- Cardoso, A.J., Andrade, G.V.Y., Haddad, C.F.B., 1989. Distribuição espacial em comunidades de anfíbios (Anura) no sudeste do Brasil. *Rev. Bras. Biol.* 49, 241–249.
- Carnus, J.-M., Parotta, J., Brockerhov, E.G., Arbez, M., Jactel, H., Kremer, A., Lamb, D., O'Hara, K., Walters, B., 2006. Planted forests and biodiversity. *J. For.* 104 (2), 65–77.
- Chao, A.Y., Jost, L., 2012. Coverage-based rarefaction and extrapolation: Standardizing samples by completeness rather than size. *Ecology* 93 (12), 2533–2547.
- Chao, A., Hsieh, T.C., Chazdon, R.L., Colwell, R.K.Y., Gotelli, N.J., 2015. Unveiling the species-rank abundance distribution by generalizing the Good-Turing sample coverage theory. *Ecology* 96 (5), 1189–1201.
- Cockle, K.L., Leonard, M.L., Bodrati, A.A., 2005. Presence and abundance of birds in an Atlantic forest reserve and adjacent plantation of shade-grown yerba mate in Paraguay. *Biodivers. Conserv.* 14, 3265–3288.
- Collins, J.P., Storer, A., 2003. Global amphibian declines: sorting the hypotheses. *Divers. Distrib.* 9, 89–98.
- Cook, J.G., Stutzman, T.W., Bowers, C.W., Brenner, K.A.Y., Irwin, L.L., 1995. Spherical densitometers produce biased estimates of forest canopy cover. *Wildl. Soc. Bull.* 23, 711–717.
- Corn, S.P., 1994. Pit-fall traps with fences. In: Heyer, W.R., Donnelly, M.A., McDiarmid, R.W., Hayek, L.A.C., Foster, M.S., (Eds.), *Measuring and Monitoring Biological Diversity. Standard Methods for Amphibians*. Washington & London, Smithsonian Institution Press, 364 pp (pp.105–112).
- Cristiano, P., Campanello, P.I., Bucci, S.J., Rodriguez, S.A., Lezcano, O.A., Scholz, F.G., Madanes, N., Di Francescantonio, D., Oliva Carrasco, L., Zhang, Y.Y., Goldstein, G., 2015. Evapotranspiration of subtropical forests and tree plantations: A comparative analysis at different temporal and spatial scales. *Agric. For. Meteorol.* 203, 96–106.
- Di Rienzo, J.A., Casanoves, F., Balzarini, M.G., Gonzalez, L., Tablada, M., Robledo, C.W., 2013. Infostat versión 2013. Grupo infostat, FCA. Universidad Nacional de Córdoba, Córdoba.
- Donnelly, M.A., Guyer, G., Juterbock, J.E.Y., Alford, R.A., 1994. Techniques for marking amphibians. In: Heyer, W.R., Donnelly, M.A., McDiarmid, R.W., Hayek, L.A.C., Foster, M.S. (Eds.), *Measuring and monitoring biological diversity. Standard methods for amphibians*. Smithsonian Institution Press.
- Duellman, W.E.Y., Trueb, L., 1994. *Biology of Amphibians*. McGraw-Hill, Baltimore and London.
- Ernst, R.Y., Rodel, M.O., 2005. Anthropogenically induced changes of predictability in tropical anuran assemblages. *Ecology* 86, 3111–3118.
- Ernst, R., Linsenmair, K.E.Y., Rodel, M.O., 2006. Diversity erosion beyond the species level: dramatic loss of functional diversity after selective logging in two tropical amphibian communities. *Biol. Conserv.* 133, 143–155.
- Eterovick, P.C., Oliveira de Queiroz Carnaval, A.C., Borges-Nojosa, D.M., Leite Silvano, D., Vicente Segalla, M., Sazima, I., 2005. Amphibian declines in Brazil: An overview. *Biotropica* 37, 166–179.
- Faith, D.P., Minchin, P.R.Y., Belbin, L., 1987. Compositional dissimilarity as a robust measure of ecological distance. *Vegetatio* 69, 57–68.
- Faivovich, J., Haddad, C.F.B., Garcia, P.C.A., Frost, D.R., Campbell, J.A., Wheeler, W.C., 2005. Systematic review of the frog family Hylidae, with special reference to Hylinae: a phylogenetic analysis and taxonomic revision. *Bull. Amer. Mus. Nat. Hist.* 294, 1–240.
- FAO, 2007. The State of the World's Forests. [Ftp.fao.org/docrep/fao/009](http://ftp.fao.org/docrep/fao/009). FAO, Rome, Italy.
- Faria, D.R., Laps, R., Baumgarten, J., Cetra, M., 2006. Bat and bird assemblages from forests and shade cacao plantations in two contrasting landscapes in the Atlantic Forest of southern Bahia, Brazil. *Biodivers. Conserv.* 15, 587–612.
- Faruk, A., Belabut, D., Ahmad, N., Knell, R.J.Y., Garner, T.W.J., 2013. Effects of oil-palm plantations on diversity of tropical anurans. *Conserv. Biol.* 27 (3), 615–624.
- Felsenstein, J., 1985. Confidence limits on phylogenies with a molecular clock. *Syst. Zool.* 34, 152–161.
- Forest Stewardship Council, 2002. FSC Principles and Criteria, FSC Document 1.2, Oaxaca.
- Fraver, S., Wagner, R.G., Day, M., 2002. Dynamics of coarse woody debris following gap harvesting in the Acadian forest of central Maine, U.S.A. *Can. J. For. Res.* 32, 2094–2105.
- Frost, D.R., 2018. Amphibian Species of the World: an Online Reference. Version 6.0 (26/02/2018). Electronic Database accessible at <http://research.amnh.org/herpetology/amphibia/index.html>. American Museum of Natural History, New York, USA.
- Frost, D.R., Grant, T., Faivovich, J., Bain, R.H., Haas, A., Haddad, C.B.F., De Sá, R.O., Channing, A., Wilkinson, M., Donnellan, S.C., Raxworthy, C.J., Campbell, J.A., Blotto, B.L., Moler, P., Drewes, R.C., Nussbaum, R.A., Lynch, J.D., Green, D.M.Y., Wheeler, W.C., 2006. The amphibian tree of life. *Bull. Amer. Mus. Nat. Hist.* 297,

- 1–370.
- Giraud, A.R., Povedano, H., Belgrano, M.J., Krauczuk, E., Pardiñas, U., Miquelarena, A., Ligier, D., Baldo, D., Castelino, M., 2005. Status da biodiversidade da Mata Atlântica de Interior da Argentina. In: Galindo-Leal, C., de G. Cámara, I. (Eds.), *Mata Atlântica, biodiversidade, ameaças e perspectivas*. Belo Horizonte: Fundação SOS Mata Atlântica. Conservação Internacional. Centro de Ciências Aplicadas a Biodiversidade, pp. 160–180.
- Guerry, A.D., Hunter, J.E., 2002. Amphibian distributions in a landscape of forest and agriculture: an examination of landscape composition and configuration. *Conserv. Biol.* 16, 745–754.
- Gutiérrez-Lamus, D., Serrano, V.H., Ramírezpinilla, M.P., 2004. Composición y abundancia de anuros en dos tipos de bosque (natural y cultivado) en la cordillera Oriental de Colombia. *Caldasia* 26, 245–264.
- Haddad, C.F.B.Y., Prado, C.P.A., 2005. Reproductive modes in frogs and their unexpected diversity in the Atlantic forest of Brazil. *Bioscience* 55 (3), 207–217.
- Harmon, M.E., Sexton, J., 1995. Water balance of conifer logs in early stages of decomposition. *Plant Soil* 172, 141–152.
- Hazell, D., Cunningham, D.L., Mackey, B.Y., Osborne, W., 2001. Use of farm dams as frog habitat in an Australian agricultural landscape: factors affecting species richness and distribution. *Biol. Conserv.* 102, 155–169.
- Heyer, W.R., Donnelly, M.A., McDiarmid, R.W., Hayek, L.A.C., Foster, M.S., 1994. *Monitoring Biological Diversity: Standard Methods for Amphibians*. Smithsonian Institution Press.
- Hillers, A., Veith, R.Y., Rodel, M.O., 2008. Effects of forest fragmentation and habitat degradation on West African leaf-litter frogs. *Conserv. Biol.* 22 (3), 762–772.
- Hsieh, T., Ma, K.Y., Chao, A., 2016. Inext: an R package for rarefaction and extrapolation of species 553 diversity (Hill numbers). *Methods Ecol. Evol.* <http://dx.doi.org/10.1111/2041-210X.12613>.
- ICPB, 1992. Putting biodiversity on the map: global priorities for conservation. ICPB. International & IUBS, Cambridge, Cambridge, UK, pp. 1–11.
- IUCN, 2017. 2017 IUCN Red List of threatened species. <http://www.iucnredlist.org> (Accessed Dec 2017).
- Jackson, R.B., Banner, J.L., Jobbágy, E.G., Pockman, T., Wall, D.H., 2002. Ecosystem carbon loss with woody plant invasion of grasslands. *Nature* 418, 623–626.
- Klomp, N., Grabham, C., 2002. A Comparison of the Avifaunal Diversity on Native Hardwood Plantations and on Pastureland in a Forested Area. Report No. 164. Johnstone Centre, Charles Sturt University, Albury, NSW.
- Kruskal, J., 1964. Multidimensional scaling by optimizing goodness of fit to nonmetric hypothesis. *Psychometrika* 29, 1–27.
- Kwet, A., Márquez, R., 2010. Sound guide of the calls of frogs and toads from southern Brazil and Uruguay, Alosa, sons de la natura, Barcelona, Spain. — *Fonoteca Zoológica* (Museo Nacional de Ciencias Naturales, CSIC), Madrid, Spain.
- Laclau, P., 1994. La conservación de los recursos naturales renovables y el hombre en la selva Paranaense. *Boletín técnico fundación Vida Silvestre Argentina* 20, 139.
- Lantschner, M., Rusch, V., Peyrou, C., 2008. Bird assemblages in pine plantations replacing native ecosystems in NW Patagonia. *Biodivers. Conserv.* 17, 969–989.
- Le Maitre, D.C., Versfeld, D.B., Chapman, R.A., 2000. The impact of invading alien plants on surface water resources in South Africa: a preliminary assessment. *Water SA* 26, 397–408.
- Ligier, H.D., 2000. Caracterización geomorfológica y edáfica de la provincia de Misiones. Informe para Fundación Vida Silvestre Argentina. Instituto Nacional de Tecnología Agropecuaria (INTA), Corrientes.
- Lindenmayer, D.B., Franklin, J.F., 2002. *Conserving Forest Biodiversity: a Comprehensive Multiscaled Approach*. Island Press, Washington.
- Lips, K.R., Reaser, J.K., Young, B.E.Y., Ibañez, R., 2001. Amphibian monitoring in Latin America: a protocol manual. *Herpetol. Circul.* 30, 1–116.
- Lynch, J.D., Suarez-Mayorga, A.M., 2011. Clave ilustrada de los renacuajos en las tierras bajas al oriente de los Andes, con énfasis en Hyllidae. *Caldasia* 33, 235–270.
- Machado, I.F., Moreira, L.F.B., Maltchik, L., 2012. Effects of pine invasion on anurans assemblage in southern Brazil coastal ponds. *Amphibia-Reptilia* 33, 227–237.
- Magura, T., Tothmeresz, B., Bordan, Z., 2000. Effects of nature management practice on carabid assemblages (Coleoptera: Carabidae) in a non-native plantation. *Biol. Conserv.* 93, 95–102.
- Marsden, S.J., Whiffin, M., Galetti, M., 2001. Bird diversity and abundance in forest fragments and Eucalyptus plantations around an Atlantic forest reserve, Brazil. *Biodivers. Conserv.* 10, 737–751.
- Matteucci, D.S., Colma, A., 1982. Metodología para el estudio de la vegetación. Secretaría General de la Organización de los Estados Americanos, Washington, D. C. 168p.
- Nuttle, T., 1997. Densimeter bias? Are we measuring the forest or the trees? *Wildl. Soc. Bull.* 25, 610–611.
- Oksanen, J., Blanchet, F.G., Kindt, R., Legendre, P., O'Hara, R.B., Simpson, G.L., Solymos, P., Stevens, M.H., Wagner, H., 2011. *Vegan: Community Ecology Package*. R package version 1.17-9.
- Pardini, R., Faria, D., Accacio, G.M., Laps, R.L., Marianno-Neto, E., Paciencia, M.L.B., Dixo, M., Baumgarten, J., 2009. The challenge of maintaining biodiversity in the Atlantic forest: a multi-taxa conservation assessment of specialist and generalist species in an agro-forestry mosaic in southern Bahia. *Biol. Conserv.* 142, 1178–1190.
- Pardini, R., Souza, S.M., Braga-Neto, R.Y., Metzger, J.P., 2005. The role of forest structure, fragment size and corridors in maintaining small mammal abundance and diversity in an Atlantic forest landscape. *Biol. Conserv.* 124, 253–266.
- Parris, K.M., Lindenmayer, D.B., 2004. Evidence that creation of a *Pinus radiata* plantation in south-eastern Australia has reduced habitat for frogs. *Acta Oecol.* 25, 93–101.
- Parris, K.M., 2004. Environmental and spatial variables influence the composition of frog assemblages in sub-tropical eastern Australia. *Ecography* 27, 392–400.
- Parris, K.M., McCarthy, M.A., 1999. What influences the structure of frog assemblages at forest streams? *Aust. J. Ecol.* 24 (5), 495–502.
- Pawson, S.M., Brockerhoff, E.G., Meenken, E.D., Didham, R.K., 2008. Non-native plantation forests as alternative habitat for native forest beetles in a heavily modified landscape. *Biodivers. Conserv.* 17, 1127–1148.
- Petranka, J.W., Brannon, M.P., Hopey, M.E., Smith, C.K., 1994. Effects of timber harvesting on low elevation populations of southern Appalachian salamanders. *For. Ecol. Manage.* 67, 135–147.
- Pineda, E., Halfpeter, H., Moreno, C., Escobar, F., 2005. Transformación del bosque de niebla en agrosistemas cafetaleros: Cambios en las diversidades alpha y beta de tres grupos faunísticos. In: Halfpeter, G., Soberón, J., Koleff, P., Melic, A. (Eds.), *Sobre Diversidad Biológica: El significado de las diversidades alpha, beta y gamma*. M3monografías 3er Milenio, vol. 4. SEA, CONABIO, Grupo DIVERSITAS & CONACYT, Zaragoza – España, pp. 177–190.
- Pineda, E., Halfpeter, G., 2004. Species diversity and habitat fragmentation: frogs in a tropical montane landscape in Mexico. *Biol. Conserv.* 117, 499–508.
- R Development Core Team. 2011. R: A Language and Environment for Statistical Computing Version 2.13.1. R Foundation for Statistical Computing, Vienna, Austria. Available at: <http://www.r-project.org>.
- Raman, R.T.S., 2006. Effects of habitat structure and adjacent habitats on birds in tropical rainforest fragments and shaded plantations in the Western Ghats, India. *Biodivers. Conserv.* 15, 1577–1607.
- Ribeiro-Júnior, M.A., Gardner, T.A.Y., Ávila-Pires, T.C.S., 2008. Evaluating the effectiveness of herpetofaunal sampling techniques across a gradient of habitat change in a tropical forest landscape. *J. Herpetol.* 42, 733–749.
- Richardson, D.M., 1998. Pines as invaders in the southern hemisphere. In: Richardson, D.M. (Ed.), *Ecology and Biogeography of *Pinus pinus**. Cambridge University Press, Cambridge, pp. 450–473.
- Richardson, D.M., Williams, P.A., Hobbs, R.J., 1994. Pine invasions in the Southern Hemisphere: determinants of spread and invasibility. *J. Biogeogr.* 21, 511–527.
- Rossa-Feres, D.C., Nomura, F., 2006. Characterization and taxonomic key for tadpoles (Amphibia: Anura) from the northwestern region of São Paulo State, Brazil. *Biota Neotropica* 6 (1), 1–26.
- Schmitz, D.C., Simberloff, D., Hofstetter, R.H., Haller, W., Sutton, D., 1997. The ecological impact of non indigenous plants. In: Simberloff, D., Schmitz, D.C., Brown, T.C. (Eds.), *Strangers in Paradise: Impact and Management of Non indigenous Species in Florida*. Island Press, Washington, DC, pp. 39–61.
- Silvano, D.L., Segalla, M.V., 2005. Conservation of Brazilian Amphibians. *Conserv. Biol.* 19, 653–658.
- Simberloff, D., Nuñez, M.A., Ledgard, N.J., Pauchard, A., Richardson, D.M., Sarasola, M., Van Wilgen, B.W., Zalba, S.M., Zenni, R.D., Bustamante, R., Peña, E., Ziller, S.R., 2010. Spread and impact of introduced conifers in South America: lessons from other southern hemisphere regions. *Austral. Ecol.* 35, 489–504.
- Skelly, D.K., Werner, E.E., Cortwright, A.N.D.S.A., 1999. Long-term distributional dynamics of a Michigan amphibian assemblage. *Ecology* 80, 2326–2337.
- Stotz, D.F., Fitzpatrick, J.W., Parker III, T.A., Moskowitz, D.F., 1996. *Neotropical Birds: Ecology and Conservation*. University Chicago Press, Chicago.
- Straneck, R., Varela de Olmedo, E., Carrizo, G., 1993. *Catálogo de voces de anfibios argentinos*. Tomo I. LOLA. Buenos Aires.
- U.S. Environmental Protection Agency, 2002. *Methods for evaluating wetland condition: using amphibian in bioassessments of wetlands*. Washington, DC, EPA-822-R-02-022.
- Urbina-Cardona, J.N.Y., Londoño-Murcia, M.C., 2003. Distribución de la comunidad de herpetofauna asociada a cuatro áreas con diferente grado de perturbación en la Isla Gorgona, Pacífico colombiano. *Revista de la Academia Colombiana de Ciencias Exactas, Físicas y Naturales* 27, 105–113.
- Vitousek, P.M., 1990. Biological invasions and ecosystem processes: towards an integration of population biology and ecosystem studies. *Oikos* 57, 7–13.
- Vitt, L.J., Cadwell, J.P., 2009. *Herpetology*, 3ra Edición. Academic Press, Burlington.
- Wake, D.B., Vredenburg, V.T., 2008. Are we in the midst of the sixth mass extinction? A view from the world of amphibians. *Proc. Nat. Acad. Sci. USA* 105, 11466–11473.
- Yan, E.R., Wang, X.H., Huang, J.J., Zeng, F.R., Gong, L., 2007. Long-lasting legacy of forest succession and forest management: characteristics of coarse woody debris in an evergreen broad-leaved forest of Eastern China. *For. Ecol. Manage.* 252, 98–107.
- Zaninovich, S.C., Fontana, J.L., Gatti, M.G., 2016. Atlantic Forest replacement by non-native tree plantations: Comparing aboveground necromass between native forest and pine plantation ecosystems. *For. Ecol. Manage.* 363, 39–46.
- Zar, J.H., 1984. *Biostatistical Analysis*, 2nd ed. Prentice-Hall Inc., Englewood Cliffs, NJ.
- Zimmerman, B.L., 1994. Audio strip transects. In: Heyer, W.R., Donnelly, M.A., McDiarmid, R.W., Hayek, L.A.C., M.S. Foster, (Eds.), *Measuring and Monitoring Biological Diversity. Standard Methods for Amphibians*. Smithsonian Institution Press, Washington & London, 364 pp (pp. 92–97).
- Zurita, G.A., Bellocq, M.I., 2010. Spatial patterns of bird community similarity: bird responses to landscape composition and configuration in the Atlantic forest. *Landscape Ecol.* 25, 147–158.
- Zurita, G.A., Bellocq, M.I., 2012. Bird assemblages in anthropogenic habitats: identifying a suitability gradient for native species in the Atlantic Forest. *Biotropica* 44 (3), 412–419.
- Zurita, G.A., Rey, N., Varela, D.M., Villagra, M., Bellocq, M.I., 2006. Conversion of the Atlantic Forest into native and exotic tree plantations: effects on bird communities from the local and regional perspectives. *For. Ecol. Manage.* 235, 164–173.