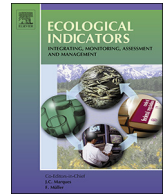




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Ferns as potential ecological indicators of edge effects in two types of Mexican forests

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ABSTRACT

The edge effect impacts the richness, composition, and abundance of plant species, as a result of environmental changes caused by forest fragmentation. Ferns are considered important indicators of environmental quality because they are directly affected by these alterations. The purpose of this study was to analyze the influence of the edge effect on ferns of two tropical lowland rainforests and two montane forests in Mexico. We sampled twelve 10 × 10 m plots at the edge and forest interior of each site, summing up 96 sampling units. In each plot, fern species were inventoried, and vegetative, microclimatic and edaphic parameters were collected. We registered a total of 51 terrestrial fern species, distributed in 33 genera and 17 families. The edge effect was considerably stronger at the montane forest sites, which had the highest fern richness and abundance in the forest interior, but experienced strong alterations of the fern species composition at the forest edge. This reduction of species richness and abundance was accompanied by significant changes in the environment at the montane forest sites, where canopy openness, temperatures, and wind speed increased, and soil moisture and air humidity significantly decreased. However, at the lowland rainforest sites the edge effect only weakly affected the abiotic environment and fern species composition. Half of the fern species were good indicators of forest interior or forest edge. We conclude that edge effects on ferns are more pronounced in montane forests than in lowland rainforests. Thus, conservation efforts to protect ferns should prioritize on threatened montane forests that are the hotspots of fern diversity, yet without ignoring lowland rainforests that contain a less diverse but distinct fern flora. Because many ferns were good indicators of the forest alterations caused by edge effects, we suggest using them as tools to evaluate the success of restoration efforts on habitat quality along forest edges.

1. Introduction

Forest fragmentation is a major cause of biodiversity loss and species extinction at a local or global scale (Bierregaard et al., 1992; Turner, 1996; Laurance and Bierregaard, 1997; Tabarelli et al., 1999). It is strongly influenced by the edge effect, since fragmentation causes abrupt ecological changes on forest remnants (Murcia, 1995). Haddad et al. (2015) revealed that approximately 20 percent of the world's remaining forest area is within 100 m of an edge, and more than 70 percent of the planet's forest areas are within 1 km from the forest edge. According to Laurance et al. (2001), there is a decrease in biological flows (e.g. inter-patch movement of organisms) as the degree of contrast between adjacent habitats and the physical flows increase. Forest fragmentation diminishes the internal heterogeneity of habitats and their natural resources, increasing the area exposed to the edge effect (Murcia, 1995; Metzger, 1999; Fahrig, 2003; Ribeiro et al., 2009).

Microhabitats differ between the edge and the interior of forest

remnants, and some of the most evident edge effects are closely linked to lower relative humidity and higher light intensity, temperature and wind exposure (Kapos, 1989; Laurance, 1991; Murcia, 1995; Tonhasca, 2005). The increasing amount of sunlight at the forest edge is the most important of these factors, because it is directly linked to lower air humidity and higher temperature (Murcia, 1995). Wind speed and light intensity are higher at the forest edge due to the lack of adjacent trees, which act as natural barriers in the forest interior (Silva et al., 2011). Litter deposition also increases along forest edges, making plant communities more vulnerable to fires (Laurance et al., 2002).

Although the edge effect has been extensively studied, few contributions in the Neotropics have investigated how ferns respond to these effects (Paciencia and Prado 2004; 2005; Barros et al., 2006; Carreño-Rocabado, 2006; Silva et al., 2011, 2014; Pereira et al., 2014; Bergeron and Pellerin, 2014; Silva and Schmitt, 2015). Nevertheless, ferns are considered good ecological indicators of habitat loss and fragmentation. Grime (1985) has shown that many fern species do not

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tolerate microclimatic changes resulting from the edge effect.

Mexican montane forests have a scattered natural distribution, which is now even more reduced due to anthropogenic factors (Challenger, 1998). It is considered one of the least common and most threatened vegetation types, covering less than 1% of the national territory (González-Espinosa et al., 2012), primarily by changes in land use (Gual-Díaz, 2014). The montane forest is also considered the richest vegetation type for ferns, comprising more than 60% of the species richness of Mexican ferns (Tejero-Díez et al., 2014). On the other hand, lowland rainforests have a potential area of approximately 10% of the national territory (INEGI, 2003), but due to habitat destruction, over-exploitation of natural resources and livestock farming, it occupies less than half of its original area (INEGI, 2005). About 40% of Mexican fern species are found in this vegetation type (Tejero-Díez et al., 2014). Mexico harbors 1,014 fern and lycophyte species, mainly in the three more humid Southeastern states: Oaxaca, Chiapas, and Veracruz, with over 500 species each (Martínez-Salas and Ramos, 2014). The state of Veracruz harbors 562 fern species (Tejero-Díez et al., 2011) of which 328 species are terrestrial.

Many groups of organisms (e.g., beetles, understory plants) have been investigated for their suitability as indicators of disturbance (Kranabetter et al., 2017). Ferns are considered important ecological indicators because they are susceptible to environmental changes (Ferrer-Castán and Vetaas, 2005; Zuquim et al., 2008; Silva et al., 2011; Silva and Schmitt, 2015), such as moisture (Kessler et al., 2001; Page, 2002; Kessler et al., 2011) and light availability (Sharpe and Mehlreter, 2010), and some species even accumulate heavy metals such as arsenic from contaminated soils (Chang et al. 2009). Certain species from the forest interior are restricted to specific microenvironmental conditions because of their susceptibility to anthropogenic forest alterations (Senna and Kazmirczak, 1997). On the contrary, some fern species that are adapted to disturbance can be favored by the edge effects and may even colonize new forest edges (Richardson and Walker, 2010).

The general objective of this study was to analyze the impact of the edge effect on terrestrial fern communities in two montane forests and two lowland rainforests. Specific objectives were to detect the edge effect (1) on species richness, composition and abundance of ferns and (2) on forest structure (i.e., number and DBH of trees, canopy openness, and herb cover), edaphic factors, and microclimatic variables; (3) to identify fern species as ecological indicators of forest interior or edge; and (4) to compare the edge effect between montane forests and lowland rainforests.

2. Methods

2.1. Study area

All sites were located in the state of Veracruz, Mexico (Fig. 1B). The montane forest site 1 (MF1) (18° 33' 14"N and 95° 09' 24"W, 1,200 m asl) was located in the municipality of San Andrés Tuxtla, Ejido 18° de Mayo, at the base of the volcano San Martín Tuxtla. The site MF2 (19° 31' 28"N and 97° 00' 36"W, 1,650 m asl) was located in the municipality of San Andrés Tlalnahuayocan. Both montane forest sites are characterized as montane cloud forests with a temperate oceanic climate without a dry season and a warm summer (Cfb, Peel et al., 2007).

The lowland rainforest site 1 (RF1, 18° 34' 58"N, 95° 03' 50"W, 110 m asl) and site 2 (RF2, 18° 35' 14"N, 95° 04' 36"W, 170 m asl) are both located at the edge of the biological reserve of the Tropical Biological Field Station "Los Tuxtlas", in the municipality of San Andrés Tuxtla (Fig. 1B), and have a tropical rainforest climate (Af) with average temperature of the coldest month above 18°C and monthly rainfall of at least 60 mm (Peel et al., 2007).

The four selected forest fragments were within the same type of environmental matrix: extensive pastures for livestock. At each site, we drew two parallel transects of 500 m length, at 5 m and 100 m distance from the forest edge. We randomly selected 12 points along each

transect, which were at least 10 m apart, and established a total of 24 plots of 10 × 10 m at each site. A total of 96 plots were sampled (4 sites × 12 replicates × 2 conditions: forest edge and interior) between March 2016 and April 2017 (Fig. 1A).

For each plot, we sampled the abundance of terrestrial ferns. Species were identified based on specialized bibliography, comparisons with herbarium vouchers and the help of specialists. We followed the taxonomic classification for families and genera of PPG I (2016). Fertile representative specimens were collected and deposited in the herbaria PACA (Brazil) and XAL (Mexico).

Additionally, we estimated the percentage cover of rocks, soil, herbs and leaf litter in each plot, according to the scale proposed by Braun-Blanquet (1979). We also counted the number of trees in the plot, measured their DBH and estimated the slope of the terrain in 5° steps. Temperature, air humidity, and wind speed were recorded during a cloudless day at hourly intervals along 12 consecutive hours (from 6 am to 6 pm) at the edge and in the interior of each site. These climate data were measured with a portable digital thermo-hygro-anemometer (Instruterm-Thal-300, model 0211), mounted on a 0.75 m high tripod (VF WT-3111). On the same day, hemispheric photos were taken at the center of each plot, using a Sony H5 camera equipped with a Raynox Digital fisheye lens, model DCR-CF 85 Pro. The camera was placed at 1 m from the ground, leveled and pointed to North (García et al., 2007). The images were analyzed for relative canopy openness using *ImageJ* 1.5i.

Soil moisture and depth of the litter layer were also measured at three points within each plot. Soil moisture content was measured using an AT Delta-T Device HH2 version 4.0 Moisture Meter. Litter depth (in cm) was measured using a ruler.

2.2. Data analyses

We performed a non-metric multidimensional scaling (NMDS) on Bray-Curtis similarity matrices of presence-absence (p/a) and abundance data of ferns to analyze the edge effect on the arrangement of plots of the two montane forest and two lowland rainforest sites. The stress values of NMDS graphs indicate the fit of the multidimensionality on the 2-dimensional plots. To detect significant changes in the floristic composition among fern communities between forest edge and interior, we run a similarity analysis (ANOSIM), which is an analogue to a standard univariate ANOVA, and performs a statistical test with 999 randomized permutations on p/a data to calculate the statistic R global with their corresponding probability p (Clarke and Gorley, 2002). To identify the fern species that contributed most to the similarity within groups, as well as to the dissimilarity between groups, we used a similarity percentage analysis (SIMPER). All three analyses were performed in the software PRIMER-E (Clarke and Gorley, 2002) version 5.2.9.

To determine the edge effect at each site, we made parametric (t) or non-parametric tests of variance (U) for each biotic and abiotic factor at a significance level of 5%, depending on data normality (Shapiro-Wilk's test). A Principal Component Analysis (PCA) was performed to investigate the correlation between abiotic parameters at each site and which separate best forest edge and interior. PCAs were performed in the software PAST (Hammer et al., 2001), version 3.14. A BIO-ENV procedure was performed to detect which ordination of samples based on similarities of environmental parameters matches best the ordination of samples based on similarities of biotic parameters (PRIMER-E (Clarke and Gorley, 2002) version 5.2.9). For these procedures, similarities of Bray-Curtis species abundances between samples were square-root transformed and normalized Euclidean similarities of abiotic variables between samples were square-root transformed (air humidity, canopy openness, and soil moisture) or used without further transformation (litter depth, temperature, wind speed).

Species-accumulation curves with 95% confidence intervals were drawn with *EstimateS* 8.2 (Colwell, 2005). Edge effects on fern

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