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Assessment of Land Use and Neotropical Herpetofauna Along Steep Gradients of Elevation in an Ecuadorian Ramsar Wetland Site #1143

by

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Assessment of Land Use and Neotropical Herpetofauna Along Steep Gradients of Elevation in an Ecuadorian Ramsar Wetland Site #1143

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Abstract

Using a comparative ecological approach, over the course of 18 days at the transition from a particularly wet to dry season in 2010, I assessed herpetofaunal assemblages and related abiotic parameters (i.e., photosynthetically active radiation, specific conductance, temperature and coarse woody debris) between contiguous forest and human impacted areas along three paired transects across the steep elevation gradient at Laguna de Cube, Ramsar site # 1143. Visual encounter surveys were used to capture herpetofauna with species being processed (e.g., weight, digit length, photographed) and identified to the lowest taxonomic level possible. After evaluating transect data for pooling (i.e., no significant difference in abiotic parameters relative to elevation or land cover, hypotheses were evaluated statistically using Chi Square and Kruskal Wallis, and adjusted for multiple comparisons, with an a priori $\alpha \leq 0.10$. I depart from convention due to the rarity of the region and accelerating human impacts. A total of 37 species (22 amphibians and 15 reptiles) were captured over 144 hours of direct sampling representative of day and night (n=4), 28 of which are newly described for Laguna de Cube, with three that have IUCN status of near threatened or endangered . As hypothesized, species richness and diversity were significantly greater in the forest than in impacted habitats [i.e., 30 forested versus 21 impacted species; χ^2 (2, N = 68) = 46.267, p = 8.9809E-11]. Similarly, abiotic conditions differed significantly by land cover with human impact exceeding forest analogs in 8 of 13 parameters (e.g., PAR; Top_F v. Top_I= , H = 27.6 df = 5, p = 0.005075) , while forests had significantly greater coarse woody debris [i.e., CWD_F = 150,731.97 kg/ha v. CWD_I = 47,819.97 kg/ha; χ^2 (1, N = 198,550.97) = 135.26, p = 2.897E-31]. Of the species collected several may serve as indicators of biotic integrity with *H. pellucens* serving as an indicator of degraded human modified land cover occurring in all of the 3 human impact transects and occurring at all elevations. Additionally, I observed morphological anomalies possibly indicative of anthropogenic habitat pollutants, with a majority of these species occurring in impacted environments routinely sprayed with pesticides. Conversely, several species may serve as indicators of native habitat affinity including *E. boulengeri* and *H. fallaciosus* both of which are described as forest obligates with risk of extirpation due to forest conversion. It should be noted that four species are not yet identified. When compared to similar herpetofaunal studies (n=6), my richness and diversity estimates meet or exceed those in the primary literature in five of the six comparisons.

Table of Contents

List of Tables

List of Figures

Figure 11a. Mean PAR for forested and impacted sampling stations represented by elevation pairs (Kruskal-Wallis: Low$_F$ v. Low $_I$= ; H = 27.6 df= 5, p = 0.07612; Mid$_F$ v. Mid$_I$= ; H = 27.6 df= 5, p = 0.07612; Top$_F$ v. Top$_I$= , H = 27.6 df= 5, p = 0.005075). Dark grey represents impacted sites and light grey represents forested sites. Significant differences identified by an asterix (*).

Figure 11b. Mean Temperature for forested and impacted sampling stations represented by elevation pairs (Kruskal-Wallis: Low$_F$ v. Low$_I$; H = 15.98 df= 5, p = 0.1689; Mid$_F$ v. Mid$_I$; H = 15.98 df= 5, p = 0.01291; Top$_F$ v. Top$_I$; H = 15.98 df= 5, p = 0.005). Dark grey represents impacted sites and light grey represents forested sites. Significant differences identified by an asterix (*).

Figure 11c. Mean Conductivity for forested and impacted sampling stations represented by elevation pairs Kruskal-Wallis: Low$_F$ v. Low$_I$; H = 16.07 df= 5, p = 0.02002; Mid$_F$ v. Mid$_I$; H = 16.07 df= 5, p = 0.04533; Top$_F$ v. Top$_I$; H = 16.07 df= 5, p = 0.1275). Dark grey represents impacted sites and light grey represents forested sites. Significant differences identified by an asterix (*).

Figure 11d. Shows the CWD class comparison between forested and impacted sampling stations represented by CWD size class (Kruskal-Wallis: 1; H = 19.79 df= 5, p = 0.4209) (2; H = 19.79 df= 5, p = 0.5204) (3; H = 19.79 df= 5, p = 0.8696) (4; H = 19.79 df= 5, p = 0.09746). Dark grey represents impacted sites and light grey represents forested sites. Significant differences identified by an asterix (*).

Figure 12a. Mean PAR for forested and impacted sampling stations at TP2 represented by elevation. Dark grey represents impacted sites and light grey represents forested sites.

Figure 12b. Mean Temperature for forested and impacted sampling stations at TP2 represented by elevation pairs. Dark grey represents impacted sites and light grey represents forested sites.

Figure 12c. Mean Conductivity for forested and impacted sampling stations at TP2 represented by elevation pairs. Dark grey represents impacted sites and light grey represents forested sites.

Figure 13a. Mean PAR for forested and impacted sampling stations at TP3 represented by elevation pairs. Dark grey represents impacted sites and light grey represents forested sites.

Figure 13b. Mean Temperature for forested and impacted sampling stations at TP3 represented by elevation pairs. Dark grey represents impacted sites and light grey represents forested sites.

Figure 13c. Mean Conductivity for forested and impacted sampling stations at TP3 over represented by elevation pairs. Dark grey represents impacted sites and light grey represents forested sites.

List of Symbols and Abbreviations

º	Degrees
‘	Minutes
α	Alpha
β	Beta
γ	Gamma
BBS	Bilsa Biological Station
Bd	*Batrachochytrium dendrobatidis*
cm	Centimeters
CWD	Coarse Woody Debris
E	Endangered
e.g.	For example
F	Forest
FAO	Food and Agriculture Organization of the United Nations
FSC	Forest Stewardship Council
GFRA	Global Forest Resource Assessment
GPS	Global Positioning System
H1	Hypothesis 1
H2	Hypothesis 2
ha	Hectare
I	Impact
i.e.	That is
IUCN	International Union for Conservation of Nature
L	Low
LC	Least concern

MCER	Mache Chindul Ecological Reserve
m	Meters
M	Middle
n	Number
N	North
N/A	Not applicable
NT	Near threatened
PAR	photosynthetically active radiation
PAST	Paleontological Statistics
SVL	Snout to vent length
T	Top
TP1	Transect Pair 1
TP2	Transect Pair 2
TP3	Transect Pair 3
$TP1_F$	Transect Pair 1 forested transect
$TP1_I$	Transect Pair 1 impacted transect
$TP2_F$	Transect Pair 2 forested transect
$TP2_I$	Transect Pair 2 impacted transect
$TP3_F$	Transect Pair 3 forested transect
$TP3_I$	Transect Pair 3 impacted transect
VES	Visual Encounter Survey
WAAS	Wide Area Augmentation System
W	West

Introduction

The future of global biodiversity is of considerable concern (Wilson 1992, Morris 1995, Morris and Heidinga 1997, Sax and Gaines 2003) and it has been suggested that we are in the midst of a sixth mass extinction event (Wake and Vredenburg 2008, Barnosky et al. 2011). Increased rates of human population growth, as well as the commensurate increases in land conversion, domesticated animal waste, and the evolution of invasive species and exotic pests continue to pressure Earth's remaining natural resources (Morris 1995, Rands et al. 2010). Today, extinction rates are hundreds of times greater than those of pre-human existence (Pimm et al. 1995, Brooks et al. 2006) and continue to greatly exceed speciation, making biodiversity exceedingly vulnerable at local and global levels (Sax and Gaines 2003). It is believed that global biodiversity will dramatically decrease (Rands et al. 2010), with some estimates suggesting that as many as half of the earth's species will become extinct (Sax and Gaines 2003, Barnosky et al. 2011). Regions where high levels of biodiversity and endemism overlap with humans are especially susceptible to extirpation and extinction (Wilson 1993), and Neotropical forests serve as a constant reminder of this.

According to Global Forest Resources Assessment (GFRA 2010) there has been a net loss of the Earth's forests of 5.2 million hectares (ha) per year since 2000 with the highest deforestation rates occurring in South America (GFRA 2010). With agricultural expansion being the leading land use change associated with 96% of all deforestation (Geist and Lambin 2002), Mosandl et al. (2008) reports that in the Neotropics, Ecuador has the highest deforestation rate, making it a global concern for conservation. Attempts have been made to provide incentives for the use of less severe silviculture techniques by the Forest Stewardship Council (FSC). There has been limited success in Brazil, Bolivia, and Uruguay showing significant areas of land protection (Ebeling and Yasue 2009, FSC 2011). However, Ecuador has not capitalized on such incentives, with less than 0.2% of the country's forests (16,686 ha) currently FSC certified (Ebeling and Yasue 2009, FSC 2011). Unfortunately, such certification incentives are rarely effective in countries where the forestry laws are largely ignored and seldom enforced (Ebeling and Yasue 2009). In Ecuador, timber permits are commonly sold and reused illegally, while the National Forestry Agency receives only enough funding from the Ministry of the Environment for four forestry control posts staffed by a total of eight personnel for the entire country (Ebeling and Yasue 2009). It should be noted that Ecuador, unlike neighboring Bolivia, has no mobile field units (Ebeling and Yasue 2009). In contrast, Bolivia has 5 forestry control posts and 16 mobile field units staffed by 171 personnel (Ebeling and Yasue 2009). In a country where the average size of the remaining forest fragments is only 50 ha (Ebeling and Yasue 2009), these illegal and negligible forest protection measures further contribute to the 1.2% of forest loss (137,000 ha) continuing to take place in Ecuador annually (FAO 2010). This is especially troubling in regions that have high biodiversity, rapidly increasing human population, and nonexistent or lax protection such as in the Ecuadorian lowland coast.

With imminent threats to global biodiversity, it is critically important to make timely and intelligent decisions on how and where to focus conservation efforts (Myers et al. 2000, Wilson et al. 2006). Myers (1988) suggested concentrating conservation efforts in areas that could potentially retain the highest biodiversity. The idea of designating areas as biodiversity "hotspots" was created, where great numbers of species, often endemic, exist in densely populated areas, often facing the greatest threat to their existence (Ginsberg 1999, Myers 1988, Myers 1990, Myers et al. 2000, Myers 2003). Initially 10 hotspots were proposed (e.g., Madagascar, the Atlantic coast of Brazil, Western Ecuador, and Colombian Chocó), but that number later expanded to include 25 areas of greatest conservation concern (Myers 1988, Myers et al. 2000). Myers et al. (2000) reported that 44% of plant species and 35% of vertebrate species were contained in these areas combined, while the total area of all hotspots accounted for only 1.4% of the Earth's land surface. Six of these hotspots are located in under-developed tropical countries where conservation and protection are limited (Laurance 1997, Myers et al. 2000, Ferraro 2001, Wunder 2007); the Choco Darien of western Ecuador is one such region.

The Choco Darien provides habitat to more than 9,000 plant and 1,625 vertebrate species with described endemics of the region accounting for nearly 1% of global plant species and 1.5% of total species, though this area remains largely unprotected (Myers et al. 2000, Mittermeier 2011). Once consisting of 260,000 km^2 of primary vegetation, the Choco has been reduced to only 63,000 km^2 (Myers et al. 2000). Though still remarkably diverse, nearly 75% of the terrestrial landscape remains unprotected and vulnerable (Myers et al. 2000). Coastal Western Ecuador has, over the past few decades, been touted as a biodiversity hotspot due to extremely high levels of diversity and presence of many endemic species (Gentry and Dodson 1987, Gentry 1992, Myers et al. 2000). For example, Valencia et al. (1991) reported that 1561 individuals of 473 tree species in 187 genera and 54 families existed in a single 1 ha plot of tropical forest. This region is also known for increasing human perturbations and the resultant threats to rare and endemic species (Neill et al. 1999).

In order to effectively conserve and manage biodiversity, baseline conditions must be quantified to include an assessment of ecological integrity to better understand how humans modify natural habitats (Keddy et al. 1993, Rader 2001, Karr and Yoder 2004) with indices developed to respond to various impairments from the range of observed human disturbance. Selection of useful indicators of disturbance (i.e., differences in biological attributes between disturbed and least disturbed sites; Fausch et al. 1990) and indices developed for biotic integrity assessment when ecosystem complexity and ecological attributes are too difficult or expensive to accurately, and directly, measure (Landres et al.1988; Hilty and Merenlender 2000). While many species have been proposed as potential indicators of biotic integrity, few clearly correlate indicator status to actual changes in the environment or habitat suitability (Hilty and Merenlender 2000, Azevedo-Ramos et al. 2005). Until multi-taxa biotic integrity models are developed (Diffendorfer et al. 2007), [building on the work of others (e.g., Marsh and Pearman 1997, Gardner 2001, Lima et al. 2001, Pous et al. 2011)] reptiles and amphibians, as a functional guild, offer great potential as indicators because of their important trophic roles (Whiles et al. 2006), linking of terrestrial and aquatic ecosystems (Lips et al. 2006, Urbina-Cardona 2008), varied reproductive and thermoregulatory strategies (Huey and Slatkin 1976, Duellman and Treub 1986, Navas 1996), and susceptibility to seemingly insignificant environmental changes (Duellman and Trueb 1986, Blaustein et al. 2001).

As natural habitat rapidly decreases, being replaced by human-dominated landscapes, conservation has become crucial for the biotic integrity of herpetofauna and their associated ecosystems (Gibbons et al. 2000, Urbina-Cardona 2008). Unfortunately, of the herpetofaunal research published through 2008 (n=12,353), only 5% make specific conservation recommendations (Urbina-Cardona 2008) associated with lands that are not currently protected by governmental or non-governmental sources, and therefore are at extreme risk of future degradation and loss. As Gardner et al. (2007a) suggest, the level of research being conducted on habitat change affecting herpetofauna is not proportionally balanced to the level of threat that amphibians and reptiles experience from ecological scales ranging from microtopographic variation (e.g., humidity variation) to landscape (land use change; Gibbons et al. 2000, Gardner et al. 2007c, Greene 2009, Sigala-Rodríguez and Greene 2009).

Deforestation and resulting habitat fragmentation affect herpetofauna by altering microhabitat environmental conditions that limit chemical cue detection and movement (Ferguson 1971, Beebee and Griffiths 2005); conversion of natural habitat has long been cited as being detrimental to herpetofauna (Cushman 2005, Gardner et al. 2007c). In addition, sediments, erosion, and pollutants resulting from agricultural pursuits have been shown to directly alter the physiology of reptiles and amphibians (Mann et al. 2009). Populations of reptiles and amphibians, especially endemics and habitat specialists, have declined due to invasive species outcompeting, hybridizing with, and consuming native species (Beebee and Griffiths 2005, Pasachnik et al. 2009). Other factors such as disease (e.g., chytridiomycosis; Weldon et al. 2004, Skerratt et al. 2007) and climate change (Gibbons et al. 2000, Carey and Alexander 2003) and the resulting interactions across ecological scales are impacting herpetofauna in complex and novel ways that further jeopardize biodiversity (Bosch et al 2006). For example, using the modest 2˚C increase in temperature predicted by 2050 (Thomas et al. 2004), ecosystems will potentially experience irreversible herpetofaunal assemblage changes ranging from simple range shifts to increased competition at elevation boundaries.

On a local scale, herpetofauna are threatened by deforestation, fragmentation, and pollution. While consequences of habitat loss and fragmentation are well documented (Cushman 2005, Gardner et al. 2007c) herpetofauna, especially amphibians, are exceedingly vulnerable to small changes in abiotic gradients such as temperature, conductivity, and photosynthetically active radiation (PAR; Overcash et al. 1981, Lund et al. 1999, Galloy and Denoel 2010). Additionally, it has been shown that pollutants (e.g., herbicides) cause a wide range of problems from drastic changes in pH to anuran feminization (Beebee and Griffiths 2005, Mann et al. 2009). There is also evidence that some amphibians use olfactory and humidity cues to reach breeding areas, making habitat degradation especially problematic (Ferguson 1971). Similarly, fragmentation creates barriers to dispersal and foraging for many reptiles and potentially increases energy expenditure in thermoregulation and mate selection (Huey and Slatkin 1976, Duellman and Treub 1986, Navas 1996).

At intermediate levels, herpetofauna are suffering from threats resulting from distribution along elevation gradients, issues with dispersal, exotic species, direct exploitation (consumption), and mortality through ignorance (e.g., removing snakes from populated areas, Fauth et al 1989, Gibbons et al. 2000, Whitaker and Shine 2000, Beebee and Griffiths 2005). The ecological changes found along elevation gradients have been reported to be a significant determinant of community structure and diversity (Rahbeck 1995, Lomolino 2001, McCain 2005). Changes in species abundance along elevation gradients may reveal important aspects of habitat preference; elevation is commonly correlated with a number of climatic variables known to constrain the distribution of amphibians and reptiles, especially in the tropics (Janzen 1967, Meik and Lawing 2008). Additionally, because humans occupy the landscape in a nonrandom pattern, typically occupying and developing flat lowlands, the consequences of human impact may be more severe for species with narrow requirements (Janzen 1967, Hofer et al. 1999). In turn this affects high elevation herpetofauna as community changes shift up gradient (Hofer et al. 1999, Janzen 1967).

The spread of disease and climate change are among the leading factors contributing to reptilian and amphibian declines on a global scale (Gibbons et al. 2000, Carey et al. 2003 Weldon et al. 2004, Bosch et al. 2006, Skerratt et al. 2007). Chytridiomycosis, which is caused by the chytrid fungus *Batrachochytrium dendrobatidis* (Bd), has been linked to amphibian declines worldwide since its discovery in the early 1900's (Weldon et al. 2004, Skerratt et al. 2007). Climate change and its associated variables (i.e., temperature change, seasonal change, and fluctuation in precipitation) affect herpetofauna directly (Gibbons et al. 2000, Carey et al. 2003) and have been investigated as factors that may be increasing the prevalence of chytridiomycosis (Bosch et al. 2006).

Despite international concerns having been raised (Neill et al. 1999), there are only a few sites within Ecuador that have received any conservation protection or adequate long-term study: Cotacachi-Cayapas and the Mache-Chindul Ecological Reserve (MCER), with MCER containing the last important remnants of coastal wet tropical forest, characterized by high species diversity and endemism (Clark et al. 2006, Ortega-Andrade et al. 2010). One organization stands out, primarily for its international cooperation and its focus on a particular ecosystem type, Ramsar.

The Ramsar Convention is an intergovernmental treaty created to help maintain ecological character and promote sustainable use of wetlands and their territories through education and outreach (Ramsar, 2012). Ramsar was created in 1971 and within the last 40 years has grown to include 2,062 wetlands covering an area of 197,258,541 ha in 163 countries (Ramsar 2012). Today Ecuador has 14 sites recognized by Ramsar for their ecological significance and research value (Ramsar 2012). These sites along with a few privately funded reserves offer protection for only a small portion of Ecuador's biologically rich landscape (Jatun Sacha 2011). One of these privately owned reserves is the 3,300 ha Bilsa Biological Station (BBS) operated by Fundacion Jatun Sacha (Ortega-Andrade et al. 2010, Jatun Sacha 2011). Jatun Sacha is a private, non-profit organization created in 1989 with the goal of promoting the conservation of Ecuador's biodiversity (Jatun Sacha 2011). One reported shortcoming to herpetofaunal research conducted at Bilsa is the failure to adequately sample the steep elevation gradients found within the region (Ortega-Andrade et al. 2010).

Ramsar recognizes one unique lacustrine wetland system in the coastal mountains of Ecuador: Laguna de Cube. The Laguna de Cube wetland complex typifies a biodiversity hotspot (Myers et al. 2000) with high reported levels of terrestrial biodiversity (Ramsar 2012), minimal but expanding human influence, and international recognition (i.e., Ramsar status, Conservation International 1991), yet it receives only minimal protection by a small non-governmental organization (NGO), Kaiman Fondacion. Additionally, Laguna de Cube forms the headwaters of the Rio Cube, which flows through the entire Mache Chindul Ecological Reserve, making conservation of Laguna de Cube a priority to human health, regional biodiversity, and everything downstream, including one of the last protected reserves in Ecuador.

The overall goal of my research is to examine herpetofaunal composition and structure along gradients of elevation and land use using a comparative ecological approach and test the following hypotheses:

H1: Forested environment will have greater abundance, species richness, and Shannon Weiner diversity than human-impacted environment.

H2: Abundance, richness, and Shannon Weiner diversity will differ as elevation changes in these environments.

Methods

Site Description

Laguna de Cube is a freshwater lacustrine wetland in the Esmeraldas Province of northwestern Ecuador (00°24’N 079°39’W; Figure 1). In 2002 it was added to the Ramsar list of wetlands of international importance due to uniqueness (i.e., the only natural permanent lotic system in the coastal region of Ecuador), hydrologic importance (e.g., only source of potable water, ongoing aquaculture), and unique biotic communities (e.g., 23 species of mammals and 40 species of birds; Ramsar report 2001). Laguna de Cube consists of a 22.41-ha, permanent freshwater oxbow lake surrounded by a variety of aquatic, semi-aquatic and flooded habitats divided into three areas: coastal area with rooted vegetation, limnic area of open water with floating hydrophilic vegetation, and deep benthic area. Together, these areas support a rich biotic community as well as a working landscape for forestry, grazing, and agriculture (Kaiman Foundation personal communication, Tosso 2009, Ramsar 2012). Current threats to the region are largely the result of human expansion and include erosion, hydrologic alteration and water removal, and disturbance to vegetation through cutting, clearing, and the application of herbicides and pesticides. Average temperatures range from 25 to 27^{o}C with annual precipitation ranging from 2000mm to 3000mm (Ramsar 2012). Since designation on the Ramsar list, minimal research has been conducted. As a deep floodplain lacustrine complex, Laguna provides critical ecosystem functions and services to the surrounding landscape and as such is important to the downstream Mache Chindul Ecological Reserve.

Sampling Design

Herpetofauna were sampled in summer 2010 (June 10th – June 28th) during a transition from a particularly rainy wet season to the dry season at Laguna de Cube. Building on Tosso (2009), three transect pairs (TP1, TP2, and TP3) were established along the eastern border of Laguna de Cube extending from the water's edge to the top of the local watershed. Each transect pair consisted of a transect through the forest (e.g., $TP1_F$) and a transect through a human-impacted area (e.g., $TP1_I$; Figure 2) with a substantial elevation gradient, such as 250 – 630 m above sea level (asl), within the local watershed. Although the forested areas were largely contiguous and mature (i.e., trees over 30m with a diverse understory, evidence of coarse woody debris, and no direct evidence of harvest), they are most likely secondary and not old-growth forests (Kaiman Foundation, personal communication). Human-impacted areas were directly and substantially modified by humans (i.e., >75% human-dominated land cover) and were representative of the three dominant human lands uses in the region: passion-fruit (*Passiflora edulis*) agriculture ($TP3_I$), a recent clear-cut that removed viable timber and left slash ($TP2_I$), and cattle rangeland ($TP1_I$). Although these transects varied in specific land use and current management, they are representative of the human modifications that occur within the Neotropics and serve as replicates of human-impacted areas for my study. Transects ranged in length from 525m to 1450m and were not significantly different by treatment (i.e., mean forested length = 975m and mean impacted length = 928m; t (2) = 0.975, p = 0.216) making subsequent comparisons defensible. Clearing for rangeland ($TP1_I$) began in the 1980s with annual and ongoing expansion using fire, herbicides, and grazing management (rotational grazing, personal observation). Initial clearing of $TP3_I$ (passion fruit agriculture) began in the spring of 2004 with a mechanical clear-cut followed by intensive fire management, weekly mechanical removal of graminoids, and intensive application of herbicides (e.g., atrazine, glyphosate) and several pesticides (e.g., diazion). Direct management has declined over time to quarterly applications of herbicides/pesticides and annual burns (Kaiman Fondacion, personal communication). $TP2_I$ was clear-cut during spring 2010 with harvestable timber removed and slash left onsite.

At each transect, three sampling stations were established along the elevation gradient from lake level to hilltop (low = ~300 m asl, mid = ~ 450 m asl, and top = ~600 m asl) to assess herpetofaunal assemblages and abiotic parameters. Sampling start points were randomly assigned by elevation and transect, then varied to capture all possible combinations of elevation and land use with transect pairs being sampled concurrently. Elevation, slope, and geographic location of each sampling station (n=18) were recorded using a handheld GPS with WAAS. Following Tosso (2009), direct and indirect sampling methods were used to examine the composition and structure of reptilian and amphibian assemblages.

Arboreal species (e.g., hylids) were sampled using small diameter bamboo tree-frog shelters (n=5) at each sampling station (Appendix 1). Tree frog shelters were capped at the bottom to allow water to accumulate; a small (0.5 cm) drainage hole was drilled approximately 5 cm from the bottom to control water level. Shelters were constructed and placed on trees at each sampling station with open ends elevated to 80 to 90 cm (Tosso 2009).

Time-constrained visual encounter surveys (VES) were conducted at each sampling station during the day and night. Searchers (n=4) were positioned equidistant 10m uphill from the centroid of a sampling station with searchers moving slowly through the site over 15 minutes until they had passed 20m through the plot, following standard practices outlined in Urbina-Cardona et al. (2006, Appendix 2). To aid in capture, a 25 m black plastic drift fence was placed at the lower boundary of each sampling station (Appendix 2). Captured organisms were photographed and identified to the lowest level possible using taxonomic keys and local experts. Snout-to-vent length (SVL) and digit length were measured for each individual. Photographs were taken of dorsal and ventral sides of torso and legs, tympanum, dorsolateral fold, eyes (to assess orientation and pupil shape), and front and rear feet (to assess toe number, shape, length and orientation; Glaw and Vences 1994).

Measures of abiotic variables, known to influence the distribution of herpetofauna, were collected once and included quantification of soil temperature, conductivity, photosynthetically active radiation (PAR), and coarse woody debris (CWD; Table 1). Following Uranowski et al. (2002), CWD was measured along a 12 m transect at each sampling station by recording the circumference of branches that intersected the transect line in four size classes (0 – 6 mm, 6 – 25 mm, 25 – 76 mm, >76 mm). These measurements were converted to tons/ha for statistical comparison by land use and elevation.

Biotic and abiotic data were assessed by land cover (i.e., forest and human impact) and elevation class to support pooling for subsequent hypotheses testing. Statistical assumptions could not be satisfied for parametric tests (normalcy violated; Shapiro Wilk W, $p<0.05$) despite multiple transformation attempts. Therefore, I used nonparametric equivalents: Fisher's exact test and/or Chi-squared for count data following Steen and Gibbs (2004), Kruskal-Wallis, adjusted for multiple comparisons, for all other data (Kruskal and Wallis, 1952, Ruxton and Beauchamp, 2008). Rarefaction was used to assess sampling adequacy and compare to other Neotropical studies while Shannon Wiener diversity, using SW diversity t-test (Magurran 2004), was compared between forest and human impacted environments in paleontological statistics software (PAST, Hammer and Harper 2001). Unbalanced/insufficient replication and/or limited sample size precluded statistical assessment among transect pairs, so I describe herpetofauna data and abiotic parameters across transect pairs as related to land cover and elevation, reporting general patterns of occurrence and trends relative to my stated hypotheses. Throughout this research I used an a priori alpha of ≤ 0.10 due to the global rarity and regional importance of this lacustrine wetland complex (Ramsar 2001, 2012). It is preferable to commit a false-change error (Type I) than a missed-change error (Type II) since Type II errors can potentially overlook irreversible ecological changes such as extirpation/extinction (Noss and Cooperrider 1994). Because I depart from convention (i.e., $\alpha \leq 0.05$), p values are reported in text, tables, and figures for individual interpretation.

Indicator species analyses

Following the general guidelines Azevedo-Ramos et al. (2005), and Barr (2008) I developed indices of habitat quality and biodiversity using a “guild” approach (sensu Blocksom 2002), my capture data divided between reference (i.e., forested) and disturbed (i.e., impacted) sites, IUCN (2012) reports, and associated literature. IUCN protection status (e.g., endangered, least concern) was given to each species and supplemented by other sources (e.g., CITES), in addition to assigning each species to a habitat class (i.e., forest or impacted and then weighted by the prevalence of my capture data). With this information, a preliminary Index of Biotic Integrity (IBI) was created to assign a quantifiable value to each species. Three categories were assigned to each species (Risk Factor, Abundance Weight Value, Habitat Distribution Factor) and summed to create this value. Risk factor represents a description of a species’ rarity and distribution [Critically Endangered (10 pts), Endangered (8 pts), Vulnerable (6 pts), near threatened (4 pts), undescribed (2 pts), least concern (1 pt)]. Abundance Weight Value represents how abundant a species was [Cumulative Abundance < 6 (1 pt), 6-10 (2 pts), 11-15 (3 pts), 16-20 (4 pts), 21-25 (5 pts), 26-30 (6 pts), 31-35 (7 pts), 36-40 (8 pts), 41-45 (9 pts), > 45 (10 pts)]. Habitat Distribution Factor indicates a species' preferred environment [Forested preference (1 pt), No preference (0 pts), Impacted preference (-1 pt)]. For example, a species of least concern pursuant IUCN, that occurs in a forested environment but never in an impacted environment, with high total abundance (i.e., > 45 individuals) would receive a very high score and potentially serve as an excellent indicator of an un-impacted forested environment.

Results

Two hundred eighty-nine individuals representing 37 species were collected over 18 days during the transition from the wet to dry season along six transects equally divided between secondary forest and human impacted areas along an elevation gradient. Each transect was sampled four times during the day and four times during the night and data were pooled across all sampling events. Pooling of data was supported by the lack of significant difference by treatment [e.g., abundance and richness did not significantly differ ($p \geq 0.10$) among forested transects]. Visualization of sampled-based rarefaction suggested that sampling was adequate (Figures 3a and 3b).

Of species captured, 22 were amphibians (all anurans of 6 families) and 15 were reptiles (11 families) consisting of eight lizard, five snake, one turtle, and one caiman species (Table 2). Of these, a total of 15 species of anurans and 13 species of reptiles were previously undocumented at Laguna de Cube (Tosso 2009). While most species encountered are considered common (i.e., least concern following IUCN Redlist 2012), two are listed as near threatened (*Pristimantis celator* and *Silverstoneia nubicola)*, one as endangered (*Pristimantis pteridophilus*), and four of the specimens captured have yet to be identified to the species level.

Fourteen species (38%) were captured in both forested and impacted habitats while sixteen species (43%) were unique to forest and seven (19%) to impacted sampling stations (Table 3). Jaccard's similarity index shows that forest and human impacted area were strongly dissimilar (0.378) though generally transect pairs were more similar than unpaired treatment comparisons (Table 4). Abundance differed significantly by treatment (154 individuals captured in forest vs. 135 in impacted sampling stations, [χ^2 (8, N = 289) = 548.53, p = 2.6835E-113], with two of the transect pairs (i.e., TP1 and TP3) showing a similar pattern (Figure 4).

TP1 had more individuals in forest than the human impacted area (40 versus 10, respectively; Figure 4). TP3 followed this pattern with more individuals captured in forests than human-impacted sampling stations (78 versus 71, respectively; Figure 4) In contrast, TP2 showed the reverse, with greater abundance in impacted sampling stations than forest (54 versus 35 respectively; Figure 4).

TRANSECT PAIR DESCRIPTIONS OF HERPETOFAUNA

Transect Pair 1

Fifty individuals were captured with twice as many species captured in the forest (12) as compared to the impacted analog (6) in transect pair one, with the low-elevation contributing greatest to overall richness (8 species in forest vs. 2 in impacted; Figure 5). The top-elevation sampling station recorded six species in the forested sampling station and one in the impacted sampling station (Figure 5). The mid-elevation in this pair recorded more species in the impacted sampling station (3) than the forested (2) (Figure 5). The most abundant species captured in the impacted transect of TP1 was *Pristimantis achatinus* with five captured at the mid-elevation. This species is listed by IUCN redlist as “least concern.” The most abundant species captured in $TP1_F$ was *Epipedobates boulengeri* with 18 total captures, one at the top-elevation and 17 at the low-elevation. This species is also listed by IUCN redlist as “least concern.”

Transect Pair 2

Transect pair 2 resulted in the capture of 91 individuals with more species captured in the human-impacted sampling stations (15) than the forested sampling stations (11). Similarly, this was also the case for two of the three elevation classes (i.e., $TP2_I$ low and $TP2_I$ mid). The impacted low-elevation sampling station produced 11 species while the forested low-elevation sampling station produced five (Figure 6). The impacted top-elevation sampling station produced four species while the forested top-elevation sampling station produced two (Figure 6). The mid-elevation sampling stations for this pair showed more species in the forest than in the impacted transect with seven in the forest and three in the impacted sampling stations (Figure 6). Like TP1, the most abundant species captured for both forested and impacted, was *Pristimantis achatinus* with 31 individuals collected in the impacted transect (i.e., 15 at the low-elevation, 10 at the mid-elevation and six at the top-elevation) and 22 in the forested (i.e., two at the low-elevation, 17 at the mid-elevation, and three at the top-elevation). This species is listed by IUCN redlist as “least concern.”

Transect Pair 3

Transect pair 3 showed the greatest difference in species richness between the two treatments (19 species in the forested and 5 in the impacted transect) across the entire elevation gradient. Fourteen species were collected at the low, nine at the mid, and three at the top compared to three, three, and two at the respective impacted elevations (Figure 7). In $TP3_I$ *Hypsiboas pellucens* was the most abundant species with a total of 65 individuals captured, 19 at the low, 15 at the mid, and 31 at the top elevations. For the $TP3_F$, *Epipedobates boulengeri* was the most abundant species encountered with 19 individuals collected, 17 at the low and two at the mid elevations. This species is listed by IUCN redlist as “least concern.”

Treatment Comparisons

Species richness differed significantly by treatment overall with forests having greater species richness than human impacted sites [i.e., 30 versus 21 species; χ^2 (2, N = 68) = 46.267, p = 8.9809E-11]. When assessed by transect pair, forests generally had higher species richness with TP1 exhibiting the lowest overall richness with 16 species ($TP1_I$ = 6, $TP1_F$=12, 2 shared), followed by TP2 with 18 species ($TP2_I$=15, $TP2_F$=11, 8 shared), and TP3 with 21 species ($TP3_I$=5, $TP3_F$=19, 3 shared; Figure 8).

Shannon Wiener diversity was significantly different between habitat types with forested transects having higher diversity than human-impacted environments (i.e., SWH'_F= 2.35 versus SWH'_I = 1.68; diversity t-test, p=0.0005), though only TP3 exhibited the same significant pattern in Shannon Weiner diversity by transect pair. When examined by sampling station, six of nine forest sites (67%) had greater Shannon Wiener diversity than impacted sampling stations (Table 5). Transect Pair 2 was a notable exception, where greater overall abundance and richness were associated with human-impacted sampling stations (Figures 4 and 6).

Overall, herpetofaunal abundance and richness were greatest at low elevation, although this pattern was not significant (Kruskal-Wallis, H = 1.904 df= 5, p = 0.3861; Figure 9 and Kruskal-Wallis, H = 2.333 df= 5, p = 0.1266; Figure 10). At each elevation, with the exception of $TP1_I$ mid, $TP2_I$ low, and $TP2_I$ top, forested sampling stations had greater richness than the human-impacted analogs (Figures 5, 6, and 7). Low-elevation forests contributed 21 species to overall richness, followed by middle (~450m asl) and top-elevations (~ 600m asl), which yielded 14 and 8 species respectively, however these differences were not statistically significant.

ABIOTIC PARAMETERS

Treatment comparisons by abiotic parameters (i.e., PAR, temperature, conductivity, photosynthetically active radiation (PAR), and CWD) were evaluated as important constraints to the distribution and composition of herpetofauna (Owens et al. 2008, Uranowski et al. 2003) though edaphic data from TP1 were lost in travel and excluded from subsequent analysis.

Abiotic parameter data including PAR, temperature, conductivity, and CWD compared statistically by habitat type and elevation class (Figures 11a-13c, Table 6). PAR between habitat types ranged from 0.88nm-50.35nm across the forest elevation gradient and from 119.65nm-1667.3nm across the human-impacted gradient (Table 6) and was significantly greater in all of the impacted sampling stations and all elevations, when compared to forests (Kruskal-Wallis: $\mathrm{Low_F}$ v. $\mathrm{Low_I}$; H = 27.6 df = 5, p = 0.07612; $\mathrm{Mid_F}$ v. $\mathrm{Mid_I}$; H = 27.6 df = 5, p = 0.07612; $\mathrm{Top_F}$ v. $\mathrm{Top_I}$; H = 27.6 df = 5, p = 0.005075, Figure 11a). Mean soil temperature also varied by treatment and elevation ranging from 23.5^{o}C-26.3 ^{o}C in the forests and from 24.3 ^{o}C-28.5 ^{o}C in human-impacted sampling stations (Table 6). While soil temperature did not differ between forested and impacted sampling stations in the low-elevation class, soil temperature was significantly higher in human-impacted sampling stations of mid and top-elevation classes when compared to forest analogs (Kruskal-Wallis: $\mathrm{Low_F}$ v. $\mathrm{Low_I}$; H = 15.98 df = 5, p = 0.1689; $\mathrm{Mid_F}$ v. $\mathrm{Mid_I}$; H = 15.98 df = 5, p = 0.01291; $\mathrm{Top_F}$ v. $\mathrm{Top_I}$; H = 15.98 df = 5, p = 0.005, Figure 11b). Specific conductivity was also significantly greater for human-impacted transects when compared to forested sampling stations at low and mid-elevations (Kruskal-Wallis: $\mathrm{Low_F}$ v. $\mathrm{Low_I}$; H = 16.07 df = 5, p = 0.02002; $\mathrm{Mid_F}$ v. $\mathrm{Mid_I}$; H = 16.07 df = 5, p = 0.04533) but was insignificant at the top-elevation ($\mathrm{Top_F}$ v. $\mathrm{Top_I}$; H = 16.07 df = 5, p = 0.1275; Figure 11c), Conductivity ranged from 0.6m/s-31.2m/s in the forest and from 17.4m/s-66m/s in the impacted sampling stations (Table 6).

Forests had greater CWD than human-impacted transects [$\mathrm{CWD_F}$ = 150,731.97 kg/ha v. $\mathrm{CWD_I}$ = 47,819.97 kg/ha; χ^2 (1, N = 198,550.97) = 135.26, p = 2.897E-31]. While CWD was typically greater in forests, only the largest size class (i.e., CWD 4, diameter >75mm) was significantly different between forested and human-impacted transects (Kruskal-Wallis: 1; H =

19.79 df = 5, p = 0.4209), (2; H = 19.79 df = 5, p = 0.5204), (3; H = 19.79 df = 5, p = 0.8696), (4; H = 19.79 df = 5, p = 0.09746; Figure 11d).

In TP2, PAR was greater in the impacted sampling stations than forested sampling stations at all elevations (Figure 12a). PAR ranged from 0.88nm-6.96nm in the forest and from 97.06nm-1667.3 nm in the impacted sampling stations. Mean air temperature is shown in Figure 12b and indicates higher temperatures at all elevations in the impacted sampling stations compared to the forested sampling stations of TP2. Temperature ranged from 24.3^{o}C-26.3^{o}C in the forest and from 26.5 ^{o}C-28.5^{o}C in the impacted sampling stations. Conductivity was higher at all elevations in impacted sampling stations compared to forested sampling stations for TP2 (Figure 12c). Conductivity ranged from 21.2m/s-31.2m/s in the forest and from 32.8m/s-47.3m/s in the impacted sampling stations. Represented in Table 7 are the CWD measurements listed by size class. Overall, forests had greater CWD than human-impacted transects in this transect pair (118,967.02 kg/ha v. 36,539.73 kg/ha).

Transect pair three followed the same trend as TP2. PAR as represented in Figure 13a is greater in impacted sampling stations compared to forested sampling stations. PAR ranged from 9.84nm-50.35nm in the forest and 119.65nm-488.3nm in the impacted sampling stations. Mean air temperature was significantly higher in impacted sampling stations than forested sampling stations (Figure 13b). Temperatures ranged from 23.5^{o}C-24.1^{o}C in the forest and from 24.3^{o}C-25.3^{o}C in the impacted sampling stations. Conductivity levels were greater in impacted sampling stations than in forested sampling stations but only significantly different at the low-elevation. Conductivity ranged from 0.6m/s-28.9m/s in the forest and from 17.4m/s-66m/s in the impacted sampling stations. CWD presence is greater in the forested sampling stations than in the impacted sampling stations for TP3 (Table 8,764.9 kg/ha v. 11,280.23 kg/ha).

INDICATOR SPECIES

Epipedobates boulengeri, *Hypodactylus fallaciosus*, *Pristimantis achatinus,* and *Pristimantis pteridophilus* all expressed a unique value in the index of biotic integrity (IBI) indicating they are suitable species for designating a habitat as an optimal forested environment (IBI value; 9, 8, 10, 8 respectively, Table 9). *Hypsiboas pellucens,* with an IBI value of -10, suggests it may indicate a level of human impact (Table 9). The remaining 32 species scored IBI values between -4 and 4 and as a result are not considered bioindicator species (Table 9).

Discussion

Over the course of 18 days, a total of 37 species (22 amphibians and 15 reptiles) representing 17 families and 289 individuals were collected from 144 hours of direct sampling divided equally between forest and impacted landscapes, along a steep elevation gradient, where I report significant differences in abundance, richness, and diversity relative to land use and elevation. While four species remain unidentified (3 Hylidae, 1 Leptodactylidae), I have substantially increased the species list by 28 (15 amphibians and 13 reptiles). In comparison, Tosso (2009) sampled Laguna de Cube two years prior and collected a total of 23 species. I report differences in species richness (37 versus 23) and total abundance (289 versus 200) compared to my overall findings suggesting that elevation contributed to richness and diversity with low elevation sampling stations contributing disproportionately. These findings lend support to the notion that community structure changes with elevation, especially in the tropics (Janzen 1967, Meik and Lawing 2007).

As hypothesized, species richness and diversity were significantly greater in forested environment than in impacted supporting the general notion that forests are more species-rich and diverse than human impacted sampling stations (Bell and Donnelly 2006, Gardner et al. 2007, Sodhi et al. 2008, Cisneros-Heredia et al. 2009, Hayes et al. 2010, Brown et al. 2012). Conversely, human impacted sampling stations had lower richness and diversity seemingly attributed to the difference in abiotic variables.

Abiotic variables may be responsible for differences in species richness among sampling stations. Overall impacted sampling stations had higher PAR, temperature, and conductivity as well as lower amounts of CWD than forested sampling stations. Although these parameters were not significantly different at every elevation, each comparison still followed this general pattern. Transect pair three showed a difference in each abiotic parameter tested (conductivity, temperature, light penetration, CWD) at every elevation. The difference in anthropogenically caused changes in physical environment is likely the cause of the significant variation in species richness between the two transects. Converting what was once forest into a passion fruit plantation drastically alters the soil conductivity, surface temperature, light penetration, and CWD present making this a less suitable habitat for most herpetofauna found at this wetland. Increased salinity, fertilizers, animal waste, and pest control application increase soil conductivity which are factors that may have been responsible for the increases in these variables observed in the current study (Overcash et al. 1981, Lund et al. 1999). Changes in local habitat (tree removal, cattle grazing, plantation, etc.) create different temperature gradients associated with forest and pasture edge effects linked to herpetofaunal habitat preference, specifically altering seasonal preferences (Lehtinen et al. 2003). CWD presence may benefit thermoregulation ability as it creates an easy way to escape heat and a lack of CWD would in turn create a problem in CWD free environments such as a pasture or plantation where temperatures are significantly higher (Whiles and Grubaugh 1996).

Transect pair 2 showed differences at nearly each elevation for conductivity, temperature, and PAR and still fit the pattern found in TP3. TP2 did not show a difference in CWD found. Unlike TP1 and TP3, TP2 did not show a greater richness in the forested transect. CWD was the only abiotic factor that did not differ between $TP2_F$ and $TP2_I$ suggesting that the amount of CWD present may be the leading factor in determining whether herpetofauna thrive in either of these types of local wetland watershed environments. CWD has been suggested to be useful for a number of herpetofaunal needs (protection from predation, thermoregulation, reproduction, feeding substrate) where more CWD is beneficial (Whiles and Grubaugh 1996). This may be the reason richness was high in only this impacted transect.

My preliminary index of biotic integrity (IBI) determined there were 5 total species suitable to be indicators of habitat quality and biodiversity. *Epipedobates boulengeri, Hypodactylus fallaciosus, Pristimantis achatinus,* and *Pristimantis pteridophilus* are all species whose presence suggests undisturbed natural forest environment as indicated by their risk factors, abundances, and habitat preferences combined. *Hypsiboas pellucens* presence on the reverse indicates that an environment has undergone human alteration or impact of some degree represented as a negative outlier due to their high abundance and preference for impacted environments. This simple index paired with habitats significantly differing in collected abiotic parameters offer a great value in selecting these species as a simple way of determining a habitats level of human impact at Laguna de Cube.

I report richness and diversity estimates that are comparable or exceed other tropical herpetofauna research often conducted over multiple years with significantly greater sampling effort (Table 10). For example, Vonesh (2001) sampled herpetofauna in Uganda at Makerere University Biological Field Station. Vonesh (2001) sampled fifty 5m x 5m plots for a period of three months and captured 18 species, 10 anurans and 8 reptiles offering comparable low richness. Wanger et al. (2010) sampled herpetofauna in Central Sulawesi, Indonesia. In this study a total of 31 plots were each sampled 6 times and resulted in 20 total species captured, 8 amphibians and 12 reptiles (Wanger et al. 2010). Urbina-Cardona et al. (2006) sampled herpetofauna in Mexico, under similar environmental conditions, reported 54 species (i.e., 21 amphibians and 33 reptiles) over the course of a year. While Gardner et al. (2007) found more species in primary (i.e., 22 amphibian 25 reptiles), their richness values were strikingly similar for secondary forest (14 amphibian and 15 reptiles) and plantations (5 amphibian and 15 reptiles) when compared to Laguna de Cube, though area sampled was far greater (average plot size comparison 1,687 ha-2,682 ha) based on assessment of rarefaction curves (Gardner et al. 2007). These data show that Laguna de Cube is exceptionally species rich, as I found as many amphibians with only one season of sampling. In contrast, Bell and Donnelly (2006) sampled herpetofauna in Costa Rica from Oct. 2003 to July 2004 and found 36 amphibian and 14 reptile species. These results offer nearly the same number of species collected in the current study suggesting the reptile species richness is particularly abundant at this site.

Bilsa Biological Station (BBS) which is located in the same geographic region only 15 km away from Laguna de Cube, experiences similar weather and human impacts (Ortega-Andrade et al. 2010). BBS conducted a six year survey (2004-2010) where they have cataloged 37 amphibian species and 72 reptile species within the 3,300 ha reserve. Comparing my study to BBS, we share 9 species of amphibians and 13 reptile species. This leaves 15 species (13 amphibian and 2 reptile) that are unique to Laguna de Cube. The major factor that may explain these differences is the presence of the permanent water body at Laguna de Cube. This body of water accounts for the presence of the two reptiles (*Caiman crocodiles* and *Chelydra serpentina acutirostris*) at Laguna de Cube not found at BBS. Many anurans require a body of water to complete the larval portion of their life stage which may be a reason these 13 species are not present at BBS (Duellman and Trueb 1986). It is also important to note that BBS is very large (3,300 ha) and has been sampled for many years while Laguna de Cube is small and has had very little sampling effort.

One of the glaring differences between my study and others (Table 10) is the scarcity of snakes found at Laguna de Cube compared to BBS. Although greater than Tosso (2009), I only collected 7 snakes representing 5 species while BBS reports 42 total species. I expect snake diversity at Laguna de Cube is higher than I was able to document due to witnessing a number of cryptically colored snakes that evaded capture, but this still falls short of the total richness at BBS. Without further research efforts I can only speculate why this is occurring, however many locals expressed their fear of snakes, and how they kill them whenever found. This fear is likely a result of injuries from encounters with *Bothrops asper* and *Lachesis acrochorda* and lead to the killing of many other harmless snake species. Another possible reason for not capturing a large number of snakes is their ability to sense low frequency ground vibrations allowing them to flea during sampling sessions (Heinen 1992).

Three of my 18 sampling stations were located in the same place as sampled by Tosso (2009). $TP1_F$ recorded 11 species when sampled by Tosso (2009) and 8 species in my study. This was the richest location in his study and two years later has declined. Our studies only share 3 species at this sampling station (*Basiliscus galeritus*, *Epipedobates boulengeri,* and *Hypsiboas pellucens*) however 5 species in his study have yet to be identified. The other two sampling stations we shared were $TP3_F$ and $TP3_I$ where I reported greater species richness at each. In Tosso (2009) $TP3_I$ did not result in the capture of the vastly abundant *Hypsiboas pellucens* documented in my study. I speculate that presence of this hylid in such high numbers is due to the changes in pesticide use in this location made since Tosso (2009) sampled. The chemical use has considerably declined since the sampling in 2008 (local personal communication).

The Coco-Darien forests of Ecuador (more recently referred to as Tumbes-Choco-Magdelena; Mittermeier et al. 2011) are among the world's most biologically diverse (Meyers et al. 2000, Shanee 2006, Mittermeier et al. 2011, Critical Ecosystem Partnership Fund 2012). The

Mache Chindul Ecological Reserve (est. 1996), in the Esmeraldas Province, has been recognized as an area rich in biodiversity (Neill et al. 1999, Ortega-Andrade et al. 2010), containing more than 11,600 species and the last significant remnants of tropical wet forest within the Choco. While several broad initiatives have been implemented for the region, there is currently no clear conservation management plan (Myers et al. 2000).

The ministry of the environment for Ecuador (MAE) is a national institution with the responsibility of creating environmental policy and coordinating strategies, projects, and programs directed toward ecosystem conservation and the sustainable use of the country's natural resources (MAE 2011). However, Ecuador's National Forestry Agency receives very little money from MAE, enough to fund only eight personnel for the entire country (Ebeling and Yasue 2009). In addition, the few Ecuadorian forestry permits that are issued are commonly sold and reused illegally with no apparent penalty (Ebeling and Yasue 2009), which contradicts the goals of the MAE. Other non-governmental organizations such as the Ramsar convention have tried to bring awareness to areas like these where biologically important habitats receive no governmental protection (Ramsar 2012). With no power, these organizations struggle to gain ground against largely unregulated resource degradation (Ebeling and Yasue 2009, Ramsar 2012).

Habitat loss is not the only result of deforestation, with habitat fragmentation, degradation and alteration problematic, by displacing species, altering home ranges (Cushman 2005, Gardner et al. 2007c), and influencing abiotic and biotic interactions. Habitat alteration leads to increased resource stress and dramatic changes in community structure including the introduction of invasive species which change the characteristics of predation and competition

(Beebee and Griffiths 2005). Furthermore, introduction of diseases, such as Chytridiomycosis (Chytrid) in amphibian populations, greatly impact native species at a global scale (Weldon et al. 2004, Skerratt et al. 2007), while predicted changes in weather patterns (e.g., precipitation) further compound ecosystem stress and threaten biotic integrity (Gibbons et al. 2000, Carey and Alexander 2003). This is especially worrisome to Neotropical anuran species with narrow environmental tolerances and highly specific breeding requirement particularly in species who engage in explosive breeding (1-14 days) (McCauley et al. 2000).

The MCER is also experiencing unprecedented human expansion (Advanced Conservation Strategies 2012), extreme poverty (e.g., annual per capita income < $4,500 US; CARE 2012), and minimal governmental support (Advanced Conservation Strategies 2012) though it contains the last remnants of tropical wet forest in the country. As more forest is lost and fragmentation increases, habitat change and edge effects will likely cause community composition and structure to change drastically (Murcia 1995, Bell and Donnelly 2006). Since MCER is largely isolated from other contiguous forest and exists in an ever increasingly hostile matrix, immigration rates are likely to decline potentially leading to local extirpation, following the general tenants of island biogeography (sensu MacArthur and Wilson 1963, 1967). Similarly, destruction of habitat islands may result in lower species richness and local extinction. Laguna de Cube, positioned at the entrance of MCER, offers an ideal opportunity to investigate the relationship between Neotropical biodiversity and the actions of humans.

Studies in Costa Rica (Bell and Donnelly 2006) and Mexico (Urbina-Cardona 2006) show complex community structure at the border of different habitats with clear differences in environmental conditions. Bell and Donnelly (2006) found that most herpetofauna were present in contiguous forests with significantly fewer in secondary forest and plantations. This pattern is also evident in northwestern Ecuador; Cisneros-Heredia et al. (2009) reported continued habitat loss and range restriction of the endemic glass frog (*Cochranella mache*). As forest area is replaced by agriculture and humans, it is extremely important that remaining forest patches are preserved, especially for habitat-sensitive species. Laguna de Cube has shown declines in species richness since Tosso's research (2009), with my data suggesting his most species rich site exhibiting a 30% reduction in richness, likely attributable to accelerating and continued human influence. Furthermore, habitat fragmentation and loss are of the highest concern in this area and should be the greatest conservation priority of the whole country (Sierra et al. 2002).

My project represents the first attempt to examine herpetofaunal composition relative to land use and elevation at Laguna de Cube. Findings were used as indicators for anthropogenic disturbance with a goal of both determining what species exist in the area and to use them as bioindicators of the local human impact in this important ecosystem. This project took place during the month of June at the end of the rainy season at Laguna and as a result only serves as a seasonal account of what species may inhabit this area year round. Sampling during other months may increase the richness for this area due to potential seasonal immigration and emigrations as well as species life stage patterns.

As with any logistically constrained study, there are several recommendations for improvement. One of the collection problems that should be addressed in further research at this site is the inclusion of juvenile anuran larval stages such as tadpoles. Due to the difficulty of identification and the extreme delicacy of this life stage, these individuals were not collected and processed, potentially leaving out additional species not classified with the collection of only terrestrial life stages. The use of timed group searches proved to be the most effective collection tool in this study and should be used for any future collections done at Laguna de Cube. The one improvement I may suggest for the group searches to maximize abundance is to remove the time limit and use methods similar to Vonesh (2001) where searches continued until 15 minutes have elapsed with no new captures. Although much more time consuming, this would ensure abundances are as high and data are as accurate as possible for this type of study. I suggest increasing the length of the entire sampling period and including captures from both wet and dry seasons. We continued to find new species on the last day of sampling suggesting that further collections would result in reports of greater richness and abundance. This would partly be solved by using the time constraints used by Vonesh (2001) suggested above, but would truly need seasonal collections for the most accurate account of the inhabitants at Laguna de Cube. As my sampling was conducted during a transition period from wet to dry season, edge effects may not be realized to their full extent as drastic changes between wet and dry season change habitat preference (Lehtinen et al. 2003).

Conclusions

<u>My time spent at Laguna de Cube has led me to the following conclusions;</u>

1- Herpetofauna prefer unaltered forested habitat.

2- Species richness and diversity appear to decrease as elevation increases. Many species may require a permanent water source, especially some anuran species reproductive patterns. This is evident as the greatest richness was found at sites at the lake level. Species distributed as elevation increases may require more specific environmental condition ranges not available at the lake margin. Also this distribution may be a result of many anurans having very specific reproductive processes that can only be achieved with conditions found at different elevations. Much of this remains speculation as collections were only gathered during one time of year and would likely change and shed more light on distribution if seasonal collections were done.

3- Habitat impact is indicated by abiotic variables. Human impact can be quantified by the resulting changes in abiotic variables associated with anthropogenic pressure. Parameters such as PAR, temperature, and conductivity all seem to be higher in areas experiencing human impact of some kind. CWD however, appears to be the leading driver of habitat selection as increases in CWD are positively correlated to species abundance and richness.

4- Not all species followed the overall patterns concluded from this study. The habitat created in the passion fruit plantation seems to be preferred by the sub adult life stage of *Hypsiboas pellucens*. This species was very abundant during night searches on these elevated vine systems running the length of this transect. Since these individuals appear to all be in a sub adult life stage, it seems this may be in response to a possible food source as a breeding preference can be ruled out at this age. This environment has seemed to create a type of microhabitat that this species is thriving in.

Overall Laguna de Cube seems to be a very diverse wetland ecosystem harboring numerous species of herpetofauna. These animals are in danger of losing this habitat to human invasion. With Laguna de Cube being the only lake in the coastal mountains of Ecuador, it is extremely important to mitigate this impact and restore this wetland as it is one of a kind. Losing this environment will likely cause extirpation of many species needing this lacustrine habitat to survive. If something doesn't change soon, then the fate of Laguna de Cube and its wildly diverse community of species may reach a state beyond repair and one of the world's unique wetland ecosystems may disappear.

Based on my research I suggest the following:

1. Increase herpetofauna research to include both wet and dry season data as habitat preference has been shown to differ by season (Lehtinen et al. 2003). Also I strongly recommend introducing research on the chytridiomycosis (Chytrid) fungus *Batrachochytrium dendrobatidis* (Bd) in anurans. Chytrid is linked to amphibian declines worldwide since its discovery in the early 1900's (Weldon et al. 2004, Skerratt et al. 2007). McCracken et al. (2009) found Chytrid present in 20 percent of the anuran captures in a study conducted in eastern Ecuador. There seems to be a number of studies looking at Chytrid presence in eastern Ecuador, Brazil, Costa Rica, and other South American countries (Lips et al. 2003, Carnaval et al. 2006, McCracken et al 2009), but there is a lack of information documenting whether it exists in western Ecuador. As this is a growing problem worldwide (Weldon et al. 2004), I feel that it is important for the future of this wetland to determine if it is present there.

2. Further study the area including investigation of the vast array of other taxa present at Laguna as a lack of study leaves classifications in this ecosystem undoubtedly underestimated. Minimal research has been documented at Laguna and limited to herpetofaunal and fish collections, leaving countless other flora and fauna unclassified here.

3. Introducing and carrying out a conservation management plan for human impact and development of this area with emphasis on mitigating disturbance of remaining forest fragments and restoration of areas already experiencing heavy anthropogenic stress. Following steps suggested by Margules and Pressey (2000), I suggest that Laguna de Cube should systematically tackle the process of implementing a conservation management plan through: 1) Further compiling data on biodiversity at this site, 2) Identifying and selecting their conservation goals, 3) Re-sampling areas where data has been collected and reviewing conservation areas at BBS, 4) Selecting specific areas to protect at Laguna de Cube, 4) Implementing these conservation strategies, and 6) Setting and tracking their conservation goals. A local community uses this area for subsistence purposes as it is the only lucustrene system in the coastal mountains in the entire country. Efforts are being made to market this area to ecotourists as a way to bring in revenue while maintaining the fractions of natural habitat still remaining. Steps are being taken to restore this area and the Kaiman Foundation have been recognized for this effort with being awarded the Green Globe Award by the World Wetland Network’s 2010 Wetland Globe Awards

4. Reach out for support and partnership with larger conservations groups with greater financial and connected resources such as Jatun Sacha, the conservation group running the successful nearby Bilsa Biological Station. The Kaiman Fundacion has been able to bring this wetland to light on not just a local but, with Ramsar and the World Wetland Network, a global awareness, however minimal conservation implementation has resulted. Partnering with a

larger organization may make it possible to enforce the management strategies that these global organizations stand for.

Tables

Table 1. Abiotic parameters and relevance for inclusion, quantified at replicated sampling stations along forested and human impacted transects over 18 days during the summer of 2010 at Laguna de Cube, Ramsar site #1143, Ecuador.

Abiotic Parameters	Units	Relevance
Temperature	Degrees C	Temp. changes associated with habitat
		alteration and herpetofaunal habitat preference.
Soil Conductivity	Meters/Sec.	Increases with salinity, fertilizers,
		animal waste, and pest control application
Photosynthetically Active Radiation (PAR)	nm	Indicates the amount of light reaching the
		Ground; important for thermoregulation.
Coarse Woody Debris (CWD)	Diameter in cm	Aids in protection from predation,
		thermoregulation, reproduction, feeding substrate.

Table 2. Species collected at Laguna de Cube, Ramsar site #1143, Ecuador over 18 days of summer 2010 reporting habitat type (i.e., forest or human impact), elevation class (i.e., low, middle, top), and IUCN Redlist conservation status(LC = Least Concern, NT = Near Threatened, N/A = Not Available, E = Endangered; IUCN 2012). Species newly described at Laguna de Cube indicated by an asterix (*).

	CLASS / Order / Family / Species		Forest/Impact	Elevation	Redlist Status
AMPHIBIA					
Anura					
	Bufonidae				
	1	*Rhinella margaritifera*	F	L	LC
	2	*Rhinella marina*	F,I	L,M,T	LC
	Craugastoridae				
	1	*Craugastor longirostris**	F,I	L,M	LC
	Dendrobatidae				

	1	*Epipedobates boulengeri*	F,I	L,M,T	LC
	2	*Silverstoneia nubicola**	F		NT
	Hylidae				
	1	*Dendropsophus* sp.*	F,I	L,T	N/A
	2	*Hypsiboas pellucens*	F,I	L,M,T	LC
	3	*Hypsiboas picturatus**	I	T	LC
	4	*Hypsiboas rosenbergi**	I	L	LC
	5	*Hypsiboas rufitelus*	F	L	LC
	6	*Hypsiboas* sp.*	F	L	N/A
	7	*Scinax ruber**	F	L	LC
	8	*Scinax* sp.*	F,I	L	
	9	*Scinax quinquefasciatus*	F	L	LC
	10	*Trachycephalus jordani**	I	M	LC
	Leptodactylidae				
	1	*Leptodactylus bolivianus**	F	L,M	LC
	2	*Leptodactylus* sp.*	F,I	L,M	N/A
	Strabomantidae				
	1	*Hypodactylus fallaciosus**	F,I	L,M,T	N/A
	2	*Pristimantis achatinus*	F,I	L,M,T	LC
	3	*Pristimantis celator**	I	L	NT
	4	*Pristimantis pteridophilus**	F	T	E
	5	*Pristimantis ridens**	F,I	L,M,T	LC
REPTILIA					
Crocodylia					
	Alligatoridae				
	1	*Caiman crocodilus*	F	L	LC
Squamata-Lacertilia					
	Corytophanidae				
	1	*Basiliscus galeritus*	F	L	N/A
	Gymnophthalmidae				
	1	*Alopoglossus festae**	F,I	L,M	N/A
	2	*Ptychoglossus gorgonae**	F	M	N/A
	Hoplocercidae				
	1	*Enyalioides heterolepis**	F,I	M,T	N/A
	Polychrotidae				
	1	*Anolis biporcatus**	I	M	N/A
	2	*Anolis granuliceps**	I	M	LC

	Sphaerodactylidae				
	1	*Lepidoblepharis buchwaldi**	F,I	L,T	N/A
	Teiidae				
	1	*Ameiva septemlineata**	I	L	N/A
Squamata-Serpentes					
	Boidae				
	1	*Boa constrictor imperator**	F	M	N/A
	Colubridae				
	1	*Coniophanes fissidens**	F	L	N/A
	2	*Tantilla melanocephala**	F	L	N/A
	Viperidae				
	1	*Bothrops asper**	F,I	L,M,T	N/A
	2	*Lachesis acrochorda**	F	M	N/A
Testudenes					
	Chelydridae				
	1	*Chelydra serpentine acutirostris**	F	L	N/A

Table 3. Summary of amphibians and reptiles captured over 18 days during the summer of 2010, at Laguna de Cube, Ramsar site #1143 including treatment collections (i.e., forested or impacted). Grey represents species presence. Complete species description and photograph contained in Appendix 3.

Species	**Impact**	**Forest**	**Species**	**Impact**	**Forest**
Alopoglossus festae	▨	▨	*Hypsiboas* sp.		▨
Ameiva septemlineata	▨		*Lachesis acrochorda*		▨
Anolis biporcatus	▨		*Lepidoblepharis buchwaldi*	▨	▨
Anolis granuliceps	▨		*Leptodactylus bolivianus*		▨
Basiliscus galeritus		▨	*Leptodactylus* sp.	▨	▨
Boa constrictor imperator		▨	*Pristimantis achatinus*	▨	▨
Bothrops asper	▨	▨	*Pristimantis celator*	▨	
Caiman crocodilus		▨	*Pristimantis pteridophilus*		▨
Chelydra serpentina acutirostris		▨	*Pristimantis ridens*	▨	▨
Coniophanes fissidens		▨	*Ptychoglossus gorgonae*		▨
Craugastor longirostris	▨	▨	*Rhinella margaritifera*		▨
Dendropsophus sp.	▨	▨	*Rhinella marina*	▨	▨
Enyalioides heterolepsis	▨	▨	*Scinax ruber*		▨
Epipedobates boulengeri	▨	▨	*Scinax* sp.	▨	▨
Hypodactylus fallaciosus	▨	▨	*Scinax quinquefasciatus*		▨
Hypsiboas pellucens	▨	▨	*Silverstoneia nubicola*		▨
Hypsiboas picturatus	▨		*Tantilla melanocephala*		▨
Hypsiboas rosenbergi	▨		*Trachycephalus jordani*	▨	
Hypsiboas rufitelus		▨			

Table 4. Jaccard's Similarity Index calculations for all transect comparisons using richness data collected at Laguna de Cube (scale 0-1). Of the transect pairs, transect pairs 1 and 3 showed the lowest similarity (0.125, 0.142857) while transect pair 2 showed the greater similarity (0.444444).

	TP1-F-tot	**TP1-I-tot**	**TP2-F-tot**	**TP2-I-tot**	**TP3-F-tot**	**TP3-I-tot**
TP1-F-tot	1	0.125	0.210526	0.421053	0.24	0.133333
TP1-I-tot		1	0.214286	0.166667	0.136364	0.1
TP2-F-tot			1	0.444444	0.2	0.066667
TP2-I-tot				1	0.307692	0.111111
TP3-F-tot					1	0.142857
TP3-I-tot						1

Table 5. Shannon Wiener diversity (H’) values for each transect pair by elevation. Greater values are shown in gray indicating 6 of the 9 transect site pairs found a greater diversity in forested environments. Entire transect values are also provided ($TP1_F$ H’ = 1.7433 versus $TP1_I$ H’ = 1.4975, p = 0.32755; $TP2_F$ H’ = 1.454 versus $TP2_I$ H’ = 1.7872, p = 0.2036; $TP3_F$ H’ = 2.3471 versus $TP3_I$ H’ = 0.434, p = 0.0001).

	TP1-F	TP1-I	TP2-F	TP2-I	TP3-F	TP3-I
Low	1.3437	0.6931	1.5498	1.8875	2.1561	0.385
Mid	0.6931	0.7963	1.1814	0.5661	1.6096	0.4438
Top	1.7479	0	0.5623	1.0027	0.995	0.2286
Transect	1.7433	1.4975	1.454	1.7872	2.3471	0.434

Table 6. All median data by elevation for PAR, Temperature, and Conductivity along with lowest and highest readings. Lowest and highest values along with overall mean for each transect are also listed. In each case impacted sites had greater values compared to forested sites.

	PAR (nm)		Temperature (°C)	
	Forest	Impacted	Forest	Impacted
Low elev.	8.4 (4.4-19.25)	321.05 (119.65-773.1)	24.6 (23.5-26.3)	25.5 (24.3-26.6)
Mid elev.	11.73 (0.89-50.53)	545.4 (345.7-1667.3)	24.15 (23.7-24.7)	25.95 (24.6-27.9)
Top elev.	7.86 (0.88-23.9)	237 (97.06-1298.5)	24.2 (23.9-24.6)	25.95 (24.7-28.5)
Lowest	0.88	119.65	23.5	24.3
Greatest	50.35	1667.3	26.3	28.5
Transect Mean	12.75	507.53	24.84	25.93
	Conductivity (m/s)			
	Forest	Impact		
Low elev.	10.95 (0.6-31.2)	34.3 (17.4-66)		
Mid elev.	22.25 (20-23.4)	39.15 (20.7-45.2)		
High elev.	27.85 (20.2-31)	33.2 (20.9-41.5)		
Lowest	0.6	17.4		
Greatest	31.2	66		
Transect Mean	20.25	36.08		

Table 7. Assessment of coarse woody debris collected from each forested and impacted sampling station. Size classes modified from Uranowski et al. (2003) and included: 0 mm-6 mm, 6 mm-25 mm, 25 mm-76 mm, >76 mm. Treatment comparisons (i.e., forest v impact) were assessed by statistical comparison of CWD biomass tons/ha. Coarse Woody Debris (CWD) size class measurements collected of each site in Transect Pair Two (TP2) (H = 15.83 df = 1, p > 0.1).

	CWD 1	**CWD 2**	**CWD 3**	**CWD 4**
TP2-I				
Bottom	20	9	7	1
Mid	22	11	3	2
Top	8	3	3	0
TP2-F				
Bottom	16	13	6	5
Mid	10	9	8	5
Top	16	4	0	2

Table 8. Assessment of coarse woody debris collected from each forested and impacted sampling station. Size classes modified from Uranowski et al. (2003) and included: 0 mm-6 mm, 6 mm-25 mm, 25 mm-76 mm, >76 mm. Treatment comparisons (i.e., forest v impact) were assessed by statistical comparison of CWD biomass tons/ha. Coarse Woody Debris (CWD) size class measurements collected of each site in Transect Pair Three (TP3).

	CWD 1	**CWD 2**	**CWD 3**	**CWD 4**
TP3-I				
Bottom	1	0	1	0
Mid	1	3	1	0
Top	0	5	1	1
TP3-F				
Bottom	17	6	1	2
Mid	4	2	3	1
Top	20	8	2	0

Table 9. Preliminary Index of Biotic Integrity (IBI). Represents a calculation to determine each species' ability to indicate an environments level of impact. Risk factor represents a description of a species' rarity and distribution (Critically Endangered (10 pts), Endangered (8 pts), Vulnerable (6 pts), near threatened (4 pts), undescribed (2 pts), least concern (1 pts)). Abundance Weight Value represents how abundant a species was (Cumulative Abundance <6 (1 pts), 6-10 (2 pts), 11 -15 (3 pts), 16-20 (4 pts), 21-25 (5 pts), 26-30 (6 pts), 31-35 (7 pts), 36-40 (8 pts), 41-45 (9 pts), >45 (10 pts)). Habitat distribution factor indicates a species' preferred environment (Forested preference (1 pts), No preference (0 pts), Impacted preference (-1 pts)). IBI value is a sum of all three numbers giving each species a quantifiable rank where the outliers, indicated in bold, indicate species as bioindicators for Laguna de Cube.

Species	Risk Factor	Abundance Weight Value	Habitat Distribution Factor	IBI Value
Alopoglossus festae	2	1	0	0
Ameiva septemlineata	2	1	-1	-2
Anolis biporcatus	2	1	-1	-2
Anolis granuliceps	1	1	-1	-1
Basiliscus galeritus	1	1	1	1
Boa constrictor imperator	1	1	1	1
Bothrops asper	1	1	1	1
Caiman crocodilus	1	1	1	1
Chelydra serpentina acutirostris	1	1	1	1
Coniophanes fissidens	2	1	1	2
Craugastor longirostris	1	1	1	1
Dendropsophus sp.	2	2	1	4
Enyalioides heterolepsis	2	1	1	2
Epipedobates boulengeri	1	9	1	**9**
Hypodactylus fallaciosus	2	4	1	**8**
Hypsiboas pellucens	1	10	-1	**-10**
Hypsiboas picturatus	1	1	-1	-1
Hypsiboas rosenbergi	1	1	-1	-1
Hypsiboas rufitelus	1	1	1	1

Hypsiboas sp. 1	2	1	1	2
Lachesis acrochorda	2	1	1	2
Lepidoblepharis buchwaldi	2	1	0	0
Leptodactylus bolivianus	1	1	1	1
Leptodactylus sp.	2	2	0	4
Pristimantis achatinus	1	10	1	**10**
Pristimantis celator	4	1	-1	-4
Pristimantis pteridophilus	8	1	1	**8**
Pristimantis ridens	1	1	0	0
Ptychoglossus gorgonae	2	1	1	2
Rhinella margaritifera	1	1	1	1
Rhinella marina	1	1	1	1
Scinax ruber	1	1	1	1
Scinax sp.	2	1	0	0
Scinax quinquefasciatus	1	1	1	1
Silverstoneia nubicola	4	1	1	4
Tantilla melanocephala	2	1	1	2
Trachycephalus jordani	1	1	-1	-1

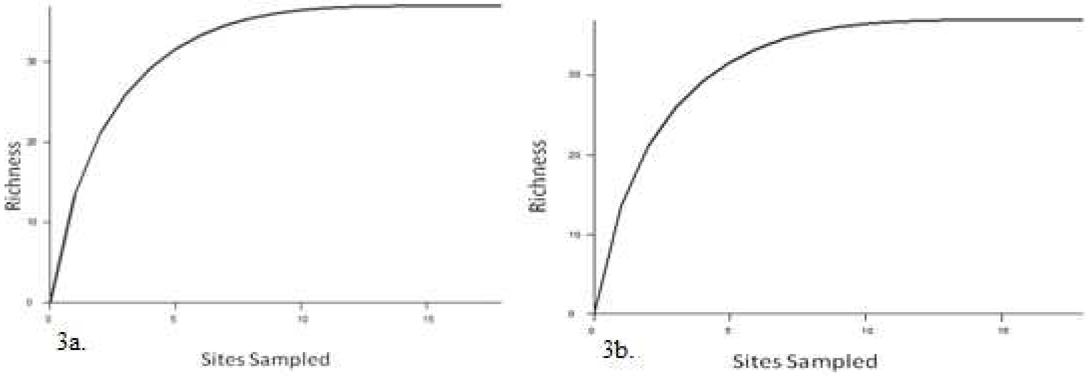
Table 10. Comparison of my study to available primary literature reporting location, treatment, study duration, and richness:(Bell and Donnelly 2006, Gardener et al. 2007, Ortega-Andrade et al. 2010, Urbina-Cardona et al. 2006, Vonesh 2001, Wanger et al. 2010).

Study	**Location**	**Study Length**	**Amphibian Richness**	**Reptilian Richness**
My Study	Ecuador	June 2010	22	15
Bell and Donnelly 2006	Costa Rica	Oct. 2003-July 2004	36	14
Gardner et al. 2007	Brazil	May-Sept. 2004,	23	30
		Jan.-June 2005		
Ortega-Andrade et al. 2010	Ecuador	Dec. 2006	37	72
		Feb.-March 2007		
		April-May 2008		
Urbina-Cardona et al. 2006	Mexico	June 2003-May 2004	21	33
Vonesh 2001	Uganda	March-July 1997	10	8
Wanger et al. 2010	Indonesia	Dec. 2007-July 2008	8	12

Figures

Figure 1. Map of the country of Ecuador. Laguna de Cube, Ramsar site 1143 (00°24’N 079°39’W) indicated by a red star.

Figure 2. An illustrated map of Laguna de Cube. Green lines represent the placement of forested transects and red lines represent locations of human impacted transects.



3a.

3b.

Figures 3a and 3b. Species accumulation curves for herpetofauna (i.e., amphibians and reptiles) in forest (3a) and impacted (3b) transects captured over 18 days during the summer of 2010 at Ramsar site #1143, Laguna de Cube, Ecuador. Accumulation curves suggest adequate sampling (i.e., reaching asymptote) by treatment.

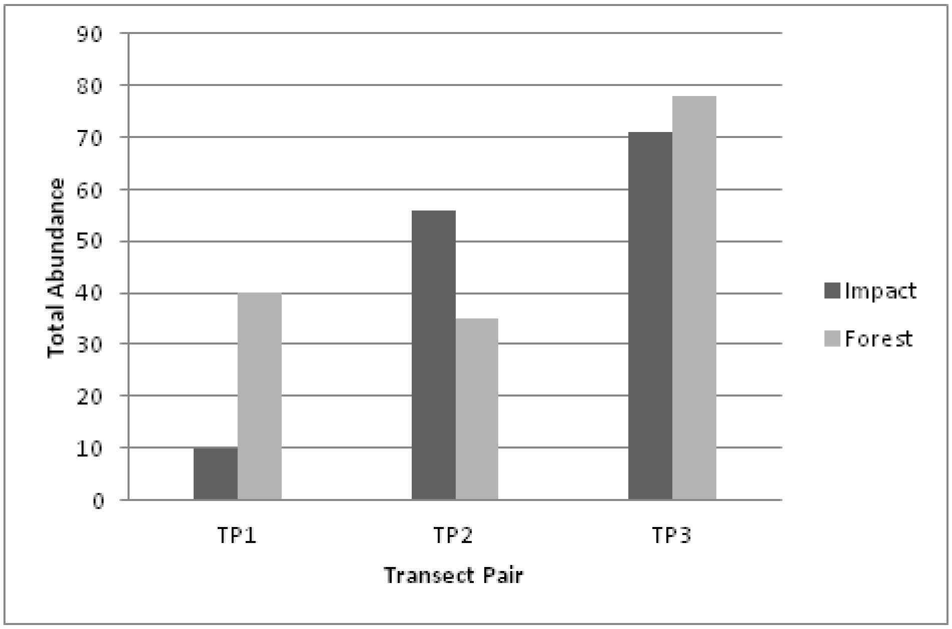


Figure 4. A comparison of herpetofauna abundance by treatment pair. Differences by treatment (i.e., TP1; 40 individuals in forest and 10 in impacted, TP2; 36 individuals in forest and 54 in impacted, TP3; 78 individuals in forest and 71 in impacted).

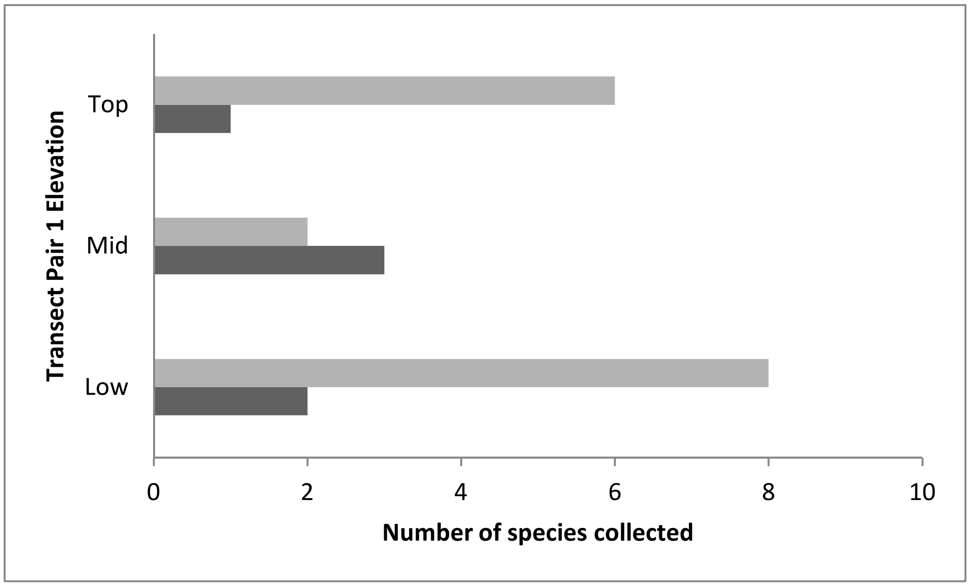


Figure 5. Species richness by elevation represented by the number of species captured at each elevation in transect pair 1. Elevations presented in pairs (n=3). Forest transect is light gray while impacted is shown in dark grey.

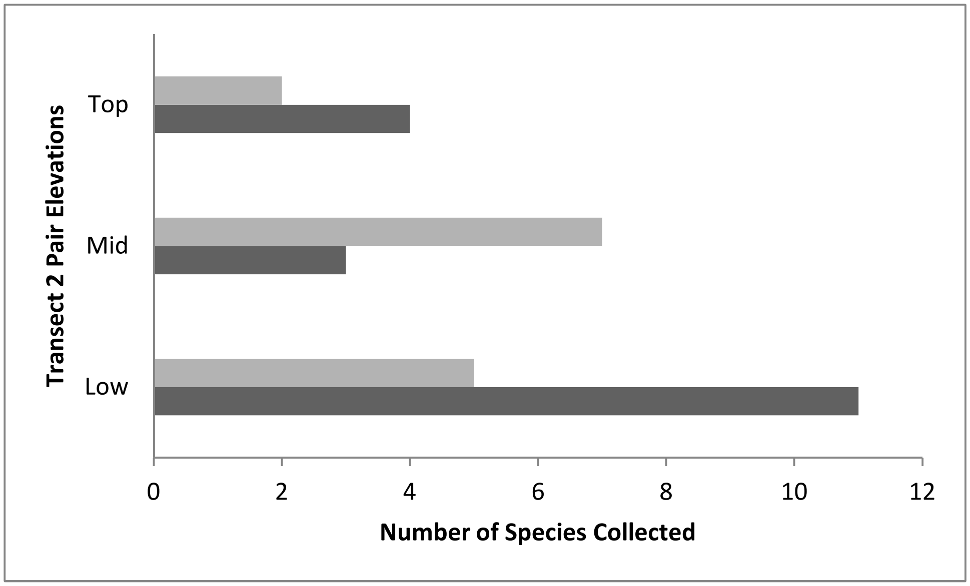


Figure 6. Species richness by elevation represented by the number of species captured at each elevation in transect pair 2. Elevations presented in pairs (n=3). Forest transect is light gray while impacted is shown in dark grey.

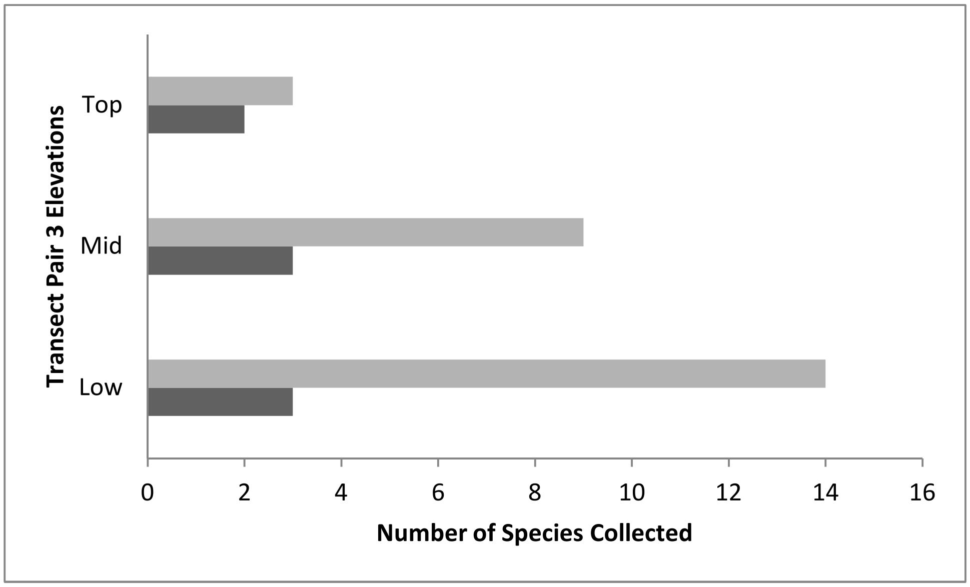


Figure 7. Species richness by elevation represented by the number of species captured at each elevation in transect pair 3. Elevations presented in pairs (n=3). Forest transect is light gray while impacted is shown in dark grey.

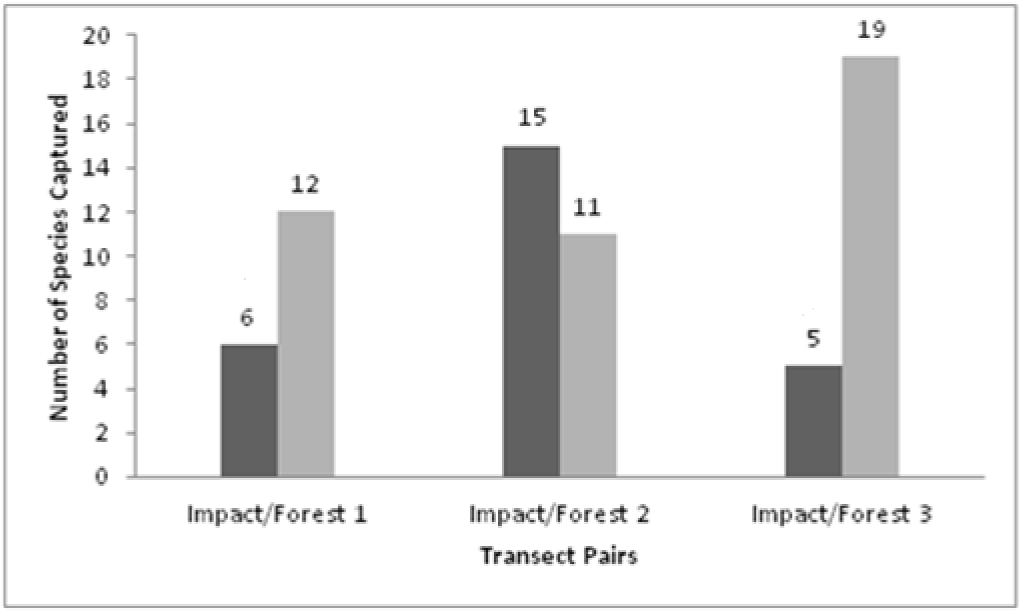


Figure 8. Comparison of species richness by treatment pair with forest transects represented by light gray and impacted sites by dark gray.

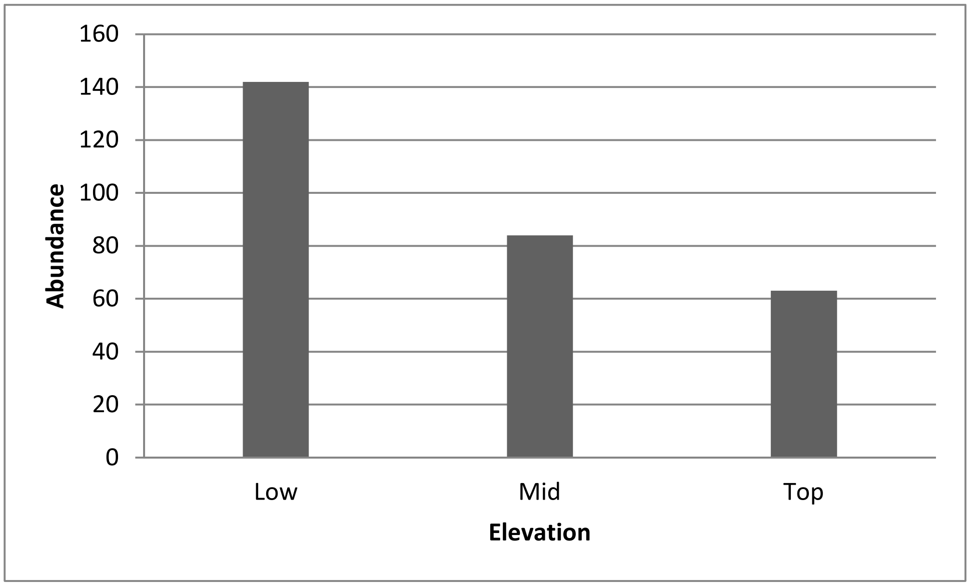


Figure 9. A comparison of total herpetofauna abundance among three elevation (i.e., low, mid and top) with summed forest and impact values. Abundance did not differ significantly by elevation with Kruskal-Wallis analysis of variance (p>0.10).

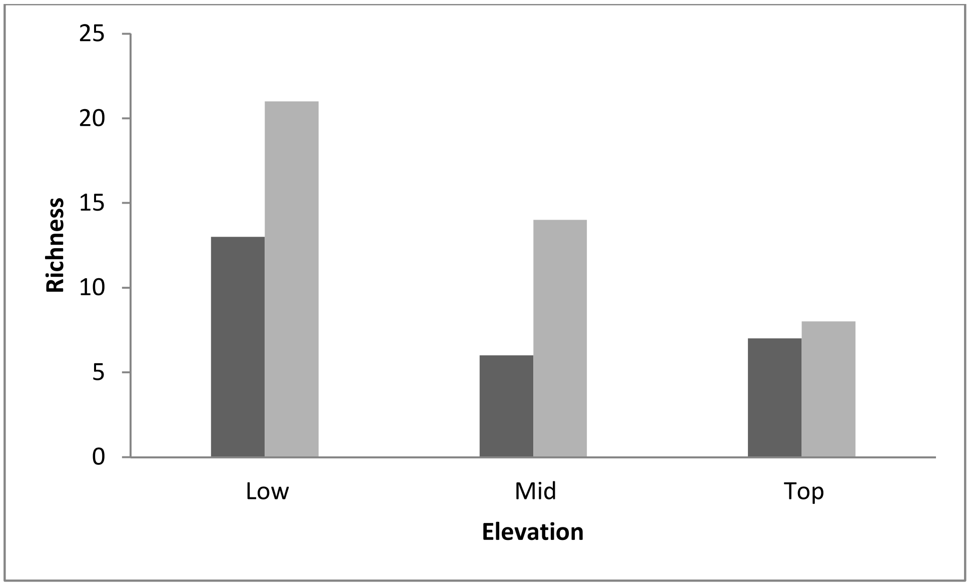


Figure 10. A comparison of total herpetofauna richness among three elevation (i.e., low, mid and top) with summed forest and impact values. Richness did not differ significantly by elevation with Kruskal-Wallis analysis of variance (p>0.10).

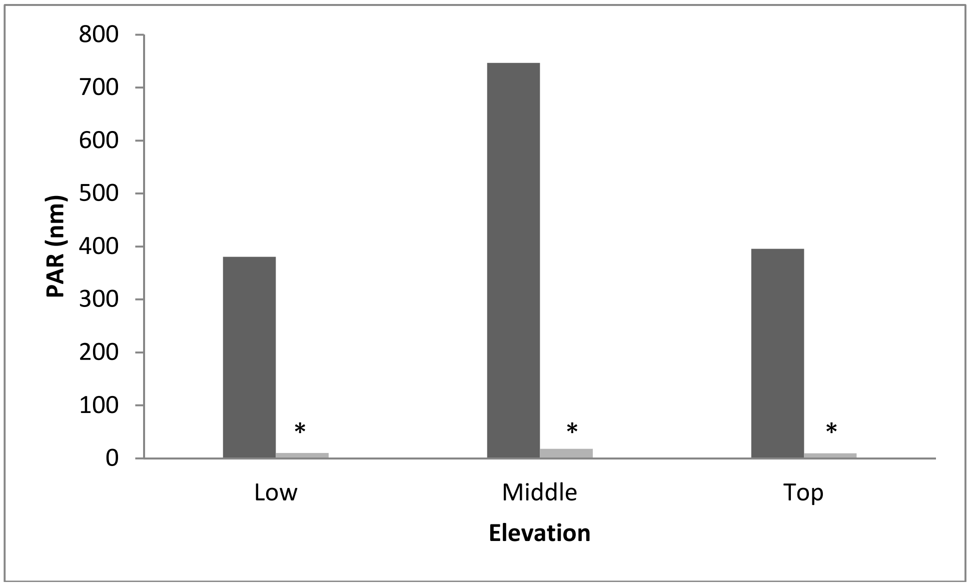


Figure 11a. Mean PAR for forested and impacted sampling stations represented by elevation pairs (Kruskal-Wallis: Low_F v. Low_I= ; H = 27.6 df= 5, p = 0.07612; Mid_F v. Mid_I= ; H = 27.6 df= 5, p = 0.07612; Top_F v. Top_I= , H = 27.6 df= 5, p = 0.005075). Dark grey represents impacted sites and light grey represents forested sites. Significant differences identified by an asterix (*).

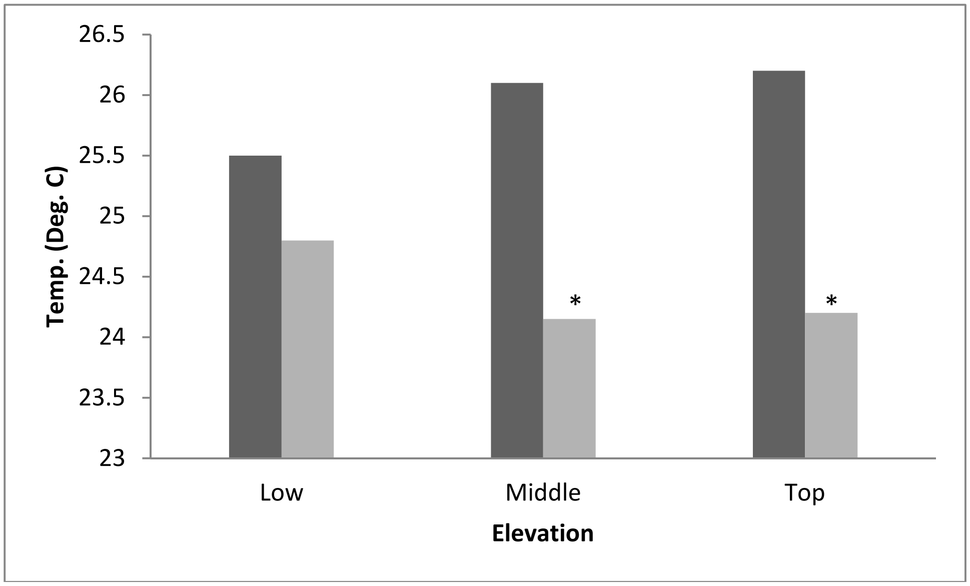


Figure 11b. Mean Temperature for forested and impacted sampling stations represented by elevation pairs (Kruskal-Wallis: Low_F v. Low_I; H = 15.98 df= 5, p = 0.1689; Mid_F v. Mid_I; H = 15.98 df= 5, p = 0.01291; Top_F v. Top_I; H = 15.98 df= 5, p = 0.005). Dark grey represents impacted sites and light grey represents forested sites. Significant differences identified by an asterix (*).

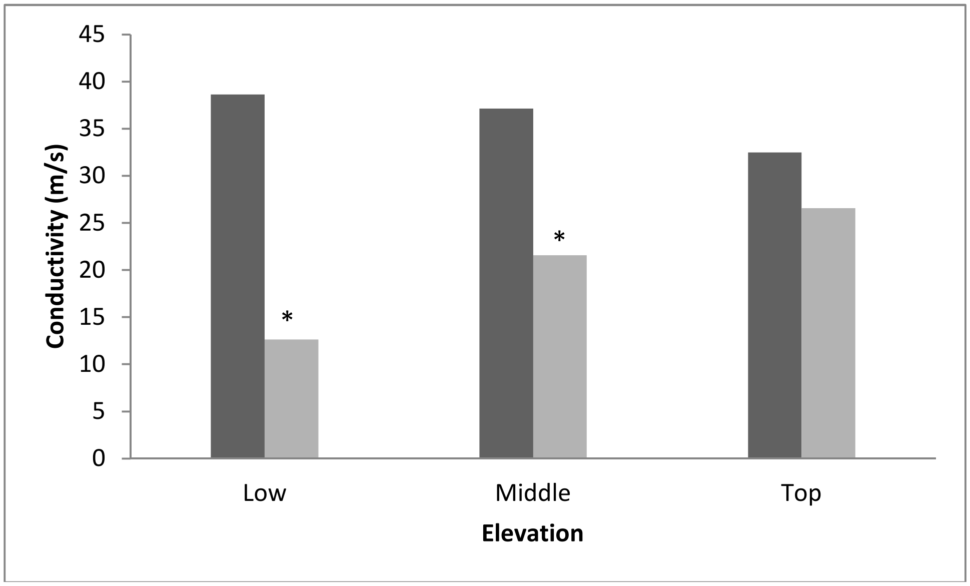


Figure 11c. Mean Conductivity for forested and impacted sampling stations represented by elevation pairs Kruskal-Wallis: Low_F v. Low_I; H = 16.07 df= 5, p = 0.02002; Mid_F v. Mid_I; H = 16.07 df= 5, p = 0.04533; Top_F v. Top_I; H = 16.07 df= 5, p = 0.1275). Dark grey represents impacted sites and light grey represents forested sites. Significant differences identified by an asterix (*).

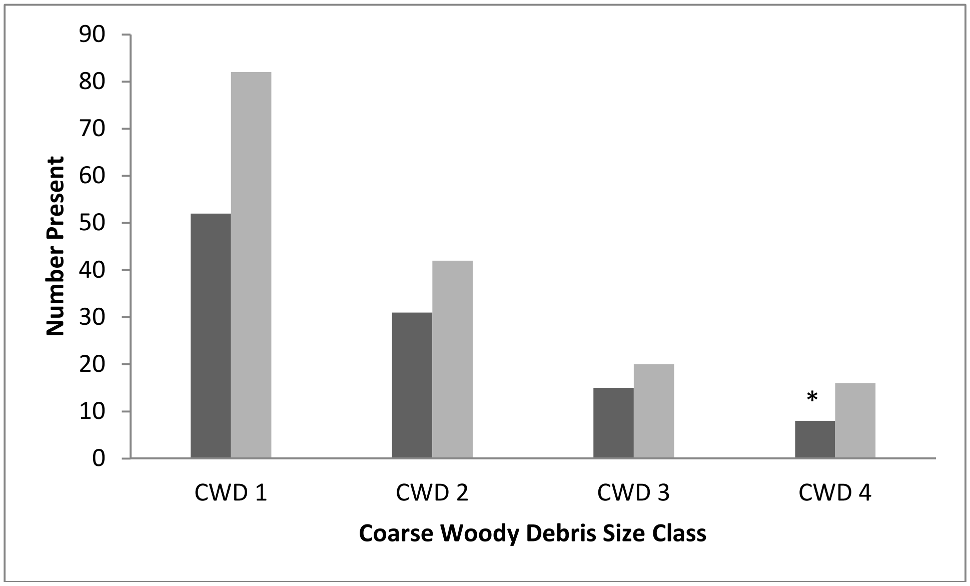


Figure 11d. Shows the CWD class comparison between forested and impacted sampling stations represented by CWD size class (Kruskal-Wallis: 1; H = 19.79 df= 5, p = 0.4209) (2; H = 19.79 df= 5, p = 0.5204) (3; H = 19.79 df= 5, p = 0.8696) (4; H = 19.79 df= 5, p = 0.09746). Dark grey represents impacted sites and light grey represents forested sites. Significant differences identified by an asterix (*).

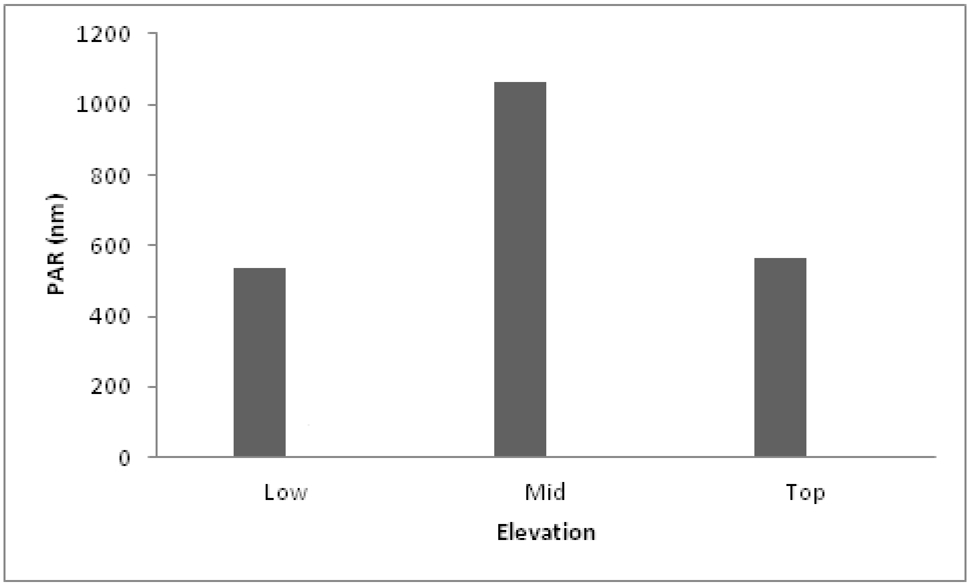


Figure 12a. Mean PAR for forested and impacted sampling stations at TP2 represented by elevation. Dark grey represents impacted sites and light grey represents forested sites.

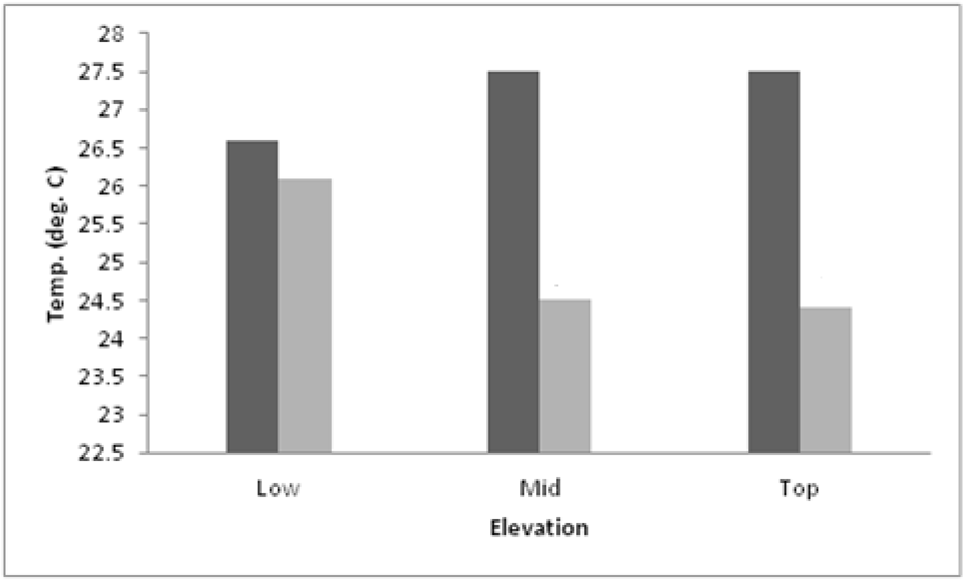


Figure 12b. Mean Temperature for forested and impacted sampling stations at TP2 represented by elevation pairs. Dark grey represents impacted sites and light grey represents forested sites.

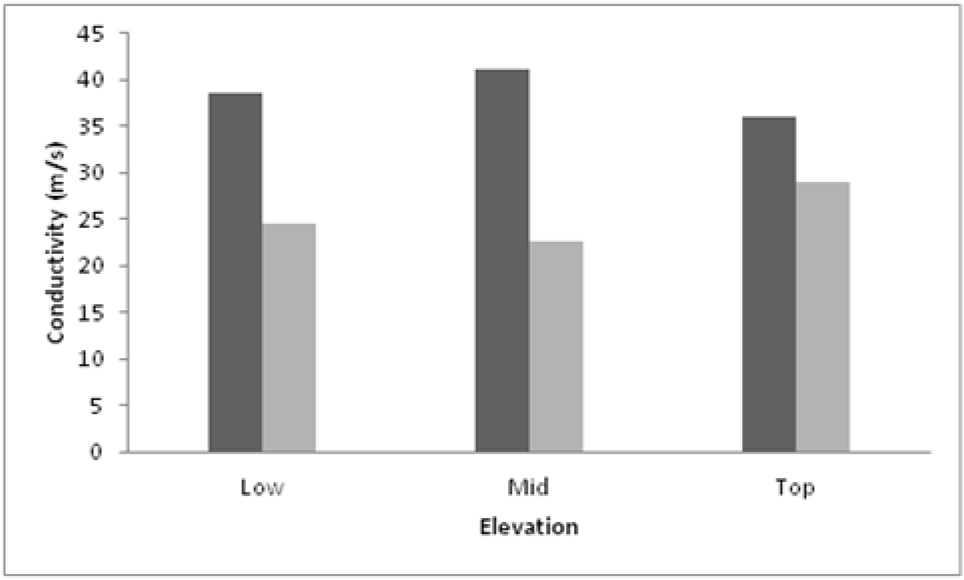


Figure 12c. Mean Conductivity for forested and impacted sampling stations at TP2 represented by elevation pairs. Dark grey represents impacted sites and light grey represents forested sites.

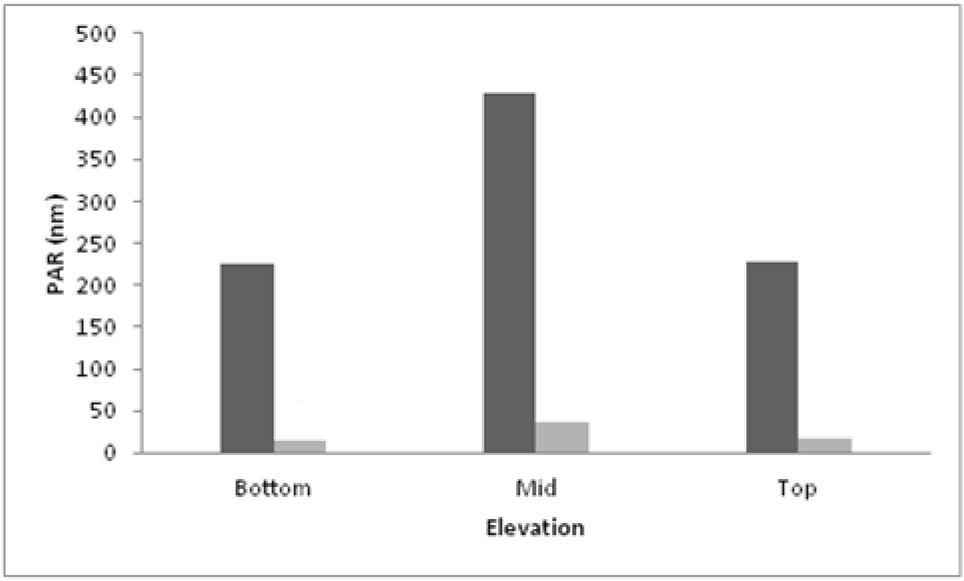


Figure 13a. Mean PAR for forested and impacted sampling stations at TP3 represented by elevation pairs. Dark grey represents impacted sites and light grey represents forested sites.

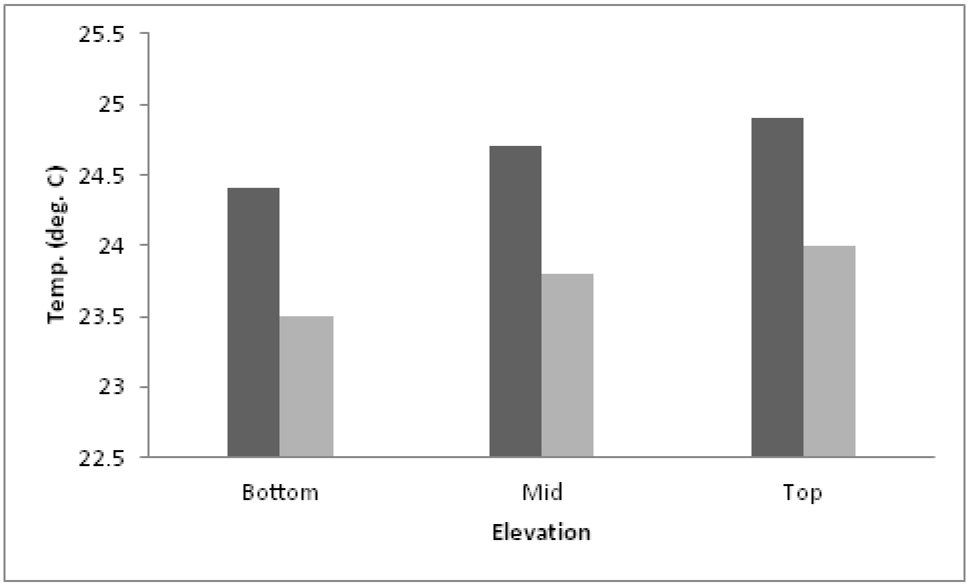


Figure 13b. Mean Temperature for forested and impacted sampling stations at TP3 represented by elevation pairs. Dark grey represents impacted sites and light grey represents forested sites.

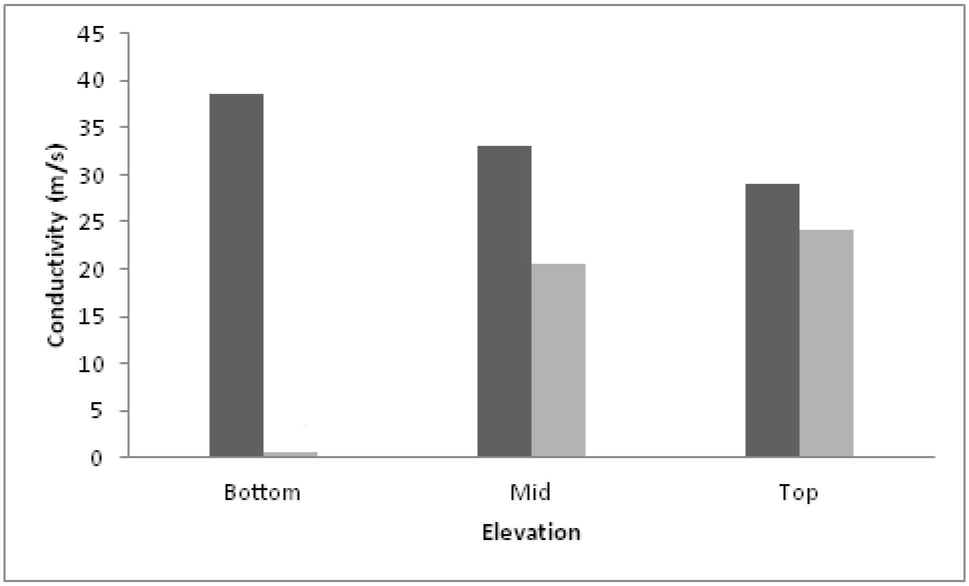


Figure 13c. Mean Conductivity for forested and impacted sampling stations at TP3 over represented by elevation pairs. Dark grey represents impacted sites and light grey represents forested sites.

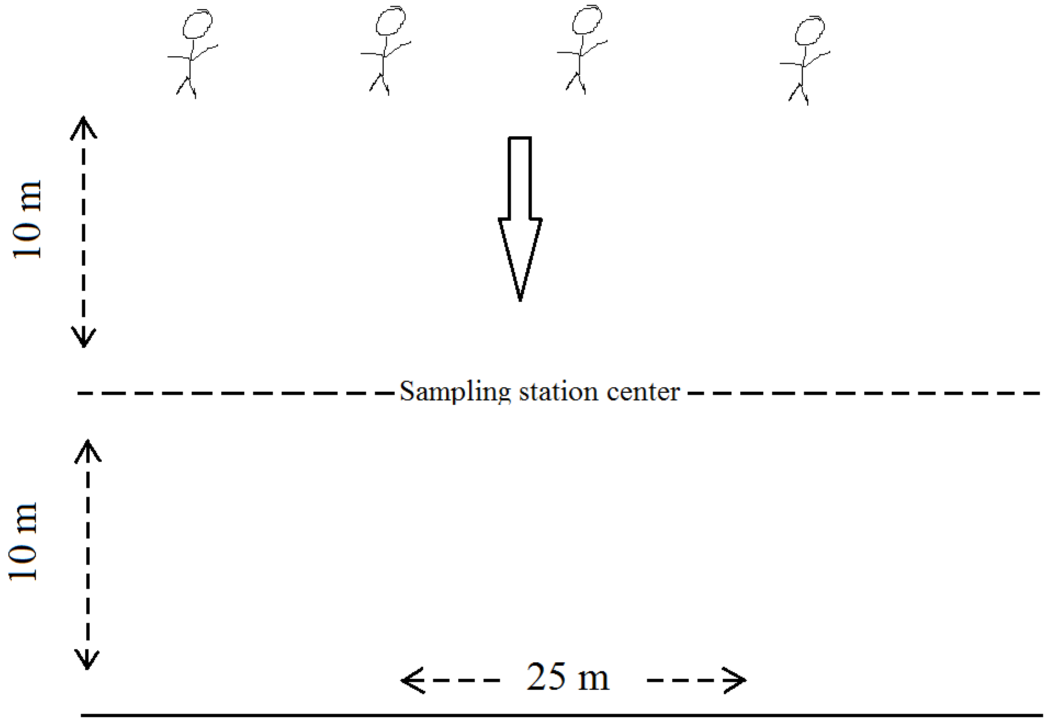
Appendix

1. Examples of the tree frog shelters constructed and the drift fences used at each of the 18 sites samples at Laguna de Cube in 2010.

An example of a tree frog shelter constructed out of bamboo with a small (0.5 cm) drainage hole drilled to reduce water accumulation between samplings.

A drift fence placed at the boundary of a sampling station to aid in capturing fleeing individuals.

2. A sampling station with a 25 m drift fence at the lower boundary and 4 searchers preparing for a VES beginning 10 m uphill from the sampling station center.



3. A list of species captured at Laguna de Cube, Ecuador over 18 days during the summer of 2010. Each species is accompanied by photographs and where accessible, the IUCN (2012) Redlist habitat and ecology description.

Amphibia

Rhinella margaritifera, Bufonidae

Terrestrial and nocturnal toad found in primary and secondary lowland, premontane and montane tropical moist forest (including terra firme and seasonally flooded forest). Also present in open areas. In Gorgona Island its distribution appears to be influenced by the understory. This species is an explosive breeder in temporary pools and streams. Males call along banks of watercourses. It is a very generalist species that can be found in disturbed areas.

Rhinella marina, Bufonidae

A nocturnal and terrestrial toad that inhabits humid areas with adequate cover, including cane fields, savannah, open forest, well watered yards and gardens. It also inhabits dry equatorial forests. It thrives in degraded habitats and man-made environments, and is occasionally found in pristine lowland and montane rainforests, but generally prefers open or disturbed habitat such as tracks, roads, low grassland and areas that are near human settlement, e.g. grazing land, suburban parks and gardens.

Craugastor longirostris, Craugastoridae

It is a species of lowland and submontane rainforest, with a few records also from dry forest. It is a terrestrial species that can survive in secondary forest, but not in open areas. It is often found along streams. It is presumed to breed by direct development, but the egg deposition site is not known.

Epipedobates boulengeri, Dendrobatidae

Silverstoneia nubicola, Dendrobatidae

It is a terrestrial and diurnal species of humid lowland, premontane and montane forest. Adults may be encountered along rocky sections of forest streams. It can occur in secondary forest and plantation forests, and occurs in degraded habitats in Colombia. The eggs are deposited in the leaf-litter and the males transport hatching tadpoles to forest streams to complete metamorphosis.

Dendropsophus sp., Hylidae

Hypsiboas pellucens, Hylidae

Juvenile Adult

It lives in a very wide variety of habitats, including forest, degraded areas, gardens, and urban areas. It can be found in the canopy, but descends to lower vegetation at night to breed. It breeds in permanent pools, including artificial ponds.

Hypsiboas picturatus, Hylidae

It is a species of closed forest, including small patches of secondary forest. However, it needs a closed canopy in order to survive. It lives on vegetation close to streams in which the tadpoles develop.

Hypsiboas rosenbergi, Hylidae

It is an adaptable species that occurs in primary and secondary forest, but also in heavily altered areas such as small strips of trees in pastureland. Breeding takes place in a shallow water-filled depression constructed by the male close to a stream. Developing tadpoles swim from this nest to streams when they become flooded.

Hypsiboas rufitelus, Hylidae

It inhabits humid lowland forest and tolerates some disturbance. It can be found in open areas, but this needs to be close to forest. It reproduces in swamps surrounded by trees.

Hypsiboas sp,. Hylidae

Scinax ruber, Hylidae

This large, arboreal, nocturnal frog inhabits a vast array of habitats, from open environments to moist forests. In the Gran Sabana region of southeastern Venezuela, males have been found calling from the ground and low vegetation around temporary ponds in the forest, and amplectant pairs were on low vegetation. Scinax ruber is a "pest" species primarily inhabiting cleared areas in the rainforest. This species generally breeds in small temporary pools. In cultivated areas, the species breeds in roadside ditches and shallow, temporary ponds. Animals have been recorded in modified environments such as gardens and parks.

Scinax sp., Hylidae

Scinax quinquefasciatus, Hylidae

It is a very adaptable species, living on the forest edge and in open habitats with some trees, even entering houses. It breeds in temporary pools.

Trachycephalus jordani, Hylidae

It is an arboreal frog of lowland and premontane wet and dry forests. It has also been found in disturbed areas. Lives in bromeliads and holes in tree trunks. It reproduces in lentic waterbodies.

Leptodactylus bolivianus, Leptodactylidae

It is a large, mostly nocturnal species of open grassy areas, humid lowland forest, dry forest, secondary forest, swamps, pasture, drainage ditches and rice fields. It is not present in coffee plantations. The adults live in a burrow retreat, although there is no evidence that they excavate these burrows. It is usually associated with permanent water sources, although may also be found in marshes and temporary ponds. The eggs of the species are laid in foam nests, often hidden in vegetation. The larvae develop in shallow, muddy ponds, where they take refuge under leaves on the bottom. It probably breeds during the early to mid rainy season.

Leptodactylus sp., Leptodactylidae

Hypodactylus fallaciosus, Strabomantidae

It has been collected in montane cloud forest, where it is a terrestrial species that breeds by direct development.

Pristimantis achatinus, Strabomantidae

It is a species that has been found in forest clearings, road cuts in forest, and banana, cacao and coffee plantations, as well as in forested areas. It is found that this species is greatly influenced by the canopy cover. It is found in both lowland and montane habitats. It is mostly terrestrial, sometimes occurring off the ground in vegetation. It breeds by direct development, the eggs are deposited either on the forest floor or low in the vegetation.

Pristimantis celator, Strabomantidae

It is a nocturnal species that has been found in terrestrial bromeliads, and may be encountered on the sides of roads and in herbaceous vegetation, in leafy cloud forests. It is presumed to be a direct developing species although the site of egg deposition is not known. It is tolerant of habitat disturbance provided bromeliads are available.

Pristimantis pteridophilus, Strabomantidae

It lives in primary forest, and is somewhat adaptable, being found on forest edges and the edges of pastures. The use of ferns by this species stands out dramatically in comparison with the limited degree to which most other Andean **Eleutherodactylus** perch on ferns. It is presumed to breed by direct development, but the site of egg deposition is not known.

Pristimantis ridens, Strabomantidae

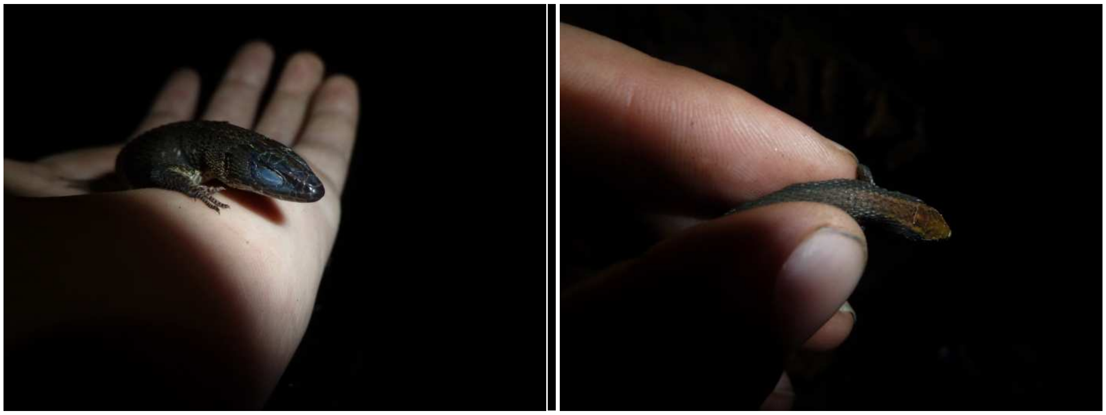
It inhabits humid lowland and montane forest, but also survives in disturbed habitats, including degraded secondary vegetation, plantations, rural gardens and urban areas. It breeds by direct development and the eggs are deposited in the leaf-litter.

Reptilia

Caiman crocodiles, Alligatoridae

Basiliscus galeritus, Corytophanidae

Alopoglossus festae, Gymnophthalmidae



Ptychoglossus gorgonae, Gymnophthalmidae

(Photo taken from Ortega-Andrade et al. 2010)

Enyalioides heterolepis, Hoplocercidae

Anolis biporcatus, Polychrotidae

Anolis granuliceps, Polychrotidae

This species is found in lowland moist forest habitat.

Lepidoblepharis buchwaldi, Sphaerodactylidae

Ameiva septemlineata, Teiidae

(Photo taken from Ortega-Andrade et al. 2010)

Boa constrictor imperator, Boidae

Coniophanes fissidens, Colubridae

Tantilla melanocephala, Colubridae

Bothrops asper, Viperidae

Lachesis acrochorda, Viperidae

Chelydra serpentina acutirostris, Chelydridae

(Photo taken from freeimagefinder.com)

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