



How informative is the response of Ground Beetles' (Coleoptera: Carabidae) assemblages to anthropogenic land use changes? Insights for ecological status assessments from a case study in the Neotropics

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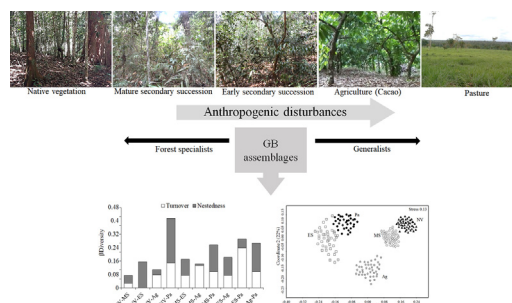
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HIGHLIGHTS

- Effects of ecosystem change on Carabidae (C) composition were investigated.
- The extent of disturbance was correlated with variation in C communities.
- Several C species were selected as indicators of the different ecosystems.
- C communities could be used to determine the status of Neotropical ecosystems.

GRAPHICAL ABSTRACT



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ABSTRACT

Ground Beetles (Carabidae) have been suggested to be used as indicators of ecological status under the effects of anthropogenic land use and land cover changes in highly biodiverse regions such as the Neotropics. In this study Ground Beetles' assemblages from a region in the Brazilian Amazon were investigated for evaluating their applicability as disturbance indicators. Differences in assemblages between ecosystems, discriminated by complementary methodologies, demonstrate a sensitive reaction from the most pristine forests to increasingly disturbed systems. Besides capturing the influence of different prevailing conditions between ecosystems, Ground Beetles are easy to communicate and to link with the other components of the system, being effortless and routinely measurable using standard methodologies. This study represents a step forward in using Ground Beetles for the purposes of planning, management and public reporting on the ecological status of Neotropical ecosystems. Additionally, the methods depicted could support projections for trends of relevant ecosystem attributes under realistic social-ecological change scenarios, which can be used to guide effective conservation planning.

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1. Introduction

Neotropical rainforests are considered biodiversity hotspots (Gardner, 2010), vital for carbon storage and regional/global regulation

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of hydrological cycles and climate (Berenguer et al., 2015). However, anthropogenic activities have been causing massive loss of biodiversity in these systems (Kim et al., 2015), associated with new ecosystem processes and combinations of species/morphospecies that are ultimately linked with land use/land cover changes (LUCC) (Titeux et al., 2016). Biodiversity loss threatens the sustainability of regional ecosystems, with major implications in the socio-ecological services delivered (Cajas and Silva, 2015, 2017). Predicting the ecological consequences of LUCC is therefore subject to scientific and political studies aimed at supporting strategic options for landscape planning and natural resources management (Turner et al., 2007). In this context the selection of the most pertinent indicators for ecological assessments and monitoring is crucial, namely to detect and evaluate possible trends with relevance to guide conservation decision-making in the scope of disturbed ecosystems management (Cajas et al., 2017a).

Terrestrial invertebrates and especially insects play a critical role in most ecological processes and are key components of ecosystems' structure and functioning (Bicknell et al., 2014; Viegas et al., 2014; Campos and Hernández, 2015). Insects' diversity is intimately related with other taxa diversity and abiotic characteristics, thus representing potential ecological indicators of the ecosystem as a whole (e.g. Nichols et al., 2008). Therefore, understanding the ecological relevance of insects in the Neotropics could support decision-making and robust management/recovery of imperilled ecosystems in the scope of the need for rapid, standardized and cost-saving assessment methodologies (Godfray et al., 1999; Cajas et al., 2017b). Ground Beetles (GB) of the Carabidae family have been suggested as promising ecological indicators in biodiversity studies and ecological integrity/status assessments, considering their morphological, behavioural, taxonomic composition, ecological traits, abundance and sensitivity to environmental changes (Koivula et al., 2002; Rainio and Niemelä, 2003; Koivula, 2011). GB represent an ecologically and taxonomically well-known group, which is straightforward to collect by using pitfall traps (e.g. Rainio and Niemelä, 2003; Niemelä et al., 2007; Koivula, 2011; Kotze et al., 2011). Furthermore, in forest ecosystems GB exhibit important trophic roles in the food webs, either as predators, decomposers or phytophagous (Qodri et al., 2016). Several GB are generalist predators (Nitzu et al., 2008), feeding on a variety of arthropod preys, such as Collembola, Acari, and larvae of Diptera (Ribera et al., 1999), providing valuable ecosystem services by controlling forest and even agriculture pests (Holland, 2002) and weeds (Bohan et al., 2011). The GB trends also reflect the consequences of ecosystem changes given their dependence on strict microhabitat conditions and specific diets (Koivula, 2002).

Various studies have used GB for indicating successional stages, the degree of ecosystems fragmentation, urbanization pressures, metal contamination in soils, disturbance gradients, responses to environmental characteristics, regeneration in natural areas and ecosystem "health" (Fujita et al., 2008; Silva et al., 2008; Uehara-Prado et al., 2009; Moraes et al., 2013). Despite the recognized importance as ecological indicators, their application in the Neotropics, such as the Brazilian Amazon (Amazonia), has been extremely limited. As a consequence, the effects of human actions (e.g. deforestation, fire, animal grazing, logging and mining) on the GB communities and their relationships with the structure and functioning of ecosystems are still poorly understood (Samu et al., 2010). Another problem is the replicability of studies' results since many areas in the region have low accessibility and several logistical and practical difficulties, which restrict the application of standardized methods of sampling and replication (Cajas et al., 2015a).

Our study examined the concealed patterns of GB assemblages through a gradient of ecosystem disturbance, including areas of primary forest, secondary forest in different stages of recovery, cocoa plantations and pastures in a region of the Amazonia. The main objective was to analyze the dissimilarity of GB assemblages by testing the following hypotheses: (1) the GB community composition, using morphospecies, is sensitive to disturbance of reference ecosystems such as primary forest; (2) increasing disturbance can be assessed by an increasing dissimilarity

of the selected ecological indicators from pristine forests, namely morphospecies richness, abundance, taxonomic diversity and ecological composition. These hypotheses were tested through representative gradients in order to provide new insights for the GB diversity standard assessments as indicators of disturbance in Neotropical ecosystems.

2. Material and methods

2.1. Study sites

The study area was located in the municipality of Uruará, state of Pará, northern Brazil ($-03^{\circ}43'27''$ S - $53^{\circ}44'8''$ W, Fig. 1). Forest (69% of the area) is the dominant land use/land cover (LULC) while deforestation is concentrated mainly in the south-central part of the territory and near the main roads. Extensive livestock production and the exploitation of timber (mostly illegal) are currently considered the most serious environmental pressures (Cajas et al., 2015b). The studied areas contain the most representative ecosystems of the region, in terms of biophysical and ecological characteristics for understanding the response of Ground Beetles (GB) assemblages to landscape disturbance, such as Native Vegetation (NV), Early Secondary succession (ES - secondary vegetation with five years of regeneration), Mature Secondary succession (MS - secondary vegetation with 15 years of regeneration), Agriculture (Ag - cocoa plantations, *Theobroma cacao* L.) and Pasture for extensive livestock (Pa). The climate is characterized as hot-humid (Köppen's classification), with annual average temperature and precipitation of 26 °C and 2000 mm respectively (Peel et al., 2007).

2.2. Carabidae sampling

Fieldworks were carried out during the year 2015 in the rainy season (February/March), intermediate season (June) and dry season (September/October), for inspecting eventual periodic differences in the activity and composition of GB. The selected locations were positioned at minimum distance of 100 m from ecotones, guaranteeing that most GB captured were associated to the monitored ecosystem. GB were trapped using pitfalls with 75 mm diameter and 110 mm deep, filled with preservative liquid consisting of formalin, alcohol, water and a few drops of detergent to break the surface tension. A roof was attached to each pitfall trap in order to prevent the destructive effect of direct rainwater. In each ecosystem per season, fifty-six traps were placed (100 m from each other), remaining installed for 48 h prior to collection. This protocol was applied to all areas and periods of collection, totalizing a sampling effort of 840 traps (see Table S2, Supplementary material). The specimens were deposited in the MCNU (Museu de Ciências Naturais da Univas - Univas Natural Sciences Museum, Brazil) and Coleção de Zoologia, Departamento de Biologia da Universidade Federal do Pará (Zoological collection of the Biology Department of the University of Pará, Brazil). The collection authorization process was issued by SISBIO system under the number 50133/2015. Sampling survey period and intensity, although inadequate for a definitive inventory, served the purpose of comparing GB general sensitivity to ongoing changes in the scope of their application as ecological indicators of disturbance (Dale and Beyeler, 2001).

2.3. Environmental variables

Fourteen environmental variables, considered with potential influence on GB communities, were measured in the sampled ecosystems (Felton et al., 2006): temperature (T), humidity (H), precipitation (P), circumference at breast height (CBH), circumference at ankle height (CAH), canopy cover (CC), richness of plants (RP), abundance of plants (AP), richness of shrubs (RS), abundance of shrubs (AS), percentage of exposed soil (PES), percentage of green (vegetation) cover (GC), percentages of leaf litter cover (LLC), height of leaf litter (HLL). Air temperature, relative humidity and rainfall of each point were measured

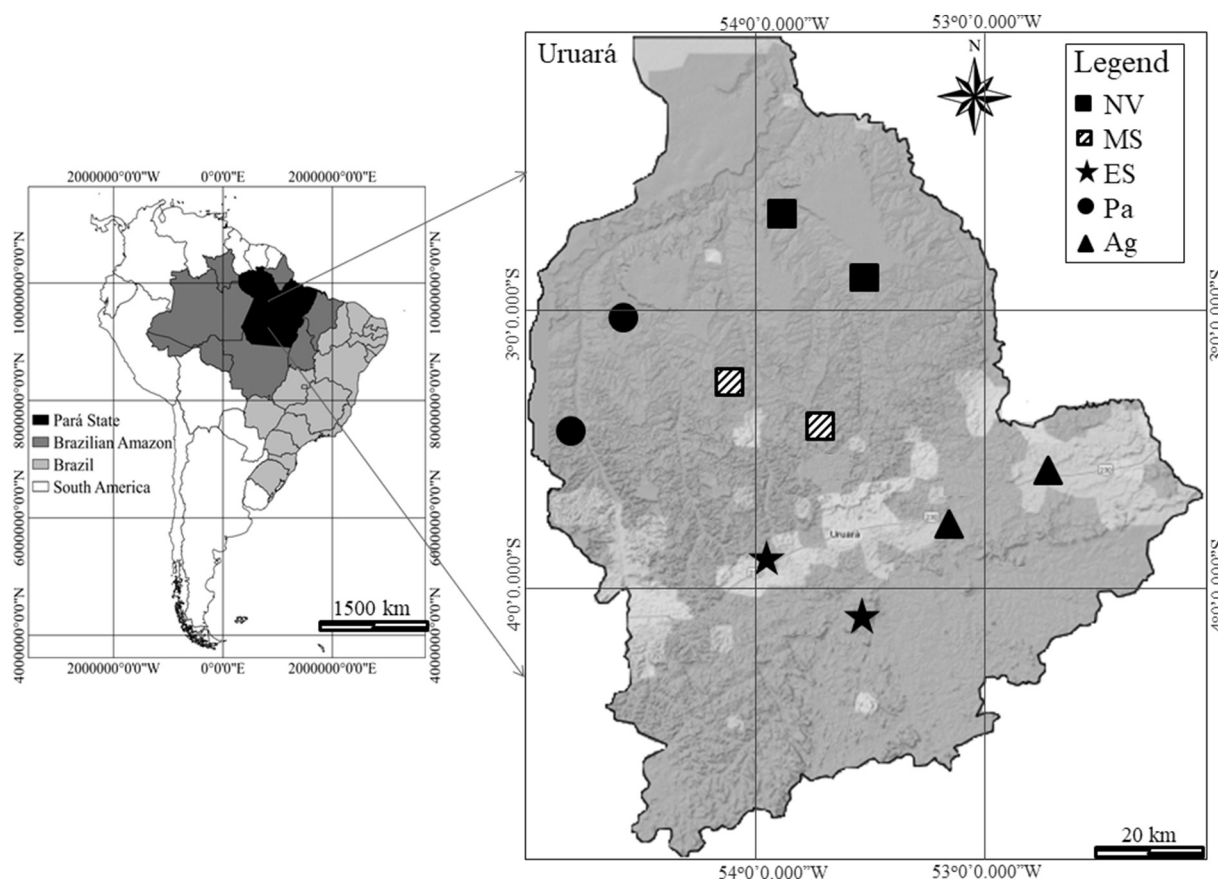


Fig. 1. Location of the study region in the municipality of Uruará, state of Pará, northern Brazil. Location of the study region, (a) Brazil, (b) Pará state, and (c) city of Uruará. Location of the sampling areas (sites) associated with each ecosystem: NV, Native vegetation; MS, Mature secondary succession; ES, Early secondary succession; Pa, Pasture; Ag, Agriculture. See Supplemental material, Table S1 and Fig. S1, for details of the distances between each sampled site.

during the traps' installation and removal by a portable weather station (model Oregon Scientific WMR200A). To assess the environmental complexity of each ecosystem, the quadrat-section method was adopted (Campos and Hernández, 2015). Using a cross as a reference, four quadrants (northeast, northwest, southeast, southwest) were marked and in each quadrant the following variables were measured: trees with circumference at breast height (CBH) > 15 cm, shrubs with CBH < 15 cm and with height > 1 m were selected and the distances to the centre of the cross, height, crown diameter and trunk diameter. Trunk diameter was taken at breast height (1.3 m) for the trees and ankle height (CAH = 0.1 m) for the shrubs. In each quadrant, the height of leaf litter in 1 m × 1 m marked square (using PVC pipe) was measured with a ruler, and the percentages of leaf litter layer, green and exposed soil area (no vegetation or leaf litter) were measured by visual estimation using the following classes, 0–5%, 6–25%, 26–50%, 51–75%, 76–95% and 96–100% (Campos and Hernández, 2015). Using the same classes, the percentage of canopy cover in the four quadrats was calculated with a convex spherical Lemmon densiometer (D). Information concerning the methodology associated with each variable is depicted in Table S3, Supplementary material.

2.4. Statistical analysis

2.4.1. Preliminary data analysis

Within the scope of their use as ecological indicators in biodiversity studies and ecological integrity/status assessments our emphasis was not on a detailed taxonomic study (species, sub-species), but to present results of a prompt survey to quantify the impacts of anthropogenic activities on the biodiversity of Ground Beetles in the Amazon region. The morphospecies concept was applied, useful for megadiverse regions

with poorly known taxa (Swart et al., 2018). Additionally morphospecies are considered important in conservation studies, mainly for environmental impact assessments and comparing diversity among sites within a region (e.g. Derraik et al., 2002).

Estimated morphospecies/species (hereinafter designated by recognizable taxonomic units of ground Beetles: RTU) richness was calculated for all sites together and for each individual site using the mean through the estimators Chao 1, Chao 2, Jackknife 1, Jackknife 2, and Bootstrap (and their confidence intervals at 95%) (Krell, 2004). The software Estimate 9.0 was used for analyses (Colwell, 2013). The mean of these estimators was used in order to minimize bias from any particular estimator, the performance of which often varies according to differences in richness, sampling effort, and community evenness (O'Hara, 2005).

To infer the independence and significance of each spatial scale of the monitored data (pitfall trap, site or ecosystem), several tests were applied (Colegrave and Ruxton, 2018). Statistical independence of the field data, organized by location of each pitfall trap within each site was assessed using the Wald-Wolfowitz runs test (Zar, 1996). Results showed that overall, RTU richness and abundance of GB monitored are random in relation to location of each pitfall trap within each site (Table 1). In addition, to complement the previous analysis, the spatial independence at all pitfall traps in the given distance class related to taxonomic composition was tested using Moran's I Analysis (Moran, 1950). We used the resulting values of the taxonomic composition for a Principal Components Analysis (PCA) where the first axis was used as response variable to the Moran's I, with coordinate variables for ten different distance classes (Dalzochio et al., 2017). A global Moran's I analysis detected no significant spatial structure of the taxonomic composition for any distance class (Moran's I = −0.48; $p = 0.74$).

Table 1

The Wald-Wolfowitz runs test results for RTU richness (a) and RTU abundance (b) of Carabidae ranked by location within each site (sample size, N = 56).

Ecosystems	Z	P	Z	P
	a) RTU richness		b) RTU abundance	
NV_1	0.060	0.952	1.545	0.122
NV_2	−0.182	0.855	0.328	0.743
MS_1	−1.084	0.279	−0.611	0.541
MS_2	−1.645	0.100	−1.375	0.169
ES_1	−1.778	0.075	0.715	0.475
ES_2	1.607	0.108	1.870	0.062
Ag_1	−1.864	0.062	−1.875	0.061
Ag_2	0.137	0.891	1.119	0.263
Pa_1	−0.440	0.660	0.456	0.648
Pa_2	−1.238	0.216	−1.120	0.263

Z value (Z) and the significance level associated (P). NV, Native vegetation; MS, Mature Secondary succession (vegetation with 15 years of regeneration); ES, Early Secondary succession (vegetation with five years of regeneration); Ag, Agriculture; Pa, Pasture. The values followed by the acronyms of sites within ecosystems (e.g. NV_1, NV_2, ...).

Furthermore, a non-parametric nested analysis of variance was performed to analyze data of GB abundance and richness per pitfall trap within each site, per site within each ecosystem and among ecosystems (Conover and Iman, 1981; McDonald, 2009). Non-parametric statistical tests assume homoscedastic error variances, and rank transformation corrects for heteroscedasticity (which is common in ecological field studies) because the variance of rank data is automatically stable (Potvin and Roff, 1993). Rank transformed data are also uniformly distributed and thus meet the assumptions of parametric statistics, which are generally more powerful than non-parametric alternatives. The results obtained show statistical significant effects among sites within ecosystems and among ecosystems (Table 2). Anyhow differences among ecosystems explained 99.9% of the variance, both for abundance and richness (Table 2). Spatial structure was therefore not included in further statistical analyses and the posterior investigation was performed at the ecosystem scale.

2.4.2. Assemblage analysis

RTU richness and abundance were determined and differences among ecosystems gauged using the Kruskal-Wallis test (a non-parametric equivalent to the multifactorial ANOVA), using the subsequent Dunn post hoc tests to check for specific differences (Sokal and Rohlf, 1995). The taxonomic composition of RTU communities between ecosystems was compared using Permutational Multivariate Analysis of Variance (PERMANOVA). In order to visualize these differences, we performed a Non-Metric Multidimensional Scaling (NMDS) analysis, with Bray-Curtis dissimilarity. The environmental variables were fitted to

Table 2

Non-parametric nested analysis of variance table testing the effect of ecosystem change on (a) carabid RTU richness and (b) carabid RTU abundance. The analyses were performed at the pitfall trap level with the 10 sites nested within the ecosystems.

Effect	DF	SS	MS	F	Var. %
a) RTU richness					
Among ecosystems	4	3,668,068	917,017	6.2*	99.884
Among sites within ecosystems	5	737,545	147,509	35.6***	0.052
Error (among traps within sites)	830	1,698,428	4143		0.064
Total	839	6,104,041			
b) RTU abundance					
Among ecosystems	4	3,220,358	805,090	5.1*	99.838
Among sites within ecosystems	5	789,355	157,871	30.6***	0.067
Error (among traps within sites)	830	2,116,942	5163		0.095
Total	839	6,126,655			

DF: degrees of freedom; SS: sums of squares; MS: Mean Square; F: Variance ratio; Var. %: variance component expressed as a percentage (Sokal and Rohlf, 1995).

* p < 0.05.
*** p < 0.001.

the two first axes of ordination by the envfit function in PAST v.3.19 (Hammer et al., 2001).

Complementary, a beta diversity partition analysis was applied to verify the dissimilarity between ecosystems. Using the partitioning framework proposed by Baselga (2010), the pairwise dissimilarity index (β_{sor}) was partitioned into two components: turnover (β_{sim}) and nestedness (β_{nes}): $\beta_{sor} = \beta_{sim} + \beta_{nes}$. This method was applied in order to evaluate whether dissimilarities in the composition of the RTU communities occurred through the substitution of some species by others (β_{sim}) or by the formation of nested subsets of more diverse communities (β_{nes}). The β_{sor} index ranges from 0 (identical species assemblages) to 1 (different species assemblages). Using this approach with our dataset allowed testing (i) differences in the values of total dissimilarity (β_{sor}) between different sites studied, but also (ii) the relative contribution of species turnover (β_{sim}) and nestedness-resultant dissimilarity (β_{nes}) in each site. Beta diversity analysis was performed using the functions *beta.pair* from *betapart* package (Baselga et al., 2017) within R 3.2.4 program (R Core Team, 2016).

2.4.3. RTU indicator value – IndVal

To determine possible RTU indicators, the single value indicator (IndVal) developed by Dufrêne and Legendre (1997) was calculated, combining specificity (patterns of relative abundance) of a given RTU in a given ecosystem with its fidelity within that ecosystem (patterns of incidence). RTU with a high specificity and high fidelity within an ecosystem are considered to achieve the highest indicator value. Only taxa with IndVal > 25% were saved in the final lists (Dufrêne and Legendre, 1997). This analysis was performed using the R 3.2.4 program (R Core Team, 2016) using the *indicspecies* package 1.7.5 (De Caceres and Jansen, 2015) with 9999 permutations, using RTU abundance.

3. Results

3.1. Preliminary analysis to Ground Beetles' communities

A total of 2378 Ground Beetles (RTU) were captured, distributed by 33 RTU. Overall, 29 RTU were identified within 859 specimens captured in Native Vegetation (NV), 25 RTU were identified from the 414 specimens captured in the Mature Secondary (MS), 18 RTU were identified within 201 individuals captured in Early Secondary (ES), 26 RTU and 590 individuals were captured in Agriculture (Ag) and 13 RTU were identified within 314 individuals captured in Pasture (Pa). The most abundant GB RTU were: *Odontocheila* sp1 (185 individuals), *Galerita* sp1 (179 individuals), *Athrostictus* sp1 (155 individuals), *Pterostichini* sp1 (151 individuals) and *Pterostichus* sp1 (142 individuals), together representing approximately 35% of total abundance (Table S4, Supplementary material).

Considering all sampled sites together, the richness estimate Jackknife 1 and Bootstrap indicated a maximum of 35.40 and 34.58 RTU, respectively (Table S5a, Supplementary material). The average of these estimates and observed RTU richness indicates that sampling efficiency was roughly 94.31% (Table S5b, Supplementary material).

3.2. Assemblage analysis, Ground Beetles RTU diversity and ecosystems

The average RTU richness revealed significant differences between ecosystems (Kruskal-Wallis nonparametric test, KW = 251.8, df = 4, p < 0.001). RTU richness was significantly higher in NV, followed by Ag, MS and finally by ES and Pa (Fig. 2A). RTU abundance had significant differences between the study ecosystems (Kruskal-Wallis nonparametric test, KW = 220.2, df = 4, p < 0.001). RTU abundance was significantly higher in NV, followed by MS and Ag (with equivalent medians), ES and lastly Pa (Fig. 2B) (see Supplementary material, Table S6, for details of the associated differences and Dunn's post-hoc values).

The results of the NMDS showed that the GB assemblages of different ecosystems could be separated from each other by ordination of the RTU composition, forming an arch changing gradually from the

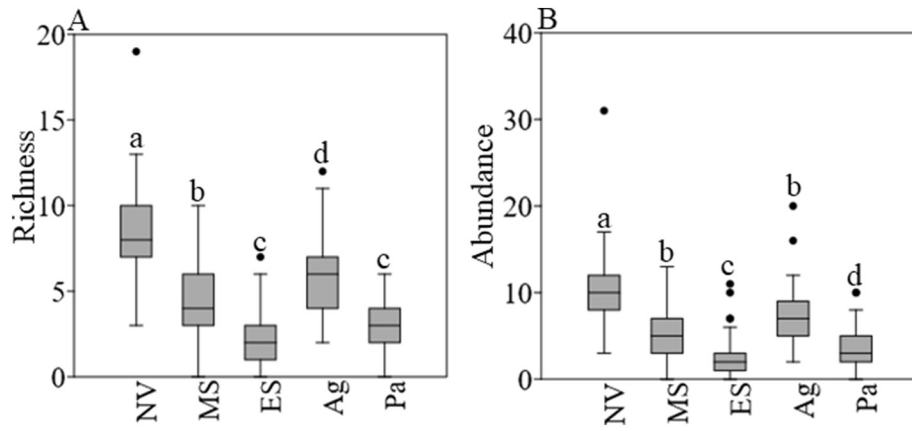


Fig. 2. Box and Whisker plots expressing the differences in Ground Beetles (A) recognizable taxonomic unit (RTU) richness and (B) RTU abundance for the different ecosystems. The values followed by different letters are significantly different according to the Dunn test. NV: Native vegetation; MS: Mature Secondary succession (vegetation with 15 years of regeneration); ES: Early Secondary succession (vegetation with five years of regeneration); Ag: Agriculture; Pa: Pasture.

most pristine (NV) to more disturbed ecosystems (Pa): RTU samples composition in the MS are clearly between reference ecosystems (NV) and Ag while ES stands clearly between the most disturbed ecosystems (Pa) and Ag also (Fig. 3). The RTU communities in Native Vegetation (NV) were mostly associated with the CBH (circumference at breast height); Mature Secondary succession (MS) was correlated with AP (abundance of plants), H (humidity) and CC (canopy cover); Native Vegetation (NV) + Mature Secondary succession (MS) were correlated with CBH (circumference at breast height), AS (abundance of shrubs) and GC (percentage of green (vegetation) cover); Early Secondary succession (ES) were correlated with AS (abundance of shrubs); finally, Pasture (Pa) was correlated with ES (percentage of exposed soil) and T (temperature) (Fig. 3).

The Permutational Multivariate Analysis of Variance (PERMANOVA) confirmed the previous results by showing that RTU taxonomic composition among the ecosystems studied was significantly different from each other ($F_{4,415} = 17.43$, $p < 0.0001$).

The beta diversity (β_{sor}) presented relatively low values (β_{sor} maximum = 0.52) (Fig. 4, Table S7a, Supplementary material). The highest dissimilarity values (β_{sor}) were found when comparing Pa with NV ($\beta_{sor} = 0.52$), Pa with ES ($\beta_{sor} = 0.44$), Pa with Ag ($\beta_{sor} = 0.40$). The lowest β_{sor} was found when comparing NV with MS (β_{sor}

= 0.11) and NV with Ag ($\beta_{sor} = 0.16$) (Fig. 4 and Table S7a, Supplementary material). The highest turnover values (β_{sim}) were found when comparing ES with Pa ($\beta_{sim} = 0.36$), and the lowest β_{sim} was found comparing NV with ES ($\beta_{sim} = 0$) (Fig. 4 and Table S7b, Supplementary material). The highest nestedness values (β_{nes}) were found when comparing NV with Pa ($\beta_{nes} = 0.39$) and the lowest nestedness was found when comparing MS with Ag ($\beta_{nes} = 0.02$) and MS with Pa ($\beta_{nes} = 0.07$) (Fig. 4 and Table S7b, Supplementary material).

3.3. IndVal results: specific RTU indicating ecosystems

Of the 33 RTU found in the present study, 12 RTU ($\approx 37\%$) were considered ecosystem indicators. According to IndVal, two RTU were significantly associated with NV, one with MS, six with Ag, and two with Pa. *Calosoma* sp1 which was an indicator of NV, was also associated with MS. No RTU was indicative for ES (Table 3).

4. Discussion

4.1. Neotropical Ground Beetles, ecosystems and environmental conditions

4.1.1. Assemblages analysis and composition of Ground Beetles' communities

Our results demonstrate that GB diversity, represented by RTU richness and RTU abundance, was not indifferent to the ongoing structural changes in the study area, reaching higher values in the pristine

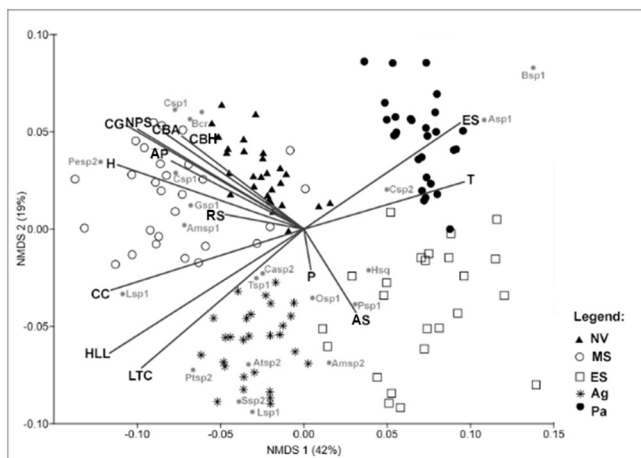


Fig. 3. Non-metric multidimensional scaling (NMDS) showing Ground Beetles (GB) recognizable taxonomic units (RTU) grouped in accordance with the ecosystems (using Bray-Curtis similarity). NV, Native vegetation; MS, Mature Secondary succession (vegetation with 15 years of regeneration); ES, Early Secondary succession (vegetation with five years of regeneration); Ag, Agriculture (Cocoa); Pa, Pasture. For a description of the abbreviations of environmental parameters, see Table S2, and for abbreviations of GB RTU see Table S3, Supplementary material.

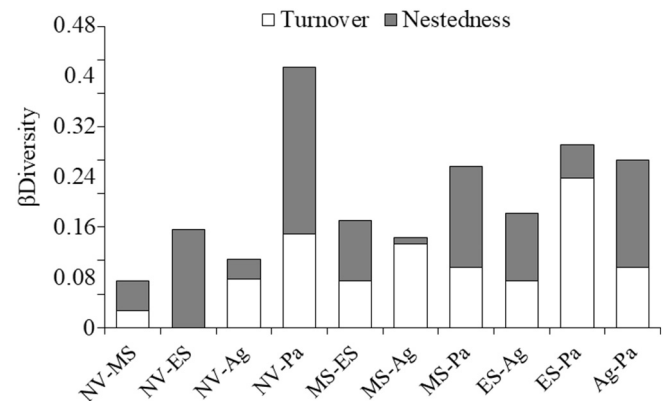


Fig. 4. Comparison of dissimilarity values for β_{sor} (overall dissimilarity), β_{nes} (dissimilarity resulting from nestedness), and β_{sim} (turnover) for the different ecosystems. NV, Native vegetation; MS, Maturing Secondary succession (vegetation with 15 years of regeneration); ES, Early Secondary succession (vegetation with five years of regeneration); Ag, Agriculture (Cocoa); Pa, Pasture.

Table 3

Recognizable taxonomic unit (RTU) selected by the IndVal method. NV, Native vegetation; MS, Mature Secondary succession (vegetation with 15 years of regeneration); ES, Early Secondary succession (vegetation with five years of regeneration); Ag, Agriculture; Pa, Pasture (only taxa with IndVal > 25% were considered).

GB RTU	Indicator value	p value	Habitat
<i>Calossoma</i> sp1	0.88	0.005	NV
<i>Brachynus crepitans</i>	0.87	0.005	NV
<i>Pelecium</i> sp2	0.73	0.001	MS
<i>Calosoma</i> sp1	0.68	0.005	NV + MS
<i>Loxandrus</i> sp1	0.79	0.005	Ag
<i>Tetracha</i> sp1	0.68	0.005	Ag
<i>Athrostictus</i> sp2	0.67	0.01	Ag
<i>Pterostichini</i> sp2	0.63	0.02	Ag
<i>Selenophorus</i> sp2	0.59	0.01	Ag
<i>Amara</i> sp2	0.53	0.005	Ag
<i>Cymindis</i> sp2	0.76	0.005	Pa
<i>Odontocheila</i> sp2	0.63	0.005	Pa

ecosystems (NV - Native vegetation), corroborating the results of Fujita et al. (2008). Although the RTU richness of cacao plantations (Ag) was considered high (even higher than MS - Mature secondary), RTU composition was quite dissimilar, as demonstrated by Permanova and NMDS. Additionally, the index β_{sor} (global beta) depicted that Pa (Pasture) was the most “poor” and divergent within all ecosystems (Table S7a, Supplementary material). Despite this, β_{sim} index (turnover) was only 22% of the NV RTU (our reference environment), i.e., circa 1/5 of species were replaced in Pa, showing that many of the RTU that inhabit NV can also be present in the other ecosystems. However, the β_{nes} (nestedness) value between NV and Pa was relatively high ($\beta_{nes} = 0.52$). Disturbed systems such as Pa and Ag may favour the presence of generalist species that tolerate the modification of their ecosystem and, thus, may allow the colonization of non-forest species from other adjacent ecosystems (Escobar, 2004). The conversion of the pristine ecosystems into pastures (Pa) or monocultures (Ag) may lead to changes in abiotic factors, with probable detrimental effects on the structure and composition of the GB communities (Medri and Lopes, 2001). These changes seem also to decrease evenness in the assemblages, leading to the dominance of some RTU since the forest RTU are unable to adapt to the conditions of disturbed ecosystems (Davis et al., 2001; Liberal et al., 2011). More disturbed environments have reduced resources and microhabitats, supporting lower richness and are dominated by generalist and/or opportunistic RTU (e.g. Moraes et al., 2013).

Nonetheless, it should be noted that high diversity itself is insufficient in indicating “healthy” ecosystems, which should be complemented by RTU composition (Zou et al., 2015). In fact, many forest RTU are not able to occur in Cocoa agroforests, despite the high diversity associated to these habitats (Schroth and Harvey, 2007). Therefore, it is clear that relying on Cocoa agroforests for the conservation of biodiversity, in vast areas of Amazonia, would be ineffective and risky for the more vulnerable and specialized groups.

4.1.2. Effects of environmental variables on Ground Beetle assemblages

Our results demonstrate that both RTU richness and abundance followed the trend expected for the disturbance gradient studied, i.e. a decrease in the respective values as the disturbance level increase. Thus, different aspects of RTU diversity suggest that GB assemblages are distinct among ecosystem, land uses and soil conditions (Moraes et al., 2013), namely because: a) litter type and litter depth are important determinants, providing GB hunting and foraging niches, protection from predators, desiccation (Pearce et al., 2003) and providing attachment points for web building (Leclerc and Blandin, 1990); b) coarse woody debris is fundamental as overwintering site and for oviposition and larval development for many GB species (Lövei et al., 2006). For example, cattle trampling in grassland areas contributes to soil compaction, resulting in reduced pathways through the litter

layer, hiding surfaces and hunting places. Additionally, reduced vegetation cover affects foraging or maintenance of water balances for GB species (Lövei et al., 2006).

It has also been suggested that the GB fauna may be affected not only by the shrub and herb cover (e.g. impeding its movement but protecting it from predation) (Brose, 2003; Taboada et al., 2008; Koivula et al., 2003) but also by plant diversity (i.e. increasing the availability of food resources) (Koricheva et al., 2000). Both the arboreal characteristics and the structural heterogeneity of the vegetation strongly influences the GB richness (Brose, 2003; Taboada et al., 2008), indicating the importance of scattered trees within agricultural areas and late successional vegetation (Koivula et al., 2002; Magura et al., 2005). The GB richness observed in forest areas and/or mature succession (NV/MS) could also be associated with the availability of niches and food resources. On the other hand, disturbed areas support mostly generalist and tolerant RTU (Kašák et al., 2017). In fact the distribution of GB along different environmental characteristics seems to demonstrate discrete associations with particular biotypes within the landscape (Silva and Hernández, 2016; Cajiaba et al., 2017a). Corroborating other studies, our results suggest richness and abundance are negatively impacted by disturbed ecosystems (Taboada et al., 2008) especially those without tree canopy cover.

4.2. Neotropical Ground Beetles as ecological indicators of anthropogenic disturbances

The increasing human population in tropical areas makes species more vulnerable to extinction due to ecosystem changes, implying effective sustainable management strategies to protect biodiversity (Hulme, 2006). For this reason, sustainable management requires a detailed understanding of the complex relationships between disturbance levels and biodiversity response trends (Hulme, 2006). Thus, the use of GB metrics as ecological indicators to evaluate the effects of anthropogenic and natural disturbances in forests has been suggested as an important and reliable tool for defining sustainable forest management (Lindenmayer et al., 2000; Work et al., 2008). In fact, policy objectives have emphasized the use of species that are vulnerable or in other ways sensitive to changing forest conditions as ecological indicators (Work et al., 2008) and the use of GB as promising ecological indicators has been supported by several studies (Niemelä et al., 2000; Larsen et al., 2003; Rainio and Niemelä, 2003; Silva et al., 2008; Work et al., 2008; Riley and Browne, 2011).

The results obtained with the IndVal signals indicator RTU for pristine and/or anthropogenic systems and might help the management and recovery of imperilled Neotropical ecosystems. In this study *Selenophorus* sp2 stands as good surrogate of disturbed environments, namely associated with Agriculture (Ag), being positively related to the number of shrubs, while *Cymindis* sp2 was good indicator for pasture areas (Pa), being positively related to the variables temperature and bare soil. On the other hand, *B. crepitans* is an indicator of native vegetation (NV) and *Calossoma* sp1 was also associated with NV + MS (Mature secondary), correlated negatively with dry areas, as identified by NMDS (Fig. 3). This relatively high number of indicator species (see Table 3) supports the applicability of RTU GB metrics as ecological indicators for the use and management of forests in the region under study, indicating that several forest-dependent species disappear if the forests are replaced by an agricultural landscape (e.g. Pa or Ag).

4.3. Perspectives and challenges on using Neotropical Ground Beetles metrics for environmental management of forest landscapes

The conservation of biological diversity has become one of the important goals of managing forests in an ecologically sustainable way (Cajiaba et al., 2017b). Ecologists and forest resource managers need measures to judge the success or failure of management regimes designed to mitigate biological diversity losses (Lindenmayer et al.,

2000). Therefore, the use of ecological indicators for monitoring environmental change/recovery becomes crucial. The use of GB represent several other environmental characteristics that have justified their relevance as ecological indicators, namely because they are influenced by temperature, moisture and shade (Thiele, 1977), food quality and abundance (Bilde et al., 2000; Bohan et al., 2011), habitat structure as reflected by the vegetation cover (Brose, 2003; Taboada et al., 2008), and substrate salts, sugars and acidity (Merivee et al., 2006; Milius et al., 2006) (see complete review in Koivula, 2011). Such studies have demonstrated that, despite the different GB species pool of particular regions, the general patterns of their functional responses to anthropogenic disturbances were very similar (Niemelä et al., 2000). This suggests the possibility of using GB, and particularly RTU universally to monitor changes in terrestrial habitats (Eyre and Luff, 1990).

Three factors not mutually exclusive, associated with the alteration of the pristine ecosystems of the Amazon rainforest, may be responsible for different trends in diversity of RTU observed in the studied ecosystems (Cajariba et al., 2017a): 1) the ability of some RTU in colonizing disturbed areas; 2) distinct ecological and ecophysiological tolerances of the RTU in relation to environmental conditions; and 3) dependence of some RTU on specific conditions and resources for reproduction. Thus, an important and urgent task is to find the GB diversity hotspots in the Amazon forest, to ensure their conservation and to manage the surroundings in the most favourable way in order to allow their populations to survive and to spread (Warnaffe and Lebrun, 2004). Despite the proved ecological relevance of GB, problems in species identification and the lack of systematic natural history studies might originate constraints for their practical use as surrogates in routine ecological studies (Freitas et al., 2006). However, in our study most of the RTU were identified until the level of morphospecies (Maveety et al., 2011). We realize that this is a simplified, preliminary and demonstrative approach, although we believe that RTU classification represents a reasonable trade-off between absolute taxonomic accuracy (which may take many years to achieve) and the ecological functional representativeness, necessary to ensure the applicability of GB diversity estimates for comparative analysis between different ecosystems (Maveety et al., 2014). In fact taxonomic surrogacy approaches have been developed to meet the short-term needs of providing scientific advice for resource managers and policy makers, reducing the time, costs and dependence on specialized taxonomists (Krell, 2004; Cotes et al., 2009; Hackman et al., 2017). Recent attempts to integrate invertebrates into biodiversity assessments utilize the taxonomic minimalism approach (e.g. Cotes et al., 2009; Obrist and Duelli, 2010; Schwerek and Dymitryszyn, 2016). This involves the use of species whose identification is based wholly on morphological differences from related species (i.e., morphospecies), or species identified to genus or other higher taxonomic ranks (Hackman et al., 2017). Our work should be complemented by ecological and behavioural studies to understand and assess the ecological status and conservation priorities of the ecosystems considered (Brown, 1997). Anyway, the responses to anthropogenic disturbances found with non-specific sampling methods are applicable per se, and should be improved in further studies with standardized methods for this group (Uehara-Prado et al., 2009). Since for hyperdiverse groups, such as Carabidae, it is difficult to obtain the complete inventory of the species that inhabit a given area, ecological assessment and monitoring based on surrogates at the community level are important procedures to address effective management of ecosystems and natural resources. Therefore, the use of holistic ecosystem indicators is considered crucial to measure and evaluate the ecological status and trends of target components and environmental conditions (Kandziora et al., 2013; Cajariba et al., 2017a).

5. Conclusions

Our approach represents a useful contribution to understand the relevance of key GB responses through a representative gradient of the

structural and functional LUCC dynamics under very complex and variable regional conditions. In fact, the present results indicate that pristine ecosystems are fundamental for conservation purposes, holding higher diversity and exclusive GB whose response was proved to be a reliable surrogate of other taxa as shown by Desender et al. (1999). Since GB composition varies according to environmental conditions and density/diversity of vertebrates' populations, GB diversity and or GB functional guilds could be used to rank the conservation status of ecosystems (Hong et al., 2017). Therefore, GB used as ecological indicators could gauge conservation actions (Koivula et al., 2002; Niemelä et al., 2007). Nevertheless, since ecological integrity of the studied ecosystems can be only partly assessed by GB indicators, our results should be complemented with information from other indicators, interactions and interferences (such as the specific agricultural practices disturbance) with precise applicability conditions. Despite the limitations inherent to a preliminary demonstration, the methodology proposed is applicable to other type of ecosystems affected by gradients of changes. In this perspective, this study represents a useful contribution for the holistic understanding of the GB community role as surrogate of other taxa responses in this region. Moreover, this approach also provides a useful starting point, allowing the precise development of more instructive protocols for environmental managers and decision-makers, based on the potential added-value of our combined metrics approach, namely in order (1) to anticipate the impacts induced by anthropogenic pressures that will characterize most of this region in the future, and (2) to provide guidance for pertinent forest restoration strategies until the effects of diverse disturbances and regime shifts in the Amazonia are reasonably understood (Cobb et al., 2007; Cajariba et al., 2017b).

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.scitotenv.2018.04.392>.

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