

Açaí palm intensification and the reorganization of woody plant assemblages at multiple spatial scales in an Amazonian estuarine forest

Maria Fabíola Barros¹

Museu Paraense Emílio Goeldi

Mário Augusto Gonçalves Jardim

Museu Paraense Emílio Goeldi

Victor Fernando Silva Soares³

Universidade do Estado do Pará

Paula Maria Melo Menezes

Universidade Federal Rural da Amazônia

Ima Célia Guimarães Vieira

Museu Paraense Emílio Goeldi

Marcelo Tabarelli (✉ marcelotabarelli9@gmail.com)

Universidade Federal de Pernambuco

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Abstract

The exploitation of non-timber forest products has been proposed as a sustainable way to exploit tropical forests, but such an opportunity remains to be examined case-by-case. We examine the impact by the intensification of the açai palm (*Euterpe oleracea*) management for fruit production via increments on palm clumps/stems density on the woody plant assemblages (adult and regenerating) in an estuarine forest landscape in the eastern Amazon region, Brazil. Across 30 forest stands açai clumps density correlated negatively with stem density and taxonomic diversity considering both adult and regenerating woody plant assemblages, but it posed a positive effect on beta diversity via species turnover. Soil fertility and forest stand accessibility played a complementary role. Moreover, açai clump density affected the abundance of particular woody plant species, including a set of species declining before the 400-açai clump density threshold imposed by current regulation. Accordingly, açai clump density influenced the taxonomic organization of woody plant assemblages at landscape spatial level. Finally, it is worth mentioning a group of 105 species, which were recorded across the regenerating assemblages but not in the adult assemblages. Among them, a large number of small-statured woody plant species typical from the forest understory (i.e., the shade-tolerant flora) such as those from *Miconia*, *Piper*, *Bactris*, *Annona* and *Geonoma* genus among many. As intensification is still spreading it can be considered a threat to the integrity of the Amazon estuarine forest. It questions the ecological sustainability of the current Amazonian “black gold” production.

1. Introduction

Tropical forests provide a wide range of ecosystem services considered essential for sustainability at multiple spatial and temporal scales (Bonan, 2008). Such a nature benefit for people includes climate regulation, carbon sequestration and storage, biomass production, water supply and purification, pollination, and biodiversity persistence (Thompson et al., 2011; Decocq et al., 2016; Liang et al., 2016; Mori et al., 2017). Moreover, the livelihood of millions of people from uncountable ethnic groups and traditional populations (i.e., a tropical forest cultural heritage) relies on forest products for subsistence and commercial activities into small and local markets (Shackleton and De Vos, 2022). Forest products offer support for a wide range of necessities, from firewood to building material and bushmeat (Naughton-Treves et al., 2007). Such a forest-dependent socio-economy is still the predominant reality across both tropical humid and dry forests globally (Shackleton and De Vos, 2022). However, ecosystem services by tropical forests and their biological and cultural heritage are threatened by conversion of old growth forests for more economically profitable land use such as the production of international commodities (e.g., soybean, beef, palm oil) and forest degradation by intense logging, hunting and fire (Barlow et al., 2016). As human populations continue to grow, particularly across tropical countries, pressure on tropical forest lands and products is likely to escalate, largely limiting the achievement of the Global Sustainable Goals.

In this context, the exploitation of non-timber forest products (NTFP; e.g., edible fruits, oil, latex, fiber, cultural artifacts and medicines) represents an opportunity to a forest-based economy able to guarantee

forest persistence, better life conditions for traditional populations and regional socioeconomic development; i.e., win-win strategies (Ruiz-Pérez and Arnold, 1996; Hernández-Barrios et al., 2014). Indeed, a review about the sustainability of NTFP harvesting states that a majority of practices is considered ecologically and economically sustainable (Stanley et al., 2012). It is important to mention that NTFPs fulfill multiple functions in supporting human wellbeing via five mechanisms: (1) direct household consumption for many of the world's poorest people (and less poor too) (Saha and Sundriyal, 2012), with this consumption reaching 60% of total household income (Babulo et al., 2009); (2) income generation for many households as additional or primary cash generation (Babulo et al., 2009) reaching over 90% of total income (Bista and Webb, 2006); (3) provide a safety-net, or insurance, for use in times of misfortune (drought, death of a breadwinner in the household) (Shackleton and De Vos, 2022); (4) some NTFPs play extensive and important roles in local cultures and spirituality (Shackleton and De Vos, 2022) and, (5) the use of NTFPs by local households represents a cash saving (Shackleton et al., 2007).

In the Amazon region, particularly in the eastern, the açai palm fruits (*Euterpe oleracea* Mart.) have been historically consumed by traditional populations to produce a beverage adopted as staple food (Brondízio, 2008). Nowadays, a myriad of food-related products based on the açai fruit pulps are already present from South America to Australia as part of a global market (Yamaguchi et al., 2015). Fruit pulp demand continues to grow leading to an intensification of fruit production via immense increments on açai palm density across natural stands managed by traditional populations, particularly in the Amazon estuarine forest, i.e., the core area of açai consumption and production (IBGE, 2021). To artificially increment palm density, forest is submitted to frequent selective thinning of non-desirable plant species, from large shrubs to emergent trees, as it reduces competition for space and light (Anderson, 1988). Intensification can increment natural açai density by many folds (from 20 to over 1000 clumps per hectare (Freitas et al., 2015)) and is able to promote a complete forest replacement by açai mono-specific stands, particularly close to households (Anderson, 1988). Intensification has been recognized to cause a decline in tree species (> 15 DBH) density and richness at forest stand level; i.e., alpha diversity (Freitas et al., 2015, 2021). However, it is reasonable to expect that intensification effects on tree assemblages are widespread, include a decline on beta diversity, impact tree recruitment and the understory assemblages composed by large shrubs and small-statured trees and palms since all this forest components are exposed to the selective thinning (Anderson, 1988; Tregidgo et al., 2020).

Here we examine the impact of the açai intensification (i.e., increment in the açai stems and clumps density) on the structure of woody plant assemblages in an estuarine-forest landscape historically devoted to açai exploitation in the eastern Amazon region (Pará state, Brazil). Adult plant assemblages (trees, palms and lianas) and regenerating assemblages were described considering a large set of community-level attributes, from plant density to taxonomic and ecological composition. Attributes were contrasted to açai density and other explanatory variable considering 30 forest stands across three landscapes. Our findings update our comprehension on the açai effects on plant assemblages, question the current regulation in support of sustainable açai fruit production, indicate the required investigation while they question the basic assumption that the extraction of NTFP is naturally sustainable as it operates as a species assembly force producing impoverished assemblages.

2. Material And Methods

2.1 Study area

This study was conducted in the state of Pará, Brazil, within three protected areas: (1) Wildlife Refuge Metr pole da Amaz nia (REVIS, 6.367.27 ha; six forest stands), (2) Comb  Island Env Protection Area/Comb  Island Extractive Reserve (APA/PAE, 1.597.2 ha; 22 forest stands) and (3) Island Grande Extractive Reserve (PAE, 922.835.7 ha; 2 forest stands). These three protected areas are part of an ecological corridor in the metropolitan region of Bel m, capital of the state of Par  (Fig. 1.). Flat and alluvial lands dominate the landscapes which are exposed to both seasonal and tide-driven floodings. Hydromorphic soils with a high clay content predominate. The climate is humid tropical, with a mean annual temperature of 27 C and average annual rainfall achieving 2000 mm without the occurrence of dry months. A tidal floodplain evergreen forest exclusively exposed to an estuarine tide regime (i.e., short-term floodings occurring daily) covers the region and thereby it has been referred as an estuarine forest (see Freitas et al., 2021). The tree and palm species flora are dominated by Fabaceae, Euphorbiaceae, and Arecaceae families, with the large tree *Ceiba pentandra* and the a a  palm as typical elements (Lau and Jardim, 2013). A a  is a canopy, light-demanding palm species (15 to 30 meters high) that occurs naturally in this forest (ter Steege et al., 2013). Biogeographically, this estuarine forest belongs to the Bel m Area of Species Endemism (BAE sensu Cracraft, 1985) and together with surrounding estuarine forests constitute the core area of a a  fruit production in the Amazon region (IBGE, 2021).

These protected areas have long been inhabited by traditional populations (i.e., riverine people or caboclos), with livelihood or familiar-based economic supported by forest extractivism and slash-and-burn agriculture; agroforestry by agro-extractivist communities (Brond zio et al., 2002, 2008). Rubber latex (from *Hevea brasiliensis*), annatto (from *Bixa orellana*), cocoa (from *Theobroma cacao*), a a  (from *Euterpe oleracea*) and valuable timber species (e.g., *Virola surinamensis*) are traditionally exploited (Anderson et al., 1985). A a  has been managed for domestic consumption but also for commercial purposes, leading to an artificial increment in clump and stem density via practices such as the elimination of undesirable woody plant species but also the spare of useful ones (Tregidgo et al., 2020). Such an intensification gained speed in the 80's and it is still in process across our focal landscapes (Freitas et al., 2021).

2.2 Woody plant assemblages

We recorded all adult woody individuals (i.e., trees, palms and lianas) with the circumference at breast height (CBH) ≥ 15 cm in 30 plots, 20 m x 20 m (0.04 ha) and across three landscapes. Within these plots, 2 subplots of 4 m x 4 m (0.0008 ha) were established to sample the regenerating plant assemblage (up to 4 m). Adults were recorded once, while regenerating individuals were recorded monthly. However, only 20 plots (out of 30) contained information for the regenerating assemblage because they were accessible during 8 consecutive months (July 2019 - February 2020), assuming that water dynamics can interfere in plant recruitment (Parolin et al., 2004). It is important to note that the location of the sub-plots was different in each sampling month. All species collected were identified to the lowest possible taxonomic

level with the aid of parataxonomists, taxonomists and by checking specimens at the regional herbarium. Vouchers are available in the João Murça Pires Herbarium and the Laboratory of Plant Ecology, both at the Museu Paraense Emílio Goeldi, Brazil. These plots were disposed into açai management gradient (i.e., number of açai clumps per ha; 25-3575 clumps per ha). We considered the forest stands with ≤ 400 clumps per ha as exposed to non-intensive management and those with ≥ 400 clumps per ha as intensive management (Queiroz and Mochiutti, 2001, EMBRAPA, 2015). Stand selection was based on semi-structured interviews with the local population, with special attention to land use. We acknowledge that (1) plots were relatively small due to limited mobility across the floodplains, and (2) the best control for the effects of açai intensification refers to landscapes covered by pristine old-growth forests rather than human-modified landscapes as we adopted here in the lack of such condition. However, both plot size and number permitted us to record over 14,000 plants and document some of the effects by açai intensification even across landscapes historically devoted to açai consumption/management (Brondízio, 2008); i.e., increments on clumps and stem density.

2.3 Forest stand accessibility and soil as explanatory variables

We calculated the distance (m) of each plot to the main river (Rio Guamá), the nearest human settlements and the closest bore e arms of the river (i.e., small drainage channels). This four Geo information was collapsed, resulting in a main axis with 69% explanatory power, adopting the method of principal component analysis (PCA). Accessibility in this study represents a proxy or measure of chronic anthropogenic disturbance (i.e., consumption of forest products) as adopted by Hawes et al. (2012) and Arnan et al. (2018) and can potentially affect woody plant assemblages in addition to açai intensification.

Soil attributes can also affect woody plant assemblage attributes (Lu et al., 2002; Baldeck et al. 2013). To estimate the soil properties of each plot, we collected five samples of topsoil (up to 20 cm deep) per plot and then pooled them into a single sample per forest stand. These soil samples were then sent to a private laboratory (EMBRAPA *Empresa Brasileira de Pesquisa Agropecuária*) to be analysed following the Brazilian protocol for soil analysis. Later, soil attributes were combined to create a single soil fertility index (*sensu* Lu et al., 2002): $SF = [Ca + Mg + K - \log(1 + Al)] * OM + 5$, where SF = Soil Fertility, Ca = Calcium (cmolc dm^{-3}), Mg = Magnesium (cmolc dm^{-3}), K = Potassium (cmolc dm^{-3}), Al = Aluminium (cmolc dm^{-3}), and MO = Organic Matter (g kg^{-1}).

2.4 Data analysis

Collinearity between açai stem density, soil fertility and accessibility (i.e., explanatory variables) was checked using the variation inflation factor (VIF) via *car* package (Fox and Weisberg, 2019); as they were not collinear (VIF values < 2; açai stem abundance = 1.34, soil fertility = 1.30 and accessibility = 1.05), all of them were included in the subsequent analyses (Legendre and Legendre, 1998; Jou et al., 2014). It is important to mention that we consider both açai stem density and açai clumps density in different analyses. We decided to use açai stems density because we consider it more accurate variable as

compared to açai clumps density (see Freitas et al., 2021) as açai clumps can support one or several stems. The explanatory-variable effects on the attributes of woody plants assemblage were assessed using generalized linear models (GLMs) from the *stats* package. For the GLMs, Binomial distribution was used for count variable (density) and Gaussian distribution was used for continuous data ($q = 0$, $q = 1$, $q = 2$ and NMDS1) (Venables and Smith, 2018). Residues were checked via *DHARMa* (Hartih, 2022) and *hnp* (Moral et al., 2017) packages, while under/overdispersion were checked dividing the residual deviance by the degrees of freedom. Here, NMDS axis 1 was used as a proxy for woody species composition.

We calculate taxonomic beta diversity and their components based on a pairwise method. Both Sorensen (presence) and Bray-Curtis (abundance) dissimilarity coefficients were used to calculate the total beta diversity, turnover, balanced variation, nestedness and abundance gradient using the *betapart* package (Baselga, 2016). To understand how açai clumps density affects woody plant-assemblage beta diversity clump density was adopted as an environmental distance matrix, which was contrast to diversity scores. In this context, a Euclidean distance was used as a pairwise index for each site and a Person Partial Mantel test with 999 permutations was run with a third spatial distance matrix. The environmental, spatial and Pearson Parcial Mantel test were performed using *vegan* package (Oksanen et al., 2019).

To examine how açai clumps density influences species abundance, positively (increasing the abundance) or negatively (reducing the abundance), a TITAN test was adopted via *TITAN2* package (Beker and King, 2010). Finally, to exam the taxonomic organization of both adult and regenerating assemblages and their response to açai density, we applied a non-metric multidimensional scaling (NMDS) followed by ANOSIM (Oksanen et al., 2019) based on Bray-Curtis dissimilarity matrices using the *vegan* package.

3. Results

A total of 1169 adult woody plants from 109 species and 34 families were recorded across all the 30 estuarine forest stands (Table S1), including 493 açai clumps. Trees dominated adult assemblages by accounting for 92.59% of all species and 52.52% of the individuals by excluding açai clumps. Stem density and species richness achieving 563.3 ± 298.1 and 18.6 ± 9.3 per 400 m^2 (mean $\pm$ SD) as açai clumps were excluded from adult assemblages. More than half of the species (55.04%) were considered rare (i.e., one or two individuals across the 30 forest stands). In fact, woody plants assemblages were dominated by few species, with the 10 most abundant species representing nearly 70% of individuals (Fig S1.). Woody plants density varied from 75 to 1125 and it was reduced by half as açai stems density increased up to 3575 (density: $\chi^2 = 5.87$, $p = 0.01$; Fig. 2a). The same occurred with species richness ($q = 0$: $\chi^2 = 7.22$, $p = < 0.01$ (Fig. 2c)) and typical species ($q = 1$: $\chi^2 = 5.26$, $p = 0.02$ (Fig. 2e)). Only dominant species ($q = 2$) were not related to açai clumps ($\chi^2 = 3.09$, $p = 0.07$ (Fig. 2g)). These same variables were positively related to accessibility (density: $\chi^2 = 5.70$, $p = 0.01$ (Fig. 2b; $q = 0$: $\chi^2 = 10.63$, $p = < 0.01$ (Fig. 2d); $q = 1$: $\chi^2 = 12.67$, $p = < 0.01$ (Fig. 4f) and $q = 2$: $\chi^2 = 9.23$, $p = < 0.01$ (Fig. 2h)). None of the response variables were correlated with soil fertility (Table S2).

Moving to the regenerating assemblages, 13,378 individuals were recorded from 168 species and 46 families across the 20 forest stands (Table S3). By excluding açai clumps, trees also dominated the regenerating assemblages: 92.26% of all species and 76.55% of the individuals. Stem density and species richness achieved 343.8 ± 108.4 and 48.4 ± 17.4 per 16 m^2 respectively by excluding açai clumps/stems. Similar to the adult tree assemblages, many species (36.3%) were considered rare across the 20 forest stands with regenerating assemblages dominated by few species, i.e., the 10 most abundant species also representing nearly 70% of individuals (Fig S2.). Plant density across regenerating assemblages varied from 170 to 568.1 and it was positively related with soil fertility (density: $\chi^2 = 8.19$, $p = < 0.01$; Fig. 3b). In fact, plant density doubled in more fertile forest stands. On the other hand, açai stem density negatively affected species richness ($q = 0$: $\chi^2 = 4.21$, $p = 0.04$ (Fig. 3c)), typical species ($q = 1$: $\chi^2 = 16.02$, $p = < 0.01$) and dominant species ($q = 1$: $\chi^2 = 18.36$, $p = < 0.01$; Fig. 3c, e and g). By considering the extremes açai stem density, species richness, typical and dominant species dropped by almost half. None of the woody plant assemblage attributes correlated with accessibility (Table S2).

Açai clump density positively affected beta diversity (i.e., pairwise dissimilarity) across adult woody plant assemblages by considering both incidence and abundance-based dissimilarity with taxonomic beta-diversity being represented more by turnover (Fig. 4b) and balanced variation (Fig. 4e) than nestedness (Fig. 4c) and abundance gradient (Fig. 4f; Table S4). In contrast, açai clump density did not affect beta diversity across regenerating forest stands (Fig. 5; Table S4), but it was geographical distance (Table S4).

Moreover, açai clump density affected the abundance of particular woody plant species positively and but also negatively considering both adult and regenerating woody plant assemblages: 11 species from adult (Fig. S3) and 6 species in the case of regenerating assemblages (Fig. S4). It is worth mentioning that with exception to *Quararibea guianensis* (Malvaceae) and *Astrocaryum murumuru* (Arecaceae), this team of species declined before the 400-açai clump density threshold (Fig. S3). Moreover, all woody species that responded positively to increments on açai clump density occurred widespread across forest stands. Accordingly, açai clump density influenced the taxonomic organization of both adult ($\chi^2 = 14.83$, $p = < 0.01$) and regenerating woody plant assemblages ($\chi^2 = 37.57$, $p = < 0.01$; Table S2). Finally, adult and regenerating plant assemblages exhibited distinct taxonomic composition (ANOSIM: $R = 0.93$, $p = < 0.01$; Fig. S5) by sharing 45 species ($\cong 17\%$) of the whole flora. It is worth mentioning a group of 105 species, which were recorded across the regenerating assemblages but not in the adult woody plant assemblages. Among them, a large number of small-statured woody plant species typical from the forest understory (i.e., the shade-tolerant flora) such as those from *Miconia*, *Piper*, *Bactris*, *Annona* and *Geonoma* genus among many.

4. Discussion

Our results suggest that açai stems and clumps density (i.e., a measure of açai management intensification) reorganize woody plant assemblages in the estuarine forest landscapes devoted to açai fruit production. Açai intensification reduces the density and alpha taxonomic diversity of both adult woody plant assemblages inhabiting forest canopy and the emergent layer, but also of those

assemblages inhabiting forest understory. We refer to assemblages by small-statured tree species, palms and large shrubs. In fact, such understory assemblage appears to be much more diverse than canopy and emergent assemblages, as they combine individuals from species occurring across all forest vertical habitats. On the other hand, açai intensification appear to increment taxonomic beta diversity, particularly due to species turnover in response to reduction on alpha diversity and increasing levels of rarity as açai density increases. The estuarine forest also supports an ecologically diverse team of sensitive species (from large trees to shrubs), which decline before the regulatory threshold of 400 açai clumps ha, while a reduced number of species respond positively. Accordingly, açai intensification reorganize plant assemblages taxonomically but also ecologically towards impoverished species sets, probably including the assemblages responsible for forest regeneration and structural integrity (i.e., saplings from canopy and emergent tree species).

Our findings reinforce the notion that açai intensification promotes de establishment of low-density and impoverished adult tree assemblages, which become taxonomically distinct as açai density increases as already documented (Freitas et al., 2015, 2021). However, we offer evidence suggesting that species loss among adult tree assemblages occurs across forest stands still supporting relatively low-density açai stands; species loss occurs as soon as intensification starts to alter açai abundance. On the other hand, few species respond positively to açai intensification; i.e., more losers than winners sensu Tabarelli et al. (2012). Furthermore, our findings indicate the occurrence of a diverse woody plant assemblage inhabiting forest understory, which combines saplings from canopy tree species, but also a variety of large shrubs, small-statured tree and palm species; i.e., a typical understorey flora across tropical forests (Whitmore, 1984; Gentry and Emmons, 1987). This neglected forest component in açai-related studies, but not necessarily less diverse than the canopy woody flora, appears to be negatively affected by açai intensification and thus call attention to other forest components being impacted by açai intensification. In fact, a reasonable portion of the Amazonian woody plant flora consists of shrubs, small-statured trees and palms inhabiting forest understory by being shade-tolerant species; i.e., the whole life-cycle in the shaded forest understory (Bloor and Grubb, 2003). The few estimates available report a minimum 16% of tree species flora belonging to this ecological group in the Amazon region (Assis and Wittmann 2011). However, by considering herbs, shrubs and small palms, the understory flowering species flora is expected to achieve over 20% of all species (see Costa and Magnusson 2002). Unfortunately, there is little information about the Amazon estuarine forest, even for the tree flora, although this unique forest represents the core area of açai production and intensification (Brondízio, 2008) and an ecologically unique component of an area of species endemism considering an immense number of taxa, from insects to primates (Moraes et al., 2020; Silva et al., 2022).

Although we did not explicitly address the baseline mechanisms driving the changes in plant assemblages, it has been argued the elimination of undesirable trees in order to provide better microclimatic conditions and the space demand to increment clump density is the main driver for the impoverishment tree assemblages experience as açai clump density increases (Freitas et al., 2015, 2021). Such a forest thinning or coppicing is one of the prescribed management procedures in order to increment açai fruit productivity (Anderson, 1988) and in a large extent responds for a complete

degradation of forest spatial structure and the impoverishment of tree assemblages, although it has never been experimentally tested. In this context, decreasing species richness and increasing rarity as açai density increases is probably the mechanism behind higher beta diversity as intensification proceeds, a similar phenomenon in response to habitat loss and fragmentation resulting in low gamma diversity as already documented; i.e., species richness at landscape/regional level (Sfair et al., 2016).

In fact, it is reasonable to propose that changes promoted by forest thinning in forest physical structure and woody plant assemblages will alter a myriad of processes limiting further plant recruitment upon which thinning operations will occur. Three examples, elimination of large trees probably results into much more illuminated habitats filtering shade-tolerant species while favoring light demanding such as sun-loving grasses and açai recruitment (Rutishauser et al., 2016; Pinho et al., 2020). Similarly, the elimination of large buttressed trees, including their large dead trunks over the ground, is likely to increase seed runoff during water descending (Junk et al., 2010). Increased runoff causing lower soil fertility is another potential effect by “açaization”, the local expression to describe intensification leading to the emergence of monospecific stands, which are not permitted according to current regulation but continue to proliferate (Hiraoka, 1993; Freitas et al., 2021). As we documented here soil fertility is positively associated with several community- and ecosystem-attributes across tropical forests, including species richness, forest productivity and aboveground biomass with cascading effects relative to support for herbivores/frugivores assemblages (Gentry and Emmons, 1987; Peres, 2008; Cleveland et al., 2011). It is interesting that forest stand accessibility was positively with adult species richness. Such a priori unexpected result may in fact result from a higher number of useful species being spared close to households such as *Theobroma cacao*, *Hevea brasiliensis*, *Spondias mombin*, *Mauritia flexuosa* (Nogueira et al., 2005; Tregidgo et al., 2020). Additionally, açai dropped fronds may cover seedlings on the ground and reduce recruitment of woody plant species as already documented for other palm species (Peters et al., 2004). Finally, there is evidence suggesting that açai intensification disturbs plant pollination (Campbell et al., 2018) and seed dispersal by frugivorous vertebrates (Moegenburg and Levey, 2002). We refer to both abiotic and biotic changes affecting plant recruitment and then contributing to forest impoverishment as “managed forest/landscapes” experience the intensification trajectory.

In this context, the current regulation requiring a maximum of 400 açai clumps per ha plus the persistence of 50 palm trees of other species and 200 trees hardwoods, 40 thick (> 45 cm DBH), 40 medium (20 to 45 cm DBH) and 120 thin (5 to 20 cm DBH) is highly welcome (see Queiroz and Mochiutti, 2001). However, to acknowledge the açai management as sustainable and then stimulate legislation compliance by riverine traditional producers, it is highly recommendable to examine in which extent these thresholds are able to guarantee forest integrity considering and all ecosystem services this forest provide, from biodiversity persistence to the apport of organic material (e.g., leaves, flowers, fruits, seeds) to a highly productive estuarine system (Sodré et al., 2011). Accordingly, it is required a reference ecosystem sensu (Suganuma and Durigan, 2015) as a baseline (probably landscapes still covered by old-growth forests) to investigate forest changes and responses to the whole package involving the açai management as it involves more forest thinning and increments on açai density.

It is true the estuarine forest has long been submitted to the extraction forest products, including management to favor cocoa, rubber tree and açai; i.e., a typical land-sharing approach (Green et al., 2005). Like in the past, such a “disturbed forest” currently experiencing açai intensification continues to be exposed to a management exclusively restricted to açai forestry/agricultural demands rather than a management able to guarantee both forest products but also forest integrity. Such a broad perspective of forest management is likely to demand land-use zoning and the establishment of *strictu sensu* protected areas in addition to more comprehensive and robust set of guidelines to manage açai stands considering multiple spatial scales (e.g., land-sparing vs. land-sharing). This more ambitious approach remains neglected due a lack of (1) a substantial research effort focused on the multiple impacts by açai intensification (from local to regional scales) such as the collapse of the understory flora, and (2) a detailed reference on the estuarine forest relative to patterns of species diversity, community organization, physical structure and ecosystem functioning, including primary productivity and biomass.

In synthesis, we shall recognize that rather than the extension of the flooding period (Parolin and Junk, 2002), açai intensification currently represents the major driver of plant species assembly and community organization operating across several floodplain forests. Applied, açai intensification moves forward into a “blind context” and thus it can be considered a major threat to estuarine/floodplain forests, offering a disturbing message about the role played by NTFPs as an opportunity for sustainability. In fact, the “açai case” inform us that NTFPs can only be sustainably extracted in the case appropriate public policies/regulations are incorporated to guarantee traditional low impact forest management approaches rather than market-oriented management stimulating intensification.

Declarations

Ethical Approval

Not applicable.

Competing interests

No conflicting/competing interests.

Authors' contributions

MFB, MAGJ, ICGV and MT conceived the idea and secured funding; MFB, MAGJ, VFSS and PMMM collected data. MFB was responsible for data analysis and writing. All authors reviewed and provided constructive feedback on the manuscript.

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Availability of data and materials

The data that support the findings of this study are available via request to the main author (Maria Fabíola Barros).

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Figures

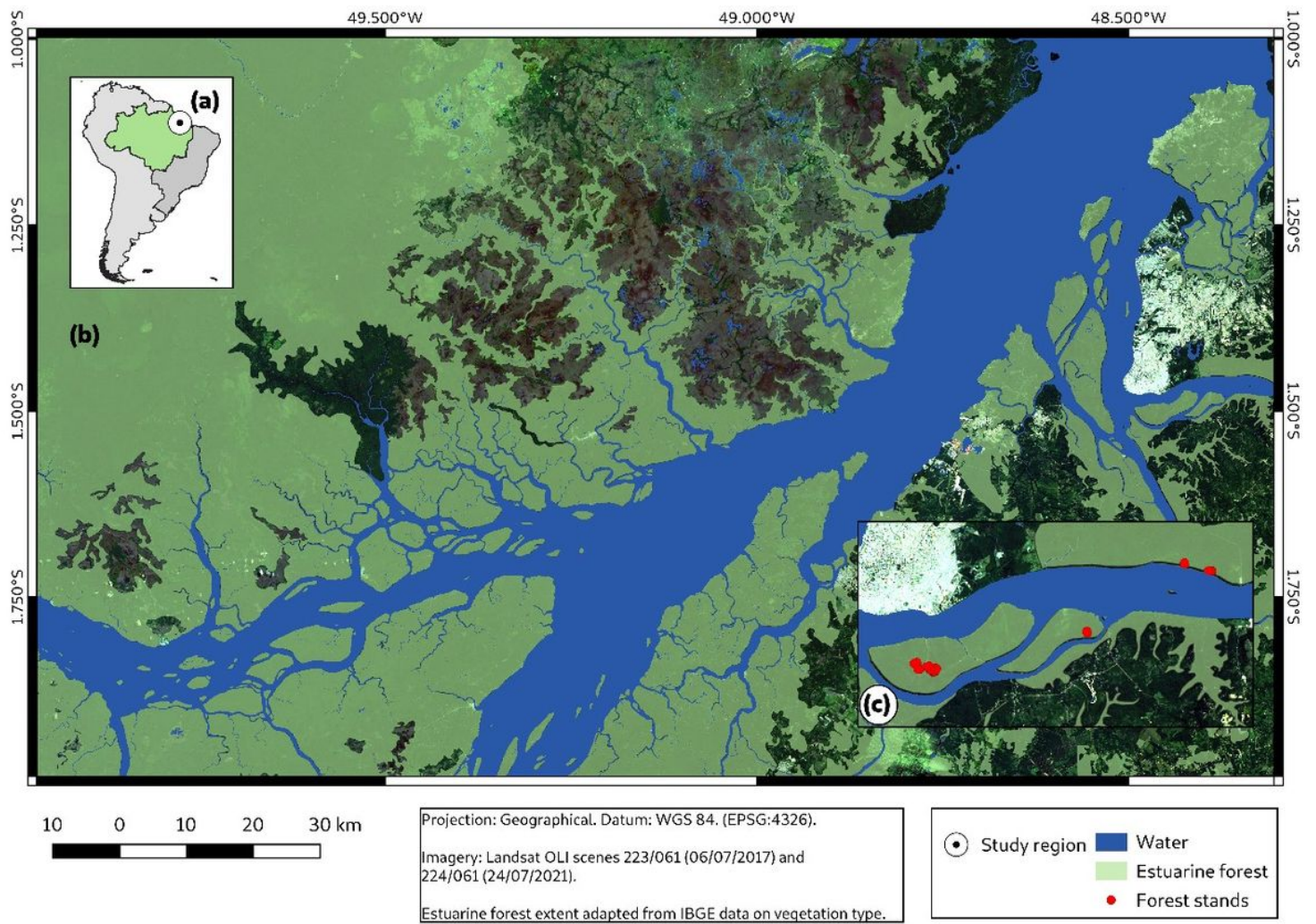


Figure 1

Location of the study area in the Amazon forest, northern region of Brazil (a), with emphasis on the estuarine forest (b), and details of the 30 forest stands studied (c).

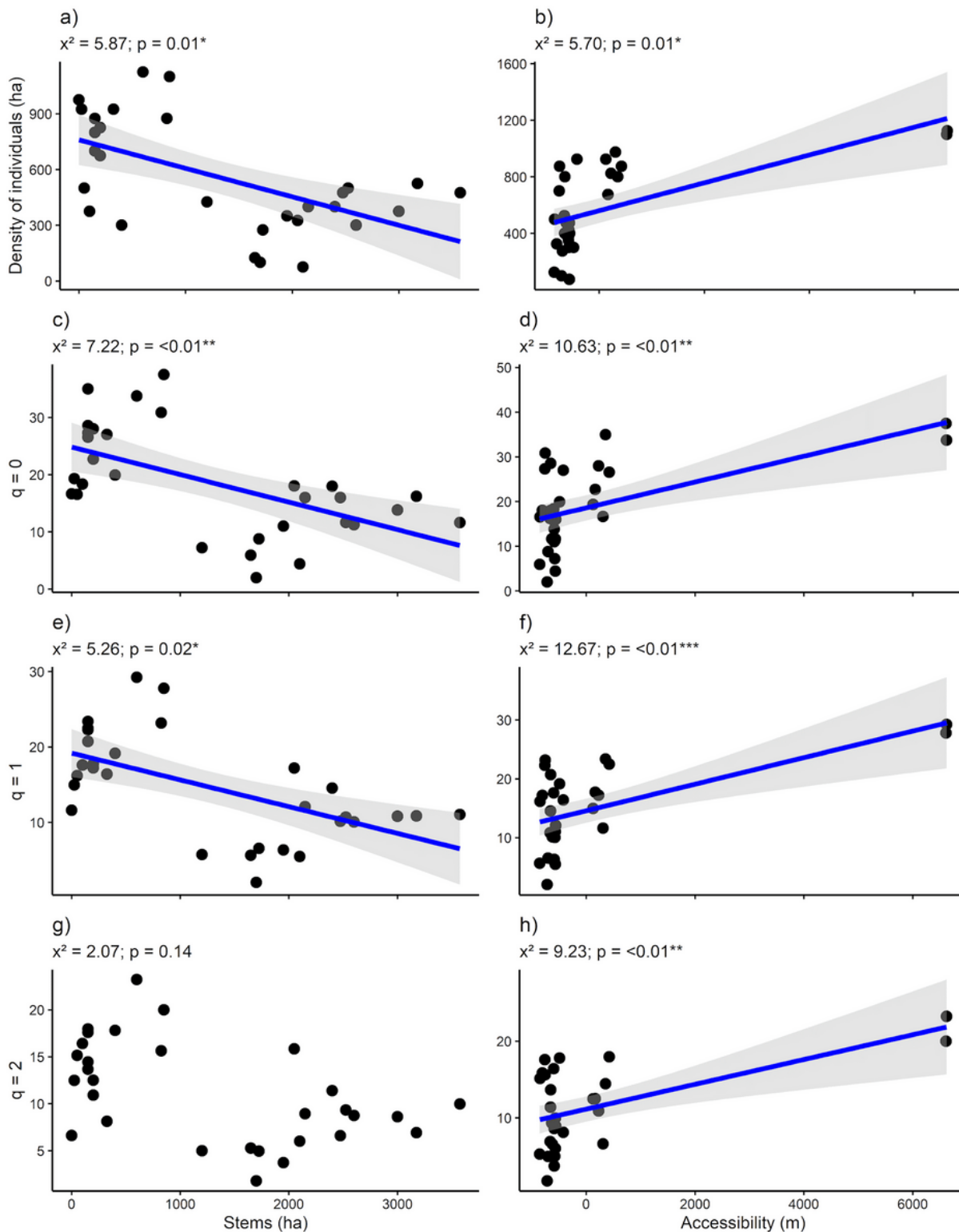


Figure 2

Generalized linear models between explanatory variables (stems (ha) and accessibility (m)) and response variables (density of individuals (ha), species richness ($q = 0$), typical species ($q = 1$) and rare species ($q = 2$)) in relation to adult woody assemblages of an Amazonian estuarine forest. Positive signal indicate significant positive relationships, whereas negative signal indicate the opposite trend. $^*p < 0.05$; $^{**}p < 0.001$; $^{***}p < 0.0001$.

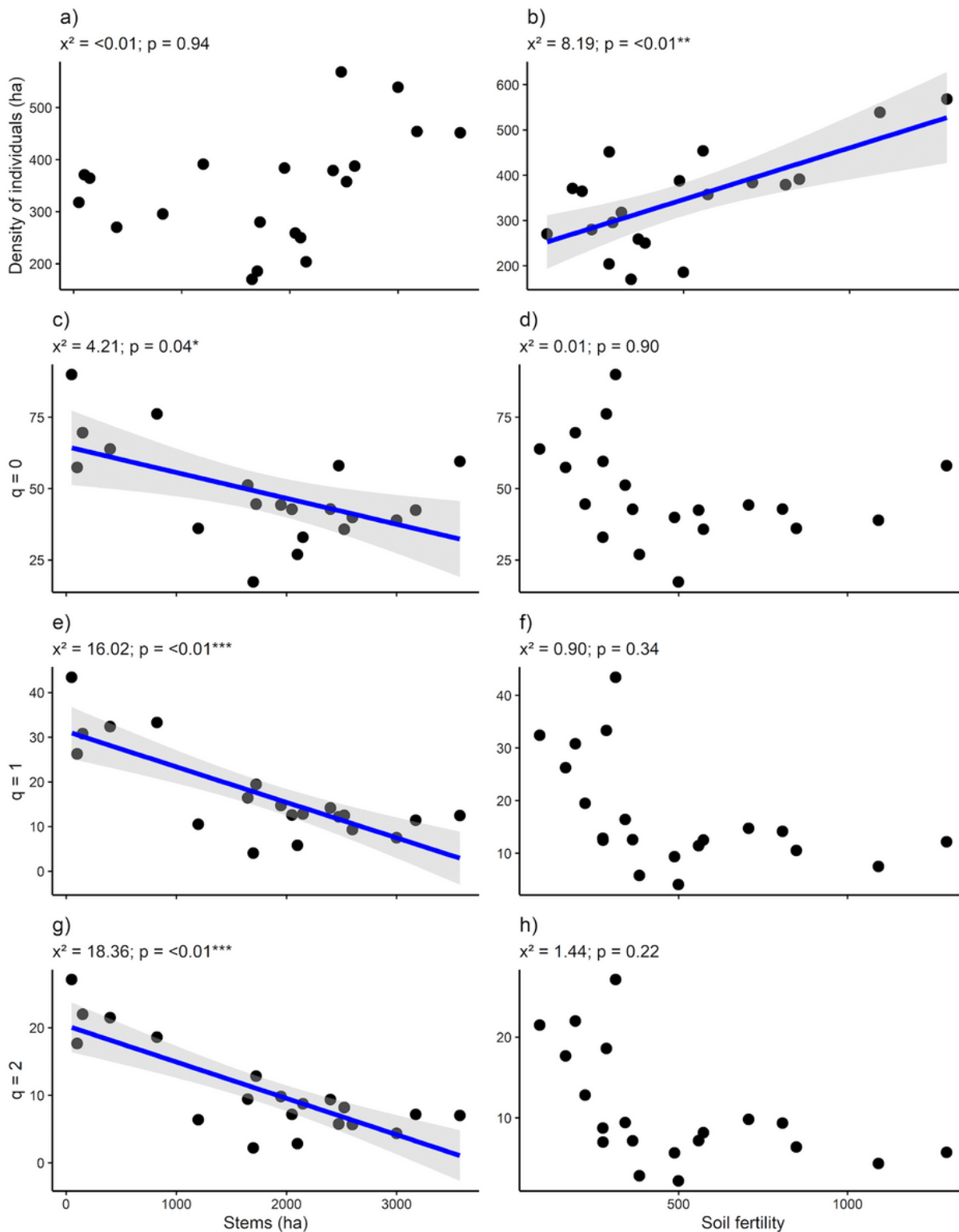


Figure 3

Generalized linear models between explanatory variables (stems (ha) and accessibility (m)) and response variables (density of individuals (ha), species richness ($q = 0$), typical species ($q = 1$) and rare species ($q = 2$)) in relation to regenerating assemblages of an Amazonian estuarine forest. Positive signal indicate significant positive relationships, whereas negative signal indicate the opposite trend. $^*p < 0.05$; $^{**}p < 0.001$; $^{***}p < 0.0001$.

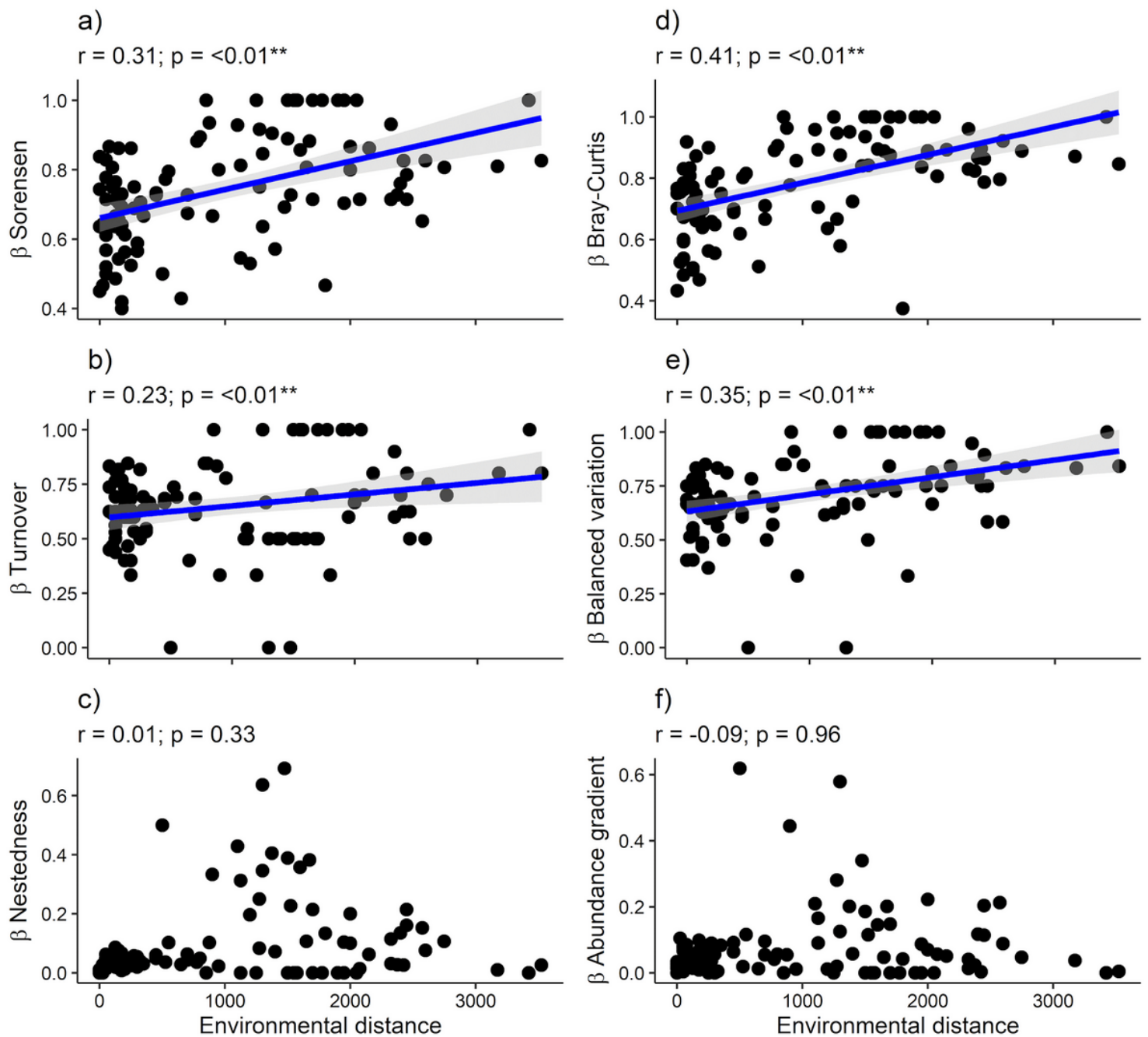


Figure 4

Relationship between biotic dissimilarity and environmental distance (açai clumps density) using incidence (a - c) and abundance (d -f) in relation to adult woody assemblages of an Amazonian estuarine forest. Total dissimilarity (β Sorensen and β Bray-Curtis) and its components (β Turnover, β Nestedness, β Balanced Variation and β Abundance gradients). Both determination coefficients (r) and significance (p) for each relationship are shown.

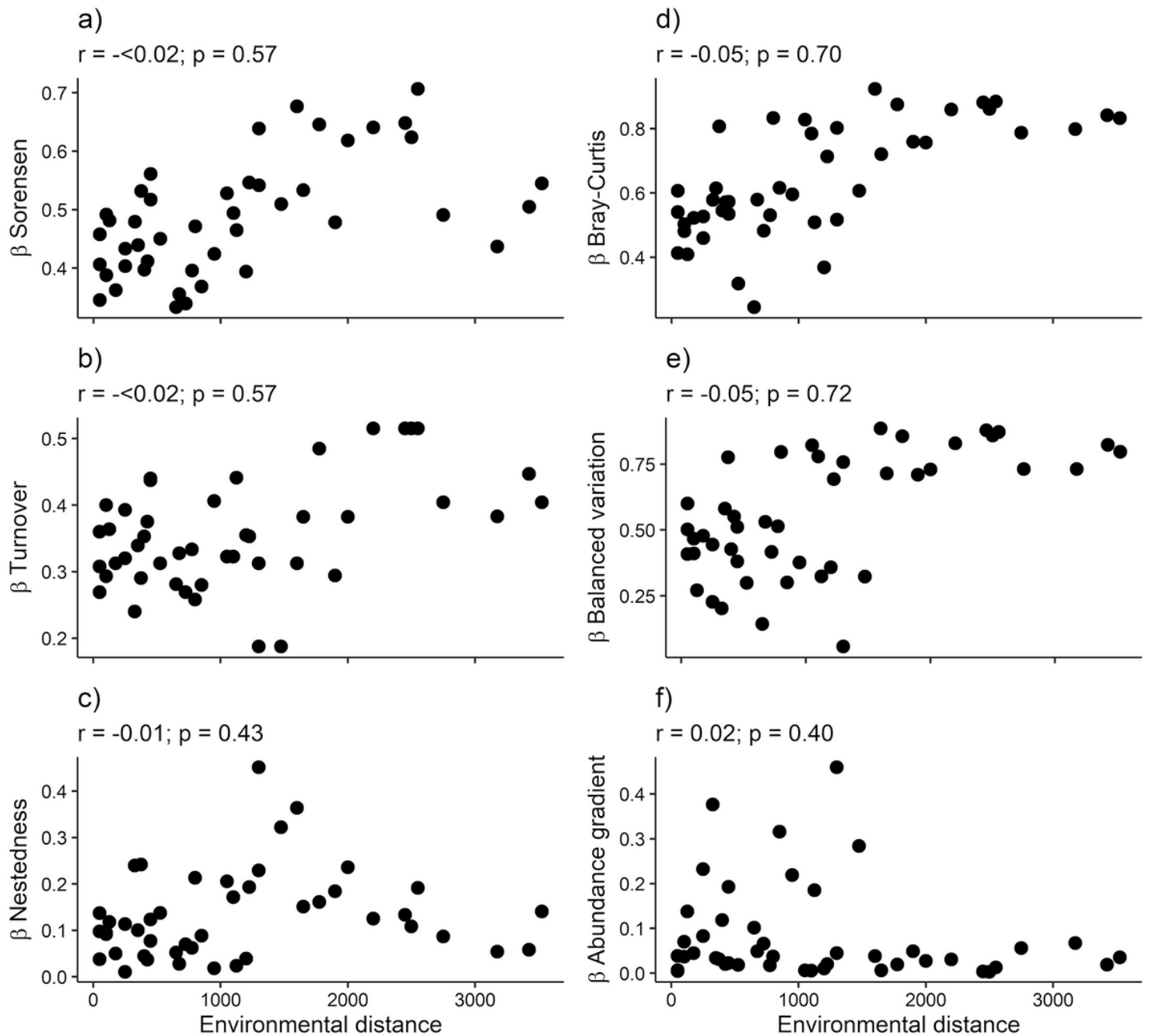


Figure 5

Relationship between biotic dissimilarity and environmental distance (açaí clumps density) using incidence (a - c) and abundance (d -f) in relation to regenerating assemblages of an Amazonian estuarine forest. Total dissimilarity (β Sorensen and β Bray-Curtis) and its components (β Turnover, β Nestedness, β Balanced Variation and β Abundance gradients). Both determination coefficients (r) and significance (p) for each relationship are shown.

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