UNIVERSIDADE FEDERAL DO RIO GRANDE DO NORTE
PROGRAMA DE PÓS-GRADUAÇÃO EM ECOLOGIA

ADRIANA PELLEGRINI MANHÃES

RELAÇÃO ENTRE A BIODIVERSIDADE DE PLANTAS E OS SERVIÇOS DO ECOSSISTEMA NA CAATINGA

NATAL, RN
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Tese apresentada ao programa de Pós-Graduação em Ecologia da Universidade Federal do Rio Grande do Norte, como parte das exigências para a obtenção do título de Doutor em Ecologia.

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Resultado: ____________________

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SUMÁRIO

INTRODUÇÃO GERAL

Os serviços do ecossistema são benefícios derivados de processos ecológicos e propriedades do ecossistema e são essenciais para o bem-estar humano. Com a crescente degradação de ambientes naturais e o desmatamento para conversão do uso da terra (principalmente agricultura e agropecuária), muitas espécies vem se extinguindo e assim, o papel que estas exercem no ecossistema também é perdido. Muito tem se discutido na literatura sobre o papel da biodiversidade na função do ecossistema e também, nos serviços do ecossistema. O entendimento de quais fatores podem afetar a provisão dos serviços do ecossistema pode auxiliar à um manejo mais adequado para que estes sejam preservados para as futuras gerações.

O uso da terra é um dos principais fatores causadores do desmatamento em todo o mundo, causando prejuízos imensuráveis, como a perda de diversas espécies, tanto de plantas como animais. Pesquisas na área de *Biodiversity and Ecosystem Functioning* (BEF) vem elucidando a importância da diversidade de plantas na produtividade primária, estoque de biomassa e no uso de recursos inorgânicos do solo. Estas propriedades do ecossistema estão relacionadas com a provisão dos serviços de captação e estoque de carbono, e também, de fertilidade e ciclagem de nutrientes no solo. Duas hipóteses são utilizadas para explicar os mecanismos derivados da relação entre a biodiversidade de plantas e o funcionamento do ecossistema: a hipótese da diversidade e da razão-massa. A primeira está relacionada com o uso complementar dos recursos pelas plantas, onde comunidade mais diversas funcionalmente tem maior complementaridade que comunidades menos diversas. Já a hipótese da razão-massa explica que a função das espécies mais abundantes na comunidade pode ter mais efeito no funcionamento do ecossistema que a diversidade das espécies.

Muitos estudos na área de BEF tem dado suporte a hipótese de diversidade, mas sua maioria é desenvolvido no campo experimental e pouco se sabe ainda sobre o papel da biodiversidade de plantas no funcionamento do ecossistema e seus serviços em sistemas naturais antropizados, e também em uma escala de paisagem. Em pequena escala, comunidades de planta em condições naturais já possuem um certo grau de distúrbio, principalmente no bioma Caatinga, onde em torno de 45% já se encontra desmatado ou com algum impacto antropogênico. Portanto, incluir o fator de distúrbio influenciando estas comunidades torna-se essencial para entender como a cobertura da vegetação e a biodiversidade de plantas respondem ao distúrbio e, ao mesmo tempo, como afetam as propriedades do ecossistema. Este foi o principal objetivo do primeiro capítulo desta tese de doutorado.

Já em uma escala maior, à nível regional, não há nenhum estudo que tenha analisado e estimado os serviços do ecossistema para o bioma Caatinga, além de suas relações espaciais com a biodiversidade de plantas. Essas informações podem amparar e subsidiar o planejamento sistemático para conservação da natureza, onde áreas prioritárias são selecionas baseadas em análises espaciais objetivando aumentar a efetividade da conservação por meio da complementaridade destas áreas. Assim, entender a congruência espacial entre a biodiversidade de plantas e serviços do ecossistema e avaliar como as atuais unidades de conservação do bioma Caatinga estão ou não inserindo as áreas de maior valor destes alvos (*hotspot*) foram os objetivos do segundo capítulo desta tese de doutorado.

Muitas pesquisas tem evidenciado uma correlação negativa (*trade*-off) entre biodiversidade e serviços do ecossistema em uma escala maior, a qual é utilizadas na tomada de decisão por conservacionistas. Assim, torna-se importante incluir os serviços do ecossistema como alvos na conservação, pois utilizando somente a biodiversidade como alvo na seleção de áreas prioritárias pode não embarcar os serviços de uma forma igualitária. Outro *trade-off* tem sido evidenciado em trabalhos de conservação da natureza, explicitando que muitas áreas importante para conservação da biodiversidade co-ocorrem com áreas de alta vulnerabilidade, como por exemplo, áreas de maior valor econômico para agricultura ou para expansão urbana. Estas áreas possuem maiores custos de oportunidade e podem ser evitadas, quando os objetivos da conservação não podem ser atendidos juntamente com os objetivos de desenvolvimento socioeconômico, como a categoria de proteção integral, por exemplo. Deste modo, o terceiro capítulo desta tese de doutorado objetivou selecionar áreas prioritárias para conservação no bioma Caatinga utilizando quatro cenários de priorização: sem custo de oportunidade, com custo econômico, com custo social e com custo socioeconômico.

Espera-se que esta tese de doutorado venha contribuir para o avanço na pesquisa sobre as relações entre biodiversidade de plantas e serviços do ecossistema, de modo que as informações possam elucidar um maior entendimento sobre o assunto. Além disso, ressaltamos a importância de sua conservação para o bem estar humano em uma escala de paisagem e, também, o desenvolvimento de um manejo mais sustentável da vegetação na caatinga para evitar maiores perdas dos serviços ecossistêmicos e diversidade de plantas em uma escala local.

CAPÍTULO I

PLANT COVER MEDIATES NEGATIVE EFFECTS OF ANTHROPOGENIC DISTURBANCE ON ECOSYSTEM PROPERTIES IN THE BRAZILIAN CAATINGA

Plant cover mediates negative effects of anthropogenic disturbance on ecosystem properties in the Brazilian Caatinga

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Summary

1. Anthropogenic disturbance can have negative impact on ecosystem properties that provide important ecosystem services for human well-being. However, how plant community mediates this trade-off is still unknown.

2. A gradient of anthropogenic disturbance (livestock density, selective logging and clear-cutting) was assessed to understand its direct and indirect effects on ecosystem properties (standing biomass, litter biomass, soil water retention, soil carbon, soil nutrients and multifunctionality). Indirect effects were measured by functional structure of plant community (community weight mean, functional diversity and richness) and plant cover. We used structural equation modeling to evaluate data suitability with the theoretical model developed to the study system.

3. Anthropogenic disturbance mainly affects the ecosystem properties and the multifunctionality through the loss of plant cover. Functional structure had weak influence on properties, however, functional diversity and richness were also influenced by plant cover. Total effect (sum of direct and indirect effects) of anthropogenic disturbance was negative for all ecosystem properties and multifunctionality with exception for soil nutrients.

Synthesis and applications: In a long period of time, the loss of plant cover caused by anthropogenic disturbance derived from economic activities in the Brazilian Caatinga may lead to desertification, that is the complete loss of the function of the land. More sustainable management practice that prioritizes the plant cover maintenance should avoid the complete loss of the ecosystem properties and multifunctionality**.**

Key-words: direct and indirect effects, functional structure, mass-ratio and diversity hypothesis, multifucntionality, structural equation modeling.

Introduction

There is a solid knowledge about the influence of biodiversity (species richness) on ecosystem functioning corroborated by numerous experiments worldwide (Hooper et al. 2005; Balvanera et al. 2006; Cardinale et al. 2011). When the magnitude of biodiversity effect from those experiments was compared to other factors such as environmental change and human-caused drivers, biodiversity had more influence (Hooper et al. 2012; Tilman et al. 2012). However, in natural systems, those relative factors presented stronger effects than biodiversity to explain ecosystem functionality. In natural grasslands, species richness had the smallest influence on biomass production and stronger effects arose from abiotic factors and disturbances (Grace et al. 2007). In a semiarid system, perennial plant cover is more influential on soil ecosystem properties related to ecosystem functioning than other biotic attributes such as richness and evenness (Maestre et al. 2010). The understanding of which factors are affecting the ecosystem functionality in natural and disturbed systems is important to develop better management practices.

Biodiversity has multiple dimensions and beyond the taxonomic dimension (species richness) the functional attributes of plant community have been evocated to explain the biodiversity effects on ecosystems functioning (Garnier et al. 2004; Laliberté & Tylianakis 2012; Lavorel & Grigulis 2012). Multifunctionality that is the provision of multiple functions is also explained by functional biodiversity (Mouillot et al. 2011). These functional attributes are derived from functional traits, that are the physiological and morphological features linked with species performance in different environments (Díaz & Cabido 2001). Response-effect traits framework integrates community response to changes (disturbance) and how the modified community influences the ecosystem processes through the modification of functional structure of plant community (Lavorel & Garnier 2002; Suding et al. 2008). This framework assumes that the functional traits are the main mediator from disturbance and ecosystem properties. However, in semiarid systems, plant cover can also explain and mediates this relationship as it was related as a key element to monitor desertification process that is the loss of ecosystem process and functions of the system (Maestre & Escudero 2009).

The aim of this study was to assess the effects of anthropogenic disturbances on ecosystem properties and multifunctionality and how the functional structure of plant community and the plant cover mediate this relation. We defined functional structure as the distribution of species and their abundance in the functional space (Mouillot et al. 2013) and ecosystem properties as one component of ecosystem functioning, related with the pool of material and fluxes of material and energy (Hooper et al. 2005). We developed one theoretical model (Fig. 1) based on knowledge about the studied system and the ecological literature (detailed below) to test our hypothesis. The study system is localized in the Brazilian seasonally dry tropical forest called *Caatinga* and inserted in the semiarid region of the country. The Brazilian Caatinga has chronic disturbances (Ribeiro et al. 2015) that is the removal of small and continuous fraction of forest biomass such as forest grazing and selective logging (Singh 1998). We hypothesized that anthropogenic disturbance has direct and indirect effects (mediated by functional structure of plant community and plant cover) on ecosystem properties (Fig. 1). Further, we assessed the magnitude of influence of functional structure and plant cover to explain each ecosystem property and multifunctionality.

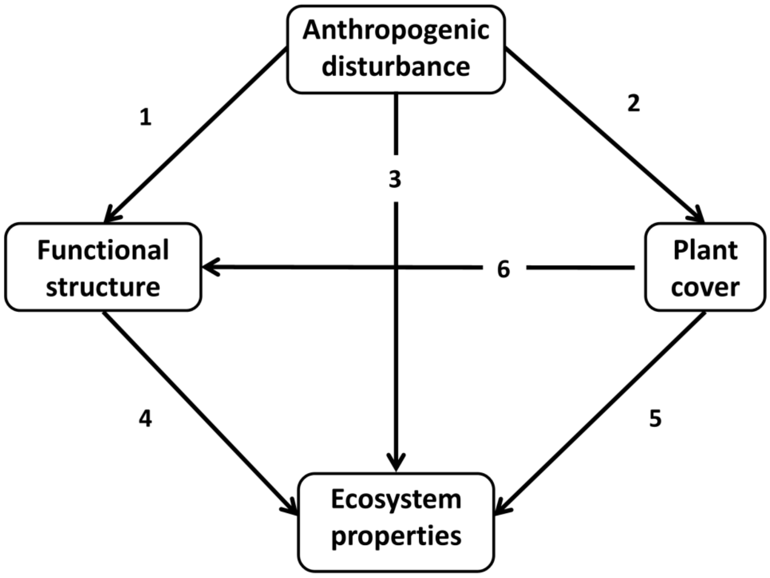

Fig. 1. Theoretical model developed to assess the effects of anthropogenic disturbance on ecosystem properties. Indirect effects mediated by functional structure occur through paths 1 and 4 while indirect effects mediated by plant cover are through the paths 2 and 5. Path 3 represents the effects of disturbance on ecosystem properties operating independent of those mediated indirectly through functional structure and plant cover. Path 6 represents the association among the mediators (functional structure and plant cover).

THE THEORETICAL MODEL

Paths 1, 2 and 3: *Anthropogenic disturbance changes functional structure of plant community, plant cover and ecosystem properties.*

Functional structure through the analysis of functional traits is capable to detect community response to different types of disturbance better than only species richness (Mouillot et al. 2013). Disturbance derived from human resources exploitation alters the traits space in a non-random way excluding preferable species (loser) and improving some winner species (Mouillot et al. 2013). Disturbance caused by land use (mostly agriculture and grazing) is the main cause of deforestation worldwide and drastically alters forest cover (Foley et al. 2005). Land use also affects directly and negatively the ecosystem properties of above-ground net primary productivity, above-ground live and dead biomass and the contents of carbon and nitrogen in the soil (Garnier et al. 2007). Further, disturbance changed local leaf traits and therefore, the ecosystem properties of litter biomass and soil carbon related to those traits (Lienin & Kleyer 2012).

Path 4: Functional structure of plant community influences ecosystem properties.

More than 20 years of biodiversity-ecosystem function (BEF) research revealed the importance of plant biodiversity on ecosystem functioning (Hooper et al. 2005; Cardinale et al. 2011). Two main hypotheses emerged to explain the underlying mechanisms: (i) diversity hypothesis, where diverse plant communities have greater complementary use of resources than species poor communities because different species use resources in distinct ways (Tilman et al. 1997) and (ii) mass-ratio hypothesis, which states that the functional effects of dominant plant species will prevail the functioning of ecosystems (Grime 1998). Diversity and mass-ratio hypotheses are not mutually exclusive (Cardinale et al. 2011). For the multifunctionality variation, both functional diversity (diversity hypothesis) and mean values of traits (mass-ratio hypothesis) were related to explain it (Mouillot et al. 2011). However, mean values of traits (mass-ratio) had more influence than functional diversity to explain the ecosystem properties of plant and litter biomass (Mokany et al. 2008; Laughlin 2011; Roscher et al. 2012), above-ground net primary productivity and soil carbon (Laliberté & Tylianakis 2012; Lienin & Kleyer 2012). The functional structure of our model was estimated using variables of functional diversity and mean value of traits that is more detailed in the methods.

Path 5 and 6: *Plant cover influences ecosystem properties and functional structure of plant community.*

Perennial plant cover has crucial role on drylands functioning (Maestre & Escudero 2009; Maestre et al. 2010). Plant cover had stronger effects on properties related to infiltration and nutrient-cycling when compared to other biotic attributes (richness and evenness) (Maestre et al. 2010). Vegetation loss also modifies hydrological and biogeochemical cycles, increasing soil water evaporation and the erosion of nutrients (Asner et al. 2004). Analysing semi-arid regions worldwide, (Soliveres et al. 2014) and co-authors found that relative woody cover has a hump-shaped relationship with diversity (species eveness). They argued that higher levels of woody cover and density increase the environmental heterogeneity and therefore niche space, favoring local diversity. From the threshold of 41-60% of relative woody cover, diversity decreases due more environmental homogeneity (Soliveres et al. 2014).

Methods

Study area

The study area is located at the Sustainable Development Reserve (SDR) called *Reserva de Desenvolvimento Sustentável Estadual Ponta do Tubarão*. The reserve is a Protected Area (PA) defined in category VI of IUCN (International Union of Conservation Nature). This type of reserve allow local people to live within reserve boundaries and traditional livelihood practices are permitted as long as these practices are managed and considered sustainable (SNUC 2000). Previous questionnaires applied on local livelihoods, showed that three main traditional activities are practiced inside the reserve: i) livestock production (goat, sheep and cattle) raised freely and fed mainly by herbaceous plants during rainy season; ii) subsistence agriculture followed by clear-cutting of small areas; and iii) selective logging for construction (fences, houses and boats) or charcoal production. The SDR is located in Macau and Guamaré municipalities, in the north of Rio Grande do Norte state, northeast of Brazil, and is placed in the Brazilian seasonally dry tropical forest biome (Fig. 2). Inside the reserve, mean rainfall is 508 $mm.year^{-1}$ which is concentrated between January and May and less than 20 mm between October and December (data available at http://www.inmet.gov.br). We conducted the study in the Caatinga vegetation of the reserve with 2,010 hectares (Fig. 2). The Caatinga vegetation of the reserve with low anthropogenic disturbance has a closed canopy cover of ~ 4 meter height, dominated by the woody species *Mimosa tenuiflora*, *Poincianella pyramidalis*, *Pytirocarpa moliniformis* and *Croton sonderianus*.

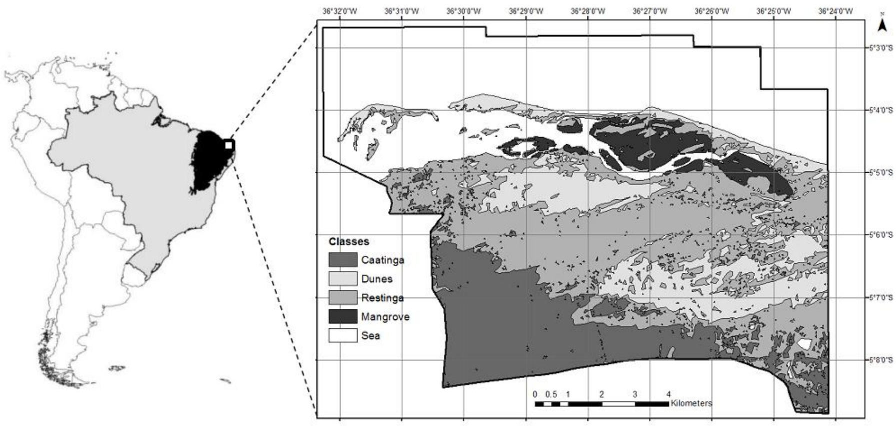

Figure 2. Location of the *Ponta do Tubarão* Sustainable Development Reserve, placed in the northeast of Brazilian seasonally tropical dry forest boundaries (black polygon). The classes of the reserves are: Caatinga, dunes, restinga, mangrove and sea.

Data collection

First, to randomize the plots location in a gradient of plant cover we classified the Caatinga vegetation of the reserve as open, intermediate and closed. We used the

Maximum Likelihood (ML) supervised classification in ArcGis v.10 (ESRI 2011) and Landsat TM5 satellite image from 2008 with resolution of 30x30m (www.inpe.br). Open vegetation has remaining trees and shrubs patches, intermediate vegetation has a more continuous forest with trees height up to 2m and closed vegetation has closed canopy with trees height of about 3-4 m.To apply the ML procedure, we selected signatures for each type of vegetation on satellite image based on field observation and then all pixels of the Caatinga vegetation of reserve were classified according to priori signatures. Then, we randomized 20 locations in each type of vegetation to place circular plots with 25 meters radius (area of 1962.5 m^2) to measure the variables of anthropogenic disturbance. We implemented square plots with 10 x 10 meters (100 m^2) following the four cardinal directions to measure the variables of plant community (functional structure and plant cover) and ecosystem properties. We used the same coordinates of circular plots to place the center of square plots. At the end, we sampled 55 plots during the rainy season of 2012 and 2013 (from March to July).

The variables measured to estimate anthropogenic disturbance were (i) livestock density: based on number of total dung pellets from goats, sheep, cattle and donkeys; (ii) clear-cutting: presence or absence of past deforestation where plot is located using Landsat satellite images from 1984-2010 (see Appendix S1 in Support Information for detailed methodology) and (iii) selective logging: estimated by total basal area of wooden stump found inside the circular plots. We estimated the anthropogenic disturbance index (AD) using an adaptation of the compound index of land-use intensity from (Allan et al. 2014) and is illustrated as followed.

$$AD = \sqrt{\frac{Ld}{mean(Ld)} + \frac{Sl}{mean(Sl)}} + Cc$$

We standardized the variables of livestock density (Ld) and selective logging (Sl) by its mean and took the square root of this sum. We summed the value of two when the plot had clear-cutting (Cc) and zero when it had not.

We estimated the percentage of plant cover by counting the number of presence or absence of vegetation in the ground and/or canopy at 25 grid points (distanced two meters among them). We identified all woody plants above 20 cm height in square plots (10 x 10 m) to estimate the functional structure of local plant community. A total of 40 woody species were identified at the Rio Grande do Norte University herbarium (Appendix S2 in Support Information). We measured five functional plant traits that are related to maintenance of ecosystem processes and provision of important services (de Bello et al. 2010). We collected five leaves from five different individuals of each species to estimate the leaf functional traits: (i) leaf area (LA), calculated from scanned rehydrated leaves using ImageJ software (Rasband 1997); (ii) leaf mass per area (LMA), measured by dividing leaf dry mass (oven dried to constant mass) by its area and (iii) leaf area per perimeter ratio (APR), calculated by dividing the leaf area per its perimeter, which was calculated using ImageJ software (Rasband 1997). We collected five branch samples from five different individuals of each species to estimate (iv) wood density, calculated by dividing branch xylema dry mass (without bark) by its volume a few hours after field collection using beakers of several sizes. We also classified the plant community according to (v) life forms: tree, treelet, shrub, sub-shrub.

We estimated four variables to represent the functional structure of plant community, two variables using the mean traits value (wood density and leaf traits) and two variables of functional diversity (richness and entropy). For the estimation of the mean traits value we used the formula of community weight mean (CWM) for each functional trait (except life forms) that is the total sum of relative abundance of species (basal area) times the value of the functional trait (Garnier et al. 2004). Principal component analysis (PCA) was used to represent the leaf traits (CWM of LA, LMA and APR). Functional richness is defined as the functional trait space that is occupied by the community and was calculated as the convex-hull volume of multidimensional trait space (Villéger et al. 2008). Functional entropy is based on Rao's quadratic entropy (Rao 1982) which is the functional difference between species pairs weighted by their relative abundance (Botta-Dukát 2005). We used all five traits to calculate the indexes of functional diversity that were estimated with multivariate species trait axes from principal coordinate analyses (PCoA) obtained using Gower dissimilarity, Podani's approach to deal with ordered factors and Calliez's method to correct negative eigenvalues of PCoA axes (Podani & Schmera 2006; Pavoine et al. 2009). We used the *FD* package (Laliberté et al. 2014) in R version 3.02 (R Core Development Team 2005) to calculate these functional variables.

We measured five ecosystem properties: (i) standing biomass, (ii) litter biomass, (iii) soil water retention , (iv) soil carbon, (v) soil nutrients (nitrogen, potassium, phosphorus and calcium). We also calculated the index of multifunctionality as proposed by (Maestre et al. 2012) that is the average of Z-scores (standardized values) of all ecosystem properties per plot. To estimate standing biomass, we calculated the stem volume (m^3) for each plant located inside plots using the cylindrical formula (basal area times height) multiplied by the factor form of 0.9 used for the Caatinga species (Gariglio et al. 2010). Then, we calculated standing biomass multiplying the stem volume times relative species' wood density ($g.cm^{-3}$ converted to $kg.m^{-3}$). Therefore, we assessed the total standing biomass (kilograms) per plot summing the biomass calculated for each plant inside the plots. We estimated litter biomass collecting the litter in four samples per plot using 0.25 x 0.25 cm subplots and then dried until constant weight. We estimated soil water retention by the percentage of moisture in the soil three days after the last local rain using the equipment *Aquaterr digital soil moisture and temperature* (model M, T & EC - 300 meters). For this ecosystem property, we measured only 33 plots and calculated the average soil moisture collecting 20 measures per plot. For soil carbon and soil nutrients, we collected four soil samples at 10 cm depth per plot and then homogenized and dried in shaded ambient conditions. Soil analysis were done at the soil laboratory of the *Empresa de Pesquisa Agropecuária do Rio Grande do Norte* (EMPARN) using methodology from (EMBRAPA 1997) to estimate the contents of carbon (C), nitrogen (N), phosphorus (P), potassium (K) and calcium (Ca). Principal component analysis (PCA) was applied to N, P, K, Ca to reduce the variables of soil nutrients into two principal components axes (PC1 and PC2).

Statistical analyses

We used structural equation modeling (SEM) to test our theoretical model developed to explain the variation of each ecosystem property and multifunctionality in the Caatinga of reserve (Fig. 1). In SEM, theoretical model is constructed based on a-priori available researcher knowledge and is rejected only if the observed data did not match the model (Grace 2006). SEM is an important statistic tool that has been used on response-effect traits framework (Minden & Kleyer 2011; Laliberté & Tylianakis 2012; Lavorel & Grigulis 2012; Lienin & Kleyer 2012) and it is a promising way to test direct and indirect effects on natural systems in a realistic gradient of perturbation (Tomimatsu et al. 2013).

We selected the final models for each ecosystem property and multifunctionality removing non-significant paths from theoretical model and they were only accepted whether the indexes of goodness of fit was improved (Lavorel & Grigulis 2012).

Although, some non-significant paths were kept in the final models when it improved the variance explained of ecosystem property and multifunctionality. The variance explained of the response variable indicates how the addition or exclusion of some paths improve the explanation of the variable of interest (Grace 2006). Goodness of fit of these models were assessed using: (i) chi-squares test to evaluate the degree to which the data deviates from the model (*P* value > 0.05); (ii) root mean square error of approximation (RMSEA > 0.05) and (iii) comparative fit index that measures the improvement of the model fit over a baseline model (CFI > 0.95) (Grace 2006; Kline 2011).

We performed analysis in R version 3.02 (R Core Development Team 2005) using the packages *lavaan* and *semTools*. Standardized values (z transformation) were used to output path coefficients in standard variation units. Variables of livestock density, standing biomass, functional richness and functional entropy were log transformed to maintain linear relationship in SEM. We used the path coefficients rules to calculate the total effect of anthropogenic disturbance on each ecosystem property, using the sum of path coefficients from direct and indirect effects (Grace 2006). Indirect effects is calculated by the multiplication of standardized path coefficients of indirect pathways (Grace 2006).

Results

The standardized coefficients (β) estimated and *P* values of all relationships from the theoretical full model and final models (paths 1, 2, 3, 4, 5, 6) of each ecosystem property and multifunctionality are in Appendix S3 in Support Information. All final models had better fit than the relative theoretical full model and were accepted to explain the ecosystem properties and multifunctionality (Table 1).

Table 1. Goodness of fitness indexes (p, RMSEA and CFI) and variation explained (R^2) of the hypothetical and final models for each ecosystem property and multifunctionality.

Ecosystem property	Model	χ^2	df	p	RMSEA	CFI	AIC	R^2
Standing biomass	Theoretical	14.61	4	0.006	0.11	0.93	62.61	0.74
	Final	12.93	7	0.074	0.00	0.95	40.93	0.72
Litter biomass	Theoretical	14.61	4	0.006	0.11	0.90	62.61	0.46
	Final	1.71	4	0.789	0.00	1.00	23.71	0.46
Soil nutrients (N,P,K,Ca)	Theoretical	14.21	4	0.007	0.10	0.88	62.21	0.17
	Final	1.85	3	0.605	0.00	1.00	25.85	0.14
Soil water retention	Theoretical	8.72	4	0.069	0.00	0.92	56.72	0.59
	Final	2.91	5	0.714	0.00	1.00	22.91	0.58
Soil carbon	Theoretical	14.21	4	0.007	0.10	0.89	62.21	0.27
	Final	2.47	5	0.781	0.00	1.00	22.47	0.25
Multifunctionality	Theoretical	14.21	4	0.007	0.10	0.89	62.21	0.29
	Final	2.94	4	0.568	0.00	1.00	24.94	0.27

Anthropogenic disturbance negatively affects functional diversity (functional richness and entropy) mediated by the loss of plant cover (Fig. 3). The total negative effect from disturbance on functional diversity variables (multiplication of indirect standardized paths coefficients) are β= -0.20 for functional richness and β= -0.16 for functional entropy. Otherwise, the mean traits value (leaf traits and wood density) was not influenced by anthropogenic disturbance, neither by direct or indirect effects. Then, the effect of disturbance on functional structure occurred through the indirect path mediated by plant cover (paths 2 and 6 in Fig. 1) not by the direct effect from anthropogenic disturbance (path 1 in Fig.1).

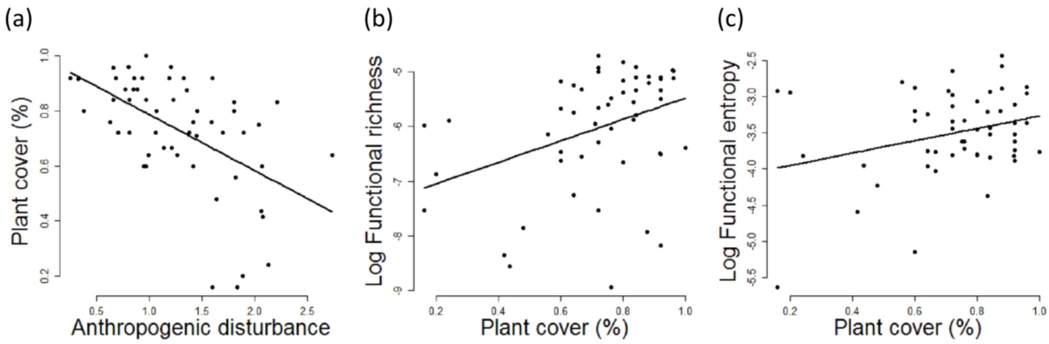

Figure 3. Relationships among (a) anthropogenic disturbance and plant cover, (b) plant cover and functional richness and (c) plant cover and functional entropy. These relationships occurred in the final models of all ecosystem properties and multifunctionality.

In the same way, the effect of anthropogenic disturbance on all ecosystem properties and multifunctionality occurred mainly through the indirect path mediated by plant cover (Fig. 4; paths 2 and 5 in Fig. 1). Indirect effect of anthropogenic disturbance through this indirect path was negative for all ecosystem properties and multifunctionality. Total effect of anthropogenic disturbance was β= -0.41 for standing biomass, β= -0.30 for litter biomass, β= -0.20 for soil nutrients (PC1), β= -0.47 for soil water retention, β= -0.29 for soil carbon and β= -0.29 for multifunctionality. The influence of disturbance through functional structure (plant cover affecting functional richness) occurred only for standing biomass but was low (β= -0.03, Fig. 3a).

Direct effects of anthropogenic disturbance occurred only on the ecosystem properties of litter biomass and soil nutrients. Summing the indirect negatives effects (mediated by plant cover) with the direct and negative effects of disturbance (β= -0.22; Fig. 4b), the total effect of anthropogenic disturbance on litter biomass was β= -0.52. For soil nutrients (PC1), the total effect of anthropogenic disturbance by the sum of direct (β= 0.42; Fig. 4b) and indirect effects (β= -0.20) remained positive (β= 0.22). The first axis of principal component of soil nutrients is represented by calcium (54.74%) , nitrogen (54.5%), phosphorus (54.5%) and potassium (22.16%).

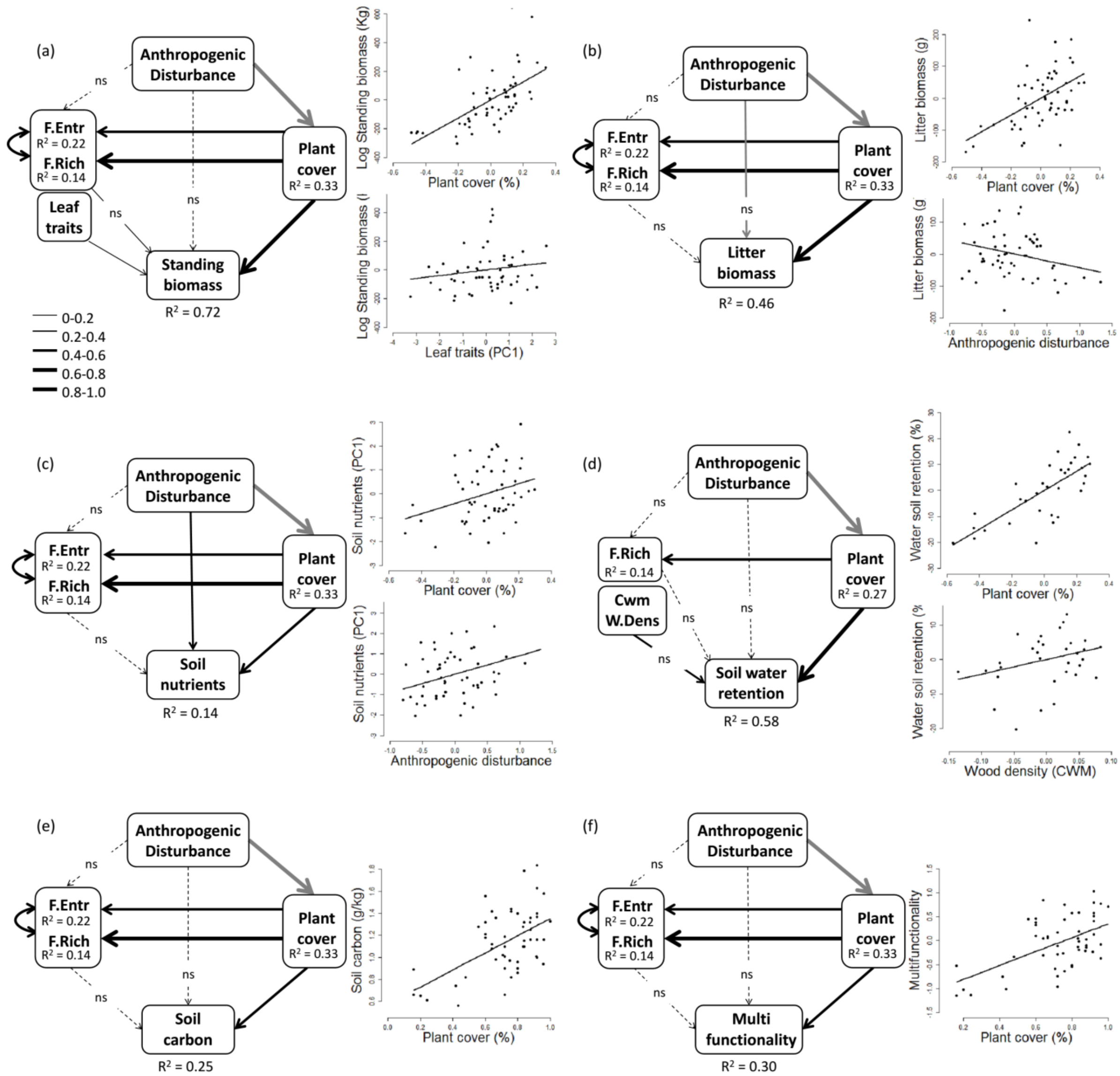

Figure 4. Final models derived from the threoretical model for each ecosystem property and multifunctionality. Grey and black lines are negative and positive associations, respectively. The thickness of lines represents the strengh of relation, dotted lines are non-significant (ns) paths that were removed from the hypothetical model and double arrows represent correlation. Partial and single regressions of the explanatory variables (plant cover, leaf traits, wood density and anthropogenic disturbance) with the ecosystem properties and multifunctionality are on the right side of each SEM model.

(a) Standing biomass, (b) litter biomass, (c) soil nutrients (PC1), (d) soil water retention, (e) soil carbon and (f) multifunctionality.

Besides the stronger effect from plant cover on all ecosystem services and multifunctionality, weaker influence of functional structure on standing biomass (Fig, 4a) and soil water retention (Fig. 4d) also occurred. Leaf traits (PC1) was positively associated with standing biomass (β= 0.144, P= 0.05) and this first axis of principal component of leaf traits is represented by area per perimeter ratio (64.7%), leaf area (62.1%) and leaf mass per area (44.2%). Functional richness had non-significant influence on standing biomass (β= 0.134, P= 0.10) but was not removed from the final model due its relative contribution on the variance explained of this ecosystem property (2%). Wood density (mean trait value) had positive but non-significant influence on soil water retention (β= 0.205, P= 0.07) but was not removed from the final model due it improved in 6% the variance explained of this ecosystem property.

Discussion

We developed one theoretical model to understand how anthropogenic disturbance is affecting ecosystem properties through direct effects or indirectly mediated by functional structure and plant cover. The main path to explain the disturbance effects on ecosystem properties and multifunctionality is through the loss of plant cover (paths 2 and 5 in Fig 1). Even functional diversity (entropy and richness) is negatively affected by anthropogenic disturbance through the loss of plant cover (paths 2 and 6 in Fig 1). However, mean traits value (leaf traits and wood density) only has weak association with standing biomass and soil water retention (path 4 in Fig 1). We evidence that in the Brazilian Caatinga, plant cover is the main factor associated to the maintenance of soil resources (nutrients and water) and aboveground biomass (live and dead). Hence, the loss of plant cover is the main negative effect caused by anthropogenic disturbance decreasing local ecosystem properties and functional diversity.

Studies in drylands comparing the magnitude of other factors effects on ecosystem properties and multifunctionality support our findings showing the importance of plant cover in these systems (Maestre et al. 2010; Soliveres et al. 2014). Analysing global drylands, Soliveres and colleagues (2014) found that total plant cover and relative woody cover had stronger influence on multifucntionality (14 variables used as proxy for key ecosystem processes) than diversity measured as species richness and evenness. Still in global drylands, abiotic factors (sand content and temperature) had same influence as species richness on ecosystem multifunctionality (Maestre et al. 2012). In a regional scale (Patagonian rangelands), grass and shrub cover is directly associated to above-ground net primary productivity but also in a indirect way through the mediation of species richness (Gaitán et al. 2014). However, in this study, relative effects were stronger from plant cover than species richness (Gaitán et al. 2014). Perennial plant cover explains more the soil properties related to infiltration and nutrient-cycling than other biotic attributes such as richness and evenness (Maestre et al. 2010). In the same way, our study in the Brazilian Caatinga highlights the importance of perennial plant cover to maintain the ecosystem functioning in this semiarid region, such as biomass production and soil resources maintenance.

The importance of plant perennial cover is overwhelming to maintain essential processes in semiarid ecosystems worldwide (Martinez-Mena et al. 2002; Bastida et al. 2008; Maestre et al. 2010). The cover offered by vegetation creates a positive feedback between plant and soil resources that usually occur in semi-arid systems (HilleRisLambers et al. 2001; D’Odorico et al. 2012). Plant cover intercepts the sunlight and raindrops and thus, avoids soil evaporation by lowering the topsoil temperature and superficial water runoff, respectively (Facelli & Pickett 1991; van de Koppel et al. 1997; HilleRisLambers et al. 2001). Vegetation also protects soil from water and wind erosion which may cause soil nutrients losses (Ludwig et al. 2005). Beyond the changing of biophysical factors, intermediate percentage of vegetation cover creates high environmental heterogeneity that increases niche availability and more species could occur in the same space (Soliveres et al. 2014).

Disturbance caused by human alteration of landscape is one of the factors besides climatic variation related to increase the desertification process in arid and semiarid regions (D'Odorico et al. 2012). Desertification is affecting around 15% of Brazilian seasonally dry tropical forest biome (Leal et al. 2005) and our study is the first empirical evidence of how anthropogenic disturbance is negatively impacting functional structure of plant community and multifunctionality through the plant cover loss. Plant cover can be a suitable indicator of desertification such it is the main factor associated to single ecosystem properties and multifunctionality in the Brazilian Caatinga. As found by (Maestre & Escudero 2009), perennial plant cover also had more explanation than the exponent of the truncated power law as suggested by (Kéfi et al. 2007) to monitor desertification.

Currently, the deforestation in the Brazilian seasonally dry tropical forest biome reached about 47% of its total area (MMA 2009). Besides, around 27 million people live in this region and they are highly dependent from natural resources harvesting (mainly for woody energy and agricultural purposes) and livestock raising (Hauff 2010). However, these traditional economic activities are chronic disturbances that change plant communities functional structure and cover and may lead to desertification in a long period of time.

Current global environmental challenge is to set up how manage inherent land use trade-offs which offer supply of human needs and at the same time could maintain ecosystems capacity to provide services in the future (Foley et al. 2005). We recommend specifically for management of the Brazilian Caatinga vegetation that livestock should be raised inside farms with fences to avoid domestic animals feeding inside forested areas. Further, abandoned clear-cut fields should be restored aiming to cover bare soil and to faster natural regeneration. Perennial plant cover is the main factor to maintain the local ecosystem properties and intrinsic services for human well being. More sustainable management of the Brazilian Caatinga lands is the way to avoid desertification expansion in Brazilian seasonally dry tropical forest biome.

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Appendix S1. Methodology of the clear-cutting estimation.

Table S1. Woody species list.

Table S2. Standardized coefficients estimated and *P* values.

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Supporting Information

Appendix S1. Methodology of the clear-cutting estimation

A fraction image of bare soil reflectance from each year was created using Spectral unmixing procedure in ENVI software v.5. Spectral unmixing is a method that decomposes the spectrum of mixed pixels into a collection of constituent spectra called *endmembers* and their correspondent abundances or *fraction*, indicating the proportion of each endmember present in each pixel of target landscape (Keshava & Mustard 2002). For each plot, the development of the fraction of bare soil (between 0 and 1) was analyzed over time. Whenever there was a sudden increase in the fraction of bare ground from one year to another, the plot was considered to have been burned. From the 55 plots, 21 were classified as clear-cut in the past at least once in previous 26 years. This satellite image classification was then verified in the subsequent field visits for vegetation assessments when we searched for evidence of past forest burning, e.g. charcoal or burned logs on the ground, and by asking local people for information.

Table S1. Woody species list that occur in *caatinga* area of *Ponta do Tubarão* Sustainable Development Reserve (SDR).

Family	Species
Apocynaceae	*Aspidosperma pyrifolium*
Boraginaceae	*Varronia globosa*
Burseraceae	*Commiphora leptophloeos*
Capparaceae	*Cynophalla flexuosa*
Combretaceae	*Combretum leprosum*
Erythroxylaceae	*Erythroxylum sp1*
Erythroxylaceae	*Erythroxylum sp2*
Euphorbiaceae	*Croton adamantinus*
Euphorbiaceae	*Croton blanchetianus*
Euphorbiaceae	*Croton heliotropiifolius*
Euphorbiaceae	*Croton nepetifolius*
Euphorbiaceae	*Croton pedicellatus*
Euphorbiaceae	*Jatropha mollissima*
Euphorbiaceae	*Jatropha mutabilis*
Euphorbiaceae	*Jatropha ribifolia*
Euphorbiaceae	*Manihot sp*
Euphorbiaceae	*Sapium sp*
Fabaceae	*Bauhinia cheilantha*
Fabaceae	*Bauhinia dubia*
Fabaceae	*Calliandra depauperata*
Fabaceae	*Calliandra spinosa*
Fabaceae	*Chamaecrista sp*
Fabaceae	*Mimosa sp*
Fabaceae	*Mimosa tenuiflora*
Fabaceae	*Piptadenia stipulacea*
Fabaceae	*Poincianella pyramidalis*
Fabaceae	*Pityrocarpa moniliformis*
Fabaceae	*Senna macranthera*
Fabaceae	*Senna splendida*
Fabaceae	*Senna trachypus*
Malvaceae	*Herissantia sp*
Malvaceae	*Pavonia varians*
Malvaceae	*Sida galheirensis*
Malvaceae	*Waltheria brachypetala*
Nyctaginaceae	*Guapira sp*
Olacaceae	*Ximenia americana*
Turneraceae	*Turnera diffusa*
Rubiaceae	*Cordiera sp*
Verbenaceae	Undefined species

Table S2. Standardized coefficients estimated and *P* values of all relationships from the hypothesis and final models (paths 1,2,3,4,5,6) of (a) standing biomass, (b) litter biomass, (c) soil nutrients (N, P, K, Ca), (d) soil water retention, (e) soil carbon and (f) multifunctionality.

(a) Standing biomass

Paths	Explanatory variable	Response variable	Theoretical model		Final model	
			Estimate	*P* value	Estimate	*P* value
1	Disturbance	Functional richness	-0.101	0.499	-	-
1	Disturbance	Functional entropy	0.046	0.765	-	-
1	Disturbance	Leaf traits (PC1)	0.032	0.844	-	-
1	Disturbance	Wood density (CWM)	0.118	0.446	-	-
2	Disturbance	Plant cover	-0.537	<0.001	-0.537	<0.001
3	Disturbance	Standing biomass	0.073	0.400	-	-
4	Functional richness	Standing biomass	0.174	0.039	0.134	0.098
4	Functional entropy	Standing biomass	-0.049	0.545	-	-
4	Leaf traits (PC1)	Standing biomass	0.195	0.018	0.144	0.054
4	Wood density (CWM)	Standing biomass	0.131	0.128	-	-
5	Plant cover	Standing biomass	0.840	<0.001	0.765	<0.001
6	Plant cover	Functional richness	0.323	0.030	0.377	0.003
6	Plant cover	Functional entropy	0.319	0.039	0.294	0.024
6	Plant cover	Leaf traits (PC1)	-0.004	0.979	-	-
6	Plant cover	Wood density (CWM)	-0.200	0.197	-	-

(b) Litter biomass

Paths	Explanatory variable	Response variable	Theoretical model		Final model	
			Estimate	*P* value	Estimate	*P* value
1	Disturbance	Functional richness	-0.101	0.499	-	-
1	Disturbance	Functional entropy	0.046	0.765	-	-
1	Disturbance	Leaf traits (PC1)	0.032	0.844	-	-
1	Disturbance	Wood density (CWM)	0.118	0.446	-	-
2	Disturbance	Plant cover	-0.537	<0.001	-0.578	<0.001
3	Disturbance	Litter biomass	-0.233	0.048	-0.216	0.077
4	Functional richness	Litter biomass	0.072	0.528	-	-
4	Functional entropy	Litter biomass	0.084	0.443	-	-
4	Leaf traits (PC1)	Litter biomass	0.166	0.136	-	-
4	Wood density (CWM)	Litter biomass	0.051	0.658	-	-
5	Plant cover	Litter biomass	0.484	<0.001	0.526	<0.001
6	Plant cover	Functional richness	0.323	0.030	0.471	<0.001
6	Plant cover	Functional entropy	0.319	0.039	0.374	0.003

6	Plant cover	Leaf traits (PC1)	-0.004	0.979	-	-
6	Plant cover	Wood density (CWM)	-0.200	0.197	-	-

(c) Soil nutrients (N, P, K, Ca)

Paths	Explanatory variable	Response variable	Theoretical model		Final model	
			Estimate	*P* value	Estimate	*P* value
1	Disturbance	Functional richness	-0.104	0.493	-	-
1	Disturbance	Functional entropy	0.054	0.729	-	-
1	Disturbance	Leaf traits (PC1)	0.012	0.943	-	-
1	Disturbance	Wood density (CWM)	0.125	0.425	-	-
2	Disturbance	Plant cover	-0.543	<0.001	-0.543	<0.001
3	Disturbance	Soil nutrients	0.397	0.009	0.416	0.006
4	Functional richness	Soil nutrients	-0.159	0.273	-	-
4	Functional entropy	Soil nutrients	0.107	0.448	-	-
4	Leaf traits (PC1)	Soil nutrients	0.084	0.554	-	-
4	Wood density (CWM)	Soil nutrients	-0.037	0.801	-	-
5	Plant cover	Soil nutrients	0.374	0.020	0.362	0.017
6	Plant cover	Functional richness	0.316	0.036	0.373	0.003
6	Plant cover	Functional entropy	0.330	0.034	0.300	0.022
6	Plant cover	Leaf traits (PC1)	-0.039	0.811	-	-
6	Plant cover	Wood density (CWM)	-0.187	0.235	-	-

(d) Soil water retention

Paths	Explanatory variable	Response variable	Theoretical model		Final model	
			Estimate	*P* value	Estimate	*P* value
1	Disturbance	Functional richness	0.026	0.898	-	-
1	Disturbance	Functional entropy	0.018	0.931	-	-
1	Disturbance	Leaf traits (PC1)	-0.07	0.749	-	-
1	Disturbance	Wood density (CWM)	0.201	0.317	-	-
2	Disturbance	Plant cover	-0.061	<0.001	-0.607	<0.001
3	Disturbance	Soil water retention	-0.146	0.305	-	-
4	Functional richness	Soil water retention	-0.109	0.417	-	-
4	Functional entropy	Soil water retention	0.088	0.496	-	-
4	Leaf traits (PC1)	Soil water retention	-0.059	0.628	-	-
4	Wood density (CWM)	Soil water retention	0.211	0.112	0.205	0.074
5	Plant cover	Soil water retention	0.732	<0.001	0.781	<0.001
6	Plant cover	Functional richness	0.412	0.040	0.397	0.013
6	Plant cover	Functional entropy	0.318	0.127	-	-
6	Plant cover	Leaf traits (PC1)	0.031	0.887	-	-
6	Plant cover	Wood density (CWM)	-0.246	0.220	-	-

(e) Soil carbon

Paths	Explanatory variable	Response variable	Theoretical model		Final model	
			Estimate	*P* value	Estimate	*P* value
1	Disturbance	Functional richness	-0.104	0.493	-	-
1	Disturbance	Functional entropy	0.054	0.729	-	-
1	Disturbance	Leaf traits (PC1)	0.012	0.943	-	-
1	Disturbance	Wood density (CWM)	0.125	0.425	-	-
2	Disturbance	Plant cover	-0.543	<0.001	-0.583	<0.001
3	Disturbance	Soil carbon	0.056	0.681	-	-
4	Functional richness	Soil carbon	0.089	0.497	-	-
4	Functional entropy	Soil carbon	0.110	0.384	-	-
4	Leaf traits (PC1)	Soil carbon	0.070	0.586	-	-
4	Wood density (CWM)	Soil carbon	0.012	0.931	-	-
5	Plant cover	Soil carbon	0.518	0.020	0.504	<0.001
6	Plant cover	Functional richness	0.316	<0.001	0.468	<0.001
6	Plant cover	Functional entropy	0.330	0.034	0.380	0.003
6	Plant cover	Leaf traits (PC1)	-0.039	0.811	-	-
6	Plant cover	Wood density (CWM)	-0.187	0.235	-	-

(f) Multifunctionality

Paths	Explanatory variable	Response variable	Theoretical model		Final model	
			Estimate	*P* value	Estimate	*P* value
1	Disturbance	Functional richness	-0.104	0.493	-	-
1	Disturbance	Functional entropy	0.054	0.729	-	-
1	Disturbance	Leaf traits (PC1)	0.012	0.943	-	-
1	Disturbance	Wood density (CWM)	0.125	0.425	-	-
2	Disturbance	Plant cover	-0.543	<0.001	-0.543	<0.001
3	Disturbance	Multifunctionality	0.120	0.377	-	-
4	Functional richness	Multifunctionality	-0.061	0.641	-	-
4	Functional entropy	Multifunctionality	0.096	0.447	-	-
4	Leaf traits (PC1)	Multifunctionality	0.146	0.256	-	-
4	Wood density (CWM)	Multifunctionality	0.076	0.569	-	-
5	Plant cover	Multifunctionality	0.518	0.020	0.543	<0.001
6	Plant cover	Functional richness	0.627	<0.001	0.373	0.003
6	Plant cover	Functional entropy	0.330	0.034	0.300	0.022
6	Plant cover	Leaf traits (PC1)	-0.039	0.811	-	-
6	Plant cover	Wood density (CWM)	-0.187	0.235	-	-

CAPÍTULO II

SPATIAL ASSOCIATIONS OF ECOSYSTEM SERVICES AND BIODIVERSITY AS A BASELINE FOR SYSTEMATIC CONSERVATION PLANNING

Spatial associations of ecosystem services and biodiversity as a baseline for systematic conservation planning

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ABSTRACT

Aim

Conservation units are frequently defined on the bases of plant and animal species occurrence. Although ecosystem services are expected to be protected when biodiversity is preserved, positive spatial associations between these two factors are still to be demonstrated at large spatial scales. We evaluated spatial associations among ecosystem services and plant biodiversity and how these variables are represented across a network of protected areas.

Location

Brazilian seasonally tropical dry forest (Caatinga).

Methods

We produced plant biodiversity maps (species richness, narrow-range species richness and beta-diversity) using species distribution modeling. We elaborated maps of ecosystem services using primary data and proxy-based approach for regulating services (water purification, carbon storage and erosion control), provisioning services (water supply, fodder, agriculture) and supporting services (water balance, net primary productivity and soil fertility). We performed spatial correlation analyses between biodiversity and ecosystem services using Pearson's correlation test. We calculated the percentage of hotspot areas of biodiversity and ecosystem services that occurred in two types of protected areas (Strict Protection and Sustainable Use) and compared it to what was expected by a null model.

Results

Positive correlations (synergies) arose among biodiversity and ecosystem services (beta-diversity with water balance, species richness with both water purification and carbon storage). Negative correlations (trade-offs) occurred among water balance with both species richness and narrow-range species richness. Strict Protection areas were well represented in terms of carbon storage and underrepresented for fodder and agriculture. Sustainable Use protected areas were important for water balance. Biodiversity variables were poorly represented in both types of protected areas.

Main conclusions

Only two ecosystem services were represented inside the protected areas network, the . biodiversity variables positively correlated with these services were not represented in conservation. Complementarity approach based on spatial correlation among targets might not be efficient to protect non-selected targets.

Keywords

Caatinga, spatial correlation, regulating, provisioning and supporting services, protected areas network, InVEST, species distribution modeling

INTRODUCTION

Systematic conservation planning is a fundamental procedure for protected areas implementation and it often uses as baseline the presence of biodiversity hotspots and/or charismatic, rare and endangered species (Margules & Pressey, 2000; Dudley, 2008). On the other hand, ecosystem services, such as clean water or erosion control, have been rarely used in conservation planning, apart from justifying biodiversity conservation needs (Balvanera *et al.*, 2001; Egoh *et al.*, 2007). Nonetheless, it is still unclear the extent to which biodiversity could function as a surrogate for ecosystem services when defining protected areas. Correlation between biodiversity and ecosystem services at large spatial scales have shown divergent results, with more negative (trade-offs) than positive correlations (synergies), depending on the scale and ecosystem services selected (Chan *et al.*, 2006; Turner *et al.*, 2007; Anderson *et al.*, 2009; Egoh *et al.*, 2009; O´Farrell *et al.*, 2010; Bai *et al.*, 2011). If these variables are not positively correlated ecosystem services might not be effectively preserved inside protected areas defined on the bases of biodiversity.

A representativeness analysis approach is frequently used to evaluate if established protected areas have been effective to reach biodiversity and ecosystem services standards, however, this factors are usually addressed separately. Biodiversity representativeness inside protected areas network is mainly assessed through gap analysis, which measures the percentage of the species distribution area that is not included inside the protected area (Rodrigues & Brooks, 2007). While ecosystem services representativeness have been analyzed by measuring the ratio between the percentage area where ecosystem services were found divided by the percentage land area covered by the same protected areas (Eigenbrod *et al.*, 2009; Eigenbrod *et al.*, 2010a; Durán *et al.*, 2013). Coupling those representativeness analysis with spatial correlation assessments of biodiversity and ecosystem services could function as a unique approach to understand whether biodiversity could be used as a surrogate for ecosystem services protection when defining protection areas.

Biodiversity can be estimated through a large variety of measures that might not always respond in the same manner (Mace *et al.*, 2012). Biodiversity has multiple dimensions such as taxonomic, phylogenetic, genetic, functional, spatial or temporal, interaction and landscape diversity (Naeem & Wright, 2003). For example, spatial mismatching among bird biodiversity components (taxonomic, phylogenetic and functional diversity and their respective turnover) showed the difficulties of finding single biodiversity measures (surrogates) that could represent all biodiversity at large spatial scales (Devictor *et al.*, 2010).

Ecosystem services might also be spatially correlated with each other (Bennett *et al.*, 2009) and multiple positive associations could support multiple services provision at the same conservation area. Provisioning services are the products obtained from ecosystems, such as fodder and wood production, and regulating services are the benefits provided by the regulation of ecosystem processes, such as carbon storage and water retention (MA, 2005). Negative spatial correlations usually occur among regulating and provisioning services (Raudsepp-Hearne *et al.*, 2010; Qiu & Turner, 2013). However, positive association might be expected from supporting services, that are those necessary to produce all other ecosystem services (MA, 2005). One example of supporting service is the net primary productivity, which could function as a potential surrogate for several provision ecosystem services (Egoh *et al.*, 2008). Therefore, it would be very useful to find a particular ecosystem service that could represent all ecosystem services when defining protected areas.

In this study we aimed to 1) Assess correlations among biodiversity and ecosystem services variables at large spatial scale and discuss the implications of our findings to evaluate a protected areas network; 2) Find possible surrogate measurements for biodiversity and ecosystem services separately, that might be applied in systematic conservation planning. We expect that protected areas with higher biodiversity would have more ecosystem services available. We also expect that surrogate measurements of biodiversity and ecosystem services can be found and used in the future as a baseline for establishing a protected areas networks.

METHODS

Study area

The northeast Brazil holds a seasonally dry tropical forest called Caatinga (Fig. 1). Seasonally dry tropical forest includes tall forest in moister sites to scrub rich succulent in driest sites, has rainfall less than 1800 mm. year^{-1}, with a period of 5-6 months receiving less than 100mm (Pennington *et al.*, 2009). The Caatinga vegetation is mostly characterized by deciduous plants that shed their leaves during the dry season and has often dense and continuous formation of tree and shrubs cover during the rainy season with herbaceous plants layer (Bellefontaine *et al.*, 2000). However, enclaves of seasonal forests, ombrophilus forests, savannas and ecotones also occur in the Brazilian seasonally dry tropical forest (Fig. 1; MMA, 2006).

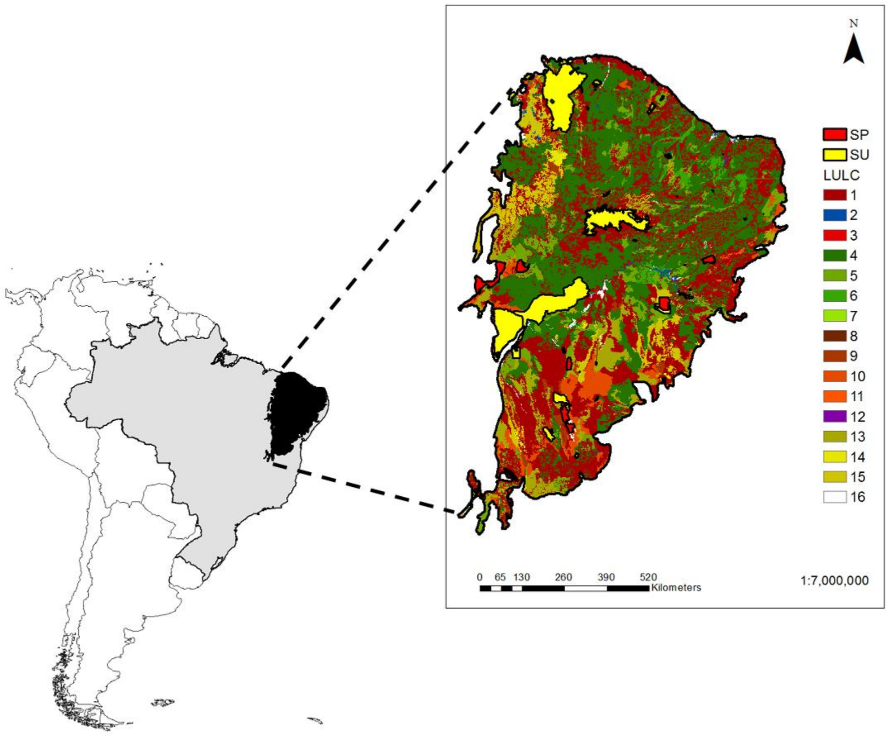

Figure 1. Location of Brazilian seasonally dry tropical forest (Caatinga, black color). In the right side, Caatinga land cover and land use map (LULC) with respective categories of non-vegetation areas (redish colors): (1) farming, (2) water and (3) urban areas; caatinga vegetation areas (greenish colors): (4) forested caatinga, (5) wooded caatinga, (6) park caatinga, (7) woody-grassy caatinga; enclave (brownish colors): (8) ombrophilus forest, (9) savannah, (10) seasonal forest; (11) secondary forest (orange color), (12) dunes (purple color); ecotone (yellowish colors): (13) caatinga/seasonal forest, (14) savannah/seasonal forest, (15) savannah/caatinga and (16) non-identified. Reserve network in Caatinga are the strict protection protected areas (SP, red color) and sustainable use protected areas (SU, yellow color).

The Caatinga has an area of 826,411 km^2 (11% of the Brazilian territory) and is mostly located in the semi-arid region (969,589 km^2). Semi-arid areas is characterized by a mean annual rainfall between 300-400 mm (dry season) and 700-800mm (rainy season) and the precipitation and evapotranspiration rate (P/PET) ranging from 0.2-0.5 (Verheye, 2006). Currently, main threats of the Caatinga are the expansion of deforestation, which has reached about 47% of its total area (MMA, 2009), and the desertification process that already extends 15% of its total area (Leal *et al.*, 2005). Conservation goals will vary with the purpose of each protected area and two broad main management strategies exist in Brazil: targeting protected areas to strict protection (which are equivalent to IUCN protected areas in categories I-IV) and targeting them to sustainable use of resources (equivalent to IUCN V and VI categories). Inside protected areas under strict protection, direct use of natural resources are forbidden, whereas in areas aiming sustainable use, traditional practices are permitted as long as these practices are planned and considered sustainable (SNUC, 2000).

Species distribution modeling

We estimated biodiversity in the Caatinga using woody species distribution modeling (SDM) with Maximum Entropy (MaxEnt) algorithm to estimate species geographical distribution, which allows to predict species suitability of occurrence in areas where information is missing using only presence records (Platts *et al.*, 2010). MaxEnt uses presence records to estimate the suitability of species occurrence based on correlations of known occurrences with environmental variables of the background landscape (Elith *et al.*, 2011). To build the SDMs, we used presence-only records for 769 Caatinga woody species from the TreeAtlan database (Oliveira-Filho, 2010). We used environmental variables from Worldclim (http://www.worldclim.org) and included a map of soil types (http://geoftp.ibge.gov.br) and height above nearest drainage (HAND) (http://www.dpi.inpe.br) as additional environmental variables to calibrate the models. Suitability of occurrence of each species were aggregated by average to 0.05

degrees. For further details on SDMs performances, see Appendix S1 in Supporting Information.

With SDMs predictions, we calculated three proxies of woody biodiversity: species richness, beta-diversity and narrow-range species richness (see Appendix S1). We estimated species richness by summing the number of species present in each pixel (0.05 degree) using the 10 percentile threshold (we considered that the species was present above this threshold). Beta-diversity was calculated by the average of species turnover between the target pixel and the eight neighboring pixels, as proposed by (Lennon *et al.*, 2001). This turnover index focuses more precisely on compositional differences, with a lower influence of local species richness on species dissimilarity (Lennon *et al.*, 2001). Based on principle of irreplaceability, that uniqueness of some species could not be protected elsewhere (Thomas *et al.*, 2013), we calculated the number of species with restricted geographic ranges (hereafter narrow-range species richness) for each pixel. To calculate narrow-range species richness we ranked species by the size of their modeled geographic distribution area. Then, we summed maps of 10% of the species with the smallest areas.

Assessment of ecosystem services

We used two types of data to map ecosystem services: primary data on ecosystem services within the study region and proxy-based data, which links land cover to ecosystem service provision (Eigenbrod *et al.*, 2010b). We mapped nine ecosystem services: three provisioning services (agriculture, fodder and water supply), three regulating services (carbon storage, water purification and erosion control) and three supporting services (net primary productivity, soil fertility and water balance) (Table 1).

Table 1. Description of ecosystem services, units of measurement (pixel of 0.5^0) and the methods and sources used to estimate the service.

Ecosystem services	Description	Unit	Methods and sources
Provisioning services			
Agriculture	Relative area covered by agricultural farms from Brazilian land use map (2010). Levels: <10%, 10-25%, 25-50%	% cover	Primary data (http://mapas.mma.gov.br)
Fodder	Native fodder production in the Caatinga vegetation estimated by weigth gain of livestock (sheeps, goats and cattle) in each vegetation type.	$kg.ha^{-1}.year^{-1}$	LULC proxy-based
Water supply	Underground water wells established for human water use that is registered on Brazilian underground water information system.	number of wells registered	Primary data (http://siagasweb.cprm.gov.br)
Regulating services			
Carbon storage	Carbon density contained in above and below ground of live woody vegetation summed to the soil organic carbon density.	$Mg.ha^{-1}$	Primary data (IPCC, 2006; Cardinale *et al.*, 2011; Hiederer & Köchy, 2011; Baccini *et al.*, 2012).
Water purification	Capacity of each LULC category to retain nutrients (N and P) avoiding their runoff to streams. We standardized and summed the maps of N and P retention.	unitless	LULC proxy-based (InVEST)
Erosion control	Ability of vegetation and soil to avoid initial nutrient and sediment loss by erosion assessed by the universal soil loss equation (USLE).	$Mg.ha^{-1}.year^{-1}$	LULC proxy-based (InVEST)
Supporting services			
Net primary productivity	Amount of atmospheric carbon fixed by plants and accumulated as biomass. We used the net primary productivity (NPP) from 2000 to 2009.	$Pg\ C.year^{-1}$	Primary data (Zhao & Running, 2010)
Soil fertility	Categories of soil fertility from Brazilian agricultural potential map. Levels: very high, high, mid and low.	unitless	Primary data (http://geoftp.ibge.gov.br)
Water balance	Annual amount of precipitation that does not evapotranspire given the water storage properties of the soil.	$mm.year^{-1}$	LULC proxy-based (InVEST)

We used primary data provided by the Brazilian Government (atlas and database) to produce the maps of soil fertility, water supply and agriculture. For soil fertility, we used the Brazilian agricultural potential map (http://mapas.mma.gov.br) that is divided in four categories of fertility (very high, high, mid and low). For water supply, we summed the number of registered underground water wells on the Brazilian underground water information system (http://siagasweb.cprm.gov.br). For agriculture, we used the Brazilian land use map of 2010, which is divided into three categories according to the relative area of agricultural farms (percentage per pixel): < 10%, 11-25% and 26-50% (http://geoftp.ibge.gov.br). We used a global assessment of net primary productivity (NPP) using MODIS satellite product MOD17A3 (Zhao & Running, 2010) to assess Caatinga's NPP average between 2000 and 2009. We used the map of carbon fixed in the aboveground live woody vegetation of tropical America (Baccini *et al.*, 2012) to estimate carbon storage aboveground (Ca). Belowground carbon storage (Cb) was calculated using the average belowground to aboveground biomass ratio (shoot-root ratio = 0.27) for tropical dry forest obtained from the Intergovernmental Panel on Climate Change (IPCC, 2006). And the soil organic carbon (Cs) was obtained from the global soil dataset of Harmonized World Soil Database (HWSD) (Hiederer & Köchy, 2011). The regulating service of carbon storage estimated in the Caatinga was calculated summing the Ca + Cb + Cs (Table 1).

When primary data was not available we estimated ecosystem services with proxy-based approach using InVEST (Integrated Valuation of Environmental Services and Tradeoffs), a modeling software used to map and value goods and services from nature developed by the Natural Capital project (www.naturalcapitalproject.org). InVEST uses land use and land cover map (LULC) and biophysical variables aiming to model the ecosystem services of target landscapes (Kareiva *et al.*, 2010; Tallis *et al.*, 2011). We used InVEST to model the supporting service of water balance and the regulating services of water purification and erosion control (Table 1). We used the LULC map of the Caatinga to estimate these three ecosystem services (Fig. 1). See Appendix S2 and Table S1 in Supporting Information for further information about the modeling of these ecosystem in InVEST. Water balance is related to the annual amount of precipitation that does not evaporate and transpire given the water storage properties of the soil (Mendoza *et al.*, 2011). Water purification is related to the capacity of each LULC category to retain nutrients (nitrogen and phosphorus) and to avoid their runoff to low lands and streams (Kareiva *et al.*, 2010). Erosion control is related to the difference of soil erosion among absence of land cover (potential soil erosion) and the presence of land cover or land management (current soil erosion) (Zhiyun *et al.*, 2011).

To estimate the provisioning service of fodder, we assumed that liveweight gain of livestock raised outside farms is directly related to native fodder consumed by them in the Caatinga vegetation areas. We calculated the total liveweight gain per pixel of free raised animals using the information of weight gain of livestock per head of sheeps, goats and cattle ($kg.ha^{-1}.year^{-}1$) provided by Filho and co-authors (2002) in the Caatinga vegetation areas (Fig. 1). Then, we multiplied the weight gain of livestock per head by the livestock density in each pixel (Robinson *et al.*, 2007) and summed the total weight gain of all type of livestock (Table 1 but see Appendix S2).

Spatial analysis of ecosystem services and biodiversity

We put all 12 single maps (three from biodiversity and nine from ecosystem services) at same resolution (0.05º), extent, datum and geographic coordinates system (WGS84) and performed analyses using *raster* and *maptools* packages in the software R 3.02 (R Core Development Team, 2005). We computed a matrix of spatial pairwise correlation between all maps of biodiversity and ecosystem services using Pearson's correlation test. Further, we created summed maps of each category (biodiversity, provisioning, regulating and supporting services) to analyze if these categories could have positive correlations as well. Summed maps were derived from the sum of z-scores of the three single maps of each category that were standardized by z transformation (original values minus the sample mean divided by standard deviation). We also analyzed spatial associations among these summed maps with Pearson's correlation test.

We first defined hotspots as the areas with high provision of ecosystem services and high biodiversity value. Then, we divided the values ranges in quantils and selected the hotspot areas those pixels with values above 5th quantile (the highest 20% values). Then, we calculated the percentage of hotspot areas from each ecosystem service and woody biodiversity map located inside the boundaries of the protected areas network (observed value) for both and each type of protected areas (strict protection and sustainable use). Then, we ran null models to test the null hypothesis that the protected areas were spatially distributed independently from hotspot areas of ecosystem services and woody biodiversity proxies. We constructed the null models randomizing the positions of protected areas network while holding the location of hotspots 999 times, and then calculating the percentage of hotspot areas inside the protected areas network (random values). Then, we tested observed values against the null distribution generated from random values for each ecosystem service and woody biodiversity map. We considered the observed values below 2.5% or above 97.5% probability of distribution different from random.

RESULTS

Spatial distribution of hotspots areas of woody biodiversity and ecosystem services variables were different even within same category (biodiversity, provisioning, regulating and supporting services) (Fig. 2).

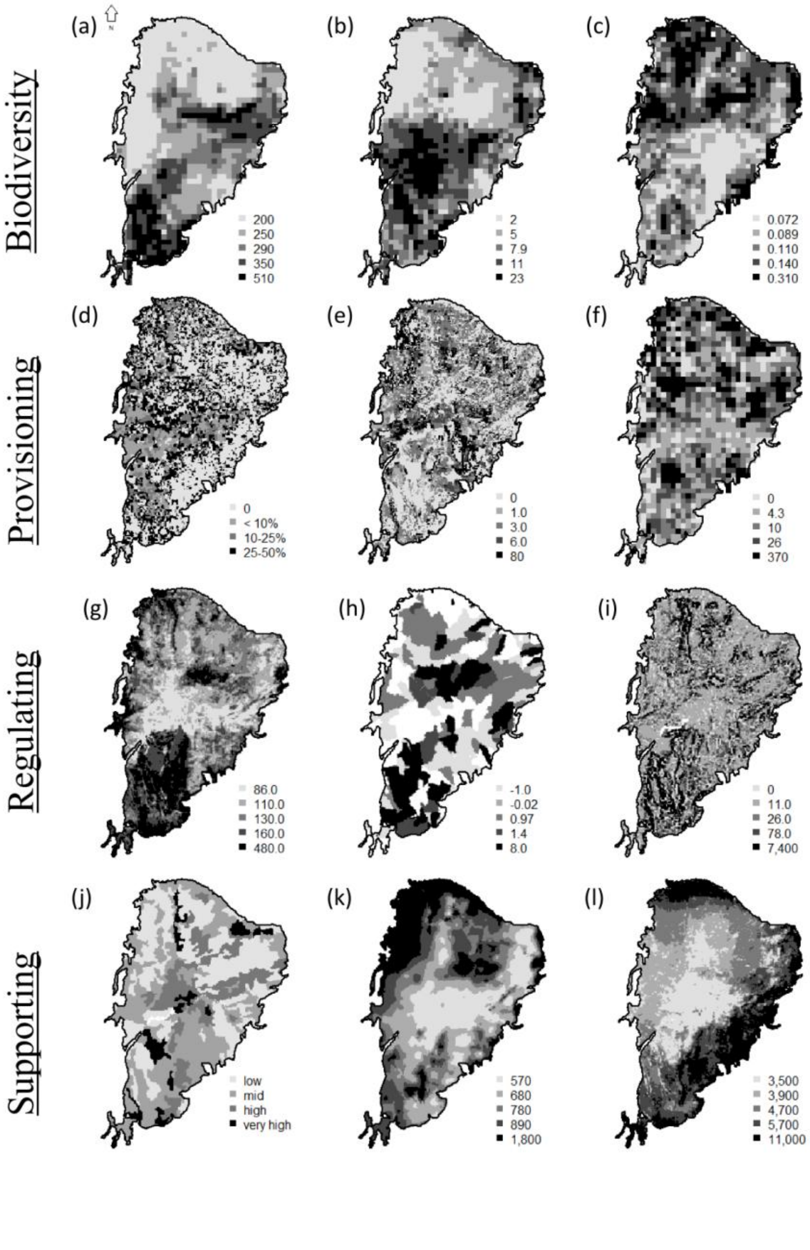

Fig 2. Single maps of biodiversity, provisioning services, regulating services and provisioning services. Biodiversity (BIO): (a) species richness, (b) narrow-range species richness, (c) beta-diversity. Provisioning services (PROV): (d) agriculture (% cover), (e) fodder (kg.ha^{-1}year^{-1}), (f) water supply (number of underground water wells). Regulating services (REG): (g) carbon storage (Mg.ha^{-1}), (h) water purification (standardized values summed from N and P retention maps), (i) erosion control (t.ha^{-1}year^{-1}). Supporting services (SUP): (j) net primary productivity (Pg C.year^{-1}), (k) soil fertility (from low to very high), (l) water balance (mm). Values higher than 5th quantile of single maps are the hotspot areas (black color).

Biodiversity vs. ecosystem services

Pairwise correlations among ecosystem services and woody biodiversity variables were all significant ($P < 0.05$) mainly because of the high amount of data. Thus, we considered $|r| \leq 0.20$ as low, $|r|$ values between 0.20 and 0.40 as intermediary and $|r|$ values ≥ 0.41 were set up as high correlations (Table 2 but see Figure S1 in Supporting Information). All biodiversity variables (Fig. 2a-c) were highly correlated with water balance (Fig. 2k). Species richness and narrow-range species richness were negatively correlated with water balance while beta-diversity was positively correlated with water balance. Species richness (Fig. 2a) had intermediary positive correlation with two regulating services, carbon storage (Fig. 2g) and water purification (Fig. 2h).

Table 2. Values of r from pairwise Pearson's correlation tests among single maps of biodiversity (BIO) and ecosystem services (PROV= provisioning, REG= regulating and SUP= supporting). Italic numbers are intermediary correlations (0.20 < |r| < 0.40) while bolded numbers are high correlations (|r| ≥ 0.40). SpRich= species richness; NarRan= Narrow-range species richness; BetDiv= beta-diversity; Agricul= agriculture; Fodder= fodder; Wsupp= water supply; CarSto= carbon storage; Wpurif= water purification; EroCon= erosion control; NPP= net primary productivity; SoilFer= soil fertility; Wbalan= water balance.

Variable		BIO			PROV			REG			SUP		
		SpRich	NarrRan	BetDiv	Agricul	Fodder	Wsupp	CarSto	Wqual	EroCon	PrimPro	SoilFer	Wbalan
BIO	SpRich	1.00	**0.47**	**-0.41**	-0.01	-0.13	-0.06	*0.24*	*0.35*	0.15	0.07	0.10	**-0.45**
	NarrRan	**0.47**	1.00	**-0.47**	0.05	-0.05	-0.13	0.00	-0.01	0.04	-0.10	0.14	**-0.58**
	BetDiv	**-0.41**	**-0.47**	1.00	0.00	-0.07	0.14	0.06	-0.06	0.04	0.09	-0.10	**0.45**
PROV	Agricul	-0.01	0.05	0.00	1.00	0.01	-0.01	0.00	-0.05	0.02	-0.13	0.02	0.03
	Fodder	-0.13	-0.05	-0.07	0.01	1.00	0.00	-0.05	-0.12	-0.05	-0.11	-0.03	0.07
	Wsupp	-0.06	-0.13	0.14	-0.01	0.00	1.00	-0.02	0.04	-0.03	0.02	0.03	0.15
REG	CarSto	*0.24*	0.01	0.06	0.00	-0.05	-0.02	1.00	0.14	0.16	*0.37*	0.05	*0.31*
	Wqual	*0.34*	-0.02	-0.06	-0.05	-0.12	0.04	0.14	1.00	0.06	-0.05	0.11	-0.15
	EroCon	0.15	0.04	0.04	0.02	-0.05	-0.03	0.16	0.06	1.00	0.04	-0.03	0.07
SUP	NPP	0.07	-0.10	0.09	-0.13	-0.11	0.02	*0.37*	-0.05	0.04	1.00	-0.10	*0.22*
	SoilFer	0.10	0.14	-0.10	0.02	-0.03	0.03	0.05	0.11	-0.03	-0.10	1.00	*-0.23*
	Wbalan	**-0.45**	**-0.58**	**0.45**	0.03	0.07	0.15	*0.31*	-0.15	0.07	*0.22*	*-0.23*	1.00

Representativeness of biodiversity hotspot areas was not different from random indicating that any biodiversity variable was found to be represented inside the protected areas network in Caatinga (Table 3). Nevertheless, hotspot areas of water balance and carbon storage were more represented inside protected areas network than at random. The category of sustainable use protected areas were more successfully allocated to protect water balance (11.9%; P = 0.025) while strict protection areas represent more the ecosystem service of carbon storage (2.9%; P = 0.002). Moreover, two provisioning services were underrepresented, observed percentage of fodder hotspot (0.5%; P = 0.979) and agriculture hotspot (0.4%; P = 0.999) were lower than the expected at random inside the strict protection areas.

Biodiversity and ecosystem services categories

Analyzing correlation among biodiversity variables, species richness (Fig. 2a) and narrow-range species richness (Fig. 2b) were highly positively correlated to each other but they were highly negatively correlated with beta-diversity (Fig. 2c). Negative spatial association also occurred within the supporting services variables, water balance (Fig. 2k) had intermediary negative correlation with soil fertility (Fig. 2j) but positive correlation with NPP (Fig. 2l).

NPP and water balance (supporting services) had intermediary positive correlation with one regulating service, the carbon storage (Fig. 2g). The summed map of standardized values for regulating services (Figure S2) had intermediary positive correlation with biodiversity (r = 0.29) and supporting services (r = 0.21).

Table 3. Percentage of woody biodiversity and ecosystem services hotspot areas observed (obs) inside the boundaries of all protected areas categories, inside the strict protection and sustainable use protected areas. Observed value is higher than expected at random when P value of the null model < 0.025 (*) and lower than expected at random when P value > 0.975 (†).

	Variables	All protected areas		Strict protection		Sustainable use	
		obs (%)	P	obs (%)	P	obs (%)	P
BIO	Species richness	6.62	0.452	1.21	0.350	5.41	0.491
	Narrow-range	11.30	0.215	1.04	0.590	10.29	0.205
	Beta-diversity	9.28	0.160	0.33	0.849	9.01	0.121
PROV	Agriculture	6.51	0.840	**0.45**	**0.999†**	6.07	0.603
	Fodder	6.16	0.780	**0.48**	**0.979†**	5.68	0.656
	Water supply	5.50	0.685	0.69	0.672	4.80	0.616
REG	Carbon storage	9.72	0.102	**2.94**	**0.002***	6.78	0.214
	Water purification	5.30	0.729	0.42	0.905	4.88	0.628
	Erosion control	9.32	0.149	1.37	0.319	7.98	0.184
SUP	Net primary productivity	7.41	0.128	0.66	0.422	6.20	0.153
	Soil fertility	3.30	0.751	1.56	0.33	1.74	0.792
	Water balance	**12.51**	**0.025***	0.73	0.449	**11.89**	**0.019***

DISCUSSION

Analysis of representativeness of woody biodiversity and ecosystem services inside the protected areas network in the Brazilian Caatinga revealed that only two ecosystem services are being represented (carbon storage and water balance). Despite of positive correlation among these ecosystem services with biodiversity (carbon storage with species richness and water balance with beta-diversity), none of the proxies of woody biodiversity were represented inside either protected areas of sustainable use or strict protection. According to complementarity approach, we were expecting to find ecosystem services positively correlated with biodiversity to be represented in protected areas for biodiversity conservation. Even though the positive correlations found among biodiversity variables and ecosystems services, the representation of those services in the protected areas network did not assure biodiversity protection. Thus, the use of complementarity approach as conservation criteria mostly based on surrogate choices might be not so effective as previously thought.

Usually, surrogates used to represent patterns of biodiversity and select conservation areas were either taxonomic (focal, umbrella or endemic species, for instance) or environmental, which includes biological and physical data (Pressey, 2004; Grantham *et al.*, 2010). In the past the criteria used in the Brazilian Caatinga to select priority areas and design protected areas network was mainly environmental, based on the size of remnant vegetation and on conservation status (Tabarelli *et al.*, 2003; Hauff, 2010). This criterion likely explains the representativeness of only carbon storage and water balance into the protected areas, once these ecosystem services are strictly dependent on the presence of vegetation. Habitats showing suitable conservation status are expected to provide higher biodiversity and regulating services than habitats with low conservation status (Maes *et al.*, 2012). However the use of conservation status in the Brazilian Caatinga prioritization was not able to include the proxies of woody biodiversity used here.

These evidences raise the importance of representing other features beyond biological biodiversity and landscape quality when planning to protect ecological aspects of biodiversity and ecosystem functioning. As a matter of fact, some authors have already claimed for new surrogates (Oliver *et al.*, 2004; Williams *et al.*, 2006) and methods when taking conservation decisions (Grantham *et al.*, 2010). We could not find a good surrogate to represent woody biodiversity and each ecosystem service category as negative correlations and weak positive correlations occurred. Although, correlation results of the summed maps highlight the importance of biodiversity and supporting services to provide the regulating services.

Positive correlation among woody species richness with carbon storage and water purification likely evidences that the role of plant biodiversity on ecosystem functioning may occur at large scales. Actually, several experiments conducted at local scale have shown the linking between plant species richness and the ecosystem functionality as biomass production and nutrient retention (Cardinale *et al.*, 2011 for review). The regulating service of carbon storage may be improved by the supporting services of NPP and water balance, also underlining prior findings of Chan and co-authors (2006), also at large scale.

Here by investigating the distribution of ecosystem services and woody biodiversity into the Caatinga network we had shown that humid areas (high water balance) inserted in the Caatinga are more represented than the semi-arid areas. As a result, ecotones and enclaves represent 35.3% from the small proportion of protected areas in Caatinga (7.4% of the total ecosystem). The arid vegetation typical of Caatinga is presented in 30.1% of this area. Nowadays, there are 34% endemic species present only in the Caatinga vegetation (Leal *et al.*, 2005). Thus, many of them are possibly not being included inside the protected areas network. Our results also highlighted the need for implementation of reserves in the strict protection category, which has proven to be effective to avoid the development of traditional economic activities as agriculture and fodder production, which are currently the main threats for Caatinga conservation.

Currently, reserves of strict protection has six times lower coverage than the sustainable use category (Hauff, 2010).

In summary, the analysis performed here pointed spatial association and representativeness of biodiversity and ecosystem services variables to be used as baseline for establishing protected areas network. The availability of these nature services has been decreased due to anthropogenic activities, even though ecosystem services are considered vital to enhance human well-being and to support economic activities (MA, 2005; Pascal *et al.*, 2010). As ecosystem services might not be used as justification for biodiversity conservation and vice-versa, the main contribution of the approach presented here is to show that positive associations among ecosystem services and biodiversity is not suitable enough to preserve ecosystem functioning and biological conservation. These evidences may guide conservation planners to better achieve conservation goals and improve human welfare by shedding light on selecting ecosystem services as additional targets on biodiversity for systematic conservation planning

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SUPPORTING INFORMATION

Additional Supporting Information may be found in the online version of this article:

Appendix S1 {Species distribution modelling}

Appendix S2 {LULC proxy-based methodology}

Figure S1 {Correlation graphs among variables}

Table S1 {Biophysical table used in InVEST}

SUPPORTING INFORMATION

Appendix S1 {Species distribution modeling}

Species distribution modeling (SDM) of all species were fitted with MaxEnt software using entire Brazilian territory as background. MaxEnt uses presence records to estimate the suitability of species occurrences based on correlations of known occurrences with the environmental variables of background landscape (Elith *et al.*, 2011).

Presence-only records

We used the woody species occurrence records from TreeAtlan 2.0 database which is a compilation of woody species records in different vegetation types in areas of tropical and subtropical extra-Andean South America (http://www.icb.ufmg.br/treeatlan/). From this database, we extracted presence records of species that occur in the Brazilian Caatinga and estimated the potential distribution area of all species selected (769 woody species).

Environmental variables

We collected the current climatic variables (average from 1950 - 2000) and altitude (Digital Elevation Model) from WordClim database (http://www.worldclim.org/current). We also used the Brazilian map of soil types provided by Brazilian Institute of Geography and Statistic (IBGE) (ftp://geoftp.ibge.gov.br/mapas_tematicos/mapas_murais/) and the variable of height above nearest drainage (HAND) available at National Institute for Spatial Research (INPE) (http://www.dpi.inpe.br/Ambdata/). We done pairwise Pearson's correlations test among all environmental variables and we selected only variables with correlation coefficients values below |0.7|. Following this criteria, we ran MaxEnt models using eight climatic variables from WorldClim (mean diurnal range, isothermality, mean temperature of warmest quarter, precipitation of wettest quarter, precipitation of driest quarter, precipitation of warmest quarter, precipitation of coldest quarter). We also used the environmental variables of altitude, HAND and soil type (Table S1). We fitted species distribution models at a 0.10° resolution.

Species richness map

We aggregated the suitability of occurrence values of each map of species distribution generated by SDMs process to a 0.05° resolution by the mean. We considered that the species was present when suitability of occurrence estimated in each pixel were above the 10 percentile presence threshold. Then, we categorized as value 1 (presence) the pixels that had values above this threshold and categorized as value 0 (absence) when values were below this threshold. We developed 769 maps of presence/absence of each species. Woody species richness was calculated summing these 769 binary maps and resulted a map ranging from 54 to 510 species.

Beta-diversity map

We calculated beta-diversity for each pixel (0.05°) using the woody species presence/absence maps. We used the number of species that occur in each target pixel and compared to the eight neighbor´s pixels using the symmetric form of Simpson´s asymmetric index ((Lennon *et al.*, 2001).

$$\beta = \frac{1}{n}\sum_{i=1}^{n}(1 - S) \qquad S = \frac{a}{a + min\ (b,c)}$$

S = resembles Sympson´s assymetric index;

n = number of pair-wise comparison (n=8 neighbor´s pixels);

a = number of species that are present in both pair-wise pixels;

b = number of species that are present only in neighboring pixel;

c = number of species that are present only in target pixel;

min (b,c) = decreases the influence of local species richness on dissimilarity index.

Appendix S2 {LULC proxy-based methodology: water balance, water purification, erosion control and fodder}

Water balance

Water balance is based on the hypothesis that water yield can be approximated by local interaction of precipitation and potential evapotranspiration given the water storage properties of the soil (Kareiva *et al.*, 2010). We used the water yield model from InVEST to estimate the supporting service of water balance and is defined as the annual amount of precipitation that does not evaporate and transpire (Kareiva *et al.*, 2010).

The InVEST methodology to model the water yield can be see here: http://www.naturalcapitalproject.org/models/hydropower.html.

Water yield (Yxj) is calculated as following:

$$Yxj = \left(1 - \frac{AET\ xj}{Pxj}\right) .\ Pxj .\ Axj$$

where $AETxj$ is the annual actual evapotranspiration in pixel x with LULC category j, Px is the annual precipitation in pixel x and LULC j and Axj is the area in pixel x and LULC j.

The evapotranspiration portion of water balance $\frac{AET\ xj}{Pxj}$ is an approximation of the Budyko curve developed by Zhang *et al.* (2004).

$$\frac{AETxj}{Pxj} = \frac{1 + \omega xj .\ Rxj}{1 + \omega xj .\ Rxj + \frac{1}{Rxj}}$$

where Rxj is the Budyko dryness index (ratio of potential evapotranspiration to precipitation) in pixel x and LULC j and ωxj is a dimensionless ratio of plant accessible water storage to expected precipitation during the year.

$$Rxj = \frac{Kcj\ .ET0x}{Pxj}$$

where Kc is the plant evapotranspiration coefficient associated with LULC j and $ET0x$ is the reference evapotranspiration in the pixel x and LULC j (based on alfafa).

$$\omega xj = Z\ \left(\frac{AWCx}{Pxj}\right)$$

where $AWCx$ is the measure of the water content in the soil available to plants and Z is a parameter applied to homogeneous basin in the landscape and is calculated with calibration.

Data needs (Tallis *et al.*, 2011) and respective sources used:

GIS raster dataset

1) Root restricting layer depth: http://www.iiasa.ac.at/Research/LUC/External-World-soil-database/HTML/ *

2) Precipitation: http://www.worldclim.org/current

3) Plant available water content: http://www.iiasa.ac.at/Research/LUC/External-World-soil-database/HTML/ *

5) Annual average reference evapotranspiration: http://csi.cgiar.org/Aridity/

6) Land use/land cover: Figure 1 in main text (MMA, 2006)

* We collected the values of root restricting layer depth and plant available water content from Harmonized World Soil Database (HWSD) according to the soil class based on FAO soil classification. We used the soil map based on Brazilian soil classes map (ftp://geoftp.ibge.gov.br/mapas_tematicos/mapas_murais/) to create two raster files (root restricting layer depth and plant available water content) based on HWSD dataset.

Shapefile

7) Watersheds: http://hidroweb.ana.gov.br/HidroWeb

8) Subwatershed: http://hidroweb.ana.gov.br/HidroWeb

Data

9) Biophysical table (Table S1)

9.1. Land use code: 1-16

9.2. Land use name: (1) farming, (2) water, (3) urban areas, (4) forested caatinga, (5) wooded caatinga, (6) park caatinga, (7) woody-grassy caatinga, (8) ombrophilus forest, (9) savannah, (10) seasonal forest, (11) secondary forest, (12) dunes, (13) caatinga/seasonal forest, (14) savannah/seasonal forest, (15) savannah/caatinga and (16) non-identified.

9.3. Root depth for each LULC class: Canadell *et al.* (1996)

9.4. *Kc:* plant evapotranspiration coefficient for each LULC class, used to obtain potential evapotranspiration by using plant physiological characteristics to modify the reference evapotranspiration (*ET0x*), which is based on alfalfa. The evapotranspiration coefficient is thus a decimal in the range of 0 to 1.5. There is only information about *Kc* for crop species and any *Kc* value was found for LULC classes of the Caatinga. Then, we used value $Kc = 1$ (Tallis *et al.*, 2011).

Water purification

More information about InVEST methodology to model water purification can be see here: http://www.naturalcapitalproject.org/models/water_purification.html. It estimates the quantity of pollutant (nitrogen and phosphorus) retained by each parcel of the landscape (watershed) based on annual average runoff from each parcel and the filtering capacity of each land use and land cover category (Tallis *et al.*, 2011) .
Annual average runoff is calculated by the Adjusted Loading Value at pixel *x* (*ALVx*):

$$ALVx = HSSx \,.polx$$

where *polx* is the export coefficient at pixel x (load P and load N in Table S1) and *HSSx* is the Hydrologic Sensitivity Score at pixel *x* which is calculated as:

$$HSSx = \frac{ʎx}{ʎw}$$

where $ʎw$ is the mean runoff index in the watershed of interest and $ʎx$ is the runoff index at pixel *x*, calculated using the following equation:

$$ʎx = Log\,(\sum_{U} Yu)$$

where $\sum_U Yu$ is the sum of the water yield (*Yxj* in water balance model) of pixel *x* along the flow path above pixel *x.*

Data needs (Tallis *et al.*, 2011) and respective sources used:

GIS raster dataset

1) Digital elevation model (DEM): http://www.worldclim.org/current
2) Root restricting layer depth: http://www.iiasa.ac.at/Research/LUC/External-World-soil-database/HTML/ *
3) Precipitation: http://www.worldclim.org/current
4) Plant available water content: http://www.iiasa.ac.at/Research/LUC/External-World-soil-database/HTML/ *
5) Annual average potential evapotranspiration: http://csi.cgiar.org/Aridity/
6) Land use/land cover: Figure 1 in the main text (MMA, 2006)

* We collected the values of root restricting layer depth and plant available water content from Harmonized World Soil Database (HWSD) according to the soil class based on FAO soil classification. We used the soil map based on Brazilian soil classes map (ftp://geoftp.ibge.gov.br/mapas_tematicos/mapas_murais/) to create two raster files (root restricting layer depth and plant available water content) based on HWSD dataset.

Shapefile

7) Watersheds: http://hidroweb.ana.gov.br/HidroWeb

Data

8) Biophysical table (Table S1):

8.1. Land use code: 1-16

8.2. Land use name: same as water balance model

8.3. Root depth for each LULC class: Canadell *et al.* (1996)

8.4. *Kc*: same as water balance model

8.5. Nutrient loading (nitrogen and phosphorus) for each LULC class (load P and load N): Young *et al.* (1996) and Jeje (2006).

8.6. Vegetation filtering value for each LULC class (eff. P and eff. N): ranging between 0 and 100, using expertise knowledge.

We ran two models, one for nitrogen (N) retention and other for phosphorus (P) retention. The output is the total amount of the nutrient (P or N) retained by each watershed (Kg/watershed). We standardized (z-scores) the values of each map of phosphorus and nitrogen retention estimated by watershed and summed to create only one map of water purification.

Erosion control

The InVEST methodology to model the erosion control can be see here: http://www.naturalcapitalproject.org/models/sediment_retention.html. The regulating service of erosion control is based on the ability of vegetation and soil to avoid initial nutrient and sediment loss by erosion (Kareiva *et al.*, 2010). We estimated erosion control as the difference of potential soil erosion (RKLS) and the current soil erosion (USLE) as described by Zhiyun *et al.* (2011). We calculated current soil erosion using the Universal Soil Loss Equation (USLE) derived from the sediment retention model in InVEST:

$$\text{USLE} = R\ .\ K\ .\ LS\ .\ C\ .\ P$$

R= rainfall erosivity;

K= soil erodibility;

LS= slope length-gradient factor;

C= cover management factor;

P= support practice factor.

Potential soil erosion was calculated using USLE equation but without C and P factors (RKLS) that are related to management of the land.

Data needs (Tallis *et al.*, 2011) and respective sources used:

GIS raster dataset

1) Digital elevation model (DEM): http://www.worldclim.org/current, to calculate LS

2) Rainfall erosivity index: Oliveira *et al.* (2012)

3) Soil erodibility: da Silva *et al.* (2011)

4) Land use/land cover: Figure 1 in main text (MMA, 2006)

Shapefile

5) Watersheds: http://hidroweb.ana.gov.br/HidroWeb.asp?TocItem=4100

Data

6) Biophysical table (Table S1)

6.1. Land use code: 1-16

6.2. Land use name: same as water balance model

6.3. C factor for each LULC class: Silva *et al.* (2007) and Farinasso *et al.* (2010)

6.4. P factor for each LULC class: Tomazoni & Guimarães (2009)

6.5. Sediment retention value for each LULC class (eff. SedRet): ranging between 0 and 100, using expertise knowledge (Table S1).

Fodder

Native fodder production in the Caatinga (woody and herbaceous) is an important provisioning service to feed livestock raised freely in native vegetation. We estimated the potential fodder production using the proxy of total weight gain of livestock (sheeps, goats and cattle) raised only in the Caatinga vegetation.

GIS raster dataset

(1) Livestock density (LVD): three maps of the total number of sheeps, goats and cattle estimated per pixel (Robinson *et al.*, 2007)

Data

(2) Weight gain of livestock: per head weight gain of sheeps, goats and cattle ($kg.ha^{-1}.year^{-}1$) in each class of the Caatinga vegetation (Filho *et al.*, 2002) related to the LULC Caatinga classes: (4) forested caatinga, (5) wooded caatinga, (6) park caatinga, (7) woody-grassy caatinga.

We calculated the total weight gain of livestock by the sum of each type of weight gain of livestock (sheeps, goats and cattle) that was calculated by the multiplication of the per head weight gain of each type of livestock (kg.ha-1.year-1) in each class of the Caatinga vegetation by respective livestock density estimated per pixel.

Table S1 {Biophysical table used in InVEST to model the ecosystem services of water purification, water balance and erosion control}

LULC description	LU code	root depth	Kc	load P	load N	eff. P	eff. N	C factor	P factor	eff. SedRet
Farming	1	2100	1	737	4225	25	25	21	533	40
Water	2	1	1	0	0	0	0	0	1	0
Urban area	3	1	1	160	3830	5	5	1	950	10
Forested caatinga	4	5100	1	178	2225	75	75	13	1	60
Wooded caatinga	5	7000	1	200	2500	80	80	13	1	60
Park caatinga	6	500	1	165	2063	75	75	13	1	50
Woddy-grassy caatinga	7	500	1	152	1020	40	40	13	1	40
Ombrophilus forest	8	1500	1	200	2500	90	90	1	1	70
Savannah	9	7000	1	90	1000	70	70	42	1	35
Seasonal forest	10	3700	1	200	2500	85	85	7	1	65
Secondary forest	11	600	1	165	2063	95	95	1	1	75
Dunes	12	1	1	0	0	0	0	1000	1	0
Ecotone (caatinga/seasonal forest)	13	5350	1	200	2500	82	82	10	1	62
Ecotone (savannah/seasonal forest)	14	5350	1	145	1750	77	77	24	1	62
Ecotone (savannah/caatinga)	15	7000	1	145	1750	75	75	87	1	48
Non-identified	16	1	1	1	1	1	1	1	1	1

Figure S1 {Correlation graphs among variables with |r| > 0.20 using the pixels number}

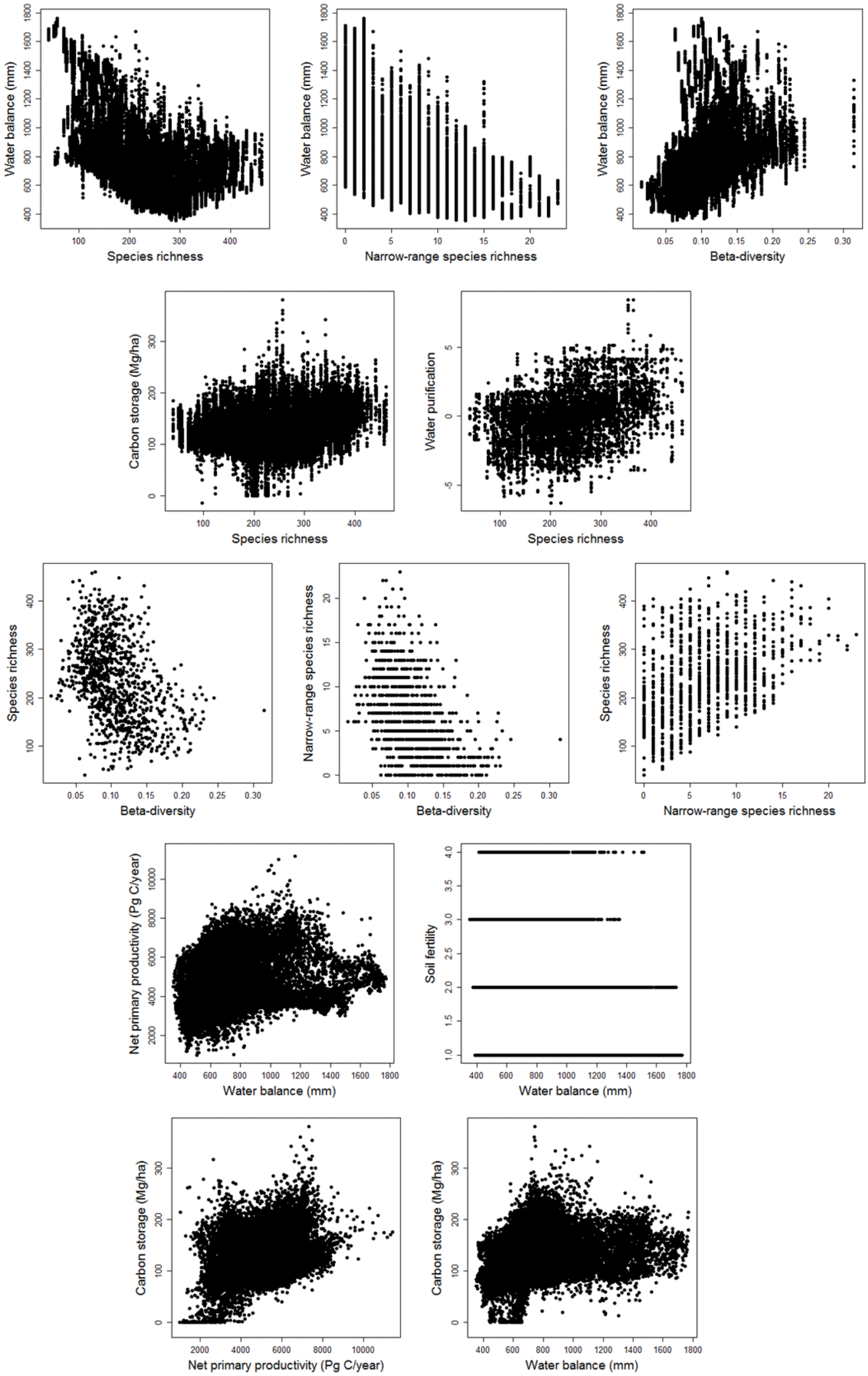
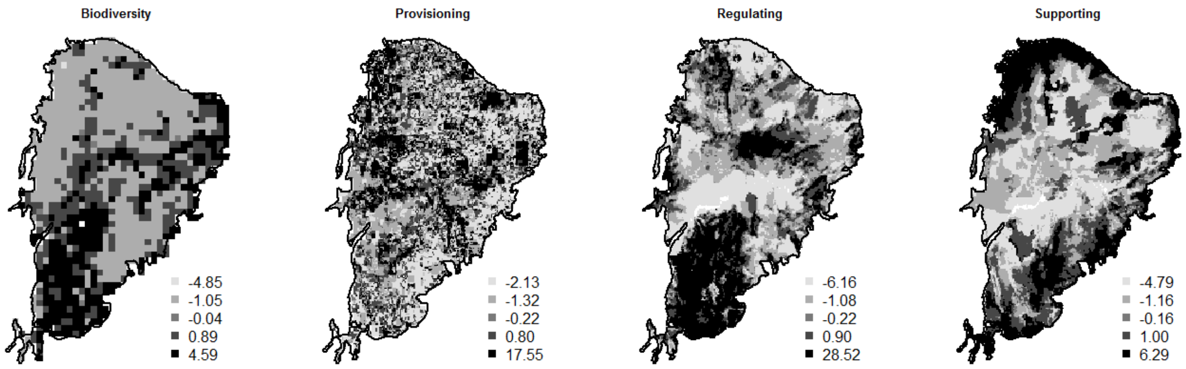


Figure S2 {Summed maps of biodiversity, provisioning, regulating and supporting services}



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CAPÍTULO III

MATCHING THE CONSERVATION OF ECOSYSTEM SERVICES AND BIODIVERSITY WITH SOCIOECONOMIC COSTS

Matching the conservation of ecosystem services and biodiversity with socioeconomic costs

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Abstract

Ecosystem services are the benefits provided for human well being derived from ecological processes. They must be included into systematic conservation planning in addition to biodiversity features to assure their provision. Here we identified priority sites for conservation investment in the seasonally dry tropical forest (known as the Caatinga) based on spatial distribution of 685 tree species and eight ecosystem services. We developed one prioritization scenario with no cost and three scenarios including opportunity costs (social, economic and socioeconomic). We used plant species and supporting services (water balance, primary productivity and soil fertility) as conservation targets, added provisioning services (water supply and fodder) to identify areas for sustainable use, and included regulating services (water purification, carbon storage and avoided erosion) to select areas for strict protection. Provisioning and regulating services had the highest decrease of proportion protected when socioeconomic costs were considered in prioritization, 54.2% and 33.4%, respectively. Biodiversity had a lower decrease, 2.8% in sustainable use areas and 10.4% in strict protection areas. Overall, spatial overlapping among priority areas and areas with high human population density and economic agriculture decreased in all cost scenarios. The choice of the best scenario will depend on the use allowed in the areas. Areas allowing economic activities may join socioeconomic and conservation goals with sustainable management while the places spared for protection must avoid overlapping with high socioeconomic development areas.

Keywords: Caatinga, conservation features, opportunity costs, Zonation

Highlights

• We developed conservation plans for ecosystem services and plant diversity.

• We used agriculture and population density as socioeconomic opportunity costs.

• The inclusion of costs reduced the representation of biodiversity and ecosystem services in the region.

• Priority areas for nature protection and those targeted for human development had low spatial overlap.

• Integration of ecosystem services in conservation planning may provide new insights for conservation policy.

1. Introduction

Systematic conservation planning has been developed to set priority areas that embrace as many biodiversity features as possible based on the concept of complementarity (Margules and Pressey 2000). It has been often assumed that ecosystem services could be protected bundled with biodiversity (Balvanera et al. 2001) but the use of biodiversity-only strategy could be not so effective to protect ecosystem services (Thomas et al. 2013). Moreover, planning outputs tend to fail when gains and losses for all stakeholders involved in different planning scenarios are not clear or not properly measured (McShane et al. 2011). Hence, trade-offs analysis may help to ally different conservation goals (biodiversity and ecosystem services) with social goals, such as poverty alleviation and economic development (Hirsch et al. 2011).

In some cases, synergies between biodiversity and ecosystem services arise, e.g. in Brazilian dry forest, plant species richness was positive correlated with both carbon storage and water purification, two important regulating services (Manhães et al. 2015). However, most studies have shown a trade-off between protecting biodiversity and maintaining ecosystem services at the landscape scale (Anderson et al. 2009; Bai et al. 2011; O´Farrell et al. 2010; Turner et al. 2007). Altogether, regulating services (e.g. water purification, carbon storage) have been positively correlated with biodiversity, whereas provisioning services (e.g. provision of food, material, water) have shown spatial incongruence (Cimon-Morin et al. 2013). Despite existent trade-offs, it is possible to ally different goals into a unified conservation planning strategy (Chan et al. 2006; Thomas et al. 2013; Wickham and Flather 2013).

Trade-offs may also take place when conservation costs are integrated in prioritization and some biodiversity or services targets may not be retained in some areas. Conservation action carries intrinsic costs that are necessary to cover all steps to implement the intervention and are classified in acquisition, management, transaction, damage and opportunity costs (Naidoo et al. 2006). Global analyses showed that conservation costs were positively correlated with human population density and with economic activities measured as mean per capita gross net product (GDP) (Balmford et al. 2003). These socioeconomic costs are related to the cost of forgone opportunities to use the land (opportunity cost), for example, urbanization and economic development. For example, conservation strategies that included social goals decreased the loss of agricultural production, but at the same time protected less biodiversity than expected when food production did constrains the selection of priority areas (Dobrovolski et al. 2014). In Brazilian Cerrado, biodiversity representation decreased 13% in proportion prioritized (relative to 17% of the Cerrado) when all socioeconomic costs were included in the analysis (Faleiro and Loyola 2013). Regardless of these explicitly trade-offs, the inclusion of conservation costs can improve the effectiveness of conservation through substantial benefits at low costs in more isolated areas (Balmford et al. 2003). Then, future conflicts and pressure on planned protected areas could be avoided.

Here we compare four prioritization scenarios, with and without considering opportunity costs, to select priority areas in the Brazilian dry forest. We used plant biodiversity and supporting ecosystem services as our main conservation goals. However, we added two provisioning services when planning for priority areas for sustainable use, and three regulating services when planning for areas where strict protection is needed. We expect that prioritization outputs that include socioeconomic costs would decrease opportunity costs derived from conservation, but at the expense of a decreasing proportion of protection for each conservation goal. Based on our results, we discuss which scenario could fit better to each type of conservation strategy (strict protection or sustainable use) according to the balance among human development and biodiversity conservation.

2. Methods

2.1. Study area

The northeast Brazil holds the seasonally dry tropical forest biome called "Caatinga" (Fig. 1). Seasonally dry tropical forest includes tall forest in moister sites to scrub rich succulent on the driest sites, has rainfall less than 1800 mm. $year^{-1}$, with a period of 5-6 months receiving less than 100mm (Pennington et al. 2009). The Caatinga dry forest is characterized by steppe vegetation, mostly deciduous during the dry season and has often dense and continuous formation of tree and shrubs cover (Bellefontaine et al. 2000). However, enclaves of seasonal forest, ombrophilus forest and savannah and ecotones occur as well (Fig. 1).

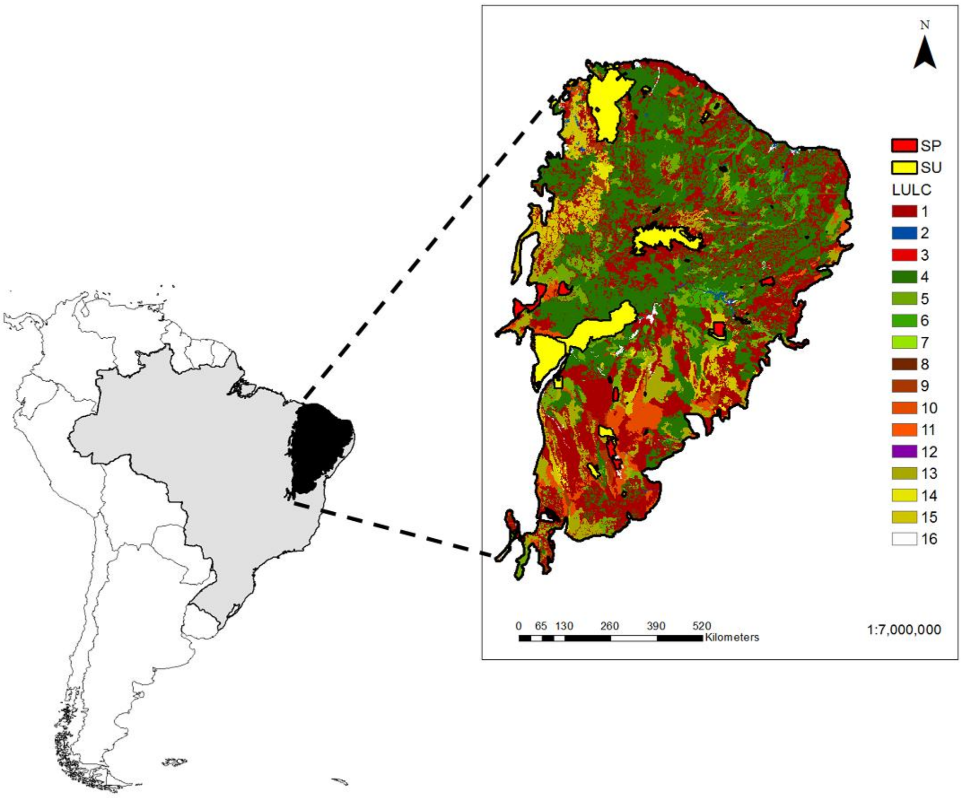

Figure 1. Location of Brazilian seasonally dry tropical forest (Caatinga, black color). In the right side, Caatinga land cover and land use map (LULC) with respective categories of non-vegetation areas (redish colors): (1) farming, (2) water and (3) urban areas; caatinga vegetation areas (greenish colors): (4) forested caatinga, (5) wooded caatinga, (6) park caatinga, (7) woody-grassy caatinga; enclave (brownish colors): (8) ombrophilus forest, (9) savannah, (10) seasonal forest; (11) secondary forest (orange color), (12) dunes (purple color); ecotone (yellowish colors): (13) caatinga/seasonal forest, (14) savannah/seasonal forest, (15) savannah/caatinga and (16) non-identified. Reserve network in Caatinga are the strict protection protected areas (SP, red color) and sustainable use protected areas (SU, yellow color).

The Brazilian Caatinga has an area of 826,411 Km^2 (11% of the Brazilian territory) in the semi-arid region (969,589 Km^2) that has the evapotranspiration rate three times higher than the rates of precipitation causing water shortage in this region (ASA - Brazilian semi arid articulation, http://www.asabrasil.org.br). Currently, the main threats of this biome are deforestation, which has reached about 47% of its total area (MMA 2009), and desertification process which extends by 15% of its total area (Leal et al. 2005). Conservation goals will vary with the purpose of a given protected area and two broad main management strategies exist in Brazil: targeting protected areas for strict protection (which are equivalent to IUCN protected areas in categories I-IV) and targeting them for sustainable use (equivalent to IUCN V and VI categories). Inside protected areas under strict protection, the direct use of natural resources are strictly controlled, whereas in those areas targeted for sustainable use, local inhabitants practices are permitted as long as these practices are managed and considered sustainable (SNUC 2000).

2.2. Data

2.2.1. Species distribution models (SDM)

We built SDM using the Maximum Entropy (MaxEnt) software that uses presence records to estimate the suitability of species occurrences on the basis of correlations of known occurrences with the environmental variables of the background landscape (Elith et al. 2011). As input for the modeling we used presence-only records for 685 woody plant species from the Caatinga, obtained from the TreeAtlan database (Oliveira-Filho 2010). Environmental variables were obtained from Worldclim (http://www.worldclim.org). We also included as environmental variables the map of soil types (ftp://geoftp.ibge.gov.br/mapas_tematicos/mapas_murais/) and the height above nearest drainage (HAND) variable (http://www.dpi.inpe.br/Ambdata/hand.php). Output of the model was the suitability of occurrence for each species and the spatial resolution was aggregated by mean to 0.05°. SDM methodology is detailed in the online Appendix A.

2.2.2. Assessment of ecosystem services

To map ecosystem services two types of data are commonly used: primary data on ecosystem services within the study region or proxy-based data, which links land cover to ecosystem service provision (Eigenbrod et al. 2010). Ecosystem services are classified in provisioning services (products obtained from ecosystems), regulating services (benefits provided by the regulation of ecosystem processes), and supporting services (those necessary for production of all other ecosystem services) (MA 2005). Here, we mapped eight ecosystem services: two provisioning services (fodder and water supply), three regulating services (carbon storage, water purification and erosion control) and three supporting services (primary productivity, soil fertility and water balance) (Table 1).

Table 1. Description of ecosystem services, units of measurement (pixel of 0.5º) and methods and sources used to estimate the service.

Ecosystem services	Description	Unit	Methods and sources
Provisioning services			
Fodder	Native fodder production in the Caatinga vegetation estimated by weigth gain of livestock (sheeps, goats and cattle) in each vegetation type.	$kg.ha^{-1}.year^{-1}$	LULC proxy-based
Water supply	Underground water wells established for human water use that is registered on Brazilian underground water information system.	number of wells registered	Primary data (http://siagasweb.cprm.gov.br)
Regulating services			
Carbon storage	Carbon density contained in above and below ground of live woody vegetation summed to the soil organic carbon density.	$Mg.ha^{-1}$	Primary data (IPCC, 2006; Cardinale *et al.*, 2011; Hiederer & Köchy, 2011; Baccini *et al.*, 2012).
Water purification	Capacity of each LULC category to retain nutrients (N and P) avoiding their runoff to streams. We standardized and summed the maps of N and P retention.	unitless	LULC proxy-based (InVEST)
Erosion control	Ability of vegetation and soil to avoid initial nutrient and sediment loss by erosion assessed by the universal soil loss equation (USLE).	$Mg.ha^{-1}.year^{-1}$	LULC proxy-based (InVEST)
Supporting services			
Net primary productivity	Amount of atmospheric carbon fixed by plants and accumulated as biomass. We used the net primary productivity (NPP) from 2000 to 2009.	$Pg\ C.year^{-1}$	Primary data (Zhao & Running, 2010)
Soil fertility	Categories of soil fertility from Brazilian agricultural potential map. Levels: very high, high, mid and low.	unitless	Primary data (http://geoftp.ibge.gov.br)
Water balance	Annual amount of precipitation that does not evapotranspire given the water storage properties of the soil.	$mm.year^{-1}$	LULC proxy-based (InVEST)

We used primary data provided by the Brazilian Government (atlas and database) to produce the maps of soil fertility and water supply. For soil fertility, we used the Brazilian agricultural potential map (http://mapas.mma.gov.br) that is divided in four categories of fertility (very high, high, mid and low). For water supply, we summed the number of registered underground water wells on SIAGAS, the Brazilian underground water information system (http://siagasweb.cprm.gov.br) (Table 1).

We used a global assessment of the net primary productivity (Zhao and Running 2010) to estimate the net primary productivity (NPP) in the Caatinga calculating the mean from 2000 to 2009 (Table 1). We used the map of carbon contained in the aboveground live woody vegetation of tropical America (Baccini et al. 2012) to estimate carbon storage aboveground (Ca). Belowground carbon storage (Cb) was calculated using the average belowground to aboveground biomass ratio (shoot-root ratio = 0.27) for tropical dry forest obtained from the Intergovernmental Panel on Climate Change (IPCC 2006). And the soil organic carbon (Cs) was obtained from the global soil dataset of Harmonized World Soil Database (HWSD) (Hiederer and Köchy 2011). The regulating service of carbon storage estimated in the Caatinga was calculated summing the Ca + Cb + Cs (Table 1).

When primary data was not available we estimated ecosystem services using the proxy-based approach in InVEST (Integrated Valuation of Environmental Services and Tradeoffs), which is a modeling software used to map and value goods and services from nature developed by Natural Capital project (www.naturalcapitalproject.org). InVEST uses land use and land cover (LULC) map and biophysical variables aiming to model the ecosystem services in the target landscape (Kareiva et al. 2010; Tallis et al. 2011). InVEST was used to model the regulating services of water purification and erosion control and the supporting service of water balance (Table 1). We used the LULC map of caatinga (MMA 2006) to estimate these three ecosystem services (Fig. 1). Water purification is related to the capacity of each LULC category to retain nutrients (nitrogen and phosphorus) and avoid their runoff to low lands and streams (Kareiva et al. 2010). Erosion control is related to the difference of soil erosion among absence of land cover (potential soil erosion) and the presence of land cover or land management (current soil erosion) (Zhiyun et al. 2011). Water balance is related to the annual amount of precipitation that does not evapotranspire given the water storage properties of the soil (Mendoza et al. 2011). See online Appendix B for further information about proxy-based maps using InVEST.

To estimate the provisioning service of fodder, we assumed that the liveweight gain of livestock raised outside farms is directly related to native fodder consumed by them inside the steppe vegetation areas. Using the LULC map, the information of weight gain of livestock per head of sheeps, goats and cattle ($kg.ha^{-1}.year^{-}1$) in each category of steppe vegetation (Filho et al. 2002) and the livestock density (Robinson et al. 2007) we calculated the total liveweight gain per pixel of all animals raised freely in caatinga vegetation (Table 1 but see online Appendix B).

2.2.3. Selection of priority areas

We ran all prioritization analyses using the Zonation Conservation Planning Software (version 4.0, Conservation Biology Informatics Group, Helsinki, Sweden; http://cbig.it.helsinki.fi/software/zonation). Zonation is a framework for conservation prioritization and planning at a large-scale which identifies areas that are important for retaining habitat quality and connectivity for multiple species (or other features) (Moilanen et al. 2012). We used the basic core-area Zonation algorithm that is based on cell remove rules determining which cell has the smallest marginal loss of biodiversity (Moilanen et al. 2012). This methodology is more detailed in online Appendix C.

We included the existent protected areas in prioritization using the shapefile of their spatial location (http://mapas.mma.gov.br) as an input mask file in the prioritization analysis. We classified opportunity costs in economic and social costs. Economic costs were estimated through gross domestic product added by agriculture per municipality measured in Brazilian currency (BRL; In January of 2010, 1.0 BRL = 0.57 USD) (http://www.ibge.gov.br) . We estimated the social cost via human population density measured by person per square kilometers (http://sedac.ciesin.columbia.edu).

With these data, we developed four prioritization scenarios: (i) a no cost scenario; (ii) an economic cost scenario, (iii) a social cost scenario, and (iv) socioeconomic scenario (using both economic and social costs). All inputs maps used in Zonation were put at same resolution (0.05º), extent, datum and geographic coordinates system (WGS84).

As conservation strategies focusing on sustainable use of natural resources or the strict protection of natural ecosystem differ greatly, we chose different conservation goals for these two complementary types of conservation strategies. Regulating services are mainly dependent on the maintenance of vegetation cover and might be safeguarded inside areas under strict protection. On the other hand, provisioning services are related to food and water provision and might be priority in areas targeted to sustainable use, which allow human settlements inside their boundaries. Supporting services like primary productivity, water balance and soil fertility are important services to support all ecosystem services and were considered as priority in both strategies, and so was plant biodiversity.

Prioritization analyses in Zonation can be parameterized weighting the goals to balance different values for each goal, but negative weights can be used for competing land uses (Moilanen et al. 2012). Each plant species was weighted by +1/ 685 (total number of species). We weighted +1/6 for the supporting and regulating services used to find areas best suited for protection and +1/5 for the supporting and provisioning services in areas targeted for sustainable use. At the end, biodiversity and ecosystem services had same aggregated weight (+1.0). In the no cost scenario, each opportunity cost (GDP added by agriculture and population density) was weighted by zero while we negatively weighted (-1.0) GDP added by agriculture in economic cost, population density in social cost and both opportunity cost in socioeconomic costs scenarios.

2.3. Analysis

Data and maps derived from Zonation were analyzed and plotted using R software 3.02 and the packages of *maptools*, *rgdal*, *raster*, *GISTools*, *maps* and *rgeos* (R Core Development Team 2005). To understand how conservation features (ecosystem services and biodiversity) differ in different scenarios, we assessed the performance curves that describes the performance of solution at given level of cell removal (Moilanen et al. 2012). A linear relation in performance curves means that for every proportion of areas protected by our prioritization plan, the same proportion of the conservation feature would be protected. Logarithmic (higher concave) and exponential (lower concave) curves indicate higher and lower percentage of the features protected relative to the proportion of priority areas protected, respectively. Using the proportion of 17% of priority areas protected based on Aichi Biodiversity Targets (http://www.cbd.int/sp/targets/), we calculated the percentage of the features (biodiversity, supporting services, provisioning services, regulating services, GDP added by agriculture and population density) that could be protected within each of the four scenarios (no cost, economic cost, social cost and socioeconomic costs). Then, we calculated the percentage of the feature protected in each opportunity costs scenarios relative to the no cost scenario.

3. Results

Maps of conservation features (ecosystem services and plant biodiversity), opportunity costs (gross domestic product added by agriculture and population density) and the mask of current protected areas used for prioritization in Zonation can be seen at Figure 2.

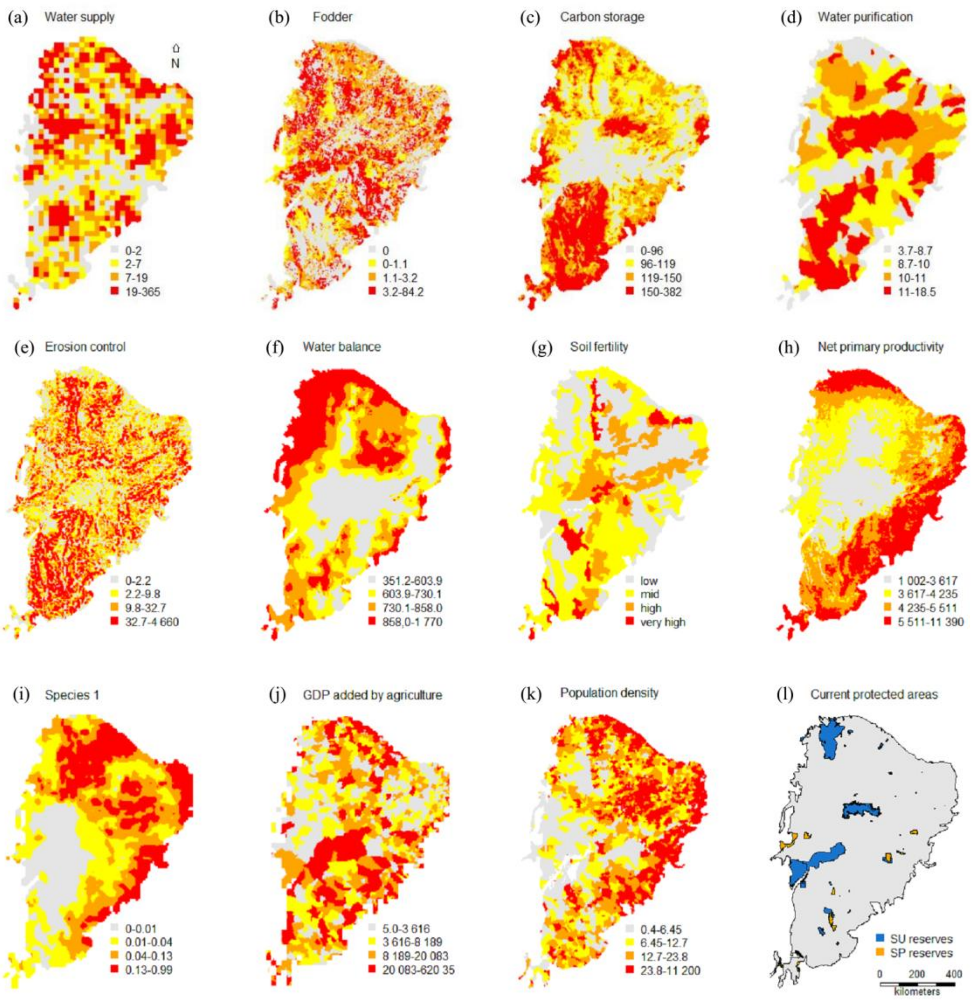

Figure 2. Maps used in Zonation: provisioning services (a) water supply (number of the underground water wells), (b) fodder ($kg.ha^{-1}year^{-1}$); regulating services (c) carbon storage ($Mg.ha^{-1}$), (d) water purification (standardized values summed from N and P retention maps), (e) erosion control ($t.ha^{-1}year^{-1}$); supporting services (f) water balance (mm), (g) soil fertility, (h) net primary productivity ($Pg\ C.year^{-1}$); biodiversity (i) suitability of occurrence of species 1; 685 species distribution maps were used to represent the biodiversity target; socioeconomic costs (j) gross domestic product added by agriculture (BRL per municipality), (k) population density (persons per km^2); (l) Sustainable Use (SU) protected areas (blue color) and Strict Protection (SP) protected areas (orange color) that were used as mask file. Red, orange, yellow and grey colors are respectively, the 100-75%, 75-50%, 50-25% and 25-0% quantile.

The spatial distribution of priority areas resulted from prioritization in Zonation (17% of the Caatinga including current protected areas) changed when opportunity costs were included for both categories of protected areas (Fig. 3). When single or both opportunity costs were negatively weighted some priority areas selected in no cost scenario were set aside, both under a sustainable use (Fig. 3a-d) and under the strict protection strategy (Fig. 2e-). The analysis of performance curves elucidates the change of each feature protection in all prioritization scenarios.

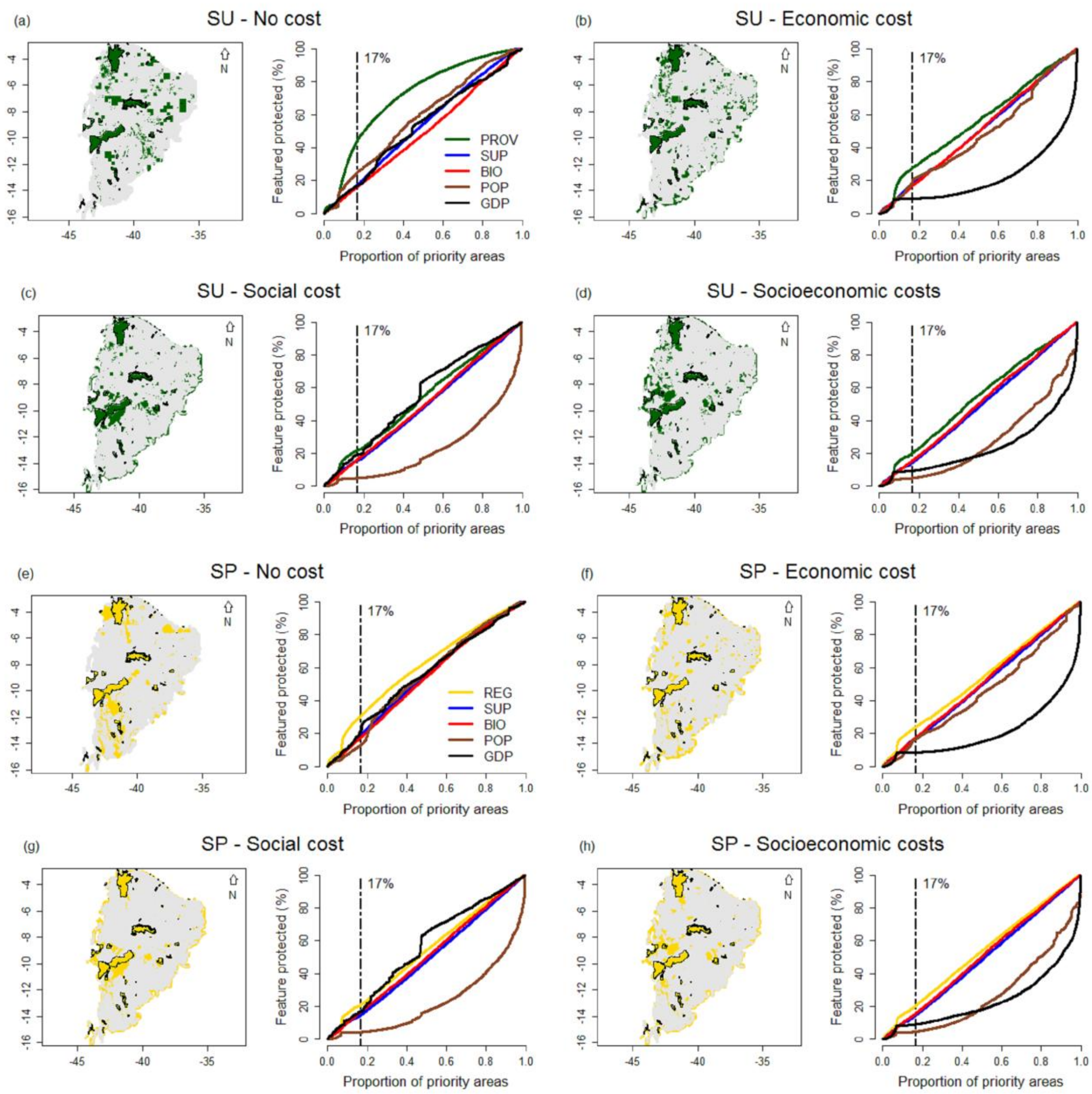

Figure 2. Priority areas for conservation in caatinga selected by Zonation (17% highest values; green color in Sustainable Use - SU and yellow color in Strict Protection- SP) and respective performance curves in four prioritization scenarios (no cost, economic cost, social cost and socioeconomic costs). Selection of priority areas for Sustainable Use protected areas in (a) no cost scenario, (b) economic cost scenario, (c) social scenario and (d) socioeconomic costs scenario. Selection of priority areas for Strict Protection protected areas in (e) no cost scenario, (f) economic cost scenario, (g) social scenario and (h) socioeconomic costs scenario. PROV = provisioning services; REG = regulating services; SUP = supporting services; BIO = plant biodiversity; GDP = gross

domestic product added by agriculture and POP = population density. We selected the threshold of 17% (Aichi Biodiversity Targets) to calculate the relative difference among scenarios. Current protected areas are delimited by black polygons.

Performance curves showed the representation of conservation features in all scenarios and the decrease of the opportunity costs when they were negatively weighted (lower concave curves in costs scenarios; GDP and POP in Fig. 3). Plant biodiversity and supporting services maintained a linear relationship in all scenarios for both conservation strategies (BIO and PROV in Fig. 3), indicating that the protection of these features did not change with the inclusion of opportunity costs. Moreover, the highest concave curve occurred for the provisioning services in no cost scenario under the strategy focusing on areas for sustainable use (Fig. 3a). When we compared the three opportunity costs scenarios, the economic cost had the lowest decreasing of the provisioning services (Fig. 3b). Although, in priority areas target for strict protection, the regulating services in no cost scenario presented just a few higher concave curve (Fig. 3e) and did not have any significant difference among opportunity costs scenarios (Fig. 3f-h). Using the proportion of 17% (Aichi Biodiversity Targets) of priority areas protected, we assessed the percentage protected of each feature for all scenarios (Appendix C) to calculate the percentage of protection relative to no cost scenario in each opportunity cost scenario (Fig. 4).

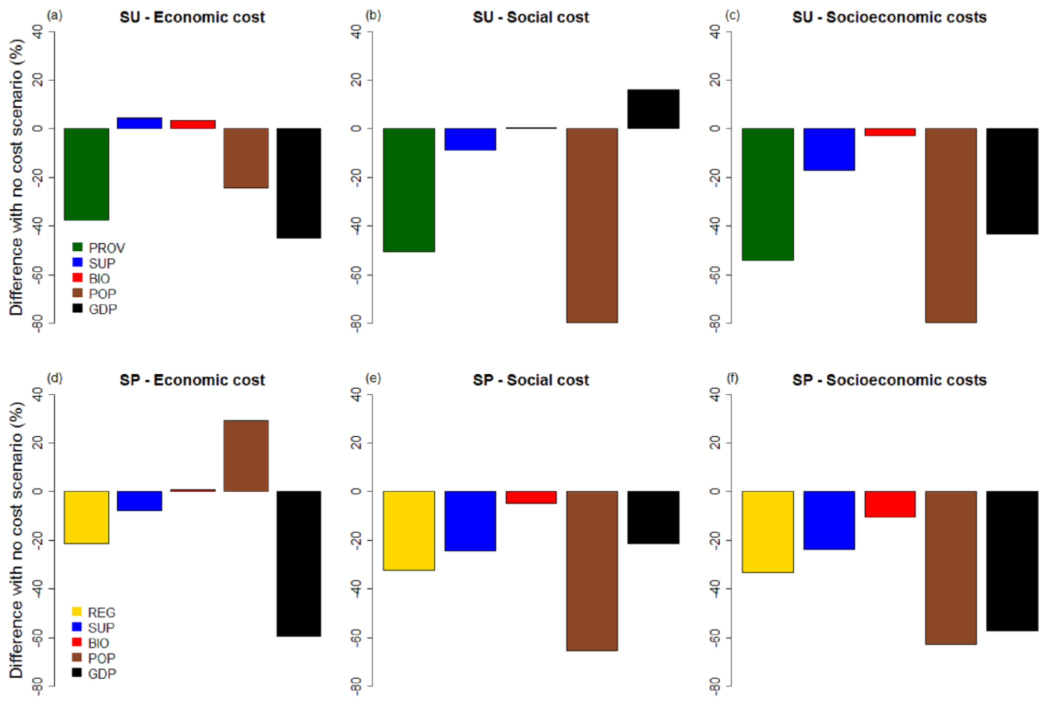

Figure 4. Percentage of protection of the conservation features (PROV - provisioning services, REG - regulating services, SUP - supporting services and BIO - plant biodiversity) and opportunity costs (GDP - gross domestic product of agriculture and POP - population density) relative to no cost scenario in each opportunity cost scenario. Relative difference for areas targeted for sustainable use (SU) in (a) economic cost scenario, (b) social scenario and (c) socioeconomic costs scenario. Relative difference for areas targeted for strict protection (SP) protected areas in (d) economic cost scenario, (e) social scenario and (f) socioeconomic costs scenario.

The representation of conservation features decreased in all opportunity costs scenarios with exception in economic cost scenario in areas targeted for sustainable use (Fig.4 a). In this scenario, supporting services increased the protection by 4.42% in relation to no cost scenario while plant biodiversity increased 3.43%. For the areas suitable for sustainable use, provisioning services were the conservation features that had the highest decrease of proportion protected when opportunity costs were taken into account (Fig. 4a-c) with higher decreasing in the socioeconomic cost (-54.23%). In priority areas targeted for strict protection, the regulating services were the conservation features with the most decreasing proportion protected in costs scenarios (Fig. 4d-f), with the higher decreasing in the socioeconomic cost (-33.39%). As expected for the opportunity costs, all costs scenarios decreased the protection of GDP added by agriculture and population density but the social cost scenario in sustainable use areas (Fig. 4b) and the economic cost scenario in strict protection areas (Fig. 4d). For the former scenario, the GDP added by agriculture increased the proportion protected by 15.9% related to no cost scenario while for the later scenario, the population density had the increase of 29.28%.

4. Discussion

The identification of priority areas for conservation must be viewed through the existent trade-offs among conservation and development goals. Ecosystem services now have been included as conservation goals in prioritization beyond the biodiversity feature since they may not co-occur in the same areas (Balvanera et al. 2001). Opportunity costs incurred from the use of the land to achieve the conservation goals are good surrogates of the development goals, as they inclusion avoids overlapping with important economic and social areas. To achieve these opposite goals, the analysis of how conservation features and costs respond in multiple scenarios sheds light which scenario could fit better for each type of conservation strategy. We discuss how our results could support the choices for priority areas selection in the Brazilian dry forest Caatinga considering both conservation and development goals in the two main conservation strategies adopted in Brazil.

The highest perceived trade-off derived from the inclusion of opportunity costs in the prioritization is the socioeconomic gain at the expense of representation of conservation goals. Conservation planning has been developed as a win-win approach in which all stakeholders involved could benefit from conservation, however, this approach changed to hard choices based on real trade-offs involving losses even for an "optimal" choice (McShane et al. 2011). Despite the decrease in representation of ecosystem services in cost scenarios, representation of plant biodiversity did not show a significant decrease indicating that some win-win situation can indeed be achieved when costs are included in prioritization. Thus, priority sites for biodiversity conservation are not co-occurring in the same development areas and as much biodiversity could be protected in more isolated areas avoiding overlapping and pressure on new protected areas. Our result differed from that found by Duran and colleagues (2014) that analyzed multi-criterion prioritization in South America, using carbon, biodiversity and agriculture features. They showed the exclusion of agriculture lands from priority sites (negatively weighted) decreased the biodiversity representation while carbon was increased.

Ecosystem services had the highest decreasing of proportion protected with the inclusion of opportunity costs, mainly the provisioning services in areas target for sustainable use, indicating the co-occurrence of this type of services in areas of higher socioeconomic costs. Provisioning services normally have highest provision in areas with medium to high level of anthropogenic disturbance but at the same time, occur in areas with medium degree of biodiversity loss (Cimon-Morin et al. 2013; Groot et al. 2010). Then, areas with high provisioning services can be associated with socioeconomic development related to agriculture and urbanization expansion. Based on reactive strategy of conservation which prioritizes areas with high vulnerability and threats (Brooks et al. 2006), we recommend to use no cost scenario for the selection of priority areas for sustainable use. In this scenario, if 17% of priority areas were actually assigned as protected, 43.48% of provisioning services could be included inside protected areas with 24.32% overlapped with high population density areas and 16.41% with high agriculture economic value. Under this type of strategy, agri-environment schemes (AES) should be encouraged since they were related to avoid biodiversity decline (Marja et al. 2014).

The association of development goals of local people with conservation goals is more difficult to achieve in the stricter categories of protected areas (Salafsky 2011). Then, based on proactive strategy that selects priority areas with lower vulnerability (Brooks et al. 2006), the socioeconomic scenario could fit better for priority areas targeted for strict protection, avoiding future pressure of agriculture and urbanization expansion on them. Despite a lower representation of biodiversity, regulating and supporting services, the overlapping with high population density areas and economic value derived from agriculture is 4.64% and 9.03%, respectively. Regulating services and supporting services are related to be maximum in natural ecosystems with low degree of human disturbance (Cimon-Morin et al. 2013; Groot et al. 2010). Only 1% of the Brazilian dry forest is covered by strict protection protected areas and remnant vegetation must be included in this category for the maintenance of important regulating and supporting services beyond the plant biodiversity.

Most natural conditions are related to a stricter management category, but choice of categories should mostly be guided by biodiversity conservation, ecosystem services delivery, needs and beliefs of human communities, land ownership, strength of governance and population levels (Dudley 2008). Moreover, the inclusion or exclusion of agriculture lands in systematic conservation planning must be viewed through the type of management approach, reactive versus proactive (Duran et al. 2014). Multiple scenarios including different conservation costs and features have demonstrated that real trade-offs among conservation and development goals must be analysed in systematic conservation planning to avoid future conflicts among stakeholders (Carwardine et al. 2008; Di Minin et al. 2013; Dobrovolski et al. 2014; Dobrovolski et al. 2011; Faleiro and Loyola 2013; Luck et al. 2004; Moilanen et al. 2011; Naidoo and Iwamura 2007; Schneider et al. 2011).

5. Conclusions

The inclusion of socioeconomic costs in the identification of priority areas for conservation can indeed avoid overlapping areas among conservation and development goals but at expense of important ecosystem services, mainly the provisioning and regulating services. The choice to include or not opportunity costs in prioritization will depend on the strategy adopted to create new protected areas (less or more strict) that is supported by trade-offs analysis in multiple scenarios approach. Effectiveness of protected areas might be improved balancing gains and losses of conservation and development goals to attend all stakeholders involved in nature conservation. In further research, other biodiversity features such as plants, vertebrates and invertebrates should be assessed to complement the information revealed at this regional scale of the Brazilian dry forest Caatinga.

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Supporting information

Appendix A

Species distribution modeling (SDM) of all species were fitted with MaxEnt software using entire Brazilian territory as background. MaxEnt uses presence records to estimate the suitability of species occurrences based on correlations of known occurrences with the environmental variables of background landscape (Elith et al. 2011).

Presence-only records

We used the woody species occurrence records from TreeAtlan 2.0 database which is a compilation of woody species records in different vegetation types in areas of tropical and subtropical extra-Andean South America (http://www.icb.ufmg.br/treeatlan/). From this database, we extracted presence records of species that occur in the Brazilian Caatinga and estimated the potential distribution area of all species selected (769 woody species).

Environmental variables

We collected the current climatic variables (average from 1950 - 2000) and altitude (Digital Elevation Model) from WordClim database (http://www.worldclim.org/current). We also used the Brazilian map of soil types provided by Brazilian Institute of Geography and Statistic (IBGE) (ftp://geoftp.ibge.gov.br/mapas_tematicos/mapas_murais/) and the variable of height above nearest drainage (HAND) available at National Institute for Spatial Research (INPE) (http://www.dpi.inpe.br/Ambdata/). We done pairwise Pearson's correlations test among all environmental variables and we selected only variables with correlation coefficients values below |0.7|. Following this criteria, we ran MaxEnt models using eight climatic variables from WorldClim (mean diurnal range, isothermality, mean temperature of warmest quarter, precipitation of wettest quarter, precipitation of driest quarter, precipitation of warmest quarter, precipitation of coldest quarter). We also used the environmental variables of altitude, HAND and soil type (Table B). We fitted species distribution models at a 0.10° resolution. We excluded species that had less than ten occurrences and we only used 685 woody species distribution maps as conservation targets in Zonation.

Appendix B

Water balance

Water balance is based on the hypothesis that water yield can be approximated by local interaction of precipitation and potential evapotranspiration given the water storage properties of the soil (Kareiva et al. 2010). We used the water yield model from InVEST to estimate the supporting service of water balance and is defined as the annual amount of precipitation that does not evaporate and transpire (Kareiva et al. 2010).

The InVEST methodology to model the water yield can be see here: http://www.naturalcapitalproject.org/models/hydropower.html.

Water yield (*Yxj*) is calculated as following:

$$Yxj = \left(1 - \frac{AET\ xj}{Pxj}\right) .\ Pxj\ .\ Axj$$

where *AETxj* is the annual actual evapotranspiration in pixel *x* with LULC category *j*, *Px* is the annual precipitation in pixel *x* and LULC *j* and *Axj* is the area in pixel *x* and LULC *j*.

The evapotranspiration portion of water balance $\frac{AET\ xj}{Pxj}$ is an approximation of the Budyko curve developed by (Zhang et al. 2004).

$$\frac{AETxj}{Pxj} = \frac{1 + \omega xj\ .\ Rxj}{1 + \omega xj\ .\ Rxj + \frac{1}{Rxj}}$$

where *Rxj* is the Budyko dryness index (ratio of potential evapotranspiration to precipitation) in pixel *x* and LULC *j* and ωxj is a dimensionless ratio of plant accessible water storage to expected precipitation during the year.

$$Rxj = \frac{Kcj\ .ET0x}{Pxj}$$

where *Kc* is the plant evapotranspiration coefficient associated with LULC *j* and *ET0x* is the reference evapotranspiration in the pixel *x* and LULC *j* (based on alfafa).

$$\omega xj = Z\ \left(\frac{AWCx}{Pxj}\right)$$

where *AWCx* is the measure of the water content in the soil available to plants and *Z* is a parameter applied to homogeneous basin in the landscape and is calculated with calibration.

Data needs (Tallis et al. 2011) and respective sources used:

GIS raster dataset

1) Root restricting layer depth: http://www.iiasa.ac.at/Research/LUC/External-World-soil-database/HTML/ *

2) Precipitation: http://www.worldclim.org/current

3) Plant available water content: http://www.iiasa.ac.at/Research/LUC/External-World-soil-database/HTML/ *

5) Annual average reference evapotranspiration: http://csi.cgiar.org/Aridity/

6) Land use/land cover: Figure 1 in main text (MMA 2006)

* We collected the values of root restricting layer depth and plant available water content from Harmonized World Soil Database (HWSD) according to the soil class based on FAO soil classification. We used the soil map based on Brazilian soil classes map (ftp://geoftp.ibge.gov.br/mapas_tematicos/mapas_murais/) to create two raster files (root restricting layer depth and plant available water content) based on HWSD dataset.

Shapefile

7) Watersheds: http://hidroweb.ana.gov.br/HidroWeb

8) Subwatershed: http://hidroweb.ana.gov.br/HidroWeb

Data

9) Biophysical table (Table B)

9.1. Land use code: 1-16

9.2. Land use name: (1) farming, (2) water, (3) urban areas, (4) forested caatinga, (5) wooded caatinga, (6) park caatinga, (7) woody-grassy caatinga, (8) ombrophilus forest, (9) savannah, (10) seasonal forest, (11) secondary forest, (12) dunes, (13) caatinga/seasonal forest, (14) savannah/seasonal forest, (15) savannah/caatinga and (16) non-identified.

9.3. Root depth for each LULC class: (Canadell et al. 1996)

9.4. *Kc:* plant evapotranspiration coefficient for each LULC class, used to obtain potential evapotranspiration by using plant physiological characteristics to modify the reference evapotranspiration (*ET0x*), which is based on alfalfa. The evapotranspiration coefficient is thus a decimal in the range of 0 to 1.5. There is only information about *Kc* for crop species and any *Kc* value was found for LULC classes of the Caatinga. Then, we used value *Kc* = 1 (Tallis et al. 2011).

Water purification

More information about InVEST methodology to model water purification can be see here: http://www.naturalcapitalproject.org/models/water_purification.html. It estimates the quantity of pollutant (nitrogen and phosphorus) retained by each parcel of the landscape (watershed) based on annual average runoff from each parcel and the filtering capacity of each land use and land cover category (Tallis et al. 2011) .
Annual average runoff is calculated by the Adjusted Loading Value at pixel *x* (*ALVx*):

$$ALVx = HSSx \, . polx$$

where *polx* is the export coefficient at pixel x (load P and load N in Table B) and *HSSx* is the Hydrologic Sensitivity Score at pixel *x* which is calculated as:

$$HSSx = \frac{\lambda x}{\lambda w}$$

where λw is the mean runoff index in the watershed of interest and λx is the runoff index at pixel *x*, calculated using the following equation:

$$\lambda x = Log \, (\sum_{U} Yu)$$

where $\sum_{U} Yu$ is the sum of the water yield (*Yxj* in water balance model) of pixel *x* along the flow path above pixel *x.*

Data needs (Tallis et al. 2011) and respective sources used:

GIS raster dataset

1) Digital elevation model (DEM): http://www.worldclim.org/current
2) Root restricting layer depth: http://www.iiasa.ac.at/Research/LUC/External-World-soil-database/HTML/ *
3) Precipitation: http://www.worldclim.org/current
4) Plant available water content: http://www.iiasa.ac.at/Research/LUC/External-World-soil-database/HTML/ *
5) Annual average potential evapotranspiration: http://csi.cgiar.org/Aridity/
6) Land use/land cover: Figure 1 in the main text (MMA 2006)

* We collected the values of root restricting layer depth and plant available water content from Harmonized World Soil Database (HWSD) according to the soil class based on FAO soil classification. We used the soil map based on Brazilian soil classes

map (ftp://geoftp.ibge.gov.br/mapas_tematicos/mapas_murais/) to create two raster files (root restricting layer depth and plant available water content) based on HWSD dataset.

Shapefile

7) Watersheds: http://hidroweb.ana.gov.br/HidroWeb

Data

8) Biophysical table (Table B):

8.1. Land use code: 1-16

8.2. Land use name: same as water balance model

8.3. Root depth for each LULC class: (Canadell et al. 1996)

8.4. *Kc*: same as water balance model

8.5. Nutrient loading (nitrogen and phosphorus) for each LULC class (load P and load N): (Jeje 2006; Young et al. 1996).

8.6. Vegetation filtering value for each LULC class (eff. P and eff. N): ranging between 0 and 100, using expertise knowledge.

We ran two models, one for nitrogen (N) retention and other for phosphorus (P) retention. The output is the total amount of the nutrient (P or N) retained by each watershed (Kg/watershed). We standardized (z-scores) the values of each map of phosphorus and nitrogen retention estimated by watershed and summed to create only one map of water purification.

Erosion control

The InVEST methodology to model the erosion control can be see here: http://www.naturalcapitalproject.org/models/sediment_retention.html. The regulating service of erosion control is based on the ability of vegetation and soil to avoid initial nutrient and sediment loss by erosion (Kareiva et al. 2010). We estimated erosion control as the difference of potential soil erosion (RKLS) and the current soil erosion (USLE) as described by (Zhiyun et al. 2011). We calculated current soil erosion using the Universal Soil Loss Equation (USLE) derived from the sediment retention model in InVEST:

$$\text{USLE} = R \,.\, K \,.\, LS \,.\, C \,.\, P$$

R= rainfall erosivity;

K= soil erodibility;

LS= slope length-gradient factor;

C= cover management factor;

P= support practice factor.

Potential soil erosion was calculated using USLE equation but without C and P factors (RKLS) that are related to management of the land.

Data needs (Tallis et al. 2011) and respective sources used:

GIS raster dataset

1) Digital elevation model (DEM): http://www.worldclim.org/current, to calculate LS

2) Rainfall erosivity index: (Oliveira et al. 2012)

3) Soil erodibility: (da Silva et al. 2011)

4) Land use/land cover: Figure 1 in main text (MMA 2006)

Shapefile

5) Watersheds: http://hidroweb.ana.gov.br/HidroWeb.asp?TocItem=4100

Data

6) Biophysical table (Table B)

6.1. Land use code: 1-16

6.2. Land use name: same as water balance model

6.3. C factor for each LULC class: (Farinasso et al. 2010; Silva et al. 2007)

6.4. P factor for each LULC class: (Tomazoni and Guimarães 2009)

6.5. Sediment retention value for each LULC class (eff. SedRet): ranging between 0 and 100, using expertise knowledge (Table B).

Fodder

Native fodder production in the Caatinga (woody and herbaceous) is an important provisioning service to feed livestock raised freely in native vegetation. We estimated the potential fodder production using the proxy of total weight gain of livestock (sheeps, goats and cattle) raised only in the Caatinga vegetation.

GIS raster dataset

(1) Livestock density (LVD): three maps of the total number of sheeps, goats and cattle estimated per pixel (Robinson et al. 2007)

Data

(2) Weight gain of livestock: per head weight gain of sheeps, goats and cattle ($kg.ha^{-1}.year^{-1}$) in each class of the Caatinga vegetation (Filho et al. 2002) related to the LULC Caatinga classes: (4) forested caatinga, (5) wooded caatinga, (6) park caatinga, (7) woody-grassy caatinga.

We calculated the total weight gain of livestock by the sum of each type of weight gain of livestock (sheeps, goats and cattle) that was calculated by the multiplication of the per head weight gain of each type of livestock (kg.ha-1.year-1) in each class of the Caatinga vegetation by respective livestock density estimated per pixel.

Table B. Biophysical table used in InVEST to model the ecosystem services of water purification, water balance and erosion control

LULC description	LU code	root depth	Kc	load P	load N	eff. P	eff. N	C factor	P factor	eff. SedRet
Farming	1	2100	1	737	4225	25	25	21	533	40
Water	2	1	1	0	0	0	0	0	1	0
Urban area	3	1	1	160	3830	5	5	1	950	10
Forested caatinga	4	5100	1	178	2225	75	75	13	1	60
Wooded caatinga	5	7000	1	200	2500	80	80	13	1	60
Park caatinga	6	500	1	165	2063	75	75	13	1	50
Woddy-grassy caatinga	7	500	1	152	1020	40	40	13	1	40
Ombrophilus forest	8	1500	1	200	2500	90	90	1	1	70
Savannah	9	7000	1	90	1000	70	70	42	1	35
Seasonal forest	10	3700	1	200	2500	85	85	7	1	65
Secondary forest	11	600	1	165	2063	95	95	1	1	75
Dunes	12	1	1	0	0	0	0	1000	1	0
Ecotone (caatinga/seasonal forest)	13	5350	1	200	2500	82	82	10	1	62
Ecotone (savannah/seasonal forest)	14	5350	1	145	1750	77	77	24	1	62
Ecotone (savannah/caatinga)	15	7000	1	145	1750	75	75	87	1	48
Non-identified	16	1	1	1	1	1	1	1	1	1

Appendix C

Basic core-area Zonation algorithm

In this methodology, cell removal is done by calculating a removal index or minimum marginal loss of biological value (δ_i):

$$\delta_i = max_j \frac{q_{ij} \ w_j}{c_i}$$

w_j = weight of species (or ecosystem service) *j*
c_i = cost of site *i*
q_{ij} = proportion of remaining distribution of species (or ecosystem service) *j* located in cell *i* for the set of cells remaining;

For each step, the program calculates δ_i value through all cells that is the maximum biological value over all species (or ecosystem service) and the cell with lowest value is removed (Moilanen et al. 2005; Moilanen et al. 2012). When part of the distribution of species is lost, the importance of remaining habitat for that species increases thus, contributing to retain species that occurs in species-poor region and to prevent common species to be removal at early stages of running (Moilanen et al. 2005). The maximum structure of equation indicates a preference to retain location with the highest occurrence levels although, species-poor regions can be spared if they have high level of occurrence of rare species (Moilanen et al. 2012).

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