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Fernando de Moura Resende

Planejamento para conservação de serviços ecossistêmicos no Cerrado

Orientador: Prof. Rafael Loyola

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Fernando de Moura Resende

Planejamento para conservação de serviços ecossistêmicos no Cerrado

Tese apresentada à Universidade Federal de Goiás, como parte das exigências do Programa de Pós-graduação em Ecologia e Evolução para obtenção do título de doutor.

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Prof. Dr. Rafael Dias Loyola
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UFRN

Dra. Nathalia Machado e Souza
CNCFlora/RJ

Dedico esta tese aos meus pais,
pela companhia e apoio constante.

“*O real não está no início nem no fim,*

ele se mostra pra gente é no meio da travessia”.

Guimarães Rosa

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RESUMO

O conceito de serviços ecossistêmicos (SE) tem atraído grande interesse científico e político nos últimos anos. Embora este conceito tenha influenciado o discurso conservacionista, ainda é pouco compreendido se abordagens baseadas em SE também são úteis para lidar com as necessidades para conservar a biodiversidade. Além disso, ainda são necessários mais esforços para integrar SE na tomada de decisão. Para desenvolver estratégias de conservação de SE em uma região, é importante compreender a efetividade de estratégias de conservação já estabelecidas nesta região e como elas protegem os SE. Essa compreensão permite o desenvolvimento de planos de conservação futuros e de suas ações. No primeiro capítulo dessa tese, discutimos oportunidades e desafios derivados do uso de SE como uma estratégia para conservar a biodiversidade. Também apresentamos formas para que abordagens baseadas em SE sejam mais alinhadas com interesses conservacionistas. Destacamos que SE e biodiversidade devem ser vistos como estratégias complementares para incentivar a conservação. No segundo capítulo, avaliamos a efetividade das unidades de conservação e terras indígenas em representar SE e biodiversidade no Cerrado, Brasil. Mapeamos seis SE (*i.e.* produção de água, retenção de sedimentos, retenção de nutrientes, estocagem de carbono, produtividade primária líquida e provisão de alimentos silvestres) e a distribuição de espécies ameaçadas de vertebrados e plantas. Encontramos que a maioria das reservas não é efetiva para capturar SE e biodiversidade do Cerrado. Ainda, a maioria das reservas efetivas foi adequada em proteger apenas um dos seis SE. No terceiro capítulo, avaliamos o impacto de adiar ações de conservação para proteger SE no Cerrado. Geramos mapas de uso do solo para o presente, 2025 e 2050 e modelamos a provisão dos seis SE para os três períodos. Identificamos áreas prioritárias para proteger SE no presente e no futuro e avaliamos mudanças nas propriedades básicas dessas áreas prioritárias. Encontramos que as mudanças no uso do solo afetarão a provisão de SE ao longo do tempo. Além disso, as áreas prioritárias identificadas no futuro incluirão maior quantidade de ambientes alterados quando comparadas a áreas prioritárias definidas no presente. Como consequência, adiar ações de conservação aumentará os conflitos entre conservação e atividades humanas. Nosso estudo é o primeiro a fornecer informações espacialmente explícitas de múltiplos SE no Cerrado. Esperamos que nossos resultados possam orientar políticas visando estabelecer um plano de conservação efetivo para SE no Cerrado.
Palavras-chave: biodiversidade; unidades de conservação; InVEST; priorização espacial; mudanças no uso do solo; Brasil.

ABSTRACT

The concept of ecosystem services (ES) has attracted great scientific and policy interest over the last years. Although the concept has influenced the conservation discourse, it is still poorly understood if strategies focused on ES are useful to address the need for biodiversity protection. In addition, more efforts are needed to integrate ES into the decision-making process. To develop ES-focused conservation strategies in a region, it is important to understand the effectiveness of conservation strategies already established in that region and how they safeguard ES. This understanding allows the developing of future conservation plans and their actions. In the first chapter of this thesis, we discussed opportunities and challenges arising from the use of ES as a strategy to conserve biodiversity. We also presented ways to build an ES approach more aligned with conservation interests. We highlighted that ES and biodiversity should be seen as complementary strategies to foster conservation. In the second chapter, we assessed the effectiveness of protected areas and indigenous lands in representing ES and biodiversity in the Brazilian Cerrado. We mapped six ES (*i.e.* water yield, sediment retention, nutrient retention, carbon storage, net primary productivity and wild food provision) and the distribution of threatened vertebrate and plant species. We found that most reserves were not effective to capture ES and biodiversity in the Cerrado. In addition, most effective reserves were suitable for safeguarding just one out of six ES. In the third chapter, we evaluated the impact of postponing conservation actions to safeguard ES in the Cerrado. We used land use maps for the present, 2025 and 2050 and modeled the provision of the six aforementioned ES for these three time steps. We identified priority areas for safeguarding ES in the present and future and evaluated changes in basic properties of those priority areas. We found that land use changes will impact ES provision over time. Moreover, priority areas identified for the future will encompass greater amounts of altered environments when compared to priority areas defined right now. As a consequence, postponing conservation actions will increase conflicts between the implementation of conservation actions and human activities. Our study is the first to provide spatially explicit information on multiple ES in the Cerrado. We hope our results might guide policies aiming to establish an effective conservation plan focused on ES in the region.

Keywords: biodiversity; protected areas; InVEST; spatial prioritization; land use changes; Brazil.

SUMÁRIO

Introdução Geral

A atual crise ambiental desafia a humanidade a implementar estratégias de conservação eficientes que evitem maiores perdas de biodiversidade e impactos no funcionamento dos ecossistemas. Para lidar com o crescente impacto das atividades humanas sobre a natureza, iniciativas baseadas em serviços ecossistêmicos (SE) têm atraído grande interesse da comunidade cientifica e tomadores de decisão ao redor do mundo (Bouwma et al., 2018; Costanza et al., 2017). Esse interesse pode ser justificado pelo fato do conceito de SE ser útil para avaliar a relação entre homem e natureza, assim como possibilitar maior ligação entre o conhecimento científico e tomadas de decisão para conservação (Busch et al., 2012; Schroter et al., 2014).

O reconhecimento de que a natureza fornece benefícios aos seres humano remonta a antiguidade. No entanto, o termo serviços da natureza foi usado pela primeira vez na década de 1970 (Westman, 1977). Na década seguinte, o termo SE começou a ser usado em referência às contribuições da biodiversidade aos seres humanos (Ehrlich and Ehrlich, 1981; Ehrlich and Mooney, 1983). Já na década de 1990, houve um aumento do interesse acadêmico em relação aos SE, principalmente após a publicação de dois estudos. Um deles foi o livro editado por Daily (1997), que abordou questões teóricas e práticas relacionadas aos SE, além de apresentar estudos sobre os serviços fornecidos pelos principais biomas do mundo. O segundo estudo foi o artigo de Costanza et al. (1997), que estimou o valor monetário dos SE em uma escala global e demonstrou que a contribuição econômica da natureza é maior do que considerado pela economia convencional.

Após a década de 1990, o número de estudos baseados em SE cresceu exponencialmente e, atualmente, pode ser encontrada uma ampla literatura sobre o tema (Chaudhary et al., 2015; McDonough et al., 2017; Seppelt et al., 2011). Uma busca na base de dados *Web of Knowledge* revela que o número de artigos publicados com o termo *ecosystem services* aumentou 230 vezes entre 1997 e 2017 (*i.e.* 96 estudos foram publicados até 1997 e 22.166 até 2017). Além disso, revistas científicas dedicadas ao tema foram lançadas (*e.g. Ecosystem Services* e *International Journal of Biodiversity Science, Ecosystem Services & Management*).

Com o avanço dos estudos, SE foram incorporados em diversas iniciativas internacionais. Tais iniciativas contribuíram para operacionalização do conceito de SE e influenciaram a agenda política internacional (Chaudhary et al., 2015). Em 2005, por exemplo, foi lançada a Avaliação Ecossistêmica do Milênio, que avaliou o status global dos SE e propôs o primeiro sistema de classificação de SE (MEA, 2005). Esta iniciativa definiu SE como os benefícios fornecidos pelos ecossistemas que são apropriados pelos seres humanos e agrupou tais serviços em quatro categorias: serviços de provisão, serviços de regulação, serviços culturais e serviços de suporte[1]. Posteriormente, foi lançado o estudo global a Economia dos Ecossistemas e da Biodiversidade (TEEB, 2010), que forneceu bases para que o setor privado incorporasse os valores econômicos provenientes do SE e biodiversidade no processo de tomada de decisão. Em 2012, foi criada a Plataforma Intergovernamental sobre Biodiversidade e Serviços Ecossistêmicos (IPBES na sigla em inglês; Pascual et al., 2017), que representa o esforço global mais recente para avaliar o status global dos SE e biodiversidade. O IPBES propõe uma abordagem mais integrada que iniciativas anteriores, em que diversos elementos biofísicos, socioculturais e econômicos devem ser incorporados nas avaliações ecossistêmicas (Díaz et al., 2018). Metas internacionais também passaram a considerar a importância dos ecossistemas e de seus serviços para a manutenção do bem-estar humano, tais como o Plano Estratégico para a Conservação da Biodiversidade de Aichi (CBD, 2010) e a Agenda 2030 para o Desenvolvimento Sustentável (UN, 2015), ambas apoiadas pela Organização das Nações Unidas.

Apesar do intenso desenvolvimento teórico e prático do conceito de SE nas últimas décadas, alguns tópicos ainda merecem maiores esforços de pesquisa (Hossain et al., 2017). Um deles está relacionado à efetividade de abordagens baseadas em SE em conservar a biodiversidade. Apesar de SE terem entrado na agenda científica e política de conservação da natureza, ainda não é bem compreendido se investir esforços em SE também irá gerar benefícios à conservação da biodiversidade (Redford e Adams, 2009).

[1]O sistema de classificação de SE proposto pela Avaliação Ecossistêmica do Milênio é o mais comumente utilizado (Lele et al., 2014) e forneceu a base para classificações posteriores, incluindo as classificações adotadas pela Economia dos Ecossistemas e da Biodiversidade (TEEB, 2010) e pela Classificação Internacional CICES (*Common International Classification of Ecosystem Services*; Haines-Young e Potschin, 2013). Este último ainda se encontra em desenvolvimento. Veja a evolução dos principais sistemas de classificação de serviços ecossistêmicos em Costanza et al. (2017).

Está discussão volta à tona com frequência e geralmente é polarizada entre duas visões. Uma delas defende que abordagens baseadas em SE são demasiadamente antropocêntricas, dão muito importância à valoração econômica e não são adequadas para atingir objetivos de conservação da biodiversidade (McCauley, 2006). A outra visão considera que abordagens mais antropocêntricas atraem maior atenção da sociedade para a importância de se conservar a natureza e, assim, geram resultados mais práticos para a conservação (de Groot et al., 2012). No entanto, discussões sobre oportunidades e desafios em se utilizar a abordagem de SE para conservar a biodiversidade estão fragmentados na literatura. Além disso, estratégias para que abordagens baseadas em SE alcancem resultados efetivos para a conservação da biodiversidade ainda foram pouco exploradas.

Outro tópico que temos pouco conhecimento é em relação à eficácia de estratégias tradicionais de conservação em proteger os SE. Mais especificamente, não sabemos a função que as unidades de conservação desempenham na provisão destes serviços. Estudos que avaliaram a habilidade de unidades de conservação em representar eficientemente biodiversidade têm se expandido nos últimos anos (Frederico et al., 2018; Nori et al., 2015), mas trabalhos com ênfase em SE são extremamente raros. Como unidades de conservação desempenham um papel chave nos esforços de conservação da natureza (Pouzols et al., 2014; Soares-Filho et al., 2010), avaliar a efetividade delas em representar SE é útil para guiar políticas públicas e garantir o bem-estar humano.

Além disso, compreender as consequências de se adiar a implementação de ações para a conservação é outro tópico que merece atenção. Em um mundo em que a pressão por recursos naturais é cada vez maior e os fundos para a conservação são limitados (Balmford et al., 2002; Waldron et al., 2013), a implementação de ações de conservação é frequentemente adiada. Em consequência, o sucesso de ações de conservação pode ser prejudicado e coloca em risco a manutenção da biodiversidade e a provisão de SE (Cimon-Morin et al., 2016; Nori et al., 2013). No entanto, o impacto de se adiar a implementação de ações de conservação de SE tem sido pouco estudado, especialmente em regiões que sofrem com a rápida expansão de atividades agrícolas. Estudos que elucidem essas consequências podem ser úteis para indicar em que momento as ações de conservação devem ser implementadas para que seu sucesso seja maximizado.

Avaliar a efetividade das unidades de conservação em representar SE, bem como as consequências de se adiar a implementação de ações de conservação, é especialmente importante em regiões que experimentam rápidas taxas de mudanças de uso de solo ou que possuem pequenas extensões incluídas em unidades de conservação. O Cerrado combina essas e outras características, o que torna esse bioma uma região prioritária para um planejamento eficiente para a conservação de SE. O bioma é um *hotspot* de biodiversidade e representa a savana tropical com maior diversidade de espécies do mundo (Klink e Machado, 2005; Mittermeier et al., 2004). Além de sua importância biológica, milhões de pessoas dependem dos SE fornecidos pelo Cerrado, notadamente devido aos inúmeros rios que nascem na região e abastecem outras áreas do Brasil e da América do Sul (Overbeck et al., 2015; Strassburg et al., 2017). Apesar da importância biológica e socioeconômica do Cerrado, as unidades de conservação cobrem apenas 8% da área do bioma (Françoso et al., 2015) e sofrem constante pressão para serem desafetadas ou se tornarem mais permissivas ao desenvolvimento de atividades humanas em seu interior (Bernard et al., 2014). Associado a baixa abrangência das unidades de conservação na região, os ecossistemas do Cerrado têm sido convertidos rapidamente em áreas agrícolas (Strassburg et al., 2017). Assim, iniciativas que subsidiem a conservação de SE no Cerrado são urgentes, sobretudo porque ações de conservação conduzidas na região têm dado pouca atenção à manutenção da provisão de SE.

Diante disso, tivemos como objetivo nesta tese discutir sobre a efetividade do uso de abordagens baseadas em SE como estratégias para conservar a biodiversidade, avaliar a efetividade das unidades de conservação do Cerrado em representar SE e realizar um exercício de planejamento de conservação de SE no Cerrado, antevendo desafios futuros.

Para isso, organizamos o estudo em três capítulos. No primeiro deles apresentamos uma discussão conceitual sobre a eficácia de abordagens baseadas em SE em alcançar também a conservação da biodiversidade. Destacamos oportunidades e desafios derivados do uso dessa abordagem. Além disso, discutimos maneiras para construir uma abordagem de SE mais alinhada com os interesses conservacionistas.

No segundo capítulo, avaliamos a efetividade das unidades de conservação e terras indígenas existentes no Cerrado em representar SE e biodiversidade no bioma. Mapeamos seis SE (*i.e.* produção de água, retenção de sedimentos, retenção de nutrientes, estocagem de carbono, produtividade primária líquida e provisão de alimentos silvestres) e a distribuição de espécies ameaçadas de vertebrados e plantas. Testamos a efetividade das unidades de conservação e terras indígenas comparando a quantidade de SE e a biodiversidade que cada reserva representa com a quantidade que seria capturada se as reservas fossem posicionadas aleatoriamente pelo Cerrado.

No terceiro capítulo, avaliamos o impacto do adiamento da implementação de ações de conservação de SE no Cerrado. Geramos mapas de uso do solo para o presente e para dois períodos futuros (2025 e 2050), usando o modelo de uso do solo OTIMIZAGRO. Utilizamos os mapas de uso do solo e bases de dados ambientais para modelar a provisão dos seis SE nos três períodos avaliados. Identificamos áreas prioritárias para representar os SE nos três períodos e, em seguida, avaliamos possíveis alterações nas redes de áreas prioritárias que poderão ocorrer com o adiamento da implementação de ações de conservação.

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Capítulo 1

The Ecosystem Services Approach as a Strategy to Conserve Biodiversity: Challenges, Opportunities and Ways Forward

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Abstract

In recent years, the conservation community started using arguments based on ecosystem services (ES) to encourage biodiversity conservation. However, it is still not clear whether ES approach is also adequate to contemplate biodiversity conservation targets. Here, we evaluated if ES approach can be useful to achieve biodiversity conservation, highlighting opportunities and challenges derived from the use of this approach. We also discussed ways to build an ES approach more aligned with conservationist interests. The ES approach produces opportunities to biodiversity conservation that would not be possible otherwise, such as it might increase environmental awareness in the society, and encourage investments in nature conservation inside and outside protected areas. Nevertheless, using this approach to foster biodiversity conservation should be treated with caution, due to some limitations and risks, including it might reduce the interest in conserving nature when it is not profitable, and suddenly reduce the incentive to conserve biodiversity because market is known to be volatile. Aiming to conciliate better ES approach with biodiversity conservation, ES approach should explicitly consider intrinsic value of nature, expand the number of ES evaluated simultaneously, and increase the period of the analyses to capture the dynamic of the ecosystems in the long run. These efforts could better encompass the complexity of ecosystems and better integrate the ES approach with biodiversity conservation. Despite the importance of using ES for biodiversity conservation, conservation community would protect biodiversity more efficiently using both ES approach and traditional arguments, which are based on moral and ethics responsibility to safeguard nature. Rather than being considered as mutually exclusive arguments, biodiversity and ES should be seen as complementary strategies to foster conservation.

Keywords: conservation biology; conservation opportunities; intrinsic value; instrumental value; natural capital; nature’s benefits to people.

Introduction

Species and ecosystems are in peril throughout the world and efforts to avert biodiversity loss are desperately needed (Butchart et al., 2010). In the last century, conservation scientists, practitioners and policymakers allocated their efforts mainly towards the proposal and implementation of protected areas (Le Saout et al., 2013; Loucks et al., 2008). Although these areas now cover 14.6% of Earth's terrestrial area (Butchart et al., 2015), this is still not enough to protect biodiversity (Nori et al., 2015; UNEP-WCMC and IUCN, 2016; Watson et al., 2014).

Most of the global protected areas were established with a argument of the intrinsic value of nature, linked to a moral obligation of preserving species and ecosystems (Justus et al., 2009; McCauley, 2006). However, with an ever-growing human population and higher demand and pressure on natural resources (Ehrlich et al., 2012; Guo et al., 2010), it seems that the intrinsic value *per se* will not be sufficient to stop nature degradation. To overcome this issue, the concept of ecosystem services (ES; *i.e.* the benefits that people obtain from ecosystems) has attracted a lot of interest from conservation scientists, economist and policymakers (MEA, 2005; West, 2015). The emergence of this concept also influenced the conservation community discourse, that is gradually moving from moral and ethical arguments to a more anthropocentric view, stressing the contribution of nature' benefit to people and human well-being (Pascual et al., 2017; Reyers et al., 2012). The expectation is that by using a more instrumental approach, arguments in favor of nature conservation will be more persuasive, raising the awareness of the society and improving the success of conservation interventions (Lele et al., 2014). Awareness would be raised because different sectors of the society would understand the goods and services nature delivers to humans (Vira and Adams, 2009), and therefore, these sectors would stand for biodiversity conservation.

Following this idea, studies and initiatives based on ES have expanded quickly in the last decades and influenced markedly biodiversity conservation efforts (Balvanera et al., 2012; Costanza et al., 2014; MEA, 2005; TEEB, 2010). For example, the Aichi Biodiversity Targets, proposed by the UN Convention of Biological Diversity (CBD, 2010), includes targets directly related to ES (see Targets 14 and 15 in CBD 2010). Prominent conservation institutions (*e.g.* Conservation International) changed their mission to encompass explicitly the contribution of nature's benefits to people.

Ecosystem services have also influenced policy implementation in several countries (Balvanera et al., 2012; TEEB, 2010). For example, in China, a mix of policies have been implemented to increase provision of ES (*e.g.* enhance carbon sequestration, control of flood and erosion) in almost half of China's territory (Liu et al., 2008). Besides sustaining human well-being, there is also an expectancy that, as biodiversity underpins the provision of several ES (Mace et al., 2012; MEA, 2005), environmental policies aiming to conserve ES would concomitantly lead to biodiversity protection.

Albeit the importance of ES and the use of its concept, there is an intense debate if fostering the ES approach will also bring positive outcomes to traditional conservation goals, *i.e.* biodiversity *per se* (see Fig. 1, for examples). Pro arguments rely on win-win opportunities, in which both ES and biodiversity are safeguarded by conservation strategies (Adams, 2014; Reyers et al., 2012). These opportunities include management actions, such as maintenance of forest patches in agricultural areas that besides contributing to biodiversity conservation could also increase natural pollination and increase crop productivity (Hipólito et al., 2018; McCauley, 2006). Meanwhile, arguments against the ES approach claim about the commodification of nature and loss of its intrinsic value importance (McCauley, 2006; Redford and Adams, 2009). Such arguments are based on win-lose situations, in which ES are maintained or enhanced although biodiversity is not necessarily seen as a conservation goal (Adams, 2014; Reyers et al., 2012). These situations include interventions in nature aiming to increase ES that lead to negative consequences to biodiversity, such as forestry plantation to increase carbon stocks (Redford and Adams, 2009). Currently, despite the development of several initiatives and global targets based on ES, it is not clear whether the ES discourse and policies are adequate to ensure biodiversity conservation or not (see Reyers et al. 2012).

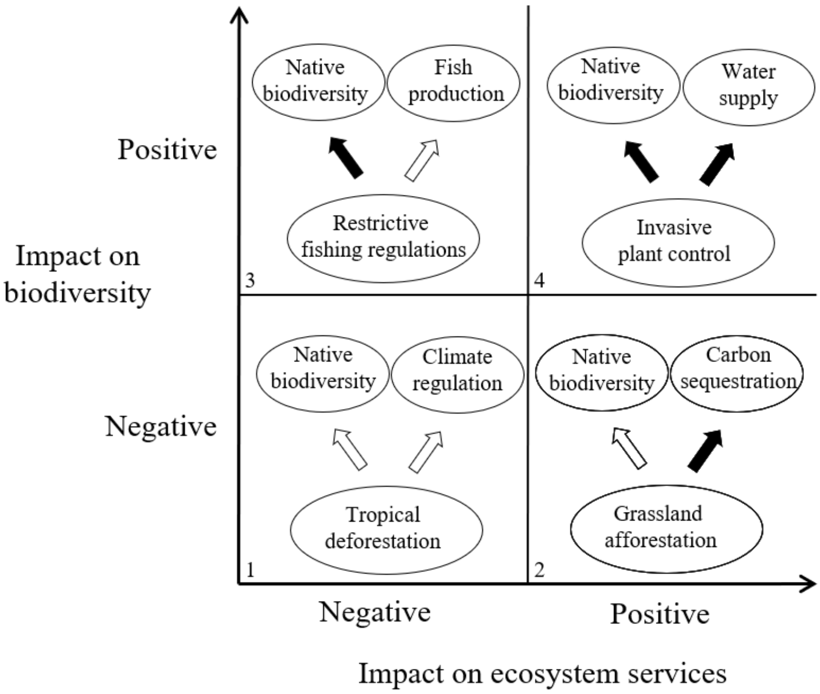

Figure 1: Impact of different drivers on biodiversity and ecosystem services. The impact of a driver on biodiversity and ecosystem services can be similar (positive or negative for both targets) or differ substantially (positive for one target and negative for other, and vice-versa). Example in quadrant 1: tropical deforestation can impact negatively both biodiversity (Barlow et al., 2016) and regional climate regulation, such as rainfall regime (Nobre, 2014). Quadrant 2: grassland afforestation can impact negatively the native biodiversity, but can increase the carbon sequestration (Veldman et al., 2014). Quadrant 3: restrictive fishing regulations can impact positively native biodiversity (Floeter et al., 2006), but might produce economic and social problems due reduction of fish production (Loring, 2016). Quadrant 4: invasive plant control can benefit both native biodiversity and water supply (Cowling et al., 1997). Black arrays indicate positive effects; white arrays indicate negative effects. Adapted from Bennett et al. (2009).

Here, we used examples from literature to evaluate the effectiveness of ES-based approach as a strategy to conserve biodiversity. We discussed biodiversity conservation opportunities and challenges derived from the use of ES as a strategy to conserve biodiversity. It was not our intention to make a complete literature revision, but the examples presented here were useful to help us to propose advances towards an ES approach more aligned with biodiversity conservation interest.

Ecosystem services approach

The ES approach encompasses several approaches. A common one is to evaluate how ES vary across a geographical area (Schagner et al., 2013) to assess synergies or conflicts between multiple ES (Casalegno et al., 2014; Naidoo et al., 2008) or the spatial congruence between biodiversity and ES (Cimon-Morin et al., 2013; Larsen et al., 2012). These studies that map ES might support policies aiming to conserve important ES provisioning areas.

There are also studies which use economic valuation technics to estimate ES flux in monetary units (Costanza et al., 2014). This latter approach may serve as guides to decision makers, because i) it is a useful tool to demonstrate that the losses caused by an intervention in nature can be bigger than the benefits, ii) it can be used to calculate negative or positive externalities of economic activities, and also iii) it can be used to evaluate the impact of certain environmental policies to different stockholders (Balmford et al., 2002; Costanza et al., 2014; de Groot et al., 2012; TEEB, 2010).

Another common approach associated with ES are market-based mechanisms, including systems of paying for ecosystem services (Balvanera et al., 2012). The general logic of paying for ecosystem services' initiatives is encouraging activities compatible with conservation that provide ES and consequently penalize unsustainable activities (Young et al., 2014).

Biodiversity conservation opportunities

Even being too soon to draw general conclusions about the contribution of the ES approach to biodiversity conservation (Adams, 2014), it is possible to point out some emerging opportunities offered by ES strategies in protecting biodiversity. In this section, we discuss four of them (see Figure 2).

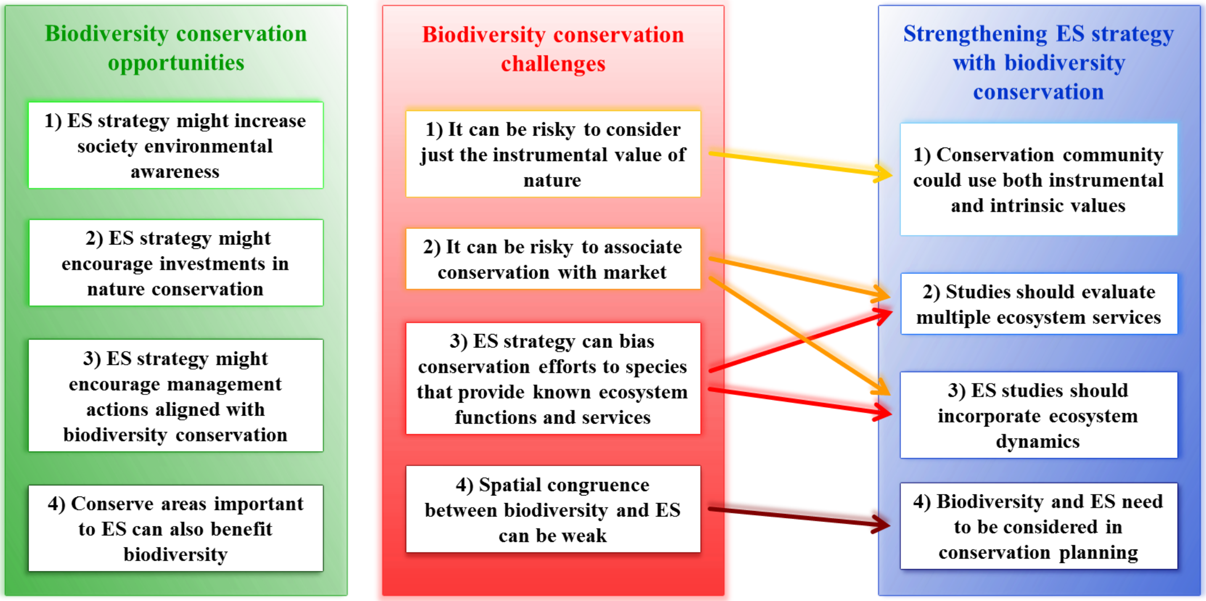

Figure 2: Overview of opportunities, challenges and ways to build an ecosystem services approach more lined up with conservationist interests. Arrows indicate the main possible strategies to deal with each challenge. ES: ecosystem services.

1) Ecosystem services approach might increase society's environmental awareness

The concept of ecosystem services can be easily understood by scientists from different fields, and even by stakeholders that have never heard the terms "ecosystem services" before (Lamarque et al., 2014). As the ES concept emphasizes how important biodiversity and ecosystems are to human well-being (MEA, 2005), ES strategy might contribute to conservation by increasing society's knowledge on the relationship between human well-being and the maintenance of natural ecosystems (Vihervaara et al., 2010).

More specifically, the ES approach could make the society aware about how human impacts, such as deforestation and habitat conversion, may affect human well-being. For example, in a recent review, Nobre (2014) showed that the Amazon forest influences the climatic system of regions located far from Amazon region, including Southern Brazil, Bolivia, Paraguay, and Argentina. In fact, high levels of deforestation in the Amazon can be associated with the recent severe drought in southern Brazil (Dobrovolski and Rattis, 2015). These results show that deforestation may directly impact water provisioning service for residential and agriculture use, as well as other economic activities that are influenced by rainfall regime. The loss of tropical forest is also associated with other negative consequences to human well-being, such as health problems due to pollution (Hewitt et al., 2009) and increase in the density of vector diseases (*e.g.* anopheline species) (Yasuoka and Levins, 2007).

2) Ecosystem services approach might encourage investments in nature conservation

As conservation funds are limited (Balmford et al., 2003; Margules and Pressey, 2000), one of the main challenges faced by conservation projects is raising enough financial funds to run its activities. Arguments based on ES might be useful to encourage private and public investments on biodiversity conservation initiatives (Balmford et al., 2002). For example, Goldman et al. (2008) showed that ES projects are able to expand opportunities for conservation, because they attract four times more financial resources and engage more investors (mainly corporate funding) than those projects based strictly on biodiversity.

The ES approach can be important to incentive investments for the maintenance of protected areas and natural ecosystems outside protected areas. For example, Medeiros et al. (2011) showed that the government investment in the National Protected Areas Systems (SNUC, in Portuguese) is considerably smaller than the contribution of the Brazilian PAs to the country's economy in terms of ES. They estimated the potential contribution of five ES provided by Brazilian PAs network, including tourism, greenhouse gas emissions avoided, and extraction of forest products. The authors argued that it would be necessary to double the investment currently made in protected areas in order to properly manage them. Moreover, ES discourse played a fundamental role to conserve the Catskill/Delaware watershed, which is responsible for purifying and providing water to the New York City’s residents. It was demonstrated that it is more profitable to maintain the region’s ecosystems than to invest in a water filtration plant (Chichilnisky and Heal, 1998; Turner and Daily, 2008).

3) Ecosystem services approach might encourage management actions aligned with biodiversity conservation

The ES approach can influence decision makers to adopt management actions with positive outcomes to biodiversity. For example, studying the mountain fynbos ecosystems in South Africa, Cowling et al. (1997) showed that a pristine ecosystem provides higher benefits to society (expressed in monetary units) than an ecosystem invaded by shrubs and trees (*Pinus* spp. and *Acacia* spp). The authors evaluated how management of invasive plants could impact the services provided by the study area, such as water provision, ecotourism and plant maintenance. They showed that the benefits derived from the eradication of invasive plants are significantly bigger than the benefits of a non-management action, even considering the costs of clearing invasive plants. Due to this work, the control of plant invasion in fynbos ecosystems became a South African government program, encouraged specially by the reduction of water supply caused by invasive plants.

Knowledge on the dynamic of ES provision could also be used to support management practices that benefit both conservation and agriculture interests. For example, it was shown that forest fragments provide pollination services that increase coffee production (De Marco and Coelho, 2004; Hipólito et al., 2018; Ricketts et al., 2004). In addition, Carvell et al. (2011) demonstrated that agri-environment schemes might boost declining pollinator populations. These findings suggest that ES studies could be used to enhance native pollination and generate positive results to the agricultural productivity and conservation (*e.g.* maintenance of forest fragments in agricultural areas).

4) Conserve areas important to ecosystem services can also benefit biodiversity

Studies evaluating spatial congruence between biodiversity and ES are increasing and have pointed out some interesting results. Generally, important areas for both biodiversity and ES can be found globally, especially in tropical region (Cimon-Morin et al., 2013). For example, Larsen et al. (2011) found that tropical forests could benefit both threatened species and carbon storage. Also in a global perspective, regulating services tend to be spatially congruent with biodiversity (Cimon-Morin et al., 2013). For example, Luck et al. (2009) found that water provision and carbon storage are positively correlated with biodiversity in a global scale. These studies suggest that investing in the conservation of areas due their importance in the provision of ES could also generate positive outcomes to biodiversity.

Challenges for biodiversity conservation

Given that the ES approach focuses on the benefits provided by nature to people, some concerns and potential limitations emerge when ES arguments are emphasized to support conservation. In this section, we discuss four aspects to which the conservation community should pay attention to in regard to ES approach (Fig. 2).

1) It can be risky to consider just the instrumental value of nature

The ES concept encompasses several types of benefits to human well-being, nonetheless some authors consider it as being just the economic benefits provided by nature (McCauley, 2006; Reid, 2006). This view, associated with the clearly anthropocentric approach of ES, can lead society and policy makers to relate the importance of biodiversity solely to economic returns, neglecting arguments not based on an economic reason, such as the intrinsic value of nature (Redford and Adams, 2009). The risk here is reducing the interest in conserving nature when and where it is not profitable.

Arguments based on the intrinsic value of nature have also influenced environmental policies and contributed to achieve some positive results. For example, it is reasonable to assume that lists of threatened species (*e.g.* IUCN Red List of Threatened Species; www.iucn.org) are built mainly based on intrinsic value arguments, because these lists are focused on species with greatest risk of extinction irrespective of the benefits that these species could provide to humans. In fact, lists of threatened species play an important role to guide conservation actions on a national and international scale (CBD, 2010; Ginsburg, 2001; Hidasi-Neto et al., 2013; Rodrigues et al., 2006). As threatened species might be protected by law, these lists could impact directly the planning of economic activities, determining, for example, areas that could not be used for mineral extraction or infrastructure expansion. Also, biodiversity *per se* is commonly used to justify the existence of protected areas (*e.g.* Ferreira and Valdujo, 2014). Therefore, the complete loss of intrinsic value appreciation could hinder conservation efforts.

2) It can be risky to associate conservation with market

As the economic market is very volatile, associating the importance of species and ecosystem conservation to monetary metrics can be risky (McCauley, 2006; Redford and Adams, 2009). Habitats or species with high economic values at a certain point of time and, therefore, with strong appeal to conservation, can dramatically lose its value over time, and thus its appeal. For example, forest patches close to pollination-dependent crops can have high economic value due to pollination service it provides, but if crops are replaced by other activity that does not depend on natural pollinators, forest patches' value may fall dramatically. This happened to forest patches located close to the coffee plantation evaluated by Ricketts et al. (2004) in Valle del General in Costa Rica (McCauley, 2006). The coffee crop was replaced by pineapple plantation some years after the study, which lead to a vertiginous reduction of the forest patches economic value.

Another risk to associate conservation with market is the fact that ES can be substituted by technologies in some level (Moberg and Ronnback, 2003). If a certain technology is an adequate substitute, the ES monetary value may also fall giddily and reduce the appeal for conservation. For example, if the arguments to conserve an area are based on its clean water provision, the argument for maintaining this area will decrease considerably if the purification by a water treatment station becomes more profitable. Pollination by native insects could be replaced by exotic/cultured pollinators. In fact, evaluating the importance of wild pollination service around the world, Kleijn et al. (2015) found that the contribution of managed bees to crop production (mean ± s.e. = \$3,251 ha $^{-1}$ ± 547) is similar to the service provided by wild bees (\$2,913 ha $^{-1}$ ± 574).

3) Ecosystem services approach can bias conservation efforts to species that provide known ecosystem functions and services

Ecosystem services approach can emphasize the importance to conserve species solely due to the importance they play in the ecosystem functioning or in the provision of ES. It can be easy to demonstrate and estimate the monetary value of species contribution to some ES (*e.g.* carbon storage, pollination, provision of food), but not for all of them (*e.g.* nutrient cycling, soil formation). Therefore, those species with known and more importantly quantified functions or services would be prioritized in conservation efforts based on ES, while species with limited or unknown roles to the ecosystem functioning could be neglected. It has been suggested that species with small populations (as many endemic and rare species) can be less important to the ecosystems' functioning than those with high density (Cardinale et al., 2006). As a result of this approach based solely on ES, the endemic and rare species could reduce conservation appeal.

4) Spatial congruence between biodiversity and ecosystem services can be weak

Although it is possible to find important areas for both biodiversity and ES (as we argued previously), spatial congruence between these targets is not the most common pattern found. For this reason, there are several situations in which the prioritization of one target (biodiversity or ES) is maximized at the expense of another. The spatial congruence between biodiversity and ES seems to be lower in continental or national scale, and especially provisioning services tend not to be congruent with biodiversity (Cimon-Morin et al., 2013). For example, Chan et al. (2011) showed that timber production is not correlated with biodiversity in British Columbia, Canada. Schneiders et al. (Schneiders et al., 2012) showed that food production is negatively correlated with biodiversity in Flanders, Belgium. Also, low (or negative) correlations are found when evaluating ES from other categories. Holland et al. (2011), for instance, found negative correlation between recreation service and biodiversity across England and Wale. These evidences reinforce the argument that efforts to conserve biodiversity cannot rely solely in places with greatest ES, being risky leave important areas to biodiversity conservation out of conservation strategies.

Strengthening ecosystem services approach with biodiversity conservation

Part of the shortcomings or apparently inconsistences of using the ES approach to deal with conservation is related to the approach traditionally used. In this section, we discuss how these two strategies are being (or could be) better integrated (Fig. 2).

1) Conservation community could use both instrumental and intrinsic values

Instrumental value may be useful to support decision making process (Justus et al., 2009; Maguire and Justus, 2008) but still, could remain compatible with the use of intrinsic value in biodiversity conservation efforts (Tallis and Lubchenco, 2014). As mentioned above, there are also positive results from the use of intrinsic value. Conservation community could use arguments based on both values to achieve better conservation outcomes. Which argument is more adequate will depend on the context and stockholders involved. People can be motivated to conserve ecosystems for several reasons, and may be touched by their heart, brain or wallet (Costanza, 2006; McCauley, 2006).

2) Studies should evaluate multiple ecosystem services

Ecosystem services studies generally focus on few ES (Seppelt et al., 2011), tending not to reflect adequately the complexity of ecosystem processes and producing unsteady arguments to conservation. Including multiple ES in the analysis could create more robust arguments less subjected to market's fluctuations. For example, the conversion of forest patches located in agricultural regions might negatively impact pollination services, but also several other ES (*e.g.* pest control, water regulation, carbon storage, ecotourism opportunities). If an ES study considers those multiple services, arguments to maintain forest patches would be less vulnerable to market oscillation (*e.g.* switch of agriculture crops in the region) than using arguments based solely on pollination services.

Considering multiple ES would also reduce problems related to the argument that humans are not highly dependent on nature because technologies could substitute ES (McCauley, 2006). For example, retaining walls could contribute to protect human settlements, being a potential substitute for coastal ecosystems to prevent flooding. However, artificial barriers do not substitute other services associated with coastal ecosystems, such as storm protection, carbon storage, plant and animal products and biodiversity nursery (Ewel et al., 1998). In fact, substituting ES can be difficult or even impossible, as showed by the Biosphere II project, which tried to replicate the natural biosphere in a greenhouse within the Arizona desert (Cohen and Tilman, 1996). Even though this was a millionaire program, which replicated miniatures of several natural ecosystems, inserting eight people inside the greenhouse, the project was cancelled because the artificial ecosystems could not sustain these people with material and physical need for them to survive during the project lifespan. The reasons were that the functioning of the artificial ecosystems collapsed.

3) Ecosystem services studies should incorporate ecosystem dynamics

Ecosystem services studies generally consider a short period of time for their analyses (*e.g.* Adams et al., 2008; Peixer et al., 2011). Nevertheless, considering the ecosystem dynamics over time would be more adequate to capture the complexity of ecosystems and to produce stronger outcomes to conservation. One of the reasons to encourage long-term studies is that the reduction of some ES can be detected solely in a long run.

For example, fish stocks may keep producing considerable quantities of fish despite being overexploited, due to the capture of juveniles (MEA, 2005).

Moreover, the importance of ES tends to increase when ecosystem dynamics are considered. In this case, benefits to human well-being might overcome the costs associated to conservation actions. For example, evaluating economic consequences of deforestation versus conservation over 30-year in Leuser National Park in Indonesia, Van Beukering et al. (2003) found that benefits generated by conservation surpassed benefits from deforestation and selective use scenarios after 10 years of analysis. In this study, using a long-term evaluation was important to foster the conservation of the region and demonstrate that conserve nature is profitable.

Finally, evaluating the ecosystem dynamics over time might be necessary to understand the ecosystem functions performed by some species, which would not be perceived in a short period analysis. For example, Isbell et al. (2011) demonstrated that higher plant diversity is necessary to maintain ES when more years are considered, because some species provide ES irregularly during the timeline. They showed that different species are important to maintain ES fluxes if different contexts are considered, including time, space and ecosystem functions.

4) Biodiversity and ecosystem services need to be considered in conservation planning

Several factors can influence the spatial congruence between biodiversity and ES, including the scale of the analysis, ES category analyzed (*e.g.* provision or regulation services), region of the world (*e.g.* tropical or temperate) and biodiversity metrics considered (*e.g.* species richness or functional diversity; Cimon-Morin et al. 2013). In situations where areas important to biodiversity and ES are not spatially congruent, planners could design priority areas that embrace those different targets (Chan et al., 2006; JéRome Cimon-Morin et al., 2016; Manhães et al., 2018). It is possible to maximize both biodiversity and ES and define complementary sites using systematic conservation planning. This approach allows planners to build different scenarios and give different weights to biodiversity or to some specific ES, depending on the study's objective (Manhães et al., 2018; Moilanen et al., 2011) .

Concluding remarks

The ES approach produce arguments and opportunities to conserve biodiversity that would not be possible to achieve through a discourse based uniquely on biodiversity. For this reason, ES is a good approach to deal with biodiversity conservation and should be used by conservation community. Nevertheless, the use of ES as a strategy to biodiversity conservation should be treated with caution, due to some limitations and risks. Efforts to overcome the ES approach limitations and to conciliate it with biodiversity conservation agenda should be made acknowledging that arguments based on intrinsic and instrumental value are not mutually exclusive. Also, ES studies should better encompass the complexity of ecosystems, including in the analyses multiple ES and ecosystems dynamics overtime.

To incorporate suggestions presented here, and to build an ES approach more aligned with biodiversity targets, a high monetary investment is needed in order to quantify ES fluxes worldwide in an adequate resolution. Due to the multidisciplinary nature of those challenges, the collaboration between scientists from different areas would be required to integrate multiple ES and values in the analyses and also to build adequate models to evaluate the dynamics of ecosystem in a long term. Current investments in ES database (*e.g.* Ecosystem Service Value Database; de Groot (2012)) and encouragement of formation of multidisciplinary teams would certainly contribute to the challenges imposed by the use of the ES approach to biodiversity conservation.

In addition, conservation community should pursue an integrated conservation in which different views should be embraced to a common objective (Tallis and Lubchenco, 2014). We need to be careful because the discussion between supporters and opponents of the ES approach may produce unwanted effects to conservation efforts, including weakening of the conservation community, and delay conservation policies development.

Despite the importance of the ES approach for biodiversity conservation, the environmental crisis is too complex and it is unlikely that a unique strategy will be able to deliver the solution. Meanwhile a more integrative ES approach should be fostered, by including the suggestions made in the present study, we also need strong institutions concerned about biodiversity, and funds directly designated to conserve biodiversity *per* *se*. Rather than being considered as exclusive arguments, biodiversity and ES should be seen as complementary strategies to foster conservation.

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Capítulo 2

Effectiveness of Protected Areas and Indigenous Lands in Providing Ecosystem Services in the Cerrado Biodiversity Hotspot

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Abstract

Evaluating the effectiveness of protected areas (PAs) to deliver ecosystem services is mandatory in a developing world where resources dedicated to conservation are scarce, especially in regions in which human activities could threats nature's benefits to millions of people. Here, we evaluated the effectiveness of PAs and indigenous lands (ILs) in representing ecosystem services and biodiversity, using as case study the Cerrado Biodiversity Hotspot, in Brazil. We mapped six ecosystem services (*i.e.* water yield, sediment and nutrient retention, carbon storage, net primary productivity and wild food provision) and the distribution of threatened vertebrates and plants species. We tested the effectiveness of each PA and IL by comparing the amount of ecosystem services and biodiversity they hold with those captured by random positions across the Cerrado, using a null model that maintained the same size, form and orientation as the corresponding PAs and ILs. We found that a small amount of PAs and ILs were more effective than randomly selected positions to capture ecosystem services or biodiversity. Yet, the majority of effective reserves are suitable in representing just one out of six ecosystem services. Moreover, the entire network of PAs and ILs captured a relatively small proportion of the services provided by the biome. Only biodiversity was relatively well captured by PAs, which represented on average 23.8% of the distribution range of threatened species. Considering that the current configuration of the network of PAs and ILs in the Cerrado poorly represent ecosystem services, explicitly planning for the representation of ecosystem services within government and private lands should be priority in future conservation initiatives for the region.

Keywords: conservation policy; ecosystem-based adaptation; nature's benefits to people; natural capital; threatened species; ecological services; Brazil.

Introduction

Currently, the global network of protected areas (PAs) covers about 14.6% of the Earth's terrestrial surface (Butchart et al., 2015) and substantial efforts and funds are continuously invested to maintain and increase this number. Being one cornerstone of conservation strategies, the global annual expenditure in the management of established PAs is estimated to US$6.5 - US$10 billion (Gutman and Davidson, 2007). However, PAs have traditionally been established with opportunism, notably toward sites with low commercial or high scenic value (Pressey and Tully, 1994). As *ad hoc* approaches to reservation persist (Baldi et al., 2017), there is an increasing interest to quantify the effectiveness of PAs for the protection of conservation features (Bertzky et al., 2012; Butchart et al., 2015; Nori et al., 2015).

Assessments of PAs' contribution have mainly focused on the safeguard of biodiversity (Jenkins et al., 2015; Nori and Loyola, 2015), being rare studies that evaluate the effectiveness of PAs to represent ecosystem services. The scarce literature challenge the effectiveness of PAs to represent plant productivity, carbon storage and soil retention (Durán et al., 2013; Xu et al., 2017). Understanding the capacity of PAs to provide a wide array of ecosystem services is critical as most of the benefits provided by nature have been at risk due to intensification of agriculture or other human-induced pressures (Foley et al., 2005). Further, knowledge on how PAs represent ecosystem services is important to guide conservation strategies aiming to secure natural capital (Ehrlich et al., 2012), especially for regions which faces intense pressure by human activities but sustain livelihood of millions of people.

Non-forest ecosystems have experienced unprecedented loss of native vegetation and associated ecosystem services worldwide (Overbeck et al., 2015). A remarkable example is the Brazilian savanna, known as Cerrado, which is a world's Biodiversity Hotspots (Mittermeier et al., 2004) and one of the most threatened biome in the world (Strassburg et al., 2017). Due to the expansion of agriculture and cattle raising activities, more than 15,000 km^2 were converted annually between 2000 and 2010 (MMA, 2014a), leaving less than 55% of the native vegetation cover in place (MMA, 2015). In addition, only 6.5% of the remaining native vegetation is included in network of PAs (Françoso et al., 2015). This raises important issues for the protection of conservation features in the region (Strassburg et al., 2017; Vieira et al., 2017).

Although poorly understood, loss of vegetation is impacting ecosystem services in the Cerrado, including reduction of biomass carbon storage and impairment of water quality (Hunke et al., 2015; Vieira et al., 2017). Nonetheless, a large number of people depend on the Cerrado's ecosystem. About 170 millions of inhabitants considering its surrounds live in the region, ~ 25 millions of which are engaged in extraction of row material or farming of low intensity (Sawyer et al., 2016). Rural population also includes traditional people, such as indigenous people, which are spread throughout the Cerrado region and rely on goods and services provided by nature (Mistry et al., 2005; MMA, 2014a).

To guarantee land possession for indigenous peoples, Brazilian government has recognized 4.8% of the Cerrado's area as indigenous lands (ILs). As ILs tend to be distant from urban centers and possibly biased to regions with less potential for economic land uses, extensive portions of these reserves are covered by native vegetation. Together with the network of PAs, ILs has played an important role to reduce habitat conversion in the biome (Carranza et al., 2014). Nonetheless, it is still unknown whether the network of PAs and ILs are effective to represent and maintain ecosystem services in the Cerrado. Here, we assessed the effectiveness of the PAs and ILs located in the Cerrado in representing six ecosystem services and biodiversity (*i.e.* threatened species). We also estimated the proportion of ecosystem services available in the Cerrado region that are included in the current network of PAs and ILs. Our study was intended to help identifying potential gaps in the strategy of nature conservation, as well guiding future conservation planning in the region.

Methods

Ecosystem services

We selected six ecosystem services provided by the Cerrado based on their importance and feasibility to quantify and map them in a good spatial resolution. These services are distributed between three of the four categories recognized by the Millennium Ecosystem Assessment (2005): provisioning (wild food provision and water yield), regulating (carbon storage, sediment and nutrient retention), and supporting (net primary productivity) services. The selected ecosystem services benefit people at different spatial scales. Wild food provided by native edible plants has predominantly local importance and contribute to the maintenance of local community's livelihood.

Water yielded in the Cerrado goes far beyond its limits and reach other regions of Brazil and other countries in South America (Strassburg et al., 2017). Sediment and nutrient retention are related to the capacity of the landscape to retain sediments and maintain the fertility of the soil in the Cerrado, but also to maintain the water quality of rivers important to other regions. Carbon storage has a globally significance and contributes to regulate climate at global scale. Finally, net primary productivity is an indicator of other important ecosystem services to local people, such as fuel wood provision, and to global community, such as the carbon sequestration (Balvanera et al., 2006; MEA, 2005).

We quantified the supply of the selected ecosystem services using biophysical units. To characterize and map water yield and sediment and nutrient retention services, we used the InVEST software v.3.3.3 (Integrated Valuation of Ecosystem Services and Tradeoffs), which is a well-recognized tool for modeling and mapping multiple ecosystem services (Kareiva et al., 2011). To map carbon storage, we used biomass and soil carbon stocks, while to map net primary productivity we used data derived from MODIS sensor. Wild food provision and biodiversity were mapped using species distribution range of wild edible plants and threatened vertebrates and plants, respectively. We associated each ecosystem service and biodiversity database to an equal-area grid of 0.1º latitude/longitude (~ 11x11 km, or 121 km^2, near the equator), totalizing 18,242 grid cells for the whole Cerrado.

Water yield. We mapped the distribution of water yield service in the Cerrado using the InVEST Water Yield model (Sharp et al., 2016), which considers data on precipitations, evapotranspiration, land-use and soil characteristics to calculate the contribution of each part of the landscape to the annual water yield of the study area. The model estimates the water yield as the amount of water from precipitations that is not lost by evapotranspiration and flows toward downstream areas.

We mapped 21 land-use covers for the study region (~ 1x1 km; Table S1). We first used the 10 land-use classes defined from TerraClass Cerrado 2013 (~ 1x1 km; www.dpi.inpe.br/tccerrado MMA, 2015): remnants of native vegetation, annual and perennial agriculture, silvicultural fields, pasturelands, water bodies, urban areas, mining pits, bare lands and non-identified lands. We further classified remnants of native vegetation into nine vegetation types (IBGE, 2012) and defined the distribution of the vegetation types using the map developed by MCTi (2010). We also mapped areas with soybean, sugar cane and corn using spatial data from Otimizagro 2013 (Soares-Filho et al., 2016). These three crops are the most extensive cultures, corresponding to 70% of the agricultural area in the Cerrado.

We used data on precipitations from WorldClim dataset (~ 1x1 km; www.worldclim.org/current), average reference evapotranspiration (1950-2000) from CGIAR-CSI (~ 1x1 km; csi.cgiar.org/Aridity/) and watersheds limits from National Water Agency of Brazil (ANA) (www.ana.gov.br). Following Manhães et al. (2016), we obtained values of root restricting layer depth (*i.e.* soil depth in which root penetration is inhibited by soil characteristics) to each soil type consulting Harmonized World Soil Database (HWSD; Hiederer and Köchy, 2012) and associated these values to the Brazil soil map (IBGE, 2001). We used the same approach to define values of plant available water content (*i.e.* the proportion of water stored in the soil profile that is available for plants use). To obtain the root depth of each land-use class (*i.e.* where 95% of the root biomass occur for each land-use class), we used values from literature and checked them consulting specialists (Table S1). Using the output of the water yield model, we calculated the value of water supply of each grid cell in mm cell^{-1}.

Sediment retention. We mapped the sediment retention service using the InVEST Sediment Delivery Ratio model (Sharp et al., 2016). The model calculates the amount of sediment delivered to the water resource and retained at each pixel due land cover, climatic and topographic characteristics. Thereby, the sediment retention service represents the capacity of the land cover to retain sediments which otherwise could be transported to downstream areas. The amount of soil loss by each cell ("A"), in ton ha^{-1} yr^{-1}, is calculated by the revised universal soil loss equation:

$$A = \mathrm{R} \cdot \mathrm{K} \cdot \mathrm{LS} \cdot \mathrm{C} \cdot \mathrm{P}$$

in which "R" represents the rainfall erosivity in MJ mm (ha hr)$^{-1}$, "K" is the soil erodibility in ton ha hr (MJ ha mm)$^{-1}$, "LS" is the slope length-gradient factor, "C" is the crop-management factor and "P" is the support practice factor.

We mapped the rainfall erosivity (*i.e.* the potential of the rainfall erodes the soil without protection) using the multiple linear regression developed by Mello et al. (2013). These authors developed equations to different regions of Brazil that allow one calculates the rainfall erosivity of any location in the country using latitude, longitude and altitude as predictors. To map the soil erodibility (*i.e.* the inherent erodibility of the soil, which is influenced by the proportion of gravel, organic matter and equivalent moisture of the soil), we obtained values for each soil type found in Cerrado (IBGE, 2001) using the literature review performed by Da Silva et al. (2011). The slope length-gradient factor is computed by InVEST using the digital elevation model provided as input by the user; that factor is dimensionless and higher values are associated with steep terrains. We obtained the digital elevation model from WorldClim dataset (~ 1x1 km; www.worldclim.org/current). We got values for both crop-management and support practice factors to each land-use cover revising the literature (see Supplementary Material). Both factors vary from 0 to 1, where land covers with values close to 1 represent land-use classes with high soil loss rate.

The InVEST assesses the sediment retention as the difference between the soil loss calculated from the input data compared to a hypothetical situation in which the watershed would be covered by bare soil (Sharp et al., 2016). We used the same land-use map considered in the water yield model (*i.e.* 21 classes of land-uses; Table S1). Using the output of the sediment retention model, we calculated the value of sediment retention in t cell^{-1}.

Nutrient retention. We used the InVEST Nutrient Delivery Ratio model to map nutrient retention service. The model calculates the amount of nutrient produced by each portion of the landscape that is transported to the streams and the capacity of vegetation and soil to retain nutrient (Sharp et al., 2016). The length of the flow, average slope gradient and retention efficiency of the downslope path influence the amount of nutrients that reaches the streams.

We obtained nutrient load and retention efficiency of each land-use class consulting literature (Table S1). We used the same land-use map (*i.e.* 21 classes of land-uses; Table S1), digital elevation model, annual precipitation and watershed limits considered before. Due the lack of knowledge in the study area, we followed the InVEST guidelines and assumed that all nutrient flows via surface only (Sharp et al., 2016). We ran models for phosphorus and nitrogen and mapped the nutrient retention service averaging retention values of both nutrients. We represented nutrient retention by an index that varies from 0 to 1, where grid cells with values close to one are more efficient to retain nutrients.

Carbon storage. We estimated carbon storage from above- and belowground biomass and soil organic carbon in the Cerrado. For above- and belowground carbon storage, we considered the aforementioned land-use map (*i.e.* 21 classes of land-uses; Table S1). However, for native vegetation of Cerrado there is detailed information about carbon storage. Thereby, instead of using nine vegetation types (see Table S1), we determined above- and belowground carbon storage for 28 vegetation types according to the database of MCTi (2010). We also took above- and belowground carbon storage values related to other land-use classes from the database of the MCTi (2010). We set different values for silvicultural fields located in each Brazilian state considering the proportional area of *Pinus* and *Eucaliptus* plantations areas per state. For simplicity, we assumed that above- and belowground carbon storage of water bodies, urban areas and mining was equal to zero.

We retrieved soil organic carbon data from the HWSD (~ 1x1 km; Hiederer and Köchy, 2012). We summed the carbon stock in above- and belowground biomass and soil organic carbon and obtained the total carbon stored in tC cell^{-1}.

Net primary productivity. We used the net primary productivity data derived from MOD17 algorithm design for the MODIS sensor (www.ntsg.umt.edu/project/mod17). MOD17 algorithm uses absorbed Photosynthetically Active Radiation and a conversion efficiency parameter (which varies among vegetation types and climate conditions) to calculate the gross primary productivity of land surface. The net primary productivity is then calculated subtracting respiration losses, assessed as daily leaf and fine root maintenance respiration, annual growth respiration and annual maintenance respiration of live cells in woody tissue (see Running, 2004; Running and Zhao, 2015) (see more details in Running, 2004; see Running and Zhao, 2015). We used the data processed by LAPIG-UFG (www.lapig.iesa.ufg.br/lapig), which represents the annual average of the net primary productivity between 2000 and 2012 (~ 1x1 km). We calculated the net primary productivity in tC cell^{-1}.

Wild food provision. We estimated wild food provision from the distribution of 16 wild edible plant species found in the Cerrado (see Vieira et al., 2006), which are of significant importance to food security and income generation (Table 1). We used occurrence records compiled from online databases (*e.g.* JABOT – Banco de dados da Flora Brasileira and Species Link) and published by Oliveira et al. (2015). We organized the occurrence records in a presence/absence matrix and then calculated the proportion of the distribution area of each species that fall within each grid cell. To estimate the ecosystem service, we calculated the mean proportion of the 16 species distribution that fall within each cell. We assumed that cells with higher mean proportion of species distribution area are more important to the supply of wild food.

Table 1: Wild edible plants used in this study and their respective growth form, traditional use and commercial importance. Based on Vieira et al. (2006).

Scientific name	Common name	Growth form	Traditional use importance	Commercial importance
Anacardium othonianum Rizzini	Caju	Tree	High	High
Ananas ananassoides (Baker) L.B.Sm.	Abacaxi do cerrado	Herb	Low	Low
Annona coriacea Mart.	Araticum	Tree	Medium	Medium
Butia capitata (Mart.) Becc.	Coquinho	Palm tree	High	Medium
Byrsonima verbascifolia (L.) DC.	Murici	Tree	Medium	Medium
Campomanesia adamantium (Cambess.) O.Berg	Gabiroba	Shrub	High	Medium
Caryocar brasiliense Cambess.	Pequi	Tree	High	High
Dipteryx alata Vogel	Baru	Tree	Medium	High
Eugenia dysenterica (Mart.) DC.	Cagaita	Tree	Low	Low
Eugenia klotzschiana O.Berg	Pêra do cerrado	Tree	Low	Low
Genipa americana L.	Jenipapo	Tree	Low	Medium
Hancornia speciosa Gomes	Mangaba	Tree	Medium	High
Hymenaea stigonocarpa Mart. ex Hayne	Jatobá	Tree	Low	Low
Mauritia flexuosa L.f.	Buriti	Palm tree	High	Low
Passiflora setácea DC.	Maracujá do cerrado	Climber	Low	Medium
Psidium guianeense Sw.	Araçá	Shrub	Low	Medium

Biodiversity

We considered mammals, birds, amphibians and plants listed as threatened by the Brazilian Ministry of Environment (MMA, 2014b). To map the distribution of threatened vertebrates (totaling 130 species), we used species distribution range obtained from the IUCN Red List of Threatened Species (www.iucnredlist.org/). For threatened plants (totaling 748 species), we used species distribution range provided by the Brazilian National Center for the Conservation of Flora – CNCFlora (Martinelli and Moraes, 2013), which is the Red List Authority for plant species in Brazil. We used the species range to make a presence/absence matrix and to calculate the proportion of the distribution of each species that fall within each grid cell. We associated to our grid the mean proportion of all threatened species distribution that fall within each cell.

Protected areas and indigenous lands

The limits of federal PAs were set according to the MMA database (mapas.mma.gov.br/i3geo/), and those of the state and municipal PAs according to the Brazilian Electricity Regulatory Agency (ANNEL; www.aneel.gov.br) (Fig. 1). We considered two categories of PAs as defined by the SNUC (Portuguese acronym for National System for Protected Areas, which is the national legislation responsible to define and orient the norms to the establishment and management of PAs in Brazil; MMA, 2000): strict protection and sustainable use. PAs of strict protection correspond to IUCN categories I to III and aim the conservation of biodiversity and natural assets, while PAs of sustainable use correspond to IUCN categories IV to VI and are intended to discipline human occupation and use of natural resources in a sustainable way (Rylands and Brandon, 2005). ILs were considered apart from PAs because they are not recognized officially by the SNUC and have been established for different goals. ILs limits were set according to the National Indian Foundation (FUNAI) database (www.funai.gov.br/index.php/shape).

To associate PAs and ILs to our grid, we only considered the grid cells that were covered by more than 55% of PAs and more than 30% of ILs (Ribeiro et al., 2016). These criteria led to similar total area covered by PAs and ILs network in our grid cells compared to the actual extent of PAs and ILs in the study region (~ 7.1% and 4.8% of the Cerrado’s area, respectively), while avoiding overestimating the area occupied by small PAs and ILs in the region. It follows that we maintained 109 PAs (occupying 1,344 grid cells) and 61 ILs (942 grid cells) in our analyses.

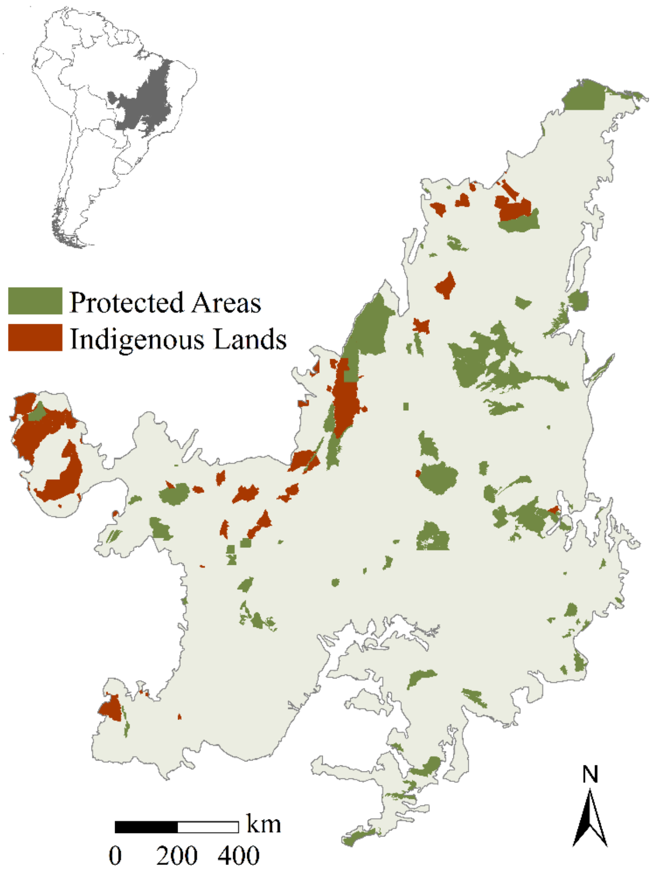

Figure 1: Location of the Brazilian Cerrado and the distribution of protected areas and indigenous lands used in the analysis. Reserves are represented as polygons.

Analyses

To evaluate if PAs and ILs are effective to capture conservation features (*i.e.* ecosystem services and biodiversity) in the Cerrado, we compared the amount of conservation features held within the limits of actual reserves (*i.e.* currently established PAs and ILs) with the amount of these features represented by random positions across the biome (Ferro et al., 2014; Lemes et al., 2014). For this, we ran a null model that defined 999 random positions for each actual reserve. These random position (hereafter referred to as randomly located reserves) maintained the same size, form and orientation as the corresponding actual reserve.

For water yield, sediment retention, carbon storage and net primary productivity, we summed the amount of ecosystem services found inside both actual and randomly located reserves. We assessed nutrient retention of each reserve by calculating the median of the index values of the grid cells associated to actual and randomly located reserves. For wild food provision and biodiversity, we averaged the proportion of the species distribution that fall within grid cells associated to actual and randomly located reserves.

We evaluated the effectiveness of actual reserves in representing ecosystem services and biodiversity considering one conservation feature at a time. An actual reserve was considered effective when the amount of a given conservation feature observed inside its limits (obs) was higher than the amount of this feature found inside the randomly located reserve ($aleat$) in at least 95% ($p < 0.05$) of the randomization runs (n), as follow (Ribeiro et al., 2016).

$$p = \sum_{i=1}^{n} \frac{(aleat \geq obs) + 1}{n + 1}$$

To avoid spatial overlapping between actual and randomly located reserves, we restricted the randomization to areas that do not hold established reserves (see Ribeiro et al., 2016). To this end, we removed areas occupied by actual PAs from the PA's null model or actual ILs from IL's null model.

We also calculated the proportion of conservation features available in the whole Cerrado region that is included in actual PAs and ILs networks. For water yield, sediment retention, carbon storage and net primary productivity, we divided the amount of each ecosystem service found inside PAs or ILs by the amount of these services supplied by the whole Cerrado region. For wild food provision and biodiversity, we calculated the mean proportion of species distribution range that is represented by PAs or ILs networks. As nutrient retention was mapped as an index, it was not possible to calculate the amount of that service inside reserves; thereafter we used solely the proportion of effective reserves to discuss the representativeness of this service. We mapped ecosystem services in R environment (R Core Team, 2016) and ArcGis 10.1 (ESRI, Redlands, CA, USA). Statistical analyses were made in R.

Results

The ecosystem services evaluated in this study had a heterogeneous distribution in the Cerrado (Fig. 2). Important areas to the provision of water yield, sediment retention and wild food were found mainly in the central region of the biome (Fig. 2A, B and F) while the nutrient retention was higher in the northern and west portions (Fig. 2C). Areas with high levels of carbon storage and net primary productivity were located mainly in the west region (Fig. 2D and E), although the latter was also higher in the south portion. Finally, the south and central portions of the biome contain higher proportion of threatened species (Fig. 2G).

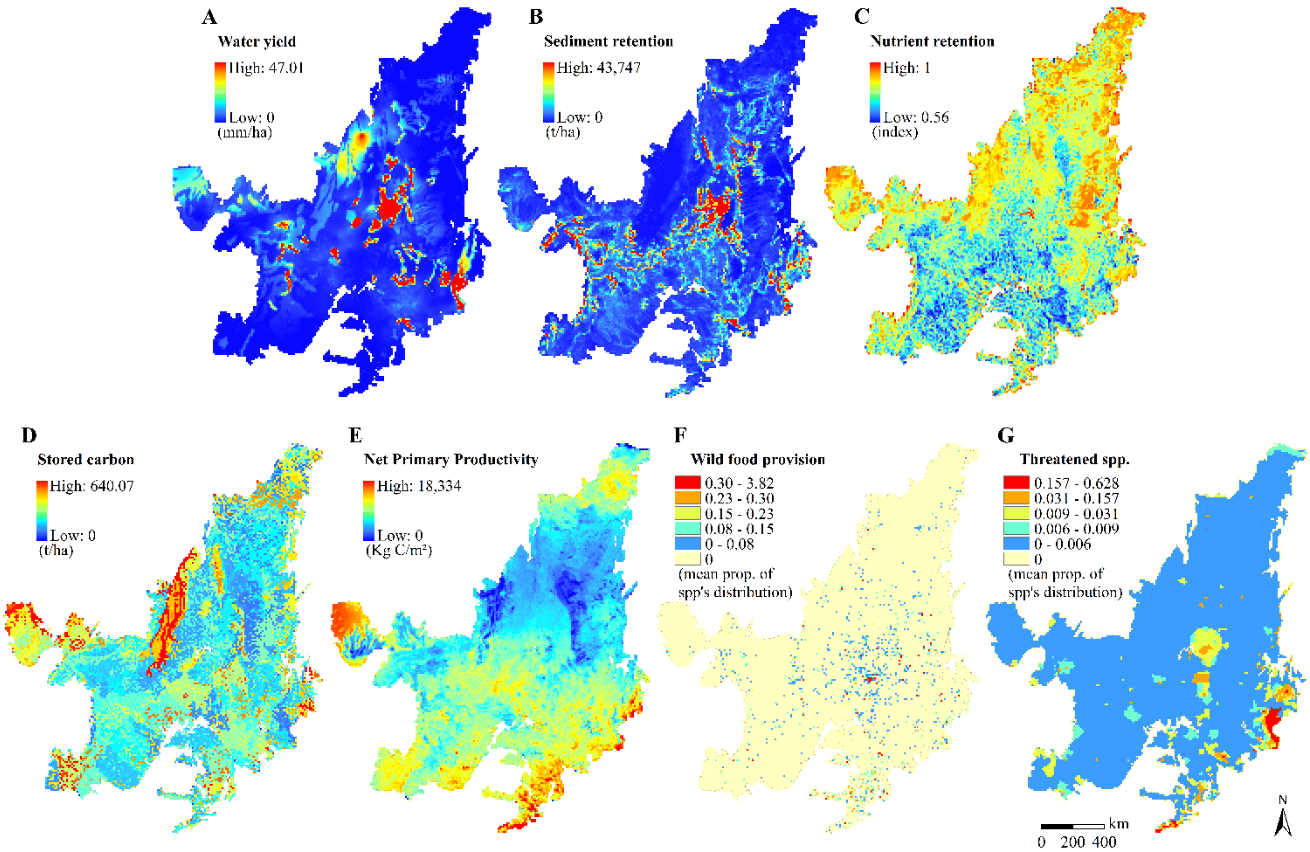

Figure 2: Spatial distribution of the six ecosystems services (A - F) and biodiversity (G) in the Cerrado.

Forty-five PAs (41.3%) and twenty-nine ILs (47.5%) did not represent effectively any of the studied ecosystem services (Fig. 3; Table 2). Most PAs and ILs were effective to provide only one ecosystem service (36.7% of the PA, and 29.5% of the ILs); while no reserve were effective to capture more than three services. The percentage of PAs of strict protection identified as effective or not was similar for PAs of sustainable use (Table 2). Also, most PAs (73.4%) and ILs (98.4%) were not effective to represent biodiversity (Table 2).

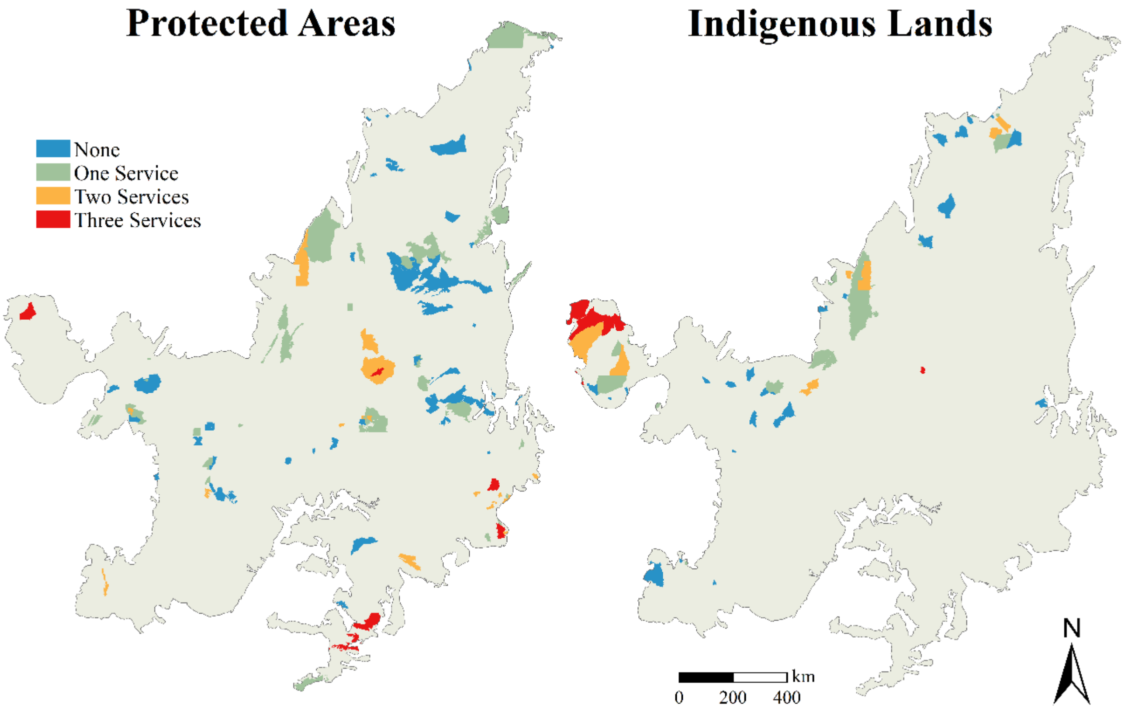

Figure 3: Distribution of protected areas and indigenous lands located at the Cerrado and their respective effectiveness to represent the six evaluated ecosystem services, namely water supply, sediment retention, nutrient retention, carbon storage, net primary productivity and wild food provision. The legend represents the amount of ecosystem services in which a reserve is effective to represent inside its boundary. A given reserve was considered effective to an evaluated ecosystem services when the amount of ecosystem services observed inside its boundary was significantly higher than the amount found in the randomly located reserves across the Cerrado, as defined by our null model ($p < 0.05$). None of the protected areas and indigenous lands was effective to represent four ecosystem services or more simultaneously. See Figure 4 and 5 for the distribution of protected areas and indigenous lands that are effective to represent each ecosystem services, as well as biodiversity.

Table 2: Percentage of protected areas (grouped in strict protection, sustainable use and all of them) and indigenous lands located at the Cerrado that are effective to represent ecosystem services, namely water supply, sediment retention, nutrient retention, carbon storage, net primary productivity and wild food provision. A given reserve was considered effective when the amount of ecosystem services observed inside its boundary was significantly higher than the amount found in the randomly located reserves across the Cerrado, as defined by our null model ($p < 0.05$).

	Protected areas			**Indigenous lands (%)**
	All (%)	**Strict protection (%)**	**Sustainable use (%)**	
Ecosystem services represented				
None	41.3	39	42.6	47.5
One	36.7	34.1	38.2	29.5
Two	16.5	17.1	16.2	11.5
Three	5.5	9.8	2.9	11.5
Four	0	0	0	0
Five	0	0	0	0
Six	0	0	0	0

The proportion of PAs identified as effective to provide each of the evaluated ecosystem services was relatively low, varying from 3.7 to 25.7% (Fig. 4, Table 3). Higher proportion of PAs was effective to provide sediment retention and water yield than other ecosystem services, but still, only 25.7% and 19.3% of them were designated as more effective than randomly located PA, respectively (Fig. 4, Table 3). The proportion of effective PAs in representing each ecosystem service was similar in both categories of PAs (*i.e.* strict protection and sustainable use), except for nutrient retention and carbon storage for which more PAs of strict protection were effective compared to PAs of sustainable use. As regard to biodiversity, the proportion of PAs identified as effective to represent threatened species (26.6%) was higher than any of the ecosystem services.

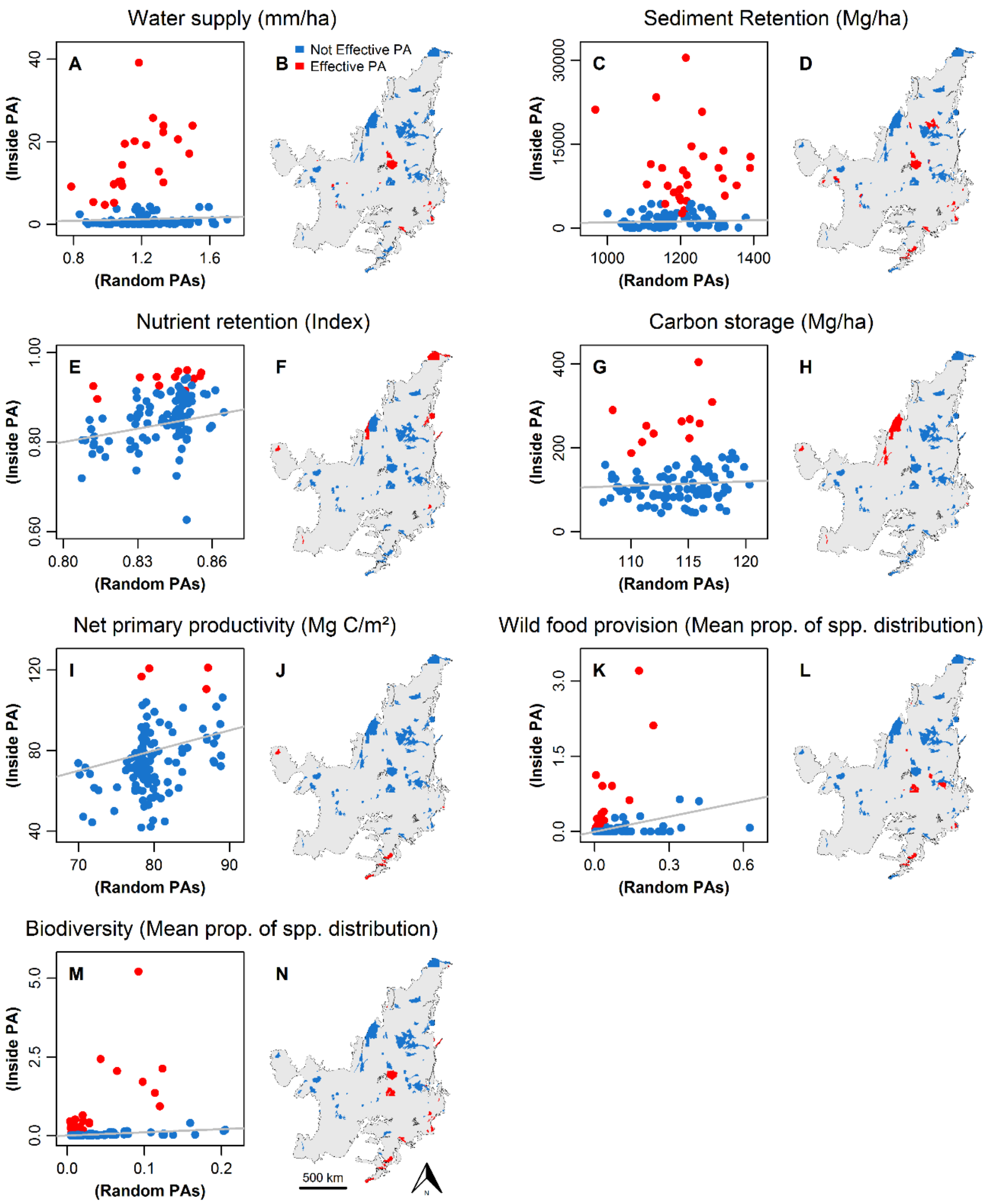

Figure 4: Protected areas effective to represent ecosystems services or biodiversity (red in graph and maps) and not effective (blue in graphs and maps). A given reserve was considered effective when the amount of ecosystem services or biodiversity observed inside its boundary was significantly higher than the amount find in the randomly located reserves across the Cerrado, as defined by our null model ($p < 0.05$). The diagonal line corresponds to 1 to 1 ratio. PA: protected area; prop: proportion and spp: species.

Table 3: Percentage of protected areas (grouped in strict protection, sustainable use and all of them) and indigenous lands located at the Cerrado that are effective to represent each of the evaluated ecosystem services and biodiversity. A given reserve was considered effective when the amount of ecosystem services or biodiversity observed inside its boundary was significantly higher than the amount found in the randomly located reserves across the Cerrado, as defined by our null model ($p < 0.05$).

	Protected areas			**Indigenous lands (%)**
	All (%)	**Strict protection (%)**	**Sustainable use (%)**	
Ecosystem services				
Water yield	19.3	17.1	20.6	1.6
Sediment retention	25.7	29.3	23.5	1.6
Nutrient retention	11.9	17.1	8.8	37.7
Carbon storage	10.1	17.1	5.9	32.8
Net primary productivity	3.7	2.4	4.4	14.7
Wild food provision	16.5	14.6	17.6	3.3
Biodiversity	26.6	29.3	25	1.6

The proportion of effective ILs was more variable between ecosystem services compared to PA, varying between 1.6 and 37.7% (Fig. 5; Table 3). The percentage of effective ILs were higher for nutrient retention and carbon storage, with 37.7% and 32.8% of the ILs providing more of these services than randomly located ILs. Only 1.6% of the ILs was effective to represent threatened species.

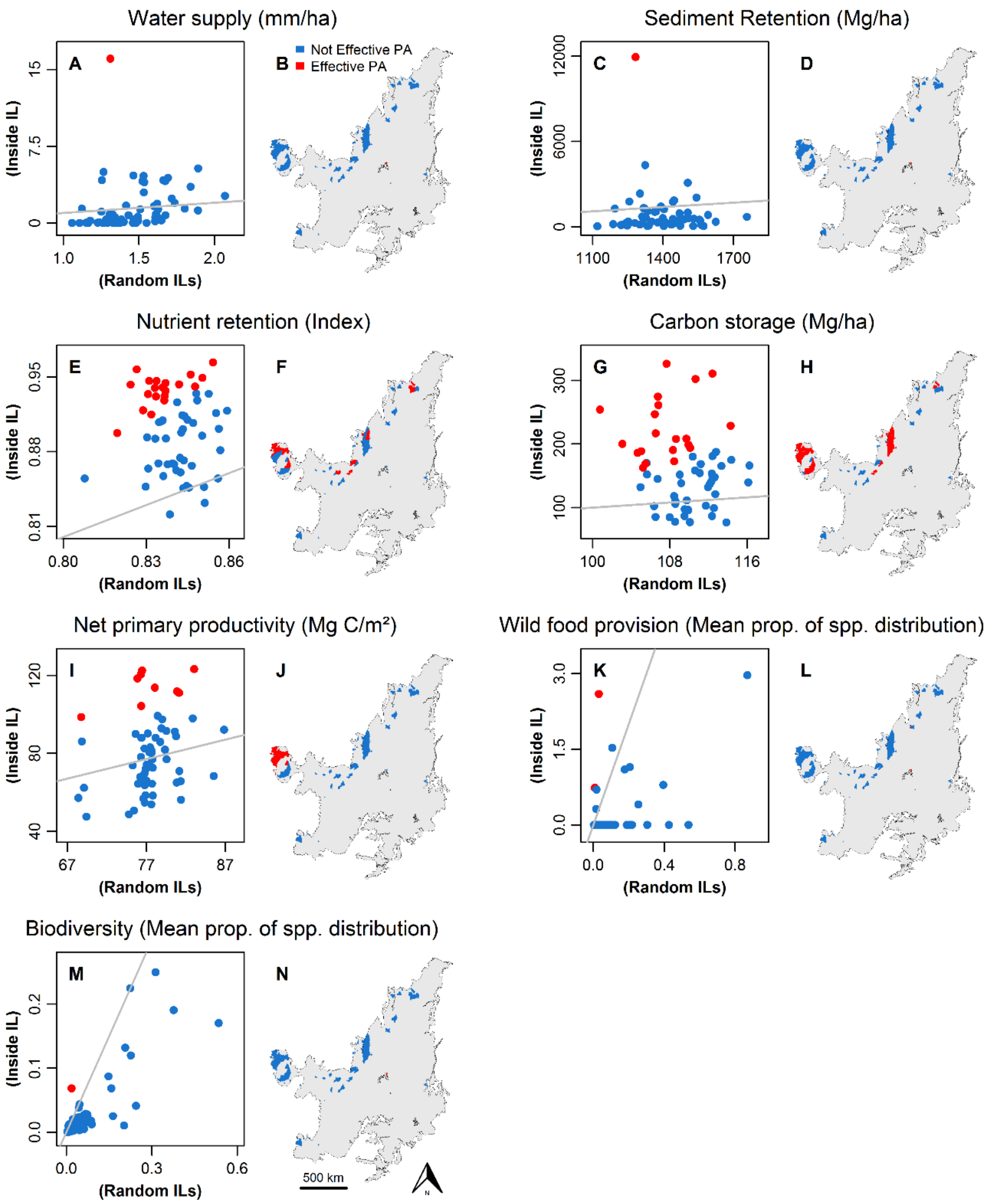

Figure 5: Indigenous lands effective to represent ecosystems services or biodiversity (red in graph and maps) and not effective (blue in graphs and maps). A given reserve was considered effective when the amount of ecosystem services or biodiversity observed inside its boundary was significantly higher than the amount found in the randomly located reserves across the Cerrado, as defined by our null model ($p < 0.05$). The diagonal line corresponds to 1 to 1 ratio. IL: indigenous lands; prop: proportion and spp: species.

Protected areas were more effective in representing water supply, sediment retention and wild food provision, capturing 17.4%, 14.6% and 14% of the amount of these services available in the whole Cerrado, respectively (Table 4). Meanwhile, PAs captured a much reduced portions of carbon storage and net primary productivity. PAs set for sustainable use tended to represent more ecosystem services than strict protection PAs, mainly for water yield and wild food provision. Compared to PAs, the amount of ecosystem services captured by ILs was smaller for all evaluated ecosystem services (ranging from 1.2% to 7.5%; Table 4). Regarding biodiversity, PAs represented on average 23.8% of the distribution range of threatened species, while ILs captured only 1.9% of the species range.

Table 4: Percentage of ecosystem services and biodiversity available in the Cerrado which is included in the protected areas or indigenous lands network. See methods for details about how the proportion of each ecosystem services and biodiversity was calculated.

	Protected areas			
	All	**Strict protection**	**Sustainable use**	**Indigenous lands**
Ecosystem services (%)				
Water yield	17.4	4.2	13.2	7.5
Sediment retention	14.6	4.7	9.9	2.5
Carbon storage	7.5	2.7	4.8	7.5
Net primary productivity	6.1	1.9	4.2	5.0
Wild food provision	14.0	1.6	12.4	1.2
Biodiversity (%)	23.8	7.0	16.8	1.9

Discussion

We showed that most PAs and ILs was not effective to capture ecosystem services and biodiversity of the Cerrado region, and the network of PAs and ILs captured a relatively small proportion of the total provision of services provided by the biome. Moreover, most effective reserves were suitable for safeguarding just one out of six ecosystem services. These results reinforce that other reasons, besides representativeness of conservation features, guide the spatial distribution of reserves (Baldi et al., 2017; Durán et al., 2013; Joppa and Pfaff, 2009).

At least three explanations could be raised to understand the poor effectiveness of PAs and ILs in capturing conservation features, as well as the low representativeness of those features in the reserve networks. First, establishment of PAs has been guided by opportunities, instead of representativeness of conservation features. For example, PAs tend to be biased towards regions inappropriate to agriculture, distant from urban centers and with low human density (Baldi et al., 2017; Pressey, 1994). The biased protection towards areas less attractive to human uses has been showed worldwide, in Latin America & Caribbean (Baldi et al., 2017) and Brazil (Joppa and Pfaff, 2009). Second, as the ecosystem services concept is relatively new to conservation science, establishment of PAs seems not to be considerably influenced by ecosystem services yet. In fact, biodiversity (*e.g.* species richness, rates of endemism and threatened species) and not ecosystem services has played important role to guide global conservation strategies (Myers et al., 2000; Olson and Dinerstein, 2002). In Brazil, threatened species have been a centerpiece in guiding the establishment of PAs or other management actions (*e.g.* MMA, 2016), although ecosystem services mapping is advancing fast (Duarte et al., 2016; Manhães et al., 2016; Vieira et al., 2017). Third, as PAs and ILs cover a small portion of the Cerrado, even if they were well planned, their system would unlikely represent high amount of ecosystem services.

Our results show that PAs of sustainable use tend to capture more ecosystem services and biodiversity then PAs of strict protection. A positive result to be highlighted is that the two ecosystem services most represented by PAs of sustainable use are provisioning services (water yield and wild food provision). As this category of PAs allows the sustainable use of resources within their limits, these services are more compatible with sustainable use than with strict protection. Nonetheless, the higher representation of all ecosystem services by PAs of sustainable use reinforces that to guarantee the conservation of ecosystem services already captured by the existing PAs, PAs of sustainable use should embrace a management strategy that guarantee the provision of the ecosystem services in a long run. PAs of sustainable use has been established in regions with consolidated human activities and allow different types of direct use of natural resources (*e.g.* wood extraction, mining and agriculture). Thus, it is a challenge to avoid the human pressure inside PAs of sustainable use and maintain the level of coverage of ecosystem services found inside this PAs category (Carranza et al., 2014; Rylands and Brandon, 2005).

Our study also contributes to understand the role that ILs play to the nature's conservation strategies. Although a smaller proportion of ecosystems services were found inside ILs network compared to PAs, the proportion of effective ILs was higher than PAs to represent three ecosystem services (*i.e.* nutrient retention, carbon storage and net primary productivity). This result suggests that the low representativeness of ecosystem services inside ILs may be related to the small coverage of ILs and not only to the ILs distribution. Apart from the considerable proportion of effective ILs for some ecosystem services, ILs are also important to reduce habitat loss in the Cerrado (Carranza et al., 2014; Paiva et al., 2015), which suggest that conservation strategies should give more attention to the role of ILs to nature conservation. Nowadays, ILs are weakly considered in conservation agenda and are not recognized by national (*i.e.* SNUC) and international (*e.g.* CDB and IUCN) conservation schemes.

On the protection of individual ecosystem services

Water yield, sediment and nutrient retention were the ecosystem services better represented by the actual reserve networks in the Cerrado. However, future conservation strategies should pay close attention to the conservation of those services, as they influence several socio-economic aspects. Water security in Brazil depends on the maintenance of those services, as the Cerrado region delivers water to eight of the 12 major watersheds of the country (Overbeck et al., 2015) and natural control of excessive sediment and nutrient loss is key to maintain water quality (Sharp et al., 2016; Walling, 2009). Moreover, the production of hydroelectric energy, which is source of 72% of energy produced in Brazil (Medeiros et al., 2011), is also highly dependent on water yield and sediment retention services. Several hydroelectric power dams are located in the Cerrado region and sediment loss control is important to expand the lifespan of dams and reservoirs (Walling, 2009). In addition, sediment and nutrient loss control are fundamental to avoid land degradation, playing an important role to maintain food security and crop productivity in the region (Montgomery, 2007).

The maintenance of carbon storage in the Cerrado is also strategic to the Brazilian climate change policy (Ribeiro et al., 2011), as the highest levels of greenhouse gases emission arising from land-use changes occur in the biome (MCTi, 2014). However, carbon storage is not properly safeguarded in reserve networks of the Cerrado. While the amount of carbon storage represented by the PAs and ILs was the same, ILs had more effective reserves capturing this ecosystem service. This result may be related to the distribution of ILs, which are mostly located close to the transition to Amazon, where the levels of carbon storage are high (*i.e.* the north and west region of the Cerrado; see Fig. 1 and Fig. 2). Both the high levels of habitat conversion and the significant reduction of deforestation in Amazon, which increased spillover towards the Cerrado in the last years, contributed to elevate the emission of greenhouse gases from the Cerrado (Gibbs et al., 2015; MCTi, 2014).

Net primary productivity was weakly represented by the reserve networks. The low protection level of this service is worrying because it determines the amount of energy that enter in the ecosystem and that is available to other species, thus playing a major role to the ecosystem functioning (Vitousek et al., 1986). Further, net primary productivity underpins the provision of many other ecosystem services (Balvanera et al., 2006; MEA, 2005). Considering evidences that net primary productivity is dependent of biodiversity levels (Balvanera et al., 2006), the loss of plant species in course in the Cerrado (Strassburg et al., 2017) may also contribute to the reduction of this service in native remnants of the region.

The two conservation features based on species range distribution, wild food provision and biodiversity, were better represented by PAs then by ILs. The low representation of wild food provision inside ILs is critical especially because the livelihood of indigenous people depends strongly on nature. Regarding to the effectiveness of PAs in representing both features, additional studies could determine if PAs are established in important areas to the occurrence of edible plant and threatened species or this result is a sampling artefact as biodiversity databases in Brazil are usually biased to regions close to roads and PAs (Oliveira et al., 2016). In any case, even with known issues, our databases used were validated by experts and have been used in other studies (Oliveira et al., 2015; Strassburg et al., 2017). In the case of threatened species, the Brazilian government has used the same database to develop conservation strategies for the region.

Improving ecosystem services conservation

The low representation of ecosystem services by PAs and ILs cannot be associated exclusively with the spatial distribution of the reserves, but also with the low extension that PAs and ILs networks cover of the Cerrado. Only 7.1% and 4.8% of the Cerrado are covered by PAs and ILs, respectively. To improve the conservation of ecosystem services, PAs network should be expanded to areas that maximize the representation of multiple ecosystem services and deliver them to local and traditional communities (Cimon-Morin et al., 2014). The priority areas for biodiversity conservation in the Cerrado was recently updated by the federal government (MMA, 2016), but it is unknown if these areas are also effective to represent multiple ecosystem services.

Efforts to conserve ecosystem services should include different types of conservation strategies and not depend exclusively on the design of PAs (Daily and Matson, 2008). Thus, besides the expansion of the network of PAs and ILs, efforts to conserve ecosystem services in the Cerrado should also focus on private lands. The majority of the remnant of native vegetation in the Cerrado are located in private lands (Soares-Filho et al., 2014), what makes the native vegetation suffer an intense anthropic pressure. However, the New Forest Code, approved in 2012, become more environmentally permissive than the previous one (Loyola, 2014; Metzger, 2010) and allows the legal conversion of 40 million ha of the Cerrado (*i.e.* 45% of the total area that can be legally deforest in Brazil) (Soares-Filho et al., 2014; Strassburg et al., 2017). As a consequence, the requirements of the New Forest Code are not enough to guarantee the maintenance of the biodiversity, with also have serious impact on ecosystem services (Brancalion et al., 2016; Metzger, 2010; Vieira et al., 2017). The limited law enforcement in the Cerrado (Gibbs et al., 2015; Soares-Filho et al., 2014) also contributes to difficult the conservation of its ecosystems and associated services.

We used a spatially explicit model that compares the representativeness of the reserves in relation to other areas of the Cerrado to evaluate the effectiveness of its reserves networks in capturing both ecosystem services and biodiversity. Our study stands out because previous assessments using similar approach around the world had focused exclusively on biodiversity (Ferro et al., 2014; Lemes et al., 2014; Ribeiro et al., 2016). We demonstrated that PAs and ILs networks still poorly represent ecosystem services in the Cerrado, as well as a small portion of ecosystem services provided by the biome is captured by these reserves. Given that spatial congruence between biodiversity and ecosystem services tends to be low (Cimon-Morin et al., 2013), systematic conservation planning could be suitable to plan simultaneously for both ecosystem services and biodiversity. Conservation policies should also focus in the maintenance of ecosystem services in private lands.

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Supplementary material

Table S1: Land-use classes and their respective biophysical variables used in InVEST models.

Land-use classes	C Factor[1]	P Factor[2]	Root depth[3]	Kc[4]	Load N[5]	Load P[5]	Eff. N[5]	Eff. P[5]	Critical length P and N[6]
Pasturelands	0.03	0.85	1000	1	5412.5	101.84	0.25	0.25	1000
Soybeans	0.0899	0.5	950	1	10767.5	1262.75	0.25	0.25	1000
Corn	0.088	0.5	1350	1	10767.5	1262.75	0.25	0.25	1000
Sugar cane	0.26	0.5	1600	1	10767.5	1262.75	0.25	0.25	1000
Annual agriculture	0.172	0.5	750	1	10767.5	1262.75	0.25	0.25	1000
Perennial agriculture	0.219	0.5	1250	1	10767.5	1262.75	0.5	0.5	1000
Silvicultural fields	0.01	1	1250	1	10767.5	1262.75	0.75	0.75	1000
Ombrophilous forest	0.001	1	1500	1	2190	142.57	0.9	0.9	75
Seasonal forest	0.01	1	3700	1	2190	142.57	0.85	0.85	75
Forested savanna	0.01	1	3700	1	2190	142.57	0.9	0.9	75
Wooded savanna	0.04	1	7000	1	1500	115	0.7	0.7	100
Park savanna	0.04	1	1500	1	1500	115	0.7	0.7	100
Steppic savanna	0.04	1	1500	1	1500	115	0.7	0.7	100
Gramineous-woody savanna	0.042	1	1000	1	1000	90	0.5	0.5	100
Mountainous vegetation	0.042	1	500	1	1000	90	0.5	0.5	100
Savanna wetland	0.042	1	1000	1	1000	90	0.5	0.5	100
Urban areas	0.06	0.98	1	1	5812.63	216.08	0.0496	0.0638	1000
Mining pits	1	1	1	1	5812.63	216.08	0.05	0.05	1000
Water bodies	0.07	1	1	1	2601.42	161.37	0	0	1000
Bare land	1	1	1	1	5812.63	216.08	0.05	0.05	1000
Non-identified	1	1	1	1	1	1	0.01	0.01	1000

[1] References to crop-management factor (C): Bertol et al. (2001), De Maria and Lombardi Neto (1997), Duarte et al. (2016), Farinasso et al. (2006), Kennedy et al. (2016), Manhães et al. (2016), Oliveira et al. (2011), and Silva et al. (2007).

[2] References to support practice factor (P): Duarte et al. (2016), Farinasso et al. (2006), Kennedy et al. (2016), Manhães et al. (2016), Oliveira et al. (2011), Silva et al. (2007), and Yang et al. (2003).

[3] References to define root depth: Manhães et al. (2016), Oliveira et al. (2005), Rodin (2004), and Castro and Kauffman (1998).

[4] References to define load and retention efficiency: Kennedy et al. (2016), and Manhães et al. (2016).

[5] References to define critical length: Mayer et al. (2007).

Table S2: Ecosystem services and biodiversity observed inside each protected areas considered in our analysis. APA: environmental protection area (IUCN category V); ARIE: area of relevant ecological interest (IV); ESEC: ecological stations (Ia); FLONA: national forest (VI); MONA: natural monuments (III); PARNA: national parks (II); RDS: sustainable development reserve (VI); REBIO: biological reserves (Ia); RESEX: extractive reserve (VI); REVIS: wildlife refuges (III); spp: species.

Protected Areas	Water supply (x 10^3)	Sediment retention (x 10^3)	Nutrient retention	Carbon storage (x 10^3)	Net primary productivity (x 10^6)	Wild food provision	Biodiversity (threatened spp.)
Strictly protected							
ESEC da Serra das Araras	0.1	90,934.8*	0.9	1,866.0	1,013.7	0	<0.1
ESEC de Iquê	635.4	141,003.1	15.1*	41,424.7*	23,385.3*	0	<0.1
ESEC de Uruçui-Una	27.7	93,298.5	8.8	10,303.4	6,809.6	0	<0.1
ESEC Serra Geral do Tocantins	339.2	1,671,818.3	46.7	42,431.9	31,244.1	0.1	<0.1
MONA das Árvores Fossilizadas	7.7	15,519.6	1.8	2,093.5	1,584.1	0	<0.1
PAREST Águas do Cuiabá	1.8	51,774.5	0.9	1,225.0	840.7	<0.1	<0.1
PAREST Biribiri	232.5*	91,897.5*	0.9	2,108.9	1,005.4	0	0.4*
PAREST Caminho dos Gerais	39.9	15,507.8	0.9	2,276.1	694.7	0	<0.1
PAREST de Terra Ronca	24.9	94,029.4	2.4	4,247.1	2,414.8	<0.1	<0.1
PAREST do Araguaia	226.4	10,965.4	15.4	53,126.1*	9,133.7	0	<0.1
PAREST do Cantão	21.2	1,903.3	4.7*	24,477.4*	4,475.7	0	<0.1
PAREST do Jalapão	58.4	798,728.8*	9.5	8,294.6	6,504.3	<0.1	<0.1
PAREST do Mirador	295.4	425,181.0	40.5	54,560.7	35,018.0	0	<0.1
PAREST do Verde Grande	0.1	169.5	0.9	2,144.8	878.4	0	<0.1
PAREST Grão-Mogol	96.5	139,794.4*	1.8	3,601.6	1,677.5	<0.1	0.3*
PAREST Lapa Grande	1.8	32,421.5	0.9	1,833.5	1,073.9	0	<0.1
PAREST Nascente Rio Taquari	219.8*	249,403.9*	1.6	3,030.5	2,571.5	0	<0.1
PAREST Rio Preto	65.0*	28,713.8	0.8	664.4	768	0	0.3*
PAREST Serra das Araras	11.5	35,859.9	0.9	1,065.9	897.4	<0.1	<0.1
PAREST Serra do Intendente	41.4	176,692.5*	0.8	1,035.4	1,137.8	0	0.4*
PAREST Veredas do Peruaçu	20.8	9,127.1	1.8	2,718.1	1,906.8	0	<0.1
PARNA Cavernas do Peruaçu	3.4	43,637.5	2.7	5,639.1	2,767.6	<0.1	0.1*
PARNA da Chapada das Mesas	9.4	288,211.8	10.1	8,995.0	8,839.0	<0.1	<0.1
PARNA da Chapada dos Guimarães	6.1	462,122.2*	2.6	3,647.1	2,602.2	0.1*	0.1*

PARNA da Chapada dos Veadeiros	2,367.6*	1,414,409.9*	4.2	6,068.8	3,628.3	0.4*	0.5*
PARNA da Serra da Bodoquena	0.3	185,072.9	5.5*	17,012.5*	7,389.0	0	<0.1
PARNA da Serra da Canastra	2,002.8*	2,481,467.9*	11.8	18,639.6	17,565.5	0	1.7*
PARNA da Serra das Confusões	58.7	165,618.1	39.2*	81,046.1	31,281.4	0	0.1
PARNA da Serra do Cipó	269.4*	368,218.5*	0.6	549.6	880.6	<0.1	0.2*
PARNA das Emas	64.6	83,432.6	9	13,143.3	11,354.0	<0.1	<0.1
PARNA das Nascentes do Rio Parnaiba	108.6	2,209,445.3*	49.3	49,275.1	36,648.3	0	<0.1
PARNA das Sempre-Vivas	2,124.8*	527,981.9*	8.0*	13,570.1	8,143.5	0	2.1*
PARNA de Brasília	1.8	85,914.9	2.5	5,169.4	2,642.7	0.3*	0.2*
PARNA do Araguaia	2,146.5	30,489.8	39.5*	139,589.1*	25,980.2	0	0.1
PARNA dos Lençóis Maranhenses	0	5,468.0	8.6*	11,554.2	4,546.2	0	<0.1
PARNA Grande Sertão Veredas	343.9	137,658.3	16.8	18,476.3	15,108.6	0.3	<0.1
REVIS Corixão da Mata Azul	8.4	1,302.1	1.8	6,370.0*	1,462.7	0	<0.1
REVIS das Veredas do Oeste Baiano	67.2	32,107.2	5.5	6,331.7	4,432.1	0	<0.1
REVIS Panela	45.6	136,851.5	3.9	6,181.7	5,281.3	0	<0.1
REVIS Quelônios do Araguaia	34.6	3,178.6	3.6	12,237.2*	3,395.8	0	<0.1
REVIS Veredas do Acari	38.7	9,722.9	2.6	3,352.0	2,442.3	0	<0.1
Sustainable use							
APA Águas Vertentes	124.1*	94,106.7*	0.9	595.9	979.3	0	0.2*
APA Carste de Lagoa Santa	27.3	29,794.8	2.5	3,161.9	3,678.5	0.2*	0.4*
APA Cavernas do Peruaçu	46.6	32,417.8	5.2	7,486.0	5,739.8	0.1	<0.1
APA Cochá e Gibão	208.5	129,387.1	20	28,322.7	18,118.9	0	<0.1
APA Corumbatai-Botucatu-Tejupa	28.9	1,139,079.6*	30.3	56,671.4	48,153.7*	2.1*	0.9*
APA da Bacia do Rio de Janeiro	94.5	255,676.8	19.2	15,867.6	14,759.0	0.2	<0.1
APA da Bacia do Rio Descoberto	37.3	45,768.6	1.6	3,177.6	1,707.0	<0.1	<0.1
APA da Baixada Maranhense	0	819.1	0.9	1,244.0	1,043.7	0	<0.1
APA da Chapada dos Guimarães	63.8	1,610,867.4*	12.6	16,664.9	13,015.0	0.3	<0.1
APA da Foz do Rio Preguiças/Pequeno Lençóis	0	9,330.4	10.4*	17,167.1	9,926.8	0	<0.1
APA da Serra da Jibóia	1.3	34,758.8	0.8	2,060.9	1,240.3	0	<0.1
APA da Serra das Araras	0.3	307,756.5*	3.3	4,987.0	3,598.1	0	<0.1

APA da Serra dos Pirineus	112.0*	51,299.8	0.9	1,084.7	1,113.7	<0.1	<0.1
APA da Serra Geral de Goiás	39.5	155,984.1*	2.5	3,270.0	2,227.1	<0.1	<0.1
APA das Bacias dos Córregos Gama e Cabeça de Veado	6.8	14,141.7	0.8	1,798.7	826.7	1.1*	<0.1
APA das Cabeceiras do Rio Cuiabá	41	1,181,600.0	31.9	42,862.0	30,187.8	0.3	<0.1
APA das Nascentes de Araguaína	3.6	8,310.7	0.8	1,032.9	681.7	0.1*	0.1*
APA das Nascentes do Rio Vermelho	296.5	266,923.3	11.1	15,950.8	10,520.0	0.9*	<0.1
APA de Cafuringa	1.2	166,474.3*	1.7	3,306.5	1,874.9	0.1*	0.1*
APA Delta do Parnaíba	0	615.4	2.8*	5,902.8	2,512.8	0	<0.1
APA do Arica-Açu	10.8	64,694.4	5.2	7,621.8	5,481.1	0	<0.1
APA do Jalapão	289.9	586,201.5	20.7	18,216.1	14,272.7	0	<0.1
APA do Pé da Serra Azul	122.9*	52,407.7	0.8	973.5	1,108.6	0	<0.1
APA do Planalto Central	127.6	841,323.6	30.2	62,340.8	34,187.2	3.2*	1.4*
APA do Pontal dos Rios Itiquira e Correntes	9.7	76,569.5	2.8*	3,222.6	2,952.7	0	<0.1
APA do Rangel	0.4	5,675.2	0.9	704.1	776.2	0	<0.1
APA do Ribeirão João Leite	7.8	89,118.3	3.9	8,221.2	5,637.1	0	<0.1
APA do Rio Dantas e Morro Verde	693.8*	154,890.5	3.4	3,958.8	3,820.7	0	<0.1
APA do Rio das Garças e Furnas do Batovi	253.1	100,860.0	5	6,236.4	6,441.9	0	<0.1
APA do Rio Pandeiros	348.9	199,865.4	25.1	29,049.2	25,894.6	0.6*	0.1
APA do Rio Preto	96.7	2,011,076.1	72.4	80,870.5	58,269.4	0.6	0.2
APA do Rio São Bartolomeu	15.5	131,407.0	4.8	11,066.5	5,418.0	0.9*	0.4*
APA do Salto Magessi	56.5*	10,955.0	0.9	1,906.6	1,058.5	0	<0.1
APA Dunas e Veredas do Baixo Médio São Francisco	0.2	72,003.4	7.6*	11,546.5	5,831.8	0	0.2*
APA Estadual da Escarpa Devoniana	0.9	433,201.4	15.3	25,287.2	27,839.6*	<0.1	5.2*
APA Foz do Rio Santa Tereza	1.4	5,213.7	3.5	7,584.5	3,042.0	0.2*	<0.1
APA Ibitinga	0.2	18,592.4	2.3	4,529.9	3,362.7	0	<0.1
APA Ilha do Bananal/Cantão	6,435.5	369,640.0	111.8	293,122.0*	96,420.5	<0.1	0.1
APA Lago de Palmas	14.9	13,143.4	2.4	2,880.8	2,137.7	0	<0.1
APA Lago de São Salvador do Tocantins, Paranã e Palmeirópolis	3,854.1*	2,857,071.9*	22.6	30,564.9	21,261.6	0	0.1
APA Meandros do Rio Araguaia	159.7	16,002.7	22.6	87,791.3*	20,694.7	0	<0.1

APA Morro da Pedreira	1,705.1*	908,390.9*	5.6	7,265.8	7,539.2	0.4*	2.4*
APA Municipal Barão e Capivara	578.2*	259,799.1*	1.5	1,073.1	1,955.7	0	0.6*
APA Municipal do Rio Taquari	0.4	1,728.2	0.9	1,294.3	988.2	0	<0.1
APA Municipal Felício	248.8*	251,176.7*	0.9	1,594.5	1,258.5	0	0.2*
APA Municipal Itacuru	413.9*	335,067.6*	1.9	2,436.5	2,344.1	0	0.5*
APA Municipal Serra do Cabral	311.7*	137,888.9*	0.7	566.6	1,030.9	0	<0.1
APA Nascente do Rio Araguaia	23.1	67,003.1	2.3	3,829.2	2,622.6	0	<0.1
APA Nascentes do Rio Capivari	0.6	32,433.0	1.8	5,391.5*	2,823.4*	0	<0.1
APA Nascentes do Rio Paraguai	8.2	221,719.3	4.4	8,514.6	4,321.8	<0.1	<0.1
APA Ninho das Águas	117.3*	255,785.2*	0.8	1,487.8	1,200.0	0	<0.1
APA Pouso Alto	19,941.9*	8,953,396.7*	57.7	83,918.2	59,686.7	0.6	2.1*
APA Ribeirão Claro, Águas Emendada, Paraíso e Rio	222.4	299,542.8*	4	6,255.8	4,472.7	<0.1	<0.1
APA Ribeirão da Aldeia e Rio das Garças	103.1	81,563.7	1.6	1,946.1	1,890.3	0	<0.1
APA Ribeirão do Sapo e Rio Araguaia	188.5*	67,586.8	2.6	4,596.6	2,810.6	0	<0.1
APA Rio Araguaia, Córrego Rico e Couto Magalhães	87.7	132,254.0	2.4	3,864.5	2,808.6	0	<0.1
APA Rio Bandeira, das Garças e Taboca	103.2	40,757.5	1.7	1,943.5	1,802.7	0	<0.1
APA Rio Uberaba	69	147,619.2	15.9	24,629.1	20,913.9	0	<0.1
APA Serra da Ibiapaba	0	2,174.3	0.9	1,653.3	905.3	0	0.1*
APA Serra da Tabatinga	2.4	38,301.4	2.5	1,830.0	1,530.8	0	<0.1
APA Serra das Galés e da Portaria	114.3	76,457.4	3.2	5,263.6	4,042.2	0	<0.1
APA Serra do Lajeado	220	742,786.4*	7.3	10,062.6	5,811.0	0	<0.1
APA Serra do Sabonetal	2.2	22,292.1	3.6	8,372.3	3,666.9	0	<0.1
APA Upaon-Açu/Miritiba/Alto do Rio Preguiças	0.1	65,221.1	67.6*	113,319.3	63,825.4	<0.1	0.4
FLONA de Cristópolis	0.1	611	0.9	666.6	814.3	0	<0.1
Florest do Araguaia	4.8	578.6	0.9	3,743.9*	668.2	0	<0.1
RESEX da Mata Grande	0	1,651.2	0.9	1,566.4	1,015.4	0	<0.1
RESEX Marinha do Delta do Parnaíba	0	159.2	1.0*	2,102.2	963.7	0	<0.1

* Significant according to the null model ($p < 0.05$).

Table S3: Ecosystem services and biodiversity observed inside each indigenous land considered in our analysis. Spp: species.

Indigenous land	Water supply (x 10^3)	Sediment retention (x 10^3)	Nutrient retention	Carbon storage (x 10^3)	Net primary productivity (x 10^6)	Wild food provision	Biodiversity (threatened spp.)
Buriti	0.2	34,309.5	1.7	4,001.2	2,368.1	0	<0.1
Bakairi	13.6	44,276.2	6	6,504.8	5,563.0	0	<0.1
Areões	168	28,721.2	16.9*	40,474.7*	19,173.2	0.2	<0.1
Kadiwéu	1.1	803,388.3	32.9	50,436.4	42,313.3	0	0.2
Merure	165.9	118,179.2	6.9	8,205.3	8,512.6	0	<0.1
São Marcos	288.5	636,518.8	14.4	20,982.8	17,707.2	0	<0.1
Cachoeirinha	0	2,956.9	0.9	2,116.7	1,122.9	0	<0.1
Chão Preto	50.6	15,468.0	0.9	1,199.7	869.2	0	<0.1
Figueiras	4.3	28,243.9	0.8	1,915.8	1,065.0	0	<0.1
Juininha	24	45,972.2	3.5	7,459.9	3,281.2	0	<0.1
Marechal Rondon	109.4	173,679.1	9.5	11,507.8	8,786.8	0	<0.1
Parabubure	212.5	152,672.7	21.2*	36,718.6	20,665.9	0	<0.1
Paresi	132	359,801.2	46.4	102,266.6*	36,757.6	<0.1	0.1
Rio Formoso	13.4	104,920.1	1.8	3,709.5	2,010.9	0	<0.1
Sangradouro/Volta Grande	117.7	54,139.2	10.2	14,093.4	10,803.3	0	<0.1
Santana	13.1	15,868.9	3.6	8,717.0	3,899.7	0	<0.1
Tadarimana	1.1	499.2	0.8	925.9	930.2	0	<0.1
Taihantesu	2.9	21,445.6	0.8	2,188.2	1,375.4*	0	<0.1
Taunay/Ipegue	0.1	6,412.1	1.9*	3,196.7	2,213.7	0	<0.1
Ubawawe	281.4	79,008.2	4.3	6,398.3	4,459.1	0	<0.1
Uirapuru	23	40,406.4	2.6	5,109.9	2,522.0	0	<0.1
Umutina	0.9	4,132.1	2.8	4,965.4	3,262.6	0	<0.1
Avá-Canoeiro	776.2*	577,152.3*	3.4	4,190.3	3,889.4	0.3*	<0.1
Xacriabá	4.7	26,126.4	1.8	3,561.5	1,756.7	0	<0.1
Xacriabá	10	44,298.7	5.3	10,045.3	5,673.2	0	<0.1
Enawenê-Nawê	2,666.5	620,305.3	57.3*	150,001.0*	88,767.6*	0	0.2
Manoki	45.7	6,102.5	1.9*	4,014.6	1,556.6	0	<0.1
Maraiwatsede	29.7	29,217.7	6.3	12,838.1	6,000.2	0	<0.1
Menkü	198.6	18,194.0	3.7*	15,026.7*	5,370.3*	0	<0.1
Menkü	658.3	65,443.3	12.0*	43,126.6*	16,394.5*	0	<0.1
Parque do Aripuanã	1,186.7	174,765.4	38.7*	122,393.4*	60,776.0*	0	0.1
Pirineus de Souza	160.3	74,381.5	2.8*	7,209.1*	4,473.3*	0	<0.1
Tirecatinga	352.8	92,236.2	9.9	22,651.2	13,124.8*	0	<0.1
Urubu Branco	579.9	44,201.6	8.3*	22,622.9*	6,215.4	0	<0.1
Apinayé	6.3	111,562.5	10.8	12,184.6	12,092.0	0	<0.1
Arariboia	0	1,550.6	1.0*	1,463.7	1,037.8	0	<0.1
Bacurizinho	0.1	38,766.5	11.1*	25,053.0*	11,889.6	0	<0.1
Xerente	238	116,390.5	14.1	28,071.9	12,048.0	0	<0.1

Cana Brava/Guajajara	0	22,602.7	11.3*	27,655.0*	13,229.2	0	<0.1
Funil	33.9	25,418.9	1.7	2,313.7	1,410.3	<0.1	<0.1
Geralda Toco Preto	0	3,291.2	0.8	1,688.6	1,091.5	0	<0.1
Governador	0.1	9,625.8	4.6*	8,947.4	5,369.2	0	<0.1
Inawebohona	1,833.4	23,503.2	30.3*	104,286.7*	21,452.5	0	<0.1
Kanela	0.1	49,137.8	9.6	13,695.3	8,641.5	0	<0.1
Kanela Memortumré	0.2	68,738.6	7.9	12,116.4	7,040.5	0	<0.1
Kraolandia	573.3	173,783.7	26.2	29,787.0	22,308.7	0.1	<0.1
Krikati	12.3	62,640.8	12.8	16,409.0	13,956.9	0	<0.1
Lagoa Comprida	0	2,149.1	0.9*	2,035.5	1,199.6	0	<0.1
Parque do Araguaia	6,049.7	87,534.9	111.1*	375,186.3*	74,613.5	0.3	0.2
Porquinhos dos Kanela Apãnjekra	0.9	138,864.4	24.3*	38,492.4	22,331.1	0	<0.1
Tapirapé/Karajá	114.2	13,351.0	6.6*	17,621.7*	4,756.9	0	<0.1
Urucu/Juruá	0	7,374.8	1.8	3,559.3	2,356.2	0	<0.1
Krahó-Kanela	60.3	949.1	0.9	3,655.8*	661.1	0	<0.1
Utaria Wyhyna/Iròdu Iràna	405.4	11,273.1	15.6	67,105.4*	9,753.6	0	<0.1
Taego Ãwa	192.8	2,885.7	3.6	11,053.2*	2,746.4	<0.1	<0.1
Vale do Guaporé	13.9	67,369.4	2.7	7,033.5*	4,055.2*	<0.1	<0.1
Nambikwara	2,868.1	944,989.8	70.8	180,206.8*	115,207.0*	0	0.2
Pimentel Barbosa	317.9	105,302.8	27.9*	55,123.5	26,199.2	0.1	<0.1
Utiariti	267.9	142,023.7	37.9*	83,896.5*	40,465.7	<0.1	<0.1
Wedezé	225.4	11,324.2	12.8*	36,730.9*	10,890.1	0	<0.1
Cacique Fontoura	53.7	4,145.9	3.7*	9,069.5	2,354.5	0	<0.1

* Significant according to the null model ($p < 0.05$).

Capítulo 3

Delaying Actions for Safeguarding Ecosystem Services will Increase Conservation Conflicts in the Cerrado

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Abstract

In a world of increasing demand of natural resources, conservation is not the most important matter of stakeholders and postponing conservation actions is frequent. As a consequence, the success of conservation strategies might be impaired and puts at risk ecosystem services (ES) and maintenance of human well-being. Here, we evaluated the impact of postponing conservation actions to safeguard ES in the Cerrado, Brazil, the most diverse tropical savanna in the world that has experienced a rapid expansion of agriculture. We generated land use maps for the present and two future time steps (2025 and 2050), using a comprehensive land use model. Based on these land use maps we modeled the provision of six ES for the three time steps: water yield, sediment retention, nutrient retention, carbon storage, net primary productivity and wild food provision. We identified priority areas for safeguarding ES to meet four conservation targets (*i.e.* 10%, 20%, 30% and 40% of each ES) in the three time steps. We found that expected land use changes diminish the ES provision over time along with changes in their spatial distribution. The spatial distribution of priority areas in the region also differed between present and future. Moreover, priority areas identified in 2025 and 2050 will encompass greater amounts of altered environments then they could currently include. The increases of altered environments inside priority areas over time may increase conflicts between conservation actions and human activities. We highlight that establishing conservation actions to safeguard ES in the Cerrado today is more effective than postponing conservation actions for the next decades.

Keywords: spatial prioritization; replacement cost; InVEST; Marxan; nature’s contribution to people; Brazil.

Introduction

The human population increase, associated with economic growth, have altered the global ecosystems dramatically (Crist et al., 2017; Rockstrom et al., 2009). Currently, land use changes are ubiquitous: at least 75% of the Earth's ice-free land are converted, mainly pasture and agriculture (Ellis and Ramankutty, 2008). The human impact on ecosystems might continue to increase in the coming years to attend the global demand for food, which is expected to increase between 60% and 70% by 2050 (Alexandratos and Bruinsma, 2012; FAO, 2009). As a consequence of land use changes, the capacity of nature to provide ecosystem services (ES) is impaired (Foley et al., 2005; MEA, 2005). For example, land use changes interfere in regional climate regulation (Nobre, 2014), diminish insect pollination (Winfree et al., 2011), impair soil and water quality (Hunke et al., 2015), and favor incidence of diseases (Yasuoka and Levins, 2007). Therefore, the challenge is to protect nature in order to sustain the provision of ES and maintain human well-being (Díaz et al., 2018), mainly in frontier landscape prone to economic development.

Systematic conservation planning (SCP) is useful to design efficient conservation strategies by maximizing the representativeness of multiple conservation features while minimizes their associated costs (Margules and Pressey, 2000; Moilanen et al., 2009). This approach uses transparent and defensible technics to select priority areas aiming to achieve different conservation targets (Kukkala and Moilanen, 2013). Historically, SCP has focused on biodiversity (Noss et al., 2009), but more recently SCP has also been used to solve spatial problems based on ES (Chan et al., 2006; Cimon-Morin et al., 2014; Manhães et al., 2018). In addition to the spatial distribution of conservation features (*e.g.* biodiversity or ES), conflicts with other land uses also drive the implementation of conservation actions and might reduce the efficiency of the selected set of priority areas (Faleiro et al., 2013; Knight and Cowling, 2007; Williams et al., 2003). As such socio-economic factors might influence the success of conservation actions (Knight et al., 2009; Naidoo et al., 2006), they should therefore be included in the prioritization process.

Implementing conservation strategies faces several challenges, such as scarce economic resources and lack of policymaker interest (Knight et al., 2009; Martín-López et al., 2009). Thereby, delaying conservation actions is frequent in real world (Drechsler et al., 2011). As areas of high conservation values could be impaired, or even lost, by land use changes and not be suitable for conservation indefinitely, delaying actions might jeopardize the achievement of conservation objectives (Cabeza and Moilanen, 2006; A. Moilanen et al., 2009). Notably, land use changes might increase the total costs of conservation networks (*e.g.* area necessary to meet conservation targets) and reduce the efficiency of conservation planning (Jérôme Cimon-Morin et al., 2016; Fuller et al., 2007; Nori et al., 2013). Nevertheless, the consequences of delaying actions to conserve ES are poorly understood and hardly available to decision makers. This information is even more critic in tropical countries where the provision of ES has just begun to be modelled and understood (*e.g.* Manhães et al., 2016), although they suffer an intense pressure by agricultural expansion (Laurance et al., 2014).

In this study, we evaluated the consequences of delaying conservation actions to safeguard ES in the Cerrado, a region severely threatened by agriculture expansion in Brazil. More specifically, we investigated the impact of future land use changes in the priority areas selected to conserve multiple ES in the study region. We explored the consequences of minimizing altered environments in the prioritization process, thereby optimizing the selection of areas where human influence is lower. Our study might assist the planning and formulation of conservation policies to the Cerrado and other regions worldwide that suffer intense land use change pressure.

Methods

Study area

We focused our analyses in the Cerrado, which is the second largest biome in South America, covering nearly 200 million hectares (*i.e.* 24% of the Brazilian territory). The Cerrado is one of the global biodiversity hotspot and the most biodiverse tropical savanna in the world (Klink and Machado, 2005; Myers et al., 2000). It includes different vegetation types, such as forests, savannas and grassland, and harbor *ca.* 12 thousand plant species, from which 44% are endemic (Forzza et al., 2010; Martinelli and Moraes, 2013).

Natural features of the region (*e.g.* flat relief adequate to mechanization and high availability of water), associated with political incentives, have boosted agricultural activities in the Cerrado. Between 2013 and 2015, for example, about 1.9 million of hectares were converted to agriculture in the region (MMA, 2014a). This deforestation rate (~ 1% of the Cerrado area per year) is the highest one among Brazilian biomes and five times higher than deforestation rate in Amazon (Strassburg et al., 2017). Although the network of protected areas (PAs) in the Cerrado is effective to reduce deforestation, it covers only 8% of the region (Carranza et al., 2014; Françoso et al., 2015), a portion lower than Amazon (24% of its surface, considering only federal and state PAs; Ribeiro et al. 2016). The limited coverage of PAs network in the Cerrado, associated with the relative low protection required inside private lands (*e.g.* 20% of private lands must be set aside for conservation in Cerrado, whilst in Amazon it is 80%; Brancalion et al. 2016), contributed to made vast areas available to agriculture expansion.

Currently, 46% of the native vegetation of the Cerrado was deforested, and 40% of the remaining vegetation can be legally converted to other land uses (Soares-Filho et al., 2014; Strassburg et al., 2017). Following land use changes projections, 1,140 threatened endemic plant species might go extinct by 2050 in the region, a number eight times higher than the plant extinctions rate recorded since the 16th century worldwide (Strassburg et al., 2017). The land use changes may also impact ecosystem functions and ES provision of the Cerrado, including changes in the hydrological cycle (Spera et al., 2016) and impairment of water quality (Hunke et al., 2015). Furthermore, carbon storage and water provision could reduce dramatically following legal conversion of native areas (Vieira et al., 2017). The fast change in the Cerrado might impair quality of life of its inhabitants. More than 29 million of people live in urban areas located in the Cerrado (IBGE, 2010) and other 25 million, including indigenous people and other minority groups, rely their livelihoods on raw material extraction or in small scale farming (Sawyer et al., 2016). Nonetheless, benefits provided by the Cerrado ecosystems go far beyond its area, mainly due freshwater provided to other regions and climate change mitigation, as discussed below.

Land use change model

To forecast the land use changes, we used outputs from OTIMIZAGRO land use model (~ 0.5 x 0.5 km; Soares-Filho et al. 2016). This model simulates the expansion of pasture, nine temporary and five perennial crops in Brazil to the coming decades based on official historical record and projected trends of the agriculture production. To define the future land use maps, the model considers the climatic suitability of each crop, historical pattern of development of the landscape and current and planned transportation network (*e.g.* roads and ports). The land use changes follow the enforcement of the Brazilian Native Vegetation Protection Law (NVPL – Lei de Proteção da Vegetação Nativa, in Portuguese; Brancalion et al. 2016), thus consider only legal deforestation. To our knowledge, it is the most updated and reliable projection of land use changes in Brazil.

Using the land use model, we use the currently available land use map for the present (2012) and generated two future time steps (2025 and 2050). For this purpose, we aggregated crops in annual and perennial agriculture classes. However, we maintained soybeans, corn and sugar cane as independent categories since these crops cover the highest extension (*i.e.* 70%) of the Cerrado's agricultural area. Thereby, we ended up with nine anthropogenic classes: five agricultural classes (*i.e.* annual, perennial, soybeans, corn and sugar cane), pasturelands, silvicultural fields, urban areas and mining pits (Table S1). As the land use model's outputs include only two classes of native vegetation (*i.e.* savanna and forest), we used the database from Ministry of Science, Technology and Innovation (MCTi, 2010) to refine the spatial classification of native vegetation. We were able to distinguish nine native vegetation types distributed among forest, savanna and grassland ecosystems (MMA, 2010). We ended up with 21 land use types at a resolution of 1 km^2 (Table S1).

Ecosystem services

We mapped six ES provided by the Cerrado in the present and future time steps: water yield, sediment and nutrient retention, carbon storage, net primary productivity and wild food provision (Fig. 1). To map these ES, we used InVEST v.3.3.3 (Integrated Valuation of Ecosystem Services and Tradeoffs) or environmental database obtained from secondary data sources (Martínez-Harms and Balvanera, 2012). InVEST is a software that includes a suit of models for mapping several ES from land use maps and biophysical data (Sharp et al., 2016). This software is useful to inform how land use changes alter the provision of ES and assist the selection of sites able to benefits humans and nature simultaneously (Kareiva et al., 2011). The ES were mapped in an equal grid of 0.1^{o} latitude/longitude (~ 11 x 11 km, or 121 km^2, near to the Equator), composed by 18,246 cells covering the entire Cerrado. To choose the grid cells size, we considered the Cerrado extension and the size of PAs in the biome, as discussed below.

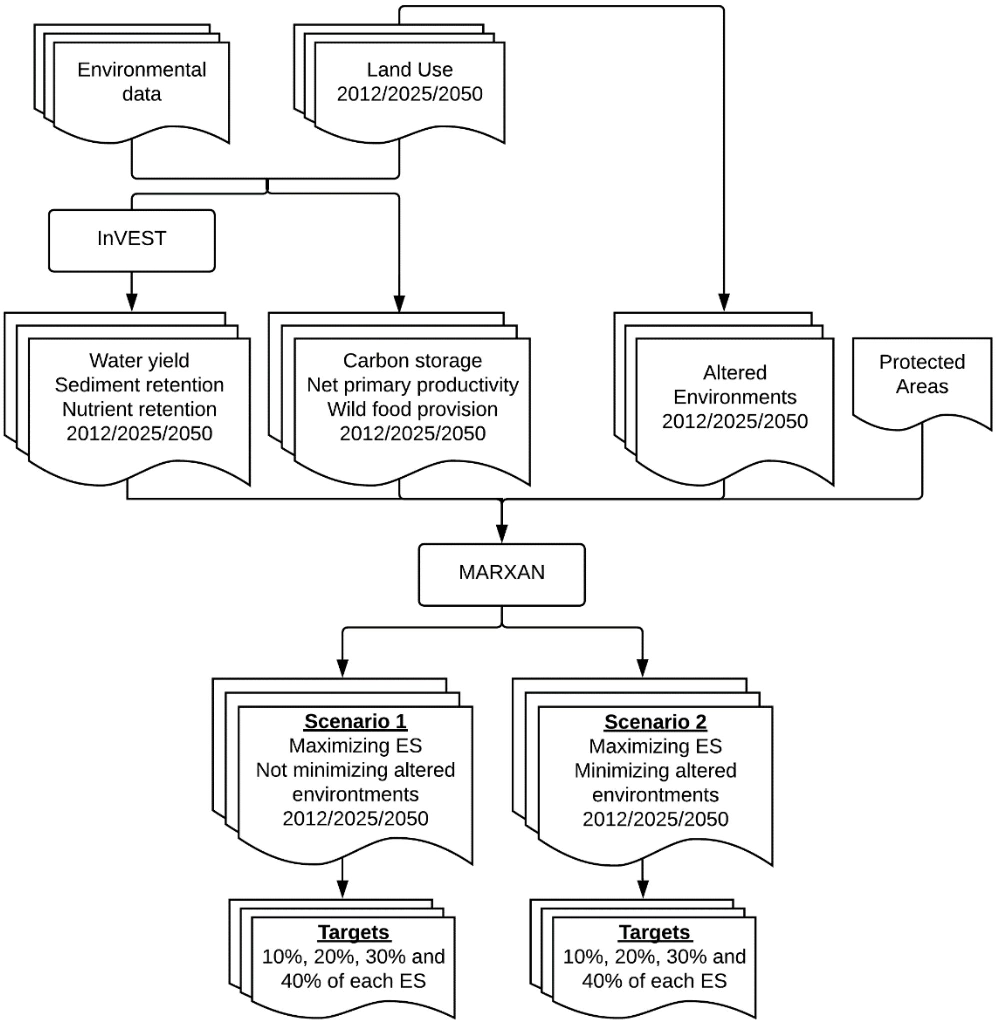

Figure 1: Schematic representation of the steps followed for identifying networks of priority areas for safeguarding ecosystem services in the Cerrado. Using environmental database (abiotic and biotic data) and land use maps for the present and future, we modelled six ecosystem services for 2012, 2025 and 2050. We used these six ecosystem services, as well as altered environments and protected areas distribution to set priority areas considering two scenarios: minimizing altered environments and not minimizing these environments. Each scenario includes priority areas settled to 2012, 2025 and 2050 to meet four conservation targets in each time step: representation of at least 10%, 20%, 30% and 40% of each ecosystem services. See Methods section for further details.

The six ES provide benefits at different spatial scales (local, regional and global). Water yield, sediment and nutrient retention are key ES to guaranty water security in the Cerrado and other regions of Brazil and South America. Waters from the Cerrado fed eight of the 12 Brazilian watersheds (Overbeck et al., 2015), including three of the biggest ones of South America. Carbon storage, which contributes to global climate regulation, represents a strategic ES for Brazil since the Cerrado is the region of highest levels of greenhouse gas emission due land use changes (MCTi, 2014). Net primary productivity is related to provision of raw materials (*e.g.* wood and fibers) for local people (Balvanera et al., 2006; MEA, 2005). Finally, wild plants represent a source of food and economic opportunities to local communities (Vieira et al., 2006).

We used InVEST Water Yield model to map water yield. This model calculates the average quantity of water produced by each part of the study region that flows to downstream areas, which represents the precipitation that does not undergo evapotranspiration (Sharp et al., 2016). To calculate the actual evapotranspiration, the model considers soil variables, such as plant available water content and root restricting layer depth, and vegetation proprieties, such as root depth (see Supplementary Material; Table S1). We used the InVEST output (~ 1 x 1 km) to calculate the value of water yield in mm $cell^{-1}$.

To map sediment retention, we used the InVEST Sediment Delivery Ratio model, that uses the revised universal soil loss equation to assess the amount of sediment generated and delivered to the streams by overland process (Sharp et al., 2016). To define the importance of the land cover to avoid loss of sediment, the model compares the difference of sediment loss between the input data and a hypothetical watershed covered only by bare soil (*i.e.* without vegetation covering). That model relies on data such as rainfall erosivity, soil erodibility and sediment retention efficiency of each land cover class (see Supplementary Material; Table S1). The sediment retention map was represented in t $cell^{-1}$.

To map nutrient retention, we used InVEST Nutrient Delivery Ratio model. Based on the mass balance approach, the model calculates the amount of nutrient produced by each part of the study region that reaches streams or that is retained by vegetation or soil (Sharp et al., 2016). To map the nutrient retention, the model uses slope, retention efficiency of each land use class and position of the pixel in the water flow (see Supplementary Material; Table S1). The nutrient retention was mapped using an index that varies from 0 to 1, where grid cells with values close to one are more efficient to retain nutrients.

We used secondary data to map carbon storage, net primary productivity and wild food provision. For carbon storage, we associated data from above- and belowground biomass to each land use class. A more refined carbon storage data is available for native vegetation. Thereby, we considered 28 types of native vegetation to map this ES according to the database provided by MCTi (2010). We associated the carbon storage of each land use class consulting the lookup table of MCTi (2010), which is the official database used by the Brazilian government to account for the national emission of greenhouse gases. We set carbon storage to silvicultural class considering the proportional area of *Pinus* and *Eucalyptus* found in each Brazilian state included in the Cerrado (MCTi, 2010). For simplicity, we assumed that above- and belowground carbon storage of water bodies, urban areas and mining was equal to zero. We calculated the carbon storage in ton C cell^{-1}.

To map net primary productivity, we used the database derived from MOD17 algorithm and processed by LAPIG-UFG (~ 1 x 1 km; www.lapig.iesa.ufg.br/lapig). MOD17 algorithm was design for the MODIS sensor (www.ntsg.umt.edu/project/mod17) and calculates the net primary productivity discounting respiration losses from the gross primary productivity of land surface. Respiration losses are calculated as daily leaf and fine root maintenance respiration, annual growth respiration and annual maintenance respiration of live cells in woody tissue. While gross primary productivity is derived from absorbed Photosynthetically Active Radiation and a conversion efficiency parameter that varies according to vegetation types and climate conditions. The database used represents the annual average of net primary productivity from 2000 to 2012. We calculated the net primary productivity in kg C cell^{-1}.

We mapped the wild food provision using the spatial distribution of 16 wild edible plant species, which are commonly used by local people as source of food and monetary income (Vieira et al., 2006; Table S2). We used occurrence records of the 16 species compiled by Oliveira et al. (2015) to build a presence-absence matrix. Using this matrix, we calculated the proportion of distribution area of each species that fall within each grid cell. Then, we calculated the mean proportion of the 16 species distribution that fall within each grid cell. We assumed that grid cells with higher mean proportion of species distribution are more important to the wild food provision.

Finally, we mapped each ES to 2012, 2025 and 2050 considering the land use map correspondent to each time step. For water yield, sediment and nutrient retention, we ran InVEST models separately to each time step. For carbon storage, we associated the above and below storage values of each land use class to the land use maps of each time step. We mapped the net primary productivity of the present (2012) using the annual average net primary productivity from 2000 to 2012. For the future time steps, we associated the average net primary productivity of each land use class in the present to the future land use maps. To reduce uncertainties in future net primary productivity maps, we calculated the average net primary productivity of land use classes for each 3rd-order watershed of the Cerrado independently. Watershed database were obtained from National Water Agency of Brazil (ANA; www.ana.gov.br). For wild food provision, we assumed that grid cells without native vegetation in the present or in the future would not provide wild food service.

Spatial prioritization

We used Marxan v2.43 to assemble priority areas to represent ES in the Cerrado. Marxan is a well-recognized conservation planning software that uses heuristic simulated annealing coupled with iterative improvement to provide efficient spatial solutions to nature conservation (Ball et al., 2009). This software uses an objective function that penalizes networks of priority areas with high cost or with low level of connection, while it achieves the representation of a set of conservation features (Ardron et al., 2010).

We selected networks of priority areas defining four representation targets, which are 10%, 20%, 30% and 40% of each ES (Fig. 1). We used the amount of ES supply corresponding to each target in 2012 to set conservation targets for 2025 and 2050. We followed Game and Grantham (2008) to calibrate the “boundary length modifier”, which regulates the trade-off between total boundary length of the network of priority areas and their total cost. We also adjusted the “species penalty factor” to guarantee targets achievement of each ES.

To achieve each conservation target, we settled priority areas as complement to the actual network of PAs. Therefore, actual PAs of the Cerrado were forcibly included in all priority areas networks. To define the spatial distribution of actual PAs, we used the database of federal PAs from the Brazilian Ministry of Environment (MMA; mapas.mma.gov.br/i3geo) and of state and municipal PA from the Brazilian Electricity Regulatory Agency (ANNEL; www.aneel.gov.br). We used the median area of actual PA in the Cerrado (*i.e.* 96 km^2) to guide the choice of the planning units' size, which corresponded to cells of the grid with ~ 121 km^2. It follows that, 44.86% of the PAs are bigger than the planning unit size. We only included planning unit in PAs networks if at least 55% of the planning unit surface area was covered by PA (see Ribeiro et al., 2016). Using this criterion, we ended up with 109 PA distributed in 1,344 planning units, which is similar to the total area covered by the actual PAs network in the Cerrado (~ 7.1% of the Cerrado's area).

To evaluate the consequences of altered environments in the prioritization process, we considered two scenarios: minimizing altered environments and not minimizing these environments (Fig. 1). In the former scenario, the proportion of altered environments inside each PU was used as a proxy of its cost (Faleiro et al., 2013). Whilst, in the latter scenario, we did not integrated cost into the spatial prioritization, thus cost was equal to one in every planning units (Naidoo et al., 2006). We considered altered environments as land use classes that are not natural (*i.e.* agricultural fields, pasture lands, urban, and mining areas). Integrating such cost measure favors the selection of less disturbed planning units, which should reduce the conflict between conservation actions and other land use. Furthermore, such cost proxy was useful in our study, since we could compute a particular cost database to each time step. For both scenarios, we assembled networks of priority areas to four conservation targets and to three time steps (Fig. 1). It follows that we ended up with 12 networks of priority areas per scenario or 24 networks in total. We ran Marxan 100 times for each prioritization scenario and considered in the analyses only the solution with the best objective value among 100 runs.

To compare networks of priority areas, we extracted the total area, boundary length and amount of altered environments of each network. We also assessed the spatial congruence among networks using the Kendall rank correlation coefficient (Brum et al., 2017). We used Dutilleul's modified t-test to account for spatial autocorrelation and not overestimate the degree of freedom in significant correlation test (Legendre et al., 2002). We used mainly "Raster" and "Maptools" packages to handle spatial data and “stats” and “SpatialPack” packages to perform correlation analyses in R version 3.2.2 (R Development Core Team, 2015).

Results

Changes in land use and ecosystem services

The OTIMIZAGRO land use model predicted notable habitat loss for the upcoming years in the Cerrado, mainly in its center and northern regions (Fig. 2). The Cerrado might lose 40.3 million ha of native vegetation by 2050, reducing the native vegetation coverage from 48.8% in 2012 to 31.6% in 2050. The main drivers of land use changes are pasture lands and soybean fields, which are expected to cover 46.3% and 12.4% of the Cerrado by 2050, respectively.

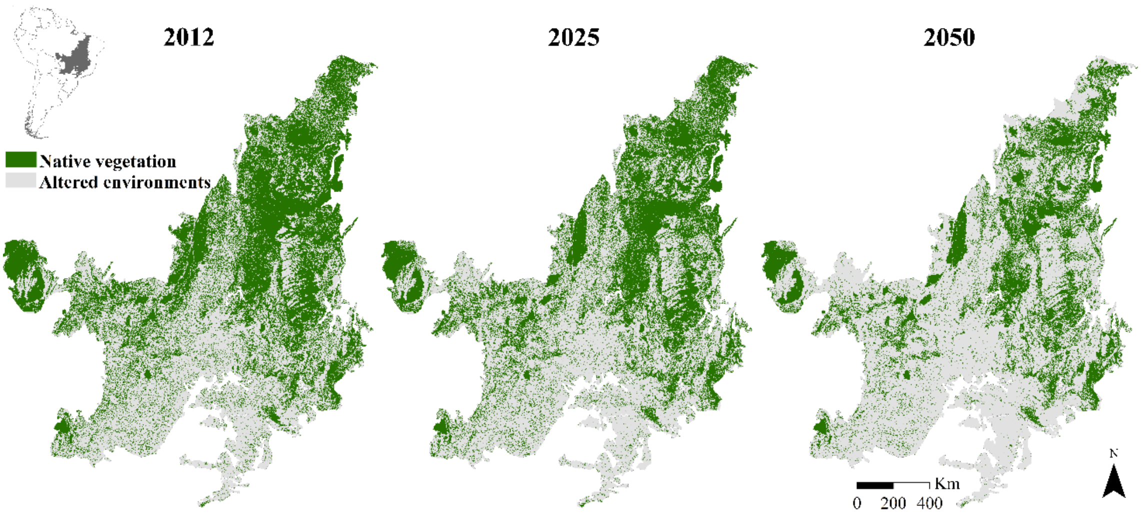

Figure 2: Land use change predicted to the Cerrado to the three evaluated time steps. Native vegetation is composed by remnants of forest, savanna and grassland ecosystems, whilst altered environments include pasture, agriculture and urban areas.

The ES provision is spread through the Cerrado and its spatial pattern varies among ES (Fig. S1). The spatial distribution pattern of ES provision tends to be similar between 2012 and 2050. However, the amount of ES provided by different region in the Cerrado is expected to change over time. Nutrient retention, carbon storage and net primary productivity provision are predicted to reduce throughout the Cerrado. Wild food provision is expected to reduce mainly in the southern portion of the study area. While, water yield and sediment retention provision tend to reduce in some areas diffusely distributed across the Cerrado.

Changes in spatial priorities

The amount of selected planning units was similar among networks of priority areas settled in 2012, 2025 and 2050. This pattern was constant among networks settled for both scenarios ("not minimizing altered environments" and "minimizing altered environments") and their respective conservation targets (10%, 20% 30% and 40%). For example, 10,7% (or 1,956 planning units) and 10,5% (1,923) of the Cerrado area was necessary to represent 10% of ES provision in the "minimizing altered environments" scenario in 2012 and 2050, respectively (Table S3). Nonetheless, postponing conservation actions is expected to change the distribution pattern of priority areas of both scenarios over time. In addition, the amount of altered environments inside priority areas is predicted to increase as result of postponing conservation actions.

The spatial distribution of priority areas tended to differ between 2012 and 2050, especially in the "not minimizing altered environments" scenario (Fig. 3 and 4). The congruence between present and future networks in the "not minimizing altered environments" scenario was low (Fig. 4), which is a consequence of change in the spatial distribution of ES provision. In another hand, the high congruence between present and future networks in the "minimizing altered environments" scenario comes from the oriented selection of priority areas to less disturbed regions (*i.e.* center and northern region of the Cerrado) (Fig. 3). In both scenarios, a higher similarity between present and future solutions was observed for lower conservation targets (Fig. 4), which occurs due to high overlapping between PA and priority areas at lower targets.

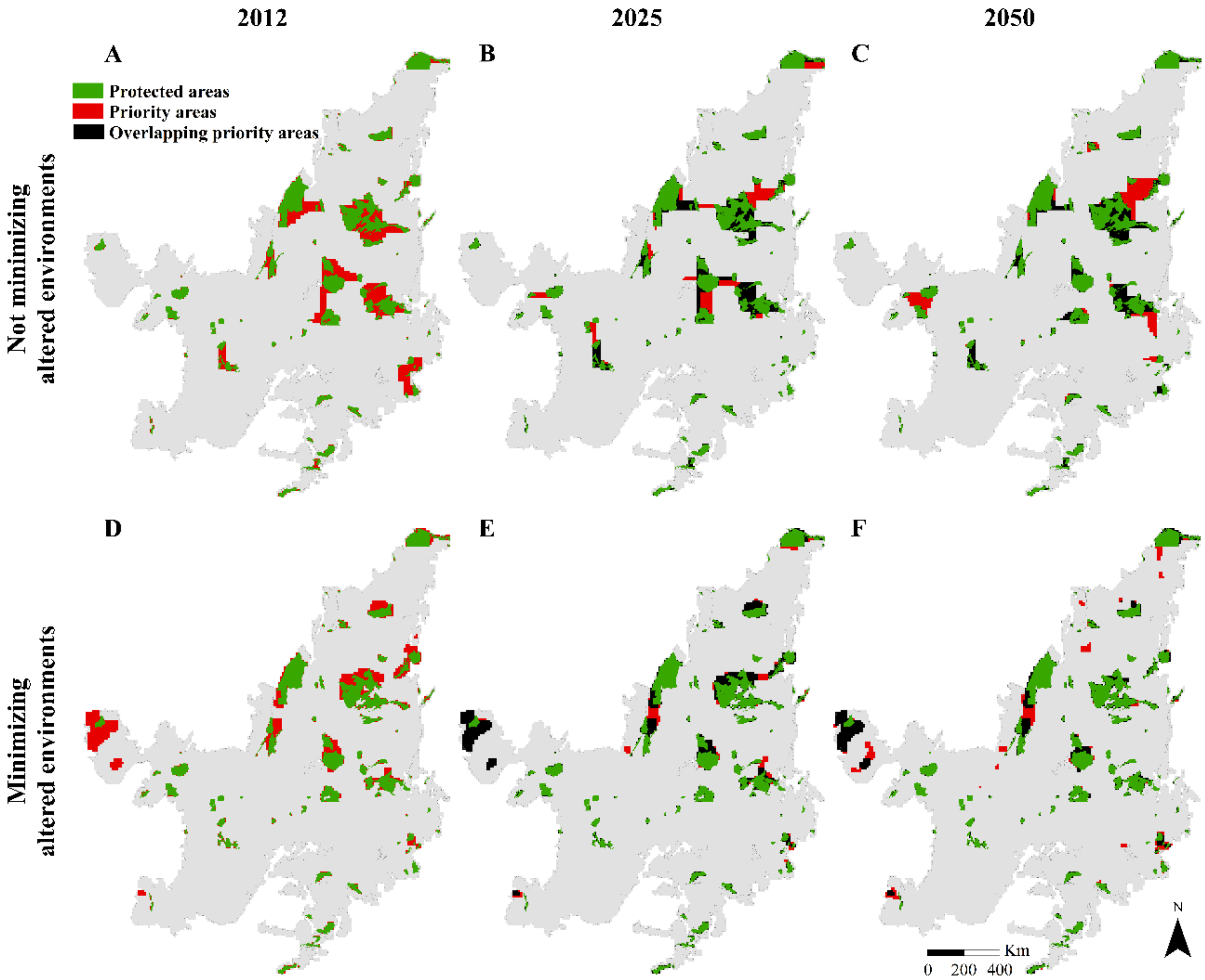

Figure 3: Networks of priority areas defined to protect 10% of each ecosystem services. In the "not minimizing altered environments" scenarios, the cost was defined equal to 1 to all planning units; while in the "minimizing altered environments" scenario, the cost was established as the proportion of altered environments inside each planning unit. Overlapping priority areas (black areas) represents planning units that were selected in future time steps (*i.e.* 2025 or 2050), as well as in the present.

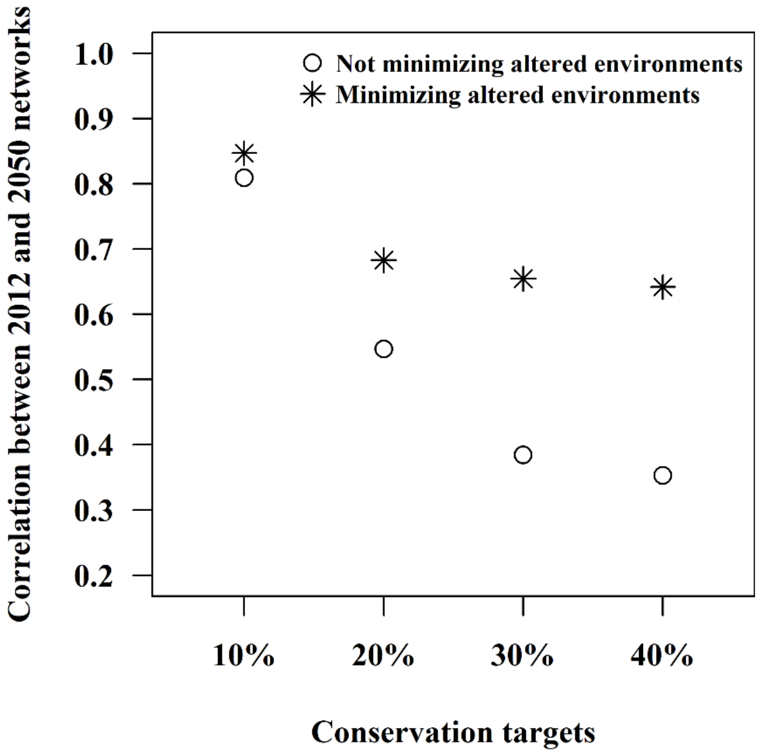

Figure 4: Correlation (Kendall rank coefficient) between networks of priority areas identified to represent ecosystem services in 2012 and 2050. All correlations were significant for $p < 0.01$.

The amount of altered environments inside priority areas increased remarkably over time in both scenarios, notably in the “minimizing altered environments” scenario (Fig. 5). Altered environments increased 79.8% between 2012 and 2050 (mean of the four conservation targets) in the “minimizing altered environments” scenario and 41.5% in “not minimizing altered environments” scenario. However, the total amount of altered environments inside priority areas was lower in the “minimizing altered environments” scenario at the three time steps. The increase of altered environments inside priority areas in the “minimizing altered environment” scenario suggests that future land use changes in the Cerrado may restrict the design of networks with high cost-benefits.

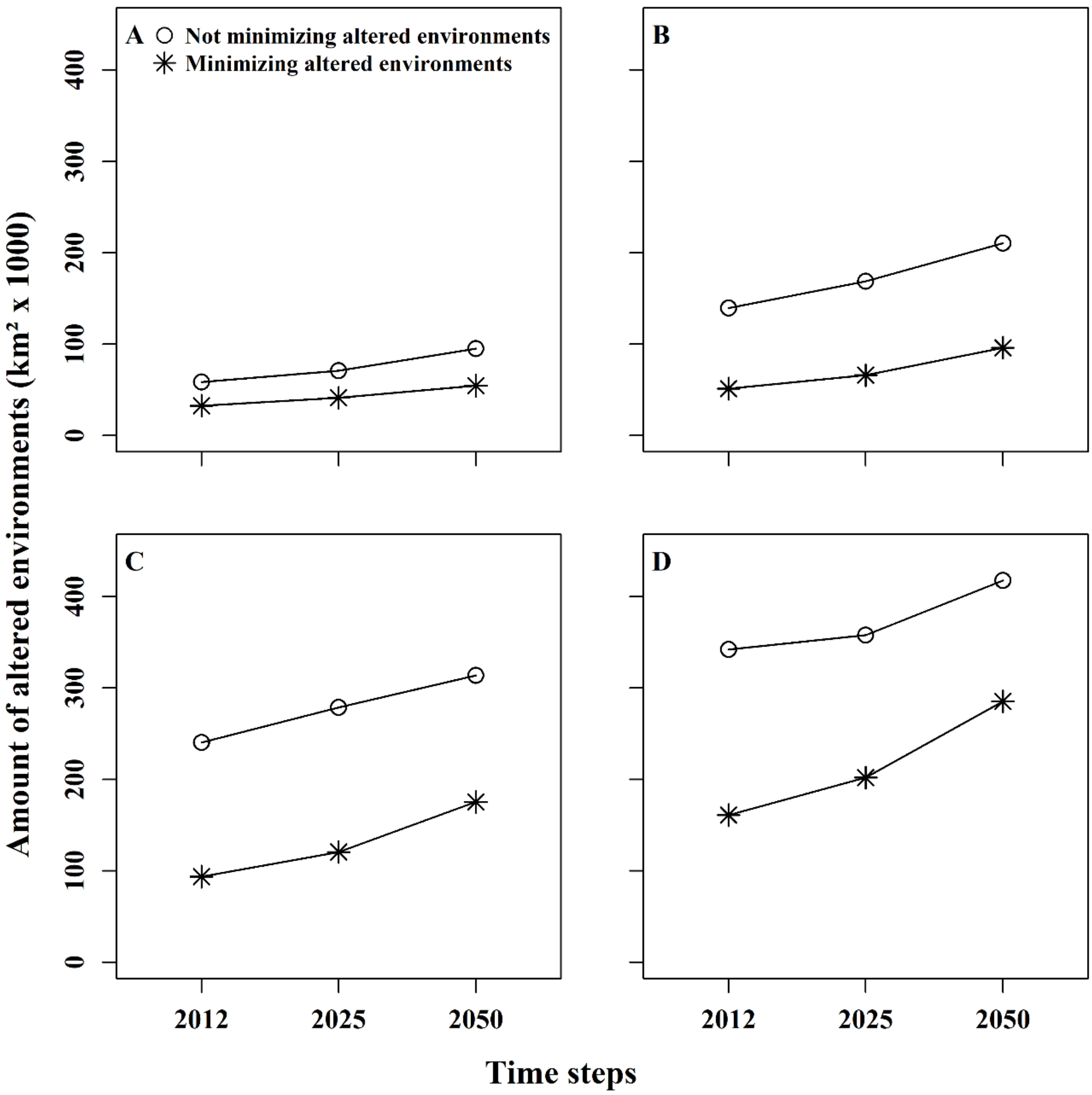

Figure 5: Amount of altered environments found inside priority areas network identified to represent ecosystem services in the present or future time steps. Each graph represents a conservation target: A) representation of at least 10% of each ecosystem services; B) 20%; C) 30% and D) 40%.

Discussion

Land use changes projected to the coming decades in the Cerrado will impose major challenges to conservation in this region. We found that delaying conservation actions will impair conservation of ES, since: i) ES provision is expected to reduce over time along with changes in their spatial distribution and ii) priority areas settled in the future will encompass greater amounts of altered environments than planning in the present. Even minimizing altered environments in the prioritization process, it may not be possible to select several less disturbed contiguous planning units in the future, given their decreasing availability.

Increases of altered environments inside selected priority areas in the future may contribute to increase conflicts between conservation actions and human activities. These conflicts may limit the establishment of conservation actions and their success. There is a plenty of evidences that conflicts between conservation actions and human activities have hindered national environmental policy and legislation in Brazil (Fearnside, 2016; Loyola, 2014), notably due to agribusiness lobby (Soares-Filho et al., 2014). For example, Bernard et al. (2014) identified 93 conflict events that changed the delineation and/or management of PA in Brazil between 1981 and 2012. The frequency of conflict events increased over time, such as 74% of them occurred from 2008 to 2012. In total, 7.3 million ha of PA were affected in the last three decades, mainly to enable the development of economic activities, such as agribusiness, real estate and infrastructure development (Bernard et al., 2014). Therefore, implementing ES conservation actions in the present may be more efficient and should face fewer conflicts with human activities than postponing them in the future.

Higher levels of altered environments inside priority areas may reduce the provision of other ES not assessed in this study. For example, the degradation of natural habitats inside PA may impact the development of recreation and ecotourism (Balmford et al., 2015; Sharp et al., 2016), and reduce revenues that could be used in conservation strategies (Balmford et al., 2015; Medeiros et al., 2011). Moreover, increases in altered environments may jeopardize efforts for conserving ES and biodiversity simultaneously. This is due to the fact that land use changes impair biodiversity, including negative effects on abundance and richness (Newbold et al., 2015), as well as functional and phylogenetic integrity (Banks-Leite et al., 2014). Therefore, delaying conservation actions undermine joint strategies to secure both biodiversity and ES in the same area (Cimon-Morin et al., 2013; Larsen et al., 2011; Naidoo et al., 2008).

We found that PA of the Cerrado poorly represents ES provided by the region. Although PA are located in areas less disturbed of the Cerrado (Françoso et al., 2015), they do not capture even the lower conservation target for representing ES. For example, the required area for meeting the 10% target in the “minimizing altered environment" scenario is 45.5% bigger than the area covered by the actual network of PAs. These results reinforce the need of conservation actions to maximize ES representation in the Cerrado.

To avoid loss of ES provision in the Cerrado over time, some strategies should be considered. First, the development of a comprehensive monitoring program of ES status over time would guide conservation actions in the biome (see Rodríguez et al., 2006). Although challenging, monitoring programs are under development in some regions around the world, such as in European Union via MAES initiative (Mapping and Assessment of Ecosystems and their Services; biodiversity.europa.eu/maes). There are also global initiatives fostered by the IPBES (Panel on Biodiversity and Ecosystem Services; www.ipbes.net), which aim a global assessment of biodiversity and ES. Second, social and environmental costs of development should be included in Brazilian accountings (*e.g.* by using the Genuine Progress Indicator), which would make visible the extension of natural capital loss in the Cerrado (see Andrade and Garcia, 2015). Using alternative indicators to Gross Domestic Product could foster a more sustainable development, in which social and environmental dimensions, beside only the economic one, could be considered in the decision making (Costanza, 2014). Third, reducing land use changes in the Cerrado is paramount to safeguarding ES provision. Strassburg et al. (2014) showed that increasing pastureland productivity in Brazil from 32-34% to 49-52% of the carrying capacity would free land to expand agricultural production to meet the demand of meet, crops, wood and biofuels by 2040 without more habitat loss. Successful initiatives to reduce land use changes in Amazon, such as the Soy Moratorium and the satellite imagery monitoring program, could also be expanded to the Cerrado (see Strassburg et al., 2017).

Our results are based on the mapping of ES and thus carry uncertainties associated to models used and assumptions made. Modeling ES through InVEST is limited by the availability of biophysical databases. However, the development of local studies in Brazil will improve the reliability of ES modelling and reduce uncertainties (Pires et al., 2017; Resende et al., 2013). Regarding carbon storage and net primary productivity modeling, our approach considered that only land use changes could impact ES provision, thus we neglected the influence of degradation and regeneration. Categorizing land use classes in different levels of degradation or regeneration could reduce uncertainties related to carbon storage and net primary productivity (Sun et al., 2017). Other factors, such as climate change, could also impact the provision of ES over time (Schirpke et al., 2017). Under climate changes, the loss of ES provision in the future might be worse than the levels reported in this study. We used the same climate variables in the present and future to maintain the comparability among time steps and assess separately the influence of land use changes.

Even with perceived caveats, we evaluated for the first time the influence of future land use changes in the provision of multiple ES in the Cerrado. Our study stands out because it differs from usual approaches that focus generally on a few ES and on spatial and temporal scales too narrow to influence land use policies (Nelson et al., 2009; Seppelt et al., 2011). Using a temporal scale of *ca*. 40 year and focusing in multiple ES in a global conservation priority biome, we showed that delaying conservation actions might impact the conservation of ES and increase conflicts between conservation actions and competing human activities. Therefore, we advocate that establishing conservation actions to safeguard ES in the Cerrado today will generate stronger conservation outputs than postponing conservation actions for the next decades.

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Supplementary material

Water supply service

The water yield model of InVEST (Sharp et al., 2016) is based on water balance and quantifies the water yield (Y_i) of each pixel i as a function of the annual actual evapotranspiration (AET_i) and the annual precipitation (P_i):

$$Y_i = \left(1 - \frac{AET_i}{P_i}\right) \times P_i$$

When pixel i is covered by vegetation, the quotient $\frac{AET_i}{P_i}$ is defined as follows:

$$\frac{AET_i}{P_i} = 1 + \frac{PET_i}{P_i} - \left[1 + \left(\frac{(PET_i)}{P_i}\right)^{\omega}\right]^{\frac{1}{\omega}}$$

where PET_i is the potential evapotranspiration and ω is a parameter that defines the natural climatic-soil properties. ω is calculated as function of the seasonal pattern of rainfall and plant available content of the soil (Donohue et al., 2012), which is defined as the proportion of water stored in the soil profile that is available for plants use. Meanwhile, PET_i is calculated as:

$$PET_i = K_c(l_i) \cdot ET_{0,i}$$

where $K_c(l_i)$ is the vegetation evapotranspiration coefficient associated to the land use l covering pixel i and $ET_{0,i}$ is the reference evapotranspiration of the pixel i. The value of K_c is defined by plant physiological characteristics and $ET_{0,i}$ is the evapotranspiration of a grass reference crop, its value depends on climatic properties (Allen et al., 1998). More information about the model is available in Sharp et al. (2016) and Natural Capital Project website (www.naturalcapitalproject.org/invest/).

To run the water yield model, we obtained average annual precipitation from WorldClim dataset (~ 1x1 km; www.worldclim.org), average reference evapotranspiration (1950-2000) from CGIAR-CSI (~ 1x1 km; csi.cgiar.org/Aridity/) and watershed limits from National Water Agency of Brazil (ANA) (www.ana.gov.br).

Following Manhães et al. (2016), we obtained both plant available water content and root restricting layer depth (*i.e.* soil depth in which root penetration is inhibited by soil characteristics) consulting Harmonized World Soil Database (HWSD; Hiederer and Köchy, 2012) and associated the values of both variables to the Brazilian soil map (IBGE, 2001). To define the root depth of each land cover (*i.e.* where 95% of the root biomass occur for each land cover), we used values from literature, which were revised by specialists afterward. As Kc values are found just to crops and is virtually absent for tropical vegetation, we used Kc = 1 to all land use classes (Manhães et al., 2016).

Sediment retention service

The sediment delivery ratio model of InVEST (Sharp et al., 2016) calculates the amount of soil loss ($usle_i$) per pixel i using the revised universal soil loss equation:

$$usle_i = R_i \cdot K_i \cdot \mathrm{LS}_i \cdot \mathrm{C}_i \cdot \mathrm{P}_i$$

where R_i is rainfall erosivity, K_i is soil erodibility, LS_i is slope length-gradient factor, C_i is crop-management factor and P_i is support practice factor. The amount of sediment exported per pixel i is calculated as follows:

$$Export_i = usle_i \cdot \mathrm{SDR}_i$$

where SDR_i represents the proportion of sediment from pixel i that reaches the catchment outlet. To each pixel:

$$SDR_i = \frac{SDR_{max}}{1 + \exp\left(\frac{IC_0 - IC_i}{k}\right)}$$

where SDR_{max} is the maximum theoretical value of SDR_i and IC_i is the connectivity index. Both IC_0 and k are calibration parameters used to set the relationship between SDR_i and IC_i. SDR_{max} is defined as 0.8 in the model (Vigiak et al., 2012) and IC_i is calculated based on proprieties of the upslope contributing area (*i.e.* area, average C factor and slope gradient) and flow path toward streams (*i.e.* length of the flow path, average C factor and slope gradient).

To assess the avoided sediment loss at the pixel-level, the model defines the sediment retention index ($index_i$) and compares the avoided sediment loss by the land use covering the pixel i with a hypothetical scenario in which the pixel is bare soil:

$$index_i = R_i \cdot \ K_i \cdot \mathrm{LS}(1 - C_i P_i) \cdot \mathrm{SDR}_{\ i}$$

See more details about the model in Sharp et al. (2016) and Natural Capital Project website (www.naturalcapitalproject.org/invest/).

As the input variables, we calculated the rainfall erosivity using multiple regressions developed by Mello et al. (2013). These authors used data from pluviographic stations to develop a suit of equations that allow calculate the rainfall erosivity of any location in the country using data of longitude, latitude and altitude. We obtained the soil erodibility consulting the review available in Da Silva et al. (2011) and associated the values of each soil type to the spatial distribution of soils in the Cerrado (IBGE, 2001)**.** The slope length-gradient factor was calculated automatically by InVEST using the digital elevation model (DEM) that we obtained from WorldClim dataset (~ 1x1 km; www.worldclim.org/current). The slope length-gradient factor is dimensionless and steep terrains have higher values. We obtain both crop-management and support practice factors of each land use class of the Cerrado revising literature (Table S1). These values range from 0 to 1 and values close to 1 indicate management and support practices that generate high soil loss.

Nutrient retention service

The nutrient delivery ratio model of the InVEST (Sharp et al., 2016) calculates the amount of nutrient exported ($export_i$) by pixel i using the nutrient load ($load_i$) associated to each land use and the nutrient delivery ratio NDR_i:

$$export_i = load_i \cdot NDR_i$$

NDR_i provides the proportion of nutrients from pixel i that reaches the catchment outlet and is defined as:

$$NDR_i = NDR_{0,i}\left(1 + \exp\left(\frac{IC_i - IC_0}{k}\right)\right)^{-1}$$

where $NDR_{0,i}$ represents the ability of downstream pixels to transport nutrient along the waterflow, IC_i is a topographic pixel that characterizes the position of the pixel in the waterflow, and both IC_0 and k are calibration parameters. $NDR_{0,i}$ is calculated according to the retention efficiency of the downslope path, which varies according to the land use classes. IC_i is calculated based on properties of upslope contributing area (*i.e.* area and average slope gradient) and downslope path (*i.e.* length of the flow and average slope gradient).

The model provides the amount of nutrient retained per watersheds and a map of NDR_i values. Pixels with lower NDR_i are more efficient to mitigate the loss of nutrient. More information in Sharp et al. (2016) and Natural Capital Project website (www.naturalcapitalproject.org/invest/).

We ran the nutrient delivery ratio model using the same DEM, annual precipitation and watershed limits defined to the other InVEST models. We obtained the nutrient load and the retention efficiency to each land use class consulting literature (Table S1). The model allows define the proportion of nutrient that are delivered via surface or subsurface flows, but due the lack of knowledge to the study area we followed the guidance and assumed that all nutrient flows via surface only (Sharp et al., 2016). We ran the model to phosphorus and nitrogen and mapped the nutrient retention service at the pixel-level averaging 1-NDR_i of both nutrients.

Table S1: Land use classes and their respective biophysical variables used in InVEST.

Land use classes	C Factor[1]	P Factor[2]	Root depth[3]	Load N[4]	Load P[4]	Eff. N[4]	Eff. P[4]	Critical length P and N[5]
Pasturelands	0.03	0.85	1000	5412.5	101.84	0.25	0.25	1000
Soybeans	0.0899	0.5	950	10767.5	1262.75	0.25	0.25	1000
Corn	0.088	0.5	1350	10767.5	1262.75	0.25	0.25	1000
Sugar cane	0.26	0.5	1600	10767.5	1262.75	0.25	0.25	1000
Annual agriculture	0.172	0.5	750	10767.5	1262.75	0.25	0.25	1000
Perennial agriculture	0.219	0.5	1250	10767.5	1262.75	0.5	0.5	1000
Silvicultural fields	0.01	1	1250	10767.5	1262.75	0.75	0.75	1000
Ombrophilous forest	0.001	1	1500	2190	142.57	0.9	0.9	75
Seasonal forest	0.01	1	3700	2190	142.57	0.85	0.85	75
Forested savanna	0.01	1	3700	2190	142.57	0.9	0.9	75
Wooded savanna	0.04	1	7000	1500	115	0.7	0.7	100
Park savanna	0.04	1	1500	1500	115	0.7	0.7	100
Steppic savanna	0.04	1	1500	1500	115	0.7	0.7	100
Gramineous-woody savanna	0.042	1	1000	1000	90	0.5	0.5	100
Mountainous vegetation	0.042	1	500	1000	90	0.5	0.5	100
Savanna wetland	0.042	1	1000	1000	90	0.5	0.5	100
Urban areas	0.06	0.98	1	5812.63	216.08	0.0496	0.0638	1000
Mining pits	1	1	1	5812.63	216.08	0.05	0.05	1000
Water bodies	0.07	1	1	2601.42	161.37	0	0	1000
Bare land	1	1	1	5812.63	216.08	0.05	0.05	1000
Non-identified	1	1	1	1	1	0.01	0.01	1000

[1] References to crop-management factor (C): Bertol et al. (2001), De Maria and Lombardi Neto (1997), Duarte et al. (2016), Farinasso et al. (2006), Kennedy et al. (2016), Manhães et al. (2016), Oliveira et al. (2011), and Silva et al. (2007).

[2] References to support practice factor (P): Duarte et al. (2016), Farinasso et al. (2006), Kennedy et al. (2016), Manhães et al. (2016), Oliveira et al. (2011), Silva et al. (2007), and Yang et al. (2003).

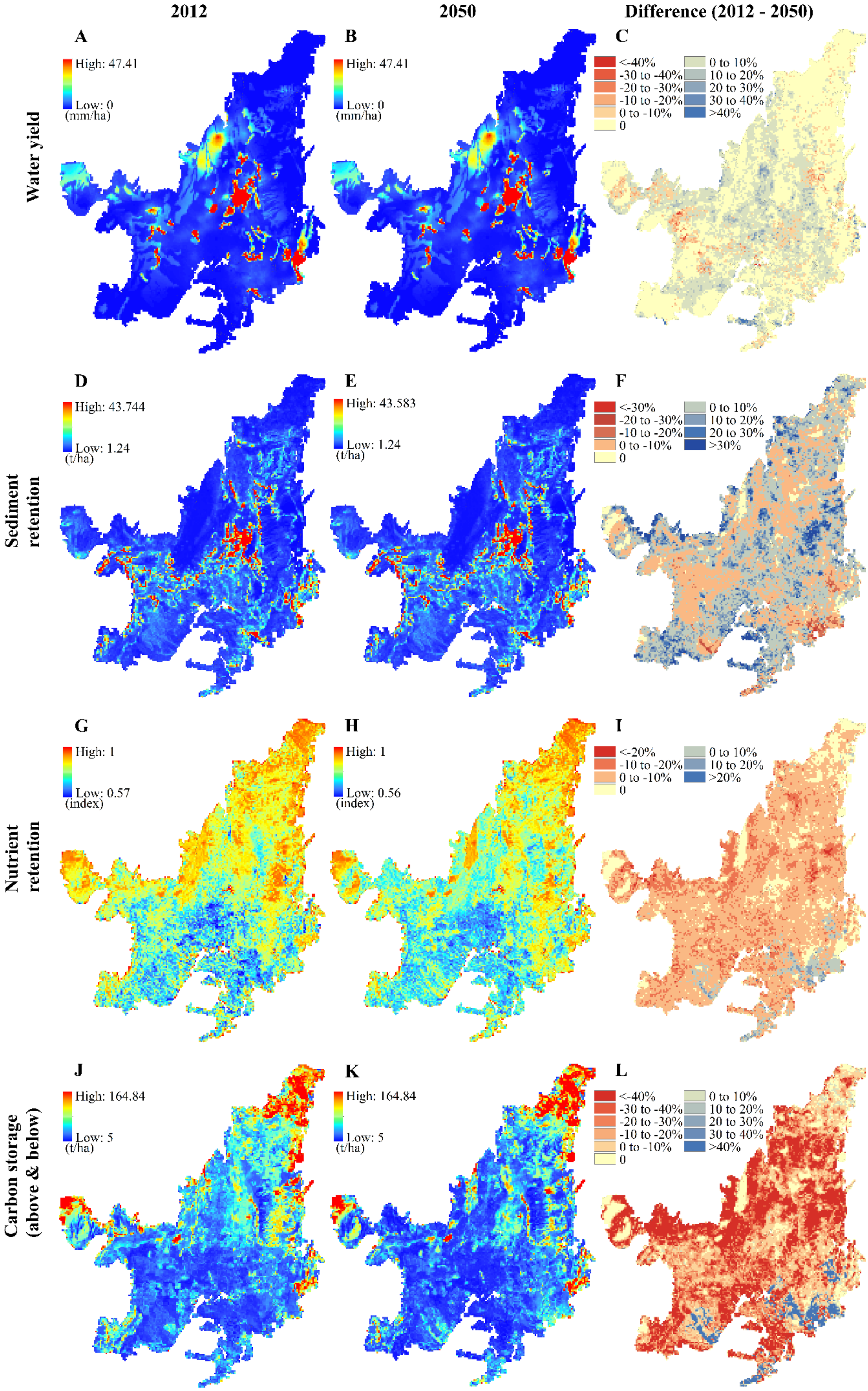
[3] References to define root depth: Manhães et al. (2016), Oliveira et al. (2005), Rodin (2004), and Castro and Kauffman (1998).

[4] References to define load and retention efficiency: Kennedy et al. (2016), and Manhães et al. (2016).

[5] References to define critical length: Mayer et al. (2007).

Table S2: Wild edible plants used in this study and their respective growth form, traditional use and commercial importance. Based on Vieira et al. (2006).

Scientific name	**Common name**	**Growth form**	**Traditional use importance**	**Commercial importance**
Anacardium othonianum Rizzini	Caju	Tree	High	High
Ananas ananassoides (Baker) L.B.Sm.	Abacaxi do cerrado	Herb	Low	Low
Annona coriacea Mart.	Araticum	Tree	Medium	Medium
Butia capitata (Mart.) Becc.	Coquinho	Palm tree	High	Medium
Byrsonima verbascifolia (L.) DC.	Murici	Tree	Medium	Medium
Campomanesia adamantium (Cambess.) O.Berg	Gabiroba	Shrub	High	Medium
Caryocar brasiliense Cambess.	Pequi	Tree	High	High
Dipteryx alata Vogel	Baru	Tree	Medium	High
Eugenia dysenterica (Mart.) DC.	Cagaita	Tree	Low	Low
Eugenia klotzschiana O.Berg	Pêra do cerrado	Tree	Low	Low
Genipa americana L.	Jenipapo	Tree	Low	Medium
Hancornia speciosa Gomes	Mangaba	Tree	Medium	High
Hymenaea stigonocarpa Mart. ex Hayne	Jatobá	Tree	Low	Low
Mauritia flexuosa L.f.	Buriti	Palm tree	High	Low
Passiflora setácea DC.	Maracujá do cerrado	Climber	Low	Medium
Psidium guianeense Sw.	Araçá	Shrub	Low	Medium



2012
2050
Difference (2012 - 2050)
A
B
C
Water yield
High: 47.41
Low: 0
(mm/ha)
High: 47.41
Low: 0
(mm/ha)
<-40%
-30 to -40%
-20 to -30%
-10 to -20%
0 to -10%
0
0 to 10%
10 to 20%
20 to 30%
30 to 40%
>40%
D
E
F
Sediment retention
High: 43.744
Low: 1.24
(t/ha)
High: 43.583
Low: 1.24
(t/ha)
<-30%
-20 to -30%
-10 to -20%
0 to -10%
0
0 to 10%
10 to 20%
20 to 30%
>30%
G
H
I
Nutrient retention
High: 1
Low: 0.57
(index)
High: 1
Low: 0.56
(index)
<-20%
-10 to -20%
0 to -10%
0
0 to 10%
10 to 20%
>20%
J
K
L
Carbon storage (above & below)
High: 164.84
Low: 5
(t/ha)
High: 164.84
Low: 5
(t/ha)
<-40%
-30 to -40%
-20 to -30%
-10 to -20%
0 to -10%
0
0 to 10%
10 to 20%
20 to 30%
30 to 40%
>40%

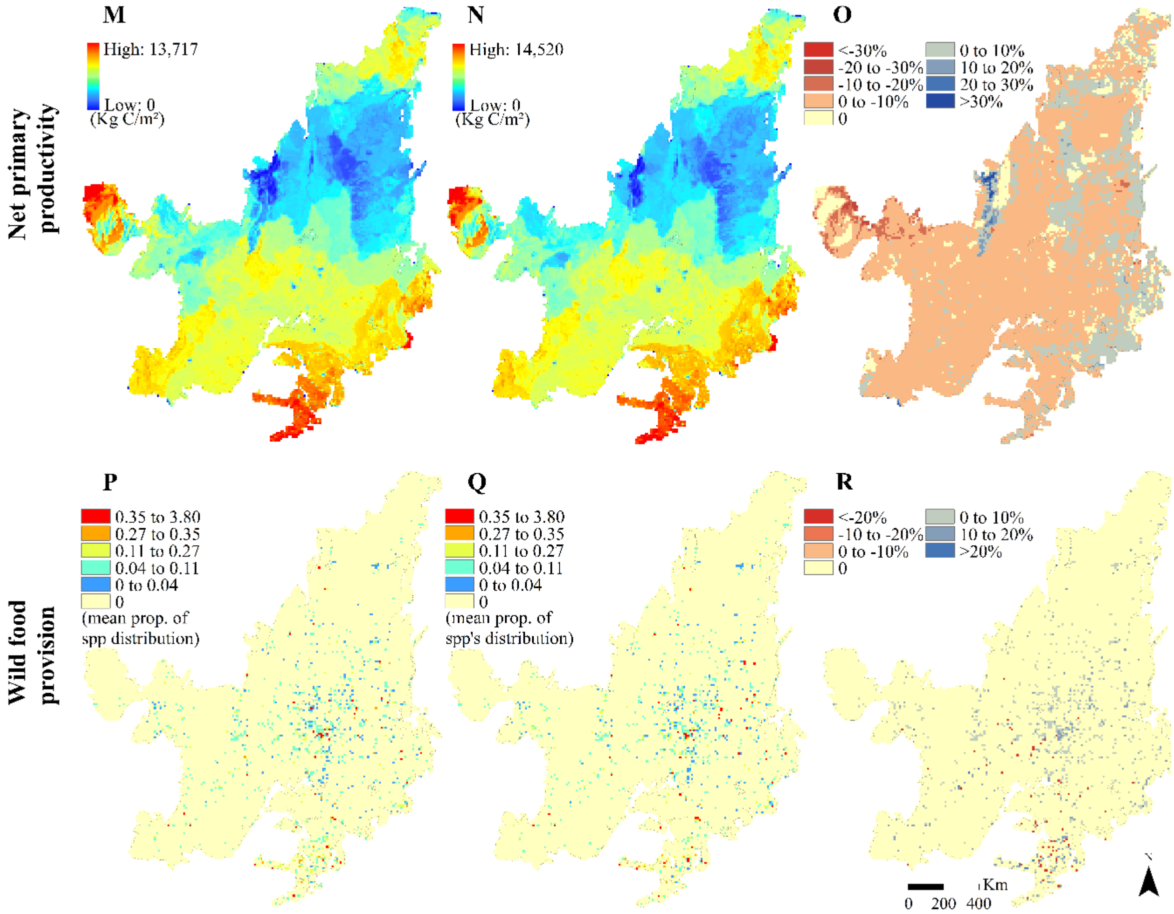


Figure S1: Spatial distribution of ecosystem services in 2012 (A, C, E, G, I and K) and difference in the provision between 2012 and 2050 (B, D, F, H, J and L).

Table S3: Information about networks of priority areas identified to represent ecosystem services. Target achievement equal to 100 means that 100% of the respective target was achieved to the given ecosystem service. Mean of percentage target achieved was calculated considering the six ecosystem services. PU: planning unit; NPP: net primary productivity; PA: protected area.

	Target	Area (number of PUs)	Altered environments ($km^2 \times 10^3$)	Perimeter (number of PUs sides x 10^6)	% of target achieved						
					Mean	Water yield	Sediment retention	Nutrient retention	Carbon storage	NPP	Wild food provision
2012											
Not minimizing	10%	2,029	58.46	13.59	166	260	201	114	121	100	203
altered	20%	3,896	139.37	20.44	138	194	159	107	114	100	156
environments	30%	5,643	240.35	27.06	128	163	145	105	107	100	145
	40%	7,374	341.97	27.42	117	132	127	102	100	100	140
Minimizing	10%	1,956	32.69	16.82	161	264	190	112	145	100	155
altered	20%	3,958	51.31	27.09	139	205	142	115	163	100	107
environments	30%	5,887	93.54	35.06	131	177	131	114	164	100	100
	40%	7,728	160.90	38.95	126	159	130	111	155	100	100
2025											
Not minimizing	10%	2,056	70.79	13.92	162	238	203	114	113	100	203
altered	20%	4,005	168.78	20.30	137	195	153	106	103	100	163
environments	30%	5,709	278.65	26.69	121	160	137	104	100	100	122
	40%	7,375	357.78	25.57	117	139	123	102	100	100	137
Minimizing	10%	1,965	41.24	17.37	166	284	195	112	141	100	167
altered	20%	3,892	66.03	28.64	139	202	150	112	159	100	111
environments	30%	5,763	120.57	36.77	131	178	145	110	155	100	100
	40%	7,580	201.86	41.85	125	159	142	108	143	100	100
2050											
Not minimizing	10%	2,067	95.07	13.80	162	243	205	114	103	100	204
altered	20%	3,982	210.37	19.51	134	173	164	106	100	100	160
environments	30%	5,687	313.79	26.41	126	171	135	104	100	100	148

	40%	7,599	417.50	23.49	117	147	134	102	100	100	116
Minimizing	10%	1,923	54.60	18.82	166	294	195	109	134	100	166
altered	20%	3,746	96.08	30.92	140	210	162	107	147	100	114
environments	30%	5,574	175.35	37.99	129	178	154	105	136	100	101
	40%	7,369	285.36	40.64	123	160	149	103	122	100	104

Conclusão Geral

Nesta tese, discutimos oportunidades e desafios relacionados ao uso de serviços ecossistêmicos (SE) como uma abordagem para a conservação da biodiversidade. Avaliamos a eficácia da rede de áreas protegidas e terras indígenas do Cerrado em representar serviços ecossistêmicos. Também, exploramos como a demora na implementação de ações de conservação pode impactar a conservação de SE no Cerrado.

Mais especificamente, no primeiro capítulo, destacamos que abordagens baseadas em SE trazem novas oportunidades para a conservação da biodiversidade, tais como aumentar a consciência ambiental na sociedade e incentivar investimentos na conservação da natureza. No entanto, o uso de SE para promover a conservação da biodiversidade pode oferecer algumas limitações e riscos, como por exemplo, reduzir o interesse na conservação da natureza quando ações de conservação não são lucrativas ou devido a variações nos interesses do mercado econômico. Considerar múltiplos SE simultaneamente e realizar análises de longo prazo são estratégias para que abordagens baseadas em SE abranjam melhor a complexidade dos ecossistemas e promovam a conservação da biodiversidade. Por fim, destacamos que SE e biodiversidade são abordagens complementares para que metas de conservação sejam atingidas.

No segundo capítulo, encontramos que poucas áreas protegidas e terras indígenas do Cerrado são eficientes em capturar SE ou biodiversidade. Ainda, a maioria das reservas efetivas é adequada para representar apenas um serviço ecossistêmico dentre os seis avaliados. Esses resultados demonstram a necessidade de incorporar os SE como um alvo explícito de conservação em futuras estratégias que abarquem múltiplos objetivos de conservação. Além disso, políticas de conservação em áreas privadas precisam ser fortalecidas para que também contribuam com a conservação de SE e biodiversidade.

No terceiro capítulo, demonstramos que mudanças de uso de solo futuras irão afetar a provisão de SE do Cerrado e, por conseguinte, a distribuição espacial das áreas prioritárias na região. Além disso, as redes de áreas prioritárias identificadas para 2025 e 2050 englobarão maiores quantidades de ambientes alterados que a rede de áreas prioritárias no presente. Portanto, adiar ações de proteção à provisão de SE tenderá a aumentar os conflitos entre ações de conservação e atividades humanas. Enfatizamos que estabelecer ações de conservação de SE no Cerrado hoje é mais efetivo que adiar a implementação dessas ações para as próximas décadas.

Este estudo é o primeiro que temos conhecimento a realizar análises espaciais de múltiplos SE fornecidos pelo Cerrado em toda sua extensão. Esperamos que nossos resultados incentivem novos estudos e auxiliem tomadores de decisão a embasar políticas para a conservação de SE no Cerrado.

"*O futuro dependerá daquilo que fazemos no presente*."
Mahatma Gandhi