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**FUNCIONAMENTO DOS ECOSISTEMAS E CONSERVAÇÃO
BIOLÓGICA: POLUIÇÃO POR LUZ ARTIFICIAL, OFERECIMENTO
DE SERVIÇOS ECOSISTÊMICOS E DIVERSIDADE FUNCIONAL**

**SÃO PAULO
2016**

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ECOSSISTÊMICOS E DIVERSIDADE FUNCIONAL

Tese apresentada ao Programa de Pós-Graduação em Ciência Ambiental do Instituto de Energia e Ambiente da Universidade de São Paulo para obtenção do título de doutora em Ciência Ambiental.

Orientador: Prof. Dr. Waldir Mantovani

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AUTORIZO A REPRODUÇÃO E DIVULGAÇÃO TOTAL OU PARCIAL DESTE TRABALHO, POR QUALQUER MEIO CONVENCIONAL OU ELETRÔNICO, PARA FINS DE ESTUDO E PESQUISA, DESDE QUE CITADA A FONTE.

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RESUMO

FREITAS, Juliana Ribeirão de. **Funcionamento dos ecossistemas e conservação biológica: poluição por luz artificial, oferecimento de serviços ecossistêmicos e diversidade funcional.** 2016. 127f. Tese – (Doutorado em Ciência Ambiental) - Programa de Pós-Graduação em Ciência Ambiental – Instituto de Energia e Ambiente da Universidade de São Paulo, São Paulo, 2016.

O funcionamento dos ecossistemas é o fluxo de matéria e de energia ao longo dos seus componentes bióticos e abióticos. A manutenção de tal funcionamento é crucial para promover os serviços ecossistêmicos dos quais a humanidade depende. Atividades antrópicas, como a agricultura e a urbanização podem alterá-los significativamente e, por isso, o entendimento dos impactos positivos e negativos de tais alterações bem como o desenvolvimento de ferramentas de avaliação da biodiversidade são importantes para a conservação. A diversidade funcional é um componente da diversidade biológica que leva em conta o papel que cada espécie desempenha no ecossistema e, portanto, deve refletir o funcionamento do ecossistema de forma mais acurada. Esta tese teve por objetivos relacionar o funcionamento do ecossistema e os seus serviços a ações antrópicas, bem como propor a abordagem funcional como indicadora da biodiversidade de áreas naturais. No primeiro capítulo, avalio padrões espaciais e temporais da exposição dos tipos de vegetação que ocorrem no Brasil à luz artificial, um dos principais símbolos da urbanização e da vida moderna. Os resultados mostram que a maioria deles apresenta alguma porcentagem de sua área expostos à luz artificial. Em alguns deles, porém, ainda é possível encontrar o brilho natural do céu noturno, o que nos permite sugerir a elaboração de políticas de desenvolvimento de distribuição de luz com foco no mínimo impacto, ao contrário das políticas de mitigação adotadas por países onde a iluminação é excessivamente difundida. No segundo capítulo mapeio padrões espaciais e temporais da provisão de dois serviços ecossistêmicos (polinização e estoque de carbono) e um serviço ambiental (espaço para viver) e avalio os impactos da agricultura sobre estes serviços, na área coberta pelo cerrado, na região central do Brasil. Os serviços ecossistêmicos avaliados descaíram significativamente ao longo do tempo e a região conhecida como MATOPIBA (entre os estados do Maranhão, Tocantins, Piauí e Bahia) foi identificada como um importante remanescente. Por isso, recomendo a criação de unidades de conservação de proteção integral na área. Identifiquei também sobreposição da disponibilidade destes serviços com áreas indígenas, o que ressalta a importância da manutenção destas áreas para a disponibilidade dos serviços. No terceiro capítulo exploro, por meio de revisão bibliográfica, o status da diversidade funcional na literatura científica no campo da Conservação Biológica. Os resultados permitiram visualizar o paradigma da transferência de conhecimento e identificar o potencial desta abordagem para elaboração de indicadores da biodiversidade. Por fim, no quarto capítulo, estabeleço uma lista de espécies de plantas que podem ser indicadoras da diversidade funcional em áreas cobertas por cerrado no estado de São Paulo. Tais indicadores devem ser usados em conjunto para monitoramento e diagnóstico da biodiversidade quando o objetivo é manter o funcionamento do ecossistema. As conclusões do estudo contribuem para elucidar a interferência das atividades antrópicas nos ecossistemas naturais, para levantar formas de minimizá-los e para aprimorar as formas de avaliação da biodiversidade.

Palavras-chave: Funcionamento dos ecossistemas, poluição por luz artificial, serviços ecossistêmicos, agricultura, diversidade funcional, Cerrado, Cerradão.

ABSTRACT

FREITAS, Juliana Ribeiro de. **Ecosystem functioning and biological conservation: light pollution, ecosystem services provisioning and functional diversity**. 2016. 127f. Doctorate Thesis. Environmental Science Program, Institute of Energy and Environment, Universidade de São Paulo, São Paulo, 2016.

Ecosystem functioning is the flow of energy and matter through the biotic and abiotic ecosystem's components. The maintenance of this functioning is essential to the ecosystem services provisioning upon which humans depend. Anthropogenic activities such as agriculture and urbanisation may change it. Thus, conservation strategies rely on the understanding of the positives and negatives impacts from this changes and on the development of measurements of biodiversity. Functional diversity is a biodiversity component which considers the role of each species in the ecosystem, and, as a consequence, may reflect the ecosystem functioning more accurately. This thesis aims the establishment of the relationships between ecosystem functioning, ecosystem services and anthropogenic activities as well as purpose functional approach as biodiversity indicator of natural areas. In the first chapter, I assessed spatial and temporal patterns of exposition of the Brazilian vegetation types to the artificial light – which is one of the urbanisation symbols. The results show that most of them are affected by artificial light. However, in some of them it is still possible to find a natural sky background, which allow suggesting the formulation of light distribution policies focused on minimal impact instead of mitigation, as adopted by countries where light is excessively widespread. In the second chapter, I mapped spatial and temporal patterns of two ecosystem services provisioning (pollination and carbon stocks) and one environmental service (living space) and assessed the impacts of agriculture in these services in Cerrado area, in the central region of Brazil. The ecosystem services I assessed declined significantly over time and the MATOPIBA region (around Maranhão, Tocantins, Piauí, and Bahia States) was identified as an important remnant. Thus I recommended the creation of strictly protected areas in that region. I also identify overlap of these services with indigenous land, highlighting its importance for the ecosystem services provisioning. In the third chapter I explored through literature survey, the status of functional diversity in the scientific literature related to the Biological Conservation field. The results allow visualise the paradigm of knowledge transfer and identify the potential use of this approach for developing indicators of biodiversity. Finally, in the fourth chapter, we established a list of plant species that are functional diversity indicators in cerrado areas in São Paulo State. These indicators should be use as a set for the monitoring and diagnosis of biodiversity when the goal is to maintaining ecosystem functioning. The conclusions of the study contribute to clarify some lack of knowledge concerned to the impacts of human activities on ecosystem functioning, to raise means to minimise them, and to improve means of ecological integrity assessment.

Key-words: Ecosystem functioning, light pollution, ecosystem services, agriculture, functional diversity, Cerrado (Brazilian Savanna), Cerradão.

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INTRODUÇÃO GERAL

1. A diversidade biológica e o funcionamento dos ecossistemas

Ecossistemas são sistemas ecológicos que englobam conjuntos de organismos de espécies diferentes com seus ambientes físicos e químicos. Ao longo de suas vidas, os organismos adquirem energia e nutrientes de seus arredores e se livram de rejeitos indesejados. Ao realizar tais tarefas, eles modificam as condições do ambiente e os recursos disponíveis para outros organismos, gerando um fluxo de energia e ciclando a matéria (BEGON; TOWNSEND; HARPER, 2007; RICKLEFS, 2003). O fluxo de matéria e energia ao longo dos componentes bióticos e abióticos do ecossistema é chamado de funcionamento dos ecossistemas (DÍAZ; CABIDO, 2001; HOOPER et al., 2001).

O funcionamento dos ecossistemas se mantém porque diferentes tipos de organismos apresentam diferentes funções. Por exemplo, as plantas utilizam a luz do sol para produzir matéria orgânica, possibilitando a entrada da energia solar nos ecossistemas; os animais consomem plantas e/ou outros animais, fazendo a transferência de energia ao longo dos níveis tróficos e devolvendo parte da energia ao ambiente em forma de calor; Fungos e bactérias são exímios decompositores, possibilitando transformações energéticas e o retorno da matéria orgânica morta aos ciclos vitais (RICKLEFS, 2003).

Todos os ecossistemas são sistemas abertos: ainda que a circulação interna de matéria e energia seja mais intensa, há trocas com sistemas externos. Portanto, o funcionamento dos ecossistemas tem relação direta com o funcionamento dos sistemas antrópicos, tais como sistemas agrícolas e sistemas urbanos (BOULDING, 1966; ODUM; BARRETT, 1971) (Figura 1). Os sistemas antrópicos são mantidos devido à *inputs* recebidos dos ecossistemas naturais, os quais geram benefícios diretos e indiretos aos seres humanos (MEA- MILLENIUM ECOSYSTEM ASSESSMENT, 2005). Por exemplo, a maior parte das plantas cultivadas atualmente depende de espécies de animais polinizadores, cujas populações são mantidas devido ao funcionamento dos ecossistemas naturais localizados no entorno das lavouras (OLLERTON; WINFREE; TARRANT, 2011; RICKETTS et al., 2008). Da mesma forma, essas mesmas lavouras não são dizimadas por herbívoros ou parasitas devido à existência de predadores, igualmente advindos dos ecossistemas naturais (BENTON; VICKERY; WILSON, 2003). Um outro exemplo é a manutenção do clima: ciclagem de carbono, oxigênio e vapor de água são processos que dependem das funções das espécies nos ecossistemas e são críticos para

a regulação do clima na Terra (BEGON; TOWNSEND; HARPER, 2007).

Figura 1: Diagrama esquemático das relações entre o funcionamento dos ecossistemas e os sistemas antrópicos. O funcionamento dos ecossistemas é o fluxo de matéria e energia que mantém o funcionamento dos sistemas antrópicos (tais como as áreas urbanas e agrícolas). Ao mesmo tempo, os sistemas antrópicos geram impactos que alteram o componente funcional da diversidade, alterando, por consequência, o seu funcionamento.

Ao mesmo tempo que dependem dos ecossistemas naturais, os sistemas antrópicos também liberam *outputs*, que frequentemente alteram o funcionamento dos ecossistemas. A diversidade de funções que as espécies podem desempenhar no ecossistema compõe o componente funcional da diversidade biológica (BLÜTHGEN, 2011). Como as funções das espécies dependem da sua interação com o ambiente e com as outras espécies presentes, o componente funcional da diversidade está fortemente relacionado ao nicho ecológico das espécies (i.e. os intervalos de condições ambientais dentro dos quais uma determinada espécie ocorre) – HOOPER et al., 2001; RICKLEFS, 2003). Cada espécie tem um nicho distinto e não há duas espécies que sejam exatamente iguais. Comunidades presentes em um ambiente alterado por tais *outputs* devem se ajustar às novas condições. Tal ajuste pode implicar no aumento ou na diminuição dos nichos de algumas espécies pertencentes à comunidade, na inclusão de novas espécies, ou ainda na eliminação de outras (SCHWARTZ et al., 2000). Ao alterar a distribuição dos nichos e/ou a composição das comunidades, os *outputs* afetam o componente funcional da diversidade, alterando assim, o funcionamento do ecossistema (Figura1).

Exemplos de *outputs* podem ser iluminação artificial noturna, ruídos urbanos (causados por tráfego de automóveis e de aviões, maquinaria de indústria, ar condicionado, etc) e geração de resíduos sólidos (ALBERTI, 2015; BAI, 2016). Por exemplo, as fronteiras do nicho de uma espécie hipotética presente em uma comunidade afetada por esses fatores poderiam se estender entre, digamos, intensidades luminosas de 10 a 50W/m², entre amplitude de ruído do ambiente entre 40 e 50 decibéis e entre 0,01 a 0,05 mg de N por litro de água. Quando essas condições são alteradas, esta espécie pode extinguir-se daquele local ou adaptar-se, ampliando seu nicho. No segundo caso, a ampliação de seu nicho possivelmente afetará o nicho de outras espécies, uma vez que novas relações de competição serão criadas. Além disso, as novas condições ambientais podem também atrair novas espécies ao local, com as quais as espécies já existentes também devem competir.

A manutenção tanto dos ecossistemas naturais quanto dos sistemas antrópicos depende, em parte do entendimento das relações dos seus *inputs* e *outputs* bem como da avaliação da integridade do funcionamento dos ecossistemas naturais.

2. O funcionamento dos ecossistemas e a provisão dos serviços ambientais e ecossistêmicos

Os benefícios que os sistemas antrópicos obtêm do ambiente natural são definidos de diferentes formas na literatura (COSTANZA et al., 1997; MEA- MILLENIUM ECOSYSTEM ASSESSMENT, 2005; WUNDER, 2005; ZHANG et al., 2007). Nesta tese, definimos como serviços ecossistêmicos (ou do ecossistema) como os benefícios que os seres humanos obtêm a partir da biodiversidade e do funcionamento do ecossistema (MEA- MILLENIUM ECOSYSTEM ASSESSMENT, 2005) e serviços ambientais como benefícios que são obtidos a partir do ambiente que não são atribuídos aos ecossistemas diretamente (CASALEGNO et al., 2014). Nesse sentido, os serviços ambientais constituem-se em uma síntese que inclui os serviços ecossistêmicos.

Os serviços ecossistêmicos são classificados funcionalmente em: serviços de provisão, aqueles obtidos diretamente dos ecossistemas, tais como alimento e combustível; serviços de regulação, obtidos a partir dos processos ecossistêmicos; serviços culturais, obtidos dos ecossistemas através de seu enriquecimento espiritual, desenvolvimento cognitivo, reflexão, recreação e experiências estéticas; e os de suporte, aqueles necessários à produção de outros serviços, como a produção de solo e a ciclagem de nutrientes (MEA, 2005).

Dois serviços de regulação têm recebido mais atenção recentemente: a polinização e a regulação da quantidade de carbono na atmosfera. A polinização é um processo crítico para manter tanto os sistemas naturais quanto os sistemas agrícolas, já que aproximadamente 87% das espécies de plantas com flores e 75% das espécies usadas em lavouras no mundo dependem, ao menos em parte, de animais polinizadores (KLEIN et al., 2007; OLLERTON; WINFREE; TARRANT, 2011).

Um declínio preocupante dos polinizadores tem sido observado ao redor do mundo nos últimos anos (IPBES, 2016; POTTS et al., 2010). Apesar desse declínio ser mais acentuado em regiões de clima temperado (SCHULP; LAUTENBACH; VERBURG, 2014), as suas consequências em regiões de clima tropical devem ser mais drásticas (VAMOSI et al., 2006). Enquanto em ambientes temperados a maior parte da polinização é promovida majoritariamente por abelhas solitárias e vespas, nos trópicos, além destes grupos, borboletas, mariposas, aves e morcegos também têm importância (GIANNINI et al., 2015; RICKETTS et al., 2008). Nas regiões de alta diversidade como os trópicos, as plantas veem co-evoluindo com um número maior e mais variado de animais polinizadores, o que acarreta em maior grau de especialização nas relações planta-polinizador e, conseqüentemente, no declínio dos polinizadores, o que exerce mais pressão à extinção das plantas (JANZEN, 1975; VAMOSI et al., 2006). Essa pressão se torna ainda mais acentuada em ambientes com altos níveis de endemismo e ameaça, como exemplos, o Cerrado e a Floresta Pluvial Tropical Atlântica (MYERS et al., 2000; VAMOSI et al., 2006).

Diante do aumento da concentração de gases de efeito estufa na atmosfera ao longo do último século, o sequestro de carbono tornou-se um serviço ecossistêmico de grande importância no cenário das mudanças climáticas (MEA, 2005; SCIENCE FOR ENVIRONMENT POLICY, 2015). Apesar de este serviço ser frequentemente associado aos sistemas florestais (por exemplo, o REDD+ - <http://redd.mma.gov.br/index.php/pt/redd/o-que-e-redd>), globalmente, sistemas não-florestais estocam tanto carbono quanto as florestas, o que tem sido negligenciado (WHITE; MURRAY; ROHWEDER, 2000). A biomassa subterrânea dos ecossistemas é menos estudada do que a biomassa aérea devido, em grande parte, à dificuldade de seu acesso (MOKANY; RAISON; PROKUSHKIN, 2006). Por isso, ecossistemas com proporções mais altas de biomassa subterrânea, como o Cerrado e a Caatinga, por exemplos, tem estimativas imprecisas de sua capacidade de estoque de carbono (LOIOLA; SCHERER-LORENZEN; BATALHA, 2015; MOKANY; RAISON; PROKUSHKIN, 2006).

O espaço geográfico habitado por pessoas lhes fornece os meios de subsistência e de

trabalho e produção, compondo a estrutura de uma sociedade (MORIN, 1996; SANTOS, 1977). É um benefício obtido do ambiente é essencial para os humanos que não apresenta relação direta com a biodiversidade ou com o ecossistema. Nesse sentido, consideramos o ‘espaço para viver’ um serviço ambiental (CASALEGNO et al., 2014). O espaço para viver como serviço ambiental surgiu na literatura representado por áreas urbanas (CASALEGNO et al., 2014). Contudo, a definição pode ser estendida a grupos humanos com outros modos de vida, já que estes grupos igualmente estabelecem relações com o espaço onde vivem (DIEGUES, 1996). Em geral, o modo de vida das populações não-urbanas, como as indígenas, é mais sustentável e, por isso, a questão da territorialidade destes povos aparece frequentemente na literatura científica relacionada à utilidade destes espaços para a proteção da biodiversidade e dos serviços ecossistêmicos e raramente à abordagem do espaço para viver como um serviço por si só, necessário à sua cultura (DIEGUES, 1996; NAUGHTON-TREVES; HOLLAND; BRANDON, 2005).

3. Impactos negativos no funcionamento dos ecossistemas: expansão das áreas urbanas e agrícolas.

Nos sistemas urbanos, os fluxos de matéria e energia interagem com os fluxos de capital, informações e pessoas gerando *outputs*. Alguns desses *outputs* geram benefícios e melhoram a qualidade de vida dos seres humanos (como produtos industriais, serviços e conhecimento) (BAI, 2016). Entretanto, grande parte deles influencia, de forma negativa e significativa, o funcionamento dos ecossistemas naturais em níveis locais, regionais e globais. A expansão das áreas urbanas em geral simplifica e homogeneiza a composição de espécies, perturba os sistemas hidrológicos e os fluxos de energia e de nutrientes por meio do uso de recursos e liberação de resíduos (ALBERTI, 2005). As áreas urbanas são fortemente associadas à presença de luz artificial. No último século o crescimento da proporção do planeta que está sujeito à luz artificial foi tão alto, que um observador colocado fora do sistema solar teria a possibilidade de ver a terra como um planeta que brilha com luz própria (BARGHINI, 2010).

As evidências de que a luz artificial altera o componente funcional da diversidade e, portanto, afeta processos que mantêm o funcionamento dos ecossistemas veem aumentando (GASTON et al., 2013, 2014; GASTON; VISSER; HÖLKER, 2015a). A luz artificial altera os ritmos circadianos e os ritmos biológicos das espécies da flora e atrai diversas espécies de insetos, potencialmente vetores de patógenos (BARGHINI, 2010). Essas alterações podem ocorrer localmente, no ambiente ao redor da fonte de luz, ou se estender por centenas de

quilômetros para além das áreas urbanas, por meio da luz dispersa na atmosfera, ou *skyglow* (KYBA et al., 2015; KYBA; HÖLKER, 2013).

Os impactos negativos dos outpus não se restringem aos sistemas urbanos. Embora os sistemas agrícolas sejam imprescindíveis à sobrevivência humana (SWINTON et al., 2007), a intensificação das práticas agrícolas gera mais impactos negativos do que positivos ao ambiente sendo, a longo prazo, mais danosa do que benéfica aos seres humanos (BALMFORD; GREEN; PHALAN, 2012; GREEN et al., 2005; ZHANG et al., 2007).

A expansão e a intensificação das atividades agrícolas têm alterado a distribuição, a abundancia e a efetividade dos polinizadores em níveis regionais e globais (GIANNINI et al., 2012; IPBES, 2015). Essas atividades alteram também o ciclo do carbono e de outros gases causadores do efeito estufa, seja por meio do seu estoque na biomassa das plantas cultivadas, ou da sua liberação, via desmatamento das florestas tropicais ou aplicação de fertilizantes sintéticos (ALBRECHT; KANDJI, 2003; FAO, 2004; GILBERT, 2012).

O espaço para viver também é afetado pela expansão da agricultura. O cultivo massivo de uma ou poucas espécies em grandes extensões contínuas de terra está associado à concentração das terras por grandes corporações, o que gera conflitos territoriais com pequenos agricultores familiares, populações tradicionais e indígenas. Além disso, as monoculturas estão associadas à diminuição da biodiversidade local, desde o nível genético até o de paisagem, e ao uso de pesticidas poluidores do solo e das águas, influenciando e alterando o modo de vida destas populações e ameaçando suas culturas (BICKEL; DROS, 2003).

4. Integridade ecológica e funcionamento dos ecossistemas

Diante das modificações intensas causadas pelos humanos, é necessário avaliar a integridade ecológica dos ecossistemas naturais, em especial dentro de áreas protegidas (MEFFE; CARROLL, 1963). Integridade ecológica é o grau no qual um conjunto de organismos mantém sua composição, estrutura e função ao longo do tempo, se comparado a um conjunto que não tenha sido alterado por ações humanas (TROMBULAK et al., 2004), e portanto, tem estreita relação com o funcionamento dos ecossistemas. Avaliar a integridade ecológica em ecossistemas terrestres é uma tarefa particularmente mais árdua do que em ecossistemas aquáticos, devido à maior dificuldade de se definir uma situação de referência pouco perturbada (METZGER; CASATTI, 2006). A avaliação da integridade ecológica nesses ecossistemas é frequentemente associada à avaliação da diversidade biológica (METZGER; CASATTI, 2006). Idealmente, avaliações da biodiversidade deveriam abranger todos os níveis

da biodiversidade, de genéticos até biogeográficos. Na prática, recorre-se, para cada nível, a indicadores de biodiversidade, também chamados de substitutos da biodiversidade (MARGULES; PRESSEY; WILLIAMS, 2002). No nível específico, existem propostas de uso de indicadores da biodiversidade baseados em múltiplas espécies (RICKETTS et al., 1999) ou em uma única espécie, tais como espécies-chave, espécies guarda-chuva ou espécies-bandeira (ANDELMAN; FAGAN, 2000; CARO; DOHERTY, 1999). A construção destes indicadores assume que é possível encontrar um ou poucos táxons capazes de representar grande parte da diversidade de uma área e, portanto, a avaliação dos atributos desses táxons (presença/ausência, abundância, riqueza, índices qualitativos, semi-quantitativos ou quantitativos) permitiria acessar as variações na complexidade biológica (CARO; DOHERTY, 1999; OLIVEIRA; TIDON, 2013; TRINDADE-FILHO; LOYOLA, 2010).

O funcionamento dos ecossistemas é mantido porque as espécies podem apresentar funções muito diferentes ou muito semelhantes. Na primeira situação, os nichos ecológicos se complementam, aumentando a eficiência dos processos. Na segunda, os nichos se sobrepõem de forma que uma espécie substitua certas funções da outra, mantendo a resiliência do ecossistema (MACARTHUR, 1972). Assim, a relação entre as espécies e o funcionamento dos ecossistemas se dá por meio das funções que cada uma delas desempenha em uma comunidade (HOOPER et al., 2005; TILMAN, 1997). Entretanto, os indicadores da biodiversidade utilizados por ecólogos e conservacionistas, bem como os atributos mais medidos (riqueza e abundância), são baseados em critérios taxonômicos (MAGURRAN, 2004). Embora essas medidas sejam importantes para descrever padrões, elas assumem que cada espécie contribui igualmente para o funcionamento do ecossistema e não levam em conta a influência que as características das espécies exercem sobre os processos ecossistêmicos (DÍAZ; CABIDO, 2001; PETCHEY; GASTON, 2002). Os indicadores de biodiversidade mais comumente utilizados não consideram o componente funcional da diversidade e, como consequência, devem falhar na avaliação da integridade ecológica.

5. Diversidade funcional

A diversidade funcional é um componente da diversidade biológica relacionado ao modelo de nichos: quanto mais alta a complementaridade, mais alta a eficiência no funcionamento do ecossistema (DÍAZ; CABIDO, 2001; MACARTHUR, 1972; PETCHEY; GASTON, 2002). Nessa abordagem, as comunidades podem ser pensadas como um espaço

multidimensional (o nicho total) no qual os nichos de todas as espécies se encaixam. A adição ou a remoção de espécies neste espaço pode aumentar ou diminuir o tamanho do nicho das outras espécies, afetando o funcionamento dos ecossistemas. As características das espécies adicionadas ou removidas são críticas para fazer previsões. A mensuração da diversidade funcional é feita por meio de índices que descrevem a distribuição dos nichos no espaço multidimensional medindo, por exemplo, o tamanho do nicho de cada espécie, a distância entre eles, quão complementares ou sobrepostos eles são (MOUCHET et al., 2010; PETCHEY; GASTON, 2006; VILLÉGER; MASON; MOUILLOT, 2008). Os nichos das espécies de uma dada comunidade são plotados com base na diversidade de seus traços funcionais.

Um traço funcional é qualquer característica mensurável de um indivíduo que potencialmente afete o desempenho da espécie: seu crescimento, reprodução e/ou sobrevivência (PETCHEY; GASTON, 2002). Os traços funcionais podem se expressar de forma física, bioquímica, temporal ou até mesmo comportamental. São exemplos em grupos de vertebrados a morfologia dos dentes de um predador, a preferência por forrageamento noturno ou diurno, o tipo de dieta e a biomassa corpórea (TRINDADE-FILHO; LOYOLA, 2010). Em invertebrados consideram-se traços funcionais o tamanho dos corpos e dos apêndices, tipo e tamanho de asas e de aparelho bucal e resistência à dessecação (HEEMSBERGEN et al., 2004; PODGAISKI; MENDONÇA JR; PILLAR, 2011). Pesquisas em ecologia vegetal estão na vanguarda do uso de traços funcionais (CORNELISSEN et al., 2003; DÍAZ; CABIDO, 2001). Em plantas os traços funcionais são características morfológicas, fisiológicas e/ou fenológicas que representam estratégias ecológicas (VIOLE et al., 2007). Eles determinam as respostas das plantas a fatores ambientais, suas relações com os outros níveis tróficos e suas influências sobre as propriedades do ecossistema (PÉREZ-HARGUINDEGUY et al., 2013).

Embora exista relação positiva entre a riqueza e o funcionamento dos ecossistemas, não há evidências científicas que embasem o uso da riqueza como substituta da diversidade funcional (DÍAZ; CABIDO, 2001; PETCHEY; GASTON, 2002). Uma vez que as medidas de diversidade funcional se baseiam na teoria de nichos, relaciona-se mais diretamente com o funcionamento dos ecossistemas e, conseqüentemente, com os serviços dos ecossistemas (DEVICTOR et al., 2010; LOREAU, 1998). Um número crescente de autores vem apontando a necessidade de incorporar a diversidade funcional na construção de indicadores da biodiversidade (DE BELLO; MUDRÁK, 2013; GILLISON et al., 2013; VANDEWALLE et al., 2010).

6. Postulados

O funcionamento dos ecossistemas é determinado pelo componente funcional da biodiversidade, disponibiliza os serviços ecossistêmicos e é afetado por sistemas antrópicos, tais como urbanos e agropecuários.

7. Objetivos gerais

Os objetivos gerais desta tese são:

- Avaliar os impactos dos sistemas antrópicos no funcionamento dos ecossistemas e na provisão dos serviços ambientais e ecossistêmicos e
- Propor a abordagem funcional como indicadora da biodiversidade.

8. Objetivos específicos e estrutura da tese

Esta tese foi organizada em quatro capítulos em formatos de artigos científicos publicáveis em periódicos. O primeiro capítulo analisa padrões espaciais e temporais da exposição dos tipos de vegetação que ocorrem no Brasil à luz artificial e está submetido ao periódico PlosOne. O segundo mapeia dois serviços ecossistêmicos (polinização e estoque de carbono) e um serviço ambiental (espaço para viver) e avalia os impactos da agricultura nestes serviços, na área coberta pelo cerrado, na região central do Brasil. Este capítulo será submetido ao periódico Ecosystem. O terceiro capítulo explora, por meio de revisão bibliográfica, o status da incorporação do componente funcional da diversidade biológica dentro do campo da Conservação Biológica e será submetido à Brazilian Journal of Biology. Por fim, o quarto estabelece quais espécies podem ser indicadoras da diversidade funcional em áreas cobertas por cerrado no estado de São Paulo e está submetido ao Journal for Tropical Conservation.

9. Justificativa

Mapear a exposição dos ecossistemas à luz artificial e seus padrões de mudanças no espaço e no tempo é o primeiro passo para determinar a severidade dos impactos nos diferentes ecossistemas, tipos vegetacionais e habitats. Em escala global todos os ecossistemas naturais veem experimentando algum nível de aumento na exposição à luz artificial (BENNIE et al., 2015). Análises em escalas mais finas podem basear estratégias de conservação e auxiliar o

planejamento da iluminação urbana visando mínimo impacto. Avaliações do padrão de exposição espacial e temporal dos ecossistemas à luz artificial foram desenvolvidos principalmente no hemisfério norte, notadamente nos Estados Unidos e nos países da Europa Ocidental (BENNIE et al., 2014, 2015; CINZANO; FALCHI; ELVIDGE, 2001), onde a presença de luz artificial é mais antiga e mais homogeneamente distribuída (BENNIE et al., 2014a; CINZANO; FALCHI; ELVIDGE, 2001).

Em países com níveis mais altos do Produto Interno Bruto (tais como aqueles localizados do norte da América do Norte e da Europa Ocidental), a presença de luz artificial é mais antiga e mais homogeneamente espalhada (CINZANO; FALCHI; ELVIDGE, 2001). Talvez por isso, os potenciais impactos da sua presença têm sido mais explorados nessas regiões, tanto do âmbito científico quanto no político (BARGHINI, 2010; BENNIE et al., 2015b; CALIFORNIA ENERGY COMMISSION, 2013; FARRINGTON; WELSH, 2002; MILLER, 2006). Comparados com eles, os países com mais altos níveis de biodiversidade no planeta (i.e. aqueles localizados na região tropical) apresentam níveis de poluição luminosas mais baixos e menos alarmantes (CINZANO; FALCHI; ELVIDGE, 2001) e talvez por isso a questão tenha recebido menos atenção nestes locais. Entretanto, é importante destacar que tais níveis não são frutos de políticas de controle da difusão da iluminação ou planejamento para a conservação da biodiversidade. Ao contrário, estão relacionados à questões econômicas, às desigualdades na densidades demográficas e ao curto tempo decorrido desde a chegada da luz artificial (BARGHINI, 2010). Assim, é possível inferir que é apenas uma questão de tempo e de mudanças na conjuntura econômica para que a luz artificial deixe de estar concentrada em áreas urbanas mais densamente populosas e se espalhar intensamente por todos os territórios de tais países.

Mapear a presença de luz artificial em relação aos seus tipos vegetacionais em países tropicais contribui para o planejamento da difusão da iluminação artificial nestes países. O mapeamento identifica os tipos vegetacionais mais expostos, indicando a necessidade de medidas de mitigação e potencialmente descobre áreas pristinas, ainda poucos expostas, para as quais um novo modelo de planejamento pode ser pensado, baseado no mínimo (ou até mesmo zero) impacto. Nesse contexto, o Brasil constitui-se um interessante objeto de estudo, pois é um país em desenvolvimento, de extensão continental, com a maior parte do seu território na zona tropical, e com níveis consideráveis de luz artificial nas áreas mais urbanizadas, particularmente entre os estados do Rio de Janeiro e São Paulo.

O Brasil lidera entre os produtores mundiais de *commodities* para exportação, tais como o

café, a cana de açúcar, a soja e a carne bovina. Aproximadamente 50% das áreas de produção de soja e 35% dos pastos usados à produção bovina no país estão situadas na região coberta pelo Cerrado, que já teve mais de 40% da sua vegetação nativa convertida para estas atividades (OCDE; FAO, 2015; RADA, 2015). Localizado no Planalto Central brasileiro, a área de ocorrência do Cerrado foi considerada imprópria para a agricultura devido, sobretudo, à pobreza de nutrientes e à acidez dos solos. Porém, a partir da década de 1970, avanços agronômicos e tecnológicos promovidos em grande parte pela EMBRAPA (Empresa Brasileira de Agropecuária), aliados à facilidade da mecanização devido ao relevo pouco acidentado e aos preços baixos das terras no mercado, impulsionaram mudanças drásticas na paisagem da região (BICKEL; DROS, 2003; REIFSCHNEIDER et al., 2010). O modelo agropecuário predominantemente implantado no Brasil Central é de caráter predatório, baseado na incorporação de extensas áreas para a agricultura comercial, para plantios homogêneos, com uso intensivo de agrotóxicos (MMA, 2003).

Ao mesmo tempo em que a região é conhecida internacionalmente pela alta produtividade agrícola¹, o Cerrado destaca-se por sua importância biológica: abriga a mais rica flora entre as savanas do mundo e é um dos *hotspots* mundiais para a conservação (CASTRO et al., 1999; MYERS et al., 2000). O termo Cerrado refere-se a um complexo de vegetação xeromorfa que ocorre sob climas diversos, com predomínio do clima tropical com estação seca no inverno (Aw de KÖPPEN, 1931) sobre solos lixiviados e aluminizados, ácidos, em geral profundos. É subdividido em três formações e dois subgrupos de savana: campo limpo (campo), campo sujo, cerrado *sensu stricte*, campo cerrado (savana) e cerradão (floresta). Estas formações e subgrupos compõem um gradiente estrutural-fisionômico e florístico do campo limpo ao cerradão. A porcentagem de cobertura das copas das árvores, a área basal e a proporção das espécies arbóreas aumentam continuamente ao longo do gradiente, ao passo que a cobertura herbácea e o número de espécies não-arbóreas diminuem. A vegetação das formas intermediárias apresenta fisionomias savânicas e é constituída tanto por espécies florestais quanto campestres (COUTINHO, 1978). Esse gradiente confere ao cerrado uma alta variabilidade de habitats ao longo do estrato horizontal, equivalente à variabilidade de habitats encontrada na Floresta Ombrófila Densa ao longo do estrato vertical (COUTINHO, 1978).

O alto valor biológico aliado à alta produtividade agrícola fazem do Cerrado brasileiro um objeto de estudo interessante para mapear a provisão dos serviços do ecossistema e avaliar os

¹ <http://www.economist.com/node/16889019>

impactos da agricultura sobre eles. O serviço ambiental ‘espaço para viver’ reduzem a área que poderia potencialmente prover serviços ecossistêmicos ou destinadas à conservação da biodiversidade. Assim, seu mapeamento concomitante com serviços ecossistêmicos facilita a visualização dos balanços espaciais e temporais entre esses espaços e possibilita a identificação de antagonismos e sinergias (BURKHARD et al., 2012; CASALEGNO et al., 2014). Os resultados devem ressaltar a importância deste ecossistema pouco conhecido e embasar o seu planejamento espacial, visando conciliar produtividade e conservação biológica (BURKHARD et al., 2012).

No estado de São Paulo, onde a ocupação é mais antiga, o nível de conservação do cerrado é ainda mais preocupante. Apenas 7% da vegetação original permanecem conservados, divididos em pequenas áreas imersas em matrizes agrícolas, como as de cana-de-açúcar, citricultura, pastagens ou silvicultura (DURIGAN et al., 2007). O Cerradão, a fisionomia mais comum do Cerrado no estado, apresenta alto valor biológico, pois é um disseminador de propágulos de espécies arbóreas para as outras fisionomias e representa a região ecotonal entre o Cerrado e a Floresta Estacional Semidecídua (DURIGAN; RATTER, 2006). Apesar disto, apresenta a mais baixa representatividade de área nas Unidades de Conservação do estado (DURIGAN et al., 2007).

As áreas remanescentes de Cerrado prioritárias para a conservação no estado de São Paulo foram estabelecidas por Durigan et al (2003) com base em 15 indicadores. O conjunto de indicadores incluiu a representação da biodiversidade segundo critérios taxonômicos e não incluíram o componente funcional. Atualmente a avaliação da performance dos ecossistemas cobertos por Cerradão no estado de São Paulo é feito com base na Resolução nº 64 da Secretaria do Meio Ambiente do Estado, que indica que o diagnóstico deve basear-se na densidade de indivíduos de espécies lenhosas, diâmetro do caule, área basal, e porcentagem de cobertura do solo por gramíneas (SÃO PAULO, 2009).

A inclusão de métricas de diversidade funcional na construção de indicadores de biodiversidade pode contribuir com estratégias de conservação quando o objetivo é proteger ou avaliar o funcionamento dos ecossistemas. Até o momento, dois estudos se propuseram a tal inclusão. Trivellone et al. (2014) elaboraram uma lista de espécies de plantas apontadas como indicadoras efetivas da diversidade taxonômica e funcional em vinícolas nos Alpes suíços. Os autores recomendaram que a lista podia ser incluída nos instrumentos de avaliação dos pagamentos por serviços ecossistêmicos que ocorrem na região. Trindade Filho et al. (2012) encontraram que aves pertencentes às ordens Apodiformes e Charadriiformes, bem como

grupos de espécies com distribuição restrita, tem alta chance de garantir a conservação dos processos ecossistêmicos e evolutivos e recomendam o uso destes grupos como atalho em planos de conservação da Mata Atlântica. A revisão bibliográfica realizada neste trabalho, explorada em detalhes no capítulo 3, mostra que nenhum trabalho até o momento se propôs a incluir o componente funcional no desenvolvimento de indicadores da biodiversidade de áreas de Cerrado do estado de São Paulo.

As plantas constituem um grupo adequado de organismos para indicar o componente funcional da diversidade em ecossistemas terrestres. Em primeiro lugar, elas ocupam o primeiro nível da cadeia trófica, por onde a energia solar entra nos ecossistemas. Segundo, a estrutura da vegetação está relacionada com a diversidade de habitats (RICKLEFS, 2003). Terceiro, elas podem ser facilmente identificadas e localizadas pelo público em geral, o que é uma característica desejável para indicadores (NIEMI; MCDONALD, 2004). E por fim, o conhecimento sobre como medir os traços funcionais de plantas é relativamente avançado (CORNELISSEN et al., 2003; PÉREZ-HARGUINDEGUY et al., 2013).

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CAPÍTULO 1

Exposure of tropical ecosystems to artificial light at night: Brazil as a case study

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Abstract

Artificial nighttime lighting, from streetlights and other sources, has a broad range of biological impacts. Understanding the spatial and temporal levels and patterns of this lighting is a key step in determining the severity of these impacts on different ecosystems, vegetation and habitat types. There have been few such analyses, particularly for regions with high biodiversity, including the tropics. We used an intercalibrated version of the Defense Meteorological Satellite Program's Operational Linescan System (DMSP/OLS) images of stable nighttime lights to determine what proportion of original and current Brazilian vegetation types are experiencing measurable levels of artificial light and how this has changed in recent years. The percentage area affected by both detectable light and increases in brightness ranged between 0 and 35% for original vegetation types, and between 0 and 25% for current ones. The most heavily affected areas encompassed terrestrial coastal vegetation types (restingas and mangroves), Semideciduous Seasonal Forest, and Mixed Ombrophilous Forest. The existing small remnants of Lowland Deciduous and Semideciduous Seasonal Forests and of Campinarana had the lowest exposure levels to artificial light. Light pollution is not often considered to be an issue in developing countries but our data show that it is an environmental concern.

1. Introduction

The nighttime environment is undergoing a dramatic transformation across the Earth's surface. The cycles of natural light (daily, lunar and seasonal) that have been major forms of environmental variation since the first emergence of life are being disrupted through the introduction of artificial lighting. A diversity of sources (including street lighting, advertising lighting, architectural lighting, security lighting, domestic lighting and vehicle lighting) are causing direct illumination as well as via skyglow, the scattering by atmospheric molecules or aerosols of artificial light at night that is emitted or reflected upwards (BENNIE et al., 2014a, 2015a; CINZANO; FALCHI; ELVIDGE, 2001; DAVIES et al., 2013; GASTON et al., 2013).

Because natural cycles of light have previously provided rather consistent resources and sources of information for organisms, artificial nighttime lighting has a broad range of biological impacts (GASTON et al., 2013; HÖLKER et al., 2010a; KYBA; HÖLKER, 2013). These span from gene to ecosystem levels (ASHKENAZI; HAIM, 2012; GASTON; VISSER; HÖLKER, 2015b). They include effects on the physiology, behaviour, reproductive success and mortality of species (e.g. 10–13), on their abundance and distribution (GASTON; BENNIE, 2014), and in turn on community structures and functioning (e.g. 2,15). Moreover, it seems likely that the impacts of artificial nighttime lighting interact with those of other pressures on biodiversity, including habitat loss, climate change, other forms of pollution, and invasive species (GASTON et al., 2014).

Determining the severity of these biological impacts rests, in part, on understanding of the spatial and temporal levels and patterns of artificial nighttime lighting, and particularly how these interact with those of different ecosystem, vegetation and habitat types (GASTON et al., 2014). At a global scale, virtually all natural terrestrial ecosystem types have experienced some level of exposure to artificial nighttime lighting, and those that have been most and least affected have been identified (BENNIE et al., 2015a). However, more detailed regional analyses have largely been wanting. A few evaluations exist of regional patterns of artificial nighttime lighting, but these have not tended to determine the interaction with ecosystem, vegetation or habitat types (e.g. 15,17). Of particular concern is that work on spatial patterns of artificial nighttime lighting has focussed predominantly on China, Europe and North America (BENNIE et al., 2014a; CINZANO; FALCHI; ELVIDGE, 2001; MA et al., 2012; MILLER, 2006) with almost no attention to global biodiversity hotspots. In particular, the potential environmental impacts of artificial nighttime lighting in tropical regions have been surprisingly little considered.

Aside from the often much greater levels of biodiversity that could be influenced, it remains unknown whether artificial nighttime lighting has different impacts in tropical regions compared with temperate ones. Obvious differences between tropical and non-tropical regions that might be significant are the short and rather invariant tropical periods of twilight, relatively low proportions of crepuscular and cathemeral species in tropical regions (BENNIE et al., 2014b), the greater specialism in tropical regions of some interspecific interactions that are known to be susceptible to influences from artificial nighttime light (e.g. plant-pollinator; (MACGREGOR et al., 2015; VAN LANGEVELDE et al., 2011)), and the prevalence of terrestrial species using bioluminescence which are known to be vulnerable to light pollution

(HAGEN et al., 2015; PICCHI et al., 2013; VIVIANI; ROCHA; HAGEN, 2010).

In this chapter, we determine the spatial and temporal patterns of artificial nighttime lighting across Brazil in relation to the distribution of vegetation types. Brazil makes a particularly valuable case study. As well as being the largest country in South America, it has the largest number of species of any country in the world for many major taxonomic groups (LEWINSOHN; PRADO, 2005), has high levels of species endemism, and two recognised global biodiversity hotspots (MYERS et al., 2000). Brazil also has the richest biodiversity of bioluminescent beetles in the world (COSTA, 2000).

2. Methods

2.1. Light Data

Following Bennie *et al.* (BENNIE et al., 2014a), we used nighttime stable lights annual composite images, created with data from the Defense Meteorological Satellite Program's Operational Linescan System (DMSP/OLS), downloaded from the National Oceanic and Atmospheric Administration archives (1992–2012, $n = 21$). These images capture upwardly reflected artificial nighttime light. The images are nominally at 1 km resolution, but are re-sampled from data at an equal angle of approximately 2.7 km resolution at the equator. These images cover spectral responses from 440 to 940 nm with the highest sensitivity in the 500 to 650 nm region. The spectral range encompasses the primary emissions from the most widely used sources for external lighting: low pressure sodium (589 nm), high pressure sodium (from 540 nm to 630 nm) and mercury vapour (545 and 575 nm) (CINZANO; FALCHI; ELVIDGE, 2001).

Each pixel is represented by a digital number (DN) of between 0 and 63. Zero represents darkness, while brightly lit areas saturate at values of 63. Images were inter-calibrated and drift-corrected following the method of Bennie *et al.* (BENNIE et al., 2014a). An average calibrated image for both the first (1992–1996) and the last (2008–2012) five years was created and the difference was calculated. To assess the changes over the full period time, we considered pixels increasing or decreasing by more than a threshold of 3 DN units of difference between the averages of the first and last years. It was previously observed that over 94% of observed increases in DN of more than 3 units and over 93% of observed decreases of the same magnitude were consistently related to the directions of changes on the ground (e.g., expansion or contraction of urban and industrial areas) (BENNIE et al., 2014a). Following Gaston *et al.*

(GASTON; DUFFY; BENNIE, 2015) and Duffy *et al.* (DUFFY et al., 2015), we considered pixels as exposed to artificial light when they had values higher than 5.5 DN units. By using a threshold effectively twice the detection limit for change, we defined a conservative estimate of lit area and limited the extent to which dark sites may be classified as lit due to noise in the data set or calibration errors (DUFFY et al., 2015; GASTON; DUFFY; BENNIE, 2015).

2.2. Vegetation type data

We used the vegetation map produced by the Brazilian Institute for Geography and Statistics (IBGE INSTITUTO BRASILEIRO DE GEOGRAFIA E ESTATÍSTICA, 2004), which is recommended as a good basis to compare with data obtained from remote sensing images (BUURMAN et al., 2015). This map presents both original and current vegetation. The former portrays the original vegetation classes in Brazil likely found at the time of Portuguese colonisation (IBGE INSTITUTO BRASILEIRO DE GEOGRAFIA E ESTATÍSTICA, 2004), and the latter describes the vegetation now present (IBGE INSTITUTO BRASILEIRO DE GEOGRAFIA E ESTATÍSTICA, 2004). Original vegetation includes 24 wider classes while the current is more detailed, including 52 classes (Table 1). The shapefile was produced by IBGE - Brazilian Institute of Geography and Statistics and accessed through REDD-PAC website (http://www.redd-pac.org/new_page.php?contents=data.csv) in WFS (web feature service) format.

Table 1. Vegetation classification for Brazil according to IBGE (2012). The third column corresponds to original vegetation and the fourth column to current vegetation.

Forest	Ombrophilous Forest	Dense Ombrophilous Forest	Alluvial Dense Ombrophilous Forest
			Lowland Dense Ombrophilous Forest
			Sub-Montane Dense Ombrophilous Forest
			Montane Dense Ombrophilous Forest
		Open Ombrophilous Forest	Alluvial Open Ombrophilous Forest
			Lowland Open Ombrophilous Forest
			Sub-Montane Open Ombrophilous Forest

		Mixed Ombrophilous Forest	Montane Mixed Ombrophilous Forest
			High-montane Mixed Ombrophilous Forest
	Seasonal Forest	Semi-deciduous Seasonal Forest	Alluvial Semi deciduous Seasonal Forest
			Lowland Semi deciduous Seasonal Forest
			Sub-Montana Semi-deciduous Seasonal Forest
			Montane Semi-deciduous Seasonal Forest
Non Forest		Deciduous Seasonal Forest	Lowland Deciduous Seasonal Forest
			Sub-Montane Deciduous Seasonal Forest
			Montane Deciduous Seasonal Forest
	Campinarana		Forest Campinarana
			Woody Campinarana
			Shurbland Campinarana
			Grassland Campinarana
	Savanna		Forest Savanna
			Woody Savanna
			Parkland Savanna
			Grassland Savanna
	Steppe-savanna		Forest Steppe-savanna
			Woody Steppe-savanna
			Parkland Steppe-savanna
			Grassland Steppe-savanna
	Steppe		Woody Steppe
			Parkland Steppe
			Grassland Steppe
	Pioneer formation		Alluvial Areas
			Restinga
			Mangrove
Other	Ecotone	Campinarana/Ombrophilous Forest	Campinarana/Ombrophilous Forest
		Steppe/seasonal Forest	Steppe/seasonal Forest

	Seasonal Forest /Primary Formations	Seasonal Forest /Primary Formations
	Dense Ombrophilous Forest/Mixed Ombrophilous Forest	Dense Ombrophilous Forest/Mixed Ombrophilous Forest
	Ombrophilous Forest/Seasonal Forest	Ombrophilous Forest/Seasonal Forest
	Steppe savanna /Seasonal Forest	Steppe savanna /Seasonal Forest
	Savanna/Seasonal Forest	Savanna/Seasonal Forest
	Savanna/Ombrophilous Forest	Savanna/Ombrophilous Forest
	Savanna/Primary Formations	Savanna/Primary Formations
	Savanna/Steppe-savanna	Savanna/Steppe-savanna
	Savanna/Steppe-savanna/Seasonal Forest	Savanna/Steppe-savanna/Seasonal Forest
Relict Vegetation	Relict Vegetation	High-montane Relict Vegetation Montane Relict Vegetation
Water	Water	Coastal Water Mass Continental Water Mass
Rocky Outcrops	Rocky Outcrops	Rocky Outcrops
-----		Agriculture Secondary Vegetation

The IBGE map divides vegetation into two broad classes: forests and non-forests (IBGE INSTITUTO BRASILEIRO DE GEOGRAFIA E ESTATÍSTICA, 2012). Forests are divided into Ombrophilous Forest and Seasonal Forest. The former is further divided in three physiognomies (Dense, Open and Mixed) and the last into two (Deciduous and Semi-deciduous). All of these can be classified by up to five formations: Alluvial, Lowland, Sub montane, Montane and High-montane (Table 1). Non-forests are divided into four formations: Campinarana, Savanna, Steppe-savanna, and Steppe, which in turn can be divided into up to four formations: Forest, Woody, Shrubland, and Grassland. The map also classifies pioneer formations - that encompass vegetation influenced by rivers (Alluvial Areas), by the sea (Restingas), and by both (Mangroves) - Ecotones, Relict Vegetation and Water. When considering the current vegetation, it also includes Agriculture and Secondary Vegetation classes (Table 1).

2.3. Processing

In order to define the proportional area of each vegetation type that has been exposed to artificial nighttime light, we overlaid both original and current vegetation shapefiles on the DMSP data for the most recent five years (2008-2012). We extracted both the number of lit pixels and the total number of pixels inside each vegetation type and divided the first by the second. To assess changes, we overlaid the two vegetation shapefiles on the difference between the first (1992–1996) and the last (2008–2012) five years of DMSP data. We extracted the number of increasing pixels, decreasing pixels and the total number of pixels inside each vegetation type. We divided the number of increasing and decreasing pixels by the total in each vegetation type, achieving the proportional area where artificial light has been increasing and decreasing respectively.

3. Results

Overall, the percentage of area of each vegetation type affected by increases in artificial light was higher than the percentages affected by ‘detectable’ light (Figs 1 and 2). Less than 0.00001% of the areas of vegetation types experienced decreases in brightness so we considered only the increases in the results.

Figure 1. Percentage of area of original vegetation types affected by artificial light. Horizontal bars show the percentage of total land surface area occupied by each original vegetation type that had more than 5.5 Digital Number (DN) units in 2008-2012 (red) or an increase of more than 3 DN units between 1992-2012 and 2008-2012 (blue).

Figure 2. Percentage of area of current vegetation types affected by artificial light. Horizontal bars show the percentage of total land surface area occupied by each current vegetation type that had more than 5.5 Digital Number (DN) units in 2008-2012 period (red) or an increase of more than 3 DN units between 1992-2012 and 2008-2012 (blue).

Spatial distribution of detectable light and increases in brightness followed similar patterns. The most affected areas were strongly concentrated along the coast, in the east, particularly in the southeast, while less affected areas were located in the west and in the central region (Fig. 3 A-B).

Figure 3. Spatial distribution of artificial light and vegetation types in Brazil. Distribution of: (A) pixels with detectable light ($DN > 5.5$); (B) pixels with increases in brightness (differences higher than 3 DN); (C) original vegetation types; and (D) current vegetation types. The figure was created using QGIS 2.12.3. Nighttime light images were created with data from the Defense Meteorological Satellite Program's Operational Linescan System (DMSP/OLS), freely available at the website of National Oceanic and Atmospheric Administration/National Geophysical Data Center (NOAA/NGDC) Earth Observation Group (<http://ngdc.noaa.gov/eog/>). The shapefile of Brazilian vegetation types was produced by IBGE (Brazilian Institute of Geography and Statistics) and is freely available at REDD-PAC website

(http://www.redd-pac.org/new_page.php?contents=data.csv) in WFS (web feature service) format.

3.1. Original vegetation

The area of original vegetation types affected by both detectable light and increases in brightness ranged between 0% and approximately 35%. Types affected by detectable light in more than 10% of their areas include pioneer formations (which encompass Mangroves, Restingas, and Alluvial Areas - Table 1), Semideciduous Seasonal Forest, Mixed Ombrophilous Forest, and six ecotones containing these ones and also Savanna, Steppe-savanna, Dense Ombrophilous Forest, and Steppe (Fig 1).

Less than 1% of the areas of three original vegetation types were affected by both detectable light and increases in exposure: Campinarana/Ombrophilous Forest, Savanna/Pioneer Formations, and Ombrophilous Forest/Seasonal Forest (Fig 1). Two out of 24 original vegetation types had levels of detectable artificial light at night below the threshold: Rocky Outcrops and Campinarana (Fig 1). The less affected original vegetation types were concentrated in the west and in the central area while the most affected were in the southeast and northeast (Fig 3 A, C).

3.2. Current vegetation

The area of current vegetation types affected by detectable light ranged between less than 1% and approximately 25%. Restingas, Mangroves, Secondary Vegetation, and Steppe/Seasonal Forest had more than 10% of their areas affected by detectable light (Fig 2). The first three were also the most affected by changes in brightness as well as Seasonal Forest/Pioneer Formations (Fig 2).

Vegetation types with less than 1% of their areas affected by both detectable light and increases in exposure were the three formations of Open and Dense Ombrophilous Forest (Alluvial, Lowland and Sub-montane - Table 1), Alluvial Semideciduous Seasonal Forest, Sub-Montane Deciduous Seasonal Forest and four ecotones involving Savanna, Ombrophilous Forest, Pioneer Formations (mainly Mangroves and Restingas), Campinarana and Seasonal Forest (Fig 2).

100% of the areas of seven of the 52 current vegetation types had levels of detectable artificial light lower than the threshold: Rocky Outcrops, the four formations of Campinarana

(i.e. Woody, Shrubland, Forest and Grassland - Table 1), Lowland Deciduous Seasonal Forest and Lowland Semideciduous Seasonal Forest (Fig 2).

The most affected current vegetation types were strongly concentrated along the coast, in the east. The less affected ones occurred in the west (where Amazonia rainforest is located) and in the central area (Fig 3 B, D).

4. Discussion

In this chapter, we provide the first assessment of the broad level of exposure of tropical and subtropical ecosystems to artificial light at night at a regional extent. Because the percentage of areas of the different vegetation types affected by increases in brightness was higher than those affected by detectable light in most of the cases (Figs 1, 2 and 3 A-B), it seems inevitable that the extent of artificial lighting will continue to increase.

The highest aggregations of artificial lights in Brazil are in the coastal regions (Fig 3 A-B) from where occupancy of Brazilian territory by Europeans started and where the larger urban agglomerations are now located (IBGE INSTITUTO BRASILEIRO DE GEOGRAFIA E ESTATÍSTICA, 2013). The three most widely lit vegetation types when considering original vegetation are ecotones and all of them involve Seasonal Forest or Mixed Ombrophilous Forest (Fig 1). Semideciduous Seasonal Forest and Mixed Ombrophilous Forest themselves are also widely lit by detectable light (16.7% and 13.6% respectively - Fig 1). These levels of coverage by artificial lighting are lower for current vegetation of the same types (6.18% for Montane Semideciduous Forest, 1.2% for Sub-montane Semideciduous Forest, 7.25% for Montane Mixed Ombrophilous Forest, and 3.5% for High-montane Mixed Ombrophilous Forest) because they have been highly converted and the current remnants are small (MORELLATO; HADDAD, 2000). Of the current vegetation types, Restingas, Mangroves and Coastal water mass are among the five with the greatest percentage coverage by artificial nighttime lighting (Fig 2).

Imagery of emissions of upward radiance are the best available data to assess both the presence and trends in artificial light at a regional scale (other artificial nighttime lighting data sets do not yet capture trends). However, as pointed by Bennie *et al.* (BENNIE et al., 2015a), trends established using these data must be interpreted with caution because the relationships between the images captured by the satellites and biologically relevant levels of light experienced by species are not straightforward. First, the spectral response of the OLS instrument covers the ranges of the most commonly used sources for external light, which

differs from the action spectra of biological processes depending on the species. Second, upwards radiance measures do not encompass horizontal emissions or skyglow. And finally, because DMSP/OLS images are approximately at 2.7 km resolution, the correspondence between the illuminated areas in the images and the areas at the ground surface where biologically significant levels of lights are present is not precise.

Whilst an impressively wide array of ecological impacts of artificial nighttime lighting have been documented (see Introduction), the most important effects on given vegetation types and their associated communities remain unknown. Nonetheless, Semideciduous Seasonal Forest may potentially be differentially impacted because the trees lose from 20% to 50% of their leaves during the unfavourable season (i.e. dry and cold season in tropical and subtropical zones respectively - 30) and street lighting has previously been shown in other contexts to affect leaf fall timing as well as the speed of leaf growth (HAN et al., 2015; MATZKE, 1936). Mixed Ombrophilous Forest, also known as araucaria forest due to the dominance of Brazilian pine (*Araucaria angustifolia*) (IBGE INSTITUTO BRASILEIRO DE GEOGRAFIA E ESTATÍSTICA, 2012), has a notably high richness and diversity of dung beetles (CAMPOS; HERNÁNDEZ, 2013). It is known that dung beetles exploit moonlight, the celestial polarization pattern and the starry sky for orientation (BYRNE et al., 2003; DACKÉ et al., 2003, 2004, 2013). Given the important role of dung beetles in decomposition and nutrient cycling in tropical ecosystems, it seems likely that the high levels of artificial light and increase in brightness found in Ombrophilous Mixed Forest will affect its functioning.

Restinga is the terrestrial pioneer vegetation that occurs on sandy shore environments, especially on dunes, and is directly influenced by the sea (IBGE INSTITUTO BRASILEIRO DE GEOGRAFIA E ESTATÍSTICA, 2012). Restinga harbours a high diversity of bats (LINS et al., 2009; LUZ et al., 2011; OPREA et al., 2009), which are known to be important for the maintenance of forests and to be disturbed by artificial light (DAY et al., 2015; LEWANZIK; VOIGT, 2014; STONE; JONES; HARRIS, 2009). Around the world, mangroves are threatened by deforestation, illegal shrimp culture, expansion of urban areas, tourism, fishing and pollution (FAO - FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS, 2007). Nine percent of the global area of natural or semi natural mangroves has seen an increase in exposure to artificial light (BENNIE et al., 2015a). In Brazil this percentage is 17% in the same period (Fig 2), with more than 15% of the mangrove area experiencing detectable light (Fig 2). Given that Brazil accounts for approximately 50% of mangroves in South America and 7% of the world's mangroves (FAO - FOOD AND AGRICULTURE ORGANIZATION OF THE

UNITED NATIONS, 2007), light pollution in these areas should be of particular concern. Both Restinga and Mangrove are coastal ecosystems and the coastal water mass itself is also highly affected by light (Fig 2). Five out of seven extant species of marine turtles in the world nest on the Brazilian coast (*Chelonia mydas*, *Caretta caretta*, *Dermochelys coriacea*, *Eretmochelys imbricata*, and *Lepidochelys olivacea*) - all of them are listed as threatened on the IUCN Red List (<http://www.iucnredlist.org/search>). Artificial lighting disrupts sea turtle hatchling orientation from the nest to the sea (LORNE; SALMON, 2007). In order to protect Brazilian coastal ecosystems, the law forbids illumination within 50m of the beach strip between Rio de Janeiro and Rio Grande do Norte States - which corresponds to approximately 2 500 km out of the 7 367 km of Brazilian coast (IBAMA - INSTITUTO BRASILEIRO DO MEIO AMBIENTE E DOS RECURSOS RENOVÁVEIS, 1995). Due to the scarcity of studies on the consequence of light pollution in these ecosystems, it is not possible to assess if the law is effective.

In most developing countries artificial nighttime lighting is relatively recent and concentrated in dense populated urban areas (FALCHI et al., 2016). In contrast, in highly industrialised countries it is much more widespread (BENNIE et al., 2015a; CINZANO; FALCHI; ELVIDGE, 2001), and often considered thus to be a much greater concern. However, our results here highlight that lighting is extensive in some developing countries, including ones with exceptionally high levels of biodiversity. These results also suggest that it is still possible to find vegetation types with natural sky background brightness. Countries in which this is the case have the opportunity to base policies, regulations, and guidelines on minimising rather than mitigating the ecological impacts of artificial nighttime lighting.

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CAPÍTULO 2

Assessing the relative impact of agricultural expansion in ecosystem services provision in Cerrado (Brazilian Savanna)

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Abstract

Agriculture is acknowledged as a highly detrimental activity on the environment and its impacts on different biodiversity levels have been widely assessed while effects on the ecosystem services have been often overlooked. Using land cover maps for 16th century period, years 2000 and 2012, we mapped provisioning of ecosystem and environment services (pollination, carbon stocks and living space) in Cerrado (Brazilian Savanna). We also compared how these services have changed across time associated to land cover changes and agricultural expansion. Provisioning of services associated with original vegetation has been reduced drastically over time in all agricultural land cover assessed. We found that conversion of original vegetation leads to higher depletion of services rather than the conversion of lands that have already been converted. We recommend the establishment of new strictly protected areas in MATOPIBA region, the newest agricultural frontier in Brazil, which has been identified as valuable area for ecosystem services provisioning.

Keywords: ecosystem services, Cerrado, Brazilian savanna, pollination, carbon stocks, living space, agriculture.

1. Introduction

Understanding the anthropogenic threats to areas of high biological importance is essential in order to turn conservation efforts effectively deployed (BELLARD et al., 2014). Acknowledged as a highly detrimental activity, the impacts of agriculture have been extensively assessed in areas of exceptional endemism (JINGGUT; YULE; BOYERO, 2012; KLINK; MACHADO, 2005; RIBEIRO et al., 2009; ZALIDIS et al., 2002), as the relative loss of

geographically constrained species can have disproportionate costs for conservation (MARGULES; PRESSEY, 2000). Such assessments, however, have been often focused on biodiversity elements, including habitats (BENTON; VICKERY; WILSON, 2003), species (BUTLER; VICKERY; NORRIS, 2007), and genetic diversity (VAN HEERWAARDEN et al., 2009), while effects on ecosystem services - i.e the benefits people obtain from biodiversity and ecosystems (MEA- MILLENIUM ECOSYSTEM ASSESSMENT, 2005) – and environmental services (i.e. benefits people obtain from the environment not directly related attributed to ecosystem *per se* - (CASALEGNO et al., 2014) – have been overlooked (DALE; POLASKY, 2007). Given the increasing interest in incorporating biodiversity and these services within one integrated conservation strategy (e.g. IPBES, CBD), it is important to understand the response of ecosystem services to key threats within areas of high endemism.

Pollination and carbon stock are ecosystem services that can be affected by intensification and expansion of agriculture (GILBERT, 2012; POTTS et al., 2010). The current declining of pollinators observed around the world is particularly concerning in regions with high levels of endemism and threats (IPBES, 2016; VAMOSI et al., 2006). In those regions, plants that have co-evolved with specialist pollinators are more prone to extinction when local pollinator populations decline (VAMOSI et al., 2006). Furthermore, belowground biomass have not received the same degree of study as the aboveground component (MOKANY; RAISON; PROKUSHKIN, 2006). Thus, carbon stocks in non-forest vegetation that has the highest proportion of its biomass underground have been underestimated (MOKANY; RAISON; PROKUSHKIN, 2006), diminishing its value for conservation.

Living space, the geographical space that enables humans' survival and their activities, has been often defined as an environmental service. Not-urban living space takes particular relevance when it benefits local communities that have historically been associated to one geographical region or habitat (DIEGUES, 1996). When the living space of such communities aims to be reconciled with the conservation of endemic species and the provision of alternative ecosystem services important socio-ecological trade-offs can arise, as the substitution or relocation of these services elsewhere often present strong ethical and logistic debate (WEST; IGOE; BROCKINGTON, 2006). Countries such as Brazil have presented great progress in this regard through the establishment of official Indigenous Lands (ARAÚJO, 2004), where the service of living space advocates protection. Nevertheless, this has been increasingly threatened by agricultural expansion (SULLIVAN, 2013). Mapping living space with ecosystem services help to visualise and identify spatial and temporal trade-offs between them (BURKHARD et

al., 2012; CASALEGNO et al., 2014).

The Brazilian Cerrado is recognized not only for its unique levels of endemism, but also for its high level of habitat conversion and degradation (i.e. biodiversity hotspot; Myers et al. 2000). It is among the largest and richest Neotropical savannas in the world, harbouring more than 2,500 species of vertebrates (KLINK; MACHADO, 2005), 4,000 species of plants restricted to Cerrado (CASTRO et al., 1999), with high levels of phylogenetic and genetic diversities (BATALHA et al., 2011; ROSATTI; SILVA; BATALHA, 2015). Up to 70% of the biomass of plants in Cerrado is underground (CASTRO; KAUFFMANN 1998). The area encompasses eleven Brazilian States with more than 1500 municipalities (MMA, 2003). Beyond urban communities, Cerrado also harbours indigenous communities, the most ancient people living there (MISTRY et al., 2005; WATANABE et al., 2003). Historically, they faced territorial conflicts with colonisers and also with other ethnicities that migrated from the coast towards the inland during the occupancy by colonisers (MMA, 2003). They still face territorial conflicts with agribusiness lands and additional indirect impacts of agriculture expansion on their lifestyle (e.g. loss of biodiversity and contamination of soil and water - BICALHO; MIRANDA, 2015; GOMIDE; KAWAKUBO, 2006).

So far more than 40% of Cerrado's native vegetation cover has been converted to agriculture, mostly to cattle and soybean production (KLINK; MACHADO, 2005; RADA, 2015). Only 8% of Cerrado's territory is under protection and the annual conversion rate achieved 1.1% since 1985 (MACHADO et al., 2004). Brazilian soy moratorium (*i.e.* a voluntary agreement of major soybean traders not to purchase soy grown on deforested lands) has successfully reduced land conversion, but this, however, applies only to the Amazon rainforest (GIBBS et al., 2015). The National Program of Conservation and Sustainable Use of Cerrado (MMA 2003) has highlighted the assessment of the effects of agricultural expansion as critical to underpin conservation strategies. While in Cerrado such effects have been assessed for habitat (CARVALHO; DE MARCO; FERREIRA, 2009), biodiversity (FRANÇOSO et al., 2015), and social costs (FEARNSIDE, 2001), few studies have addressed the impact of agriculture on ecosystem services and none of them has mapped ecosystem services in Cerrado at regional scale.

Here, we contribute in filling this knowledge gap by mapping two ecosystem and one environmental service (pollination, carbon stocks and living space, respectively) over time. We also assessed the relative impact of agricultural expansion on these services. Specifically, we asked: 1) How were pollination, carbon stocks and living space distributed within the original

vegetation in Cerrado? 2) How are pollination, carbon stocks and living space distributed in recent land cover in Cerrado? and 3) To what extent pollination, carbon stocks and living space has increased or decreased in areas where natural vegetation has been converted to agricultural lands?

2. Methods

2.1. Land cover maps in Cerrado

Using land cover maps for 16th century period (hereafter ‘original vegetation map’), years 2000 and 2012, we mapped ecosystem and environmental services provisioning in these periods and compared how agricultural land cover types has affected the provision of them. We obtained the Brazilian original vegetation shapefiles produced by the Brazilian Institute for Geography and Statistics (IBGE, 2004), downloaded without cost from REDD-PAC website (http://www.redd-pac.org/new_page.php?contents=data.csv) in WFS (web feature service) format. Original vegetation map presents 31 classes of vegetation including 10 ecotones (see Table S1 on Supplementary Material).

Land cover maps for the years 2000 and 2012 were obtained from IBGE website (ftp://geofp.ibge.gov.br/mapas_tematicos/mapas_murais/shapes/) in a vector format. The map contains 13 classes of land cover: agriculture, artificial area, open area, continental water, mosaic agriculture and forest vegetation, mosaic non-forest vegetation and agriculture, mosaic forest vegetation and agriculture, natural pastures, planted pastures, silviculture, non-forest vegetation, flooded non-forest vegetation, and forest vegetation. Both original vegetation and land cover map were produced in 1:5000000 scale. We clipped both maps using official boarder of Cerrado vegetation. All maps were processed and analysed using South American Albers Equal Area Conic projection.

2.2. Mapping services

The quantification and mapping of pollination, carbon stocks and living space were based on the land cover maps for the three different periods (16th century, 2000 and 2012; see details below). Resulting services maps were rasterized and resampled (average values weighted by the area) at 10 km resolution. This resulted in a total of 9 rasters (3 ecosystem

services in 3 periods), which allowed us to assess changes on services' spatial distribution and provision across time.

The original vegetation cover map presents natural vegetation only. We considered no presence of crop land cover at that time as farming practices by indigenous before Europeans arrived in South America were synchronized with Cerrado's natural fire regimes, hence not been highly detrimental (DENEVAN, 1992; MISTRY et al., 2005; SILVA et al., 2011).

2.2.1. Pollination

We measured pollination by adapting values from Kennedy *et al* (2013). Assuming that wild bees are the most important pollinators in natural and agricultural areas, the authors analysed data from 605 field sites in 39 studies in different biomes, including tropical and subtropical. The authors of each study provided values of nesting suitability and floral resources for different land cover types. Kennedy *et al* (2013) totalled nesting and floral values across different bee taxa and multiple seasons respectively, rescaled from 0 to 1 within each study and then calculated the average for standardized global land cover types. Highest overall habitat suitabilities were assigned to natural and semi-natural areas (i.e. Shrubland, grassland, forest and woody wetlands) and to a lesser extent certain croplands (i.e orchards and vineyards, pasture and fallow lands) and low density developed and open spaces.

Based on the description of each land cover types provided the authors, we established correspondence between the global land cover types used by Kennedy (2013) and the land cover types from IBGE (2012) used here. Then, we assigned values of nesting suitability and floral resource for wild bees for each land cover class in both original vegetation and land cover maps of Cerrado. The values range from 0 to 1 such that a cover type with the highest overall nesting and floral resources was assigned as 1 and the lowest as 0 (Kennedy and others 2013). The correspondence between land cover types is exposed with details in the Supplementary Material (Tables S1 and S2).

2.2.2. Carbon stocks

Following (Durán *et al.* (2014), we calculated carbon stocks from above and below ground vegetation biomass (in tonnes of carbon). Aboveground vegetation data were obtained following the IPCC GPG Tier-1 method for estimating vegetation carbon stocks using the global default values provided for above ground biomass (AALDE et al., 2006; LASCO et al.,

2006). Below ground vegetative biomass (root) carbon stock was added using the root-to-shoot ratios for each vegetation type obtained from the same IPCC document (AALDE et al., 2006). Total living vegetation biomass was converted to carbon stock using the carbon fraction for each vegetation type. All estimates and conversions were specific to each of the three ecofloristic zones in Cerrado (FAO 2001). More details about the carbon calculation are provided in the Supplementary Material (Tables S3 and S4).

2.2.3. Living space

While living space has been previously defined as urban developed areas (CASALEGNO et al., 2014), in this study we expanded this definition incorporating areas occupied by indigenous communities. Thus, we defined three types of living space: i) Only Urban, ii) Only indigenous, and iii) Both urban and indigenous (hereafter *total living space*). Following Casalegno et al. (2014) we calculate living space as the proportion of each grid with urban areas or indigenous land according to the definition above. Urban areas were obtained from land cover map provided by IBGE. Indigenous land shapefile was downloaded from FUNAI – Fundação Nacional do Índio website (<http://www.funai.gov.br/index.php/shape>).

2.2.4. Temporal change on ecosystem services

For a given ecosystem service, we calculated the difference between the original and 2000 ecosystem service rasters, and between 2000 and 2012 ecosystem service rasters. The rasters obtained as a result of the difference (i.e. hereafter *difference rasters*) had, in each pixel, the change in a given ecosystem service during that period. Pixels with positive values indicate an increase in the ecosystem service while pixels with negative values indicate a decrease. Zeroes indicate no change. In order to assess the extent to which ecosystem services provision has changed due to agricultural expansion, *difference rasters* were overlapped with agricultural land cover classes and the average of the pixel values falling within each polygon boundary was calculated. Agricultural land classes were: agriculture, mosaic agriculture and forest vegetation, mosaic non-forest vegetation and agriculture, mosaic forest vegetation and agriculture, planted pastures, and silviculture (Fig 1).

Figure 1: Study area. Location of Cerrado in Latin America and land cover in Cerrado in 2012.

3. Results

The provision of ecosystem and environmental services associated with original vegetation has been reduced drastically over time (Figs. 2, 3 and 4). Not surprisingly the highest reduction for all ecosystem services occurred between 16th century (original vegetation) and year 2000. In this period, a significant reduction in pollination was observed. When Cerrado was covered by original vegetation, approximately 52%, 47% and 1% of the area had high (>0.75), medium (0.26-0.75) and low (0.1-0.25) values of pollination provisioning, respectively (Fig. 2A). In year 2000, areas with high pollination values (>0.75), were absent, areas with medium values (0.26-0.75) declined in 25% and low value areas (0.1-0.25) increased in 77% (Fig. 2B). Strong reduction on carbon stocks was also observed in this period. Values higher than 100 tonnes/ha were originally found in 96% of the area and this was reduced to 5%, while values lower than 100 tons/ha, originally observed in 4% of the area, increased to 95% in 2000 (Fig. 3 A and B).

Figure 2. Assessment of pollination provision in original and current land cover in Cerrado. Spatial distribution values of nesting suitability and floral resource per pixel in: a) Original vegetation cover, in 16th Century. b) in the year 2000 and c) in the year 2012.

Figure 3. Assessment of carbon stocks in original and current land cover in Cerrado. Spatial distribution of values of tons of carbon per hectare per pixel in: a) original vegetation cover, in 16th Century. b) the year 2000. c) the year 2012.

Figure 4. Assessment of area of original and current living space ecosystem service in Cerrado. Spatial distribution of percentage of area of: a) indigenous land per pixel in 16th Century. b) indigenous land and urban areas per pixel in 2000. c) indigenous land and urban areas per pixel in 2012.

As we assumed indigenous communities could use the entire extension of the country before colonisation, original living space covered the whole extent of Cerrado evenly (Fig. 4 A). In 2000, living space was restricted to indigenous lands and urban areas. Indigenous land reduced from 100% to 8% mainly concentrated in East of Cerrado (in Mato Grosso and Tocantins States). Urban areas represented less than 1% of Cerrado, observed widespread in the South (in Goiás, Minas Gerais and São Paulo States - Figs. 4 B and C). Indigenous land and urban areas distribution are exposed in detail in Supplementary Material (Figs. S1 and S2).

Between 2000 and 2012 the pollination service presented a further decline (Fig. 2 B and

C). Areas with low values increased from 78% to 84%, while areas with medium values declined from 22% to 16% (Fig. 2 B and C). This reduction was particularly high in the North-East of Cerrado (i.e. Tocantins State - Fig. 2C). For carbon stocks similar patterns were found, with a small reduction (1%) in areas with less than 51 tonnes/ha whilst areas with values between 51 and 100 tonnes/ha increased (Fig. 3C). This increment in carbon stock was associated to the expansion of the silviculture and the mosaic non-forest vegetation and agriculture land cover classes (Table 1). Even after the reductions, the highest values for both pollination and carbon stocks were concentrated in the North-east and in the West of Cerrado. Total living space increased approximately 6% (Fig. 4).

Table 1: Differences of ecosystem services provisioning between 16th Century, 2000 and 2012. The values are the average ($\pm$ SD) of the pixel values falling within polygon's borders of each class of agriculture activity. Positive values indicate increase and negative values indicate decrease.

		Agriculture	Mosaic Agriculture and forest vegetation	Mosaic non-forest vegetation and agriculture	Mosaic forest vegetation and agriculture	Planted pastures	Silviculture
16th Century – 2000 period	Pollination	-0.62 ($\pm$ 0.29)	-0.63 ($\pm$ 0.30)	-0.61 ($\pm$ 0.30)	-0.62 ($\pm$ 0.30)	-0.60 ($\pm$ 0.28)	-0.67 ($\pm$ 0.30)
	Carbon	-95.40 ($\pm$ 32.01)	-80.58 ($\pm$ 37.78)	-91.79 ($\pm$ 35.26)	-79.26 ($\pm$ 38.11)	-97.20 ($\pm$ 29.41)	-86.26 ($\pm$ 31.48)
	Indigenous living space	-0.96 ($\pm$ 0.17)	-0.94 ($\pm$ 0.23)	-0.96 ($\pm$ 0.19)	-0.92 ($\pm$ 0.26)	-0.96 ($\pm$ 0.17)	-0.97 ($\pm$ 0.15)
	Urban living space	0.00 ($\pm$ 0.01)	0.00 ($\pm$ 0.01)	0.00 ($\pm$ 0.01)	0.00 ($\pm$ 0.01)	0.00 ($\pm$ 0.01)	0.00 ($\pm$ 0.01)
	Total living space	-0.96 ($\pm$ 0.17)	-0.92 ($\pm$ 0.25)	-0.96 ($\pm$ 0.17)	-0.91 ($\pm$ 0.27)	-0.96 ($\pm$ 0.17)	-0.97 ($\pm$ 0.15)
2000 - 2012 period	Pollination	-0.02 ($\pm$ 0.07)	-0.01 ($\pm$ 0.04)	-0.03 ($\pm$ 0.08)	-0.01 ($\pm$ 0.05)	-0.03 ($\pm$ 0.09)	-0.02 ($\pm$ 0.07)
	Carbon	-0.18 ($\pm$ 10.90)	-4.56 ($\pm$ 12.56)	7.57 ($\pm$ 11.89)	-4.87 ($\pm$ 13.70)	-0.16 ($\pm$ 9.86)	2.10 ($\pm$ 8.71)
	Indigenous living space	-0.96 ($\pm$ 0.17)	-0.94 ($\pm$ 0.23)	-0.96 ($\pm$ 0.19)	-0.92 ($\pm$ 0.26)	-0.96 ($\pm$ 0.17)	-0.97 ($\pm$ 0.15)
	Urban living space	0.00 ($\pm$ 0.00)	0.00 ($\pm$ 0.00)	0.00 ($\pm$ 0.00)	0.00 ($\pm$ 0.00)	0.00 ($\pm$ 0.01)	0.00 ($\pm$ 0.00)
	Total living space	0.00 ($\pm$ 0.00)	0.00 ($\pm$ 0.00)	0.00 ($\pm$ 0.00)	0.00 ($\pm$ 0.00)	0.00 ($\pm$ 0.01)	0.00 ($\pm$ 0.00)

All ecosystem services were reduced in most of agricultural land types. Increases were observed only for carbon stock, in areas covered by silviculture and mosaic non-forest vegetation and agriculture when comparing 2000 and 2012 (2.10 ton/ha and 7.82 ton/ha respectively -Table 1).

Between 16th century and 2000, planted pastures presented the lowest decline of pollination service (-0.60 units) and the highest for carbon, indigenous and total living space (-97.2 tonnes/ha and -0.96% of area respectively). Silviculture was the most detrimental activity to pollination (-0.67 units) and mosaic forest vegetation and agriculture were the less damage carbon, indigenous and total living space (-79.26 ton/ha and -0.92% of area respectively; Table 1).

Between 2000 and 2012, mosaic non-forest vegetation and agriculture had the highest reduction for pollination (-0.03 units) and increased carbon stocks (+7.57 ton/ha). Mosaic agriculture and forest vegetation presented the lowest reduction of pollination (-0.01 units) and mosaic forest vegetation and agriculture had the highest reduction for carbon and the lowest for indigenous living space (-4.87 ton/ha and -0.92% of area respectively). No significant changes were observed for total, urban, and indigenous living space in this period (Table 1).

4. Discussion

In this paper, we assess the variation of ecosystem and environmental services through time and space as well the relative impacts of agricultural activities on the provisioning of these services. Provisioning of all ecosystem services assessed has diminished in the Cerrado area through time. Here, we identify spatial remnants with high levels of services provisioning as well as overlaps and discuss their location focusing on conservation actions. We found decays in services in most of agricultural lands in both periods – although we also identify some unexpected increases. We expose the causes for the variations and raise their implications for conservation.

We found high levels of pollination and carbon stocks in the North of Cerrado (Fig. 2 and 3). This area is located in the newest agricultural frontier in Brazil, named MATOPIBA (an acronym with the initials of the states involved: Maranhão, Tocantins, Piauí, and Bahia – BRASIL, 2015). As we also show most of agricultural activities area associated with declines in ecosystem provisioning (Table 1), we suggest the creation of strictly protected areas in that region. The creation of new protected areas in MATOPIBA was already recommended by

Brazilian Environment Ministry in 2003 (MMA, 2007). Once their assessment disregarded ecosystem services, our findings strengthen such indication.

The conservation of ecosystem services in MATOPIBA region could also be reinforced with the extent of the soy moratorium to the Cerrado (WWF - Brasil 2012; Gibbs and others 2015). Currently, more than 50% of the natural vegetation suitable to agriculture in Cerrado can be legally converted (Gibbs and others 2015). Although these areas are spread all over Cerrado, conversion in MATOPIBA is more prone to occur due to the low prices of the land in that region (WWF - Brasil 2012). Soybean crops are the highest threat in Cerrado and particularly high in MATOPIBA (WWF - Brasil 2012). By incentivising its expansion into already cleared areas, soy moratorium may lead to replacement of other land cover - possibly pastures, as observed in Amazonia - instead of natural vegetation (Gibbs and others 2015). This would reduce the pressure for conversion of original vegetation lands in MATOPIBA and maintain ecosystem services provision in this region. In addition, according to our approach, conversion from planted pastures to agriculture would represent a lower decline in pollination and carbon stocks in comparison with the conversion of any original vegetation in Cerrado to agriculture.

Remnants of high values of pollination and carbon were also found in the West of Cerrado, which overlaps indigenous living space (Figs 1, 2 and 3). This highlights the importance of those areas not only for its cultural value for indigenous, but also to maintain ecosystem services provisioning. As FUNAI makes available only one Indigenous land shapefile, the increase in living space in 2000-2012 period are due to urban expansion. This also explains why no change in living space is observed within all agricultural land cover in that period (Table 1). Our approach contributes to visualize the dimension in which agriculture has been affected indigenous living space through time.

Pollination has declined in all land cover assessed in both periods - albeit the variation in reduction within land covers was not large. In the first period, pollination presented the lowest reduction in planted pastures due to high scores attributed to this land cover in comparison with the other agricultural land cover assessed (see supplementary Material Tables S1 and S2). However, in the second period, pollination presented the highest reduction in planted pastures. Between 16th Century and 2000, the replacement among land cover was not selective: all agricultural land cover has replaced all original vegetation types in the same rates. In turn, between 2000 and 2012, planted pastures has replaced mainly nature pastures (IBGE, 2015), which has higher score (see Table S2 in the supplementary material). These findings highlight the importance of maintaining natural pastures. Even though natural and planted pastures are

similar physiognomies, the conversion in one to another implies in high reduction of pollination supply.

We expected to find highest reductions in carbon stocks in the most intensively managed land covers - agriculture and planted pastures. On the contrary, carbon stocks increased and presented higher reductions in land cover composed by mosaics of natural vegetation and agriculture than by Agriculture and Planted pastures (Table 1).

During the second period, the Mosaic non-forest vegetation and agriculture replaced mainly original vegetation, probably Savannas and Steppe Savanna while Agriculture have replaced mainly land covers already cleared (i.e. Mosaic non-forest vegetation and agriculture, and planted pastures). As the difference in pollination value in first situation is higher than in the second, our results suggest that conversion of original vegetation usually leads to larger depletion of ecosystem services provisioning in comparison with conversion of already cleared areas. This suggestion is reinforced by the fact that all services decayed more between 16th Century and 2000 than between the years 2000 and 2012 within agricultural land cover.

In sum, we recommend the establishment of new strictly protected area in MATOPIBA region and highlight the importance of maintaining indigenous living space for ecosystem services provisioning. We also show that conversion of original vegetation may lead to larger depletion of ecosystem services, which accentuate the importance of creating and maintain strictly protected areas.

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Supplementary Material

Assessing the relative impact of agricultural expansion in ecosystem services provision in Brazilian Savanna (Cerrado)

Table S1: Nesting suitability and floral resources values produced by Kennedy (2013) for standardised global land types and their correspondence with IBGE vegetation types (2004) used to calculate the original pollination in Cerrado.

Standardized land cover types (KENNEDY et al., 2013)			Original vegetation type in Cerrado (IBGE 2004)	Nesting suitability and floral resources values (KENNEDY et al., 2013)
Level 1	Level 2	Level 3		
Natural and semi-natural			Primary formations (restingas)	0.6
			Savana/Seasonal Forest	0.6
			Steppe Savanna/Seasonal Forest	0.6
			Seasonal Forest/Primary Formations	0.6
			High-montane relict vegetation	0.6
			Steppe Savanna/Seasonal Forest	0.6
			Savana/Seasonal Forest	0.6
			Savanna/Ombrophilous Forest	0.6
			Savanna/Primary formations	0.6
			Savanna/Steppe Savanna	0.6
			Savanna/Steppe Savanna/seasonal Forest	0.6
		Broadleaved forest	Montane Deciduous Seasonal Forest	0.71
			Sub-Montane Deciduous Seasonal Forest	0.71
			Lowland Deciduous Seasonal Forest	0.71
			Montane Semideciduous Seasonal Forest	0.71
			Sub-Montane Semideciduous Seasonal Forest	0.71
			Alluvial Dense Ombrophilous Forest	0.71
			Dense Ombrophilous Forest	0.71
			Deciduous Seasonal Forest	0.71
			Semideciduous Seasonal Forest	0.71
			Forest Savanna	0.6
			Grassland Savanna	0.64
	Shrubland	Shrubland	Savanna	0.8
			Woody Savanna	0.8
			Parkland Savanna	0.8
			woody Steppe-savanna	0.8
	Wetlands	woody wetlands	Primary formations (mangroves)	0.61
			Primary formations (alluvial areas)	0.38
Unsuitable	Open water		Coastal water mass	0
			Continental water mass	0

Table S2: Nesting suitability and floral resources values produced by Kennedy et al., (2013) for standardised global land types and their correspondence with 2000 and 2012 land use-land cover maps (IBGE, 2015) used to calculate current pollination.

Standardized land cover types (KENNEDY et al., 2013)			Land use-land cover classes in the years 2000 and 2012 (IBGE 2015)	Description of the land use- land cover classes	Nesting suitability and floral resources values (KENNEDY et al., 2013)
Level 1	Level 2	Level 3			
Natural and Seminatural	<i>Forest</i>		Mosaic forest vegetation and agriculture	Polygons that contains more than 50% and less than 75% of its area occupied by forest and the rest by agriculture	0.6
			Forest vegetation	More than 75% of the polygon is occupied by forest, (i.e. vegetation where woody species are higher than 5m) including Dense Ombrophilous Forest, Open Ombrophilous Forest, Seasonal Forest, Mixed Ombrophilous Forest, and mangroves.	<i>0.6</i>
	<i>Grassland/h erbaceous</i>		Natural pastures	Polygons with non-forest vegetation with low intense anthropic impacts	<i>0.64</i>
	<i>Shurbland</i>	Shurbland	Non-forest vegetation	More than 75% of the polygon is occupied by non-forest physiognomies vegetation such as Savannas, Steppes, Steppe-Savanna, Primary formations and Relict vegetation.	0.8
	<i>Wetlands</i>	herbaceous wetlands	Flooded non-forest vegetation	Areas where natural grasslands are permanently or seasonally flooded for at least two months	0.29
	<i>Cultivated/R uderal vegetation</i>		Mosaic Agriculture and forest vegetation	Polygons that contains more than 50% and less than 75% of its area occupied by agriculture and the rest by remnants of natural vegetation or regeneration of forests and natural pastures.	<i>0.38</i>
			Mosaic non-forest vegetation and agriculture	Polygons that contain more than 50% and less than 75% of its area occupied by non-forest vegetation and the rest by perennial or annual crops, pasture and silviculture.	<i>0.38</i>
	<i>Grassland</i>	Pasture/fall ow fields	Planted Pasture	Areas dominated by cultivated herbaceous vegetation for livestock grazing, intensively managed.	<i>0.42</i>
Cultivated	<i>Cropland</i>		Silviculture	Polygons where more than 75% of the area is occupied by exotic trees.	<i>0.36</i>
			Agriculture	More than 75% of the polygon is occupied by annual and perennial crops, irrigated or not. It is all the lands used for fibre and commodities production for agribusiness, including flooded and fallow lands.	<i>0.33</i>
Developed			Artificial area	More than 75% of the polygon occupied by urban uses where buildings and road systems dominates the landscape, instead of agriculture. Includes cities, towns, villages, highways, power networks, industrial and commercial complexes, indigenous communities and mining areas.	0.33
Unsuitable	<i>Barren or sparsely vegetated</i>		Open area	Rocky outcrops, clives, rockies, reefs and eroded areas. Includes abandoned mining areas without vegetation, and mobile and permanent dunes.	<i>0.21</i>
	<i>Open water</i>		Continental water	Includes all continental water such as rivers, streams, lakes, lagoons and artificial tanks or pools (related to irrigation, flood control, water provision and power generation).	<i>0</i>

Table S3: Correspondence between Brazilian original vegetation types and Ecofloristic zones used to assign original carbon stock values from Aalde et al., (2006). For the ecotones we assigned the most conservative value

Original vegetation types (IBGE 2004)	Ecofloristic zones (FAO 2001)	Carbon values (tonnes C/ha ⁻¹)
Dense Ombrophilous Forest	Tropical moist deciduous forest (TAwa)	128
Deciduous Seasonal Forest	Tropical moist deciduous forest (TAwa)	128
Savanna	Tropical moist deciduous forest (TAwa)	128
Coastal water mass	Water	0
Continental water mass	Water	0
Montane Deciduous Seasonal Forest	Tropical moist deciduous forest (TAwa)	128
Sub-Montane Deciduous Seasonal Forest	Tropical moist deciduous forest (TAwa)	128
Lowland Deciduous Seasonal Forest	Tropical moist deciduous forest (TAwa)	128
Montane Semideciduous Seasonal Forest	Tropical moist deciduous forest (TAwa)	128
Sub-Montane Semideciduous Seasonal Forest	Tropical moist deciduous forest (TAwa)	128
Seasonal Forest/Primary Formations	Tropical moist deciduous forest (TAwa)/Tropical rain forest (TAra)	128
Alluvial Dense Ombrophilous Forest	Tropical rain forest (TAra)	193
Semideciduous Seasonal Forest	Tropical rain forest (TAra)	193
High-montane relict vegetation	Tropical rain forest (TAra)	193
Woody Savanna	Tropical moist deciduous forest (TAwa)	128
Steppe Savanna/Seasonal Forest	Tropical dry forest (TAwb)/Tropical moist deciduous forest (TAwa)	126
Savana/Seasonal Forest	Tropical moist deciduous forest (TAwa)	128
Forest Savanna	Tropical moist deciduous forest (TAwa)	128
Grassland Savanna	Tropical moist deciduous forest (TAwa)	128
Parkland Savanna	Tropical moist deciduous forest (TAwa)	128
woody Steppe-savanna	Tropical dry forest (TAwb)	126
Savana/Seasonal Forest	Tropical moist deciduous forest (TAwa)	128
Steppe Savanna/Seasonal Forest	Tropical moist deciduous forest (TAwa)	128
Savanna/Ombrophilous Forest	Tropical moist deciduous forest (Tawa)/Tropical rain forest (TAra)	128
Savanna/Primary formations	Tropical moist deciduous forest (Tawa)/Tropical rain forest (TAra)	128
Savanna/Steppe Savanna	Tropical moist deciduous forest (Tawa)/Tropical dry forest (TAwb)	126
Savanna/Steppe Savanna/seasonal Forest	Tropical moist deciduous forest (Tawa)/Tropical dry forest (TAwb)/Tropical rain forest (TAra)	126
Primary formations (alluvial areas)	Tropical rain forest (TAra)	193
Primary formations (mangroves)	Tropical rain forest (TAra)	193
Primary formations (restingas)	Tropical rain forest (TAra)	193

Table S4: Assumptions and references used to assign carbon stock for each the land use-land cover class from 2000 and 2012 land use-land cover maps (IBGE, 2015).

Land use-land cover classes in Cerrado in the years 2000 and 2012 (IBGE 2015)	Description of the land use- land cover classes	Assumptions/References	Values (tonnes/ha -1)
Agriculture	More than 75% of the polygon is occupied by annual and perennial crops, irrigated or not. It is all the lands used for fibre and commodities production for agribusiness, including flooded and fallow lands.	As cropland encompass annual and perennial croplands, we considered the default value for above ground biomass carbon stock at harvest for perennial species (LASCO et al., 2006).	21
Artificial area	More than 75% of the polygon occupied by urban uses where buildings and road systems dominates the landscape, instead of agriculture. Includes cities, towns, villages, highways, power networks, industrial and commercial complexes, indigenous communities and mining areas.	Assumed as zero	0
Open area	Rocky outcrops, clives, rockies, reefs and eroded areas. Includes abandoned mining areas without vegetation, and mobile and permanent dunes.	Assumed as zero	0
Continental water	Includes all continental water such as rivers, streams, lakes, lagoons and artificial tanks or pools (related to irrigation, flood control, water provision and power generation).	Assumed as zero	0
Mosaic Agriculture and forest vegetation	Polygons that contains more than 50% and less than 75% of its area occupied by agriculture and the rest by remnants of natural vegetation or regeneration of forests and natural pastures.	Consider the dominate (agriculture)	21
Mosaic non-forest vegetation and agriculture	Polygons that contains more than 50% and less than 75% of its area occupied by non-forest vegetation and the rest by perennial or annual crops, pasture and silviculture.	Consider the dominant (non-forest vegetation)	128
Mosaic forest vegetation and agriculture	Polygons that contains more than 50% and less than 75% of its area occupied by forest and the rest by agriculture	Consider the dominant (forest vegetation)	128
Natural pastures	Polygons with non-forest vegetation with low intense anthropic impacts	Default value for above ground biomass carbon stock at harvest for perennial species (LASCO et al., 2006).	21
Planted pastures	Polygons where the landscape is dominated by planted grasslands used as fallow for the cattle.	Default value for above ground biomass carbon stock at harvest for perennial species (LASCO et al., 2006).	21
Silviculture	Polygons where more than 75% of the area is occupied by exotic trees.	We considered the default value above ground biomass for Eucalyptus sp in Americas (AALDE et al., 2006) and the root-to-shoot ratio. in the Cerrado, Eucalyptus account for forest plantations in percentages significantly higher than any other species (SERVIÇO FLORESTAL BRASILEIRO, 2015).	90
Non-forest vegetation	More than 75% of the polygon is occupied by non-forest physiognomies vegetation such as Savannas, Steppes, Steppe-Savanna, Primary formations and Relict vegetation.	Values for Tropical moist deciduous forest (TAwa) – (FAO, 2001)	128
Flooded non-forest vegetation	Areas where natural grasslands are permanently or seasonally flooded for at least two months	Values for Tropical rain forest (TAra) – (FAO, 2001)	193
Forest vegetation	More than 75% of the polygon is occupied by forest, (i.e. vegetation where woody species are higher than 5m) including Dense Ombrophilous Forest, Open Ombrophilous Forest, Seasonal Forest, Mixed Ombrophilous Forest, and mangroves.	Values for Tropical moist deciduous forest (TAwa) – (FAO, 2001)	128

Percentage of area

Figure S1: Percentage of area of indigenous land per pixel in Cerrado in (A) 2000 and (B) 2012.

Percentage of area

Figure S2: Percentage of area of urban areas per pixel in Cerrado in (A) 2000 and (B) 2012.

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CAPÍTULO 3

An overview of the applicability of functional diversity in Biological Conservation

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Abstract: Functional diversity is increasingly pointed as a useful approach to reach Biological Conservation goals. Here, we provide an overview of the status of functional diversity approach in Biological Conservation field. We seek for peer-reviewed papers published along twenty years (from 1994 until 2014) by using, firstly, the general topic "functional diversity". Next, we refined our search with the key-word "conservation". We identified the conservation strategies addressed the organism studies and the continent of study site in each paper present in the results. We found that the number of papers regards to functional diversity published per year were higher than the ones that attached it with conservation. Thirteen classes of conservation strategies were identified. Plants was the group of organism most commonly studied and most of study-sites were located in Europe. Functional diversity is a new and promisor approach to improve biodiversity assessments and to develop ecological indicators of habitat quality, conservation value and ecosystem services. Although the number of papers that operationalise the inclusion of this approach in conservation strategies being still low, it is expected that the contribution of extended peer community, typical from interdisciplinary fields improve operationalisations and applications. Future studies should focus on the variation of functional diversity across scales and across taxa, its relationships with ecosystem services, and the improvement of traits measurements of other taxa instead of plants. Testing hypothesis in tropical ecosystem would also contribute to expand current understanding of applicability of functional diversity in conservation strategies.

Keywords: trait-based approach, biodiversity assessments, ecosystem functioning, conservation strategies, biodiversity measures.

1. Introduction

Biological Conservation aims to stop species extinction and maintain ecosystem functioning and services (MEFFE; CARROLL, 1963). Ecosystem functioning is the set of matter and energy flows through biotic and abiotic ecosystem components (DÍAZ; CABIDO, 2001). As it is assumed that biological diversity reflects ecosystem functioning reasonably, managers employ measures of species diversity in order to assess ecosystem functioning

(MAGURRAN, 2004; RICKLEFS, 2003). Indices of biodiversity more commonly used are based on number of species and their abundance, thus, they assume that each species contributes equally to ecosystem functioning (MAGURRAN, 2004). However, ecosystem functioning is maintained by the interactions among species and with the environment such as photosynthesis, herbivory, pollination, seed dispersal, and nutrient cycling (BLÜTHGEN, 2011; HOOPER et al., 2005). As a consequence, the contribution of each species to the ecosystem functioning rely on its characteristics and interactions and not only on the species number (TILMAN, 1997).

Functional diversity is a component of biodiversity that encompass species features and as a consequence may predict ecosystem functioning more precisely (CIANCIARUSO; SILVA; BATALHA, 2009; DÍAZ; CABIDO, 2001). Measures of functional diversity are based on functional traits, i.e. characteristics of species that can be measured at individual level and are related to species growth, reproduction and survival (CORNELISSEN et al., 2003; PÉREZ-HARGUINDEGUY et al., 2013). The trait-based approach is closely related to the niche models: different species complement each other, raising efficiency of processes while similar species substitute each other, raising ecosystem resilience (BLÜTHGEN, 2011; MACARTHUR, 1972). Indices of functional diversity measure how species are distributed in the multidimensional niche space (MOUCHET et al., 2010; PETCHEY; GASTON, 2002, 2006; VILLÉGER; MASON; MOUILLOT, 2008).

The concept of functional diversity and the trait-based approach are not new (GITAY; NOBLE, 1997; GRIME, 1998). Until the 1990s ecologists were mainly interested in creating classification systems focused on the role of species in ecosystems, explaining how traits were influenced by biotic and abiotic factors, and finding patterns of distribution of these groups (LAURETO; CIANCIARUSO; SAMIA, 2015). The consolidation of Biological Conservation as an interdisciplinary science field during the 1980's, (FRANCO, 2013; MEINE, 2010) the United Nations Conference on Environment and Development in 1992 (so-called Rio-92) raised the necessity of understanding the impacts of human activities on the biological diversity and its consequences. Thereafter scientists gradually changed the focus of functional diversity questions towards the relationships between species traits and ecosystem functioning and services (DÍAZ et al., 2007; DÍAZ; CABIDO, 1997; HOOPER et al., 2005; SRIVASTAVA et al., 2009).

An increasing number of authors has pointed out functional diversity approach as useful to reach biological conservation goals (e.g. CADOTTE; CARSCADDEN; MIROTECHNICK, 2011; CADOTTE, 2011; GILLISON et al., 2013; ISBELL et al., 2011; SCHWARTZ et al., 2000; TRINDADE-FILHO; LOYOLA, 2010, 2011; TRINDADE-FILHO et al., 2012;

TRIVELLONE et al., 2014).

Here, we review the literature in order to evaluate the status of functional diversity approach in Biological Conservation issues discuss trends and gaps of knowledge. We oriented our review by following three questions: 1) How many papers regarding to functional diversity have been published over time? 2) Among them, how many are related to biological conservation? 3) On which conservation strategy are they focused? 4) How these studies are distributed within different organisms? 5) How study sites are distributed among the continents?

2. Methods

We considered peer-reviewed papers published along twenty years (from 1994 until 2014) in journals concerned about Agricultural and Biological Sciences and Environmental Sciences using Science Direct search engine. We began with the general topic "functional diversity". Next, we refined the search with the key-word "conservation". After reading the abstract, we excluded papers that were not concerned about ecology (for instance, conservation of functional diversity of structure of enzymes and proteins). We also excluded papers out of this revision scope - where authors measured functional diversity using taxonomic indices and/or defined 'functional diversity' unclearly, as well as papers in which the relationship of the findings and biological conservation were not clear. After exclusions, we identified the conservation strategies addressed in each paper. Finally, we classified them according to the organism studied and the continent of study site.

3. Results

In total we compiled 621 documents regarding functional diversity and from those, 60 papers (approximately 9.5%) regarding functional diversity and biological conservation (Fig. 1). Nine papers were out of the scope so that we considered 51 in this revision. The number of papers regarding to functional diversity published per year were higher than the ones attached with conservation. Functional diversity has been present in academic issues since the 1994, and the number of papers published has been increasing (Fig. 1). In conservation issues, functional diversity appeared for the first time in 1998. It appeared once more in 2001, remained absent until 2005 and scarce until 2012, when it starts to increase significantly.

Figure 1 - Number of papers published per year between 1994 and 2014 regards to functional diversity (blue) and functional diversity and conservation (orange).

Regards to conservation strategies, we identified thirteen classes: biodiversity assessment (16 papers), impacts of land use (11), management (8), spatial conservation planning (8), indicator (7), ecosystem services (6), habitat loss (3), restoration (3), monitoring (2), priority areas (2), biological control (1), economic valuation of biodiversity (1), and resilience (1). Some papers were classified as more than one strategy, so that the sum is not equal to 51 (Fig. 2A). According to the organisms studied we found nine classes: plant (16), multitaxa and/or networks (8), birds (7), Arthropoda (6), fishes (4), soil organisms (4), theoretical (4), algae (1), and reef (1). Soil organism class included all organism that lives in the soil, excluding Arthropoda - for instance, earthworms, bacteria, fungi, lubricious, nematodes, and saprophytic organisms. Theoretical was the group that encompassed papers concerned about theoretical issues and discussion, without focusing on a particular organism or group. Networks and/or Multitaxa encompassed papers concerned about more than one organism, considering interactions between them or not (Fig. 2B). Regards to the study site we found nine categories: Europe (25), theoretical (6), Latin America (4), all over the world (4), Australia (2), Africa (1), Asia and Europe (1), Asia (1), and North America (1). Similarly to organism classification, theoretical was the group that encompassed papers concerned about theoretical issues and discussion, without focusing on a particular study site. The class all over the world encompassed papers that used metadata or data used samples for more than three continents and focused the discussion in global level (Fig. 2C).

Figure 2 - Distribution of papers focused on functional diversity and conservation published between 1994 and 2014 according to (A) conservation strategies, (B) organism studies, (C) continent of the study site.

4. Discussion

Publications concerned about functional diversity has increased over time. After pioneer papers, an uptake of the knowledge by other research groups occurs, which in turn, lead to more papers published in that field. The number of papers per year regards to functional diversity was considerably higher than the ones that relate it to conservation (Fig.1). This may be resultant of the paradigm of knowledge transfer in which scientists may work on specialised issues and translate the results into useful knowledge to whom are responsible for solving practical problems (FRANCIS; GOODMAN, 2009; TURNHOUT; HISSCHEMÖLLER; EIJSACKERS, 2007).

Until the early 2000-decade functional diversity concept has been developed in a disciplinary approach and was restrict to academic community (Fig.1). Once it enter into the Biological Conservation field, which is interdisciplinary, the paradigm is expected to change: an extended peer community may contribute to develop, apply and improve the knowledge, including academic from other fields, governmental agencies and non-governmental organisation (FRANCIS; GOODMAN, 2009; FUNTOWICZ; RAVETZ, 1993; TURNHOUT; HISSCHEMÖLLER; EIJSACKERS, 2007). Though this change depends on how effective is the transfer of knowledge – i.e. in which dimension the knowledge produced *de facto* reach the extended peer community (FRANCIS; GOODMAN, 2009). This is not to say that the development of the disciplinary knowledge is expected to cease. On the contrary, it might proceed, mainly curiosity-motivated, as it is expected for pure sciences, with occasional support to the application of functional diversity on conversation (FUNTOWICZ; RAVETZ, 1993).

In the first paper regards to functional diversity and conservation, published in 1998, authors compared functional patterns with plant composition and concluded that functional diversity does not necessarily follow the same patterns as the species diversity (Fig.1). Thus, they recommended the inclusion of functional trait-based approach in conservation strategies (MÉDAIL; ROCHE; TATONI, 1998). This recommendation was enhanced in 2001, in a review that evidence links between plant functional diversity and ecosystem functioning (DÍAZ; CABIDO, 2001). It became a key-paper regards to conservation and functional diversity.

Biodiversity assessments were the most common issue found in the papers analysed (Fig. 2A). Most of them focused on the responses of different metrics of biological diversity to environmental and spatial variation including functional diversity and also phylogenetic diversity, relative rarity, ecological specialisation, acoustic diversity (BENNETT et al., 2014; DENGLER et al., 2014; GASC et al., 2013; GRANGER et al., 2014; LEROY; LE VIOL; PÉTILLON, 2014; SCHIRMEL; BLINDOW; BUCHHOLZ, 2012; SCHULDT; ASSMANN; SCHAEFER, 2013; WINTER; DEVICTOR; SCHWEIGER, 2013). Overall, authors recommend the inclusion of other facets of biodiversity beyond traditional indices on conservation strategies, with emphasis on functional diversity.

Papers classified as impacts of land use, management, and spatial conservation planning were the second, third and fourth classes of conservation strategies respectively (Fig. 2 A). Overall authors focused on find which traits of organisms can provide benefits to humans as well as identify the impacts of the management of land on functional diversity. These findings are crucial to support spatial conservation planning and to conciliate economic activities and conservation of nature. Among the papers regards to impacts of land use, only one was

addressed to the impacts of urbanisation (FILIPPI-CODACCIONI; CLOBERT; JULLIARD, 2009). They observed that extreme ages of urbanisation (young and old) cause higher niche differentiation concerning specialisation of birds, leading to a more vulnerable state for the ecosystem.

The high number of papers concerned to agricultural land use is justifiable as farming is highly damaging to wild nature (BALMFORD; GREEN; PHALAN, 2012; SWINTON et al., 2007). Functional diversity responds differently and eventually more precisely to changes in management than other measures, particularly the traditional ones (BENNETT et al., 2014; DENGLER et al., 2014; JIGUET et al., 2012; KATOVAI; BURLEY; MAYFIELD, 2012; PELOSI et al., 2014; VAN CAPELLE; SCHRADER; BRUNOTTE, 2012). Overall, land use intensification such as tillage, mowing, use of tracked mowers and increase nitrogen deposition had negative impact on functional diversity of different organisms (KRAUSE; CULMSEE, 2013; LILLESKOV; HOBBIE; HORTON, 2011; PELOSI et al., 2014). While there is a relative consensus that reducing the intensification of land use may contribute to maintain functional diversity, abandonment of lands also seems to have negative impacts (FONTANA et al., 2014). Thus the challenge thereafter is to find the optimum intensity management of land in order to maintain functional diversity.

High spatial heterogeneity found in mosaics landscapes seems to be effective in maintaining functional diversity (ATHAYDE et al., 2015; AZCÁRATE et al., 2013; CHILD; CUMMING; AMANO, 2009; GRIFFITHS et al., 2007; KRAUSE; CULMSEE, 2013; MARTIN et al., 2012). In an extraction versus conservation competing scenario, two approaches have been suggested (FISCHER et al., 2008; GREEN et al., 2005): (i) land sparing (protect some land very strictly and exploit the rest intensively) and (ii) land sharing (protect less land but exploit the remainder with friendly practices). Our results suggest that land sharing approach has been corroborated when functional diversity is considered. Nevertheless, we emphasize that functional diversity is influenced by spatial scale patterns (AZCÁRATE et al., 2013; MAIRE et al., 2013; RICKLEFS; SCHLUTER, 1994) and its variation is not completely understood yet (DEVICTOR et al., 2010). More comparative results are necessary to corroborate these ideas and to support generalization of the results (DENGLER et al., 2014).

Functional diversity has been pointed as a promisor approach to develop ecological indicators, the fifth class of conservation strategies (Fig. 2A - DE BELLO; MUDRÁK, 2013; KACHERGIS; ROCCA; FERNANDEZ-GIMENEZ, 2013). Maintaining the quality of ecosystems functioning is one of the aims of biological conservation (SODHI; EHRLICH, 2010; TROMBULAK et al., 2004). Given quality is a subjective concept, ecological indicators

enable to turn it into objectives parameters connecting it with conservation (TURNHOUT; HISSCHEMÖLLER; EIJSACKERS, 2007). Ecological indicators are biological parameters that respond to chemical, physical, and biological phenomena (NIEMI; MCDONALD, 2004). Metrics of functional diversity of different organisms has been used as indicator of habitat quality and conservation value, and thus it bases recommendation of management and selection of priority areas to conservation (BISHOP; MYERS, 2005; LEROY; LE VIOL; PÉTILLON, 2014; MAIRE et al., 2013). Likewise, it has been incorporated to broader indicators framework such as the driver-pressure-state-response approach proposed by European Environment Agency (KADOYA et al., 2011). Functional diversity is also used to find biodiversity indicators (also known as surrogates - CARO; DOHERTY, 1999). A group of birds is recommended as a shortcut to conservation planning in Atlantic rainforest (TRINDADE-FILHO et al., 2012) and a list of plant species is pointed as effective indicator of biological diversity of agroecosystems in the Alps (TRIVELLONE et al., 2014).

After indicator, the next class of conservation strategy ranked in our results is ecosystem services (Fig.2A). Presence and production of ecosystem services are more effectively explained by functional diversity rather than species number (CIMON-MORIN; DARVEAU; POULIN, 2013). However, the relationships between the traits and ecosystem services are not simple and direct. A given ecosystem service can be provided by one functional trait existent only in dominant species, while other originate from the range of variety of functional traits in the community. Likewise, the same trait can be linked to several ecosystem services (CIMON-MORIN; DARVEAU; POULIN, 2013; LAURETO; CIANCIARUSO; SAMIA, 2015). Understanding the multiple associations between traits and ecosystem processes along through different trophic levels can help to identify predictable trait–service clusters, for instance clusters of traits of plants and soil organisms that underlie nutrient cycling. Thus, assessments of trait–service clusters would represent a step in ecosystem service monitoring and in balancing the delivery of multiple ecosystem services (DE BELLO et al., 2010).

As plant ecology field started the first studies with functional diversity (DÍAZ; CABIDO, 2001; MÉDAIL; ROCHE; TATONI, 1998), the high number of papers using plants is not surprisingly (Fig. 2B). Also, structure of vegetation are assumed to be associated with habitat diversity, which underscore its importance to conservation (RICKLEFS, 2003). Indeed, functional groups of plants describes robustly diversity of mammals, birds and termites (GILLISON et al., 2013). On the other hand, it has also been found that successful restoration of plant diversity does not necessarily imply successful restoration of soil nematode functional diversity (KARDOL et al., 2005). This contradiction raises the importance of studies that

disentangle the relationships between functional diversity of different taxa. Fortunately, multitaxa and/or network is the class with higher number of papers after plants (Fig. 2B).

Birds and Arthropoda are together the third class of organism studied (Fig. 2B). As birds act as genetic linkers, carrying genetic information between habitats and most of Arthropoda are strongly associated with nutrient cycling (LUNDBERG; MOBERG, 2003), functional diversity of these organisms have a great potential to be applied in conservation strategies. Basic knowledge of birds functional diversity is fairly well explored (PETCHEY et al., 2007; TRINDADE-FILHO et al., 2012). Though the inclusion of functional diversity of Arthropoda in conservation issues still needs more improvements such as defining appropriate functional diversity index for benthic macroinvertebrate (KADOYA et al., 2011).

The number of papers published that has studies sites located in Europe are considerably higher than any other (Fig 2C). As most part of Europe lies in the northern temperate climate zone, current conclusions and recommendations regards to functional diversity and conservation tends to be more suitable to ecosystems from that climate. Because of its high biological diversity, the relationships between species in the tropics are prone to be more complex (JANZEN, 1975; VAMOSI et al., 2006). More studies on functional diversity and conservation in tropical region as a study site are necessary as well as assessing in which extent usually accepted assumptions are suitable for these environments.

In sum, we show that the inclusion of functional diversity approach in biodiversity assessments is pointed as a promisor approach to improve conservation strategies. Although this consensus, the number of papers that operationalise such inclusion by developing models and systems that encompass functional diversity is still low (KADOYA et al., 2011; MAIRE et al., 2013; MOUILLOT et al., 2008; PERRY, 2010; ROBINSON et al., 2014). This may be explained by the short time since functional diversity entered to the conservation issues. It is expected that the contribution of extended peer community, tipical from interdisciplinary fields improve operationalisarions and applications. We also show that metrics of functional diversity has a great potential to develop indicator of habitat quality, conservation value and ecosystem services. Future studies should focus on the the variation of functional diversity across scales and across taxa, its relationships with ecosystem services, and the improvement of traits measurements of other taxa instead of plants. Testing hypothesis in tropical ecosystem would also contribute to expand current understanding of applicability of functional diversity in conservation strategies.

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CAPÍTULO 4

Incorporating functional diversity as indicator of conservation in the Cerrado

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Abstract It is recommended that conservation strategies embraces other components of biodiversity than taxonomic. Functional diversity is an approach that consider the characteristics of species in a given community, thus is associated with ecosystem functioning. We aim to find plant species associated with high values of taxonomic and functional diversity in Cerrado (Brazilian Savanna), Brazil. We placed 20 plots (100m²) in two protected areas in São Paulo State, where we sample all woody species. We calculated two taxonomic and three functional indices for each plot, which were, in turn, labelled as high (H), medium (M) and low (L) biodiversity according to each index. Using the Indicator Value analysis, we identified plant species significantly associated with high biodiversity (H) plots. In total, 65 plants species were sampled and six were significant indicator of high values of biodiversity indices. All of them are locally abundant, widespread distributed and none of them are listed as rare species. Our study highlights the role of common species in the ecosystem and the importance to include functional component in conservation strategies. We indicate a possible way to incorporate plant functional diversity into conservation strategies in Cerrado areas. Also we indicate how to improve our approach and adapt it for other vegetation type and other organisms.

Keywords: Biodiversity indicators, Functional diversity, Cerrado, Brazilian Savanna

1. Introduction

Assessing the status of biodiversity and the integrity of ecosystems are indispensable tasks in order to develop effective conservation actions (MARGULES; PRESSEY, 2000; MEFFE; CARROLL, 1963). Measures of biodiversity or surrogates that represents the whole biodiversity of a region are recommended as indicators of the status of conservation (CARO; DOHERTY, 1999; MARGULES; PRESSEY; WILLIAMS, 2002; NOSS, 1990). However most of them are based on taxonomic criteria and might fail in reflect ecosystem functioning, i.e. the flow of matter and energy through biotic or abiotic ecosystem components (DÍAZ; CABIDO, 2001; HOOPER et al., 2001). Ecosystem functioning is maintained because species interact with the environment and with each other in many ways, including pollination, seed dispersal, herbivory, nutrient cycling, and photosynthesis (BLÜTHGEN, 2011). Therefore, ecosystem

functioning may change depending on differences and similarities in species functions: different species complement each other, raising efficiency of processes while similar species substitute each other increasing ecosystem resilience (HOOPER et al., 2005). Once biodiversity measures based the number of species assume that each species contributes equally to ecosystem functioning (MAGURRAN, 2004), they disregard the role of each species in the community (HOOPER et al., 2005; TILMAN, 1997).

Functional diversity is an approach in which communities can be thought as a multidimensional space (the total niche) in which the niches of all species fit. Thus, adding or removing species in these space has consequences for the ecosystem functioning as the niche size of other species may either increase or decrease (DÍAZ; CABIDO, 2001). The characteristics of the species included or removed are essential in predicting such consequences. Measures of functional diversity are based on functional traits, i.e. characteristic of species that can be measured at individual level and are related to species growth, reproduction and survival (CORNELISSEN et al., 2003; PÉREZ-HARGUINDEGUY et al., 2013). Indices of functional diversity measure differences between species in the multidimensional space (MOUCHET et al., 2010; PETCHEY; GASTON, 2002, 2006; VILLÉGER; MASON; MOUILLOT, 2008). A growing number of authors have been exploring the potential of functional diversity as indicator of biodiversity (BADY et al., 2005; DE BELLO; MUDRÁK, 2013; GILLISON et al., 2013; TRINDADE-FILHO et al., 2012; TRINDADE-FILHO; LOYOLA, 2011; TRIVELLONE et al., 2014; VANDEWALLE et al., 2010). No studies have been addressed to incorporate plant species as indicator of functional diversity in natural tropical ecosystems.

Cerrado (also known as Brazilian Savanna), a hotspot for biodiversity conservation (MYERS et al., 2000), is composed by different physiognomies: Cerradão (Forest Savanna), Cerrado sensu strictu (Woody Savanna), Campo sujo (Parkland savanna) and Campo limpo (Grassland – COUTINHO, 1978). There is a continuous gradient from Cerradão (Forest) to Campo limpo (Grassland) in which canopies, basal area, and woody species tends to decrease whereas herbaceous cover in soil increase (COUTINHO, 1978; IBGE, 2012). In São Paulo state, Cerradão (Forest) is the commonest physiognomy (DURIGAN et al., 2003). It has a high ecological value because it is the transition between Savanna and Semidecidual Seasonal Forest, propagating seeds of trees to other Cerrado physiognomies (DURIGAN; RATTER, 2006). It is highly threatened and it has the lowest representativeness in protected areas in São Paulo State (DURIGAN et al., 2007). Studies focused on conservation of Cerrado in this state are the rarest among the terrestrial ecosystems (METZGER; CASATTI, 2006).

Officially, the assessment of the conservation status of Cerrado includes density of

woody individuals, basal area, stem diameter and percentage of herbaceous land cover (São Paulo 2009). It is unknown in which extend these traits encompass functional diversity. Ecological indicators have been proposed to selected priority areas for conservation in Cerrado. Although they involve important criteria for conservation strategies, such as biophysical features, external influences, and integrity of natural resources, they do not include functional diversity directly (DURIGAN et al., 2006). In this paper, we incorporate functional diversity as indicator of conservation in Cerradão. We seek for plant species related to sites with high values of functional and taxonomic diversity in Cerradão in the São Paulo State.

2. Material and Methods

2.1. Study area Experimental design

We collect data in two protected areas covered by Cerradão in São Paulo State (Fig. 1): Moji-Guaçu Biological Reserve (22°18'S e 47°11'W; 600m of altitude; 470ha), and Mogi Mirim Experimental Station (22°26'S 46°57'1W; 617m of altitude; 145.6ha). The climate in this region is classified as Cwa (KÖPPEN, 1931) with wet summer (from October to March) and dry winter (from April to September). The mean annual temperature varies from 20,5°C to 22,5°C, and the mean annual rainfall varies from 1400mm to 1600mm - achieving maximum average of 1250mm from October to March (in the rain season) and minimum average of 250mm from April to September (during the dry season - SMA 2015).

Figure 1 - Study area. **A.** Area occupied by cerrado in São Paulo State, Brasil. **B.** Moji-Guaçu Biological Reserve in Mogi-guaçu (SP), Brasil. **C.** Mogi Mirim Experimental Station, in Mogi-Mirim (SP), Brasil.

In each area, 10 square plots of 100m² were randomly established (20 in total). All individuals of the tree layer with at least 3 cm of base diameter (SMA 1997) were identified to the species level (APG III, 2009) using an identification guide (MANTOVANI; LEITÃO-FILHO; MARTINS, 1985) and specialist knowledge. For eventual corrections and updates we used Species List of Brazil's Flora (<http://floradobrasil.jbrj.gov.br>) through the package "Flora" (CARVALHO, 2015) available in R environment 3.1.3 (R DEVELOPMENT CORE TEAM, 2014).

2.2. Biodiversity Indices

From February 2013 to December 2014 we surveyed six functional traits for all species: height, basal area, leaf area, leaf phenology, reproductive phenology and dispersal syndrome

(Table 1). For quantitative traits (height, basal area, leaf area), we considered the mean of five individuals of each species randomly sampled (CORNELISSEN et al., 2003; PÉREZ-HARGUINDEGUY et al., 2013). For qualitative traits (leaf phenology, reproductive phenology and dispersal syndrome), we consulted literature available (for details see Supplementary Material 1).

Table 1. Functional traits surveyed in this paper, its definition and details of measures.

Functional trait	Definition	Type	Unit	NA entries
Height	The shortest distance between the upper boundary of the main photosynthetic tissues (excluding inflorescences) on a plant and the ground level.	Continuous	M	1
Basal area	The area of stem near the ground (considering the stem as a cylinder)	Continuous	cm ²	2
Leaf area	The one-sided or projected area of an individual leaf	Continuous	mm ²	2
Leaf phenology	The number of months per year that the leaf canopy is green.	Categorical	3 categories: Evergreen, Semi-deciduous and Deciduous	4
Reproductive phenology	Months of flowering.	Categorical	Dummy variable 0-1 for Flowering or Not flowering in each month	10
Dispersal syndrome	The mode of the propagule (unit of seed, fruit or spore) is dispersed	Categorical	3 levels: Autocoric, Anemocoric, Zoocoric	2

^aWe considered each month one trait and weighed it by 1/12.

For each plot we calculated two taxonomic and three functional diversity indices: Richness (Rich), Shannon-Wiener Index (H'), Functional Divergence (FDiv), Functional Evenness (FEve) and Functional Dispersion (FDis). Richness is the number of species within a given area (MAGURRAN, 2004). Shannon-Wiener Index account the evenness in which abundances are distributed among species (MAGURRAN, 2004). Functional Evenness quantifies the regularity with which the functional space is filled weighted by the abundance of each species. It decreases either when abundance is less evenly distributed among species or when functional distances among species are less regular (VILLÉGER; MASON; MOUILLOT,

2008). Functional Divergence takes the relative abundances of the species into account and it is related to how abundance is distributed within the volume of functional multi-space. Functional Dispersion is the mean distance of each species to the centroid of the multidimensional space weighted by species abundances: it can change the position of the centroid toward the most abundant species (LALIBERTE et al., 2010). Each functional index measures a different aspect of functional diversity and is statistically independent of each other (MOUCHET et al., 2010). Functional indices were calculated using the dbFD function from FD package (LALIBERTÉ; LEGENDRE; BILL SHIPLEY, 2014) in R environment (R DEVELOPMENT CORE TEAM, 2014).

2.3. Data Analysis

Following Trivelloni (2014), we first defined groups of sampling plots with more similar value for the biodiversity indices, then we identified plant species significantly associated with the high biodiversity groups of plots. We ranked the plots according to the values of each index in three levels by using the percentiles 33 (p33) and 66 (p66). Plots that had values lower than p33 or higher than p66 for a given index were labelled as L (low biodiversity) or H (high biodiversity) respectively to that index. Plots that had values between p33 and p66 were labelled as M (medium biodiversity - for details see Supplementary Material 2).

Plant species associated to plots with high values of each index (hereafter indicator species) were identified using Indicator Value analysis (DUFRÊNE; LEGENDRE, 1997). Indicator Value (IndVal) is an index that measures the strength of the relationship between one species and a given group of sites. It is based in two components: A (specificity) and B (fidelity). Component A (specificity) seeks for species that are present in at least one site belonging to one group and occurs only in that group. Component B (fidelity) seeks species that are present in all sites belonging to a given group although it can be found in sites of other groups. The value of the association varies between 0 (no association) and 1 (when all individuals of one species are found in all sites belonging to one group - DUFRÊNE; LEGENDRE, 1997). Statistical significance (p-value <0.05) was obtained by randomization procedure with 999 permutations (DE CÁCERES; LEGENDRE, 2009). We performed the analysis using Indic-species package (DE CACERES; JANSEN, 2015) in R environment (R DEVELOPMENT CORE TEAM,

2014).

3. Results

We sampled 798 individuals, belonging to 65 species and 34 families in both areas. The most common species were *Siparuna guianensis* Aubl. (Siparunaceae) with 165 individuals, *Ocotea pulchella* (Nees & Mart.) Mez (Lauraceae) with 118 individuals and *Xylopia aromatica* (Lam.) Mart (Annonaceae) with 63 individuals. There were 20 species with only one individual. The richest families were Fabaceae (with nine species), Myrtaceae (with six species), Primulaceae and Myrtaceae (with five species each). Families with only one species were 22 (For details on samples in each area, see Supplementary Material 3).

Six species were related to four out the five indices considered: *Lacistema hasslerianum* Chodat (Lacistemataceae), *Miconia albicans* Sw Triana (Melastomataceae), *Miconia paucidens* DC (Melastomataceae), *Myrsine umbellata* Mart. (Mysinaceae), *Rudgea viburnoides* (Cham.) Benth (Rubiaceae), and *Vochysia tucanorum* Mart (Vochysiaceae - Table 2). No species were related to Functional Evenness. All indicator species were more abundant than the average in both areas (Table 3). Most of them occurred in less than 50% of the plots - the exception was *Vochysia tucanorum*, which occurs in 70% of the plots (Table 3).

Table 2. Indicator species found in IndVal analysis for each index considered. Stat: strength of the relationship. A: Component A of IndVal (Specificity). B: Component B of IndVal analysis (Fidelity). P value: statistical significance.

Indicator Species	Index	A	B	Stat	p-value
<i>Lacistema hasslerianum</i> Chodat	FDis	0.708	0.857	0.779	0.016 *
<i>Miconia albicans</i> Sw Triana	Rich	0.9114	0.5714	0.722	0.049 *
<i>Miconia paucidens</i> DC.	Rich	0.8750	0.7143	0.791	0.005 **
<i>Myrsine umbellata</i> Mart.	H	0.7895	0.7143	0.751	0.030 *
<i>Rudgea viburnoides</i> (Cham.) Benth	H	0.7879	0.8571	0.822	0.012 *
<i>Rudgea viburnoides</i> (Cham.) Benth	Rich	0.7425	0.8571	0.798	0.017 *
<i>Vochysia tucanorum</i> Mart.	FDis	0.607	1.000	0.779	0.024 *
<i>Vochysia tucanorum</i> Mart.	FDiv	0.718	1.000	0.847	0.003 **

Table 3. Abundance of species selected as indicator species. Relative abundance: number of individuals of the species divided by the total number of individuals in the sample. Average of abundance per plot: number of individuals of that species in the sample divided by the number of plots in which that species occurs. Proportion of occurrence: percentage of plots in which that species occurs. Mean: the mean of the whole species sampled.

Indicator Species	Abundance	Relative Abundance	Average of abundance per plot	Proportion of occurrence (%)
<i>Lacistema hasslerianum</i> Chodat	23	0.029	2.56	45
<i>Miconia albicans</i> Sw Triana	23	0.020	2.6	25
<i>Miconia paucidens</i> DC.	8	0.010	1.33	30
<i>Myrsine umbellata</i> Mart.	25	0.030	2.5	49
<i>Rudgea viburnoides</i> (Cham.) Benth	48	0.060	5.33	45
<i>Vochysia tucanorum</i> Mart.	57	0.071	4.07	70
Mean	3	0.004	1	10

4. Discussion

This paper presents a set of plant indicator species that cover more than one aspect of biodiversity that can be used for monitoring and diagnosis biodiversity conservation in cerrado at local scale, particularly when the goal of conservation is to maintain both taxonomic and functional diversity. As each species is associated with a different biodiversity index, we recommend their use together, as a set. The absence of species related to Functional Evenness can be related to the low sensibility of this index in communities with small number of species. Functional Evenness performed inefficiently in detecting assembly rules and in predicting plant biomass in communities with less than 30 species, which is the case of our sample (LOIOLA, 2014; MOUCHET et al., 2010 - Table 2).

All indicator species found are common, i.e. they are locally abundant (Table 2) and widespread distributed (FREITAS, 2015; GOLDENBERG, R.; CADDAAH, 2015; MARQUETE; MEDEIROS, 2015). Only *Lacistema hasslerianum* is restricted to Cerrado vegetation type (MARQUETE; MEDEIROS, 2015), all other species occur in at least two vegetation types (FREITAS, 2015; GOLDENBERG, R.; CADDAAH, 2015). Moreover, none of them are included in the List of Rare Species of Cerrado (MARTINELLI; MESSINA; FILHO, 2014).

Finding common species as indicators for functional diversity endorses recent understandings. Given they make up a disproportionate amount of assemblage biomass, one may expect a higher contribution of common species to ecosystem functioning than the rare ones (GASTON, 2011; GRIME, 1998). Regards particularly to plant species, commonness is

related to primary production, aboveground carbon stocks, and resistance to invasion (BUNKER et al., 2005; EMERY; GROSS, 2007; SMITH; KNAPP, 2003). In this context, the significance of rare species would lie mainly in buffering changes in that function (the insurance hypothesis), by providing other species to become common when environmental conditions change instead of contribute to the functioning directly (GASTON, 2011). In fact, abundant plant species has also been found associated with high taxonomic and functional biodiversity values in temperate agroecosystems when using a similar approach used in this study (TRIVELLONE et al., 2014). Also, common species of insects are associated with ecosystem services delivery (WINFREE et al., 2015) and rare species of herbaceous plant presents low functional complementarity (LOIOLA et al., 2010).

On the other hand, our results contradict other recent findings. It has been shown that rare species are likely to carry the most distinct set of traits, supporting the most vulnerable functions in the ecosystems (MOUILLOT et al., 2013). Moreover, functional diversity is highly affected by the loss of rare species (LEITÃO et al., 2016) and can be efficiently represented by a set of restricted range species of birds in Atlantic Forest (TRINDADE-FILHO et al., 2012). The contribution of rare and common species in different assemblages remain uncertain. Our results can contribute to the debate as it points to an association of common species to functional diversity.

All indicator species found are native and *Miconia paucidens* is endemic of Brazil (GOLDENBERG, R.; CADDAGH, 2015). None of them are included in Brazilian Flora Red List (MARTINELLI; MESSINA; FILHO, 2014). This fact highlights that conservation strategies focused only on endemic and threatened species can disregard the functional component of biodiversity. This is not to say that they may not be important. On the contrary, they might be highly efficient when the target is to protect themselves due to their intrinsic value (CARO; DOHERTY, 1999). Nevertheless, it is known that focusing solely on these species may not be sufficient to preserve overall species diversity in country scale (BONN; RODRIGUES; GASTON, 2002). Our results underscore that protect endemic and threatened species does not necessarily maintain ecosystem functioning.

It is worth to notice that our study was designed at local scale and therefore results cannot be unrestrictedly extrapolated (MAGURRAN, 2004). While the variation of traditional biodiversity measurements across scale is relatively well established in the literature, for functional diversity such variation remains inconclusive (DEVICTOR et al., 2010; MOUCHET et al., 2010; VANDEWALLE et al., 2010). Other studies are necessary to investigate which species would be associated with indices in regional scale.

Functional diversity studies are relatively new and as a consequence, it still presents many gaps (LAURETO; CIANCIARUSO; SAMIA, 2015). However, as species richness can miss more than 88% of the biodiversity information (LYASHEVSKA; FARNSWORTH, 2012) and functional diversity may be under-represented in protected areas (DEVICTOR et al., 2010), its inclusion on conservation issues has been already widely recommended (e.g. SCHWARTZ et al. 2000; TRINDADE-FILHO et al. 2012b; ISBELL et al. 2011; CADOTTE 2011; TRINDADE-FILHO; LOYOLA 2011). Here, we show a possible way to incorporate plant functional diversity into conservation strategies in Cerrado areas. Further studies could help to improve our approach, test it in broader scales and adapt it for other vegetation types and organisms.

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Supplementary Material 1

Incorporating functional diversity as indicator of conservation in the Cerrado - criteria to choose trait values based on literature

In this chapter, we selected traits related to the ecosystem functioning in Cerrado, particularly Cerradão. Height and basal area are associated with growth form, position of the species in the vertical light, gradient of the vegetation, competitive vigour and reproductive size (PÉREZ-HARGUINDEGUY et al., 2013). Also, it is related with whether a species is able to establish and attain reproductive size between two disturbance events (PÉREZ-HARGUINDEGUY et al., 2013), such as fire, which is a key component in Cerrado functioning (COUTINHO, 1978). Leaf area is related to the water and energy balance within the leaves so that it is linked to ecological strategies in nutritional stress and disturbs situations (PÉREZ-HARGUINDEGUY et al., 2013).

In Cerrado, vegetative phenology determines the seasonality of the community and the spatial and temporal variability of habitats. During the dry season, woody species can lose their leaves partially or completely (Semi-deciduous and Deciduous species) or they can maintain their leaves during the whole year (Evergreen species). Thus, the seasonality in leaves production in Cerrado is not determined by the amount of water in the soil (BATALHA; MANTOVANI, 2000; SARMIENTO; MONASTERIO, 1983).

Reproductive phenology gives some information about the establishment of species, their growing and reproduction periods (flowering and fruit production). Thus these traits are linked to provision of resources for pollinators and dispersors (MORELLATO, 1991). Moreover, phenology reflects co-evolution relationships between animal and plant species - herbivorous, pollinators and dispersors - which is critical for ecosystem functioning (SARMIENTO, 1984). Propagule dispersal mode is directly related to the distance that it can achieve, so that it is related to the potential of colonisation and regeneration of the ecosystem (VAN DER PIJL, 1982).

The quantitative traits were widespread in the literature. We systematised the search following the steps below:

1. First of all we consider the traits available in Mantovani; Martins (1993) measured in the same study site as this paper.
2. For the species and trait that were not available in that paper, we seek for papers in Science Direct. using the following key-words: "Mogi-Guaçu", "Mogi-Mirim", "Cerrado", "Phenology". When we found two information for the same species, we choose the closest study

site.

3. For the ones that couldn't be found by now, we seek for papers using the keyword described in the step 2 plus the name of the species, attend to eventually synonyms or old names.

4. Finally, we consulted Lorenzi (1992), which presents averages for the whole country.

5. The ones that were not found by this step were considered as NAs.

Table S1 - Literature consulted to determine quantitative traits of species sampled

Species	Family	Reproductive phenology	Reference and place	Vegetative phenology	Reference and place	Dispersal Syndrome	Reference and place
<i>Amaioua guianensis</i> Aubl.	Rubiaceae	(LORENZI, 1992)	Brasil	(LORENZI, 1992)	Brasil	(LORENZI, 1992)	Brasil
<i>Anadenanthera peregrina</i> var. <i>falcata</i> (Benth.) Altschul	Fabaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Andira cujabensis</i> Benth.	Fabaceae	(LORENZI, 1992)	Brasil	(LORENZI, 1992)	Brasil	(LORENZI, 1992)	Brasil
<i>Austroplenkia polpunea</i>	Celastraceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Bauhinia rufa</i> (Bong.) Steud.	Fabaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(MORELLATO, 1991)	Campinas (SP)	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Brosimum gaudichaudii</i> Trécul	Moraceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Byrsonima coccolobifolia</i> Kunth	Malpighiaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Byrsonima verbascifolia</i> (L.) DC.	Malpighiaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Caryocar brasiliense</i> Cambess.	Caryocaraceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Casearia decandra</i>	Salicaceae	(MORELLATO,	Campinas	(LORENZI, 1992)	Brasil	(MANTOVANI;	Brasil

Species	Family	Reproductive phenology	Reference and place	Vegetative phenology	Reference and place	Dispersal Syndrome	Reference and place
Jacq.		1991)	(SP)			MARTINS, 1993)	
<i>Casearia sylvestris</i> Sw.	Salicaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Connarus suberosus</i> Planch.	Connaraceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Copaifera langsdorffii</i> Desf.	Fabaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Cordia macrophylla</i> (K.Schum.) Kuntze	Rubiaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Cordia sessilis</i> (Vell.) Kuntze	Rubiaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Coussarea hydrangeifolia</i> (Benth.) Müll.Arg.	Rubiaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Ficus citrifolia</i> Mill.	Moraceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(PEREIRA; RODRIGUES; MENEZES JR, 2007)	Londrina (PR)	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Handroanthus ochraceus</i> (Cham.) Mattos	Bignoniaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Hymenaea stigonocarpa</i> Mart. ex	Fabaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)

Species	Family	Reproductive phenology	Reference and place	Vegetative phenology	Reference and place	Dispersal Syndrome	Reference and place
Hayne							
<i>Ixora gardneriana</i> Benth.	Rubiaceae	(MORELLATO, 1991)	Campinas (SP)	(MORELLATO, 1991)	Campinas (SP)	(LORENZI, 1992)	Brasil
<i>Lacistema hasslerianum</i> Chodat	Lacistemataceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Lafoensia pacari</i> A.St.-Hil.	Lythraceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Leandra lacunosa</i> Cogn.	Melastomataceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Leptolobium elegans</i> Vogel	Fabaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Luehea divaricata</i> Mart.	Tiliaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(MORELLATO, 1991)	Campinas (SP)	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Mabea fistulifera</i> Mart.	Euphorbiaceae	(LORENZI, 1992)	Brasil	(LORENZI, 1992)	Brasil	(LORENZI, 1992)	Mogi Guaçu (SP)
<i>Machaerium acutifolium</i> Vogel	Fabaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Magonia pubensis</i>	Sapindaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Matayba elaeagnoides</i> Radlk.	Sapindaceae	(LORENZI, 1992)	Brasil	(LORENZI, 1992)	Brasil	(LORENZI, 1992)	Brasil
<i>Miconia albicans</i> (Sw.)	Melastomataceae	(MANTOVANI;	Mogi Guaçu	(PAULA, 2002)	São Carlos	(MANTOVANI;	Mogi Guaçu

Species	Family	Reproductive phenology	Reference and place	Vegetative phenology	Reference and place	Dispersal Syndrome	Reference and place
Triana	ae	MARTINS, 1993)	(SP)			MARTINS, 1993)	(SP)
<i>Miconia paucidens</i> DC.	Melastomataceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	-	-	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Miconia rubiginosa</i> (Bonpl.) DC.	Melastomataceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	-	-	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Myrcia bella</i> Cambess.	Myrtaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	-	-	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Myrcia guianensis</i> (Aubl.) DC.	Myrtaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	-	-	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Myrcia multiflora</i> (Lam.) DC.	Myrtaceae	(PIRANI; SANCHEZ; PEDRONI, 2009)	Barra do Garças (MT)	(PIRANI; SANCHEZ; PEDRONI, 2009)	Barra do Graças (MT)	(PIRANI; SANCHEZ; PEDRONI, 2009)	Barra do Garças (MT)
<i>Myrcia splendens</i> (Sw.) DC.	Myrtaceae	-	-	(LENZA; KLINK, 2006)	Brasilia (DF)	(LENZA; KLINK, 2006)	Brasilia (DF)
<i>Myrcia tomentosa</i> (Aubl.) DC.	Myrtaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Myrcia venulosa</i> DC.	Myrtaceae	(LORENZI, 1992)	Brasil	(LORENZI, 1992)	Brasil	(LORENZI, 1992)	Brasil
<i>Myrsine coriacea</i> (Sw.) R.Br. ex Roem. & Schult.	Primulaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Myrsine gardneriana</i> A.DC.	Primulaceae	-	-	-	-	-	-

Species	Family	Reproductive phenology	Reference and place	Vegetative phenology	Reference and place	Dispersal Syndrome	Reference and place
<i>Myrsine guianensis</i> (Aubl.) Kuntze	Primulaceae	(TANNUS; ASSIS; MORELLATO, 2006).	Itirapina (SP)	(LORENZI, 1992)	Brasil	(LORENZI, 1992)	Brasil
<i>Myrsine lancifolia</i> Mart.	Primulaceae	(LORENZI, 1992)	Brasil	(LORENZI, 1992)	Brasil	(FRANCISCO; GALETTI, 2001)	São Carlos (SP)
<i>Myrsine umbellata</i> Mart.	Primulaceae	(LORENZI, 1992)	Brasil	(LORENZI, 1992)	Brasil	(LORENZI, 1992)	Brasil
<i>Ocotea minarum</i> (Nees & Mart.) Mez	Lauraceae	(MORELLATO, 1991)	Campinas (SP)	(LORENZI, 1992)	Brasil	(LORENZI, 1992)	Brasil
<i>Ocotea pulchella</i> (Nees & Mart.) Mez	Lauraceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Ormosia arborea</i> (Vell.) Harms	Fabaceae	(LORENZI, 1992)	Brasil	(LORENZI, 1992)	Brasil	(LORENZI, 1992)	
<i>Ouratea spectabilis</i> (Mart.) Engl.	Ochnaceae	(MANTOVANI; MARTINS, 1993)		(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Pera glabrata</i> (Schott) Poepp. ex Baill.	Peraceae	(MANTOVANI; MARTINS, 1993)	Campinas (SP)	(MORELLATO, 1991)	Campinas (SP)	(FREITAS et al., 2011)	São Carlos (SP)
<i>Platypodium elegans</i> Vogel	Fabaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Pouteria ramiflora</i> (Mart.) Radlk.	Sapotaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Protium heptaphyllum</i>	Burseraceae	(MANTOVANI;	Mogi Guaçu	(LORENZI, 1992)	Brasil	(MANTOVANI;	Mogi Guaçu

Species	Family	Reproductive phenology	Reference and place	Vegetative phenology	Reference and place	Dispersal Syndrome	Reference and place
(Aubl.) Marchand		MARTINS, 1993)	(SP)			MARTINS, 1993)	(SP)
<i>Qualea grandiflora</i> Mart.	Vochysiaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Qualea multiflora</i> Mart.	Vochysiaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Qualea parviflora</i> Mart.	Vochysiaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Roupala montana</i> Aubl.	Proteaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LENZA; KLINK, 2006)	Brasilia (DF)	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Rudgea viburnoides</i> (Cham.) Benth	Rubiaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Schefflera vinosa</i> (Cham. & Schltdl.) Frodin & Fiaschi	Araliaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Siparuna guianensis</i> Aubl.	Siparunaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(MORELLATO, 1991)	Campinas (SP)	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Styrax camporum</i> Pohl	Styracaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Syagrus flexuosa</i> (Mart.) Becc.	Arecaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	-	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Tapirira guianensis</i> Aubl.	Anacardiaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)

Species	Family	Reproductive phenology	Reference and place	Vegetative phenology	Reference and place	Dispersal Syndrome	Reference and place
<i>Virola sebifera</i> Aubl.	Myristicaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Vitex polygama</i> Cham.	Lamiaceae	(MANTOVANI; MARTINS, 1993)	Brasil	(LORENZI, 1992)	Brasil	(LORENZI, 1992)	Brasil
<i>Vochysia tucanorum</i> Mart.	Vochysiaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)
<i>Xylopia aromatica</i> (Lam.) Mart.	Annonaceae	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)	(LORENZI, 1992)	Brasil	(MANTOVANI; MARTINS, 1993)	Mogi Guaçu (SP)

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Supplementary Material 2

Incorporating functional diversity as indicator of conservation in the Cerrado - details on ranking plots method

Percentiles are measures that divide a sample in 100 parts, following a crescent order. For instance, the first percentile (P1) determines the 1% lowest values in the sample, and the 98° determines the 98% lowest values in the sample (MAGALHÃES; LIMA, 2000). We classified all the plots in H (high biodiversity), M (medium biodiversity) and L (low biodiversity) for each index considered by using percentiles values for each index. Plots that had values lower than p33 for a given index were lable as L to that index. Plots that had values higher than p66 for a given index were lable as H to that index. And plots that had values between p33 and p66 were labled as M.

Table S2 - Diversity indeces values for each plot and their labels according to percentiles values - H (high), M (medium) e L (low).

Plot	FEve	Label for FEve	FDiv	Lable for FDiv	FDis	Lable for FDis	Rich	Lable for Rich	H'	Lable H'
1	0.57395120	L	0.69118411	L	0.19941128	L	6	L	0.83	L
2	0.77321339	H	0.88390895	H	0.47021649	H	17	H	2.46	H
3	0.77763352	H	0.84381480	H	0.45259662	M	13	M	2.36	H
4	0.77650662	H	0.81190031	M	0.45743303	H	18	H	2.65	H
5	0.69315378	L	0.84428291	H	0.46049947	H	12	M	2.10	M
6	0.76440483	H	0.88111125	H	0.49127618	H	12	M	2.19	M
7	0.73469405	L	0.83970216	H	0.42397851	L	14	H	2.13	M
8	0.75383158	M	0.83166726	M	0.42860870	M	17	H	2.33	H
9	0.64964953	L	0.83018192	M	0.42637757	M	18	H	2.44	H
10	0.75011159	M	0.77950249	L	0.44445835	M	18	H	2.48	H
11	0.75759310	M	0.79206445	M	0.45445980	H	17	H	2.59	H
12	0.52870204	L	0.82802644	M	0.37121397	L	5	L	1.20	L
13	0.70769398	L	0.77934362	L	0.38146100	L	11	M	2.10	M
14	0.73812656	M	0.76965789	L	0.43661417	M	8	L	1.87	L
15	0.76347229	H	0.79025901	L	0.45926804	H	13	M	2.20	M
16	0.75184337	M	0.89729051	H	0.40500220	L	10	L	1.88	L
17	0.82152815	H	0.70884463	L	0.44781594	M	10	L	2.15	M
18	0.83267881	H	0.85892800	H	0.46009421	H	8	L	1.86	L
19	0.65844181	L	0.78528418	L	0.41618872	L	10	L	1.92	L
20	0.76280816	H	0.82541080	M	0.36454896	L	13	M	1.72	L

Plot	FEve	Label for FEve	FDiv	Lable for FDiv	FDis	Lable for FDis	Rich	Lable for Rich	H'	Lable H'
P33	0.7356208		0.7907465		0.4246263		10.27		1.9686	
P66	0.7631668		0.8360061		0.4536027		13.54		2.2702	

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Supplementary Material 3

Incorporating functional diversity as indicator of conservation in the Cerrado - details on species sampled

In Moji-Guaçu Biological Reserve (22°15'S e 47°08'W) we sampled 483 individuals belonging to 45 species (26 families) within which 31 occurred exclusively in that area. Most abundant species were *Siparuna guianensis* Aubl. (Siparunaceae) - 115 individuals; *Xylopia aromatica* (Lam.) Mart (Annonaceae) - 59 individuals and *Ocotea pulchella* (Nees & Mart.) Mez (Lauraceae)- 41 individuals. 11 species were represented by only one individual. The richest families were Fabaceae (5 species), Melastomataceae, Rubiaceae and Vochysiaceae (4 species each). 17 families were represented by only one species.

In Mogi Mirim Experimental Station (22°26'S e 46°57'W) we sampled 315 individuals belonging to 39 species (12 families) within which 19 occurred exclusively in that area. Most abundant species were *Amaioua guianensis* Aubl. (Rubiaceae) and *Siparuna guianensis* Aubl. (Siparunaceae), (50 individuals each) and *Vitex polygama* Cham. (Lamiaceae) - 20 individuals. 13 species were represented by only one individual. The richest families were Fabaceae and Myrtaceae (4 species each) and Rubiaceae (3 species). 17 families were represented by only one species.

Table S3 - Number of individuals from each species sampled in total and in each area.

Family	Species	Total	Mogi guaçu	Mogi Mirim
Anacardiaceae	<i>Tapirira guianensis</i> Aubl.	6	6	0
Annonaceae	<i>Xylopia aromatica</i> (Lam.) Mart.	63	59	4
	<i>Schefflera vinosa</i> (Cham. & Schltdl.)			
Araliaceae	Frodin & Fiaschi	3	3	0
Arecaceae	<i>Syagrus flexuosa</i> (Mart.) Becc.	3	2	1
	<i>Handroanthus ochraceus</i> (Cham.)			
Bignoniaceae	Mattos	4	1	3
	<i>Protium heptaphyllum</i> (Aubl.)			
Burseraceae	Marchand	2	2	0
Caryocaraceae	<i>Caryocar brasiliense</i> Cambess.	2	0	2
Celastraceae	<i>Austroplenkia polpunea</i>	1	1	0
Connaraceae	<i>Connarus suberosus</i> Planch.	3	3	0
Euphorbiaceae	<i>Mabea fistulifera</i> Mart.	1	0	1
	<i>Anadenanthera peregrina</i> var. <i>falcata</i> (Benth.) Altschul			
Fabaceae		1	0	1
Fabaceae	<i>Andira cujabensis</i> Benth.	1	0	1
Fabaceae	<i>Bauhinia rufa</i> (Bong.) Steud.	2	2	0

Family	Species	Total	Mogi guaçu	Mogi Mirim
Fabaceae	<i>Copaifera langsdorffii</i> Desf.	17	2	15
Fabaceae	<i>Hymenaea stigonocarpa</i> Mart. ex Hayne	1	0	1
Fabaceae	<i>Leptolobium elegans</i> Vogel	2	2	0
Fabaceae	<i>Machaerium acutifolium</i> Vogel	1	0	1
Fabaceae	<i>Ormosia arborea</i> (Vell.) Harms	1	1	0
Fabaceae	<i>Platypodium elegans</i> Vogel	2	2	0
Lacistemataceae	<i>Lacistema hasslerianum</i> Chodat	23	13	10
Lamiaceae	<i>Vitex polygama</i> Cham.	20	0	20
Lauraceae	<i>Ocotea minarum</i> (Nees & Mart.) Mez	1	0	1
Lauraceae	<i>Ocotea pulchella</i> (Nees & Mart.) Mez	118	41	77
Lythraceae	<i>Lafoensia pacari</i> A.St.-Hil.	1	0	1
Malpighiaceae	<i>Byrsonima coccolobifolia</i> Kunth	3	3	0
Malpighiaceae	<i>Byrsonima verbascifolia</i> (L.) DC.	1	1	0
Melastomataceae	<i>Leandra lacunosa</i> Cogn.	3	3	0
Melastomataceae	<i>Miconia albicans</i> (Sw.) Triana	13	13	0
Melastomataceae	<i>Miconia paucidens</i> DC.	8	7	1
Melastomataceae	<i>Miconia rubiginosa</i> (Bonpl.) DC.	6	6	0
Moraceae	<i>Brosimum gaudichaudii</i> Trécul	1	1	0
Moraceae	<i>Ficus citrifolia</i> Mill.	1	1	0
Myristicaceae	<i>Virola sebifera</i> Aubl.	26	24	2
Myrtaceae	<i>Myrcia bella</i> Cambess.	1	1	0
Myrtaceae	<i>Myrcia guianensis</i> (Aubl.) DC.	3	2	1
Myrtaceae	<i>Myrcia multiflora</i> (Lam.) DC.	10	0	10
Myrtaceae	<i>Myrcia splendens</i> (Sw.) DC.	2	0	2
Myrtaceae	<i>Myrcia tomentosa</i> (Aubl.) DC.	2	2	0
Myrtaceae	<i>Myrcia venulosa</i> DC.	1	0	1
Ochnaceae	<i>Ouratea spectabilis</i> (Mart.) Engl.	6	1	5
Peraceae	<i>Pera glabrata</i> (Schott) Poepp. ex Baill.	2	2	0
Primulaceae	<i>Myrsine coriacea</i> (Sw.) R.Br. ex Roem. & Schult.	2	2	0
Primulaceae	<i>Myrsine gardneriana</i> A.DC.	1	0	1
Primulaceae	<i>Myrsine guianensis</i> (Aubl.) Kuntze	4	4	0
Primulaceae	<i>Myrsine lancifolia</i> Mart.	2	0	2
Primulaceae	<i>Myrsine umbellata</i> Mart.	25	21	4
Proteaceae	<i>Roupala montana</i> Aubl.	2	0	2
Rubiaceae	<i>Amaioua guianensis</i> Aubl.	50	0	50

Family	Species	Total	Mogi guaçu	Mogi Mirim
Rubiaceae	<i>Cordia macrophylla</i> (K.Schum.) Kuntze	1	1	0
Rubiaceae	<i>Cordia sessilis</i> (Vell.) Kuntze	9	3	6
Rubiaceae	<i>Ixora gardneriana</i> Benth.	4	0	4
Rubiaceae	<i>Rudgea viburnoides</i> (Cham.) Benth	48	48	0
Rubiaceae	<i>Coussarea hydrangeifolia</i> (Benth.) Müll.Arg.	20	14	6
Salicaceae	<i>Casearia decandra</i> Jacq.	1	0	1
Salicaceae	<i>Casearia sylvestris</i> Sw.	3	1	2
Sapindaceae	<i>Magonia pubescens</i>	2	2	0
Sapindaceae	<i>Matayba elaeagnoides</i> Radlk.	1	1	0
Sapotaceae	<i>Pouteria ramiflora</i> (Mart.) Radlk.	12	0	12
Siparunaceae	<i>Siparuna guianensis</i> Aubl.	165	115	50
Styracaceae	<i>Styrax camporum</i> Pohl	2	2	0
Tiliaceae	<i>Luehea divaricata</i> Mart.	1	0	1
Vochysiaceae	<i>Qualea grandiflora</i> Mart.	6	2	4
Vochysiaceae	<i>Qualea multiflora</i> Mart.	9	9	0
Vochysiaceae	<i>Qualea parviflora</i> Mart.	3	3	0
Vochysiaceae	<i>Vochysia tucanorum</i> Mart.	57	48	9
Total		798	483	315

CONSIDERAÇÕES FINAIS

Os objetivos gerais desta tese foram avaliar os impactos dos sistemas antrópicos no funcionamento dos ecossistemas e na provisão dos serviços ambientais e ecossistêmicos e propor a abordagem funcional como indicadora da biodiversidade. Os resultados revelam que o funcionamento de ecossistemas altamente biodiversos e endêmicos, com componentes pouco conhecidos podem estar expostos à riscos cujas consequências não podem ainda ser precisamente mensuradas. Em escala global, o Brasil, um país megadiverso, longe de conhecer completamente sua biota (LEWINSOHN; PRADO, 2005), já apresenta a maior parte dos seus tipos vegetacionais expostos à luz artificial. Em escala regional, o cerrado, um domínio fitogeográfico com alto grau de endemismo, já reduziu significativamente a disponibilidade dos serviços ambientais e ecossistêmicos medidos. As recomendações de inclusão da diversidade funcional em estratégias de conservação parecem estar embasadas em padrões enviesados devido à poucos estudos realizados em áreas tropicais. Contribuímos para diminuir esta lacuna apresentando um conjunto de espécies indicadoras da diversidade funcional em áreas cobertas por Cerrado no estado de São Paulo. A seguir, apresentamos uma síntese das principais conclusões de cada capítulo bem como as implicações e recomendações para a conservação. Apresentamos também as limitações de cada estudo e sugestões para pesquisas futuras.

No capítulo 1 mostramos que no Brasil ainda existem tipos vegetacionais que usufruem do brilho natural do céu e que a extensão de áreas afetadas por luz artificial tende a crescer caso os padrões atuais sejam mantidos. Estes resultados apontam para a necessidade de repensar os modelos de instalação e distribuição da energia elétrica, em especial, da iluminação pública. Atualmente, o número de pessoas sem acesso à energia elétrica no mundo é estimado em aproximadamente 1.3 bilhões (IEA, 2015) e é inegável que essa privação traz malefícios à qualidade de vida dessas pessoas (DOLL; MULLER; MORLEY, 2006; RIBEIRO, 2015). A ampliação do acesso à energia elétrica tem sido alvo de esforço nacional (i.e. Programa Luz para Todos - BRASIL, 2003) e internacional (ONU, 2013;). Este trabalho não se propõe a defender um modelo de desenvolvimento excludente, no qual o acesso à energia elétrica deva ser negado ou desigual. Apenas ressaltamos que assim como os ciclos biogeoquímicos, os ritmos circadianos são necessários ao bom funcionamento dos ecossistemas e da saúde humana (BARGHINI, 2010; GASTON et al., 2014) e por isso o fornecimento e o uso da luz artificial devem se dar de forma adequada, econômica, igualitária e sustentável (ONU, 2013).

No Brasil, as poucas iniciativas de controle e planejamento da iluminação pública

consistem em regulamentações pontuais, sem repercussão no planejamento energético do país como um todo. Por exemplos a portaria IBAMA de 1995, que protege toda a costa brasileira onde há desova de tartarugas e a lei em vigor no município de Campinas-SP que protege o entorno do observatório astronômico (IBAMA, 1995; PREFEITURA MUNICIPAL DE CAMPINAS, 2001). Fora do Brasil, já existem leis mais severas e menos pontuais, englobando o planejamento da distribuição da iluminação. Na República Tcheca por exemplo, iluminação em excesso é definida legalmente como uma forma de poluição e é proibida (HOLLAN, 2003). Na Califórnia, a legislação impõe limites de potência instalada para a iluminação pública e particular dependendo do tamanho da comunidade e do tipo de ocupação do espaço (CALIFORNIA ENERGY COMMISSION, 2013). Além disso, diferentemente do que ocorre no Brasil, a preocupação com os potenciais danos ambientais causados pela iluminação artificial pode surgir das próprias empresas concessionárias de distribuição de energia elétrica. Na Flórida por exemplo, a Flórida Power and Lightning Co. financiou estudos sobre o efeito da iluminação artificial sobre as tartarugas marinhas e a elaboração de manuais sobre a utilização da iluminação de forma a minimizar os impactos sobre as desovas destes animais (FLORIDA POWER & LIGHTNING COMPANY, 1998; SALMON; WYNEKEN; FOOTE, 2002).

As potenciais consequências da poluição luminosa nos tipos de vegetação que levantamos nesse trabalho devem ser interpretadas com cautela porque as relações entre a luz artificial observada a partir do satélite e os impactos no funcionamento dos ecossistemas são complexas. Quando emissões de luz ocorrem no sentido horizontal, as ondas podem se espalhar pela atmosfera e o *skyglow* pode afetar áreas localmente não iluminadas. Além disso, sombras verticais podem criar áreas discretas de escuridão não identificadas (FALCHI et al., 2011). Ainda que houvesse perfeita correspondência espacial entre as áreas iluminadas detectadas no satélite e os pontos de iluminação no chão, a intensidade de radiação média na escala estudada deve ser percebida de forma diferente por diferentes organismos. Apesar destas limitações, as imagens baseadas nas emissões de luz para cima são o melhor material disponível para avaliar os padrões espaciais e temporais da luz artificial em escalas globais e regionais (DOLL; MULLER; MORLEY, 2006).

A discussão apresentada no capítulo 1 contribui para a busca do equilíbrio entre os benefícios e os malefícios da exposição a luz artificial na medida em que levanta os potenciais riscos aos quais o funcionamento dos ecossistemas brasileiros estão sujeitos. A compreensão da questão, porém, vai além de estudos ecológicos mais aprimorados. As lacunas de conhecimento em relação aos riscos da exposição à luz artificial são variadas. A segurança, por exemplo,

parece ser o argumento mais obvio em defesa da necessidade da iluminação pública. De fato, Farrington and Welsh (2002) relatam correlação positiva entre redução da criminalidade e aumento da iluminação pública em alguns países. Entretanto, a redução não difere entre os períodos diurno e noturno, o que sugere que tal redução se deve mais à métodos informais de controle social devido ao aumento do orgulho e da sensação de pertencimento da comunidade do que à maior visibilidade das infrações. Além disso, é necessário que o foco da poluição luminosa seja ampliado para além da eficiência energética (IEA, 2012). As lâmpadas LED (do inglês, *light emitting diodes* - diodos emissores de luz), por exemplo, tem sido indicada como um caminho para a sustentabilidade devido à sua eficiência energética e consequentemente à economia de energia (TSAO 2002). No entanto, pesquisa recente mostra que as ondas emitidas por essas lâmpadas alteram a produção e secreção da melatonina, um hormônio produzido pela glândula pineal e responsável pela regulação do sono (FALCHI et al., 2011). Nesse sentido, a compreensão do questão depende de pesquisa interdisciplinar, que considere a representação simbólica da luz artificial para as diferentes sociedades humanas atuais, o bem-estar humano, o funcionamento dos ecossistemas as consequências socioeconômicas dos impactos positivos e negativos da luz artificial (HÖLKER et al., 2010).

Em relação ao campo da ecologia, pesquisas futuras devem identificar eventuais alterações no comportamento das espécies em função da presença de luz artificial e prever as consequências de tais alterações para o funcionamento dos ecossistemas. Tais pesquisas devem considerar as características e a natureza da luz, tais como comprimento de onda, intensidade e direcionalidade (FALCHI et al., 2011). Além disso, estudos que definam os padrões espaciais e temporais da distribuição da luz artificial em outros países em desenvolvimento e altamente biodiversos permitirão a comparação com nossos resultados, a identificação de padrões e de diferenças e contribuirá para o aumento do conhecimento do assunto. Em relação ao Brasil, enfatizamos a urgência de estudos e de proteção preventiva dos tipos vegetacionais localizados próximos à costa (restingas, manguezais, Florestas Estacionais e a própria massa de água costeira), que são os mais afetados.

Dentre as ações práticas preventivas, recomenda-se o controle do fluxo luminoso e da intensidade luminosa, restringindo o uso da iluminação pública apenas em locais e horários onde ela é realmente necessária, com intensidade luminosa mais baixa possível e com direcionamento do feixe luminoso voltado para o objeto ou a área que precisa ver visualizada, evitando dispersão dos raios. Deve haver também, modulação do espectro da irradiância, dando preferência para comprimentos de onda mais próximos das ondas naturais, por meio do uso de filtros seletivos (BARGHINI, 2010; GASTON et al., 2012). Em se tratando de áreas naturais

protegidas, em particular as de proteção integral, recomenda-se o estabelecimento de zona de amortecimento onde a iluminação artificial deve ser proibida, ou, pelo menos reduzida significativamente (GASTON; DUFFY; BENNIE, 2015).

No capítulo 2 mostramos que os serviços do ecossistema vêm decaindo significativamente ao longo do tempo no Cerrado brasileiro, o que ressalta a importância da área em termos de conservação. Dada a relevância das exportações agrícolas no cenário nacional atualmente, e a importância da polinização e da manutenção do ciclo do carbono para a agricultura, nossos resultados reforçam a necessidade de repensar o modelo predatório de exploração do Cerrado, sob risco de prejuízo para a própria agricultura a longo prazo. Os resultados também apontam o MATOPIBA como um importante remanescente de provisão de serviços do ecossistema e sugerem a necessidade de criação de unidades de conservação nessa área, em particular de proteção integral. Embora tal necessidade já tenha sido apontada por outros autores - não só no MATOPIBA como também em toda a área do cerrado – nenhum deles baseou-se em serviços ecossistêmicos, como apresentado neste trabalho.

Os valores globais padronizados para medir polinização e estoque de carbono neste trabalho (AALDE et al., 2006; KENNEDY et al., 2013) é menos eficaz em refletir a realidade local do que se houvessem valores específicos para a região. Por exemplo, os valores de adequabilidade para nidificação e recursos florais de abelhas nativas - utilizados como forma de medir polinização – desconsideram o papel de borboletas, mariposas, aves e morcegos, que desempenham papel importante na região (GIANNINI et al., 2015; RICKETTS et al., 2008). Nesse sentido, o desenvolvimento de medidas específicas para o cerrado poderia aperfeiçoar o mapeamento apresentamos aqui.

A literatura científica do campo ambiental frequentemente assume que o valor das terras indígenas está na sua eficiência em conservar a biodiversidade e evitar o desmatamento (CARRANZA et al., 2014). A abordagem utilizada neste trabalho parte do princípio do valor que essas terras têm para os povos que dependem diretamente delas (embora, tenhamos concluído que elas são importantes para a provisão dos serviços ecossistêmicos medidos, dadas as sobreposições encontradas). Pesquisas futuras poderiam estender o mapeamento apresentado aqui a outros serviços ecossistêmicos presentes no cerrado (como a recarga de aquíferos, por exemplo - DURIGAN, 2010) e incluir o espaço para viver de outras comunidades tradicionais presentes na região (quilombolas, quebradeiras de coco, geraizeiros – MMA, 2003). Para tanto, é necessário o mapeamento e georreferenciamento do território de tais comunidades.

No capítulo 3 observamos a raridade de artigos que incorporem a diversidade funcional na conservação de ecossistemas tropicais e identificamos o potencial desta abordagem para

elaboração de indicadores da integridade ecológica - o que levou à elaboração do capítulo 4. O termo diversidade funcional é relativamente recente na literatura científica e sua introdução no campo da conservação mais recente ainda. Nossos resultados permitem visualizar o paradigma da transferência de conhecimento. A utilização mais ampla dos conceitos e das medidas de diversidade funcional em estratégias de conservação dependem de mais estudos sobre os padrões espaciais da diversidade funcional e sobre as relações dos traços funcionais com os serviços ecossistêmicos.

No capítulo 4, apresentamos um conjunto de espécies de plantas indicadoras de mais de um aspecto da biodiversidade no cerrado em escala local. Estes indicadores devem ser usados em conjunto para monitoramento e diagnóstico da biodiversidade quando o objetivo é manter o funcionamento do ecossistema. Outros indicadores com propósitos diferentes devem ser somados a esse na implementação das estratégias de conservação. Como todas as espécies indicadoras que encontramos são consideradas comuns, o trabalho também contribui para o debate, ainda longe de um consenso, sobre a importância relativa de espécies raras e comuns quando se trata de funcionamento do ecossistema e conservação (GASTON, 2011; MOUILLOT et al., 2013).

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