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IMPACTO DA EXPANSÃO AGRÍCOLA SOBRE A PRODUTIVIDADE PRIMÁRIA DA FLORESTA ATLÂNTICA NORDESTINA

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Dissertação apresentada ao Programa de Pós-Graduação em Biologia Vegetal, Departamento de Botânica, da Universidade Federal de Pernambuco, como parte dos requisitos à obtenção do grau de mestre em Biologia Vegetal.

Orientador: Profº Dr. Bráulio Almeida Santos

Coorientadora: Profº Dra. Maria Grisel Longo

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RESUMO

A expansão da agricultura de larga escala tem mudado a estrutura da paisagem de florestas tropicais e colocado em risco a provisão dos serviços ecossistêmicos. No entanto, pouco se sabe como essas mudanças podem afetar serviços intermediários como a Produtividade Primária (PP), sobretudo em antigas fronteiras agrícolas. Em uma região produtora de cana de açúcar na Floresta Atlântica nordestina, investigamos o efeito da composição e configuração da paisagem sobre a PP da floresta remanescente e cultivos agrícolas, bem como da paisagem como um todo e sua variabilidade espacial. Através de imagens de satélite dos anos de 2011 a 2016, amostramos 120 paisagens de 1 km² durante as estações chuvosa e seca. Em cada paisagem, mensuramos a porcentagem de cultivos (medidas de composição), a densidade de borda e o número de manchas florestais (medidas de configuração). Como *proxies* da PP, estimamos o Índice de Vegetação por Diferença Normalizada (NDVI) e o Índice de Vegetação Melhorado (EVI) da floresta, das áreas de cultivos e da paisagem como um todo. O NDVI e EVI da paisagem foram influenciados negativamente pela porcentagem de cultivos em ambas as estações, mas foram pouco influenciados pela densidade de borda e número de manchas florestais. Separadamente, o NDVI da floresta e dos cultivos, foi negativamente influenciado pela porcentagem de cultivos da paisagem, sugerindo sinergia entre produção e conservação florestal. A variabilidade espacial do NDVI e EVI não foi influenciada pela estrutura da paisagem em nenhuma das estações. Em antigas fronteiras agrícolas fragmentadas, para a PP a composição da paisagem é mais importante do que a configuração, deixando clara a importância de conservar todos os fragmentos da floresta. A sinergia entre a produção agrícola e a conservação florestal é um caminho indispensável para compensar os passivos ambientais da extensiva e antiga produção de cana-de-açúcar em florestas tropicais.

Palavras-chave: Serviços ecossistêmicos intermediários. Estrutura da paisagem. Perda de habitat. Fragmentação per se. Densidade de borda. Paisagens tropicais modificadas humanas.

ABSTRACT

The expansion of large-scale agriculture has changed the landscape structure of tropical forests and threatened the provision of ecosystem services. However, little is known about how changes can affect intermediate services such as Primary Productivity (PP), especially in advanced agricultural frontiers. In a sugar cane producing region in the Northeast Atlantic Forest, we investigated the effect of composition and landscape configuration on the PP of the remaining forest and agricultural crops, as well as the landscape as a whole and its spatial variability. Through satellite images from the years 2011 to 2016, we sampled 120 landscape plots of 1 km² during both rainy and dry seasons. In each landscape, we measured the percentage of crops, the density of forest edges and number of forest patches. As a proxy for PP, we estimated the Normalized Difference Vegetation Index (NDVI) and the Enhanced Vegetation Index (EVI) of the forest, crop areas and landscape as a whole. The NDVI and EVI of the landscape were negatively influenced by the percentage of crops in both seasons, but are influenced little by the density of forest edges and number of forest patches. Separately, the NDVI of forest and crops was negatively influenced by the percentage of crops in the landscape, suggesting synergy among production and forest conservation. A spatial variability of NDVI and EVI was not influenced by landscape attributes. In old, heavily deforested agricultural frontiers, landscape composition is more important than configuration in determining the PP, highlighting the need to conserve forests in agricultural landscapes. The synergy between agricultural production and forest conservation provides a pathway for offsetting the environmental liabilities of long-lasting, large-scale sugarcane production.

Key-words: Intermediate ecosystem services. Structure landscape. Loss habitat. Fragmentation per se. Edge density. Human-modified tropical landscapes.

SUMÁRIO

1 INTRODUÇÃO

A expansão agrícola tem mudado a estrutura de paisagens tropicais causando perda e fragmentação de habitat, e ameaçado a prestação de serviços ecossistêmicos como como a regulação climática e o sequestro de carbono. Essas mudanças, em geral, provocam amplas alterações na paisagem em termos de composição e configuração espacial, onde funções ecossistêmicas que contribuem para o atendimento das demandas humanas podem ser afetadas. Enquanto a composição se refere à proporção dos diferentes tipos de uso da terra inseridos na paisagem (e.g. quantidade de habitat), a configuração se refere ao arranjo espacial em que esses diferentes tipos de uso estão organizados, como a densidade de bordas florestais e o número de fragmentos. A Produtividade Primária (PP), um serviço ecossistêmico intermediário responsável pela provisão de vários serviços finais indispensáveis para a vida humana, está entre os serviços ecossistêmicos menos estudados, sobretudo em antigas fronteiras agrícolas.

A Floresta Atlântica teve mais de 80% da sua cobertura original substituída por cidades e estradas, e principalmente por cultivos agrícolas. Desde o século XVI, a agricultura de larga escala tem causado perda e fragmentação de habitat, e ignorado seus efeitos sobre os serviços ecossistêmicos. Isso nos remete a necessidade de avaliar até que ponto a segurança alimentar e energética geradas pelos cultivos compensam as perdas no funcionamento e capacidade de prestação dos serviços dos ecossistemas. Além disso, avaliar como a PP de ambientes florestais responde à reestruturação da paisagem pode nos ajudar a identificar possíveis sinergias e demandas conflitantes entre a produção e conservação florestal.

Desta forma, essa dissertação é composta, além da fundamentação teórica, por um manuscrito no qual abordamos os impactos da estrutura da paisagem sobre a PP ao longo de um gradiente de transformação agrícola, na Floresta Atlântica Nordestina. Mais precisamente, analisamos o efeito da composição e configuração da paisagem sobre a PP, em nível de paisagem, e nos ambientes de floresta e cultivos de cana de açúcar, bem como sua variabilidade espacial ao longo das paisagens. Entender os padrões da transformação da paisagem em antigas fronteiras agrícolas como a da Floresta Atlântica Nordestina pode nos ajudar a propor estratégias mais eficazes que conciliem o uso sustentável dos recursos ambientais, além de entender os caminhos futuros de fronteiras mais recentes como a Amazônia.

2 FUNDAMENTAÇÃO TEÓRICA

2.1 PRODUTIVIDADE PRIMÁRIA

Todos os ecossistemas são formados por organismos que necessitam de matéria e energia para a construção de suas atividades. A importância dos fluxos de energia e de matéria significa que todo e qualquer processo biótico, em ecossistemas terrestres e aquáticos, estão relacionados a fatores abióticos (DEANGELIS, 1980). Em ecossistemas florestais, a Produtividade Primária (PP) é a força que determina os fluxos de energia através da taxa de energia solar estocada por plantas e sua disponibilização para o resto da cadeia alimentar (ROY; SAUGIER; MOONEY, 2001), o que ocorre, essencialmente, através de processos assimilativos no compartimento biológico do ecossistema (CHAPIN; MATSON; VITOUSEK, 2011a; CLARK et al., 2001a). Em florestas tropicais, a produtividade primária é a maior responsável pela ciclagem do carbono, sendo que os outros processos assimilativos como dissolução e deposição em ambientes aquáticos, muito menos expressivos (DEL GROSSO et al., 2008a; GRACE, 2004). Estima-se que a PP de florestas tropicais represente aproximadamente 50% do carbono absorvido pela copa florestal (PAN et al., 2011). Além disso, corresponde como base energética para vários serviços ecossistêmicos como a regulação climática e o sequestro de carbono (COSTANZA et al., 1997; HABERL et al., 2004).

Há que se considerar, que a PP pode ser dada como o somatório das entradas e perdas de matéria orgânica de diferentes compartimentos (e.g. folhas, troncos e raízes). Acima do solo, consiste na matéria orgânica produzida e perdida em um dado intervalo de tempo como a serapilheira e as emissões de compostos voláteis e lixiviáveis (CHAPIN et al. 2002). Abaixo do solo, corresponde a medida do incremento e da perda da biomassa de raízes grossas e finas, como compostos orgânicos voláteis, materiais exsudatos e os estoques de novos carboidratos incorporados pela simbiose entre rizóbios e micorrizas (CLARK et al., 2001a; MALHI; DOUGHTY; GALBRAITH, 2011). Um ponto importante é que parte da PP sintetizada ao longo de um determinado período pode também ser perdida através da herbivoria e da decomposição da matéria orgânica do solo (CHAPIN et al. 2011).

Ao longo de várias décadas, diferentes abordagens foram desenvolvidas para estimar a PP dos ecossistemas (CHAPIN; MATSON; MOONEY, 2002; CLARK et al., 2001b). Ora baseando-se em modelos estatísticos, os quais relacionam níveis de assimilação de carbono e a disponibilidade de nutrientes, ora em modelos que combinam a detecção remota de dados e processos ecológicos como estimativas da PP tais como a radiação solar absorvida e o índice de área foliar (PENG et al., 2016; RUIMY; SAUGIER, 1994). Essas medidas podem ser relevantes para representar os padrões da PP dos ecossistemas, mas como em qualquer técnica, podem ser imprecisas ou limitadas quando avaliam as entradas e saídas de matéria orgânica, seja em modelos locais ou globais (FIELD; RANDERSON; MALMSTRÖM, 1995). Um exemplo claro é a incapacidade de mensurar a totalidade da PP de um ecossistema, uma vez que uma parte da matéria orgânica é perdida através de processos como a herbivoria e decomposição de matéria, embora existam métodos conhecidos para a correção desses efeitos (CLARK et al., 2001b; KEELING; PHILLIPS, 2007).

Em sistemas florestais, a avaliação da PP através de mensurações periódicas da biomassa vegetal pode ser útil tanto para medições acima, como abaixo do solo (QUINTO-MOSQUERA; MORENO, 2017). No entanto, em avaliações da biomassa acima do solo, as medições priorizam basicamente o incremento de biomassa em árvores em função do diâmetro (CHAPIN; MATSON; MOONEY, 2002). Outros grupos de menor porte, como arbustos e musgos, não são mensurados, de forma que, além de subestimar a PP local, essa abordagem desconsidera a contribuição destes grupos na produtividade do ecossistema (BONA et al., 2016; CHAPIN; MATSON; VITOUSEK, 2011a). Em estimativas que utilizam o conteúdo de serapilheira como um *proxy* da produtividade, um dos problemas mais notáveis são os fatores envolvidos na decomposição da matéria orgânica que não são considerados (GIRARDIN et al., 2014).

Já em mensurações da biomassa abaixo do solo, um dos maiores desafios consiste na quantificação de processos que incluem a produção de raízes e materiais exsudatos, somando-se as perdas de matéria através da volatilização de compostos orgânicos, lixiviação e consumo de biomassa (SALA et al., 1988). Um viés que esse tipo de abordagem pode trazer é que no caso das raízes, a maioria dos trabalhos considera apenas a mensuração em camadas mais superficiais do solo, mesmo reconhecendo seu papel em camadas mais profundas (CANADELL et al., 1996). Estudos que avaliam a PP abaixo do solo são relativamente escassos, quando comparados com aqueles realizados acima do solo. O que pode estar relacionado a esse baixo número de estudos podem ser os obstáculos tecnológicos e econômicos, assim como a limitação de ferramentas para a mensuração das raízes (VOGT et al., 1996). De qualquer forma, ambas as abordagens, acima e abaixo do solo, tendem a estimar apenas o incremento de biomassa, mas descartam as perdas de matéria relacionadas aos processos acima e abaixo do solo (CHAPIN; MATSON; VITOUSEK, 2011b).

Outras técnicas têm sido propostas para avaliação dos fluxos de carbono entre o dossel e a atmosfera (BURBA; ANDERSON, 2010) provendo séries temporais com alta resolução espacial, além de apresentar falhas mínimas e medidas em longo prazo (BALDOCCHI, 2003; CAMPIOLI et al., 2016). Porém, a dificuldade de se capturar os fluxos de CO2 de baixa frequência, geralmente relacionados à respiração noturna em copas densas e de ambientes com topografia irregular, põe em risco as medições do fluxo de carbono (AUBINET, 2008; BALDOCCHI, 2003).

Com o surgimento das ferramentas de sensoriamento remoto, passou a ser possível expandir nossos conhecimentos sobre a PP em grande escala. Essas ferramentas permitem a utilização de outros parâmetros como *proxies* da PP. Muitos estudos optam pela utilização de parâmetros em conjunto, como o índice de área foliar (IAF) e a fração absorvida da radiação fotossintéticamente ativa (fAPAR). Enquanto o IAF é um *proxy* para área de folhas do dossel por unidade de terreno (MYNENI et al., 2002), a fAPAR é um índice derivado do NDVI que tem como base a mensuração da magnitude da radiação disposta nos comprimentos de onda (400-700) absorvidos pela vegetação fotossintéticamente ativa (TIAN, 2004). Como a PP reflete a combinação entre FPAR e IAF, a junção dessas variáveis podem fornecer subsídios importantes para interpretar os efeitos da vegetação sobre a PP dos ecossistemas terrestres (WU et al., 2016).

Usados conjuntamente, os índices de vegetação NDVI (abreviação de *Normalized Difference Vegetation Index*) e EVI (abreviação de *Enhanced Vegetation Index*) oferecem uma ampla sensibilidade sobre o estado da vegetação de diversos tipos de ecossistemas (HUETE et al., 2002), e trazem a possibilidade de fazer relações sobre a produtividade de diferentes coberturas vegetais e perspectivas em diferentes escalas temporais e espaciais (PETTORELLI et al., 2005; PHOMPILA et al., 2015). Enquanto o NDVI apresenta maior sensibilidade em relação à clorofila, o EVI corrige distorções causadas por artefatos atmosféricos e é perceptível às variações na estrutura e tipo de dossel, o que inclui o índice de área foliar (IAF) (Gao et al. 2000; Phompila et al. 2015). Um ponto desfavorável na utilização inicial desses índices era a interferência de elementos atmosféricos como nuvens e aerossóis que podem ter alguma influência na medição da cobertura vegetal, porém atualmente já existem algoritmos que utilizam bandas específicas de reflectância que amenizam esses efeitos (VERMONTE et al., 2016; VERMOTE; KOTCHENOVA, 2008).

Ainda assim, mesmo que com algumas limitações, a principal vantagem das estimativas baseadas em sensoriamento remoto, como os índices NDVI e EVI, é que as mesmas são determinadas pelas propriedades da vegetação. Essas podem ser mensuradas mensal ou semanalmente e oferecerem a possibilidade de calcular a PP média durante maiores períodos de tempo (HUETE et al., 2002). Além disso, tornam possível mensurar a PP por um período selecionado, não apenas sobre médias anuais, como no caso dos modelos baseados na biomassa acima e abaixo do solo (QUINTO-MOSQUERA; MORENO, 2017). A diferença entre a confiabilidade da PP baseada no NDVI e EVI pode ser interpretada como evidência de outros fatores não avaliados em campo como a exclusão de determinados grupos vegetais ou avaliações em função do diâmetro e que subestimam a PP total do ambiente (CHAPIN; EVINER, 2014). Naturalmente, dado o avanço das mudanças de uso da terra, esses índices operam como valiosas ferramentas na avaliação de serviços ecossistêmicos como PP, tanto em maiores períodos de tempo ou escalas mais amplas.

Diversos estudos têm mostrado a importância da PP como suporte para vários outros serviços ecossistêmicos, como o sequestro de carbono e a regulação climática, por exemplo (CAO et al., 2004; VINET; ZHEDANOV, 2011), os quais são imprescindíveis para a manutenção de diversos processos ecológicos e permanência de grande parte dos grupos biológicos existentes, incluindo a espécie humana (COSTANZA et al., 1997; DE GROOT; WILSON; BOUMANS, 2002; VINET; ZHEDANOV, 2011). Estes serviços podem ser classificados em intermediários, quando funcionam como suporte para outros serviços e em serviços finais os quais provem benefícios diretos para o bem-estar humano (BOYD; BANZHAF, 2007; FISHER; TURNER; MORLING, 2009).

Como serviços ecossistêmicos finais fornecidos pela PP, a regulação climática destaca-se como um dos mais notórios. Através do sequestro de carbono pela vegetação, as áreas florestais constituem um dos mais importantes reservatórios do carbono global, e naturalmente, da mitigação dos gases do efeito estufa (CHEN et al., 2014; FEARNSIDE, 2018; PAN et al., 2016). O armazenamento de carbono atmosférico, na forma de biomassa, ocorre ao longo do período de crescimento vegetal e, principalmente, em florestas tropicais não perturbadas, apresenta relações positivas com a regulação climática, através da redução do aquecimento em diferentes escalas espaciais (ALAMGIR et al., 2016; DE GROOT et al., 2010). As altas taxas de evapotranspiração desses ecossistemas auxiliam na redução da temperatura e no aumento da precipitação, proporcionando diferentes condições microclimáticas e microhabitats para diversos grupos biológicos (CORDEIRO; SOUZA; MENDONZA, 2008). Além disso, há materiais derivados da biomassa florestal, como madeiras, fibras e combustíveis, os quais também são considerados serviços finais fornecidos pela PP (MACE; NORRIS; FITTER, 2012).

De qualquer forma, muitos serviços são relacionados à PP e apresentam ligações indiretas e indispensáveis à manutenção de comunidades vegetais e ao funcionamento dos ecossistemas (DE GROOT et al., 2010; DE GROOT; WILSON; BOUMANS, 2002). Em nível ecossistêmico, a manutenção da fertilidade do solo e a ciclagem de nutrientes são a base do crescimento florestal e da produtividade (COSTANZA et al., 2007). Nesse sentido, podem ser observadas relações positivas entre a vegetação e a proteção contra a erosão do solo, uma das atividades mais consideráveis quando aspectos estruturais do ecossistema são levados em conta (LAL, 2014). A proteção contra a erosão dos solos que é fornecida pelas florestas é um serviço importante dos ecossistemas por dois motivos. Primeiro, o sistema radicular é fundamental à estabilidade do solo, enquanto a matéria orgânica (e.g. folhas e galhos) impede sua compactação pela água das chuvas (BARTENS et al., 2008). De maneira igualmente interessante, a vegetação impede a erosão em áreas próximas às encostas, além de contribuir com os processos de sedimentação do solo (DE GROOT; WILSON; BOUMANS, 2002). Indiscutivelmente a PP é um serviço intermediário indispensável à manutenção na vida na Terra, mas, por trás desse fluxo de serviços fornecidos pelas florestas, existe a necessidade de condições ambientais adequadas em troca da manutenção de altos níveis de produtividade.

No geral, o ganho de carbono fotossintético nos ecossistemas é controlado por fatores fisiológicos, como a disponibilidade de luz e água, e ambientais, como oferta de nutrientes, que operam em distintas escalas de tempo alterando a disponibilidade de recursos (CHAPIN; MATSON; VITOUSEK, 2011a). Muitas vezes, a obtenção de um recurso gera demandas que podem custar o crescimento vegetal e, consequentemente, provoca alterações na produtividade que podem ser percebidas em diferentes escalas de tempo (FIELD; RANDERSON; MALMSTRÖM, 1995). Em curtos intervalos de tempo, recursos fundamentais, como luz e água, são os principais controles fisiológicos que moderam o ganho de carbono fotossintético e o crescimento vegetal (FRIEND et al., 2014). Em ocasiões em que a luz é o recurso mais limitante, as folhas ajustam a condutância estomática e a capacidade fotossintética para maximizar o ganho de carbono em ambientes com diferentes disponibilidades de luz (CHAPIN; EVINER, 2014). No entanto, quando a água é o principal fator na determinação da fotossíntese, as plantas aumentam a eficiência no uso da água e reduzem as perdas hídricas (CRAMER et al., 2011; WARING; RUNNING, 1994) ou produzem hormônios que inibem o crescimento e a fotossíntese enquanto as condições ainda são hostis (DAVIES; ZHANG, 1991; NGUYEN et al., 2016).

Por outro lado, durante maiores espaços de tempo, plantas de ambientes com baixa disponibilidade de recursos enfrentam demandas conflitantes entre a absorção de carboidratos e nutrientes necessários para a produção de hormônios e o crescimento (WU et al., 2017). Como estratégia alternativa, as plantas ajustam sua área foliar e capacidade fotossintética que, em associação com o aporte de nutrientes do solo, são responsáveis por grande parte da entrada de carbono nos ecossistemas tropicais (CHAPIN; EVINER, 2014; CLEVELAND et al., 2011; NEPSTAD, 2002). De maneira geral, têm-se assumido que os controles que influenciam a taxa de crescimento, aliados aos nutrientes disponíveis no solo, irão determinar a expansão foliar e capacidade fotossintética e, consequentemente, a entrada de carbono anual na PP em escalas de tempo mais amplas (CHAPIN; MATSON; MOONEY, 2002; FIELD; RANDERSON; MALMSTRÖM, 1995). Lewis et al. (2004) explicam que, quando a disponibilidade de nutrientes é favorável, são esperados aumentos no ganho de carbono, acréscimos nas taxas fotossintéticas e incremento na biomassa florestal, o que implica em mudanças positivas sobre a PP dos ecossistemas.

Se por um lado os controles fisiológicos e ambientais implicam em restrições no crescimento vegetal, por outro, os controles climáticos em escalas globais determinam as diferenças na PP dos ecossistemas. Em florestas tropicais, por exemplo, os fatores mais limitantes para a PP são a disponibilidade de nutrientes no solo e a precipitação (HEISLER-WHITE; KNAPP; KELLY, 2008; LEBAUER; TRESEDER, 2008). A PP associada à estrutura da vegetação e aos recursos abaixo do solo tende a apresentar valores máximos em níveis intermediários de precipitação (CHAPIN; MATSON; VITOUSEK, 2011a; SCHUUR; MATSON, 2001). Isso ocorre em função do aporte de nutrientes no solo, uma vez que valores muito altos de precipitação poderiam provocar processos de lixiviação e alterações nas taxas de decomposição de matéria orgânica, e naturalmente, a perda de nutrientes essenciais para a produtividade (SCHUUR; MATSON, 2001).

As características climáticas de um ecossistema ainda podem ampliar as informações sobre as condições edáficas quando a tipologia vegetal é considerada. Essas características ajudam a prever os efeitos das condições do solo sobre processos fisiológicos e a PP (BATJES, 1996). Por outro lado, a influência do clima não fornece subsídios sobre as características do solo em ambientes submetidos aos efeitos de manejo agrícola. A fragmentação florestal por cultivos agrícolas tem sido retratada em muitos estudos como um importante controle da PP por vários motivos (CRAVEN et al., 2016). Alguns deles estão relacionados às alterações dos controles ambientais (i.e. temperatura, precipitação e aporte de nutrientes) (DEL GROSSO et al., 2008b; FIELD; RANDERSON; MALMSTRÖM, 1995; LEBAUER; TRESEDER, 2008). Outros descrevem os efeitos altamente deletérios na composição de plantas dos fragmentos remanescentes (LÔBO et al., 2011) e as consequências da estruturação de florestas secundárias nas alterações do funcionamento do ecossistema (COJOC et al., 2016; GRACE et al., 2016).

Nesse sentido, a perda da diversidade de espécies é outro fator que traz implicações importantes sobre a PP em ecossistemas florestais perturbados (RUIZ-BENITO et al., 2014; TILMAN et al., 1997). Há mais de um século, Darwin havia proposto que a diversidade de espécies teria uma relação particularmente importante na produtividade dos ecossistemas através da divisão de funções entre as espécies (CARDINALE et al., 2011). E, atualmente, algumas hipóteses sugerem que esse efeito pode estar relacionado à assembleia de espécies e seu desempenho individual para a produtividade, já outras retratam sobre as relações entre fatores bióticos e abióticos como as condições de luz, nutrientes, escala espacial e taxas de crescimento associadas aos gradientes de riqueza das espécies (FLOMBAUM; SALA, 2008; HABERL et al., 2004; NAEEM et al., 1996). De qualquer forma, quando um ambiente apresenta diversos grupos funcionais desempenhando o mesmo papel no ambiente, mesmo com perturbações, o ambiente pode ser menos impactado por causa da redundância dentro de grupos funcionais (TSCHARNTKE et al., 2012). Similarmente, a hipótese da relação de biomassa prevê que a extensão em que os traços das espécies afetam as propriedades do ecossistema estão associadas ao número, abundância relativa e identidade das espécies do ambiente (GRIME, 1998). Naturalmente, o funcionamento do ecossistema é determinando na maior parte pelo o peso dos traços das espécies dominantes para a biomassa vegetal (LAVOREL; GARNIER, 2002).

No entanto, em ambientes fragmentados, a pressão da agricultura, muitas vezes, é a principal força que leva à homogeneização da floresta (ARROYO-RODRÍGUEZ et al., 2013; LÔBO et al., 2011), conduzindo a perdas graduais na diversidade que, consequentemente, podem causar déficits nos níveis da PP local e no funcionamento do ecossistema. Essas perdas devem ocorrer através da substituição de árvores de alta densidade de madeira e de crescimento antigo, por espécies próprias de estágios de sucessão secundária, como por exemplo, lianas e espécies pioneiras com ciclo de vida curto (MELITO; METZGER; DE OLIVEIRA, 2017; SANTOS et al., 2008; TABARELLI; PERES; MELO, 2012b). Invariavelmente, essas substituições alteram a biomassa vegetal, uma vez que as espécies pioneiras e com baixa densidade de madeira não conseguem repor a biomassa perdida por espécies de crescimento tardio, o que pode ocasionar consequências sérias sobre a biomassa vegetal e sobre a PP (MELITO; METZGER; DE OLIVEIRA, 2017).

Futuramente, a transformação da floresta em cultivos agrícolas deve modificar a capacidade na prestação de serviços ambientais (LAURANCE; SAYER; CASSMAN, 2014), como a PP. Em serviços finais como o sequestro de carbono, estima-se que florestas tropicais sejam responsáveis por cerca da metade da absorção do carbono global (PAN et al., 2011). Um papel crucial na mitigação dos gases do efeito estufa quando outros ecossistemas são comparados (LAURANCE et al., 2018). No entanto, tendo em vista como as consequências das mudanças no uso da terra têm impulsionado perdas líquidas de C em regiões tropicais em decorrência da eliminação de florestas para agricultura, pastagens e extração de madeira, essas alterações devem repercutir como uma das forças mais prejudiciais sobres os ecossistemas em um futuro próximo. Por diversas vezes, essas ações têm ocorrido para a maximização de serviços agrícolas enquanto serviços florestais são deteriorados (LAURANCE; SAYER; CASSMAN, 2014; TILMAN et al., 2002). Isso remete a necessidade de planejamento e gestão territorial mais eficientes que permitam uma utilização sustentável dos serviços das florestas tropicais (LAURANCE; SAYER; CASSMAN, 2014). Logo, é indispensável entender como a reestruturação da paisagem pode moldar a PP. Além disso, é necessário avaliar até que ponto a segurança alimentar e energética gerada pelos cultivos agrícolas compensam os prejuízos no funcionamento e na provisão de serviços ecossistêmicos.

2.2 FLORESTA ATLÂNTICA: CONSEQUÊNCIAS DA EXPANSÃO AGRÍCOLA NA ESTRUTURA DA PAISAGEM

Ao deslumbrar a Floresta Atlântica em 1500, Caminha a descreve com tamanho entusiasmo de quem acabara de encontrar o paraíso: “tal era a sua formosura, arvoredo, infinitude, altivez que fazia perder as vistas” (HOLANDA, 2010). Originalmente, a Floresta Atlântica era submetida a diferentes contextos ambientais e cobria uma área de aproximadamente 150 milhões de hectares, integrando uma das maiores florestas tropicais das Américas (METZGER, 2009). Rapidamente, o deslumbre inicial do “Novo Mundo” não foi suficiente para evitar a exploração constante de diversos animais e árvores como o Pau Brasil, uma das primeiras espécies vegetais que deram início à grande retirada de madeira da “Terra Brasilis” (COLONELLI, 2009). A partir disso a Floresta Atlântica foi submetida ao saque de suas árvores, o que deu espaço a maior de todas as “descobertas”: seus solos poderiam abrigar diversos tipos de culturas agrícolas que pareciam ser tão rentáveis quanto os demais recursos naturais da floresta. Café, trigo e pastagens foram algumas das principais práticas agrícolas implementadas (COLONELLI, 2009). Mas, posteriormente, com a queda dessas culturas, a cana de açúcar provou ser a maior fonte de economia e de devastação desse ecossistema.

Séculos mais tarde, com a chegada do Proálcool em 1975, através do decreto n. 76.593, com as promessas de uma cultura promissora, sustentável e limpa, a cana de açúcar foi estabelecida como a principal cultura energética para substituir os combustíveis derivados do petróleo (CORTEZ, 2016). O histórico dessa energia “limpa” foi vivenciado por períodos intermitentes que permaneceram estáveis durante uma década, mas que foram suprimidos em 1990 com a desregulamentação do setor sucroalcooleiro. Novamente, durante três anos, período que compreende os anos de 2003 a 2005, a implementação do etanol como biocombustível trouxe de volta a cana de açúcar e com ela uma nova fase marcada pela mecanização da colheita e pelo aumento da atenção com as mudanças climáticas (CORTEZ, 2016; GALDOS et al., 2013).

De fato, o Proálcool trouxe consideráveis ganhos tecnológicos e econômicos ao país, mas, ambientalmente, os resultados dos avanços no setor sucroalcooleiro tornaram-se uns dos maiores desafios na tentativa de unir práticas agrícolas sustentáveis à conservação florestal (ZAMBERLAN et al., 2014). Por trás da “filosofia” de uma possível fonte de mitigação dos gases do efeito estufa, os cultivos de cana de açúcar enfrentam diversos paradoxos que vão, desde a utilização de pesticidas e fertilizantes, até a própria emissão de gases durante a queima, uma vez que em muitos cultivos até hoje se utilizam queimadas na pré-colheita. Apesar de ser considerada uma fonte de energia limpa, a cana de açúcar não é necessariamente uma cultura sustentável (DUARTE et al., 2015).

Os problemas ambientais causados por cultivos agrícolas podem ser percebidos basicamente em todas as fases do manejo. Em monoculturas de cana de açúcar, por exemplo, durante fase de plantio, a utilização de fertilizantes e pesticidas para a extensão e garantia da produção, podem levar ao acúmulo de metais pesados e à contaminação do solo (CONCEIÇÃO; BONOTTO, 2005; MARULL et al., 2017). Atualmente, têm-se reconhecido os agravantes causados por pesticidas de uso comum, como os neonicotinoides, que representam riscos substanciais tanto para abelhas comuns, como para as selvagens (WOODCOCK et al., 2017). Alterações nos sistemas de polinização podem causar efeitos drásticos tanto no setor socioeconômico, com a contaminação de alimentos, como no funcionamento dos ecossistemas, através da contaminação de pólen e néctar e da criação de déficits na manutenção de diversos processos ecossistêmicos, inclusive nos serviços críticos de polinização das próprias lavouras (MCART et al., 2017; NASREDDINE; PARENT-MASSIN, 2002; SIMON-DELSO et al., 2017).

Outro fato preocupante sobre os pesticidas utilizados durante o estágio de crescimento das lavouras são os impactos causados em áreas próximas aos cultivos agrícolas. Esses efeitos podem ser agravados pela irrigação e pelos processos de lixiviação, os quais promovem a salinização, a contaminação de lençóis freáticos e alterações dos fluxos hídricos e das características químicas e físicas do solo (FOLEY, 2005a; KIBBLEWHITE; RITZ; SWIFT, 2008; LIBUTTI; MONTELEONE, 2017; YEN; LIN; WANG, 2000). Outro paradoxo são as queimadas durante a pré-colheita, uma vez que essa prática ainda é bastante comum em várias regiões do Brasil, incluindo o Nordeste (RIBEIRO, 2008). A queima da folhagem e da matéria orgânica do solo é considerada uma das principais fontes emissoras de gases na atmosfera (FEARNSIDE, 2018; FEARNSIDE et al., 2009; GEORGESCU; LOBELL; FIELD, 2009). Além disso, os resíduos lançados pelas queimadas causam impactos negativos na qualidade do ar e aumentam as chances de aparecimento de diversas doenças respiratórias nas populações próximas aos canaviais (LE BLOND et al., 2017; RIBEIRO, 2008).

É notória a importância das práticas de usos da terra, uma vez que fornecem recursos para o abastecimento de alimentos, fibras e combustíveis (VINET; ZHEDANOV, 2011). Por outro lado, a perda de hábitat causada por cultivos agrícolas pode trazer prejuízos sem precedentes aos ecossistemas (RANTA et al., 1998a). A "formosura" da Floresta Atlântica, retratada anteriormente por Caminha, apesar de ainda ser bastante fascinante, atualmente encontra-se reduzida a pequenos fragmentos cada vez mais ameaçados e incertos para a manutenção de serviços ecossistêmicos e uma grande variedade de grupos biológicos (HADDAD et al., 2015; RIBEIRO et al., 2009). As consequências da fragmentação florestal, em geral, são paisagens formadas por remanescentes de vegetação nativa, embebidas por uma matriz agrícola ou outro tipo de uso da terra (SAUNDERS; HOBBA; MARGULES, 1991). Comumente, essas paisagens tendem a se tornar matrizes inóspitas manejadas por humanos, onde a pressão de diversos efeitos deletérios, como a interrupção da conectividade biológica e o isolamento de fragmentos utilizados como hábitat se tornam cada vez mais frequentes (LAURANCE et al., 2002; SANTOS et al., 2008; SAUNDERS; HOBBA; MARGULES, 1991).

Na maior parte das vezes, a "involução" da fragmentação florestal sobre as paisagens se inicia com a criação de bordas florestais e se intensifica conforme ocorre a perda de habitat e da conectividade entre os fragmentos remanescentes (FAHRIG, 2003). Os efeitos da criação de bordas incluem alterações microclimáticas como a redução da umidade, aumento da luz e maior variabilidade da temperatura, as quais podem penetrar até 60 m nos interiores das manchas de habitat (HARPER et al., 2005; KAPOS et al., 1989). Como efeitos primários da criação de bordas são esperados danos na vegetação e no solo através de mudanças na evapotranspiração e aderência de nutrientes no solo. Essas alterações promovem a dessecação do habitat e afetam negativamente a estrutura florestal, incluindo a cobertura do dossel, área foliar, densidade de madeira e biomassa vegetal. Posteriormente, efeitos secundários causam mudanças na regeneração e mortalidade de árvores, alimentando ainda mais as perdas de espécies de floresta madura e de alta biomassa (HARPER et al., 2005; LAURANCE et al., 2002).

Em contrapartida, espécies pioneiras e de baixa densidade de madeira são beneficiadas e dão início ao processo de sucessão secundária (TABARELLI; LOPES; PERES, 2008). A partir desse ponto, as paisagens que antes eram formadas por espécies de crescimento tardio dão espaço aos grupos de crescimento rápido e ao colapso da biomassa (LAURANCE et al., 2002; PIMM, 1998). Conforme os processos de sucessão retrogressiva e o colapso de biomassa avançam, o desmonte de assembleias de árvores nas bordas florestais promove a simplificação da vegetação e domínio de espécies pioneiras. Futuramente, a simplificação da paisagem será uma das principais forças no empobrecimento de espécies e homogeneização regional da vegetação (LÔBO et al., 2011; SANTOS et al., 2008; TABARELLI; LOPES; PERES, 2008). As consequências da simplificação da vegetação são florestas ralas, com prováveis déficits no funcionamento ecológico e com rupturas da provisão de serviços ecossistêmicos em paisagens totalmente alteradas estruturalmente (LAURANCE et al., 2002).

Atualmente, vários estudos têm demonstrado que a estrutura da paisagem é uma característica potencial na forma com que o funcionamento e a provisão de serviços dos ecossistemas são moldados (TSCHARNTKE et al., 2012). Paisagens tropicais modificadas por ações humanas são reconhecidas por suas estruturas altamente heterogêneas e amplas variações em termos de composição e configuração espacial (FAHRIG, 2003). Enquanto a composição se refere aos tipos e proporções de diferentes usos da terra inseridos na paisagem (e.g. quantidade de cobertura florestal ou outro tipo de uso da terra), a configuração está relacionada ao arranjo espacial desses tipos de uso (e.g. número de fragmentos florestais e densidade de borda) (DUNNING; DANIELSON; PULLIAM, 1992). Por outro lado, apesar dessas medidas já serem antigas conhecidas na ecologia de paisagens, os meios nos quais a estrutura da paisagem influencia a provisão dos serviços dos ecossistemas têm apresentado resultados contrastantes ao longo dos estudos (FAHRIG, 2003; FAHRIG et al., 2011; MITCHELL et al., 2015; TSCHARNTKE et al., 2012; TURNER; DONATO; ROMME, 2013).

Para diversos serviços ecossistêmicos como sequestro de carbono (LAURANCE et al., 2011), polinização (RICKETTS et al., 2008) e, incluindo vários grupos biológicos como aves (CARRARA et al., 2015; DE CAMARGO; BOUCHER-LALONDE; CURRIE, 2018), morcegos (ARROYO-RODRÍGUEZ et al., 2016) e besouros (SÁNCHEZ-DE-JESÚS et al., 2016), as medidas de composição (e.g. perda de habitat) têm se mostrado relativamente mais limitantes e menos variáveis quando comparadas às medidas de configuração da paisagem (FAHRIG, 2003). De fato, a forma particular em que o habitat de cada paisagem é moldado após os processos fragmentação, assim como sua localização específica (RICKETTS et al., 2008) e diferentes efeitos de borda (LAURANCE et al., 2007; RIES; SISK, 2004), sugerem que a configuração espacial pode apresentar efeitos altamente variáveis (i.e. negativos, positivos ou neutros) no tempo e no espaço (FAHRIG, 2003; LAURANCE et al., 2007). Obviamente, as influências da composição e configuração da paisagem sobre os serviços dos ecossistemas devem ocorrer por vários motivos.

Dentre as medidas de composição, a perda de habitat tem surgido como uma das principais ameaças aos serviços ecossistêmicos (FAHRIG, 2003, 2013). Em regiões de habitat contínuo a prestação de serviços pode ser mais elevada por diversas causas: fragmentos maiores são menos impactados por eventos de perturbação crônica (CHAPMAN; PERES, 2001), os quais conseguem manter maiores populações e com baixas probabilidades de extinção (HANSKI, 1998; JOERN; LINDENMAYER, 2007), uma vez que possuem maior disponibilidade de recursos. Conjuntamente, esses fatores podem significar a persistência de fluxos bióticos e abióticos, cruciais para o funcionamento do ecossistema e prestação de serviços como o estoque de carbono (MAGNAGO et al., 2017). Tradicionalmente, desde a teoria de biogeografia de ilhas (MACARTHUR; WILSON, 1988), têm se reconhecido que a riqueza de vários grupos taxonômicos é proporcional à quantidade de habitat nas paisagens, como visto para plantas (PIESSENS et al., 2004), aves (CARRARA et al., 2015) e mamíferos (GARMENDIA et al., 2013). Embora as relações entre espécie-área sejam, teoricamente, aplicadas na análise entre manchas florestais e riqueza de espécies, esse discurso é bastante antigo e pode contribuir para a compreensão de como a quantidade de habitat pode influenciar os processos e a provisão de serviços dos ecossistemas (MATTHEWS; COTTEE-JONES; WHITTAKER, 2014).

Mas, se por um lado, os efeitos da composição da paisagem sobre a provisão de serviços estão, principalmente, relacionados à perda de habitat, por outro, várias causas podem explicar as flutuações dos efeitos da configuração da paisagem (FAHRIG, 2003). Em relação aos efeitos de borda, no geral, espera-se que seus efeitos deletérios sejam mais fracos e menos penetrantes em fragmentos com maiores proporções de habitat e área de núcleo (EWERS; DIDHAM, 2007; MURCIA, 1995). Isto porque paisagens com maior área de interior capturam menos as alterações microclimáticas dos efeitos de borda, que poderiam ser ainda mais amplos em fragmentos com menor área de núcleo (MAGNAGO et al., 2017). Da mesma forma, a idade das bordas também pode ser um fator importante na magnitude dos seus efeitos (KAPOS et al., 1989; LAURANCE et al., 2007). Ao longo do tempo, é de se esperar que a mortalidade de árvores diminua, principalmente, porque as bordas são fechadas por espécies tolerantes e porque as diferenças microclimáticas entre interior e borda de fragmentos são minimizadas (LAURANCE et al., 2007).

Em contraste, fragmentos menores e com formas mais complexas são mais impactados pelos efeitos de borda, pelo fato de que o estabelecimento de espécies pioneiras e de madeiras de baixa densidade reduzem a biomassa e a estratificação vegetal (CHAPLIN-KRAMER et al., 2015; LAURANCE et al., 2002; MELITO; METZGER; DE OLIVEIRA, 2017), o que pode influenciar diretamente nos serviços finais derivados da PP como o estoque de carbono (BRINCK et al., 2017; MAGNAGO et al., 2017). Outro efeito negativo são as secas periódicas que podem aumentar a variabilidade dos efeitos da criação de bordas, dada a inerente vulnerabilidade das florestas tropicais à dessecação. Na Amazônia, onde secas foram causadas por oscilações do El Niño, a criação de bordas florestais estendeu seus efeitos causando a mortalidade de árvores em resposta das alterações nas condições microclimáticas (KAPOS et al., 1989; LAURANCE et al., 2007). De toda forma, essas alterações podem ser ainda mais severas, quando reduzem a qualidade e a quantidade de matéria orgânica no solo e a biomassa aérea e subterrânea, repercutindo na redução de cerca da metade da quantidade do carbono florestal (MA et al., 2017). Possivelmente, todas essas variações surgirão com efeitos em cascata, nas quais são esperadas alterações no fluxo de serviços dos ecossistemas, muitas vezes, com efeitos integrados entre a perda de habitat e a proliferação de bordas (MELITO; METZGER; DE OLIVEIRA, 2017).

No que diz respeito ao efeito da fragmentação *per se* (i.e. número de fragmentos), estudos realizados em florestas tropicais e temperadas, indicam que efeitos negativos em relação ao aumento no número de manchas podem estar relacionados à perda da conectividade e ao isolamento entre fragmentos na paisagem (HADDAD et al., 2015, 2017). Nessas circunstâncias, a redução da conectividade entre os fragmentos remanescentes pode resultar na limitação do movimento de dispersores de sementes e no aumento da densidade de predadores. Tais eventos podem conduzir ao declínio de serviços que dependam diretamente de organismos móveis (e.g. polinização e regulação de pragas) (KREMEN et al., 2007; LAURANCE et al., 2002; MENDES; RIBEIRO; GALETTI, 2015). Por outro lado, quando o aumento da fragmentação resulta em efeitos positivos, é esperado que os mesmos estejam ligados desde a criação e expansão de subpopulações e persistência de espécies nas paisagens mais fragmentadas (HANSKI, 1998), à redução do isolamento entre manchas florestais e acesso aos recursos localizados nas bordas (DUNNING; DANIELSON; PULLIAM, 1992).

Quanto aos efeitos neutros da configuração da paisagem, vários estudos demonstram que, quando a qualidade da matriz adjacente aos fragmentos florestais é considerada, os impactos da fragmentação sobre o funcionamento e prestação de serviços podem ser amortecidos (TSCHARNTKE et al., 2005, 2012). A natureza da matriz determina o quanto impede ou facilita a dispersão de organismos entre as manchas de habitat (CRONIN, 2007). Basicamente, as matrizes de paisagens tropicais modificadas através de ações humanas podem variar desde as mais complexas estruturalmente (e.g. sistemas agroflorestais e florestas secundárias), até as mais simples (e.g. monoculturas) (TSCHARNTKE et al., 2007). Teoricamente, vários serviços como polinização, controle de pragas e dispersão de sementes podem ser mantidos em paisagens com matrizes de alta qualidade (EKROOS et al., 2015; THIES; TSCHARNTKE, 1999). Uma razão para isso é que, muitas vezes, os organismos de paisagens fragmentadas podem explorar recursos que estão espalhados nas manchas de habitat. Paisagens compostas por matrizes complexas podem manter o movimento de polinizadores e dispersores entre manchas não adequadas para habitat, desde que a matriz tenha recursos complementares ou suplementares fora do habitat (DUNNING; DANIELSON; PULLIAM, 1992).

Por outro lado, paisagens com matrizes homogêneas, como monoculturas de cana de açúcar, são caracterizadas pela baixa capacidade de prestação de serviços (TAKI et al., 2011). Uma resposta relacionada é que os efeitos da configuração da paisagem podem ser mais penetrantes em virtude da dureza da matriz e seus efeitos altamente hostis para o movimento de organismos entre as manchas de habitat mais isoladas, além das grandes diferenças de microclima (PINTO et al., 2010; TSCHARNTKE et al., 2005, 2012). Naturalmente, essas características podem custar a permanência de grupos taxonômicos necessários para a continuidade da prestação de serviços. No caso da PP, isso pode significar muito já que para ser mantida funcionalmente necessita, principalmente, da continuidade da dispersão de sementes de árvores de floresta madura (CARDINALE et al., 2012; MAGNAGO et al., 2017; PRIMACK; MIAO, 1992).

Da mesma forma, os efeitos da simplificação da paisagem devem ser ainda mais impactantes quando alteram as características funcionais das plantas, as quais consistem em componentes-chave para funções do ecossistema (FALSTER et al., 2011; LAVOREL; GARNIER, 2002; TSCHARNTKE et al., 2012). Alterações em traços funcionais como altura média das plantas, cobertura de área foliar e densidade de biomassa podem ser considerados como os principais efeitos deletérios sobre a PP, uma vez que são os traços mais relevantes para produtividade de um ecossistema (FALSTER et al., 2011; GRIME, 1998). Normalmente, mudanças nos traços das plantas que afetam a produtividade tem consequências que se estendem além do ecossistema onde a planta ocorre e podem repercutir em grandes escalas (CHAPIN, 2003).

Além disso, conforme a matriz inóspita aumenta, a simplificação da vegetação exerce efeitos potencialmente prejudiciais diminuindo cada vez mais a heterogeneidade estrutural da paisagem. Paisagens mais heterogêneas têm maior capacidade de fornecer serviços e sustentar maior diversidade biológica do que paisagens mais simples estruturalmente, ainda que estas tenham a mesma quantidade de habitat daquelas mais complexas (TSCHARNTKE et al., 2012). No entanto, tem sido proposto que devem haver diferentes intensidades de perturbação (de baixo a médios), nos quais os distúrbios da perda e fragmentação de habitat não são suficientemente altos para provocar colapsos nos níveis de provisão de serviços e na biodiversidade (MELO et al., 2013). Basicamente, a capacidade de prestação de serviços da paisagem é caracterizada pela intensidade de distúrbios, dentro dos quais, até certo ponto, a perturbação pode aumentar sua provisão de serviços como biocombustíveis de madeiras.

Por outro lado, se expansão da agricultura, exige novos eventos de perda e fragmentação do habitat, essas paisagens tornam-se cada vez mais homogêneas pela predominância de grupos generalistas que irão provocar colapsos na variabilidade espacial de serviços ecossistêmicos e repercutir em estados irreversíveis de degradação ecológica (ARROYO-RODRÍGUEZ et al., 2013; LÔBO et al., 2011; MELO et al., 2013; TSCHARNTKE et al., 2012). A simplificação da paisagem, atualmente é reconhecida como uma das maiores forças associadas às perdas de serviços, essas perdas podem gerar efeitos em cascata para afetar múltiplos serviços da paisagem, como a polinização e o controle de pragas (MEEHAN et al., 2011). Futuramente, a simplificação de paisagens tropicais causada pela perda e fragmentação de hábitat, e em meio às pressões causas pelas demandas alimentares da população humana e seu crescimento exponencial (TILMAN, 2001), trarão obstáculos ainda mais ameaçadores para os ecossistemas, à medida que as demandas bioenergéticas aumentam e o provisionamento de serviços ecossistêmicos diminui

(MCDONALD et al., 2009; MITCHELL et al., 2015; RICKETTS et al., 2008; ROBERTSON et al., 2008).

Esses obstáculos incluem os efeitos da intensificação agrícola e suas inúmeras práticas invasivas sobre os ecossistemas que terão como resultado dramático o abandono de terras agrícolas e a criação de “novos ecossistemas”, os quais consistem em paisagens compostas por espécies invasoras e estado de conservação difícil ou irreversível (CRAMER; HOBBS; STANDISH, 2008; MURCIA et al., 2014). Basicamente, é um novo ambiente em grande escala com processos completamente modificados e ajustados às condições atuais de perturbação, tendendo-se a auto organizar e manter seus padrões que são novos e sem intervenção humana extensiva (HOBBS et al., 2006). Embora existam opiniões altamente divergentes sobre os novos ecossistemas (ARONSON et al., 2014; HOBBS; HIGGS; HARRIS, 2014; MURCIA et al., 2014), a remodelação da paisagem implica não somente em ecossistemas não resilientes, mas também na interrupção de serviços ecossistêmicos, uma vez que uma mudança no fluxo de um serviço pode desencadear várias pressões ecológicas (BENNETT; PETERSON; GORDON, 2009). Além disso, mesmo que houvessem formas eficazes de restauração, não existem certezas sobre quais espécies e características funcionais poderiam se adequar as novas condições biofísicas, exibir resiliência, manter a biodiversidade e fornecer serviços às pessoas (MAESTRE et al., 2012).

Como visto anteriormente, a produtividade pode ser influenciada por diversos controles, que, em geral, envolvem desde fatores fisiológicos e ambientais até a perda de traços funcionais das plantas. Em cenários atuais, onde as mudanças na estrutura da paisagem beneficiam espécies pobres em ganho de carbono (ARROYO-RODRÍGUEZ et al., 2013; HARPER et al., 2005; LÔBO et al., 2011) em detrimento de árvores tolerantes à sombra e com alta densidade de madeira, e que contribuem com os maiores ganhos de carbono e produtividade dos ecossistemas (ANDERSON-TEIXEIRA et al., 2016; MAGNAGO et al., 2017), não seria surpreendente esperar que essa reestruturação tivesse influencias sérias na PP.

Dadas as circunstâncias, é urgente entender os efeitos da agricultura de larga escala sobre a PP. Ainda que diversos estudos descrevam a influência da estruturação da paisagem sobre os serviços ecossistêmicos, a maioria se concentra nos serviços de polinização, controle biológico e biodiversidade (ALVARADO et al., 2017; ARROYO-RODRÍGUEZ et al., 2016; LAMY et al., 2016). Enquanto que os efeitos da reestruturação da paisagem promovidos pela agricultura de larga escala sobre a produtividade, raramente são revelados. Apesar das consequências da expansão agrícola sobre o funcionamento e prestação de serviços ecossistêmicos já serem esperados esses efeitos são até hoje mal compreendidos. Ao mesmo tempo, entender as modificações na estrutura da paisagem de antigas fronteiras agrícolas pode ser bastante útil para compreender os caminhos futuros em fronteiras mais recentes como a Amazônia.

3 LANDSCAPE PATTERNS OF PRIMARY PRODUCTION REVEAL AGRICULTURAL BENEFITS FROM FOREST CONSERVATION

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Landscape patterns of primary production reveal agricultural benefits from forest conservation

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Abstract

Primary Production (PP) is a fundamental ecosystem function that influences several ecosystem services, but little is known about how changes in landscape structure resulting from agricultural expansion affect forest and cropland production. We investigated the effect of landscape composition and configuration on the PP of forests and sugarcane crops in an old agriculture frontier of the Brazilian Atlantic forest. Using satellite images and a comprehensive dataset of vegetation indices, we sampled 120 landscapes of 1 km² during the rainy and dry seasons of 2011-2016. In each landscape we quantified the percentage of croplands (composition measure), density of forest edges, number of forest patches (configuration measures), and estimated the Normalized Difference Vegetation Index (NDVI) and the Enhanced Vegetation Index (EVI) of the remaining forest, croplands and the entire landscape. The NDVI and EVI of the entire landscape were negatively influenced by the percentage of croplands in both seasons and less affected by edge density and number of forest patches. When analyzing the NDVI of forests and croplands separately, both were negatively affected by the percentage of croplands, suggesting negative effects of cropland on forest production and positive effects of forest on sugarcane production – a synergy between conservation and production. The spatial variability of NDVI and EVI was not affected by landscape attributes. Our findings indicate that landscape composition (i.e. forest cover) is more important than spatial configuration in determining the PP of forest and sugarcane crops. Thus, to ensure productive forests and croplands in these old, heavily deforested agricultural frontiers, we should protect any forest remnant in the region and restore abandoned lands to increase landscape forest cover. We also recommend that large-scale sugarcane production in Brazil avoids forest fragmentation in forest-dominated landscapes and protects the remaining forest from fire during sugarcane harvesting. The potential synergy between conservation and production should guide future policies aimed at offsetting the environmental liabilities of this long-lasting, large-scale agricultural system.

Keywords: ecosystems service, forest biomass, habitat loss, habitat fragmentation, human-modified landscapes, landscape composition, landscape configuration, landscape structure

Introduction

Tropical forests have been widely converted into agricultural fields, causing habitat loss and fragmentation and reducing the provision of some ecosystems services such as climate regulation, biological control of pests and pollination (Birkhofer et al. 2018, Fearnside 2018, Laurance et al. 2014, Mcart et al. 2017). This global phenomenon modifies landscape composition and configuration and affects the remaining ecosystem in all levels of biological organization, including those ecosystem functions that directly or indirectly satisfy human needs (Fahrig et al. 2011, Tscharntke et al. 2012). While landscape composition refers to the percentage of area covered by different land uses, landscape configuration reflects the different spatial arrangement of land uses, such as the number of habitat patches and the density of edges within the landscape (Fahrig et al. 2011). Some ecosystem services may respond more strongly to landscape composition (Arroyo-Rodríguez et al. 2016, Carrara et al. 2015), others are more susceptible to changes in landscape configuration or respond to both (Lamy et al. 2016). Primary production (PP), an intermediary ecosystem service responsible for the provision of final services (i.e. food, fiber, biofuel), is amongst the most important ecosystem services in agricultural landscapes (Costanza et al. 2007). Nonetheless, to our knowledge, there is no study relating agricultural PP with landscape composition and configuration, especially in old agricultural frontiers. Understanding how PP of natural habitats and croplands responds to changes in landscape structure may help to identify synergies and trade-offs between food production and nature conservation.

When a forested landscape is partially converted to large-scale agriculture, the remaining forest is split into patches, forest edges are created elsewhere in the landscape and croplands becomes inhospitable to many forest-dependent organisms. These structural changes may increase precipitation in the landscape at the expense of forest moisture (Avissar and Liu 1996, Baidya and Avissar 2000, Chen and Avissar 1994). In a process known as vegetation breeze, the moist air is pulled away from forests into adjoining croplands and condenses into rain-producing clouds, then is recycled as dry air back over the forest (Cochrane and Laurance 2008). The vegetation-breeze phenomenon causes forest desiccation especially along the edges and imposes severe physiological constraints to trees, causing species loss and subtle biomass collapse as the recruitment of species that benefit from the new conditions does not surpass the mortality of sensitive species in the long run (Harper et al. 2005, Laurance et al. 2018). Such biotic and abiotic changes are likely to reduce the PP of agricultural landscapes with low forest cover, high edge density and elevated number of small (<10 ha), edge-dominated forest patches. The reduction should be even stronger during the dry seasons due to low water availability for photosynthesis and limited vegetation breeze.

The structural modifications may also reduce the spatial heterogeneity within and among the land uses that make up the landscape. In a simplistic view, the land uses of agricultural landscapes in tropical regions may be classified as forest or cropland, which are mutually dependent to each other and ultimately determine the PP of the entire landscape. Because forest loss and fragmentation favor the dispersal of a few disturbance-adapted tree species with limited capacity to store carbon (Chaplin-Kramer et al. 2015, Lôbo et al. 2011), the PP of forests is expected to be lower and spatially more homogeneous in cropland-dominated landscapes. The PP of croplands is also expected to be lower and more homogeneous in these landscapes because vegetation breeze weakens as forest is gradually eliminated (Cochrane and Laurance 2008) and planted cultivars vary little in terms of photosynthetic activity and productivity. However, agricultural landscapes actually encompass a continuum of landscape composition from forest-dominated to cropland-dominated landscapes with varying spatial configuration. In one extreme of the gradient there are totally forested landscapes that provide carbon storage, climate regulation and other ecosystem services but little food or biofuel; in the other extreme there are totally cultivated landscapes, which produce much food and biofuel at the expense of the other ecosystem services (Foley 2005). Assessing how the PP of croplands and forests respond to landscape composition and configuration helps to design possible solutions to reduce the environmental liabilities of large-scale food production (Haddad et al. 2015; Johnson et al. 2014; LAURANCE et al. 2014).

The Brazilian Atlantic Forest is one of the most threatened global biodiversity hotspots (Myers et al. 2000). More than 80% of its original cover have been occupied by cities, cut by roads, fragmented into almost 250,000 forest patches, but mainly replaced by vast agricultural fields (Ribeiro et al. 2009). Forest conversion for agriculture started in the 16th century following Portuguese colonization and intensified in the last decades of the 20th century (Colonelli 2009). As in many tropical regions, small-scale agriculture in the Atlantic Forest has been replaced by large-scale monocultures to address industrial requirements for considerable supplies of uniform and reliable products, largely ignoring the depletion of ecosystem services (Fearnside 2018, Ferreira et al. 2015, Foley et al. 2005, Power 2010). As much as agricultural intensification offers food and energy security, their costs have not compensated for potential losses in forest service capacity and even crop benefits (Laurance et al. 2014).

Although forest conservation plays a crucial role in the agricultural benefits (Power 2010, Ricketts et al. 2008, Tscharntke et al. 2005), this relationship has not been well understood. Vegetation indices derived from remote sensors have been widely used to study carbon uptake patterns among vegetation types at various spatial and temporal scales (Aragón and Oesterheld 2008, Guerschman et al. 2003, Huete et al. 2008, Paruelo et al. 2001, Shi et al.

2017), given their relatively low-cost data acquisition for a wide temporal and spatial range (Pettorelli et al. 2005). Despite this, we do not know any study describing landscape configuration effects on vegetation greenness or carbon gain. Landscape configuration is an important and immediate variable that changes with land use change. By describing the vegetation index patterns in landscapes with different configurations, we may uncover possible landscape configuration effects on PP not observed in previous studies.

In this work we assess the landscape patterns of PP in a 5,000-km^2 region of the Brazilian Atlantic Forest that is among the oldest frontiers of sugarcane plantation in the country (Ranta et al. 1998, Ribeiro et al. 2009). Using satellite images and a large, reliable dataset of vegetation indices (NDVI and EVI from 2011 to 2016), we sampled 120 landscapes of 1 km^2 within the region and test for the effect of the percentage of croplands, density of forest edges and number of forest patches on the PP of forests, croplands and the entire landscape. We also examined how these landscape metrics affect the spatial variability within croplands and the remaining forest and assessed the PP in rainy and dry seasons to evaluate how precipitation may mediate the impact of landscape attributes on production (Del Grosso et al. 2008).

Methods

Study region

Located in the Paraiba coast of Northeast Brazil, the study region covers 525,556 ha of the Pernambuco Endemism Center of the Brazilian Atlantic Forest (Fig. 1), which is a national priority for conservation (Ranta et al. 1998, Ribeiro et al. 2009). The region is covered by a mosaic of different vegetation types, including tropical rainforests, less humid forests and cerrado-like vegetation, locally called ‘*mata de tabuleiro*’ (Silva-Junior et al.

2018). The vegetation types grow on two predominant soil classes: latosols and argillosols, and in smaller proportions on neosols and spodosols (IBGE, 2008). Climate is warm and humid tropical (Koppen classification), with annual precipitation of 1800 mm approximately. Rainy season usually spans from March to August and dry season from September to February. The thermal amplitude is low, with maximum average temperature reaching 27.5ºC among the months of January and March and minimum average temperature of 24ºC in the months of June and July (Pereira et al. 2012).

The sugarcane monocultures were established in the coastal region of Paraiba a couple of centuries ago (Moreira and Targino 1997), but intensified in the 1970's owing to the National Alcohol Program (Pro-Álcool), whose main goal was to produce sugarcane at large scale to support the replacement of gasoline and other petroleum-based fuels by sugarcane-derived ethanol (Cortez 2016). The program accelerated deforestation and forest fragmentation at alarming rates and homogenized the agricultural landscapes with vast sugarcane crops. Currently, forest patches of up to 4,366 ha remains, but more than 95% are less than 45 ha and as in other parts of the Northeastern Atlantic Forest, forest patches are almost all, if not all, are dominated by secondary forests (Santos et al. 2008). This environmental liability has been underappreciated with the excuse that ethanol is a biofuel produced with environmental-friendly techniques (Cortez 2016). However, the use of fire is still allowed in sugarcane harvest in many Brazilian states; the sugarcane to be harvested is totally burnt to facilitate manual harvest, negatively impacting the soil and the forest-dependent biota that eventually supplements their habitat in adjoining croplands. In our study region, harvest usually begins in September and coincides with the dry season, while sugarcane growth usually takes place in the rainy season, if irrigation is not implemented.

Proxies of Primary Production

We used the Normalized Difference Vegetation Index (NDVI) and the Enhanced Vegetation Index (EVI) as proxies of PP. These indices are strongly correlated with various plant properties, including (1) amount of photosynthetic active radiation, (2) chlorophyll content, (3) leaf area, (4) amount of green biomass and (5) canopy structure (Huete 2011). Together, these properties provide an important measure of the vegetation photosynthetic capacity and, therefore, the primary production of the ecosystems (Huete et al. 2002, Pettorelli et al. 2005). While the NDVI is related to chlorophyll content, EVI is sensitive to variation in canopy structure, which includes leaf area index (LAI), plant physiognomy and canopy architecture (Gao et al. 2000). Therefore, the use of both indices confer robustness in pattern observation. Both indices vary from -1 to +1; values close to +1 are related to environments with high dense vegetation, elevated photosynthetic capacity and high production, while values close to zero indicate low production and are related to environments with little vegetation cover or naked soil (Pettorelli et al. 2005). Negative values usually indicate waterbodies.

Vegetation indices were obtained from the MODIS sensor (Moderate Resolution Imaging Spectroradiometer), product MOD13Q1. This product provides a fine temporal resolution through the observation of the entire Earth’s surface every 1 to 2 days and vegetation indices at 16-day intervals with a spatial resolution of ~6.25ha. The 16-day composite data minimizes atmospheric and bidirectional reflectance distribution function (BRDF) influences (Huete et al. 2002).

Tropical regions are affected by cloudy condition all year long, therefore we thoroughly analyzed available data in the following way. For this region and for the period

2000-2016, we first filtered residual cloud and aerosol contamination based on the quality assurance (QA) flags and the mixed cloud, possible shadow and aerosol quality flag provided with the MOD13Q1 product. Then we selected those months that had more quantity of good quality data and were representative of vegetation seasonality, which were June-July for the rainy season and November-December for the dry season. In spite of using gap-filling data methods to cope with missing data, we preferred to maximize available good quality data for all landscapes, by pooling 6 years of vegetation indices (2011-2016, 6 years average) for the humid and the dry season. This allowed us to randomly sample 60 landscapes from a total of 356 that had at least 80% of the landscape (13 out of 16 pixels), without aerosol, clouds or shadow effect. About 75% of the landscapes were covered by 16 pixels (see Appendix S1 for more details on the stratified random sampling design we have used).

Landscape sampling

In each landscape, we measured the percentage of croplands – our metric of landscape composition – and two metrics of landscape configuration: density of forest edges and number of forest patches (Carrara et al. 2015). To estimate the edge density, we corrected eventual spatial errors in the file and converted it to raster format with a 5-m pixel resolution. We performed the calculation of edge density with QGIS and FRAGSTATS v4 and visually counted the number of forest patches in each landscape.

To estimate the PP of croplands and forests within each landscape, we were very conservative and selected only those pixels completely covered by the respective land use. This criterion avoided that a pixel dominated by sugarcane but with a small fraction of forest had its vegetation indices overestimated, as forests are presumed to be more productive than sugarcane crops. Similarly, it also avoided that a forest-dominated pixel with pieces of sugarcane crops had its indices underestimated. This procedure reduced the number of pixels from which the values of mean and coefficient of variation (CV) of land uses were calculated, but ensured confident comparisons across the gradient of landscape composition and configuration.

We used the mean value of NDVI and EVI as proxy of PP and their CV as proxy of spatial variability in PP. The greater the CV, the greater the spatial variability and functional heterogeneity of the land use. Landscapes with only one or no pixel of a given land use was not sampled for spatial variability because CV required at least two pixels to be calculated. Similarly, totally or very forested landscapes had one or no sugarcane pixel, thus we were unable to calculate the CV of their crops. The CV of forest and croplands were calculated from 22 and 27 landscapes during the rainy season and 25 and 29 landscapes during the dry season, respectively.

Data analyses

We performed generalized linear models to test for the effect of landscape metrics on the PP of the remaining forest, croplands and the entire landscape (Crawley 2007). All models had the percentage of croplands, density of forest edges and number of forest patches as explanatory variables. The mean value and the CV of the vegetation indices were set as response variables. We fitted the models with Gaussian distribution after verifying that the residuals of the response variables had normal distribution, visualized through the graphic analysis of the ‘qqnorm’ and ‘qqline’ functions of R software. Using the car package of R version 3.0.1, we calculated the variance inflation factor (VIF) to estimate the multicollinearity among the predictors. VIF > 4 indicates possible collinearity, whereas VIF > 10 indicates high collinearity (Kutner 2004). All VIF values were lower than 2 (ranging from 1.00 to 1.90 in rainy season and from 1.00 to 1.45 in dry season), indicating independence among the explanatory variables. In previous exploration of the data, we verified that soil type had no effect on PP and therefore did not include it as a covariable in the models. All analyzes were performed in R and JMP 8.0.

Results

During the rainy season, the increase in the percentage of croplands negatively affected the mean NDVI and EVI of the entire landscape and the mean NDVI of croplands and forest (Table 1, Fig. 2), indicating that in sugarcane-dominated landscapes both cropland and forest are less productive per unit of area than they are in forest-dominated landscapes. Density of forest edges and number of forest patches had no significant effect on any vegetation index, except for the NDVI of the forest, which decreased in more fragmented landscapes (Table 1, Fig. 2).

During the dry season, the pattern remained quite similar, i.e. the increase in the percentage of croplands negatively affected the mean NDVI and EVI of the entire landscape and the mean NDVI of the forest (Table 1, Fig. 3). However, the percentage of croplands did not affect the NDVI of croplands, indicating that during the dry season the sugarcane crops in forest-dominated landscapes are as productive as they are in sugarcane-dominated landscapes. In addition, during this season we also observed that the mean EVI of croplands increased with the increase in forest edge density, suggesting that landscapes with more forest edges favor sugarcane production.

Landscape metrics did not significantly affect the spatial variability in PP of croplands and forest at any season (Table 1). During the rainy season, the coefficient of variation (CV) varied from 0.5 to 19.1% in terms of forest NDVI, 1.6 to 23.0% in terms of forest EVI, 1.7 to 30.1% in terms of cropland NDVI, and 2.4 to 39.1% in terms of cropland EVI. During the dry season, a similar amplitude of variation in CV was observed for the four indices: 0.9-32.5%, 1.9-28.3%, 9.1-49.3%, and 5.7-36.7%, respectively. However, no one varied consistently with the percentage of croplands, density of forest edges and number of forest patches in the landscape (Table 1).

Discussion

Our results reveal that landscape composition is more important than spatial configuration in determining the PP of agricultural landscapes embedded in an old frontier of sugarcane plantation, as NDVI and EVI of the entire landscape were negatively influenced by the percentage of croplands in both seasons. Nonetheless, the increase in landscape fragmentation negatively affected the NDVI of the landscape, indicating that splitting the forest into many remnants is not beneficial to landscape production. When we analyzed the NDVI of croplands and forest separately, we also observed that both land uses responded more to landscape composition, even though the EVI of croplands had a positive relationship with density of forest edges during the dry season. Surprisingly, neither landscape composition nor configuration affected the spatial variability of cropland and forest production. These findings not only support the notion that landscape attributes mediate ecosystem functions and services (Nascimento and Laurance 2004, Magnago et al. 2017, Melito et al. 2017), but also point out that we should conserve as many forest remnants as we can in agricultural landscapes because croplands and forests themselves benefit from this environmental-friendly scenario.

Despite the scarcity of landscape-level studies in the Brazilian Atlantic Forest (Melito et al. 2017), there is plenty of evidence demonstrating that agricultural expansion has led to large-scale floristic simplification and biotic homogenization (Lôbo et al. 2011, Tabarelli et al. 2012). This is possibly a consequence of the re-arrangement of tree communities following biomass collapse and other edge-related changes (Tabarelli et al. 2008), in which pioneer, softwood trees dominate the landscapes at the expense of the hardwood, old-growth flora (Oliveira et al. 2008, Tabarelli et al. 2010). In Northeast Brazil, where sugarcane crops have replaced almost all forest cover (Ribeiro et al. 2009), the pioneer plants represent more than 80% of the floristic composition (Tabarelli et al. 2010). The spread of pioneer trees is paralleled by a persistent and substantial impoverishment of the large-tree stand, including the structural collapse of forest emergent layer (Oliveira et al. 2008). The growth rate of fast-growing pioneer trees is greater than late-successional shade-tolerant trees, but their shorter life cycle largely limits long-lasting biomass accumulation (Brienen et al. 2015, Santos et al. 2012). Altogether, the floristic and structural shifts in forest structure may explain why forest production is smaller in cropland-dominated than forest-dominated landscapes.

The reduction in forest production also took place in face of increased landscape fragmentation. Based on other studies in landscapes with a long history of disturbance, we suggest that this response is related to disruptions in ecological interactions, such as pollination and seed dispersal. Most late-successional tropical trees are highly dependent on large vertebrates or specialized insects for seed dispersal and pollination, which in turn influence their demography, define their rates of reproduction and immigration, and ultimately their contribution to forest biomass (Galetti and Dirzo 2013, Lopes et al. 2009, Peres et al. 2016). In this regard, the increase in forest fragmentation may have reduced seed output due to pollination failure and prevent the seeds to arrive at safe sites and establish as new trees (Costa et al. 2012). It has been observed that low diversity in tropical forests reduces carbon storage (Poorter et al. 2015). In our study region, most forest patches have less than 45 ha and are kilometers apart from each other, exacerbating dispersal failures of late-successional species typical of more productive forests. During the dry season, these species reduce their chlorophyll content and photosynthetic activity due to water shortage (Malhi et al. 1998, Phillips et al. 2009), possibly reducing the PP of less fragmented landscapes and hindering the effect of landscape fragmentation on forest PP.

A promising result from our analyses was the increased production of croplands in forest-dominated landscapes, demonstrating a potential synergy between sugarcane production and forest conservation. Although not novel in the literature (Garibaldi et al. 2013, Kennedy et al. 2013, Zhang et al. 2007), this synergy had not been yet described for large-scale sugarcane plantations in the Brazilian Atlantic Forest. The main evidence arose from the cropland NDVI during the rainy season, which increased in more forested landscapes. The underlying mechanism is likely to be the vegetation breeze (Cochrane and Laurance 2008; Pinto et al. 2010), as the more forest in the landscape, the greater the capacity of storing water from the rainfall and ameliorate conditions in adjoining croplands. However, during the dry season, rainfall is not available at sufficient amounts and the sugarcane crops need to be irrigated to maintain high sucrose yields (Abreu et al. 2013, Carr and Knox 2011, Inman-Bamber and Smith 2005). This artificial supply of water into the system possibly replaces the vegetation-breeze service provided by the forest in the preceding rainy season, making sugarcane crops of forest-dominated landscapes as productive during periods of water scarcity as crops in cropland-dominated landscapes (Silva et al. 2007).

However, some croplands keep being positively affected by the forest during the dry season, but uniquely in landscapes with greater forest edge density. This metric of configuration does not vary linearly with the amount of forest in the landscape, being maximum in landscapes with intermediate amount of habitat and minimum at 0% or 100% of forest cover (Fahrig 2003). In landscapes with intermediate forest cover, more irregular forest patches also contribute to increment the density of forest edges. This is the case of most forest patches in our region (Ranta et al. 1998), which are very irregular in shape and somehow may supplement the benefits provided by the irrigation during the dry season (Cochrane and Laurance 2008, Laurance et al. 2007), possibly explaining the positive relationship between cropland EVI and density of forest edges. The mechanism behind this supplementation is uncertain yet, given that the vegetation breeze is unlikely to take place in the dry season, thus further studies are needed to elucidate this phenomenon. Importantly, our finding does not imply that creating forest edges will increase crop production, because under field conditions there is no way to create forest edges without losing forest cover and changing landscape composition (Fahrig 2003).

Finally, we expected that the spatial variability of NDVI and EVI would reduce in landscapes with greater proportions of croplands, since forest loss leads to floristic homogenization (Lôbo et al. 2011) and production of larger crops are leveled at the top anywhere in the landscape (Vieira et al. 2012). However, the coefficient of variation in NDVI and EVI of croplands and forest did not vary with any metric of landscape composition and configuration in both seasons. This suggests that the photosynthetic activity of forests is as spatially variable in cropland-dominated landscapes as it is in more forested landscapes. Regarding sugarcane production, this finding suggests that yields take place irrespective to forest cover and spatial arrangement of land uses in the landscape, maintaining the spatial variability of PP regardless landscape modification.

A previous study in the Brazilian Atlantic Forest shows that although disturbed forests host about half of the tree species they could support, they may be similar or even more divergent in functional and taxonomic terms than conserved forests (Sfair et al. 2016). We believe that this also holds for forest production, which is overall reduced with agricultural expansion over the landscape, but the spatial variation among remaining forest stands is not affected. The great range of values in the CV of cropland NDVI and EVI reflects differences in the stage of crop growth rather than differences in production among different varieties.

Caveats on interpretation and future research

Contrasting patterns have been observed on tropical forest greenness using remote spectral information (Huete et al. 2006, Morton et al. 2014). Although this technology still needs to go through evaluation and refinement (Dong et al. 2017) it is a readily confident tool to have a first insight on ecosystem functioning, especially in the Atlantic Forest of Northeast Brazil where no patterns of PP have been described at the landscape or regional scale. While future research should seek to adjust PP estimations at these spatial scales, our results helps to delineate a possible path that is logistically possible and could reduce uncertainties on remote sensing application for PP estimations. Sugarcane PP using biomass harvest methods are possible to carry out and should consider the spatial distribution of forest patches in order to evaluate the patterns observed in this work and, at the same time, contribute to adjust PP modeling derived from vegetation indices.

Consistent with other studies on landscape structure (Arroyo-Rodríguez et al. 2016, Carrara et al. 2015), our findings indicate that landscape composition is more important than landscape configuration in shaping the PP. Accordingly, we should conserve all forest remnants of the region to ensure the positive effects of forest on croplands. Unfortunately, we were unable to measure crop yield in situ to estimate how much forest is needed to achieve maximum yield at the 1-km^2 scale. However, we have enough information to recommend that sugarcane production in the region (1) avoids forest fragmentation of forest-dominated landscapes, (2) restores forests in cropland-dominated landscapes to increase forest cover, and (3) protects the remaining forest from fire during sugarcane harvesting. These actions should ultimately increase the yield per unit of area while conserving ecosystem functions.

Finally, our results also suggest that landscape-level analyses are needed to accurately assess the impact of agricultural expansion on ecosystem functions and services (Fahrig 2013, Carrara et al. 2015, Cong et al. 2016). Further studies should go deeper and better explore the synergy between sugarcane production and forest conservation, involving scientists, decision-makers and practitioners. Land-owners should also be part of such initiatives and share data on crop productivity. Other commodities, such as oil palm and soy, have increasingly replaced large tracts of tropical forests worldwide, with little concern with biodiversity losses and ecosystem degradation (Haddad et al. 2015, Johnson et al. 2014, Laurance et al. 2014). Whether these new agricultural frontiers will impair similar shifts in natural ecosystems remains to be known, but we encourage the replication of our study in other agricultural frontiers to assess the generality of the landscape patterns described here.

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Supporting Information

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

Appendix S1. Detailed description of landscape sampling in Northeast Brazil.

Table 1. Results of generalized linear models examining the effect of the percentage of croplands, density of forest edges and number of forest patches in the landscape on the mean value of the vegetation indices NDVI and EVI and their coefficient of variation during the rainy and dry seasons of 2011-2016. The estimate of model terms is shown in bold for significant relationships only (see also Figures 2 and 3). Models were based on 60 landscapes.

Response variable	**Whole model X^2**	**Model factors**								
		Percentage of croplands			**Density of forest edges**			**Number of Forest patches**		
		X^2	***P***	**est**	**X^2**	***P***	**est**	**X^2**	***P***	**est**
Rainy season										
Landscape NDVI	54.43	50.25	**<0.05***	**(-0.001)**	0.05	0.81		0	$0.9^{n.s.}$	
Landscape EVI	33.01	31.79	**<0.05***	**(-0.001)**	0.42	0.51		0	$0.97^{n.s.}$	
Forest NDVI	12.02	7.4	**<0.05***	**(-0.001)**	0.02	$0.88^{n.s.}$		4.93	**<0.05***	**(-0.243)**
Forest EVI	1.62	1.58	$0.2^{n.s.}$		0.3	$0.58^{n.s.}$		0.09	$0.76^{n.s.}$	
Cropland NDVI	11.73	8.88	**<0.05***	**(-0.001)**	0.42	$0.51^{n.s.}$		2.36	$0.12^{n.s.}$	
Cropland EVI	1.51	1.09	$0.29^{n.s.}$		0.06	$0.79^{n.s.}$		0.26	$0.6^{n.s.}$	
Forest CV NDVI	2,81	2.66	$0.10^{n.s.}$		0.81	$0.36^{n.s.}$		0.43	$0.51^{n.s.}$	
Forest CV EVI	2.46	1.76	$0.18^{n.s.}$		0.19	$0.65^{n.s.}$		0.57	$0.44^{n.s.}$	
Cropland CV NDVI	0.51	0.09	$0.75^{n.s.}$		0.01	$0.89^{n.s.}$		0.41	$0.52^{n.s.}$	

Cropland CV EVI	0.38	0.09	$0.75^{n.s.}$		0.16	$0.68^{n.s.}$		0.13	$0.71^{n.s.}$
Dry season									
Landscape NDVI	80.65	79.55	**<0.05***	**(-0.003)**	0.25	$0.61^{n.s.}$		1.93	$0.16^{n.s.}$
Landscape EVI	54.77	52.95	**<0.05***	**(-0.002)**	0.12	$0.72^{n. s}$		2.64	$0.1^{n.s.}$
Forest NDVI	16.63	11.74	**<0.05***	**(-0.001)**	2.2	$0.13^{n.s.}$		0.47	$0.48^{n.s.}$
Forest EVI	7.15	2.61	$0.10^{n.s.}$		3.07	$0.07^{n.s.}$		0	$0.02^{n.s.}$
Cropland NDVI	8.59	4.28	$3.5^{n.s.}$		3.5	$0.06^{n.s.}$		0.78	$0.37^{n.s.}$
Cropland EVI	5.75	0.06	$0.88^{n.s.}$		5.37	**<0.05***	**(0.001)**	0.02	$0.87^{n.s.}$
Forest CV NDVI	5.99	1.66	$0.19^{n.s.}$		3.2	$0.07^{n.s.}$		0.01	$0.95^{n.s.}$
Forest CV EVI	5.26	1.15	$0.28^{n.s.}$		2.95	$0.08^{n.s.}$		0.16	$0.68^{n.s.}$
Cropland CV NDVI	1.37	1.31	$0.25^{n.s.}$		0.16	$0.68^{n.s.}$		0	$0.97^{n.s.}$
Cropland CV EVI	0.75	0.47	$0.49^{n.s.}$		0.57	$0.44^{n.s.}$		0.1	$0.74^{n.s.}$

Figure legends

Fig. 1 Map of the study region in the state of Paraíba, Northeast Brazil, showing the remaining Atlantic forest patches (dark grey), the agricultural crops (light grey; most sugarcane), and the 60 landscapes of 100 ha (black squares) sampled during the rainy seasons of 2011-2016. In detail, a forest-dominated landscape with only 18.11% of its area covered by sugarcane plantations and a cropland-dominated landscape with 92.09% of sugarcane cover. Grey lines represent municipality limits. Other 60 landscapes (not shown) were sampled during the dry seasons of the same period.

Fig. 2 Relationships between the percentage of croplands, density of forest edges, number of forest patches and the mean value of the vegetation indices (NDVI and EVI) of croplands, forests and the entire landscape during the rainy seasons of 2011-2016. Statistical significance of the relationships is shown in Table 1.

Fig. 3 Relationships between the percentage of croplands, density of forest edges, number of forest patches and the mean value of the vegetation indices (NDVI and EVI) of croplands, forests and the entire landscape during the dry seasons of 2011-2016. Statistical significance of the relationships is shown in Table 1

Fig. 1

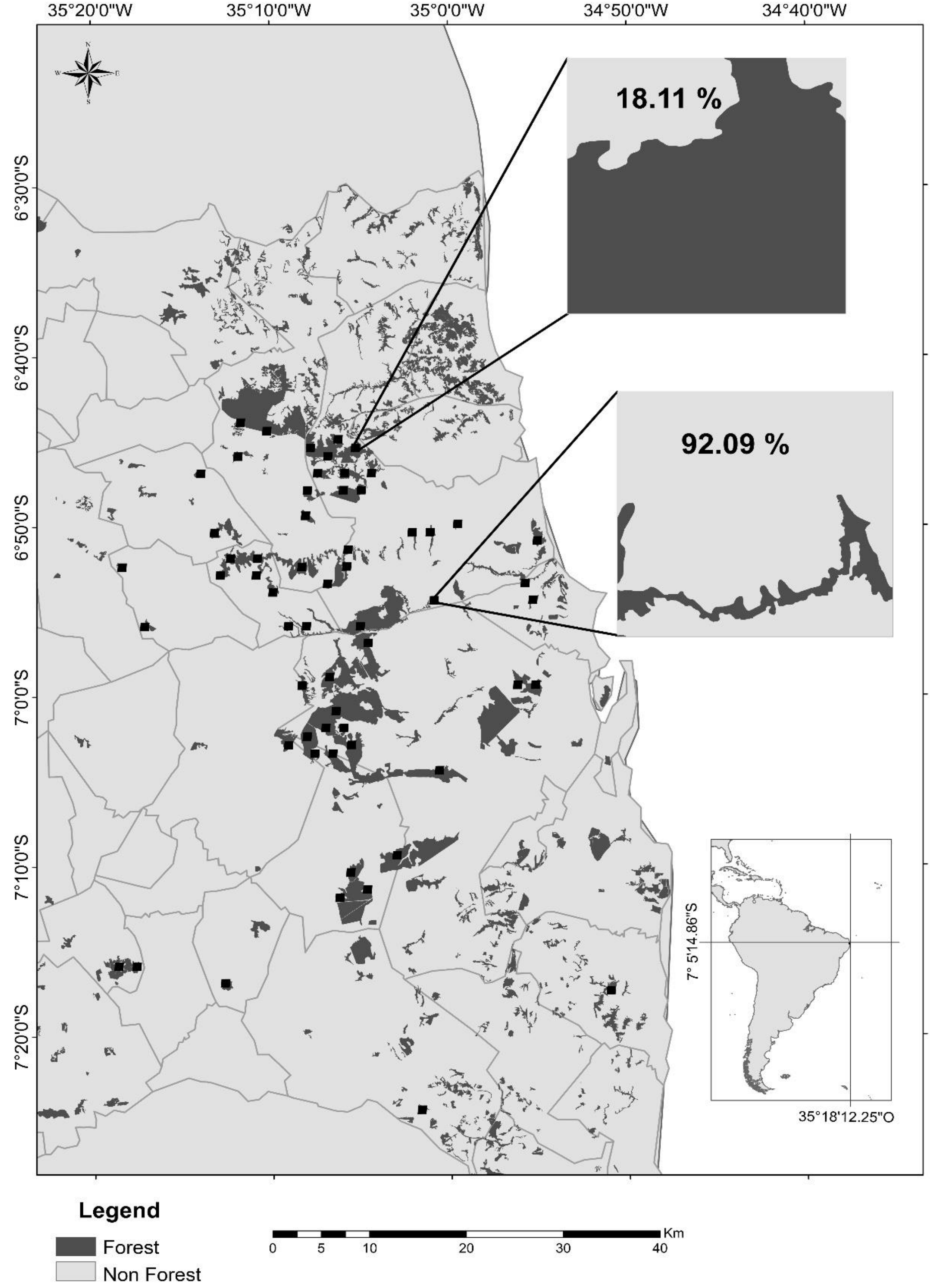

35°20'0"W
35°10'0"W
35°0'0"W
34°50'0"W
34°40'0"W
6°30'0"S
6°40'0"S
6°50'0"S
7°0'0"S
7°10'0"S
7°20'0"S
N
S
E
W
18.11 %
92.09 %
7° 5'14.86"S
35°18'12.25"O
Legend
Forest
Non Forest
0
5
10
20
30
40
Km

Fig.1

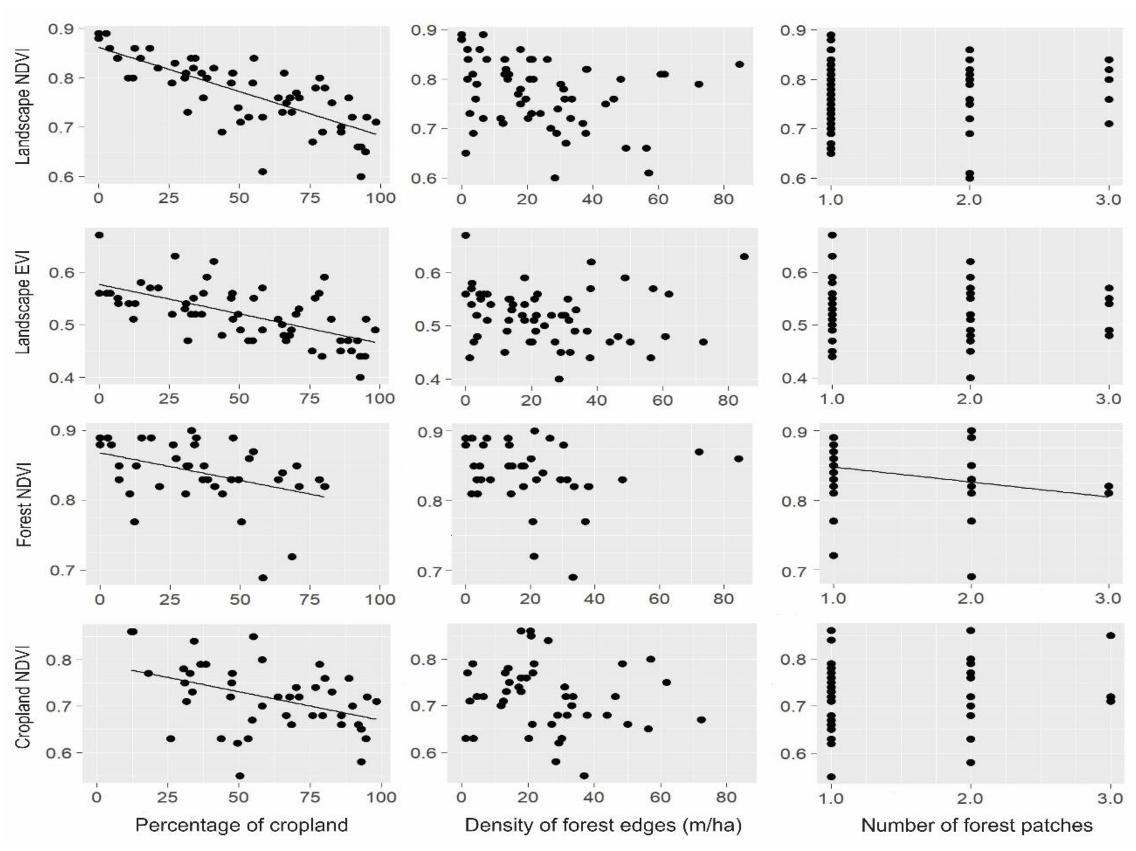
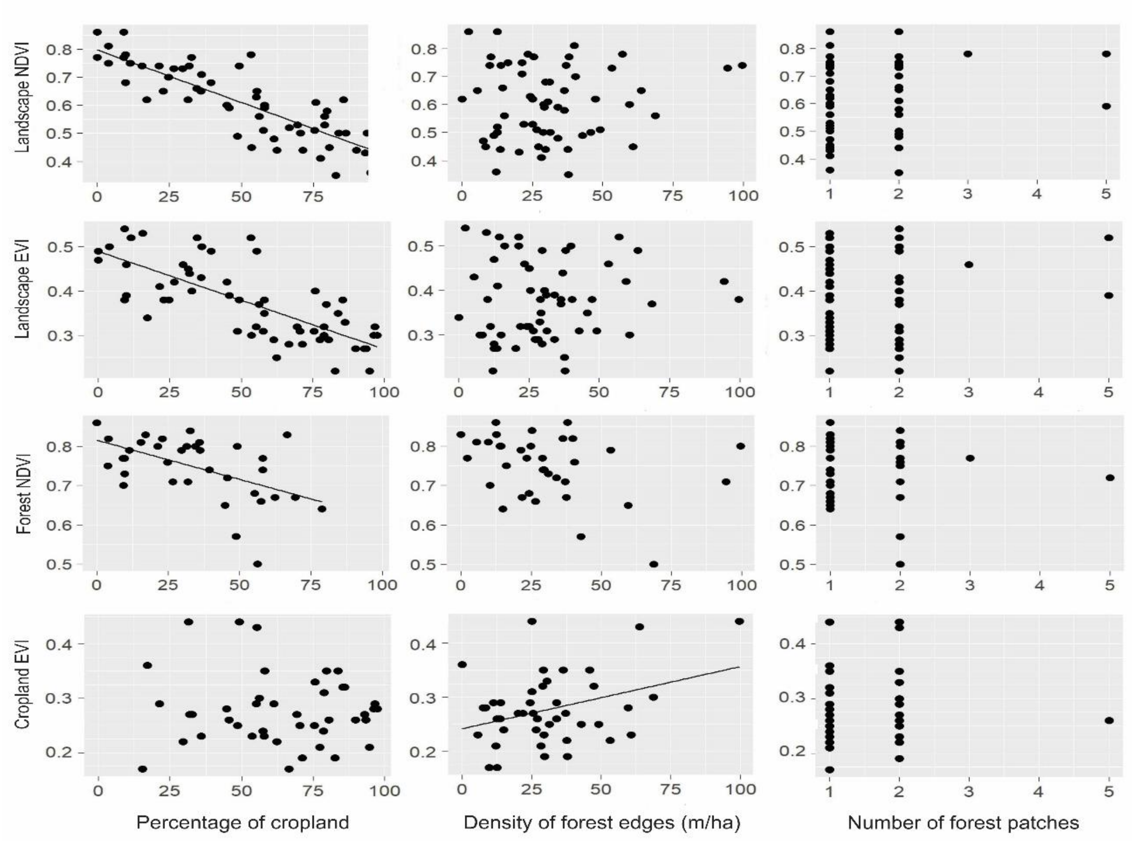

Landscape NDVI
Landscape EVI
Forest NDVI
Cropland NDVI
0.9
0.8
0.7
0.6
0.5
0.4
0
25
50
75
100
20
40
60
80
1.0
2.0
3.0
Percentage of cropland
Density of forest edges (m/ha)
Number of forest patches

Fig.2


Landscape NDVI
0.8
0.7
0.6
0.5
0.4
Landscape EVI
0.5
0.4
0.3
Forest NDVI
0.8
0.7
0.6
0.5
Cropland EVI
0.4
0.3
0.2
0
25
50
75
100
1
2
3
4
5
Percentage of cropland
Density of forest edges (m/ha)
Number of forest patches

4 CONCLUSÕES

A literatura que aborda o efeito da expansão agrícola sobre os serviços ecossistêmicos tem crescido nos últimos tempos, no entanto, a maioria dos estudos trata sobre os serviços de polinização, controle biológico e biodiversidade. Em geral, esses estudos são realizados em pequenas escalas, sendo muitas vezes difícil mensurar as perspectivas futuras da provisão de serviços em paisagens com histórico de expansão agrícola e propor ações de manejo sustentável. Nesta dissertação, buscamos contribuir para o conhecimento dos efeitos da reestruturação da paisagem sobre a PP utilizando como modelo uma antiga fronteira agrícola da Floresta Atlântica Nordestina. Com base em medidas de composição e configuração da paisagem, nós encontramos fortes evidências de que a quantidade de habitat é o fator mais limitante para a PP. Em primeiro lugar, reforçamos a ideia de que todos e quaisquer fragmentos inseridos nas paisagens são indispensáveis para manter a PP da paisagem como um todo. Em segundo lugar, destacamos que alterações na composição da paisagem resultam em efeitos negativos para tanto para a PP florestal como das monoculturas, o que torna evidente que para efeitos sinérgicos entre produção e conservação as decisões no manejo da agricultura de larga escala devem priorizar práticas agrícolas que conciliem a conservação da floresta e seu uso sustentável. Por fim, alertamos que em antigas fronteiras agrícolas a sinergia entre a produção agrícola e a conservação florestal é uma estratégia fundamental para compensar os passivos da agricultura de larga escala da cana-de-açúcar. Os tomadores de decisão e os profissionais devem estar cientes dos benefícios fornecidos às suas terras agrícolas. Nós incentivamos fortemente outros estudos futuros em outras fronteiras agrícolas para avaliar a generalidade dos padrões de paisagem descritos aqui.

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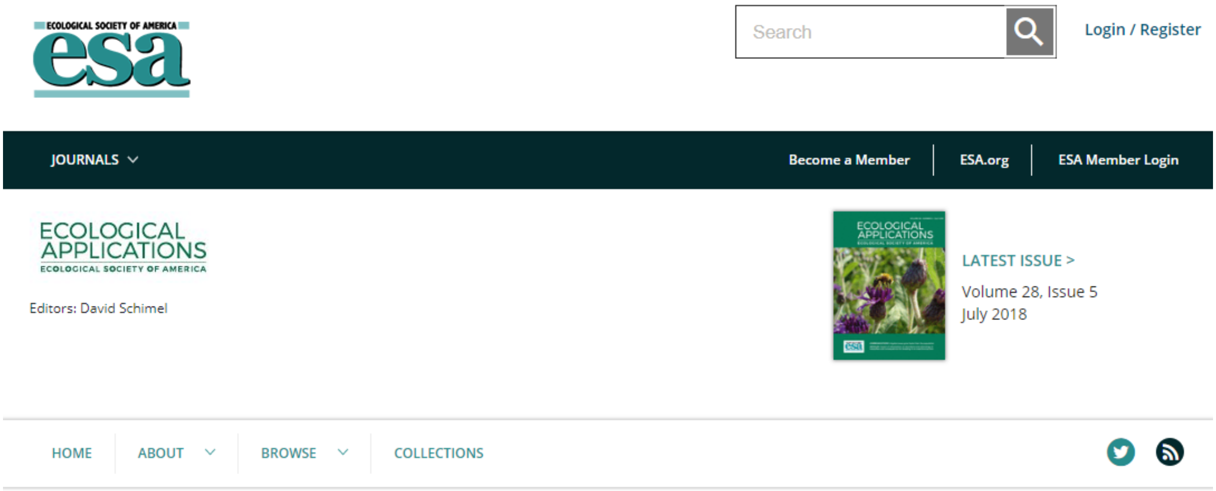
ANEXO A - NORMAS DA REVISTA ECOLOGICAL APPLICATIONS



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Types of Contributions

Length limits with respect to manuscripts refer to all pages, including title page, text, literature cited, tables and figures. Each figure will count as one page.

Final decisions on article types are at the discretion of ESA. Papers submitted/accepted as Communications that require more than 8.0 PDF pages in the typeset version will instead be published as Articles.

Articles. Articles describing significant original research comprise the core of the journal. Articles generally should not exceed 60 manuscript pages (double-spaced, 12-point Times New Roman font, including everything from the title page through the last figure). Abstracts should be no longer than 350 words. Articles exceeding 60 manuscript pages may be considered for transfer to *EcologicalMonographs* at the discretion of the Editor-in-Chief.

Communications. Communications are shorter papers (up to 8 journal pages) of immediate impact in fast-moving scientific debate or urgent practical application. These papers will be fast-tracked through the editorial, review, and production processes so as to make possible a relatively rapid publication.

These papers must be accompanied by a cover letter clearly indicating that the paper is being submitted for consideration for the Communications section and explaining why the paper warrants this special treatment. Papers deemed unsuitable for Communications may be considered for publication as regular articles.

As added incentives for authors to cast their submissions in the concise Communications format:

* All Communications will be freely available for no extra charge (i.e., one need not be a subscriber or have institutional access in order to view full text online).

* Authors are encouraged to utilize color figures. This journal is published online only beginning in 2018 and color figures incur no additional charges.

By making the Communications more accessible and visually appealing, we hope to increase our readership and cement *Ecological Applications'* reputation as the place where authors publish the most important and exciting research findings in our science.

In order to assure rapid publication, papers must conform to a strict page limit and format. Communication submissions may contain up to 20 manuscript pages (double-spaced, 12-point Times New Roman font, including everything from the title page through the last figure). The abstract can have a maximum of 200 words.

Invited Features. Invited Features are intended to address various aspects of a theme that is likely to be of broad interest to applied ecologists. Ideally, a feature should inform a broad audience about an unfamiliar topic or an area in which there has been considerable recent progress, or it should cause the audience to re-examine an issue that is not as settled as most have presumed. Proposals for Invited features should be addressed to the Editor-in-Chief. See Invited Features for additional advice and instructions that explain how to propose and prepare Invited Features.

Forums. *Ecological Applications* occasionally publishes Forums. A Forum can take a number of forms but always includes a series of commentaries solicited from a number of experts. These commentaries represent personal responses to a paper (or papers) considered to be of very broad interest and significance within the field of applied ecology.

Possible variations include but are not limited to:

- An original (published for the first time) research, review, or discussion paper, followed by a number of commentaries,
- A paper republished (with permission) from another journal, followed by commentaries,
- An ESA report, followed by commentaries,
- Several original papers on a common topic, followed by commentaries,
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The number of commentaries can depend on the nature of the Forum, but normally would not exceed six to eight. The commentaries are reviewed by the Editor-in-Chief (EIC).

An individual interested in proposing a Forum should contact the EIC to discuss the idea. If the EIC believes the idea should be pursued, a short proposal must be submitted for review by one or more member of the Board of Editors.

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Letters may be assigned to a Subject-matter Editor for evaluation. As with regular articles, typically two reviews will be sought. Every effort will be made to expedite the review process.

At the time a Letter is assigned to a Subject-matter Editor, a copy will be sent by the Publications Office to the corresponding author of the original paper. The authors of the paper being commented upon will be offered the opportunity to submit a signed review to the Subject-matter Editor, who will view such a review as supplemental to and not a replacement for the regular reviews. The signed supplemental review will be forwarded to the author(s) of the Letter, together with the two regular reviews and the Subject-matter Editor's decision letter.

When the Letter is first sent to the authors of the manuscript being commented upon, they will be informed by receipt of this policy document that at such time as the Letter is accepted for publication, they will be sent a copy of the final version and will be invited by the Subject-matter Editor to prepare a response. The response must be completed within four weeks if simultaneous publication is to be guaranteed. Authors should submit the response online and it will be sent to the same Subject-matter Editor who handled the Comment or Letter. The response will typically be reviewed by one reviewer, but the Subject-matter Editor may choose to accept the response without review in cases where the response is short and simple.

We will attempt to publish all letters on a particular paper plus any response together. We generally neither invite nor publish responses to responses.

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Although it is the Editor's responsibility to decide whether specific duplications are useful or wasteful, these decisions are generally based on information supplied by the authors. ESA journals have adopted a policy to facilitate this process. At the time of submission, authors must provide information describing the extent to which data or text in the manuscript have been used in other papers that are published, in press, submitted, or soon to be submitted elsewhere. In cases of overlap with other publications or submissions, authors should include copies of said papers along with the current submission.

Sometimes it is difficult to assess whether a work has truly been published previously. If a previous work was published in a journal or book that is already available in libraries, or is available for libraries to purchase, we expect no more than one-third overlap between the previous publication and the submission to ESA. Reference should be made to any closely related previous publication, especially if a table or figure is reproduced. If any data in a manuscript have been included in other published or unpublished manuscripts, the legend of each table or illustration reporting such data must cite those manuscripts. When in doubt, an author should supply copies of the previous publication to be sent to the Subject matter Editor and reviewers, who would then be asked to consider this matter. Authors are responsible for obtaining permission to reproduce previously published material.

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* Assemble the parts of the manuscript in this order: title page, abstract, key words, text, acknowledgments, literature cited, tables (one table per page), figure legends (on separate page preceding the first figure), figures (one figure per page; label each figure, i.e., Figure 1, Figure 2, etc.), and lastly any Appendices. Please note that if your manuscript is accepted, the appendices would only be published online, but for review purposes, appendices need to be merged into one file together with the manuscript, for the convenience of reviewers and editors. Other supporting information such as raw data or code will need to be in a separate file.

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* All pages of text must have line numbers as well.

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Tables: Tables in doc, xls, or csv format (or tables may be included in the manuscript file).

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1. The author submits a manuscript via ScholarOne.
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5. The Reviewers review the manuscript and make comments and a recommendation (generally expected within 3 weeks).
6. Once the Reviewers have submitted their reports, the SME reviews the comments and makes a decision: either (a) Accept, (b) Revise, (c) Reject with resubmission invitation, (d) Reject or (e) Reject with transfer offer to another ESA journal.
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8. The decision letter is proofread, finalized, and forwarded to the corresponding author.
9. If the decision is "revise", the authors revise the manuscript and resubmit via ScholarOne as soon as possible, preferably within six weeks. If the revised manuscript has not been submitted within three months of the decision, any subsequent revision must be submitted as a new manuscript and will be given a new manuscript number. Revisions will be sent back to the original SME so long as he/she is still available.
10. The SME reviews the revised manuscript and either requests further reviews (repeating steps 4–8 above) or makes a decision (steps 7 and 8 above).
11. The steps may be repeated as many times as necessary until a final decision is rendered. In the event the paper is accepted for publication, authors are provided a final opportunity to prepare the files related to the article for page production and publication.

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Running Head.—A running head of no longer than 40 letters and spaces should be provided at the top of the title page.

Title.—Titles should be concise, informative, tell what the paper is about, and what it found. It should contain key words necessary for digital search and retrieval methods. Avoid vague declarations (e.g., "effects of ..."); strive for information content (e.g., “fungi kill tardigrades"). The maximum length is 120 characters, including spaces. Do not include the authority for taxonomic names in the title or in the abstract. Titles may not include numerical series designations. The first letter of the first word in the title is capitalized. All other words, except for proper nouns, are lowercase.

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Abstract

The abstract should explain to the general reader why the research was done and why the results should be viewed as important. It should provide a brief summary of the research, including the purpose, methods, results, and major conclusions. Do not include literature citations in the Abstract. Avoid long lists of common methods or discursive explanations of what you set out to accomplish.

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If appropriate, organize your article in sections labeled Introduction, Methods, Results, and Discussion. You may need to add a section for Conclusions. Brief articles usually do not require a label for the Introduction. If the nature of your research requires a different organization, specify the level of each section heading (1st-order head, 2nd-order head, etc.) by using unique type styles (italics, boldface) consistently for each heading in the hierarchy.

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Tables should supplement, not duplicate, the text. They should be numbered in the order of their citation in the text. Start each table on a separate page. Provide a short descriptive title at the top of each table; rather than simply repeating the labels on columns and rows of the table, the title should reveal the point of grouping certain data in the table. Statistical and other details should be provided as footnotes rather than appearing in the title. Never repeat the same material in figures and tables; when either is equally clear, a figure is preferable. Do not include any class of information in tables that is not discussed in the text of the manuscript.

At the submission and review stages, embedded image files are acceptable for tables. Final versions of accepted manuscripts must have “true” tables in an editable format, created by using the “Insert Table” function, rather than using tabs or spaces. In Microsoft Word you should select “Insert”, “Table”, then specify the number of rows and columns and fill in the individual cells.

Tables cannot contain colors, shading, or graphics. If such enhancements are needed, the information should be formatted as a figure.

Figures

Figures should be submitted in the following formats for accepted manuscripts:

- TIFF
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- PPT/PPTX
- DOC/DOCX
- AI

Files should adhere to the following resolution requirements:

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- 600 dpi for combination images (photographs that also contain line art, text, or thin lines)

General Appearance:

- Do not embed figures in the body of your article document. Number each figure with Arabic numerals in order of their citation in text. Label multipart figures with consecutive letters of the alphabet, using a lowercase letter (a, b, c, etc.). Place this letter in the upper left corner of the figure, outside the figure itself (not in the figure).
- Because this journal is published online only beginning in 2018, color figures incur no additional charge and are encouraged for your submission. We suggest using a colorblind-friendly palette.
- Grayscale patterns do not reproduce as well as solid colors or lines. Avoid small dotted lines, thin lines, multiple levels of gray shading, and stippling. For bar graphs, use black, white, striped, hatched, or colored designs, but only if they are sufficiently widely spaced to appear distinct from one another.
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figure itself. Symbol keys and scale bars should appear on the figures, not in the figure legends. Make figures as simple as possible; avoid gridlines and boxes.
- Maps generally should include longitude and latitude, an indication of compass direction, and a thin outer line as a border. Make lines on maps bold and distinct while eliminating information not pertinent to the subject.

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- Figure sizes should be no more than 6 inches wide and 7 inches high. When possible, submit figures in the size you wish to have them appear in the journal. Most illustrations, except some maps and very wide graphs, should be 1-column width (3 inches) at a resolution of 600 dpi.
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Image quality will be drastically reduced, possibly impacting readability, if you do not supply your images in the preferred formats and resolutions.

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- Are all figures included in your submission as separate files or in an inclusive PDF/Word document/LaTeX suite?
- Do all figures have an accompanying legend that describes the content and explains any abbreviations or symbols?
- Are all figures cited in the main text of your article in numeric order
- Are all words or symbols appearing in your figures large enough for easy reading?
- Is each individual figure file less than 10 MB?

Checklist

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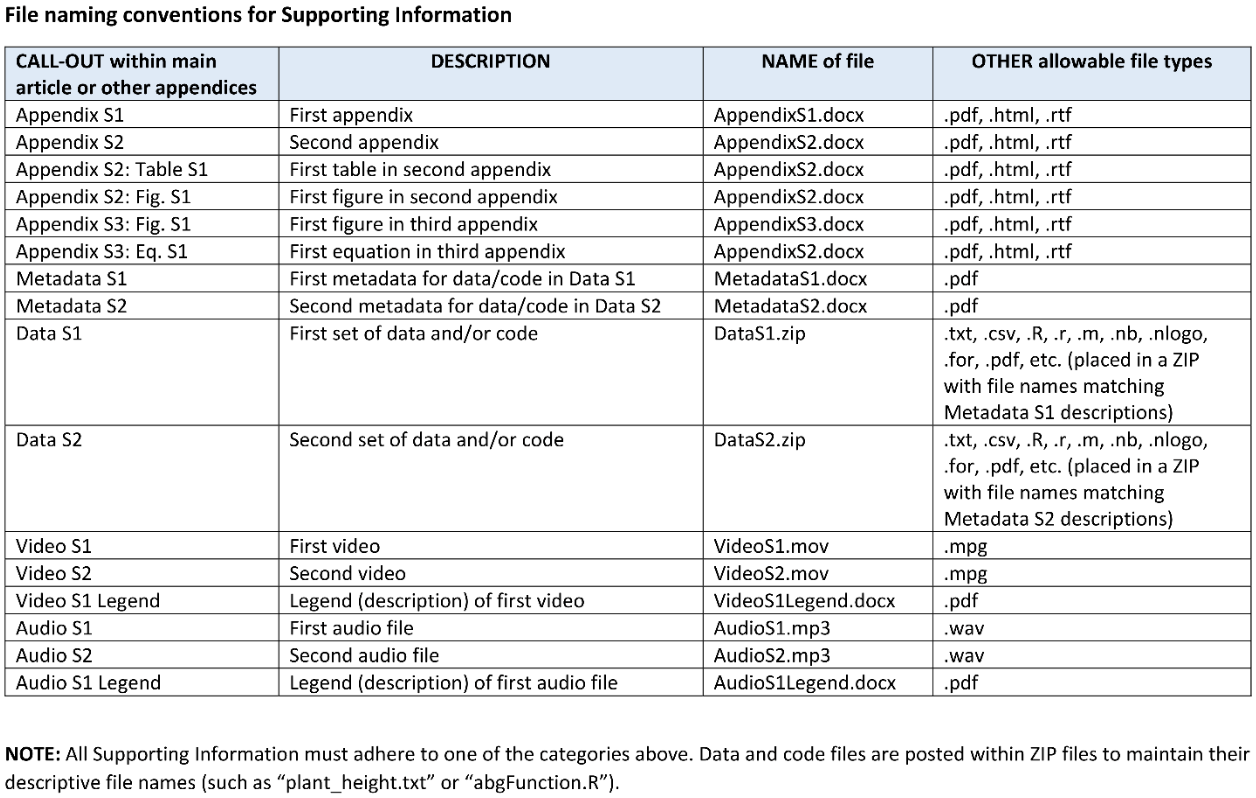

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The following standard treatments are recommended.

- *Phyla* -- Margulis, L & K.V. Schwartz. 1998. Five kingdoms: an illustrated guide to the phyla of life on Earth. Third edition. W.H. Freeman and Co., New York. 520 pp.
- *Birds* -- American Ornithologists’ Union (AOU). 1998. Check-list of North American birds, Seventh edition. American Ornithologists’ Union, Washington, D.C. 829 pp. (For North America). Monroe, B.L. Jr. & C.G. Sibley. 1993. A world checklist of birds. Yale University Press, New Haven, CT. 393 p. (For remainder of the world.)
- *Butterflies* -- Cassie, B. et al. 1995. North American Butterfly Association (NABA) checklist and English names of North American butterflies. Morristown, NJ. 43 pp.
- *Dragonflies* -- Paulson, D.R. & S.W. Dunkle. 1999. A checklist of North American Odonata including English names, etymology, type locality, and distribution. Slater Museum of Natural History Occasional Paper 56, Univ. Puget Sound, Tacoma, WA.

- *Fishes* -- Eschmeyer, W.N., C.J. Ferraris & M.D. Hoang. 1998. Catalog of Fishes. California Academy of Sciences. Robins, C.R. et al. 1991. Common and scientific names of fishes from the United States and Canada. Fifth Edition. American Fisheries Society Special Publication No. 20. 183 pp.
- *Lichens* -- Esslinger, T. L. & R. S. Egan. 1995. A sixth checklist of the lichen-forming, lichenicolus, and allied fungi of the continental United States and Canada. The Bryologist 98: 467-549.
- *Mammals* -- Wilson, D. E., and D. M. Reeder (editors). 1993. Mammal species of the world: a taxonomic and geographic reference. Smithsonian Institution Press, 1206 pp.
- *Mollusca* -- Turgeon, D.D. et al. 1998. Common and scientific names of aquatic invertebrates from the United States and Canada: mollusks. Second edition. American Fisheries Society Special Publication. No. 26. 526 pp.
- *Mosses* -- Anderson, L.E., H.A. Crum & W.R. Buck. 1990. List of the mosses of North America north of Mexico. Bryologist 93: 448-499
- *Reptiles and Amphibians* -- Collins, J.T. 1997. Standard common and current scientific names for North American Amphibians and Reptiles, Fourth Edition, Society for the Study of Amphibians & Reptiles. Herp. Circular No. 25. 40 pp. Frost, D.R. 1985. Amphibian species of the world: a taxonomic and geographic reference. Allen Press, Inc. Lawrence, KS. 732 pp. Petranka, J.W. 1998. Salamanders of the United Staes and Canada. Smithsonian Institution Press, Washington, D.C. 587 pp.
- *Vascular Plants* -- Flora of North America Editorial Committee. 1993-. Flora of North America North of Mexico. Oxford University Press, New York, New York, (where completed). Kartesz, J.T. 1994. A synonymized checklist of the vascular flora of the United States, Canada, and Greenland. Second Edition. Timber Press, Portland, Oregon. 622 p.

Equations, symbols, and abbreviations

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