Universidade Federal de Goiás

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Programa de Pós-Graduação em Ecologia e Evolução

Amazon forest dieback:

assessing vulnerabilities and threats

Caroline Corrêa Nóbrega

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RESUMO

Nos últimos anos, tem surgido cada vez mais evidências que a Amazônia pode estar ameaçada em decorrências das mudanças regionais no clima e no uso do solo. Por causa de um sistema de retroalimentação positivo, em que impactos intensificam outros impactos, alguns modelos projetam para o final do século uma substituição da floresta amazônica por formações tipo savana ou semiáridas (*forest dieback)*. Diversas evidências têm indicado uma alta vulnerabilidade da Amazônia às mudanças climáticas globais e aos impactos ambientais locais desencadeados por atividades humanas (e.g. conversão de áreas para agropecuária, construção de estradas e queimadas). Entretanto, são ainda profundamente desconhecidos os mecanismos e padrões gerais de como esses impactos afetam a floresta. Nesta tese, desenvolvi trabalhos que têm como objetivo contribuir para as discussões do tema. Em cada capítulo, abordarei uma ameaça que, de alguma forma, está contribuindo para a degradação da Amazônia. Cada uma das três ameaças discutidas nos próximos capítulos são, frequentemente, apontadas como importantes propulsores de *forest dieback*. No primeiro capítulo avaliei o impacto de incêndios florestais em diferentes níveis de diversidade de árvores de uma área florestal próxima à transição Amazônia-Cerrado. Nossos resultados sugerem que as comunidades de árvores das áreas queimadas estão perdendo mais diversidade filogenética e funcional por unidade de espécie do que nas áreas não queimadas. Nossos resultados indicam a existência de seleção de espécies com base em características filogenéticas e funcionais, que representam uma importante força de mudança e de empobrecimento (funcional e filogeneticamente) dessas comunidades. No segundo capítulo, usando imagens de alta resolução (LiDAR e hiperespectral), eu avaliei o impacto de um período de seca em áreas florestais próximas a Madre de Dios, Peru. Nossos resultados dão suporte a hipótese de que mudanças no clima regional modificam a estrutura e função da floresta. No terceiro

capítulo, eu avaliarei de que forma a construção de estradas na Amazônia brasileira tem contribuído para o desmatamento que incide em um importante grupo de áreas protegidas da Amazônia: as Terras Indígenas. Com base na análise dos impactos observados, nós propomos o estabelecimento de zonas de amortecimento (*buffers)* em torno dessas reservas para reduzir os impactos negativos da construção de estradas planejadas para serem construídas.

ABSTRACT

In recent years, it is increasing evidences about Amazon vunerability due to land use and climate changes. Because of a positive feedback system, in which impacts intensify other impacts, some models project to the end of the century a replacement of the Amazon forest by savanna formations or semi-arid (forest dieback). Several evidence has indicated a high vulnerability of the Amazon to global climate change and local environmental impacts triggered by human activities (eg conversion to agricultural areas, construction of roads and burns). However, they are still deeply unknown the general mechanisms and standards about how these impacts affect the forest. In this thesis, I developed works that aim contribute to discussions of the subject. In each chapter, I will consider a threat that is contributing to the degradation of the Amazon. Each of the three threats discussed in the following chapters are often cited as important drivers of forest dieback. In the first chapter I evaluated the impact of forest fires at different levels of diversity of trees in a forest area next to the Amazon-Cerrado transition. Our results suggest that communities of trees in burned areas are losing more phylogenetic and functional diversity per unit of species than in unburned areas. Our results indicate the existence of selection of species based on phylogenetic and functional characteristics, representing a major force of change and impoverishment (functional and phylogenetically) of these communities. In the second chapter, using high resolution images (LiDAR and hyperspectral), we evaluated the impact of a intense drought in forest areas near Madre de Dios, Peru. Thus, our results support the idea that changes in regional climate may change the structure and function of the forest. In the third chapter, I evaluated how the construction of roads in the Brazilian Amazon has contributed to deforestation in an important group of protected areas of the Amazon: the Indigenous Lands. Based on the analysis of observed impacts, we propose the establishment of buffer zones (buffers) around

these reserves to reduce the negative impacts of road construction planned to be built.

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Agradeço à Universidade de Stanford e ao *Carnegie Institution of Washington* que me receberam durante meu estágio no exterior. Agradeço ao professor Dr. Greg Asner que abriu as portas de seu laboratório e possibilitou aprendizados e experiências que não seriam possíveis no Brasil. Também agradeço à sua equipe, Robin, Claire, Dave, Chris, e todos os outros, que me acolheram e ajudaram em momentos decisivos.

Paulo De Marco Jr., a você serei eternamente grata por ser meu professor, mestre e amigo. Tenho muito orgulho de dizer que meu orientador, também é o professor que mais admiro e me espelho. Sei que seus exemplos, piadinhas bobas e histórias me acompanharão por toda minha vida e carreira (ah, o pobre do pardalzinho... rs). A você devo meus sonhos de contribuir para um mundo melhor, mesmo depois de tanto tempo, você me ajudou a mantê-los sempre vivos. Obrigada por encorajar meus voos e amparar minhas quedas.

Ao meu co-orientador, Paulo Brando, sou muito grata pela orientação, paciência e todas as portas que abriu em minha carreira. A você devo grande parte das melhores experiências que tive durante meu doutorado. Obrigada por todo aprendizado e novas experiências! Sem você eu nunca teria tido a oportunidade de fazer meu estágio em Stanford e nem teria colocado fogo na Amazônia!

Agradeço ao IPAM (Instituto de Pesquisa Ambiental da Amazônia) e todos os seus colaboradores por uma das mais gratificantes experiências que tive durante minha carreira profissional: a oportunidade de ver meus trabalhos serem usados de maneira prática! Agradeço de maneira especial aos meus parceiros de caminhada – Ane Alencar, Paulo Moutinho, Fernanda Bortolloto, Martha Fellows, Julia Shimbo e tantos outros - por permitirem que eu pudesse desenvolver meu trabalho como pesquisadora e o doutorado ao mesmo tempo. Também sou imensamente grata a toda equipe do IPAM de Canarana que me acolheu durante minhas viagens à fazenda Tanguro e executou a coleta de muitos dos dados usados nessa Tese. Wanderley Rocha, Divino Silvério, Claudinei Oliveira-Santos e toda a equipe de campo da Tanguro, não tenho palavras para expressar toda minha gratidão.

Aos meus amigos, agradeço aos que fizeram parte de minha vida acadêmica e que de alguma forma contribuíram para o desenvolvimento deste trabalho. Em especial agradeço aos amigos do TheMetaLand que muito me ensinaram ao longo dos últimos quatro anos. Não citarei nomes, pois não assumirei o risco de ser injusta com alguém. Também agradeço aos amigos que não estiveram diretamente ligados ao meu trabalho, mas que foram fundamentais para manter a minha sanidade mental nos momentos mais difíceis. Em especial agradeço às amigas Flavia Lima e Letícia Alves.

Agradeço à minha família paulistana por tantas alegrias, carinho e apoio. Muito obrigada por me apoiarem mesmo quando deixei meu marido no Brasil para correr atrás dos meus sonhos! Aos meus pais, irmãos e sobrinhos do coração, com vocês minha vida é mais rica em experiências e mais alegre. Vocês continuam sendo o melhor presente que o Du já me deu...

Agradeço à minha família por toda compreensão e paciência em minhas ausências devido ao doutorado. Mas, principalmente, agradeço aos meus pais, Antônio e Daguimar. Palavras são insuficientes para expressar todo amor, admiração, orgulho e respeito que tenho por vocês. Obrigada por estarem sempre ao meu lado me apoiando e encorajando. Em minha vida acadêmica tive a oportunidade de conhecer pessoas extremamente inteligentes, mas vocês são meu maior exemplo de sabedoria. O artigo de divulgação científica desta tese, eu fiz para vocês.

Durante um doutorado, pressupõe-se que devemos executar atividades e tomar decisões muito inteligentes, mas se me perguntarem qual foi a decisão mais inteligente e importante que fiz durante meu doutorado, a resposta é uma só: eu me casei! Muitas vezes me perguntaram como eu conseguia fazer doutorado e ao mesmo tempo trabalhar e ser casada. Meu amor, sei que você detestava essas perguntas (eu apenas achava graça, rs). Agora que estou terminando meu doutorado, posso dizer que não terminei meu doutorado apesar de você; sem sombra de dúvidas, terminei meu doutorado por causa de você. Você me fez tranquila quando fiquei nervosa, foi meu porto seguro quando me senti insegura, se alegrou com minhas conquistas, me encorajou a ir mais longe do que eu achava possível, e quando tudo dava errado e eu ficava triste, você apenas esteve ao meu lado. Por suas palavras e seus silêncios, serei eternamente grata. Ao meu companheiro de todas as horas eu dedico essa tese.

SUMÁRIO

MEMORIAL

“Nenhum homem realmente produtivo pensa como se estivesse escrevendo uma dissertação.” Albert Einstein

Se as atividades de um pesquisador não se resumem à escrita e publicação de artigos científicos, também a formação de um doutor não deveria estar centrada única e exclusivamente apenas nessa atividade. Ao longo do meu doutorado, além de desenvolver esta tese, também desenvolvi outras atividades que enriqueceram muito a minha formação. A maior parte dessas atividades contribuíram direta ou indiretamente para o desenvolvimento do produto final de meu doutorado, a presente tese. Entretanto, mesmo as atividades que não estiveram de alguma forma ligadas a meu doutorado também foram fundamentais para meu amadurecimento e desenvolvimento como pesquisadora. Se a frase acima de Einstein está correta, fico feliz em afirmar que meu doutorado foi muito maior que minha tese.

O desenvolvimento desta tese possibilitou que eu fizesse diversas viagens à Fazenda Tanguro, em Canarana (Mato Grosso) onde o IPAM realiza diversos experimentos de campo. Pude conviver em estado de “semi-confinamento” com diversos pesquisadores, nacionais e estrangeiros, e com uma experiente equipe de campo. A fazenda está no ecótono entre Cerrado e Amazônia e possui grandes áreas preservadas entre grandes extensões produtivas. Dentre as atividades que pude participar na fazenda Tanguro estão a visita à torre que mede trocas de carbono entre a floresta e a atmosfera (Figura 1) e a participação nas queimadas controladas no experimento de fogo (Figuras 2 e 3). Convêm ressaltar que a própria viagem para Canarana, na qual são necessários enfrentar 12 horas de ônibus e mais 2 horas de carro, são vivências importantes que ajudam a compreender e valorizar o trabalho de campo.

Figura 1: Torre que mede trocas de carbono da floresta com a atmosfera instalada na Fazenda Tanguro.

na área.

Figura 2: Área recém queimada pela equipe do IPAM na Fazenda Tanguro.

Figura 3: Experimento de fogo desenvolvido na Fazenda Tanguro. Todos os anos áreas de florestas são queimadas de acordo com diferentes tratamentos e o impacto dessas queimadas é monitorado. Esses experimentos têm gerado importantes publicações na área.

Além de contribuir para a publicação de outros artigos, tenho grande interesse em contribuir com a tomada de decisão de políticas públicas. Nesse campo, tive a oportunidade de co-coordenar o projeto de Vulnerabilidades, realizado por uma parceria entre o laboratório do Dr. Paulo De Marco Jr., e o Instituto Chico Mendes de Conservação da Biodiversidade (ICMBio). Nesse projeto nós avaliamos a vulnerabilidade de todas as espécies ameaçadas de extinção da fauna brasileira.

Tenho participado de um projeto financiando pela NSF (*National Science Foundation*), fruto de uma parceria entre meu co-orientador Dr. Paulo Brando e o Dr. Greg Asner, da Carnegie Institution for Science, localizada dentro da Universidade de Stanford. Frequentei o laboratório do Dr. Asner durante um mês em 2013, pouco mais de 2 meses em 2014 e durante o doutorado sanduíche por 4 meses em 2015. Durante essas temporadas no laboratório do Dr. Asner pude conviver com diversos pesquisadores provenientes de diferentes partes do mundo (África do Sul, México, Espanha, Estados Unidos), auxiliar em atividades de campo, participar de seminários e de um mini-curso promovido pelo Dr. Asner sobre escrita de artigos científicos, além de participar das atividades cotidianas do laboratório (Figuras 4 a 8).

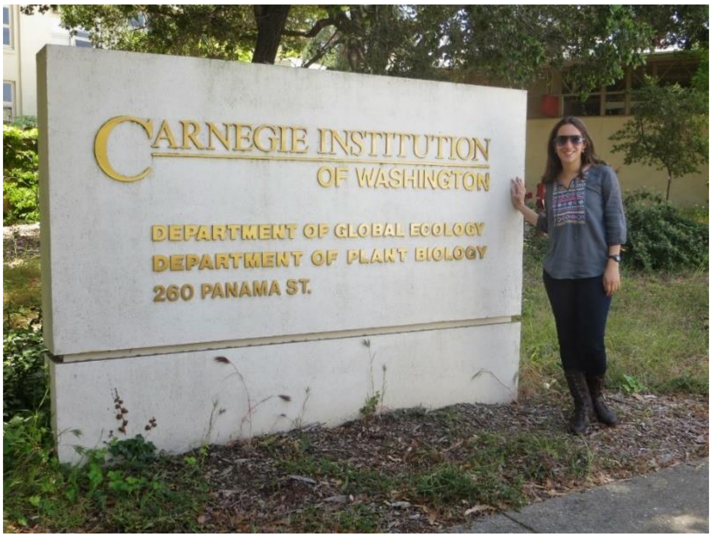


Figura 4: Fiz meu estágio no exterior e desenvolvi trabalhos do meu doutorado no Carnegie Institution of Science na Universidade de Stanford.

Figuras 5 e 6: Trabalho de campo com a equipe do Dr. Greg Asner na reserva Jasper Ridge Biological Preserve. Essa reserva está localizada próxima à Universidade de Stanford, e nela são desenvolvidos diversos experimentos da universidade.

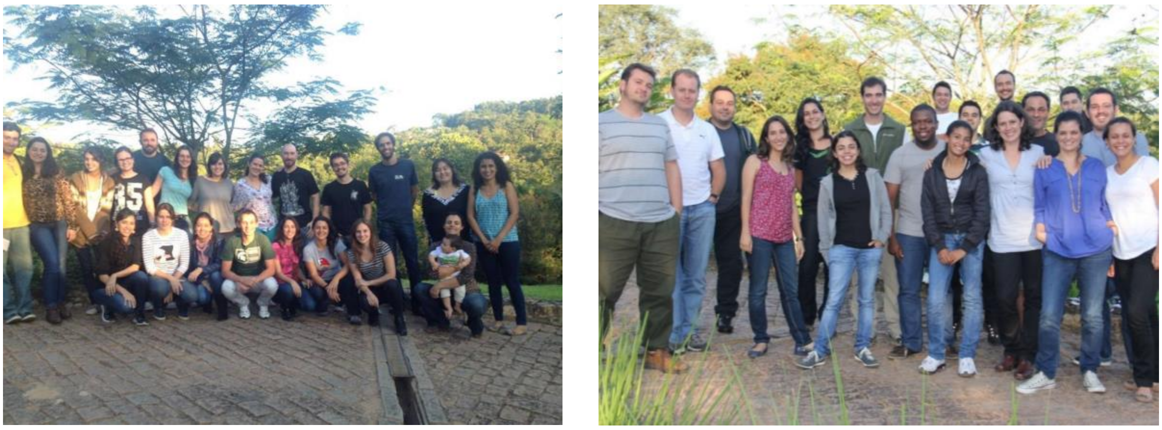
Figuras 7 e 8: Inauguração de novo avião que passará a realizar a coleta de novas informações do Carnegie Airborne Observatorie (CAO).

Em 2014, representando o IPAM, participei de um curso oferecido pela Google na própria sede da empresa em Mountain View, na Califórnia. No curso aprendemos sobre as possibilidades que temos com as ferramentas do Google para o terceiro setor, trocamos experiências e contatos com outras ONGs e conhecemos as dependências do inovador complexo da Google (Figura 9).

Figura 9: Participação de curso na sede da Google em Mountain View, Califórnia, representando o IPAM em 2014.

No Carnegie Institution, totalmente ao encontro com meu interesse em divulgar ciência, tive a oportunidade de colaborar com a tradução de um documento do Intergovernmental Panel on Climate Change (IPCC) sobre mudanças climáticas na América Latina para o português em 2014.

Outra rica experiência que tive durante meu doutoramento foi ser orientadora em parceria com a professora Natália Mundim (UFU) em cursos de curta duração na Escola Superior de Conservação Ambiental e Sustentabilidade (ESCAS) do Instituto de Pesquisas Ecológicas (IPÊ) em Nazaré Paulista (SP). Em 2012 e 2013 fui orientadora do curso de Modelagem de Distribuição de Biodiversidade (Figuras 10 e 11).



Figuras 10 e 11: Em parceria com a professora Natália Mundim (UFU), ministrei os cursos sobre Modelagem de Distribuição de Biodiversidade no Instituto de Pesquisas Ecológicas (IPE). As fotos mostram fotos das turmas de 2012 (esquerda) e 2013 (direita).

Integrar um laboratório com muitos alunos e ter um orientador que incentiva a troca de experiências e a ajuda entre seus orientados também permitiu que eu contribuísse com diversas teses e dissertações.

Além de uma experiência profissional riquíssima, o IPAM foi importante pois me fez encontrar a área de pesquisa que quero seguir: a vulnerabilidade de povos indígenas. Dentro dessa temática, pelo IPAM, participei do desenvolvimento de uma plataforma online de divulgação sobre ameaças climáticas e ambientais aos povos indígenas da Amazônia brasileira. A plataforma, denominada SOMAI (Sistema de Observação e Monitoramento da Amazônia Indígena), é financiada com recursos da embaixada da Noruega e já está online, com acesso aberto a todos os usuários (www.somai.org). Entre os produtos gerados a partir desse projeto, está a divulgação das ameaças aos indígenas da Amazônia brasileira nas COPs 20 e 21, realizadas respectivamente em Lima (2014) e Paris (2015). Senti grande satisfação de realizar diversas análises que foram divulgadas em eventos paralelos nessas COPs.

Em 2015 tive a oportunidade de divulgar na Universidade Federal de Goiás alguns resultados dos trabalhos da equipe que integro no IPAM. Realizei uma palestra na Semana do ICB sobre o efeito das mudanças climáticas nos povos indígenas. Também ministrei uma aula e uma palestra no curso de Educação Intercultural para indígenas alunos da UFG sobre mudanças climáticas (Figura 12).

Figura 12: Palestra ministrada para alunos indígenas do Curso de Educação Intercultural da UFG. A palestra tratava sobre o impacto das mudanças climáticas nas Terras Indígenas.

No final de 2015 tive a oportunidade de exercitar outras habilidades importantes para um pesquisador. Participei ativamente da concepção e escrita do projeto sobre Mudanças Climáticas e Povos Indígenas coordenado pelo Dr. Paulo De Marco Jr. em parceria com o IPAM. O projeto foi aprovado do CNPq.

INTRODUÇÃO GERAL

Antes de começar...

A introdução geral dessa tese está no formato de um artigo de divulgação científica para a revista Ciência Hoje. Essa decisão teve sua origem em duas insatisfações que me acompanharam por todo meu doutorado. Com o tempo percebi que minha primeira insatisfação não era apenas minha, mas também de boa parte dos pós-graduandos com os quais tive contato durante minha formação acadêmica. O que dizer para sua família e seus amigos, que não fazem parte do meio acadêmico, quando eles faziam a fatídica pergunta: “Mas o que você faz no seu doutorado mesmo? ”. Pânico. Para não cansar meu público tão simpático e bem-disposto, que ousou fazer uma pergunta tão terrível sem perceber o tamanho do seu erro, eu comecei a desenvolver respostas bastante simples e superficiais. Se depois de um tempo a pessoa continuava interessada, eu arriscava um pouco mais, mas logo caía em termos complexos e pouco interessantes como diversidade filogenética e funcional, *forest Dieback*, *hyperspectral*, mudanças climáticas e por aí vai.... Acredito que minha incapacidade de elaborar respostas simples e interessantes para tais questões complexas fez com que minha família e meus amigos continuassem a ter pouca ideia do que fiz no meu doutorado.

A segunda insatisfação surgiu de minha tentativa de defender para esse mesmo público a preservação da Amazônia e dos povos indígenas cuja sobrevivência dela dependem. Quando a discussão está focada na importância de preservação ambiental, não há problema, todos estão de acordo. O problema começa quando eu tentava discutir a importância de mudarmos comportamentos e padrões de consumo. Todos concordavam que precisamos preservar a Amazônia, mas ninguém se convencia que mudar alguns padrões de consumo seria importante, ou mesmo, que um custo tão alto valeria a pena para conservar a Amazônia. Entendi que minha revolta não era justa quando li a seguinte frase em um artigo: *“Science can do no more than provide an informed context for what is utimately a societal choice”* (Gardner et al., 2009). Após profunda reflexão aceitei que as decisões das pessoas e da sociedade não são minhas. Como cientista meu principal papel é gerar tais informações e, sempre que possível, fazer com que essas informações cheguem às pessoas.

Dessa forma, a introdução geral desta tese tem como principal objetivo ser simples e interessante, sem deixar de ser informativa e profunda. Apesar de destinada ao público em geral, esse artigo de divulgação aborda todos os principais elementos da minha tese. Assim, nas próximas páginas discutirei como a Amazônia tem sido ameaçada pela expansão das obras de infraestrutura e o desmatamento a elas associado, pela expansão das queimadas florestais e pelos eventos de seca que devem ser intensificados com as mudanças climáticas globais. Espero que aproveitem a leitura!

Gardner, T. a., J. Barlow, R. Chazdon, R. M. Ewers, C. A. Harvey, C. A. Peres, and N. S. Sodhi. 2009. Prospects for tropical forest biodiversity in a human-modified world. Ecology Letters 12:561–582.

O QUE AMEAÇA A AMAZÔNIA?

Abertura

Apesar de sua enorme importância ecológica, social e climática, o futuro da Amazônia está ameaçado. São diversos impactos humanos afetando negativamente o bioma, como mudanças climáticas, desmatamento, queimadas, construção de obras de infraestrutura e agropecuária. Individualmente, cada um deles já ameaça a Amazônia, mas ao interagirem entre si, esses impactos geram consequências ainda piores. Impactos negativos favorecem o surgimento de outros impactos, causando um ciclo vicioso negativo que pode ameaçar a sobrevivência da floresta.

Maior floresta úmida tropical existente hoje, a Amazônia é um dos ecossistemas de maior importância ecológica no planeta. Em nenhum outro lugar da Terra existem tantas espécies de plantas e animais – são aproximadamente 10% das espécies conhecidas no mundo ocorrendo na Amazônia, sendo que ainda existem muitas espécies desconhecidas. A floresta amazônica também é fundamental para manutenção de diversos povos tradicionais, como comunidades ribeirinhas e povos indígenas. Só na Amazônia são faladas cerca de 200 línguas indígenas, incluindo alguns povos que nunca tiveram qualquer contato com não indígenas (conhecidos como "povos indígenas isolados"). A Amazônia ainda exerce papel fundamental na regulação climática do planeta, mantendo um clima regional úmido e permitindo, por exemplo, a existência de um clima ameno ao leste da Cordilheira dos Andes.

Se a Amazônia é tão importante e valiosa, deveríamos zelar e protegê-la, correto? Porém, apesar da elevada importância ambiental, social e climática, nas últimas décadas a sobrevivência da floresta Amazônica tem sido ameaçada por diferentes fontes de degradação ambiental provocadas direta ou indiretamente por atividades humanas. Mudanças climáticas, altas taxas de desmatamento, expansão das queimadas, construção de grandes obras de infraestrutura (como estradas, portos e hidrelétricas), expansão da agricultura e pecuária, mineração e exploração de petróleo são algumas das principais ameaças que a Amazônia brasileira sofre atualmente... A lista das ameaças é tão grande que fica difícil citar todas. Mas a grande questão não é entender o efeito de cada ameaça que coloca em risco a sobrevivência da Amazônia. O que os cientistas estão descobrindo agora é que todas essas ameaças juntas podem interagir entre si gerando um resultado final muito pior que a simples soma das partes. Em outras palavras, nesse caso, dois mais dois pode ser algo muito maior que quatro.

O desmatamento da Amazônia

Pressões econômicas e sociais têm provocado o desmatamento da Amazônia. Até hoje, aproximadamente 20% de toda sua cobertura florestal da Amazônia brasileira já foi perdida. Isso representa as áreas somadas de Espanha, Inglaterra, Portugal e Bélgica. A criação de gado é a principal destinação das áreas desmatadas, com aproximadamente 75% das áreas de floresta removidas convertidas em pastagem para a pecuária. A produção de soja pelo agronegócio (segundo colocado) também é responsável por milhares de hectares desmatados todos os anos.

A maior parte do desmatamento está localizada nas regiões sul e sudeste do bioma, que não por acaso é conhecida como a região do ‘arco do desmatamento’. A Amazônia central foi pouco desmatada até o momento e o principal motivo é a proteção gerada por Terras Indígenas e Unidades de

Conservação (a Amazônia brasileira é o bioma com maior número de áreas protegidas do país – 54% do bioma está sob alguma forma de proteção) e ao fato de que muitas áreas da Amazônia são quase inacessíveis à ocupação humana. Essa dificuldade de acesso tem ajudado a preservar a floresta já que limita o desenvolvimento de atividades humanas na região.

Quando são construídas estradas há um aumento no desenvolvimento de atividades humanas, e, consequentemente, se tornam maiores também as taxas de desmatamento e o número de queimadas florestais. Estradas são construídas para promover a acessibilidade e dar suporte às atividades humanas como, por exemplo, expansão da agropecuária, extração de petróleo, mineração e operação de grandes usinas hidrelétricas. Apesar de promover alguns benefícios sociais e econômicos no curto prazo, a construção dessas grandes obras de infraestrutura está relacionada com o aumento no desmatamento, causando intensos problemas locais como a mudança na dinâmica dos rios, ameaçando plantas e animais e todos os povos que dependem da floresta para sua sobrevivência.

Entretanto, esse efeito cascata negativo gerado pela intervenção humana não se limita a ter um impacto apenas no local na região modificada. Regiões da Amazônia que possuem grandes áreas desmatadas se tornam mais quentes e secas. Nesse ponto, chegamos em uma grande contradição: se essas áreas foram desmatadas para expansão da agropecuária e construção de usinas hidrelétricas, um clima mais seco e quente gera menos água e pode comprometer a eficiência dessas atividades... O tiro sai pela culatra e a área que parecia muito adequada para atividades agropecuárias e para termos usinas hidroelétricas tem essas características modificadas quando alteramos o ambiente. As condições climáticas não são permanentes nem no espaço e nem no tempo. Por isso, a presença de Terras Indígenas e de Unidades de Conservação que consigam manter a floresta protegida, e consequentemente amenizem a mudança climática local, desempenham um papel importantíssimo para a viabilidade de boa parte das atividades econômicas que estão sendo desenvolvidas na região.

Como o fogo altera toda a estrutura da floresta amazônica

Para ocorrerem incêndios são necessários três elementos: combustível (tudo que é suscetível de entrar em combustão), comburente (elemento que ao se associar ao combustível é capaz de fazê-lo entrar em combustão, o principal é o oxigênio) e calor (temperatura para que a combustão se processe). Incêndios naturais na Amazônia são raros, porém os humanos mudam o ambiente Amazônico e o tornam mais propício aos incêndios. Com o aquecimento global, desmatamento, e as outras fontes de impactos já descritas anteriormente, estamos aumentando o material combustível e as fontes de ignição que geram fogo na Amazônia. Imagine o fogo passando pela floresta Amazônica. Diversas plantas e animais não suportarão o fogo e haverá uma mudança na composição de espécies, já que várias plantas e animais sucumbirão, e no clima local, que ficará mais seco. Mas não é apenas isso. Imagine uma enorme árvore no meio de uma floresta caindo por causa de um incêndio. Ao cair ela derrubará pelo menos algumas outras árvores e formará as chamadas “clareiras”. Essas áreas com árvores caídas no meio da floresta formam ambientes mais abertos, mais secos e com maior entrada direta de luminosidade, causando uma grande mudança na estrutura da floresta.

Os galhos e folhas secas das árvores mortas servirão como combustível para alimentar futuras queimadas, que por sua vez serão mais fáceis de serem iniciadas em um ambiente mais seco e quente. Além disso, um ambiente mais claro também favorece o estabelecimento de gramíneas invasoras originárias de pastagens vizinhas. Essas gramíneas podem avançar ao longo da floresta suprimindo o desenvolvimento de novas árvores e aumentando a chance de ocorrerem novos incêndios na floresta. Como resultado dessas mudanças, áreas de floresta já queimadas se tornam muito mais propensas a serem queimadas no futuro. Em outras palavras, na Amazônia, fogo gera mais fogo!

O fogo também é um vilão das florestas de maneira indireta. Durante o período da seca, em regiões que ocorrem muitas queimadas a poluição do ar gerada pelos incêndios reduz intensamente a radiação solar que chega à floresta. Como as plantas precisam da luz solar para fazer fotossíntese (processo indispensável à sobrevivência das plantas), essas mudanças na atmosfera causadas pelos incêndios aumentam o estresse e a mortalidade das árvores, contribuindo para a degradação da floresta.

O número de incêndios e a área anual queimada na Amazônia brasileira têm aumentado nas últimas décadas, mas os anos de seca são os que geram mais queimadas. Mesmo nos anos em que o combate ao desmatamento conseguiu reduzir as taxas de desmatamento, a incidência de queimadas continuou a apresentar uma tendência de aumento. Uma preocupação muito séria é que nas próximas décadas a frequência de incêndios e a consequente degradação florestal gerada devem aumentar bastante uma vez que diversas regiões da Amazônia devem ficar mais secas por causa das mudanças climáticas globais.

Se o sertão vai virar mar, a floresta pode virar sertão?

Você tem sentido que sua cidade está mais quente nos últimos tempos? Se comparado a uns 10 ou 15 anos atrás, o clima de sua cidade mudou? Se sua resposta foi positiva para as perguntas anteriores, saiba que isso não aconteceu apenas com sua cidade. Infelizmente, nos últimos anos o planeta ficou tão quente como nunca havíamos observado desde que começamos a fazer medições de temperatura no século XIX. As evidências dessas mudanças são fortes, por isso, a ampla maioria dos cientistas que estão estudando as mudanças do clima estão frequentemente interessados em saber **como** essas mudanças estão acontecendo, e pouca dúvida existe **se** tais mudanças estão ocorrendo. Em outras palavras, sabemos que mudanças estão acontecendo, o que nós ainda não sabemos é como o clima irá mudar...

Quando falamos de clima não falamos apenas sobre temperatura, mas também nos padrões de chuvas que caem sobre a floresta. Nos últimos anos da Amazônia, por exemplo, tem sido observado um aumento na incidência de eventos de seca e, infelizmente, isso pode piorar. Os estudos que tentam entender a mudança do clima no planeta indicam que essas secas podem se tornar ainda mais intensas e frequentes no futuro. Os indícios de que essas mudanças devem acontecer são grandes já que, em diferentes partes do mundo, grupos independentes de pesquisa têm chegado a resultados parecidos: a frequência e extensão geográfica de tais eventos de seca devem aumentar nas próximas décadas na Amazônia. Para toda a região amazônica, estudos sugerem que a área afetada por secas muito intensas irá triplicar até 2100. Caso se concretizem as mudanças esperadas por esses estudos, a maior floresta úmida do planeta se tornará um ambiente muito mais seco.

Apesar de fundamental para a manutenção de um clima equilibrado, a floresta Amazônica é um bioma muito sensível às alterações climáticas globais ou aquelas geradas por desmatamento. Isso acontece porque esses eventos de secas desencadeiam a expansão de incêndios florestais e o aumento da mortalidade de árvores, iniciando um processo de degradação ambiental severo. Como já vimos, uma floresta mais quente e seca facilita a incidência de queimadas florestais que geram... um ambiente ainda mais quente, seco e propenso a ser queimado no futuro! Se o ambiente muda tanto, é inevitável que algumas espécies de plantas e animais não conseguirão mais sobreviver nessas áreas alteradas e a floresta deixará de ter muitos elementos que a caracterizam como Amazônia.

Essa cadeia de eventos em que um impacto inicial gera outro impacto, que por sua vez gera outro impacto, e por aí vai, é conhecida pelo termo em inglês *forest dieback*. Não existe uma tradução adequada para esse termo em português, mas poderíamos definir *forest dieback* como um sistema de retroalimentação positivo em que uma sucessão de impactos gera outros impactos e, em última instância, provoca a mortalidade da floresta. Notem que nesses casos os impactos provocados são iniciais e todos os outros impactos são secundários e já não dependem mais de uma fonte de impacto externa. Ou seja, atividades humanas provocam desmatamento e alteram o clima do planeta (incluindo o clima da Amazônia). Mas após um determinado ponto, que não sabemos se já passamos, estamos pertos de passar ou ainda distantes, não seremos capazes de cessar esses impactos, mesmo que todos os impactos iniciais sejam suspensos. A floresta degradada não conseguirá mais voltar ao seu estado inicial.

Em geral, gostamos de terminar nossas leituras com uma conclusão. Mas pesquisadores gostam de ler, estudar, compreender e, no final, terem novas perguntas, para entenderem ainda mais o processo. Seguindo a lógica dos pesquisadores, após entendermos que existe um sistema que se retroalimenta e impacta negativamente a Amazônia, duas questões principais surgem: 1- Quanto a floresta deverá ser modificada para que ela deixe de ser a Amazônia que conhecemos? 2- Quanto impacto podemos gerar antes que a floresta entre no sistema de retroalimentação e passe a ser degradada sem a influência humana? Cientistas por toda parte agora estão debruçados sobre essas questões... E você? Qual a sua opinião?

Um guia para os incautos

Essa tese tem como eixo conceitual que unifica seus três capítulos o tema *Amazon forest dieback*. Nos últimos anos, tem surgido cada vez mais evidências que a Amazônia pode estar ameaçada em decorrências das mudanças regionais no clima e no uso do solo. Por causa de um sistema de retroalimentação positivo, em que impactos intensificam outros impactos, alguns modelos projetam para o final do século uma substituição da floresta amazônica por formações tipo savana ou semiáridas (Cox et al. 2004, Barlow and Peres 2008, Nepstad et al. 2008, Bennett et al. 2013). Diversos modelos climáticos globais projetam para as próximas décadas uma intensa redução da precipitação na Amazônia provocada pelas mudanças climáticas. (Cox et al. 2004, Duffy et al. 2015, Trumbore et al. 2015). Outras evidências suportam a ideia de que o desmatamento e a degradação florestal desencadeados por atividades humanas (e.g. conversão de áreas para agropecuária e construção de estradas) produzem impactos que são intensificados por eventos de seca, ao mesmo tempo que alteram o clima regional (Laurance 2000, Nepstad et al. 2004, Bagley et al. 2014, Brando et al. 2014). Apesar de todas as evidências, *forest dieback* é um conceito muito novo e que é baseado, em grande parte, em projeções derivadas dos modelos climáticos futuros. Como consequência, são ainda profundamente desconhecidos os mecanismos e padrões gerais de como esses impactos afetam a floresta. Nesta tese, desenvolvi trabalhos que têm como objetivo contribuir para as discussões do tema. Em cada capítulo, abordarei uma ameaça que, de alguma forma, está contribuindo para a degradação da Amazônia. Cada uma das três ameaças discutidas nos próximos capítulos são, frequentemente, apontadas como importantes propulsores de *forest dieback*.

No primeiro capítulo avaliei o impacto de incêndios florestais em diferentes níveis de diversidade de árvores de uma área florestal próxima à transição

Amazônia-Cerrado. Embora os incêndios florestais na Amazônia sejam naturalmente raros, eles têm se tornado mais frequentes, intensos e amplamente distribuídos. Essas mudanças de regimes de fogo podem conduzir a degradação florestal generalizada na Amazônia, com consequências importantes não só para a riqueza de espécies, mas também para a diversidade funcional e filogenética das comunidades. Usando informações de um experimento de 9 anos de queimadas controladas. Nossos resultados sugerem que as comunidades de árvores das áreas queimadas estão perdendo mais diversidade filogenética e funcional por unidade de espécie do que nas áreas não queimadas. Nossos resultados indicam a existência de seleção de espécies com base em características filogenéticas e funcionais, que representam uma importante força de mudança e de empobrecimento (funcional e filogeneticamente) dessas comunidades. Compreender os mecanismos e padrões gerais de como estes incêndios estão afetando a floresta é essencial para entender o que pode acontecer com grandes áreas de floresta amazônica se os incêndios florestais continuarem a aumentar.

No segundo capítulo eu usei imagens de alta resolução para avaliar o impacto de um período de seca em áreas florestais próximas a Madre de Dios, Peru. Com as mudanças no clima regional, as secas na Amazônia estão se tornando mais frequentes e intensas, no entanto, as árvores não são afetadas da mesma maneira. Alguns compostos químicos presentes nas folhas podem aumentar a resistência de algumas árvores ao stress hídrico provocado por esses eventos de seca. Assumindo que a presença desses compostos possa ser detectada em imagens hiperspectrais de alta resolução, nesse capítulo eu avaliei se é possível detectar a vulnerabilidade do dossel florestal aos eventos de seca com base em índices hiperspectrais. Para isso, eu avaliei se a seca de 2010 aumentou a formação de grandes *gaps* no dossel da floresta, se índices espectrais foram capazes de predizer a formação desses *gaps* e, finalmente, de que forma esses índices foram afetados ao longo dos anos. Nossas reultados mostraram que a formação de novos *gaps* dobrou durante o período de seca e que baixos valores de NDVI e altos valores de PRI representaram dosséis mais vulneráveis à formação de novos gaps. Nossos resultados dão suporte a ideia de que mudanças no clima regional pode modificar a estrutura e função da floresta.

No terceiro capítulo, eu avaliarei de que forma a construção de estradas na Amazônia brasileira tem contribuído para o desmatamento que incide em um importante grupo de áreas protegidas da Amazônia: as Terras Indígenas. Usando uma metodologia objetiva nós propomos o estabelecimento de zonas de amortecimento (*buffers)* em torno dessas reservas para reduzir o impacto das estradas planejadas para serem construídas pelo governo brasileiro no futuro. Nesse capítulo nós propomos que a expansão da rede de estradas considere as informações dos impactos existentes atualmente como subsidio para um planejamento eficaz que minimize os efeitos negativos da construção de estradas futuras em populações indígenas da Amazônia.

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CHAPTER 1

BURNING BIODIVERSITY: FIRE EFFECTS ON TAXONOMIC, PHYLOGENETIC AND FUNCTIONAL DIVERSITY OF A NEOTROPICAL FOREST [2]

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[2] Proposed journal target: Ecological Applications.

ABSTRACT

Although tropical forest fires are naturally rare, they have become more frequent and intense in response to recent land and climate change. This shift in fire regimes may drive widespread forest degradation in Amazonia, with important consequences not only for species richness, but also for functional and phylogenetic diversity. However, most studies focus on species richness alone, ignoring other diversity metrics. Here, we address this gap based on a large-scale, long-term experiment (9 years) comprised of three 50-ha plots located in southeast Amazonia: unburned control and two different fire regimes (annual and triennial fires). Overall, we found that compared with our unburned control, tree sub assemblages exposed to experimental fires lost more phylogenetic and functional diversity per species-unit. However, while fire-induced changes in diversity of small trees were similar between the two fire treatments, large trees exposed to triennial fires generated the higher decrease of phylogenetic diversity per species-unit. Our results indicate the existence of species selection based in phylogenetic and functional traits. This selection represents an important force of change and impoverishment (functionally and phylogenetically) of tree sub-assemblages. Understanding mechanisms and general patterns of how these fires are affecting the forest is essential to understand what can happen to large areas of the Amazon forest if forest fires continue to increase.

Keywords: Amazon, forest dieback, plant traits, tree mortality, tropical forests

INTRODUCTION

Amazonia’s biodiversity is threatened by fire, logging, deforestation, and, other drivers of forest degradation. As people clear, degrade, and fragment forests, they become more flammable and more prone to experiencing high-intensity fires. These drivers of Amazon forest degradation cause major losses in biodiversity (Bradshaw et al. 2009, Dirzo et al. 2014). Droughts superimposed upon these activities increase the likelihood of widespread fires, which can further reduce forest diversity. Between 1999 and 2010, for example, more than 85,500 km^2 of Amazonian forest burned — almost 3% of the total forested area of the basin. Given that tropical forest fires peak during dry-warm periods and that sources of fire ignition are not lacking, projected climate change strongly suggest that forest flammability will increase over the next decades across much of the eastern Amazonia (Duffy et al. 2015). However, the potential effect of this change in fire regime on the regional biodiversity remains unclear.

The dense canopy of tropical forests creates a moist environment in the understory that suppress fires during most non-drought years (Hoffmann et al. 2012). This helps explain why the natural incidence of fire in pre-Columbian times was so rare (Alencar et al. 2011, Hoffmann et al. 2012). As a result, fires occurring every 400-700 yrs exerted minor or no selective pressure towards species with adaptations to resist fire (e.g., thick bark). Thus, when tropical forest fires do occur, tree mortality rates tend to be high across a wide range of taxonomic groups, although a few large, thick-barked trees species resist such fire events (Barlow and Peres 2008, Balch et al. 2011, Brando et al. 2012, Silvério et al. 2013).

Although Amazonian tree species tend to be vulnerable to understory fires (Barlow et al. 2003, Brando et al. 2014), post-fire tree mortality can be highly variable (Balch et al. 2011, Brando et al. 2016). Two interrelated processes explain this high variability. First, fire intensity is controlled by processes that vary spatially and temporally (e.g., fuel characteristics and forest understory microclimate) (Cochrane et al. 1999, Hole et al. 2005, Cochrane and Laurance 2008, Brando et al. 2014). Second, fire-severity is strongly influenced by plants traits such as bark thickness and wood density, which vary widely within and across tree species (Gerwing 2002, Balch et al. 2011, Martins et al. 2012). While fire-induced tree mortality influence several types of diversity, most studies on this topic quantify only fire-related effects on species richness (Balch et al. 2008, 2011). Thus, a broader viewpoint that also includes phylogenetic and functional diversities could yield additional insights into the causes and consequences of observed community responses to fire (Cavender-Bares and Wilczek 2003, Arroyo-Rodríguez et al. 2012, Cianciaruso et al. 2012).

If fire kills trees in a random way, theory suggests that any reductions in fire-related tree density should drive proportional reductions in (i) number of species, (ii) functional, and (iii) phylogenetic diversity. Nevertheless, even in a system poorly adapted to fire such as Amazonian forests, non-random fire-induced tree mortality is common (Brando et al. 2012, Lawes et al. 2013). When fires select species with specific functional characteristics (e.g., large diameter), losses in functional diversity are expected to be higher than in taxonomic diversity. Otherwise, if the resistance traits are overdispsersed in the phylogeny, the loss of phylogenetic and taxonomic diversity should occur at similar rates. Nevertheless, the fire regimes may vary in different aspects (e.g. large vs small intervals; localized intense, large scale and low intense), what can add a new dimension of complexity that challenge the above simple predictions. This should happen especially if the resistance of the species varied according to different fire regimes considered.

Fire regime metrics vary widely in the tropics, resulting in fire-induced tree mortality rates that range from 8% to 90%. This high variability in fire-induced mortality makes it difficult to understand the effects of fire on tree diversity. To address this issue, we take advantage of large-scale, long-term experiment located in southeast Amazonia, which is comprised of a long-term fire controlled experiment with annual and triennial burn regimes(Balch et al. 2011, Brando et al. 2014). In the context of this experiment, we evaluate how two different fire regimes affect three levels of diversity: richness, phylogenetic, and taxonomic. Given that previous studies showed that fire intensity and fire-induced tree mortality were higher for small, thin barked individuals and for trees growing in treatment burned every three years (B3yr) than in the burned every year (B1yr), we expect that: (1) fire-induced losses in functional and, phylogenetic diversity will be higher than losses in taxonomic diversity; (2) overall losses in diversity will be higher in B3yr than in B1yr, and lowest in the unburned control; (3) trees in small class size should present less fire resistance traits, so the selection of species and functional traits will be more random compared with large-sized trees. Due the lack of a functional/phylogenetic structured species loss, the diversities decrease should be more intense and homogeneous among species.

METHODS

Study area and fire experiment

The fire experiment was established in 2004 in transitional area between the Cerrado and Amazonia, southern Amazon basin (13° 04′S, 52° 23′N). In this region, the average annual precipitation of 1700 mm falls predominantly during the wet season (October-May). Experimental burns were conducted at the end of the dry season (August-September), when the climatic conditions are driest. Detailed field methodology is presented in (Balch et al. 2008, 2011, Brando et al. 2012), here we present a summary of the most important points.

The experiment was established in an area with no signs of recent fires and was comprised of three 50-ha plots: an unburned control, a plot that was experimentally burned every three years (2004, 2007 and 2010; triennial fire), and a plot that was experimentally burned every year from 2004 to 2010 (annual fire) with the exception of 2008. We classified every tree individual according to the stem diameter at breast height (dbh) into two size classes: 10–20 cm (small size class) and ≥40 cm (large size class). For each 50-ha plot, we conducted yearly a mortality census of trees presented at the beginning of the experiment. We also mapped new recruitments in the years 2007, 2008, 2010, and 2012. Within each 50-ha plot we collected information for small- (10–20 cm) and large-sized trees (≥40 cm) across transects and the entire experimental area, respectively. The experimental fires of 2007 and 2010 were ignited during unusual dry-warm conditions and, therefore, were more intense than the average fire intensity of other years (Brando et al. 2014).

A total of 9321 individuals were monitored throughout this experiment. 7151 individuals were sampled at the beginning of the experiment, in 2004, and were monitored all the years until 2012. Of all individuals sampled, 5813 were small size (dbh 10-20 cm) and 3508 were large size trees (dbh higher than 40 cm) in 2004.

Functional variables

To evaluate how the experimental fires affected the forest functional diversity, we used eight plant traits at the species level: maximum plant height, maximum basal area, bark thickness, wood density, pollination syndromes, leaf type (simple or compound) and leaf phenology (evergreen, semi deciduous or deciduous). To do so, we measured tree size (height and diameter) of 7744 individuals in the experimental area. Because we focused on species-level analysis, we used the maximum observed tree size for a given species. We also compiled data on wood density data from a list of wood density for neotropical tree species generated by Chave and collaborators (Chave et al. 2006). The other traits data were survey from literature using different data sources.

Phylogenetic relations

We constructed a phylogenetic tree for the sampled species using the Phylomatic megatree (PHYLOMATIC tree version: R20120829) (Webb and Donoghue, 2005) at the Phylomatic website http://www.phylodiversity.net/phylomatic/). Phylomatic tree is a megatree generated by hand, rather than by an automated algorithm, thus conflicting branching patterns were resolved subjectively. However, it is intend to represent an approximation of the true phylogeny of seed plants (Webb and Donoghue, 2005). If any genus is missing from the megatree, the program returns a polytomy of genera within that family. In Phylomatic 's megatree species are arranged as polytomies within genera because we do not have the branch length, we used the patristic distance among species in all analysis.

Figure 1. The phylogenetic tree for all sampled species has constructed a using the Phylomatic software, a phylogenetic database and toolkit for the assembly of phylogenetic trees (Webb and Donoghue, 2005).

Phylogenetic and functional diversity measures

To understand how different fire regimes affect the phylogenetic structure of the Amazon tree communities we calculated the phylogenetic diversity index (Faith 1992a). Phylogenetic diversity (PD) represents the sum of branch lengths of the phylogenetic tree to the all species in a community. To the functional analysis, we used a similar metric of phylogenetic diversity. For this, we built a functional dendrogram transforming functional traits into functional distances. Because our data consisted in qualitative and quantitative plant traits, we used a generalisation of Gower distance (Pavoine et al., 2009) considering the treatment of mixed data and UPGMA clustering. Because this final dendrogram present a similar data structure than the phylogenetic tree, the same index applied to a phylogenetic tree can be applied to a functional dendrogram (Pavoine and Bonsall, 2011). Thus, following the same logic of the PD index (Faith 1992a), to functional traits we used the functional diversity index (Petchey and Gaston 2006).

Data analyses

To compare how the successive fires affect the underlying process that control the sub-assemblage, we measure the relationships changes between richness (taxonomic diversity) and functional/phylogenetic diversity. We considered two distinct components of the community change: the intensity of the effects and process that was acting. Intensity of diversity loss was measured looking how much the diversity measures have change over the years. Operationally intensity was considered as the line size in the regression analysis. The process, otherwise, was evaluated acording the slope of the regression analysis. In this case, the process represents how the burn treatments are affecting the relationship between functional/phylogenetic diversity and taxonomic diversity in sub-assembleges. The process indicates how the functional/phylogenetic diversity decreased per species-unit and must be compared to the expected whether only natural process are acting in sub-assemblege (observed in control sub-assemblage). Thus, the higher the slope, the higher the functional/phylogenetic diversity loss per species-unit. Thus, this slope may represent the differences in the underlying process that control those sub-assemblages.

All the analysis was executed separately to small-sized and large-sized trees. The statistical analysis was performed by a linear regression between taxonomic diversity and functional/phylogenetic diversity over the years. Finally, the significance of the regression slopes was measured by an ANOVA.

RESULTS

Fire-related tree mortality was higher in the triennial than in the annual treatments for both small and large trees between 2004 and 2012 (Figure 2). Individual mortality for the small tree size-class presented the same trend than large trees, but mortality rates were higher among smaller individuals. Species abundance, taxonomic and functional diversity decreased following a similar trend in over time. Although decreases in these diversities (abundance, taxonomic and functional) tended to be higher in the triennial treatment than in the annual one, the decrease in phylogenetic diversity is higher in the annual treatment.

Overall, we found that 30% (unburned), 64% (B3yr) and 59% (B1yr) of the initially sampled small-sized trees died between 2004 and 2010. Mortality rates were lower for large trees than for small ones. To large trees we found quite lower values of tree mortality, 16% (unburned), 39% (B3yr) and 30% (B1yr) of the initially sampled. Following this same pattern observed to mortality, to species loss we also observed higher rates of species loss in triennial fire (16%) than in the annual fire (9%). For small trees, however, despite the abundance decrease be higher in triennial fire, the proportional reduction of species was similar for both fire regimes (29%). Over the years, the general trend of functional diversity reduction was not the same than the taxonomic diversity decrease. However, at the end, the result was very similar. For both fire regimes, small trees functional diversity has decreased 19%. To large trees, the reduction was of 7% to annual area and 16% to triennial area.

However, we have observed a different temporal trend to the phylogenetic diversity. Despite the higher individual tree loss under triennial fire, the annual fire regime yields higher decrease in phylogenetic diversity. While the species reduction was the same to small trees, in the annual fire the community have lost 16% of the phylogenetic diversity against 12% in triennial fire. To large trees, the phylogenetic diversity reduction was 16% and 17% to triennial and annual fire regimes, respectively. This was the only diversity measure in which large trees presented proportionally a higher reduction of diversity than small trees.

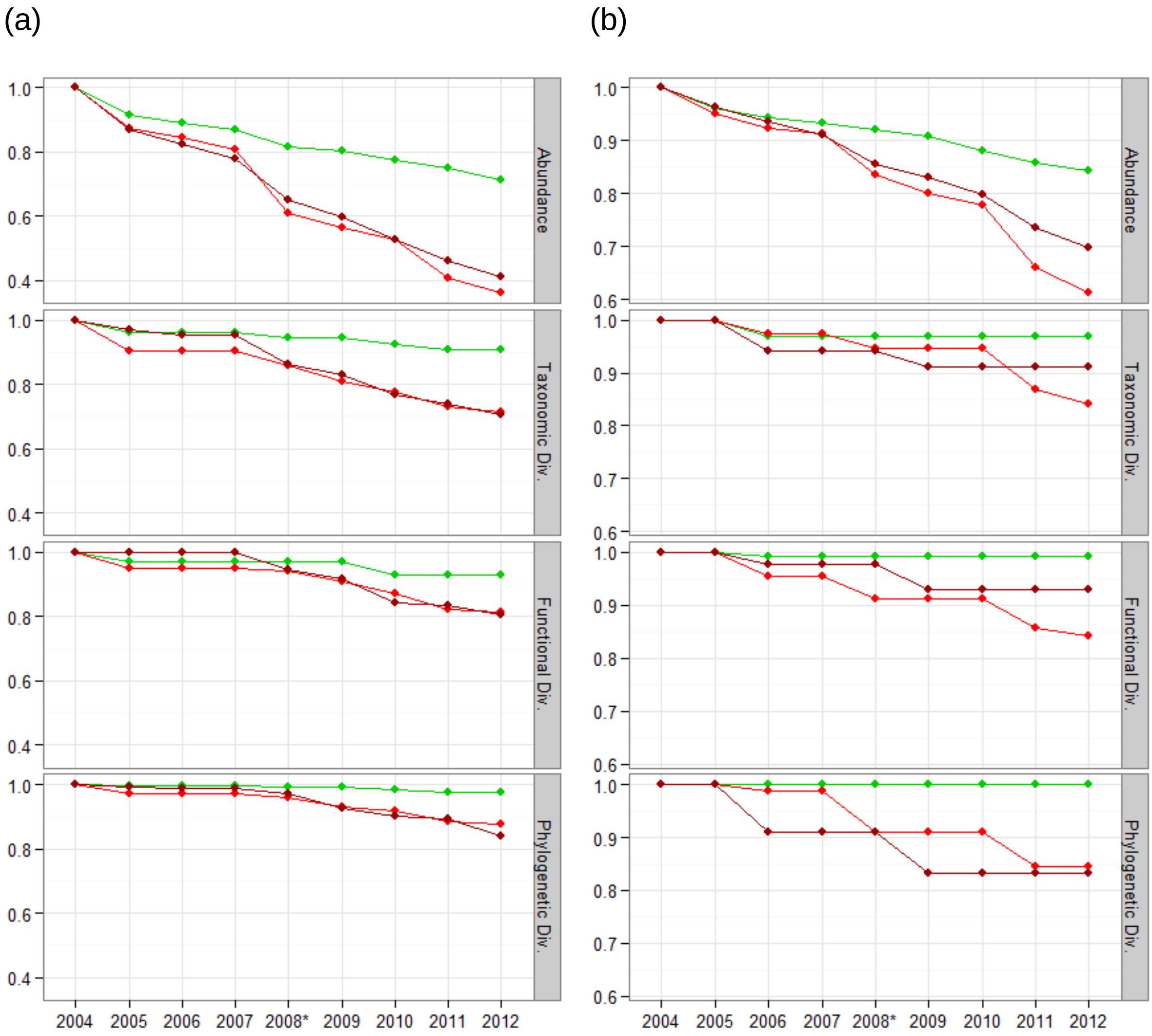

Figure 2. Temporal pattern of diversity changes for Control (green), B1yr (dark red) and B3yr (light red) sub-assemblages of trees present at the beginning of the fire experiment**.** We divided the stem diameter (dbh) in two size classes: (a) 10–20 cm and (b) ≥40 cm. B1yr was burned from 2004 to 2010, except 2008, when we did not conduct the experimental fires. B3yr was burned in 2004, 2007, and 2010.

Underlying relationships changes of the initial sub-assemblages

To evaluate the underlying relationship between taxonomic diversity and functional/phylogenetic diversity, we consided two components of change: intensity of changes and process that is driving the changes. The slope of the regression between richness and functional diversity remained the same throughout the experiment in all sub-assemblages to small size trees (Figure 3a). This pattern indicates that the process acting in tree species in burned sub-assemblages is simillar to that occurring in unburned and the diversity measeres are been lost in similar rates. However, the decrease of taxonomic and functional diversity is much more intense in burned than in unburned sub-assemblages (Figure 3a). To large trees, slope values are higher in burned than in unburned areas, indicating that these sub-assemblages are losing more functional diversity per unit species lost (Figure 3b). In this case, despite present a simillar pattern (similar slopes), the process acting in triennial area is more intense than in the annual area (Figure 3b).

In burned sub-assemblages, we lose phylogenetic diversity at faster rates than taxonomic diversity if compared to unburned sub-assemblages (Figure 3c), for both small and large trees. To small trees we did not observe a difference between the burned areas concerning the slope and arrow size, sugessting a similar process and intensity (Figure 3c). To large trees, however, the annual area show a faster decrease in phylogenetic diversity than the triennual area, even though the intensity of decrease was similar in both burned areas (Figure 3d).

Using the inference based on confidence intervals, it is possible to observe that those sub-assemblages are losing functional diversity at faster rates than losing phylogenetic diversity for small-size trees in burned areas (slopes near 0.7 for functional and below 0.5 for phylogenetic diversity; confidence intevals not covering the means of each other; Figure 3a and 3c). For large trees, triennial burns appear to exert similar effects on both diversity measures, but phylogenetic diversity is lost at faster rates than functional diversity for annual burned area (Figure 3b and 3d).

For the small trees the analysis of the slopes show that in both fire regimes there is a larger loss in functional diversity than in phylogenetic similarity. For the large size-class, the analysis of slopes shows different responses in relation to the fire regimes. There is a very similar change in both phylogenetic and functional diversity for the three-year fire regime, but the annual fires caused a higher loss in phylogenetic than in functional diversity. The last result supports the view that annual fires reduce phylogenetic similarity through the loss of functionally redundant species.

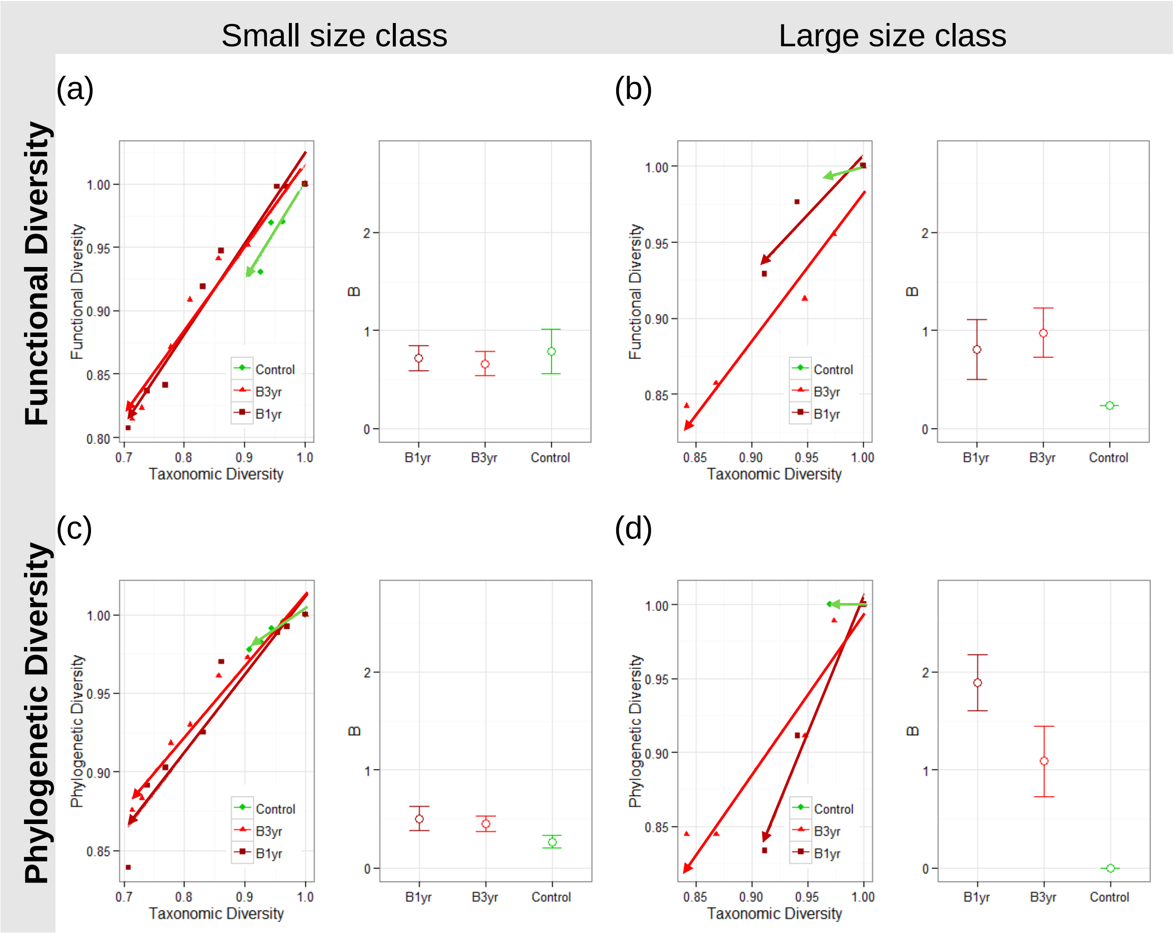

Figure 3. Changes in relationship between taxonomic diversity and functional/phylogenetic diversity in relation to initial sub-assemblages. Analysis were separated by diversity measure, functional (a and b) and phylogenetic (c and d), and by tree size class, small trees (a and c) and large trees (b and d). Sub-assemblages burned in annual (B1yr) and triennial (B3yr) fire regimes were compared to unburned sub-assemblage (Control).

Underlying relationships considering new recruitment

After incorporating the new recruitment in the analysis, some diversities rise higher values in initial years than those observed at the beginning of the experiment. In fact, for large trees, at the end of the nine years experiment we observe higher values of taxonomic and functional diversity than at the beginning. The relatioship between richness and functional diversity remained the same throughout the experiment in burned and unburned areas of small and large trees (Figure 4a and 4b).

For large trees, we observed similar patterns in phylogenetic and taxonomic diversity, with reductions occuring for all the treatments (Figure 4d). However, for small trees, the triennial fire caused a higher decrease in phylogenetic diversity than the annual fire regime. However, the relationship between small-tree phylogenetic and taxonomic diversity in annual area was similar to the unburned area (Figure 4c). The confidence-interval inference also support the claim that small-tree funcional diversity is lost at faster rates in the annual buned site (slopes near 0.7 for functional and near 0 for phylogenetic diversity; confidence intevals not covering the means of each other; Figure 4a and 4c).

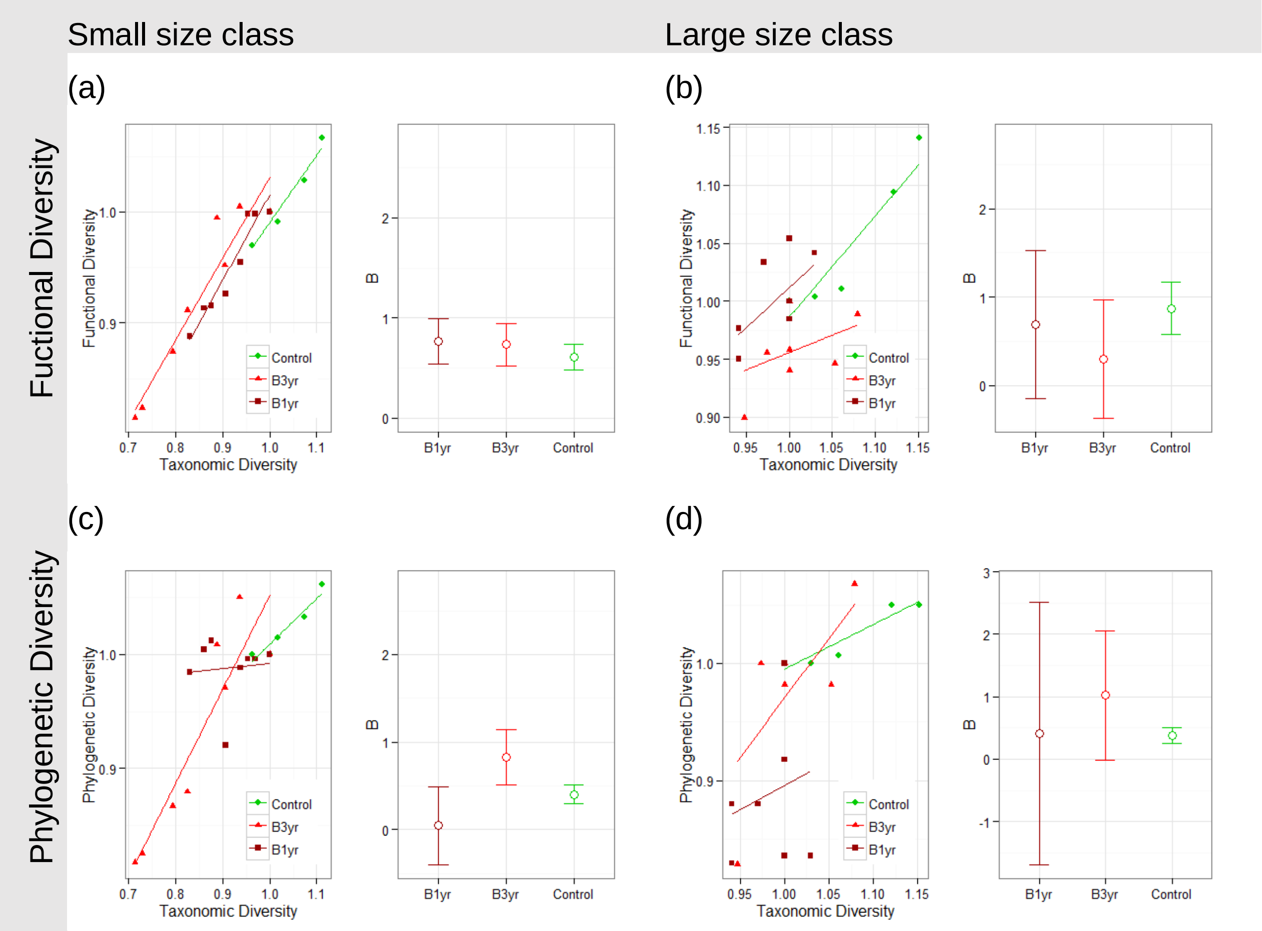

Figure 4. Changes in relationship between taxonomic diversity and functional/phylogenetic diversity incorporating the new recruitment individuals. Analyses were separated by diversity measure, functional (a and b) and phylogenetic (c and d), and by tree size class, small trees (a and c) and large trees (b and d). Sub-assemblages burned in annual (B1yr) and triennial (B3yr) fire regimes were compared to unburned sub-assemblage (Control).

DISCUSSION

We found that tree sub assemblages in the burned plots lost more phylogenetic and functional diversity per species-unit than in the unburned plot. This indicates the existence of species selection based on phylogenetic and functional traits representing an important force of change and impoverishment (functionally and phylogenetically) of these sub-assemblages. Moreover, these differences in species selection are not detectable when just the abundance or species richness is considered or when functional and phylogenetic diversities area considering separately. Our results emphasize the importance of consider and combine several level of diversities to reveal processes that impact the compositions of local communities (Pavoine and Bonsall 2011, Swenson et al. 2012). As expected, small trees experienced higher mortality rates than large trees. Besides the higher mortality, small-sized trees also presented more similar decline in phylogenetic/functional diversity per unity of species, despite the fire regime, than the observed in large-sized trees. This corroborates our previous hypothesis that the selective pressure by fire should occur mainly among large trees.

In one of the few existing fire experiments in tropical forests, it was reported that triennial burns caused more tree mortality than annual buns, because of higher fuel accumulation and associated fire intensity in the triennial burn treatment (Brando et al. 2014). We observed the same and losses in abundance and richness of tree species were higher in the plot burned every three years (Balch et al. 2011, Brando et al. 2014). However, our study incorporated others components of community diversity and also considered the effects of fire regimes in phylogenetic and functional diversity. While cumulative mortality was higher in the plot burned every three years between 2004 and 2010, reductions in others trees diversity were not necessarily more pronounced in that treatment. For example, large trees growing in the annually burned area experienced more losses in phylogenetic diversity per unit of species loss. This suggests that frequent fires killed species that were not affected by less frequent fire regime. The differential effects of the fire regimes in tree mortality are more explicit for large trees sub-assemblages, indicating that resistance traits are generating functional and phylogenetic selection according the fire regime.

Fire-induced tree mortality is an important process shaping the structure and diversity of tropical forests. However, other demographic processes also play important roles in shaping in fire-disturbed community assembles. Tree recruitment and growth of surviving individuals, for example, can be essential for population maintenance of some species (Enright et al. 2014). The contrasting effects of those processes shaping the entire plant community are expressed by the changes in overall conclusion as we included the new recruitments in the analysis. For example, in some years, after the fire disturbance, the species richness increased in burned sub-assemblages (if compared to the beginning of the experiment). The increase of species richness after disturbance can be explain by reduction of competition and changes in local conditions (e.g. more light available) that benefit news species. This study area is close to boundary Cerrado-Amazon what makes reasonable to think that species with Cerrado characteristics are been benefited by fires. There is an emerging consensus that fire drives a fundamental role to determine the transition from savanna to forest (Hoffmann et al. 2003, Silva et al. 2013, Dantas et al. 2013, Lehmann et al. 2014). A model based on alternative stable state theory was proposed to explain the dynamics of savanna-forest, where both vegetation types are alternative stable states with distinct structures and functions that are maintained by positive plant–fire feedbacks (Hoffmann et al. 2012, Murphy and Bowman 2012). Thus, in Amazon, fire is not necessarily just about mortality and loss of functional and phylogenetic diversity, but also about turnover in species and functional and phylogenetic characteristic of the community.

Contrary to our initial hypothesis, after nine years of controlled experiment, taxonomic diversity was more affected by successive fires than functional and phylogenetic diversity. It means that the burn assemblages had more intense decrease of individuals and species trees than the functional and phylogenetic diversity. Nevertheless, a close analysis comparing relative effects in burned and unburned sites shows a different pattern. Each species lost in the burned plots generated higher reduction in functional/phylogenetic diversity than the species lost by natural process observed in the unburned area. This is a by-product of the impoverishment in the burned area and may reveal an important issue to understand ecosystem function in those sites. Unique species do not necessarily play the most important and central roles to the functioning of a community (Lai et al. 2012). However, the losses of unique species can generate the loss of functions not provided by other species. The loss of complementarity among species can compromise important ecosystem function and is pointed to reducing heavily the ecosystem resilience (Petchey and Gaston 2009).

Based on the premise that traits evolves under phylogenetic restrictions and closely related species have more similar characteristics (Peterson et al. 1999, Wiens and Graham 2005, Pearman et al. 2010), it is expected that functional and phylogenetic diversity be directly related (Wisheu and Keddy 1992, Flynn et al. 2011, Baraloto et al. 2012). There is several evidences suggesting that phylogenetic diversity should be reflected in functional diversity (Felsenstein 1985, Harvey and Pagel 1991, Faith 1992b). However, there are several reasons for the variation in one of these diversity metrics will not be transferred to others. For example, the replacement of a phylogenetically less redundant species in the phylogeny for a more redundant, will decrease overall community phylogenetic diversity, with increase functional diversity but with no effect on the taxonomic diversity. On the other hand, some important functional traits may have little phylogenetic signal, and their loss will not necessarily means a reduction in the phylogenetic diversity in the community. A general view from our results was that our phylogenetic diversity is not reflecting the functional diversity. In addition to the explanations presented above, we could not sort out the possibility that phylogenetic diversity could be related to other functional traits not observed in this study (but see Vane-Wright et al. 1991)

Land use and climate change are driving intense changes in forest health. To identify the mechanisms and general patterns of how fire regimes are degrading the forest, it is important to develop comparative approaches that integrated multiple facets of diversity. Based on an experimental long-term approach, we found evidence of species selection based in phylogenetic and functional traits, which represent an important force of change and impoverishment (functionally and phylogenetically) of communities. Understanding how tree communities respond to fire regimes is essential to predict the impacts and develop conservation efforts against the wildfires expansion in Amazon forest.

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CHAPTER 2

PREDICTING FOREST CANOPY FAILURE DURING AMAZON DROUGHT WITH IMAGING SPECTROSCOPY [3]

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[3] Proposed journal target: Remote Sensing in Ecology and Conservation.

ABSTRACT

Amazon droughts are expected to become more frequent and intense in the near future as climate change, with important implications for forest structure, biodiversity, and functioning. However, our ability to detect and predict large-scale drought-related changes in ecosystem properties across a wide range of environmental gradients is still limited. Here, we use repeated airborne lidar-hyperspectral data collected in 2009, 2011 and 2013 to map plant response to drought in the Madre De Dios region. Our overarching goal was to evaluate how droughts affect proxies of canopy cover (NDVI) and photosynthetic light-use efficiency (PRI) and whether these variables are good predictors of drought-induced gap formation. NDVI and PRI indices were able to successfully predict the new gaps formation, indicating that theses indices reflect crown characteristics of trees vulnerability. We also had evidence that the drought period affected the structure and function of forest, since spectral indices values had been altered after the drought. Finally, our results show that the gap formation doubled during the drought period. Successive drought periods are expected to become more frequent in the future; our results support the idea that changes in regional climate could modify the structure and function of the forest.

Keywords: Rainforest; hyperspectral remote sensing; forest gaps, drought stress

INTRODUCTION

Drought events exert strong influences on global forest health (Aragão et al. 2007, Cox et al. 2008, Coe et al. 2013a, Trumbore et al. 2015). Drought-induced stress can kill trees (Adams et al. 2009, Phillips et al. 2010, McDowell 2011), alter function and structure of ecosystems (Belk et al. 2007, Nepstad et al. 2007, Brando et al. 2008, Gatti et al. 2014) and, ultimately, cause severe forest degradation (Cox et al. 2004, Nepstad et al. 2008, Malhi et al. 2009). Land use changes intensify those effects (Coe et al. 2013b) by altering regional precipitation patterns, reducing local moisture (Coe et al. 2009, Langerwisch et al. 2013). In response to accumulation of greenhouse gases in the atmosphere and land use change, most global climate models project warmer-drier climate over the Amazon in the near future (Duffy et al. 2015). Despite overwhelming evidence that drought episodes may become more common with climate change, it is still unclear how Amazonian forests will respond to such events.

Current evidences suggest that Amazon can be highly vulnerable to increasing moisture stress (Huntingford et al. 2004, Cox et al. 2004, Phillips et al. 2009). The major information about the impact of drought in rainforest systems comes from large-scale monitoring (Chen et al. 2009, Balch 2014, Hao et al. 2014, AghaKouchak et al. 2015) or model simulations (Chen et al. 2009, Poulter et al. 2010, Galbraith et al. 2010, Steinkamp and Hickler 2013). These works present wide spatial extension, however, they lose spatial resolution. On the other hand, using a more mechanistic approach, a small number of works is focused in plant response to water stress (McDowell 2011, Adams et al. 2013). These works, comes from local field drought simulated experiment (Nepstad et al. 2002, 2007, Belk et al. 2007) or physiologic plant experiment for few species (Adams et al. 2013, O’Brien et al. 2014), and use controlled conditions. Studies that integrate process in different scales are still very limited, therefore, we are missing something fundamental: how environmental changes occurring in large scales, in time and space, are affecting the individual trees and the forest structure?

Across the forest canopy, there is a high variability of plants traits, which varies within and among species. If we can identify plant traits that determine the forest vulnerability/resistance to drought episodes, we would be able to understand the vulnerability and to predicting the effects of the environmental changes that are occurring in tropical forests. The concentration of chemical components in leaves is determined by the species composition (Townsend et al. 2008, Asner and Martin 2011) and by environmental factors (McGroddy et al. 2004, Asner and Martin 2011). Thus, we can also consider the concentration of those chemicals as important plant traits related to individual species performance in a particular environment. Some of these traits can improve the survival of species and the forest resistance to drought episodes. This is the case of non-structural carbohydrates (NSC). Regardless the mechanism, high concentration of NSC in leaves reduce the mortality provoked by water stress (McDowell 2011, McDowell et al. 2011, O'Brien et al. 2014).

One innovative and important recent methodological advance to understand vegetation patterns and ecosystem properties is the use of high resolution and spatial remote sensing (Curran 1989, Curran et al. 1992, Asner and Martin 2008, Asner et al. 2014), and the use of vegetation indices to indicate the distribution of plant functional traits, related with physiological processes. Vegetation indices have been developed to simplify and maximize the information content in high-dimensional spectral data. Thus, traditional vegetation indices are commonly used with different aims. NDVI and SR, for example, are sensitive to canopy greenness, which is related to canopy cover and leaf area (Myneni et al. 1995). Other indices, such as, photochemical reflectance index (PRI) are used to study the photosynthetic light-use efficiency, which is related to net primary production (Gamon et al. 1992). When theses indices are used from satellite images, they bring general information about the ecosystem, because each pixel contains an average of many trees. To high spatial resolution spectroscopy, in the other side, each value concerns parts of a tree and the response is measured in individual level. Thus, it is possible to study functional response to environmental change in individual level. If by one side, these measures present wide extension data, in the other side, they also have the same quality and precision of data collected in field surveys, linking local and regional processes. Thus, the improvement in spectral and spatial high resolution data has allowed researchers to link remote sensing data with field data (Kerr and Ostrovsky 2003).

Once high resolution spectroscopy is an individual level, the spectral metrics reflects two components. The first is a species specifics signature, mainly related with an phylogenetic structure (Asner and Martin 2011, Asner et al. 2014). The second one is related with environmental factors, such as climate, relief and soil type (Aerts 1997, Bonan et al. 2002, Wright et al. 2004). Thus, even than environmental changes will affect the forest canopy greenness (Xu et al. 2011), some species presents, naturally, higher canopy cover and leaf area than others. Consequently, NDVI index, for example, should reflect both components: the species variation and the environmental response. If this variability reflects different levels of tree vulnerability to stress events - such as drought periods -, it is possible to predict which trees will be more affected for future stress events. Using high spatial extension and resolution data from hyperspectral, we evaluate how the drought stress affected the forest hyperspectral metrics and ask if some hyperspectral traits affected the canopy vulnerability to new gaps formation. We work in the following main questions:

i- Can hyperspectral indices predict the new gaps formation?

ii- Can the drought stress generate directional changes in the canopy hyperspectral metrics?

iii- Can high-resolution hyperspectral metrics predict gaps formation after a drought period?

METHODS

Study area

This study was conducted in Madre de Dios, Peru, near to the border with the Brazilian State of Acre (Figure 1). This region of Amazon rainforest contains variable geology and topography (Räsänen et al. 1990) and, relatively, high soil fertility and quick tree turnover in comparison to the northeast on the Guyana Shield and to forests in Brazilian central Amazon (ter Steege et al. 2006). The entire data collection was performed by Carnegie Airborne Observatory (CAO) system (Asner 2007). The CAO system include a High-Fidelity Imaging Spectrometer (HiFIS) and a Light Detection and Ranging (LiDAR) scanner. The CAO system was flown over this region throw a time series in 2009, 2011 and 2013, in July/August. During these periods two forest blocks were flown by CAO and entirely mapped. We excluded of our data any area inside these blocks that presented evidence of anthropic disturbance - such as logging, mining or fire. At the end, these block presents similar size, 2486.74 and 2368.15 ha, and are distant approximately 102 km.

Inside each block there is a variability of topography, geology and stream distance not controlled in this study. Because this, the analysis were performed in subplots of, approximately, 440x440 meters (Figure 1). Thus, these subplots are big enough to include several trees/ but, at the same time, they are small enough to comprise a limited spatial variability (eg. soil type, relief, and stream distance). In each block we did random samples of subplots and repeat each analysis 500 times.

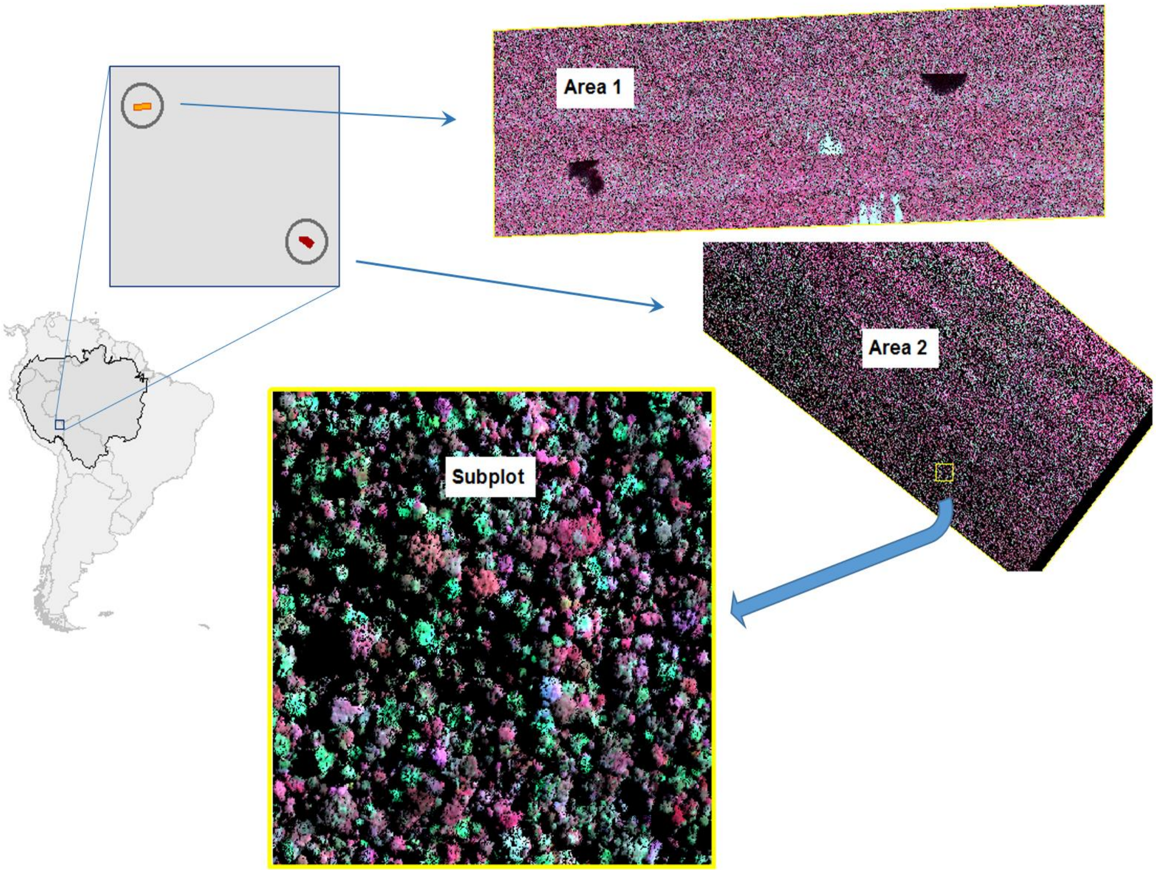

Figure 1. Study areas in Madre de Dios, Peru. In each area we sampled subplots of, approximately, 440x440 meters were the analysis were performed.

Drought episodes

Since 1990's, at least three severe and wide drought events were reported in Amazon forest. The first drought event occurred during the El Nino of 1997/98 which elevated the annual mortality rate of central Amazonian trees (Williamson et al. 2000) and turned wide Amazon forest areas susceptible to burning (Nepstad et al. 2004, Alencar and Vera 2006). The second was caused by the warming of the North Atlantic, in 2005, and extended over 37% of the Amazon (Lewis et al. 2011). The third drought event, in 2010, was considered a once-in-a-century event and, approximately, 57% of the Amazon experienced severe c/umulative water stress in this period (Lewis et al. 2011). This drought event also was associated with the warming of the North Atlantic.

To differentiate the intensity of drought between ours studies areas we used the Standard Precipitation Evapotranspiration Index (SPEI) from SPEI Global Drought Monitor (http://sac.csic.es/spei). This index is based in a multiscalar water balance approach, which allow us to examine wet and dry periods over different time scales (Vicente-Serrano et al. 2010). This index incorporates Potential Evapotranspiration from Thornthwaite's formula (Thornthwaite 1948), allowing to evaluate water supply and plant water demand (Vicente-Serrano et al. 2010). Thus, it is possible to quantify the duration, magnitude and spatial extent of the precipitation anomalies. Vegetation activity is reported to better respond to drought episodes in short SPEI time-scales in humid biomes (Vicente-Serrano et al. 2013). Thus, we choose a short, three months, SPEI time series, that present a good relationship with hydric stress in humid forests (Vicente-Serrano et al. 2013).

Mapping canopy profile and gaps

Canopy gaps are dynamic process caused by branch loss or tree fall that generate openings in forest canopies. These gaps are yield by several different process, reflecting since individual tree health and environmental variability until large scale environmental disturbances. The gap formation assessment in the forest canopy was done through Light Detection and Ranging (LiDAR) scanner. The LiDAR scanner emit pulses that penetrate the forest canopy, generating a three dimensional profile of the forest. Thus, it is possible estimate remotely the height of the trees in high-resolution images and low associated error (less than 0.15 m to 0.36 m vertical and horizontally spatial error) (Asner 2007). This technology has been employed to different uses including estimate carbon stock (Mascaro et al. 2011, Asner et al. 2012) and forest disturbance (Asner et al. 2013, Espírito-Santo et al. 2014). For more details see Asner (2013).

The distribution of size and frequency of canopy gaps can reflect the intensity of disturbance regimes (Asner et al. 2013). Here we evaluated the effect of a widespread drought event in large canopy failure, which is caused mainly by break of large branches and tree mortality, thus we focused just in large canopy gaps. We calculated the gap formation by the difference of the tree height between two years. Thus, we consider the gap formed during the periods 2009-2011 and 2011-2013. Because in this study we are interested in big tree failure or tree mortality, we focused our analysis only in gaps higher than 10 meters. We excluded small gaps (between 5 and 10 meters). The frequency of new gaps was calculated in relation to stable canopy areas, characterized by little variation in canopy high. We consider stable those areas that presented height variation under 5 meters (increase or decrease).

High-fidelity Imaging Spectrometer and Spectral indices

The High-Fidelity Imaging Spectrometer (HiFIS) provided spectroscopic images comprising the visible–near infrared spectral range between 385 and 1055.3 nm, with 72 bands at a bandwidth of 9.4 nm spectral resolution. This high resolution spectral image mosaic is obtained from an airborne vehicle that flown at an attitude of 2 km above ground, providing a ground sampling distance of 1.12 m pixel resolution, in an imaging array with 1500 cross-track pixels. Before to calculate the indices, a shade mask was generated and applied to eliminate any noisier in spectral images. Thus, spectral data was filtered to contain only well-lit and leafy vegetation pixels with NDVI higher than 0.5 and mean NIR reflectance (850–1016 nm) higher than 20%.

To enable monitoring changes in structure, physiology and composition of tropical rainforests, CAO has developed a methodology that integrates the HiFIS, high resolution LiDAR sensors and a Global Positioning System-Inertial Measurement Unit (GPS-IMU). This makes possible to detect and map chemical characteristics found in the leaves (Asner and Martin 2011, Asner et al. 2014), associating them with the forest structure profile in a high resolution images. Spectral metrics have been developed to simplify high-dimensional spectral data while maximizing the information content of the observations. The most commonly used spectral metric is the NDVI, this index is based on the difference in canopy reflectance at red (~680 nm) and near-IR (~800 nm) wavelengths (Eq. 1). This metric is related to the fraction of photosynthetically active radiation absorbed (fAPAR) by canopies. Thus, NDVI is reported to be sensitive to canopy greenness, which is a composite property representing canopy cover, leaf area, and canopy architecture (Myneni et al. 1995). The Simple Ratio index (SR) is calculated using the same wavebands than NDVI (Eq. 2). However, instead the normalized difference, it is calculated by the simple rate between red and near-IR. Thus, the SR–fAPAR relationships also have been used with multispectral data analysis.

$$NDVI = \frac{R_{800} - R_{680}}{R_{800} + R_{680}} \qquad \textbf{Eq. 1}$$

$$SR = \frac{R_{680}}{R_{800}} \qquad \textbf{Eq. 2}$$

Beyond the NDVI and SR, narrow-band sampling of the optical spectrum allows the development of metrics sensitive to other biochemical properties of canopies. This is the case of the Photochemical Reflectance Index (PRI). Based in the same general equation than NDVI, the PRI is calculated as a normalized difference reflectance index using a physiologically active green band (~531nm) and another physiologically insensitive green reference band(~570nm) (Eq. 3). This index has been confirmed in many studies as being strongly related to photosynthetic light-use efficiency (LUE) (Gamon et al. 1992), which is associated to a fundamental determinant of net primary productivity and ecosystem functioning (Gamon et al. 1992).

$$PRI = \frac{R_{531} - R_{570}}{R_{531} + R_{570}} \qquad \textbf{Eq. 3}$$

Data analyses

The analysis of remote sensing data in such large scale and resolution pose two important statistical issues. First, there are very large sample sizes generating extremely high statistical power. In these circumstances, very small effect-sizes could be considered significant even if they do not represent relevant practical effects (Murtaugh 2014). The second problem is the spatial autocorrelation since, naturally, geographically closest areas tend to have characteristics more similar than distant areas. On the other hand, this also creates an interesting opportunity. Usual statistical tests and traditional field inventories are based on the selection of sample units to constitute a sample that is assumed to be the realization of a statistical experiment. It is difficult to replicate the entire experiment so to have a better control of spatial effects to be included in the model. The large amount of spatial information present in remote sensing data allow to design a study that could include this possibility. To do so, we use each of 500 random 400x400 pixel subplots as sample for the statistical analysis. The subplots were selected for each area (Area 1 and Area 2) and period (2009-2011 and 2011-2013). After to filter the images by the shade mask, we excluded the subplots which presented less than 20% of valid cells. The random distribution of these subplots in the area allow to evaluate if the observed effects had any spatial dependence or variation within the study area.

A logistic regression was used to test if vegetation index was able to predict the formation of new gaps for each subplot. The vegetation indices values (NDVI, SR and PRI) were used to predict the new gaps formed in the subsequent years, for example, the NDVI values from 2009 were used to predict the new gaps generated between 2009 and 2011. We evaluated the success to predict the new gaps calculating the proportion of subplots where the new gaps were successfully predicted (p values< 0.05). The intensity of the effect was evaluated using Odds Ratio. The slope of the regression curve was used to indicate the trend of change. Once logistic regression coefficients are affected by the magnitude of vegetation indices, the indices were standardized before the logistic regression analysis. General Linear Models (GLM) were used to evaluate how the vegetation indices values and proportion of new gaps have changed over the years in each area. All the analysis was performed in R programs.

RESULTS

Evidence of Drought

In 2010 several areas in Amazon region have gone through an intense drought event (Lewis et al. 2011). This drought event was also recorded in our study areas, near to Madre de Dios, Peru. To both areas we have evidence of low precipitation regimes and hydric stress, mainly, in the second half of 2010. However, our studies areas were not affected in the same way. From 2009 to 2011 there were seven consecutive months with SPEI negative values in both areas, indicating a below average precipitation regime. During the drought period of drought, in the Area 2, SPEI had two months of values lower than -1, indicating moderate drought. In Area 1, SPEI values below -1 were observed in four months, representing a more intense and longer sequence of months with moderate drought than Area 2. In both areas, we observed lower SPEI values in 2009-2011 than in the following years, 2011-2013 (Figure 2). During the second period considered (2011-2013), the SPEI increased to both areas. In a general way, the SPEI values to Area 1 were positives and, apparently, were not affected by hydric stress from 2011 to 2013. On the other hand, Area 2 the SPEI values remained still low in 2011, even though higher than in 2010. Despite this, we will refer to the first period as a drought period and the subsequent years as a non-drought period.

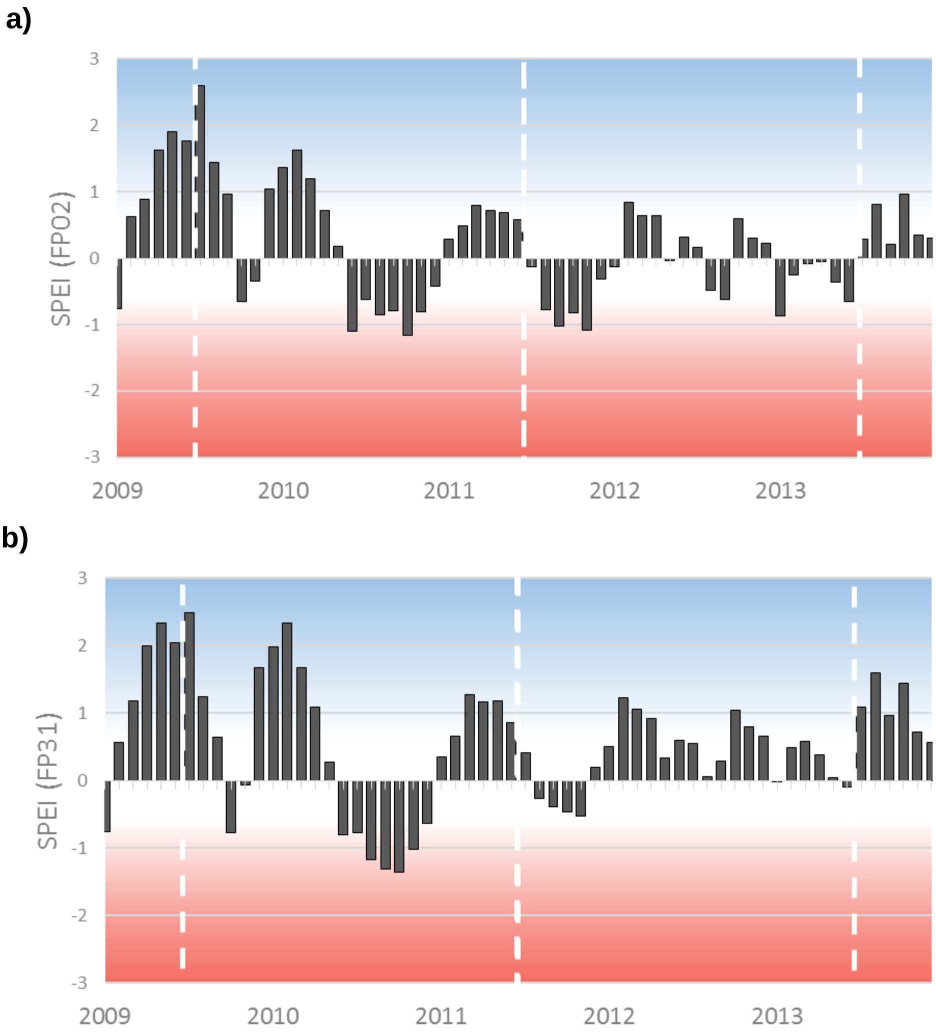

Figure 2. SPEI series during the study period, showing the presence of a drought regime in 2010 for both areas. Area 1 (b) presented a more intense drought regime than Area 2 (a) in 2009-2011 period.

Gap formation

The bellow average precipitation regime observed in 2010 affected the forest structure and the formation of large gaps. During the period of lower precipitation regimes, both areas presented similar gap formation. At about 9% of the area was observed formation of large gaps between 2009 and 2011. In Area 1 and 2 the proportion of gaps formed during this period was higher than the observed in the subsequent period (2011-2013) (Figure 3). Thus, the drought period generated a forest canopy less stable, probably for yield more tree fall and large branches break. During the period of higher precipitation regime, 2011-2013, Areas 1 presented the higher reduction of gap formation and just 4.7% of the area generated new gaps. In Area 2, despite the lower gap formation if compared to the previous years, this reduction was not so intense as observed in Area 1, and new gaps were formed in 7% of the Area 2 (Figure 3). This pattern is pretty consistent with the SPEI records. Despite the higher precipitation regimes, Area 2 yet was affected for hydric stress between 2011 and 2013, so the reduction of gap formation was not so intense as in Area 1 that, apparently, was not affected by hydric stress after 2011 (Figure 3).

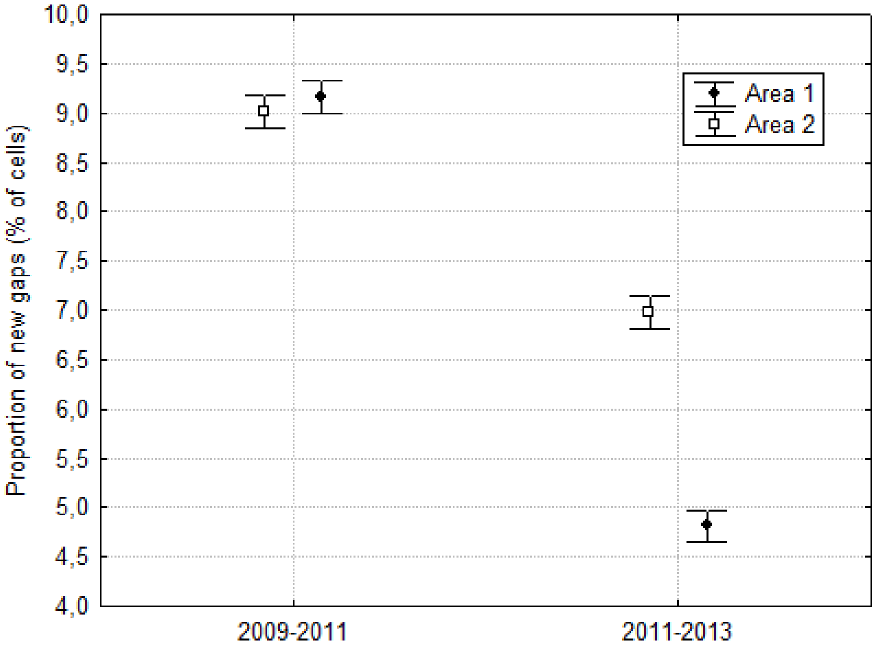

Figure 3. Proportion of new gaps formation in drought period (2009-2011) and in non-drought period (2011-2013).

How the areas changed after the drought?

The mean values of the spectral indices of the canopy forest changed over the years. At the beginning of the records, 2009, average of NDVI values were 0.857 and 0.876, to Area 1 and 2 respectively. This values decreased and in 2011 the average of NDVI values had declined to 0.856 and 0.863, respectively, to Areas 1 and 2 (Figure 4). However, while this trend of reduction was maintained to Area 1, in 2013 was observed an increase in average of NDVI to the Area 2. The SR index presented a trend of change over the years, similar to observed to NDVI. Both areas presented a decreased of SR between 2009 and 2011. As in NDVI, Area 2 showed more significant reductions of SR values than Area 1. In 2013 the average values of SR increased in Area 2, while Area 1 values remained stable compared to 2011. While the NDVI and SR showed a similar trend over the years, the average of PRI in subplots varied in a different way. The PRI index also had their values reduced after the drought period, however, the trend of strong reduction was maintained in the years of non-drought regardless the area considered (Figure 4).

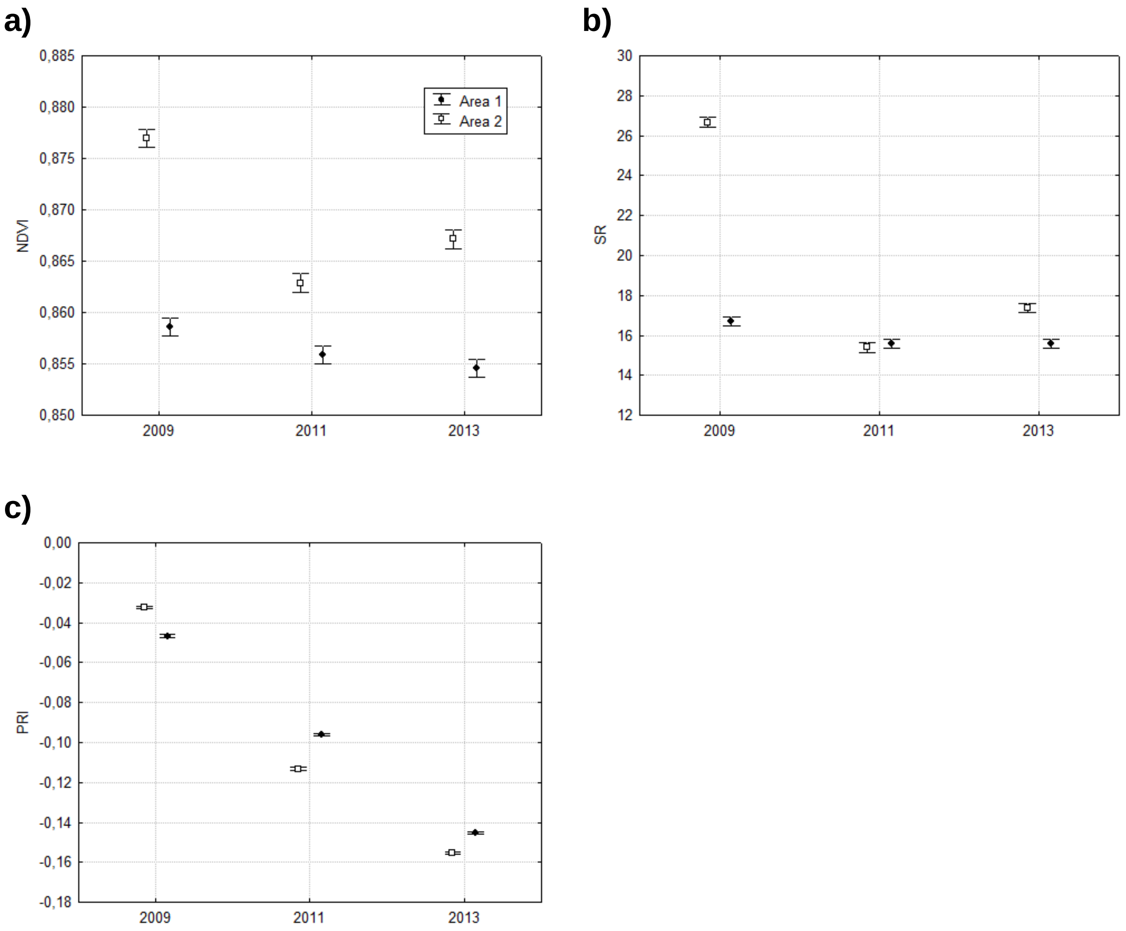

Figure 4. Changes in spectral index values over the years, from 2009 to 2013, for the areas 1 and 2.

Spectral determinants (NDVI, SR and PRI)

The lower NDVI values the higher the probability of new gaps formation in subsequent years (Figure 5). This pattern was observed to both periods, however, in 2011-2013 period it was stronger. Similarly to NDVI, lower values of SR also generated more gaps in the subsequent years. However, to SR this negative relation was weaker and close to zero. To PRI, in the other hand, higher values of PRI generated more gap formation in the subsequent years (Figure 5). This relationship was stronger in 2011-2013 period than in 2009-2011 period.

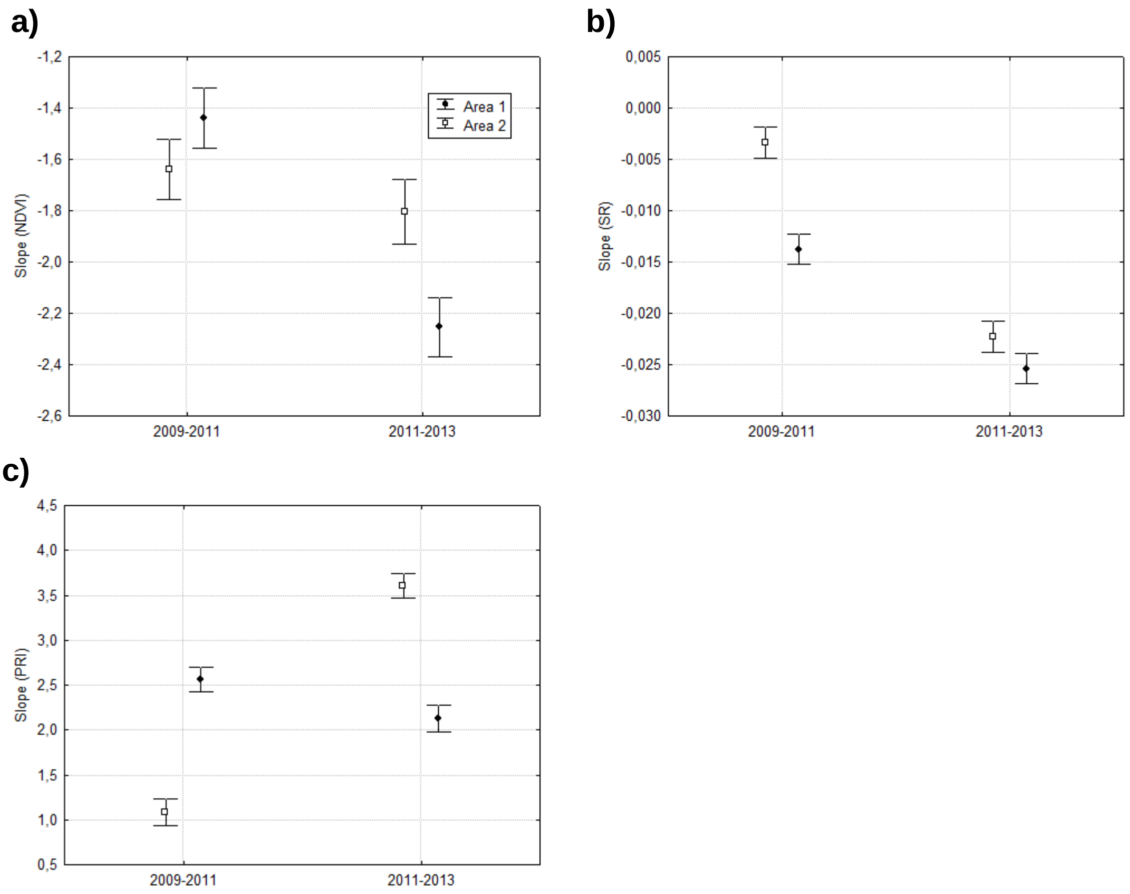

Figure 5. Slope values of logistic regression to predict the gaps formation using spectral indices, (a) NDVI, (b) SR and (c) PRI.

Among all the indices considered, NDVI was the index that more predicted successfully the new gaps formation. For both areas, NDVI values in 2009 was able to predict the new gaps formed during the drought (2009-2011) in more than 90% of the subplots (Table1). NDVI values in 2011 were able to predict more than 90% of the new gaps formed in 2013 in Area 2, however, to Area 1 this prediction ability decreased to 80%. In general, in 2009, lower values of NDVI generated 3.97 and 4.97 more gaps than higher values of NDVI to Area 1 and 2, respectively. These values increased to the followed period and low NDVIs generated 5.48 and 9.08 more gaps than high NDVI values. Despite the logistic regressions using SR to predict new gaps are frequently significative, in general, the Odds Rate of the logistic regression was near to 1, what means that in average the rate that high values of SR produced gaps was not so different than the observed to low values. Thus, SR was not a good predictor to new gaps formation for either areas considered.

PRI also was able to predict the new gaps yield in subsequent years, however, it had the highest rates of no significant logistic regressions (Table 1). While in Area 1 the first period had higher rates of significant relationships, in Area 2 during the second period it was possible to predict with greater success the formation of new gaps. These cases also were those with the highest values of odds ratio. The odds ratio values generated from the logistic regression based on PRI varied more than the other indices (Table 1). This indicates that PRI was more dependent on the spatial variation existing in each area.

Table 1. Odds ratio analysis to evaluate the new gaps prediction capacity of spectral indices.

Index	Area	Period	Significative regressions (%)	Odds Ratio (Median)	Interquartile range (25%-75%)
NDVI	Area 1	2009-2011	0.92	3.97	2.56-6.98
NDVI	Area 1	2011-2013	0.94	9.08	4.66-17.06
NDVI	Area 2	2009-2011	0.90	4.97	2.40-10.21
NDVI	Area 2	2011-2013	0.80	5.48	2.53-12.17
SR	Area 1	2009-2011	0.88	1.01	1.01-1.02
SR	Area 1	2011-2013	0.92	1.02	1.02-1.04
SR	Area 2	2009-2011	0.79	1.00	1.00-1.01
SR	Area 2	2011-2013	0.83	1.03	1.01-1.04
PRI	Area 1	2009-2011	0.84	11.12	4.65-23.32
PRI	Area 1	2011-2013	0.73	8.12	4.61-16.43
PRI	Area 2	2009-2011	0.73	3.11	2.03-4.84
PRI	Area 2	2011-2013	0.90	19.33	8.53-41.15

DISCUSSION

Our results show that vegetation indexes were able to predict the new gaps. NDVI and PRI successfully predict the new gaps formed in subsequent years, while SR was a poor predictor. Lower values of NDVI increased the probability of large gaps formation in forward years. This means that trees with lower leaf area and canopy cover were more prone to die or lose much of branches after a drought event. NDVI is sensitive to canopy greenness and represent canopy cover, leaf area, and canopy architecture (Gamon et al. 1995, Myneni et al. 1995), has been frequently used to measure the forest health (Ismail et al. 2007, Wang et al. 2010) and to evaluate environmental impact, including drought stress (Breshears et al. 2005, Mänd et al. 2010). Despite SR is based in the same wavelength and should present the same properties than NDVI (Gamon et al. 1992), it was surprisingly less efficient.

The use of vegetation indexes derived from high resolution data allow to interpret variations at individual plant level (Nagendra and Rocchini 2008). At this level, the NDVI is a surrogate of lower leaf area and canopy cover which, in turn, are related to the tree stress and physiological changes (Ismail et al. 2007). The tree health can be related, for example, with herbivory, competition and disease. Thus, drought is acting as another stress factor, which can be determining tree failure, mortality or damage of large branches. Here, low values of NDVI were more prone to produce future gaps allowing to infer that characteristics of trees associated with low NDVI are probably related with tree vulnerability. Thus, in the after stress situation (drought stress), canopy failure in low NDVI values is more intense but follows the general pattern observed in non-drought periods. While NDVI has been used as source of information about vegetation health, it is not useful to identify the causes of vegetation stress. This happens because many factors such as land cover change, plant disease, and flooding can affect NDVI similarly to that observed by drought (Wang et al. 2010, AghaKouchak et al. 2015). Thus low values of NDVI can be understand as a general indicator of crown vulnerability, not only to drought stress.

While NDVI is related to canopy structure and biomass, PRI responds mainly to changes in physiological and structural properties of canopy (Gamon et al. 1992, 1995). PRI is indicated to be related to photo protective mechanisms (Penuelas et al. 1995, Filella et al. 1996) , photosynthetic efficiency (Nichol et al. 2000) and chlorophyll fluorescence (Buddenbaum 2015). PRI has been demonstrated to be sensitivity under water stress situations (Suárez et al. 2008, Mänd et al. 2010), what is also related to chlorophyll fluorescence under water stress conditions (Evain 2004, Dobrowski et al. 2005). However, similarly, the observed with NDVI, we do not have support to affirm that PRI determines the specific tree sensitivity to drought stress. Instead, our results show that these vegetation indices are indicating the vulnerability of canopy to large gaps formation, which is intensified under the drought stress situation. Despite to be able to predict the gap formation in the following years, PRI values are frequently related with a short term response (Naumann et al. 2008, 2010). We could observe this in our study, the average of PRI were highly associated with the cumulative SPEI observed in each area during a six-month period before the measures.

NDVI index saturates at medium to high LAI, becoming insensitive to changes in canopy structure when LAI is large (Gamon et al. 1995, Pontailler et al. 2003). This is the case of our study areas, in Amazon forest, and can be intensified in high resolution studies (Asner 2000). PRI, in the other side, is showed to do not present saturation problem at medium to high LAI (Mänd et al. 2010). The saturation in NDVI are expected to lower its ability to discriminate among some environmental conditions and lowering its ability to predict tree failure in relation to PRI. Nevertheless, in our study, these both vegetation indices presented similar ability to predict the future gaps. The efficiency of NDVI may suggest that the thresholds for discriminate areas with drought or differences in crown vulnerability are below the level of NDVI saturation. Thus, we do not have evidence of NDVI saturation plays any important role in the prediction, and our results indicate that NDVI and PRI values were sensitive enough to predict the new gaps formed.

It was observed an expressive increase of new gap formation between 2009 and 2011 period when the Amazon region suffered an intense drought event (Lewis et al. 2011). Additionally, using SPEI index from NOOA satellite data, we were able to quantify the drought intensity, comparing with the subsequent period. A high proportion of new large gaps formed during the drought period was similar between both areas and was consistent with the results obtained in other studies. In a regional scale, there is evidence that this drought affected the carbon cycle and reduced the net primary production in Amazon forest (Potter et al. 2011). However, here we went beyond and quantify the gap formation in a non-drought period. Hydric stress is clearly a good predictor of gap formation, especially because the release of such conditions in the non-drought period in Area 1 (2011-13) is followed by a 50% reduction of the gaps. It is important to notice that we did not considered small gaps, which is also an important component of the gaps formed and follow a different pattern (Asner et al. 2013). Our results show that the gap formation can double in a period without drought to a period with drought stress. As a consequence, successive drought periods, similar to the 2010 event, could change the hole forest structure and reduce the carbon storage of the rainforests.

Since gap formation increased, it is an evidence that the drought period affected the structure and function of forest. Lower NDVIs were prone to yield new gaps, however, the drought stress also reduced the average NDVI values of the areas. First, this can indicate that these forests became more vulnerable to future stress, after the drought stress. Secondly, at the end of the measures in 2013, none index has returned to 2009 values, what can indicate that the forest recovery could happen in a slow way. Our results support the idea that this more intense and frequent drought period could yield a multiplicative impact in the forest, and not simply an additive impact. Additionally, an important question that remains is how individual species variation and resistance affect the whole process. Theoretically, recurrent drought events will weaken species resistance in increase tree fall in a non-random sequence. Resistant plant species that bear some traits of drought resistance such as high concentration of non-structural carbohydrates (McDowell 2011, Adams et al. 2013, O'Brien et al. 2014) and water storage mechanisms (Borchert 1994) will be less affected by drought. This line of reasoning suggests that the variation in response observed in our analysis will decrease in time, as the reaming plant community become increasingly functionally homogeneous.

Large spatial variation in logistic coefficient is consistent to the dependency of all vegetation indices on the individual species characteristics. Our design allows to estimate coefficients that are directly related to the variation of composition in subplot plant communities. This compositional variation may explain why coefficients vary both between and within areas. This may highlight that the actual value that represent a threshold for predict individual tree failure vary between plant species. This is highly expected as individual trees are affected by biotic (e.g. soil, climate, topography) and abiotic factors (e.g. floristic composition) that yield different structural, architectural and physiological characteristics and affect tree fall (Denslow 1987, Young and Hubbell 1991, Lertzman et al. 1996) . Obviously, this pose a very important limitation on prediction of gaps in a forest composed by more than 300 different species. Additionally, our design reduced the environmental variation in our analysis. Because our study areas were so large, the environmental variation related to river distance, soil type and relief are present as source of variability. Thus, our design reduced this variation once the analysis was performed in smaller subplots, large enough to contain many trees but with a far smaller environmental variability.

CONCLUSION

Drought episodes are predicted to become more intense, frequent and widely distributed in Amazon forest in the coming decades (Duffy et al. 2015). Many studies have hypothesized about a positive feedback system of forest dieback where drought events change the forest structure, composition and process, becoming the forest more susceptible to future droughts. However, the mechanisms of how this forest dieback should happen is not clear. High resolution and high spectral data allow us the monitoring in regional scale using high quality and precision data and is an important tool to improve the knowledge. Our study provides an initial step to understand the mechanisms and patterns behind the forest vulnerability to drought stress. Additionally, our study relights the fundamental importance of the temporal series to evaluate the drought impact, differentiating of other sources of impact and the natural environmental variability.

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CHAPTER 3:

INFRASTRUCTURE INDUCTION OF DEFORESTATION IN INDIGENOUS LANDS IN BRAZILIAN AMAZON: HOW TO BUFFER ITS IMPACTS?[1]

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[1] Proposed journal target: Global Change Biology

ABSTRACT

Indigenous Lands are created to ensuring the conservation of environmental resources necessary for indigenous well-being, and the physical and cultural reproduction of indigenous cultures. However, in Amazon this importance goes beyond the indigenous people. They also provide important environmental services to non-indigenous people, such as carbon stock, regional hydric regulation and protection of the mega biodiversity in Amazon region. Despite this, Indigenous Lands have been threat by expansion of deforestation caused by economic activities. Among these activities, roads are pointed to be one of the strongest drivers of deforestation in Amazon. Since roads generates collateral deforestation, they also can affect the deforestation observed inside Indigenous Lands. Here, using the current roads network in Amazon, we evaluate how the current roads have impacted the deforestation inside Indigenous Lands. We also propose an objective methodology of buffer zoning to reduce the potential impact of roads planned to be built by Brazilian government in the future. The closer the roads, the higher the deforestation inside Indigenous Lands. Distance to roads and cities and reserve size is able to explain the half the deforestation observed in indigenous Amazon. Indigenous Lands located less than 20 km from roads are much more likely to present high rates of deforestation. We propose that the road network expansion consider the current information of roads impacts and use an effective planning to minimize the impact of future building roads in Amazonian indigenous populations.

Keywords: Indigenous people; roads; Amazon forest; buffer zones; collateral deforestation

INTRODUCTION

Tropical forests are essential to the physical and cultural survival of many traditional communities whose depends on forests for their livelihood. Indigenous people, for example, depends on natural resources for food, medicine and cultural and religious rituals (Schwartzman *et al.*, 2013). As a consequence, their way of life, traditions and cultural identity are closely related to the territory for them occupied and the availability of species of plants and animals presents in each region (Gadgil *et al.*, 1993). Due to the strong connection with the environment, these people are more vulnerable to environmental impacts that degrade and threaten the ecosystem where they live (Green & Raygorodetsky, 2010; Schwartzman *et al.*, 2013). For instance, indigenous peoples in Brazil are known to be exposed to a variety of territorial threats including land invasions by loggers, miners and farmers, and illegal deforestation (Nepstad *et al.*, 2006; Le Tourneau, 2015). Similarly the observed in Brazil, threats to the indigenous rights and territories is a frequent problem in others countries of Latin American and continents (Psacharopoul & Patrinos, 1994; Walker *et al.*, 2006). Regardless the region considered, all the activities that impact these territories rarely generate benefit to themselves and leave behind social conflicts, environmental degradation, diseases and violence.

Amazon regions in Brazil are known to holds a huge biodiversity of plants and animals and home hundreds of indigenous ethnics groups, including some of the world’s last uncontacted peoples living in voluntary isolation (Kesler & Walker, 2015). Despite exposed to so many threats, in Amazon region these territories are responsible to provide important environmental services. Indigenous Lands (IL) were recorded to present much lower deforestation rates than the observed in their surroundings, similar to the observed in conservation unities (Nepstad *et al.*, 2006). Indigenous Lands and Conservation Unities represents 54% of the remains forest area in Brazilian Amazon, but stocking 56% of the forest carbon storage of the region (Soares-Filho *et al.*, 2010). In addition, indigenous areas act as barriers to the expansion of deforestation (Nepstad *et al.*, 2006; Soares-Filho *et al.*, 2010), and was considered increasingly important to conserve Amazonian biodiversity (Fagundes *et al.*, 2016). Indigenous lands are reported to occupy one-fifth of the pristine forests of the Brazilian Amazon, which is five times the area designated as parks (Nepstad *et al.*, 2006). Despite to play a similar conservation role than other protected areas, sustainable use or full protection conservation unities, these territories have a different dynamic and should consider specific management strategies. In addition to be related to the cultural history of the indigenous groups that inhabit each area (e.g. some ethics groups use fire to hunting), the legislation that regulate the use and protection of conservation unities is different than that regulate the ILs. Because this, the effective management of the indigenous territories involve a complex legal and social environment, and could be considered an important challenge to advance conservation policies in Brazilian Amazon.

Brazilian Indigenous Lands are created to ensure that territories traditionally occupied are kept by the indigenous people, ensuring the "conservation of environmental resources necessary for their well-being and for their physical and cultural reproduction, according to their uses, customs and traditions" (Brazilian Federal Constitution - Brazil, 1988). For these reserves play their role and guarantee the physical and cultural survival of these communities, it is essential that they maintain healthy ecosystems and viable populations of plants and animals. One of the main driver of local fragmentation and habitat loss yield by deforestation and construction of infrastructure projects (e.g. roads, cities and hydroelectric dams) (Peres, 2001; Geist & Lambin, 2002). Many studies have shown that high levels of fragmentation and habitat loss can create a local extinction threshold, from which there is an abrupt increase the probability of local extinction species (With & Crist, 1995; Pardini *et al.*, 2010; Swift & Hannon, 2010;

Rueda *et al.*, 2013). Thus, areas that reached this habitat loss threshold would be losing a considerable proportion of its fauna and flora. Many studies have shown the existence of these extinction threshold to several species and biological groups (Swift & Hannon, 2010), including large mammals that play important role to food security of many indigenous groups.

Roads and their concomitant traffic affect directly the biotic and the abiotic components of the ecosystem by killing animals, changing the dynamics of populations of plants and animals, promoting habitat fragmentation, facilitating the introduction exotic species, and changing the availability of resources, such as water and light (Mader, 1984; Forman & Alexander, 1998; Laurance *et al.*, 2004a). Indirectly*,* roads play a fundamental role to Amazonian forest deforestation and degradation once they make the forest accessible to human occupation (Laurance *et al.*, 2009; Barber *et al.*, 2014) and support activities such as, agriculture, oil production, mining, logging or hydroelectric operations (Caro *et al.*, 2014). In fact, roads were reported to be one of the strongest correlates of deforestation (Laurance *et al.*, 2002). Thus, in Amazon forest roads could be the main way for collateral deforestation as an indirect effect of altering environmental and social-economic environment, increasing human colonization, illegal logging and forest burns (Nepstad *et al.*, 2001; Laurance *et al.*, 2009; Ewers, 2014). Collateral deforestation is a spatial explicit “distance effect”, that could affect deforestation rates even many kilometers away from the existing impact (Chomitz & Gray, 1996).

Despite they are less impacted than unprotected areas, protected areas are also very affected by the construction of roads (Barber *et al.*, 2014). In an attempt to reconcile regional land-use change and environmental conservation, buffer zones can be established around protected areas to order the land use change of human activities that could impact these areas directly (Margules, 2000; Alexandre *et al.*, 2010). In Brazil, for example, Conservation Units National System (SNUC) has established buffer zones of transition areas that should minimize negative impacts in conservation unities. In a specific resolution (by National Environment Council Resolution – CONAMA), it was defined a 10km buffer zone around conservation unities, where any human activity that could affect the biota should be licensed. However, this buffer size is arbitrary and was not based in studies that considered the biomes particularities and negative impacts of human activities in protected areas (Alexandre *et al.*, 2010). Additionally, these buffer zones just were established to conservation unities, once indigenous lands are regulated by a different legislation, there is no study or law that aim to regulate and minimize the social and environmental impacts of human activities expansion around IL.

Considering the expansion of infrastructure that has taken place in the Brazilian Amazon, this study aims to evaluate how the current net of roads have affected the deforestation in indigenous territories in Brazilian Amazon. Using this knowledge, we expect: i- evaluate how the current net of roads existent in brazilian Amazon have impacted the Indigenous Lands, and ii- contribute to planning future roads focusing in reduce the impact of roads expansion in Indigenous Lands, using an objective methodology of buffer zoning.

METHODS

Indígenous Lands and Infrastructure Maps

The Indigenous Lands maps were obtained from Socio-Environmental Institute (ISA; http://ti.socioambiental.org/). We choose this database because it is quite complete to Indigenous Lands in Brazilian Amazon and has been constantly updated. From the 382 Indigenous Lands listed in Brazilian Legal Amazon, 48 presented less than 20% of its area characterized like forest. The official program of government deforestation monitoring just consider information about Amazon forest deforestation (see explanation in the next topic, *Deforestation in Indigenous Lands*). Because this, we excluded the reserves that had little proportional forest area, in this case, less than 20% of forest area.

All the infrastructure information was obtained from official databases. To evaluate the effects of infrastructure projects in deforestation inside Indigenous Lands, we considered the following maps and source of information. Small and large hydropower dam were obtained from National Electric Energy Agency (ANEEL - http://sigel.aneel.gov.br/sigel.html). Small hydroelectric plans are defined by an installed power from 1.1 to 30 MW (megawatts), in this case theirs dams can reach a maximum of 3 km². Large hydroelectric plans, in the other side, have more than 30 MW of installed capacity and dams areas of more than 3 km^2. The most famous example is Belo Monte hydroelectric power plant; it is the largest project that Brazilian government has built in Amazonia. Belo Monte has become famous, among others reasons, due the conflict generated with local indigenous populations who will be affected by this project. In this case, the installed capacity will be 11000 MW and an area around 516 km² will be flooded.

Cities and large rivers informations were obtained from Brazilian Institute of Geography and Statistics (IBGE - www.ibge.gov.br/), the governmental agency responsible for geosciences and social, demographic and economic statistics. Harbors map was downloaded from Transport Ministry website (http://www.transportes.gov.br). Roads map was obtained from National Department of Transport Infrastructure (DNIT - www.dnit.gov.br/) and comprises all the official roads existing in Amazon region, including paved and not paved roads.

We verified an inconsistence in some official data. Some roads, for example, presented an inconsistence between the shape on the map and the observed shape on satellite maps. This was not a common problem, but when we verified this kind of inconsistence we manually corrected the maps. When possible, large hydroeletric dams were also verified. However, verify most of the infrastructure maps by satellite images was not possible. Thus, we assumed as correct the information we could not check.

The distance of each infrastructure project to the Indigenous Lands was calculated using Euclidean distance. Here we consider the smaller distance of each individual IL to the closer piece of the infrastructure projects. Because the areas between TIs vary considerably, we decided to make this calculation using the distance of the infrastructure projects to the edge of ILs, and not the centroid. Thus, it was possible to evaluate roads that crossed or should cross the ILs in the future.

Deforestation in Indigenous Lands

The deforestation information was obtained from PRODES project (Amazon Deforestation Monitoring Project), developed by INPE (National Institute for Space Research - http://www.obt.inpe.br/prodes). This is a project of satellites deforestation monitoring by clearcutting in the Amazon forest. PRODES uses satellite images from Landsat class with 20-30 meters spatial resolution and frequency of 16 days. Estimates of PRODES is deemed reliable by national and international scientists and the annual rates of deforestation yield by this project are used by the Brazilian government for the establishment of public policies. For more information about PRODES products see http://www.obt.inpe.br/prodes.

The proportional area of deforestation was calculated for each Indigenous Land. We excluded from this calculation any area classified like non forest area, such as water and other vegetation types. We assumed clouds areas as forest area. Thus, the proportional deforestation was calculated using the area deforested divided by the area originally expected to be forest (forest and deforested areas). We define as highly deforested areas the ILs with threshold of deforestation higher than 30%.

We also consider the outside proportional deforestation to evaluate if ILs are playing an active role in the inhibition of deforestation observed outside indigenous areas. To each IL, the outside areas were stablished in buffer zones of 25 kilometers to the edge of the ILs. Since our aim was to compare deforestation observed in each territory and the outside unprotected area, we removed of these buffers all the surrounding areas that were also indigenous lands and other conservation unities.

Statistical Analysis

To found the best model that explain the proportional deforestation observed inside Indigenous Lands, we performed a series of linear multiple regression. The linear multiple regression estimation computes the relationship between a set of independent predictors and a dependent variable according to a linear model (Chatterjee & Hadi, 2006). This allows us to finding the best fitting relationship between the observed deforestation values and the set of predictors.

To control the lack of normality in our data and improve the residual distributions, we implement logistic transformation in distance to roads and area of ILs.

One of the assumptions of these models is the independency between the predictors (Chatterjee & Hadi, 2006). Because this, we performed a tolerance analysis to evaluated the redundancy among the predictors proposed as drivers of deforestation in Amazon forest (Chatterjee & Hadi, 2006). Low values of tolerance of a variable indicate a more redundant contribution to the regression. The tolerance of the predictors is defined as 1 minus the squared multiple correlation of this variable against the other independent variables in a multiple regression equation. We evaluated the tolerance of different sets of variables to select variables to a final model with low redundancy.

Considering the need to differentiate areas with high deforestation, we transform the proportional deforestation variable inside ILs into a binary variable: high and low deforestation. Several studies claim that even 30% of habitat loss is enough to affect the persistence of several species, affecting the maintenance of local species (Swift & Hannon, 2010). Thus, we have established as high deforestation all ILs which had more than 30% of its forest area cleaned. Using a logistic regression model (Hosmer & Lemeshow, 1989) we predict the probability of ILs be highly deforested according the distance from roads.

To evaluate how the individual ILs deforestation is contributing to the deforestation observed throughout the indigenous Amazon, we compute the cumulative deforestation sorting these areas based on distance to the nearest road. However, it is possible that the high contribution to general deforestation of initial classes of distance is because there is more ILs near roads than distant. Thus, we designed a Monte Carlo simulation test (Manly, 1991) to evaluate whether the high contribution to deforestation of ILs close to roads were explained by road distance and not by chance.

RESULTS

A high proportion of Indigenous Lands are located near to roads in Brazilian Amazon. Of the 329 IL considered, 67 are crossed by some kind of official roads and others 15 are located less than 1 km from a road. This means that 25% of the IL in Amazon are located until 1km of any official roads. There is a high variability in deforestation observed in ILs located near some roads. In a general way, we observed from very low to very high deforestation rates in IL located near to a road. Almost always, ILs located far from roads were little deforested. However, there is one outlier easily identified where three IL very close each other presented high deforestation rates clearly related with a harbor presence (Supplementary Material). Because they are characterized as a case of outliers, they were removed from the analysis in this work.

Regardless the distance to roads or the others infrastructure projects, the areas occupied by indigenous people presented much lower deforestation than its surrounding. While all the areas delimited as IL had 1.48% of these area deforested, the deforestation observed outside the areas that surrounding ILs were 29.8%. This difference indicates a high pressure of these outside areas to the inside ILs forests. Considering the difference between outside and inside deforestation, in average, deforestation observed outside ILs was 15% higher than that observed inside.

We observed a high redundancy among predictors when all the considered infrastructure projects were evaluated. The tolerance among predictor of deforestation varies from 0.6 to 0.8. Since low tolerance values indicate high correlation with others variables, the set of all variable initially considered were quite redundant. Hydroeletric dams, small and large, and roads presented the lower tolerance values indicating possible collinearity among those predictors

(Table 1). In the second set of predictors we exclude hydroeletric dams and large rivers variables. The removal of these variables from the model resulted in higher tolerance values for all remaining predictors. The tolerance values of the second set of variables (cities, harbors and roads) were larger then 0.89 (Table 1), indicating low collinearity. Due to the low collinearity in this set of variables it was possible developing the subsequent analysis.

Table 1. Redundancy among predictor of deforestation inside Indigenous Lands. The first set of variables, where all predictors were considered, presented low values of tolerance, indicating high redundancy. This redundancy decrease expressively in the second set of variable.

	Toleran.	**R-square**	**Partial Cor.**	**Semipart Cor.**
First set				
Small hydr. dams	0,674	0,326	0,111	0,103
Big hydr. dams	0,604	0,396	-0,163	-0,152
Large rivers	0,833	0,167	0,017	0,016
Cities	0,836	0,164	-0,162	-0,150
Harbors	0,781	0,219	-0,180	-0,168
Log (Roads)	0,644	0,356	-0,187	-0,175
Second set				
Cities	0,900	0,100	-0,138	-0,130
Harbors	0,927	0,073	-0,187	-0,177
Log (Roads)	0,886	0,114	-0,263	-0,254

The distance to roads, harbors and cities was able to explain more than 17% of deforestation observed inside Indigenous Lands (Table 2). In all these cases, the shorter the distance to these infrastructure projects, the higher deforestation rates observed in ILs. However, while roads, harbors and cities have been able to explain 17% of the deforestation observed inside the ILs, the impact of these infrastructure projects outside the ILs was much bigger (Table 2). These three projects explained 30.5% of the deforestation observed outside ILs (Table 2). This difference reinforces the importance of ILs in reducing forest clearing. Although these variables are directly related to the increase of deforestation, the impact observed is smaller inside than outside of the indigenous territories. Among these variables, harbors played a minor role in predict deforestation inside ILs and it had a non-significant effect in predict outside deforestation. Because this we exclude harbors of the final model.

Table 2. Multiple regression models to explain inside and outside Indigenous Lands deforestation. Stars indicate significant p values.

Model	Estimate	SE	$T_{(315)}$	P	Variance explained
Outside IL					**30,452%**
Intersept	0,055	0,141	0,394	0,693	
Log (Roads)	-0,778	0,096	-8,101	<0,001*	
Cities	-0,012	0,004	-3,312	0,001*	
Harbors	0,001	0,001	1,130	0,259	
Inside IL					**17,068%**
Intersept	-0,408	0,187	-2,185	0,030*	
Log (Roads)	-0,637	0,137	-4,663	<0,001*	
Cities	-0,023	0,008	-3,073	0,002*	
Harbors	-0,006	0,002	-2,916	0,004	

Considering only the deforestation inside IL, it is possible to add variables directly related to the territory. The most important possible predictor in this case is the area, since larger areas may work passively like a buffer of collateral deforestation in remote areas inside IL. In this case, after remove harbor and include IL area, the redundancy among variables become even smaller, and all the tolerance raise to values higher than 0.91 (Table 3). The explanation power of the model also has increased. As a result, road and cities distances and the IL size were able to explain almost 50% of the deforestation observed inside ILs of the Brazilian Amazon (47.1% of explained variance, Table 4). In the same way that roads and cities proximity generate higher rates of deforestation, smaller ILs have higher proportional area deforested. Thus, the smaller the ILs, the more vulnerable to deforestation they are.

Table 3 – Redundancy analysis among predictor of deforestation inside ILs.

	Toleran.	R-square	Partial Cor.	Semipart Cor.
D_CapCidad	0,907	0,093	-0,093	-0,075
Log(Area)	0,968	0,032	-0,536	-0,508
Log(Roads)	0,933	0,067	-0,301	-0,253

Table 4 – Final multiple regression models that better explained the deforestation inside Indigenous Lands. This model was able to explain 47.1% of variance of the deforestation in ILs.

Variable	**Estimate**	**SE**	**$T_{(315)}$**	**p**
Intersept	1,440	0,260	5,536	<0,001*
Log(Roads)	-0,718	0,131	-5,467	<0,001*
Log(Area)	-0,425	0,047	-8,964	<0,001*
Cities	-0,028	0,007	-4,062	<0,001*

Indigenous Lands very close to or crossed by roads had, approximately, 20% of chance to being highly deforested (more than 30% of the deforested area – Figure 1). On Amazon, 60.9% of the ILs are located until 30 km to a road and

52.6% are closer than 20 km. In these ILs located until 20 km to a road (more than a half of the ILs), the logistic regression predicts, approximately, 10% of chance to being highly deforested. Except for the previously described outliers (Supplementary Material), no indigenous territory distant more than 30 km to a road, presents more than 30% of deforested area (Figure 2). Even when we added the territories considered outliers and excluded from the analysis, the highly deforested areas correspond only to 1.5% of the ILs located more than 30km from any road. In the other side, when we consider the ILs located less than 30 km to any road, 16.6% of these lands were highly deforested. When we consider the distance of 20km, the proportion of those ILs considered highly deforested rises to 17.9%.

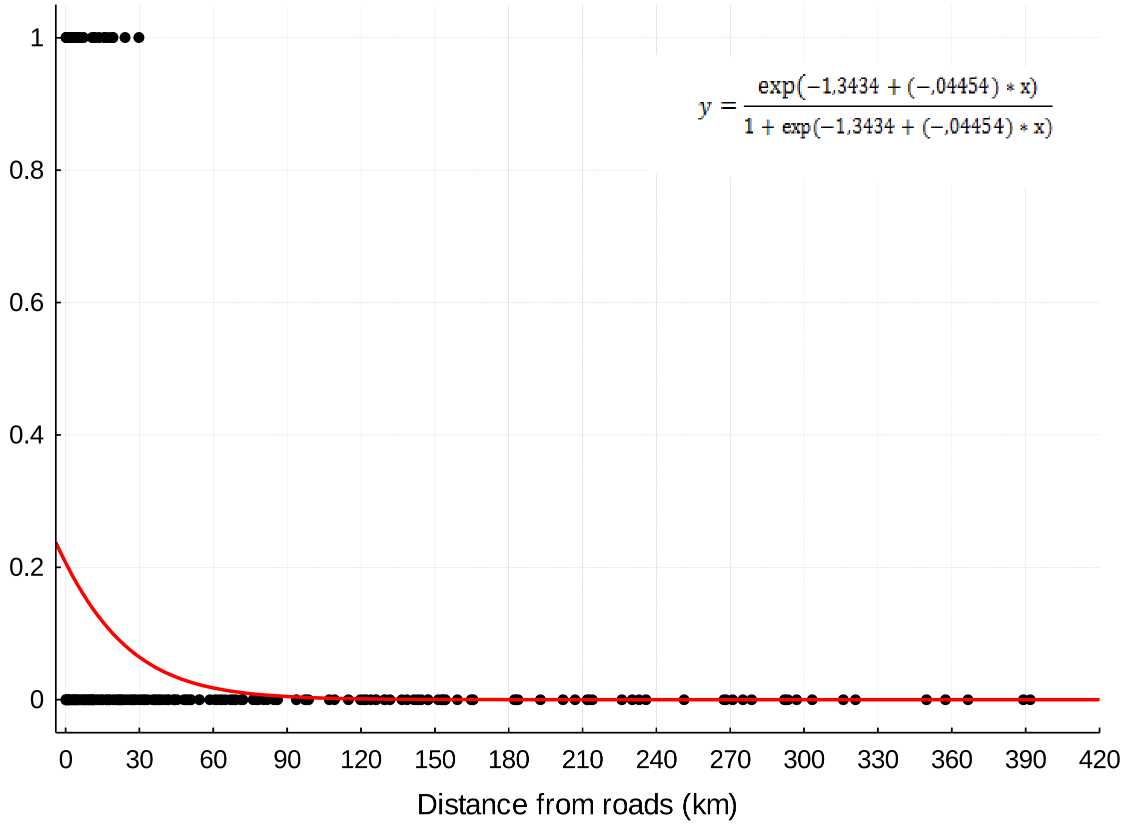

Figure 1. Logistic regression to predict high deforestation in ILs based in distance to roads.

Although 52.6% of ILs are located less than 20 km from a road, the deforestation in these ILs were responsible for 88% of deforested area observed in indigenous territories of Amazon (Figure 2). When we consider all lands located less than 30 km, this values rises to 93% of all deforestation observed in the Brazilian ILs. These values are much higher than would be expected by a null model in which the deforestation would be randomly distributed in the ILs through Amazon (Figure 2). If the deforestation had occurred randomly, the territories located 20 km from a road should have 28% less deforestation. Indigenous Lands located until 30 km from a road contributes in a marked way to the total deforestation on ILs. At distances higher than 30 km the contribution of new deforestation decreases sharply. The contribution of the territories located more than 50km to any highway is only 5% of the total deforestation.

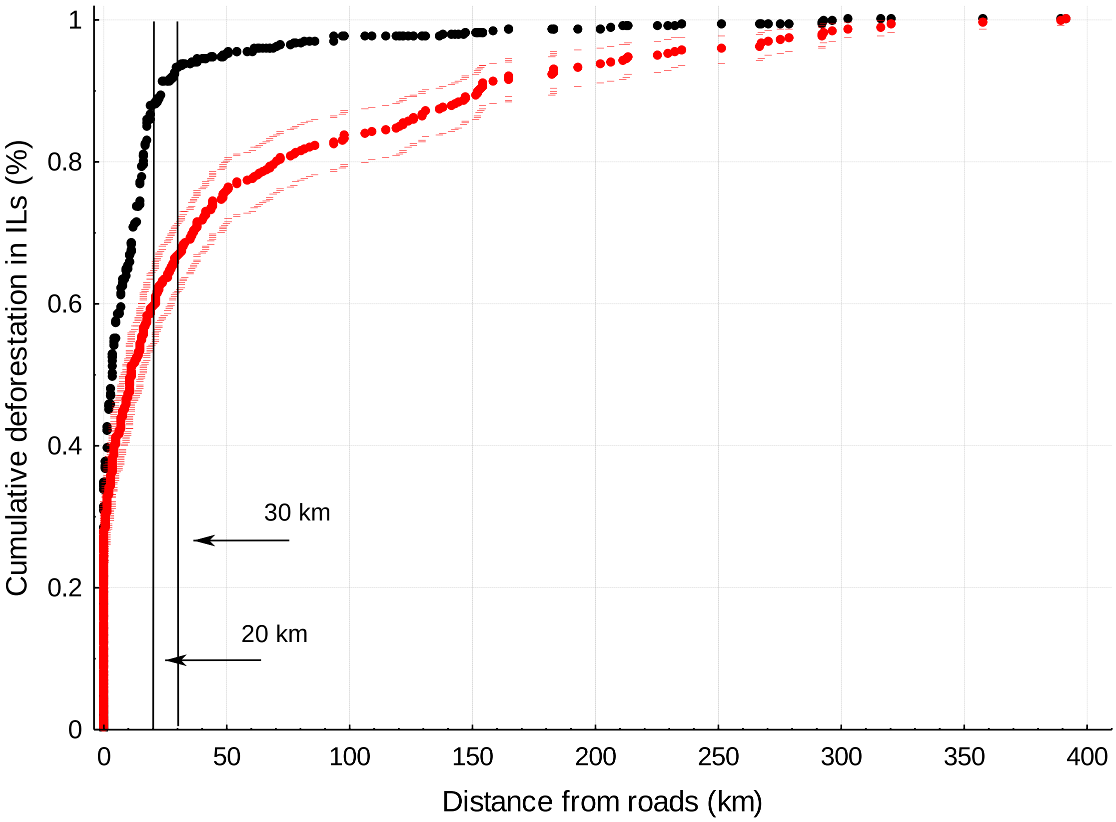

Figure 2. Cumulative deforestation of ILs in Amazon based in distance to roads. ILs located closer to roads have higher proportional deforestation and contribute more to deforestation observed in indigenous Amazon. This cumulative deforestation observed

(black points) are much higher than the randomization analysis (red points) and are not included in its confidence interval (red dash).

According to official government data, there are 491 sections of roads planned to be built in the future. If these roads were built, 64 territories would be crossed by highways. Considering the 67 ILs that are already crossed by existing highways, 38% of the Brazilian Amazon ILs would be crossed by future roads (Figure 3). In addition to the roads that would cross the ILs, the new roads would also reduce the distance of many indigenous territories and the nearest highway. In this case, 78.1% of ILs would be located less than 20 km from the nearest road in the future, an increase of 25.5% to the current reality (Figure 3). There are currently 30 indigenous territories demarcated where it is recognized the presence of isolated indigenous populations (people that do not have done no contact with non-indigenous people). Of these, 9 ILs are already crossed by some kind of roads. If the planned roads were built, 22 of the 30 ILs with recorded isolated peoples could become crossed by the new roads. Moreover, all ILs with isolated peoples would be located less than 20 km from a road. Currently, 53.3% of the territories with isolated populations records are located less than 20 km to roads.

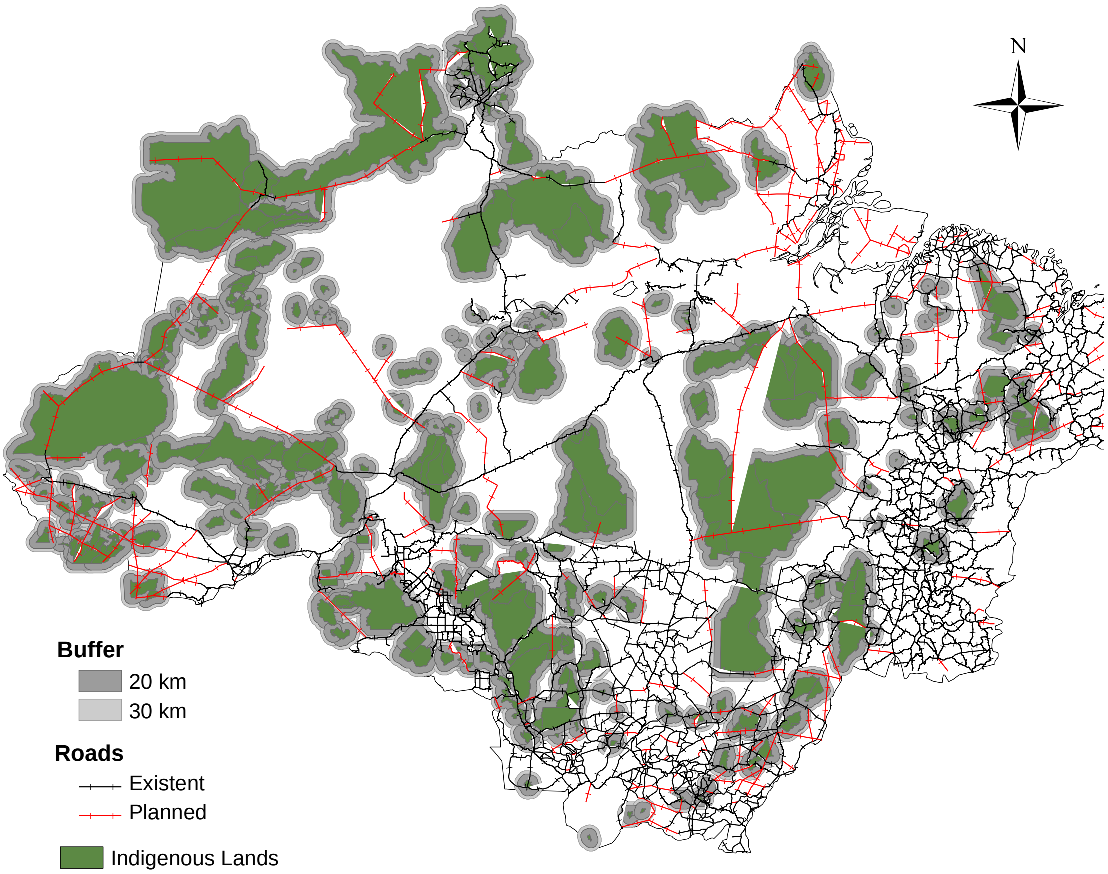

Figure 3. Indigenous Lands (green) with buffer zones (grey) in Amazon forest and all the existent (black lines) and planned (red lines) roads.

DISCUSSION

The roads network increased significantly the deforestation inside Indigenous Lands, indicating an active role of these infrastructure projects to generate collateral deforestation in indigenous protected lands. Many studies have described the major impact of roads in Amazon deforestation (Nepstad *et al.*, 2001; Laurance *et al.*, 2002; Barber *et al.*, 2014; Ewers, 2014). In general, these studies are based on the increase deforestation near the roads, and are focused in road impact in a general way. Here, we used a different approach, where our mainly focus was on the protected areas affected by roads rather than the road effects in regional deforestation. As a consequence, we were able to quantify the potential and collateral impact on ILs of individual roads and use this knowledge to plan future road network expansion. Based on our results, we propose that buffer zones around ILs should be stablished to reduce the effects of roads network expansion. Many roads are planned by federal government to be built in future, supporting agriculture expansion and others infrastructure projects (including many hydroelectric plans). If this projects were implemented, the use of buffer zones around ILs could be a powerful tool to reduce the collateral impacts in this reserves.

There is a large amount of literature shown the several impacts of roads expansion in Amazon forest (Laurance *et al.*, 2004b; Monteiro Júnior *et al.*, 2013; Ewers, 2014; Whitworth *et al.*, 2015). Our results support previous studies that found higher deforestation pressure in protected areas located near roads (Nepstad *et al.*, 2001; Barber *et al.*, 2014). Besides this, ILs located near to roads also presented high variability on deforestation rates, suggesting that other local processes have a decisive role in deforestation observed inside these lands. Otherwise, the deforestation inside indigenous lands remains far below that observed outside. The importance of ILs to Amazon conservation was already reported and shown that these lands have played a decisive role to reduce the

Amazon deforestation (Nepstad *et al.*, 2006; Ricketts *et al.*, 2010; Barber *et al.*, 2014). This protection role is even more valuable since most of the ILs are located in the arc of deforestation and suffer a more intense anthropic surrounding pressures than other protected areas – not located in arc of deforestation (Nepstad *et al.*, 2006).

Our best model of Indigenous Lands deforestation is based on distance to roads and cities distance with the addition of the ILs size. This model was able to explain a large part of the deforestation observed inside ILs in Brazilian Amazon. Smaller ILs presented higher proportional rates of deforestation, indicating that these areas are more vulnerable to roads and possibly to the construction of others infrastructures. Smaller protected areas also are naturally more vulnerable to loss of species (Margules, 2000; Parks *et al.*, 2002; Alexandre *et al.*, 2010). Besides to be more vulnerable to surrounded land use changes pressure, the high deforestation in small IL can also limit the remained forest, resulting ultimately in areas not enough to support population of some species (Fahrig, 2001; Vance *et al.*, 2003; Carvalho *et al.*, 2009) . Thus, the closer to roads and cities and the smaller the size, the more vulnerable the indigenous lands are.

Since building roads enhances local deforestation, the consequent increase of fragmentation and habitat loss can rises the probability of population extinction of many species (Fahrig, 2003; With & Crist, 2011). Critical thresholds of habitat loss, above which we should expect a suddenly species richness loss, can vary between species and taxonomic groups, however, to many species, even 30% of habitat loss have been reported to yield species extinctions (Swift & Hannon, 2010). Roads and its collateral deforestation also increase the probability of successful alien plant invasions into conservation areas (Monteiro Júnior *et al.*, 2013; Silvério *et al.*, 2013). Since some species of plants and animal are essential to food, medicine and cultural and religious rituals of the indigenous cultures (Schwartzman *et al.*, 2013), the extinction or even the abundance decrease of these species can threat, among others, the food security and human health of these human populations (Foley *et al.*, 2014).

Beyond to reduce the local biodiversity, roads and deforestation can also increase local diseases incidence, change the local climate, and increase the incidence of forest fires (Nepstad *et al.*, 2001; Cochrane & Laurance, 2008; Coe *et al.*, 2013; Hahn *et al.*, 2014). During drought periods, high incidence of fires in Amazon forest have been associated to increase of respiratory diseases in region (Smith *et al.*, 2014), although no study has been done directly with indigenous peoples. Similarly to fire incidence, many infectious diseases are associated to climate and land use change (Patz *et al.*, 2004, 2008). This is the case of malaria, a significant public health threat in the Brazilian Amazon; the risk of this disease increase with the presence of roads, fires and selective logging (Hahn *et al.*, 2014). The expansion of infrastructure listed as planned by the government, can intensify deforestation in ILs that are still inaccessible.

Roads distance is highly associated to other infrastructure projects, including small and large hydroelectric dams. Despite the causal relationship is not always clear and constant, in general, the implementation of roads is associated to construction of hydroelectric dams and expansion of cities. Both hydroelectric dams are built in areas already accessible by roads or new roads can be implemented to enable the construction of hydroelectric plants. The association between different infrastructure projects can generate synergistic consequences, for example, to building hydroelectric dams and roads it is necessary bring people from others parts of the country. However, when these projects are finished several unexpected impacts can emerge, including: irregular and disorganized settlements; water usage conflicts; intensification of predatory timber extraction; loss of fishing zones downstream from the dam; and destruction of social bonds of the local indigenous communities (Soito & Freitas, 2011). In sum, roads facilitate land invasion and the enticement of indigenous people to economic activities, associated to the extraction of natural resources.

The future of the Brazilian Amazon depends on the planned and consistent government policies, based on effective management of human activities and their impacts on landscapes (Vandermeer & Perfecto, 2007; Caro *et al.*, 2014). The Brazilian government has invested in monitoring policies of deforestation, degradation and forest fires. These actions led to a significant reduction in Amazon deforestation rates in the past decade and has been pointed as an example of success in the Brazilian environmental policies (Arima *et al.*, 2014). On the other hand, the Brazilian government has also invested in the expansion of settlements and construction of infrastructure projects in the Amazon, which promotes the occupation and regional deforestation (Nepstad *et al.*, 2002). However, the incentive in the Amazon occupation and changing of Forest Code have caused the weakening of these environmental policies (Soares-Filho *et al.*, 2014). So even though the reduction of deforestation in the Amazon can be seen as a success in the Brazilian environmental policy, this success is very fragile and has been threatened by current government policies.

Roads are associated to the promotion of regional economic development, however, while positive impacts are always considered in the decision to build new roads, the economic costs caused by negative social and environmental impacts are not understood in all its complexity. In this study, we quantified the potential negative impacts of deforestation in ILs generated by the roads. We propose a tool to measure the potential impact of roads and supports effective planning to minimize the impact of future building roads in indigenous populations. Building roads more distant to ILs has great effect in reducing impact in these territories, however, in cases where roads have to be built next to the ILs, economic costs for environmental compensation, investment in public health and increase control against the land invasion should be a responsibility of the governments and the companies responsible for the infrastructure projects.

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SUPPLEMENTARY MATERIAL:

In a general way, we observed low rates of deforestation in IL located far from roads, however, there was one exception to this general pattern. There is one particular case where three IL far away from highways have high deforestation rates. These three ILs are located very close to each other so, probably, they are under a similar social and cultural pressure very specific. Despite being far away from highways, these IL are located next to a harbor and close to two cities, which are probably the main drivers of deforestation observed in these ILs (Figure A1). Because they are characterized as a case of outliers, they were removed from some analysis in this work (Figure A1).

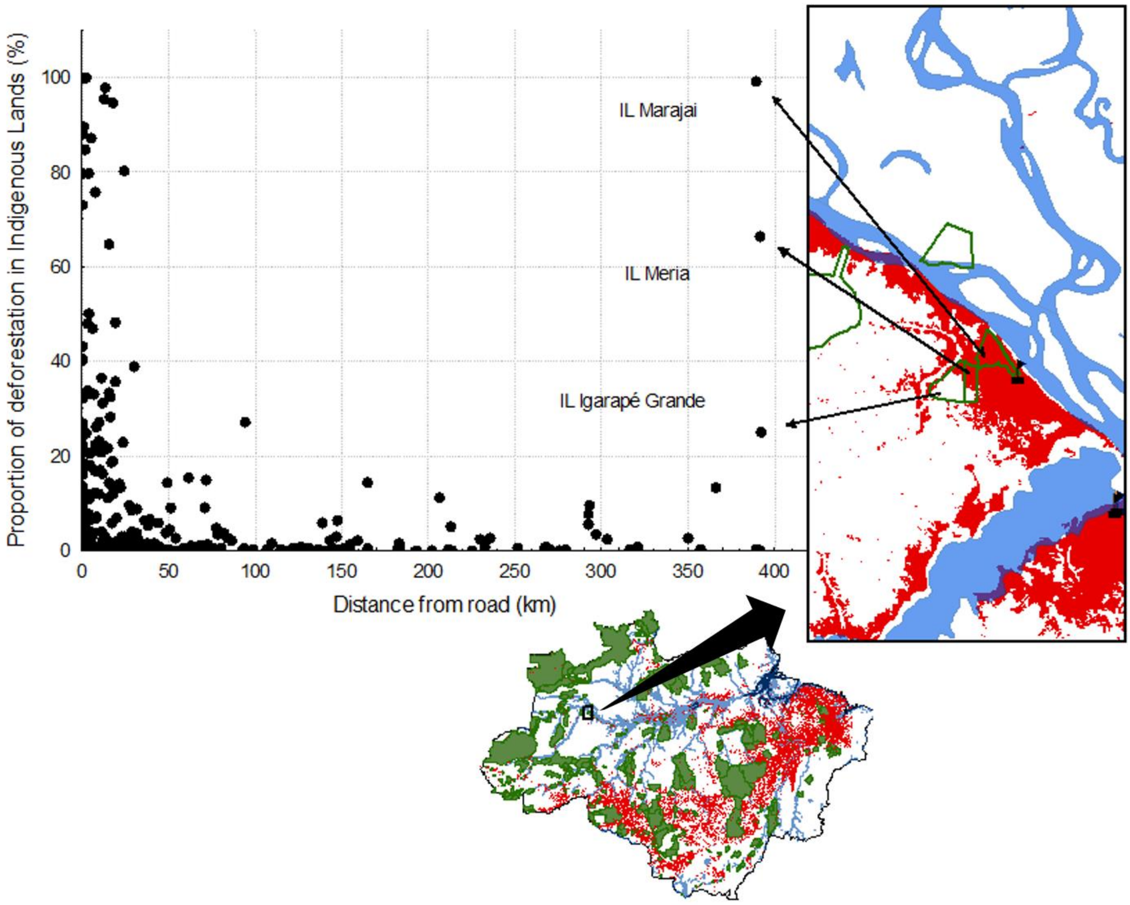


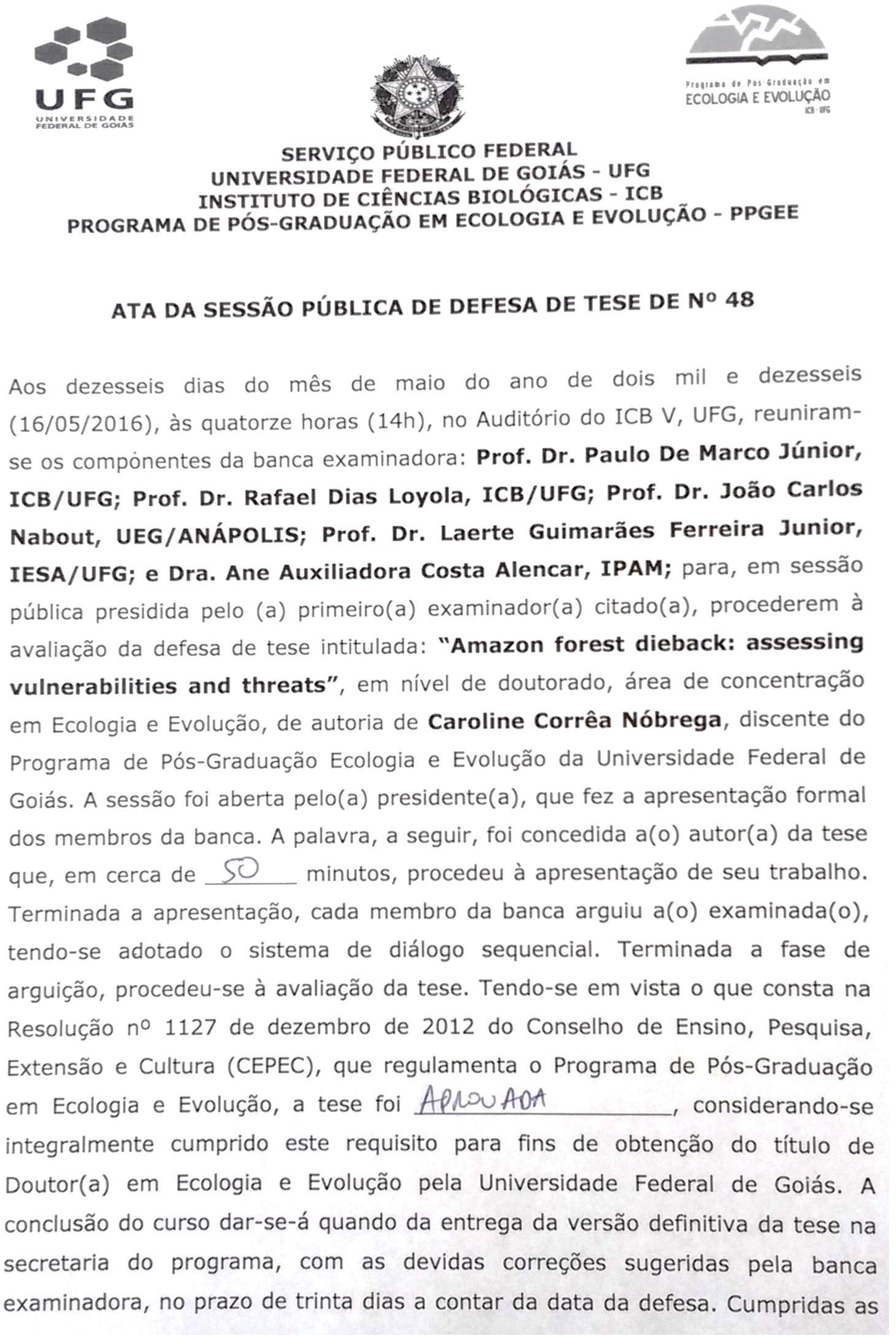
Figure A1. Relationship between distance of roads and deforestation observed in ILs. Three ILs were characterized as outliers and removed from the analysis.

CONCLUSÃO GERAL

Apesar de ser a maior floresta tropical úmido do mundo e possuir uma das maiores biodiversidades do planeta, a floresta amazônica também tem sido apontada como um dos ecossistemas mais vulneráveis aos impactos antrópicos. Mudança de uso da terra e alterações no clima global podem ameaçar o futuro da floresta. Entretanto, assim como grande parte da diversidade amazônica, também é desconhecida de que forma os diferentes impactos tem afetado a Amazônia. Para identificar os mecanismos e padrões gerais de como diferentes fontes de impacto estão degradando a floresta, é importante o desenvolvimento de abordagens comparativas que integraram múltiplas facetas da diversidade. Com base em uma abordagem experimental de longo prazo, encontramos evidência de como diferentes regimes de fogo podem influenciar a seleção de espécies com base em características filogenéticas e funcionais. Tais mudanças representam uma importante força de mudança e de empobrecimento (funcional e filogeneticamente) dessas comunidades arbóreas florestais. Entender como comunidades de árvores responder ao fogo regimes é essencial para prever os impactos e desenvolver os esforços de conservação contra a expansão incêndios na floresta amazônica.

Uma das principais causas da expansão de incêndios florestais na Amazônia é devido a intensificação de episódios de seca nas últimas decadas. Infelizmente, tais episódios são preditos para se tornarem mais intensos, frequentes e amplamente distribuídos na Amazônia com as mudanças climáticas globais. Nesse estudo, usando ferramentas de alta resolução espacial, LiDAR e hiperespectral, nós pudemos avaliar como um importante evento de seca afetou a estrutura e função da floresta. Assim, nós fornecemos um importante passo para compreender os mecanismos e padrões por trás da vulnerabilidade da floresta ao estresse hídrico.

O desmatamento é uma importante fonte de impacto, que além de impactar diretamente a floresta, também é capaz de intensificar a incidência de incêndios e episódios de seca. Desta forma, políticas efetivas de combate ao desmatamento são fundamentais para assegurar a integridade florestal. Estradas estão associados à promoção do desenvolvimento económico regional, no entanto, enquanto os impactos positivos são sempre considerados na decisão de construir novas estradas, os custos económicos causados por impactos sociais e ambientais negativos não são entendidos em toda a sua complexidade. Neste estudo, nós quantificamos os potenciais impactos negativos do desmatamento em TIs gerados pelas estradas e propomos uma ferramenta para medir o impacto potencial de estradas que dá suporte a um planejamento eficaz para minimizar o impacto da construção de estradas futuras em populações indígenas. Nós observamos que a construção de estradas mais distantes para TIs tem grande efeito na redução do impacto nesses territórios. No entanto, nos casos em que as estradas precisam ser construídas próximas às TIs, também é possível quantificar seus impactos potenciais tornando possível estimar os custos económicos para a compensação ambiental e social para populações locais afetadas.



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ATA DA SESSÃO PÚBLICA DE DEFESA DE TESE DE Nº 48

Aos dezesseis dias do mês de maio do ano de dois mil e dezesseis (16/05/2016), às quatorze horas (14h), no Auditório do ICB V, UFG, reuniram-se os componentes da banca examinadora: **Prof. Dr. Paulo De Marco Júnior, ICB/UFG; Prof. Dr. Rafael Dias Loyola, ICB/UFG; Prof. Dr. João Carlos Nabout, UEG/ANÁPOLIS; Prof. Dr. Laerte Guimarães Ferreira Junior, IESA/UFG; e Dra. Ane Auxiliadora Costa Alencar, IPAM;** para, em sessão pública presidida pelo (a) primeiro(a) examinador(a) citado(a), procederem à avaliação da defesa de tese intitulada: **"Amazon forest dieback: assessing vulnerabilities and threats"**, em nível de doutorado, área de concentração em Ecologia e Evolução, de autoria de **Caroline Corrêa Nóbrega**, discente do Programa de Pós-Graduação Ecologia e Evolução da Universidade Federal de Goiás. A sessão foi aberta pelo(a) presidente(a), que fez a apresentação formal dos membros da banca. A palavra, a seguir, foi concedida a(o) autor(a) da tese que, em cerca de 50 minutos, procedeu à apresentação de seu trabalho. Terminada a apresentação, cada membro da banca arguiu a(o) examinada(o), tendo-se adotado o sistema de diálogo sequencial. Terminada a fase de arguição, procedeu-se à avaliação da tese. Tendo-se em vista o que consta na Resolução nº 1127 de dezembro de 2012 do Conselho de Ensino, Pesquisa, Extensão e Cultura (CEPEC), que regulamenta o Programa de Pós-Graduação em Ecologia e Evolução, a tese foi APROVADA, considerando-se integralmente cumprido este requisito para fins de obtenção do título de Doutor(a) em Ecologia e Evolução pela Universidade Federal de Goiás. A conclusão do curso dar-se-á quando da entrega da versão definitiva da tese na secretaria do programa, com as devidas correções sugeridas pela banca examinadora, no prazo de trinta dias a contar da data da defesa. Cumpridas as formalidades de pauta, às 18 h e 0 min., encerrou-se a sessão de defesa e, para constar, eu, Suely Ana Ribeiro, secretária executiva da Universidade Federal de Goiás - UFG, lavrei a presente ata que, após lida e aprovada, será assinada pelos membros da banca examinadora em três vias de igual teor.

Paulo De Marco Júnior

Prof. Dr. Paulo De Marco Júnior
ICB/UFG

Prof. Dr. Rafael Dias Loyola
ICB/UFG

Prof. Dr. João Carlos Nabout
UEG/ANÁPOLIS

Laerte G

Prof. Dr. Laerte Guimarães Ferreira Junior
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Ane Auxiliadora Alencar

Dra. Ane Auxiliadora Costa Alencar
IPAM

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