UNIVERSIDADE FEDERAL DE PERNAMBUCO
CENTRO DE BIOCIÊNCIAS
PROGRAMA DE PÓS-GRADUAÇÃO EM BIOLOGIA VEGETAL

LUCAS DE FARIAS CORDEIRO SIQUEIRA ALENCAR

DINÂMICA DA COBERTURA FLORESTAL E SUSTENTABILIDADE NA CAATINGA

Recife
2023

LUCAS DE FARIAS CORDEIRO SIQUEIRA ALENCAR

DINÂMICA DA COBERTURA FLORESTAL E SUSTENTABILIDADE NA CAATINGA

Tese apresentada ao Programa de Pós-Graduação em Biologia Vegetal da Universidade Federal de Pernambuco, como requisito para obtenção do título de doutor. Área de concentração: Ecologia e Conservação

Orientador: Dr. Felipe Pimentel Lopes de Melo

Coorientador: Dr. Luke Parry

Recife

2023

Dados Internacionais de Catalogação na Publicação (CIP) de acordo com ISBD

Alencar, Lucas de Farias Cordeiro Siqueira.
Dinâmica da cobertura florestal e sustentabilidade na Caatinga/ Lucas de Farias Cordeiro Siqueira Alencar– 2023.

104 f. : il., fig., tab.

Orientador: Filipe Pimentel Lopes de Melo
Coorientador: Luke Parry

Tese (doutorado) – Universidade Federal de Pernambuco. Centro de Biociências. Programa de Pós-Graduação em Biologia Vegetal, Recife, 2023.
Inclui referências e apêndices.

1. Recursos naturais 2. Ecologia agrícola 3. Desenvolvimento rural I. Melo, Filipe Pimentel Lopes de (orient.) II. Parry, Luke (coorient.) III. Título

333.7 CDD (22.ed.) UFPE/CB – 2023 -198

Elaborado por Elaine C Barroso - CRB-4/1728

LUCAS DE FARIAS CORDEIRO SIQUEIRA ALENCAR

DINÂMICA DA COBERTURA FLORESTAL E SUSTENTABILIDADE NA CAATINGA

Tese apresentada ao Programa de Pós-Graduação em Biologia Vegetal da Universidade Federal de Pernambuco, como requisito para a obtenção do título de Doutor em Biologia Vegetal.

Aprovada em 12/07/2023

BANCA EXAMINADORA

____
Prof. Dr. Felipe Pimentel Lopes De Melo (Orientador)
Universidade Federal de Pernambuco

____
Prof. Dr. Rodrigo Felipe Rodrigues Do Carmo (Examinador Externo)
Universidade Federal Rural de Pernambuco

____
Dr. Aldrin Martin Perez-Marin (Examinador Externo)
Instituto Nacional do Semiárido

____
Dra. Camila Linhares De Rezende (Examinadora Externa)
Fundação Brasileira para o Desenvolvimento Sustentável

____
Dr. Henrique Fernandes De Magalhães (Examinador Externo)
Universidade Federal de Pernambuco

à Claudio Cordeiro, meu avô, que em seu último ato de amor aqui na Terra, (re)uniu a família e me deu forças para concluir essa tese

AGRADECIMENTOS

Eu não seria nada e nem teria feito essa tese sem um incontável número de pessoas que passaram pela minha vida antes, durante e depois desse processo se concluir. Seria muito bom ter tempo e energia o bastante para dizer para cada um o quanto me ajudaram, como me ajudaram e o pedacinho de cada um que levo comigo. Mas a relatividade do tempo faz com que tudo se torne urgente e o trabalho penoso quando os prazos chegam e a vida, no seu ritmo próprio, atravessa você.

Gostaria de começar agradecendo à família, ou às famílias, que formam a base que a gente precisa para decolar. À minha mãe e minha irmã que sempre foram meu porto seguro e sempre me recebem de coração aberto e com muito amor quando retorno das minhas andanças. Ao meu pai, grande amigo que me deu o espírito livre para explorar o mundo. E à Pâmella, a aventureira que resolveu se juntar a mim nessa grande jornada que é viver.

Muitos outros membros da família também tem um espacinho na bagagem que carrego. Dentre eles meus avós, que me ensinaram muito e ajudaram de tantas formas que nem sei dizer. À família de Pâmella que hoje carrego no coração. E aos tios, tias, primas e primos, maravilhosos no seu jeito de ser, que também me amam e me querem bem.

Não vou dedicar espaço aos amigos porque nenhum presta. Se prestassem, não seriam meus amigos e eu seria somente um grande vazio de carne e osso. Mas alguma coisa precisa ser dita. Então gostaria de agradecer a todas essas pessoas que sempre estiveram comigo, na alegria ou na tristeza, na bebida ou na salada, e que muito me ajudaram com toda felicidade que proporcionam. E também aos novos que entraram no barco e que viraram amigos para a vida.

Tem um único amigo que presta e esse merece bastante espaço. Isolado, numa casinha no meio das montanhas, no meio da pandemia, longe de tudo e de todos, foi um fiel companheiro e um ser humaninho especial. Chico Mendes, não o seringueiro revolucionário, mas o cachorro que me ensinou que é possível amar um animal, ocupa um espaço enorme no meu coração e a ele agradeço enormemente pela companhia nos dias frios e sem fim da Serra Gaúcha. Uma pena ele não poder ler isso, mas eu sinto que ele sente. E isso é o suficiente.

E para não dizer que não falei das flores, gostaria de agradecer aos meus orientadores que, difíceis e espinhosos como são, me tornaram uma versão muito melhor do que eu era antes. Também não posso deixar de agradecer pelas várias portas e janelas que os dois abriram para mim, mostrando o mundo de possibilidades que é a vida acadêmica e como navegar nisso tudo. Espero ter feito por merecer o tempo, a dedicação e acima de tudo a amizade que eles me proporcionaram.

Por fim, gostaria de agradecer às instituições que permitiram a execução da tese (UFPE e PPGBV). Mesmo com todas suas limitações, me fizeram um ser resiliente e capaz de produzir ciência de qualidade em meio às adversidades. Também agradeço à CAPES e à FACEPE pelas bolsas de estudo que me permitiram concluir minha tese de forma mais ou menos são e mais ou menos salvo.

“Isso aqui é tudo muito bonito, mas não mata a fome”

Nômade Orquestra e Edgar

Ocidentes Acontecem (2018)

RESUMO

A melhoria das condições socioeconômicas e a segurança alimentar são objetivos de desenvolvimento buscados por diferentes atores, como governos, comunidades e ONGs. No entanto, seu impacto na cobertura florestal geralmente é negativo. A contribuição das florestas para o desenvolvimento socioeconômico e a segurança alimentar também não está clara. Além disso, as mudanças na população rural e seu efeito na cobertura florestal e nas condições socioeconômicas são pouco conhecidas. Neste estudo, analisamos a relação entre a mudança na cobertura florestal e o desenvolvimento humano em municípios da Caatinga, uma floresta seca no nordeste brasileiro. Também investigamos como a variação na cobertura florestal afeta a segurança alimentar da região e como as mudanças ambientais se relacionam com as mudanças populacionais. Para o primeiro objetivo, utilizamos dados de cobertura da terra do MapBiomas e dados socioeconômicos do IBGE para os anos 1991, 2000 e 2010. Realizamos análises espaciais para comparar a evolução dos índices socioeconômicos em municípios com diferentes níveis de cobertura florestal. Para o segundo objetivo, criamos um índice de segurança alimentar multidimensional com dados do censo agropecuário de 2006 e 2017. Analisamos como as mudanças na cobertura florestal se relacionam com as mudanças na segurança alimentar da região. No terceiro objetivo, usamos dados de cobertura da terra do MapBiomas e dados de distribuição populacional do WorldPop. Estimamos o tamanho da população rural na Caatinga próxima às florestas e examinamos as mudanças populacionais e ambientais, levando em consideração a estrutura espacial. Descobrimos que ao longo do tempo, todos os índices socioeconômicos melhoraram, especialmente em municípios com níveis intermediários de cobertura florestal. Também encontramos uma relação quadrática entre cobertura florestal e desenvolvimento socioeconômico. Em relação à segurança alimentar, a pobreza e a desigualdade foram os principais fatores influentes, e também observamos uma relação quadrática com a cobertura florestal. Não encontramos uma relação direta entre mudanças populacionais e ambientais na Caatinga, mas fatores como oportunidades urbanas e rurais parecem influenciar as mudanças populacionais. Concluímos que o desmatamento pode trazer ganhos econômicos, desde que questões de desigualdade econômica sejam abordadas. No entanto, desmatar além de um certo limite não contribui para o desenvolvimento dos municípios e pode levar a piores condições socioeconômicas e de segurança alimentar, resultando, inclusive, em despovoamento de áreas rurais da Caatinga. Portanto, é crucial considerar as dinâmicas ambientais e populacionais, juntamente com medidas de combate ao desmatamento e redução da pobreza e desigualdade, ao planejar projetos de desenvolvimento sustentável na região.

Palavras-chave: Boom-Bust; Fronteiras Agrícolas; Sistemas Socioecológicos; Recursos Naturais; Desenvolvimento Rural.

ABSTRACT

Improving socio-economic conditions and ensuring food security are developmental objectives pursued by various stakeholders, including governments, communities, and non-governmental organisations. However, the impact of improving human development conditions on forest cover is often detrimental. Moreover, the contribution of forests to socio-economic development and food security remains unclear. Additionally, the changes in rural population and their effects on forest cover and socio-economic conditions are poorly understood. In this study, we examine the relationship between forest cover change and human development in municipalities within the Caatinga, a dry forest in northeastern Brazil. We also investigate how variations in forest cover relate to regional food security and how environmental changes are linked to population shifts. For our first objective, we utilised land cover data from MapBiomas and socio-economic data from IBGE for the years 1991, 2000, and 2010. We conducted spatial analyses to compare the evolution of socio-economic indices across municipalities with varying levels of forest cover. For our second objective, we constructed a multidimensional food security index using data from the agricultural census of 2006 and 2017. We analysed how changes in forest cover correlate with shifts in regional food security. For the third objective, we employed land cover data from MapBiomas and population distribution data from WorldPop. We estimated the rural population size in the Caatinga near forests and examined population and environmental changes while considering spatial structure. Our findings indicate that over time, all development indicators improved, particularly in municipalities with intermediate levels of forest cover. Furthermore, we identified a quadratic relationship between forest cover and socio-economic development. Regarding food security, poverty and inequality emerged as the primary influencing factors, and we also observed a quadratic relationship with forest cover. While we did not uncover a direct relationship between population and environmental changes in the Caatinga, however, factors such as urban and rural opportunities appeared to influence population shifts. In conclusion, deforestation can yield economic benefits, provided that issues of economic inequality are addressed. However, deforestation beyond a certain threshold does not contribute to municipal development and may lead to worsened socio-economic and food security conditions, potentially resulting in depopulation of rural areas in the Caatinga. Therefore, it is crucial to consider environmental and population dynamics alongside efforts to combat deforestation and reduce poverty and inequality when planning sustainable development projects in the region.

Keywords: Boom-Bust; Agricultural Frontiers; Social-ecological Systems; Natural Resources; Rural Development.

LISTA DE ILUSTRAÇÕES

people (FPP) at landscape scale (n=5748 forested landscapes) classified as: deforestation and rural population growth, reforestation and rural population growth, deforestation and rural depopulation and reforestation and rural depopulation. Annotations inside the figure are the number of landscapes in each category. b) Spatial distribution of the four types of population-environment transitions at municipality scale. PI, CE, RN, PB, PE, AL, SE, BA, MG are acronyms of Brazilian states.

LISTA DE TABELAS

SUMÁRIO

1 INTRODUÇÃO

O crescimento econômico e o desenvolvimento humano tem sido acompanhado, na maioria dos lugares do mundo, por desmatamento, degradação ambiental e repartição desigual dos benefícios desse processo (ROCKSTRÖM et al., 2009). A abordagem dos serviços ecossistêmicos prediz que a degradação ambiental deve levar à perda de serviços que são fundamentais para as atividades humanas e o bem-estar (e.g. agricultura, saúde, lazer) (MILLENNIUM ECOSYSTEM ASSESSMENT (PROGRAM), 2005). Paradoxalmente, a maioria dos países do mundo tem experimentado um aumento do bem-estar (no período pré-covid) independentemente da degradação ambiental (RAUDSEPP-HEARNE et al., 2010). Ainda, as florestas tropicais são vistas em alguns contextos como armadilhas de pobreza (ERIKSSON et al., 2021), estimulando ainda mais a conversão de florestas para atividades que supostamente promovem o desenvolvimento. As evidências sobre o papel das florestas na redução da pobreza são bastantes mistas e dependem fortemente do contexto geográfico, social e econômico (RAZAFINDRATSIMA et al., 2021). Por tanto, ainda é muito presente na mentalidade de tomadores de decisão e políticos a ideia de que apenas através da expansão das atividades humanas que será possível erradicar a pobreza e melhorar a qualidade de vida (JAGGER et al., 2022). Isto aumenta a dificuldade em alcançar objetivos do desenvolvimento sustentável simultaneamente, tais como a erradicação da pobreza, fome zero e conservação dos ecossistemas terrestres (SCHERER et al., 2018). Isso também demonstra a falta de conhecimento que existe sobre a relação entre desmatamento, populações rurais e as condições de desenvolvimento humano de uma região, lacuna de conhecimento que esta tese pretende preencher.

A relação entre florestas e desenvolvimento socioeconômico é por vezes mais clara em fronteiras agrícolas, onde rápidas mudanças na cobertura da terra e nas economias locais acontecem (BARBIER, 2012). Uma hipótese proposta sobre essa relação, chamada ‘boom-bust’, prediz que municípios (ou comunidades, regiões, países) com alta cobertura florestal (i.e. estágios iniciais da fronteira agrícola) têm baixos índices de desenvolvimento. No entanto, a exploração dos recursos naturais (e.g. extração seletiva de madeira, expansão da área agrícola, mineração) deve gerar mais riqueza e movimentar a economia local, além de atrair novos imigrantes e investidores. Isso deve melhorar os índices socioeconômicos do local, como renda, educação e longevidade. Esta fase de transição de uma fronteira inativa/inicial para uma fronteira de intensa atividade extrativista e agrária representa a fase de ‘boom’ na economia local (RODRIGUES et al., 2009; SACHS; WARNER, 1998). Esta hipótese prediz ainda que após a exaustão dos recursos naturais que sustentam o ‘boom’, se a economia local não passar por uma transição para uma economia mais industrializada e de serviços, a economia local vai passar por um ‘bust’, em que as condições socioeconômicas pioram junto com a diminuição da cobertura florestal, resultando em um cenário de alta degradação ambiental, baixa atividade econômica e piora das condições socioeconômicas da região (BARBIER, 2020; SACHS; WARNER, 1998).

Um claro exemplo da aplicação desta hipótese se deu no estudo de Rodrigues et al (2009) em que os autores mostraram que municípios da Amazônia com níveis extremos de cobertura florestal (i.e. < 25% ou >75%) possuem condições socioeconômicas igualmente baixas quando comparados com municípios com níveis intermediários de cobertura florestal (>25% e <75%). No entanto, esses achados foram contestados por outros autores, afirmando que a relação de boom-bust encontrada para municípios da Amazônia é resultado de autocorrelações espaciais (WEINHOLD; REIS; VALE, 2015) e que este padrão se modificou após os anos 2000 (TRITSCH; ARVOR, 2016). Por outro lado, o padrão boom-bust foi observado em estudo mais recente realizado com 98 países subdesenvolvidos, mostrando que países que continuam desmatando através da expansão agrícola e que não distribuíram os benefícios, falharam em reduzir a pobreza e desigualdade, além de esgotarem os recursos florestais (BARBIER, 2020).

Um outro caminho que algumas economias desenvolvidas seguiram foi o da substituição da matriz econômica de uma economia extrativista para um economia industrial e de serviços (RUDEL et al., 2005). Isso permitiu que estas economias continuassem crescendo ao mesmo tempo em que a recuperação da cobertura florestal aconteceu, fenômeno conhecido como transição florestal (MATHER; NEEDLE, 1998). A hipótese da transição florestal diz que o crescimento das economias ao longo do tempo, em conjunto com o desmatamento, aumenta o valor relativo dos recursos florestais, estimulando o reflorestamento a partir de um certo limiar de cobertura florestal (CARAVAGGIO, 2020). Um segundo caminho se dá quando se torna mais atrativo para os produtores abandonarem a produção agrícola em troca de trabalhos fora da fazenda, permitindo a regeneração natural (RUDEL et al., 2005). Este padrão já foi observado para alguns países desenvolvidos, tais como Japão e Coreia do Sul, e em países subdesenvolvidos, como Índia e Vietnã, enquanto que outros países da região ainda estão em fases iniciais do processo (LIU et al., 2017). No início desta transição, as taxas de recuperação da cobertura florestal são altas, mas devem atingir um patamar (CARAVAGGIO, 2020). Ao atingir esse patamar, o ganho de cobertura florestal começa a competir por espaço com outros usos da terra, como produção agrícola e pode afetar negativamente a segurança alimentar (DOELMAN et al., 2020).

A relação entre cobertura florestal e segurança alimentar pode apresentar sinergias e/ou trade-offs de acordo com o sistema alimentar (YANG; GERGEL; BAUDRON, 2020) e com a quantidade e disposição espacial da floresta (RASMUSSEN et al., 2020). Por um lado, as florestas podem contribuir para a segurança alimentar de um local por promoverem serviços ecossistêmicos importantes para a produção alimentar (e.g. polinização, controle de pragas), ou de forma mais direta, através de fonte de caça ou como forragem para gado (BAUDRON et al., 2019). Por outro lado, áreas com grandes quantidades de floresta podem ter uma baixa produção agrícola, já que poucas partes foram convertidas para um uso agrícola mais intensivo. Em regiões com a estrutura agrária mais consolidada, como na Grã Bretanha, por exemplo, o ganho de cobertura florestal tem o alto potencial de comprometer parte da produção agrícola e da segurança alimentar da região se os trade-offs não forem considerados (WILKES et al., 2020). Em países subdesenvolvidos, a especulação de terra para projetos de restauração florestal também pode ser um importante promotor de insegurança alimentar local se estes projetos não considerarem essa possibilidade (GARCIA-POLO et al., 2021). Entender como a pobreza e desigualdade de acesso aos sistemas alimentares influencia na relação entre florestas e segurança alimentar também é um fator chave, tendo em vista que o acesso à comida é considerado a principal causa de insegurança alimentar no mundo (FAO, 2020).

Ainda não é claro a relação entre florestas e a segurança alimentar de uma região, mas é possível que essa relação siga um padrão semelhante ao do 'boom-bust', tendo vista que apenas as florestas não são o suficiente para garantir a segurança alimentar de uma região, mas desmatar demais pode colapsar os sistema ecológico responsável por manter os serviços necessários para a produção de comida (BAUDRON et al., 2019). Por tanto, níveis intermediários de cobertura florestal em uma paisagem (ou municípios, país) podem garantir a produção agrícola enquanto ainda mantém preservada parte do habitat original. Entender essas relações e identificar como evitar trade-offs entre florestas e segurança alimentar é fundamental para que os sistemas alimentares se tornem mais sustentáveis, garantindo não só a manutenção da cobertura florestal, mas também a melhoria na qualidade de vida das pessoas.

Outra relação não muito clara na literatura é a relação entre as mudanças na população rural e a alteração da cobertura florestal. Também pouco se sabe sobre as condições de desenvolvimento humano em que estas transições estão ocorrendo. A degradação ambiental foi recentemente apontada como uma consequência direta do crescimento da população humana (GREEN et al., 2022). No entanto, os países com um PIB per capita mais elevado tendem a ter uma maior pegada ambiental, independentemente do tamanho da população (HUGHES et al., 2023). Entender se o tamanho da população rural ou urbana tem maior influência nas alterações ambientais também é objeto de debate (DEFRIES et al., 2010). Sendo assim, compreender a relação entre o crescimento ou o declínio da população rural e a alteração da cobertura florestal é importante para atingir os objetivos de desenvolvimento sustentável.

O crescimento da população rural nas fronteiras agrícolas pode levar ao desmatamento se esse aumento expandir a força de trabalho e a demanda por alimentos (RODRÍGUEZ GARCÍA et al., 2021), aumentando os incentivos para expandir as áreas de cultivo. Por exemplo, em partes do sudoeste da Amazônia brasileira, um aumento na densidade populacional rural esteve associada a um maior desmatamento (JUSYS, 2016). Entretanto, o desmatamento em 41 países tropicais foi impulsionado principalmente pelo crescimento da população urbana (DEFRIES et al., 2010). Em outros contextos, o crescimento da população rural foi associado ao aumento da cobertura florestal. Em Gana e na Guiné, as práticas agrícolas (e.g., manejo do fogo, agroflorestas) e até mesmo novos assentamentos (e.g., por meio do plantio ativo de espécies de árvores úteis) foram associados com ganho na cobertura florestal (LEACH; FAIRHEAD, 2000), desafiando a ideia de que o crescimento populacional necessariamente causa o desmatamento. As consequências do despovoamento rural sobre a cobertura florestal dependerão do tipo de agente que está ocupando a terra e de como a terra é explorada. As remessas de dinheiro internacional, por exemplo, promovem a regeneração florestal no Nepal, combinada com o despovoamento rural e o declínio do setor agrícola (OLDEKOP et al., 2018). No entanto, o acúmulo de terras e a desapropriação de pequenos proprietários por grandes latifundiários são importantes fatores de desmatamento tropical (DAVIS et al., 2015) e podem causar o despovoamento rural à medida que as fronteiras agrícolas avançam (LÓPEZ-CARR, 2021). As diversas tipologias das transições população-ambiente revelam como essas trajetórias podem ser variáveis e imprevisíveis.

Além da já mencionada hipótese que prediz melhores condições socioeconômicas em locais com níveis intermediários de floresta, existe o argumento de que paisagens com níveis intermediários de distúrbio devem prover mais serviços ecossistêmicos que paisagens com pouco distúrbio ou completamente degradadas (MELO et al., 2013). Esta ideia pode ser expandida para o contexto de desenvolvimento socioeconômico: locais completamente florestados só possuem serviços e produtos prestados pelas florestas, tais como madeira, comida, plantas medicinais, mas não tem acesso a serviços “humanos” tais como assistência médica, produção agrícola, escolas. Enquanto que locais com baixa cobertura florestal, não possuem importantes serviços ecossistêmicos para o desenvolvimento socioeconômico e segurança alimentar, tais como lazer, regulação hídrica e “safety nets” para populações mais pobres. Sendo assim, é razoável pensar que locais com níveis intermediárias de cobertura florestal devem ao mesmo tempo ter níveis adequados de serviços prestados pelas florestas e, ainda assim, as populações locais devem ter mais acesso a bens e serviços providos pelas atividades humanas, resultando em melhores condições socioeconômicas e mais segurança alimentar.

Com isso, esta tese tem como objetivo geral entender a dinâmica da cobertura florestal e sua relação com o desenvolvimento humano na Caatinga. Especificamente, pretendo identificar os padrões espaciais de desmatamento na Caatinga, entender a relação da mudança florestal com o desenvolvimento socioeconômico de municípios da Caatinga, entender a relação entre cobertura florestal e segurança alimentar nos municípios da região e, por fim, entender como as mudanças na população rural estão associadas com mudanças na cobertura florestal. Nossas hipóteses são 1) Municípios com níveis intermediários de cobertura florestal devem ter melhores índices socioeconômicos pois devem ter mais serviços ecossistêmicos que municípios com baixo níveis de cobertura florestal e mais infraestrutura que municípios com altos níveis de cobertura florestal; 2) Municípios com maior área agrícola (mais desmatados) não devem ter mais segurança alimentar porque esta depende mais das condições de acesso da população ao sistema alimentar do que da produção de comida em si; e 3) a direção conjunta da mudança na cobertura florestal e no tamanho da população deve estar associada a um contexto específico de mudanças em indicadores de desenvolvimento.

2 ARTIGO 1 – Deforestation and human development in the Brazilian Caatinga dry forest (resubmitted to Global Environmental Change)

Authors

Lucas Alencar,

Laboratory of Applied Ecology, Department of Botany, Federal University of Pernambuco, Recife, Pernambuco, Brazil

Corresponding author: lucas.alenc@ufpe.br. +55 (81) 98864-1253. Av. Prof. Moraes Rego, 1235 - Cidade Universitária, Recife - PE - CEP: 50670-901. Centro de Biociências, Departamento de Botânica, Laboratório de Ecologia Aplicada

Luke Parry,

Lancaster Environment Centre, Lancaster University, Lancaster, United Kingdom. luke.parry@lancaster.ac.uk

Felipe Melo

Laboratory of Applied Ecology, Department of Botany, Federal University of Pernambuco, Recife, Pernambuco, Brazil. felipe.plmelo@ufpe.br

Highlights

- Evaluated boom-bust hypothesis in a densely populated agricultural frontier
- Deforestation~development relationship assessed using panel and cross-sectional data
- Panel analysis showed weaker poverty reduction in highly deforested municipalities
- Propensity score weighting showed a quadratic deforestation~development relationship

- Spatial models confirmed better development outcomes at intermediate deforestation

Abstract

The relationship between tropical deforestation and human development is unclear and contested. There is evidence, mostly from relatively new agricultural frontiers in Amazonia, of a boom-bust pattern of development associated with deforestation, indicating the failure of business-as-usual development policies based on agricultural expansion. The generality of the boom-bust development pattern is yet to be tested for other forest biomes, especially for densely populated and consolidated relatively old farm-forest frontiers. We evaluated this relationship in Brazil's Caatinga, one of the largest dry forests in the world and home to 28 million people. We used panel data (1991, 2000 and 2010), and cross-sectional data (2010) from 1207 municipalities to assess how development indicators are linked to deforestation. Our main finding is that deforestation in the Caatinga is associated with a boom-bust development pattern. Municipalities at the advanced stage of deforestation (<33% of forest cover remaining) in 1991 generally had higher development indicators than the initial stage (>66% remaining), but differences between these groups disappeared by 2010. Intermediate stage municipalities (33-66% remaining) consistently outperformed initial and advanced stage municipalities in four out of six development indicators (longevity, monetary income, extreme poverty prevalence, and child mortality), indicating a temporary 'boom' during frontier advance, followed by a stagnation. Evidence of a boom-bust was supported by cross-sectional analysis of 2010 data using propensity score weighting and a spatial autoregressive model. Overall, our findings contribute to on-going debate, and strengthen the boom-bust hypothesis. By implication, the consumption of natural resources alone is insufficient for achieving sustained development progress. Sustainability at Brazil's agricultural frontiers cannot be achieved by depoliticized technical 'fixes', and instead requires the state, non-state institutions and society at large to confront the nation's entrenched inequalities and uneven power relations.

Key-words: Sustainable development, land cover change, semi-arid region, longitudinal analysis, econometric model

1 Introduction

This paper engages with the contested hypothesis that tropical deforestation leads to a 'boom-bust' in human development. Our novel contribution is examining this relationship across agricultural frontiers in the Brazilian Caatinga, the largest dry forest in South America, and arguably the most populated on earth (Silva et al., 2017a). In doing so, we shift empirical testing of the boom-bust hypothesis beyond Amazonia (Caviglia-Harris et al., 2016; Weinhold, Reis and Vale, 2015), or national-scale analysis (Barbier, 2004, 2020a; Miyamoto, 2020). Early affirmation of the hypothesis came from Sachs and Warner's (1998) finding that natural resource booms in Latin American countries – during 1960-1994 – mostly failed to catalyse long-term improvements in per capita GDP. Later, analysis of municipalities in the Brazilian Amazon revealed an apparent boom-bust development pattern across the deforestation frontier in 2000 (Rodrigues et al., 2009), interpreted as the failure of the "business as usual" model of unsustainable regional development. However, their cross-sectional analysis (i.e., a space-for-time substitution) was criticised for neither controlling for spatial auto-correlation nor providing causal insights (Weinhold, Reis and Vale, 2015).

Establishing consistent linkages between changes in tropical forest cover and human welfare remains elusive, and variation may be explained by development stage and political-economic forces; both context- and scale-dependent. Agricultural frontier advancement is an important driver of tropical deforestation (Seymour and Harris, 2019) and tends to occur in poorer sub-national regions where land remains relatively cheap and abundant. Indeed, political ecologists typically understand frontiers as spaces at the limits of the power of state institutions, and are often the subject of conflicts over resource extraction (Watts, 2018). In early stages, agricultural frontiers may be dominated by migrants in search of a better life (Carr, 2009). This movement of people is central to frontier advance but complicates attempts to measure cause-effect relationships between deforestation and human welfare. As frontiers advance, environmental and social change may reflect processes of agricultural mechanisation, land concentration and smallholder dispossession, infrastructure development (e.g., road-building), and changes in market access and policy environments (Thaler, Viana and Toni, 2019; Thypin-Bermeo and Godfrey, 2012). Moreover, any changes in welfare will likely reflect altered relationships with forests, and the distribution of economic benefits from deforestation activities and agricultural expansion.

Whether boom-bust development patterns are general phenomena or instead dependent upon specific conditions mostly found in new and hyper-dynamic agricultural frontiers remains uncertain. Silva, Prasad, and Diniz-Filho (2017b), for example, suggests that a boom-bust only occurs in specific areas of Amazonia and cannot be generalised for the entire biome. We address this key gap by examining the relationship between deforestation and development in a tropical dry forest, a type of social-ecological system often characterised by environmental degradation, multidimensional poverty, and scant research attention (Schröder, Rodríguez and Günter, 2021; Sunderland et al., 2015). The Caatinga dry forest is located in the least developed part of Brazil, a semi-arid hinterland in the country's Northeast. Municipalities in the Caatinga have the worst development metrics in Brazil, including severe poverty and inequality (Silva et al., 2017a). Currently, agricultural land in the Caatinga is mostly occupied by resource- and land-poor smallholders (Finan and Nelson, 2022), known as Caatingueiros (traditional inhabitants of the Caatinga). This is a historically neglected people (Nelson and Finan, 2009) that still remains so, exemplified by low levels of investments from the federal program for supporting family agriculture, PRONAF (Dyngeland, Oldekop and Evans, 2020). In environmental terms, the current narrative blames the Caatingueiros' livelihoods for causing most degradation of the Caatinga's vegetation and severely disturbing forest remnants (Antongiovanni et al., 2020; Silva et al., 2017a; Tabarelli et al., 2017). This discourse side-steps recent evidence that cattle-ranching, a farming practice commonly attributed to large-landowners, is the predominant land use in deforested areas, occupying most of the cleared areas in this region (MapBiomas, 2022). This limited understanding about the proximate causes of deforestation of the Caatinga, and other global drylands with similar social-environmental histories, reflects scant research on dry forest biomes from a development and environmental change perspective (Schröder, Rodríguez and Günter, 2021), hampering efforts to promote sustainable futures for drylands.

The boom-bust hypothesis – as applied to forested regions – makes two assumptions; first, that highly-forested areas tend to be ‘poverty traps’; second, that deforestation fails to alleviate poverty in the long-term. The hypothesis does not, however, include explicit caveats or assumptions regarding the relationship between development and forest type (i.e., rainforest, dry forest, or even a temperate forest), the level of anthropogenic disturbance (i.e., old-growth forests, managed-yet-disturbed forests, or secondary forests) or the age of frontier colonisation. Instead, the boom-bust hypothesis simply states that natural resource consumption, including non-forest resources, such as minerals (Sachs and Warner, 1998), fails to promote sustained long-term gains in human development. In the Global South, there is convincing evidence that living in or near highly forested regions (i.e., the ‘pre-frontier’, with little deforestation) is generally associated with monetary poverty. Drawing on subnational data from seven tropical countries, Sunderlin, Dewi and Puntodewo (2008) found a positive correlation between an area’s remaining forest cover and poverty rate. However, this correlation can also be interpreted as evidence that tropical forests provide safety nets for the rural poor (Angelsen and Wunder, 2003), justifying investments in conservation and forest livelihoods. Less contentious is that persistent forest poverty poses a wicked problem which must be overcome to achieve Sustainable Development Goals (SDGs) (Pritchard et al., 2019).

Does existing research support the expectation of rapid increases in welfare gains during the intermediate stage of frontier advance, followed by a bust or stagnation in poverty alleviation? At the global scale, indicators of human well-being have consistently improved in tandem with urbanisation and the rapid loss and advanced degradation of natural habitats (Navarro, D’Agostino and Neri, 2020; Raudsepp-

Hearne et al., 2010); paradoxical because unsustainable economic growth has led humanity to the edge of Earth's ecological limits (Rockström et al., 2009; Steffen et al., 2015). For industrialised, richer nations, net forest regrowth has generally occurred alongside long-term development gains (Ewers, 2006); affirming the so-called environmental-economic 'win-win' predicted by Forest Transition Theory (Mather and Needle, 1998; Rudel, 1998). In contrast, Barbier's (2004) economic analysis of Latin American countries demonstrates that although forest clearance initially allows for rapid development, the economic benefits of logging and agricultural land expansion appear temporary, plausibly due to distorted land and resource markets, and ineffective land rights.

Contrasting with Rodrigues et al.'s (2009) municipality-scale boom-bust findings, property-scale research shows that declining on-farm forest cover is linked to welfare improvements, yet these are mediated by access to infrastructure and institutions (Guedes et al., 2014). Garrett et al.'s (2017) study of two late-stage Amazonian deforestation frontiers found that farmers' subjective well-being is positively correlated with time-on-property and increasing agricultural footprint. They showed this expanding 'footprint' typically involved expansion of extensive cattle-raising; an environmentally degrading land-use associated with low income. In South-East Asia, deforestation~development linkages appear to vary by country. Deforestation in Indonesia failed to reduce rural poverty, and instead the emergence of large plantations dispossessed smallholders and increased social inequities (Miyamoto, 2020). Conversely, the same study found that decades of deforestation in Malaysia was linked to reductions in rural poverty, ostensibly linked to government policies spreading the benefits of rubber and oil-palm plantations, and timber-export. Analysis of 98 land-abundant countries (most in the Global South) concluded that deforestation through agricultural expansion fails to mitigate poverty or inequality long-term, and may perpetuate boom-bust cycles (Barbier, 2020a).

Social inequities often increase as deforestation frontiers advance. Early frontier stages tend to have more egalitarian societies, albeit poorer (Barbier, 2020a; Rodrigues et al., 2009). De Waroux et al. (2018) describe the tendency of tropical agricultural frontiers to evolve from a populist stage of smallholder settlement (i.e., colonisation), to a consolidated stage characterised by smallholder dispossession and export-oriented production. Perhaps unsurprisingly, Amazonia's soy-farming boom, infamous for its association with deforestation in the 1990s including through displacing cattle-ranching to new frontiers, increased average incomes but amplified social inequality (Weinhold, Killick and Reis, 2013). Of course, the development gains experienced at agricultural frontiers also depend on the distribution of economic benefits from natural resource extraction (e.g., logging of hard-woods) (Rodrigues et al., 2009). Because these economic gains (including land-grabbing and related speculation) are spatially and socially uneven (i.e., through differential access to farmland, supply chains, urban markets and educational opportunities), overall development improvements may slow or even reverse in post-frontier contexts (Dawson, Martin and Camfield, 2019; Smith et al., 2019). Hence, prolonged or intense deforestation is argued to perpetuate a cycle of poverty and inequality, exacerbated by the loss of natural capital (Barbier and Hochard, 2018). Achieving more sustainable development in the forest tropics requires the equitable sharing of benefits from deforestation, and limits to deforestation in order to not disrupt their provision of ecosystem services (Neufeldt et al., 2013; Steffen et al., 2015). Ultimately, the complex interactions between lives, livelihoods and land cover change in the forested tropics are hard to measure but vital to understand in order to achieve the SDGs (Timko et al., 2018; Wan Mahari et al., 2020).

Examining forest cover and development trajectories in the Caatinga may provide new insights on the boom-bust hypothesis. We perform (1) longitudinal analysis and (2) cross-sectional analysis utilising propensity score weighting. The latter represents a quasi-experimental approach which attempts to improve our inference power about any observed relationship. Thus, our aim was to examine, at the municipality scale, the association between development and environmental change in a densely populated dry land, with many poor people heavily dependent on forest resources. We tested the hypothesis that development indicators are highest during the intermediate stage of deforestation, based on the premise that in this social-ecological system there is greatest economic activity from agriculture, services sector and resource extraction (e.g., wood-fuel collection), and provisioning of other ecosystem services from native vegetation. We expect to find a non-linear, inverted U-shaped relationship between remaining forest cover and development outcomes. In other words, that population-level socio-economic outcomes improve with deforestation, until a point beyond which further land degradation is related to overall declines in human welfare.

2 Methods

2.1 Study area and the Caatinga socio-ecological system

The Caatinga is a seasonally dry tropical forest located in the Brazilian semi-arid where vegetation varies from very dense forest with small trees and shrubs, to savanna-like vegetation with sparse shrubs and cactus, but without grasses (Queiroz et al., 2017). Official estimates from 2008 indicated that the Caatinga's vegetation cover had already declined by 45.4% (MMA, 2010), largely due to clearance for agricultural expansion, more selective vegetation removal for charcoal production, and habitat destruction for road-building (Silva and Barbosa, 2017). Satellite imagery from 2019 suggests slightly lower aggregate deforestation, 40% of original cover lost, of which 88% was replaced with agricultural land-uses (MapBiomas, 2022). The apparent decrease in cumulative deforestation likely reflects methodological differences. Whereas the official 2008 deforestation classification relied on visual inspection of Landsat-5 and CBERS satellite images, the MapBiomas land-cover estimates are derived from a Random Forest approach combined with several pre- and post-processing quality controls (Souza Jr et al., 2020). However, it is plausible that an overall increase in forest cover occurred between 2008 and 2019, given MODIS data indicates net regeneration occurring between 2000 and 2010 (Aide et al., 2013).

Typically, the Caatinga experiences several months of drought each year although there is high inter-annual variation in rainfall linked to severe, sometimes prolonged, and unpredictable droughts (Sena et al., 2018). Soil conditions are very diverse with mixed quality and suitability for agriculture (Sampaio, 2010) which, compounded by the region's unstable and dry climate, largely prevents large-scale agriculture, apart from in riverine areas where irrigated agriculture has been increasing (Sampaio et al., 2017). The principal use of the Caatinga forest is for grazing livestock, strongly influenced by the historical Portuguese colonisation process. The combined rural and urban population of the Caatinga was approximately 28 million people in 2010, according to the last official Brazilian census. These Caatingueiros are spread across 1200 municipalities in nine States. Municipalities are the lowest political level in

Brazil governed by an elected mayor who has considerable responsibilities for delivering public services and administration of public life. In each municipality there is an 'urban' centre (sometimes very small, but always containing public administration offices, commerce, transportation links, etc.) and a surrounding rural area of influence.

The Caatinga is one of the world's most densely populated seasonally dry tropical forests (Silva et al., 2017a) and according to the Brazilian government, 72.8% of the municipalities are considered as predominantly rural and only 11.9% as predominantly urban in 2010 (IBGE, 2017), which indicates a large number of 'forest-proximate people', a policy-relevant term defined by Newton et al. (2020) as people that live in and around forests. Brazilian municipalities, although they have urban and rural zones within them, were classified along a rural-urban gradient according to the total number of inhabitants and the share of the total population living in the urban centre (IBGE, 2017). However, this demographic-centric classification probably overestimates the number of urban municipalities (Rodrigues, 2014), which implies that rural municipalities have been under-counted. Many Caatingueiros continue to meet their basic needs by relying on the ecosystem services provided by native vegetation, including harvesting forage for livestock, firewood and medicinal plants (Melo, 2017).

During the 2000's, Brazil experienced an economic boom, due to rising commodities exports and market prices (Souen and de Souza Campos, 2019) together with major public (and private) investments in infrastructure (e.g., energy production, industries, hospitals, universities), with positive effects on reducing poverty (Medeiros, Ribeiro and Amaral, 2021). Combined with economic growth, Brazil's social welfare investments reduced hunger and poverty (Marques, Ximenes and Ugino, 2018), and improved health and general well-being (de Souza et al., 2021). Although public investments were not uniform across Brazil (Dyngeland, Oldekop and Evans, 2020), municipalities in the Caatinga benefited significantly. The Bolsa Família conditional cash transfer program accrued particular social benefits, including education attainment (Dyngeland, Oldekop and Evans, 2020). Despite the Caatinga experiencing sizable socioeconomic improvements this century, it remains under-developed relative to most other Brazilian regions (Cardozo and Martins, 2020; Medeiros, Ribeiro and Amaral 2021).

2.2 Data gathering

We gathered data on native vegetation cover from the open database MapBiomas collection 5.0 (MapBiomas, 2022). This project uses satellite imagery to map land-use and cover changes across all Brazilian biomes, from 1985 onwards, with data aggregated at different scales (municipality, states, macro-regions). We chose municipality as the unit of analysis because most official data related to development indicators are aggregated at this level and because municipalities are key to public policy delivery in Brazil, a highly decentralised country. Mapbiomas has two main databases: one that classifies land cover each year, and the other records transition from one land cover type to another. Each land cover type is classified in three different levels, but we considered only the highest classification levels since we are interested in broader patterns of each category (e.g., forest, agriculture) instead of more specific ones (e.g., mangroove, perennial agriculture). Thus, the percentage of native vegetation cover of each municipality was calculated as the area occupied by all types of remaining native vegetation cover in relation to the total area originally covered by native vegetation. Likewise, we calculated the annual deforestation rates for each municipality as the areal sum of transitions from all types of native vegetation cover to any other land cover. We considered a municipality as being in the Caatinga if it is completely or partially inside the 2019 Caatinga boundaries defined by the federal government (IBGE, 2019), resulting in a total of 1210 municipalities (of which we analysed 1207 for the longitudinal analysis and 1050 for the cross-sectional analysis due to missing data).

We used six development indicators including the three individual sub-dimensions (i.e., not a weighted composite measure) of the MHDI, a Brazilian adaptation of the Human Development Index to calculate this index at the municipality scale (M stands for municipality) (Atlas Brasil, 2020). The HDI-Income sub-dimension equivalent of the MHDI is the mean per capita income of the municipality; the HDI-Longevity is a person's mean life expectancy at birth in a given municipality; and the HDI-Education is a geometric mean between the percentage of people older than 18 years with completed elementary school and the percentage of students in the expected grade according to age. The three other indicators were the Gini Index (a measure of the inequality of monetary income distribution within a population), rate of extreme poverty (% of households, based on official categories of per capita income, whose thresholds increase over time) and under-five mortality (per thousand children per year). Despite the HDI's shortcomings, it is comparable across regions and easy to understand, and analysing the three sub-dimensions elucidates important aspects about a population's life conditions (Sagar and Najam, 1998). We choose the Gini Index because it is widely used to represent income inequality, is easy to understand and allows comparison with other regions (Giorgi and Gigliarano, 2017). We analysed extreme income poverty because it represents the most vulnerable share of the population and therefore widely used in poverty research (Castañeda et al., 2018; Roser and Esteban, 2013). Under-five mortality serves as a good proxy for overall population health and relative access to government assistance (Rutherford, Mulholland and Hill, 2010). We gathered all six development indicators data from the Atlas Brasil project, which compiles and specialises the data derived from the IBGE decadal census (Atlas Brasil, 2020). Thus, we had comparable data for three time periods; 1991, 2000 and 2010. Combined, these data reflect variation in development outcomes across time and space, therefore providing proxies of the potential social, economic, and health consequences of native vegetation loss for the *Caatingueiro* population.

2.3 Data analysis

2.3.1 Longitudinal panel analysis

We grouped municipalities into three deforestation stages based on the percentage of original native vegetation cover deforested, in a particular year (i.e., initial stage has a total deforested area between 0-33%; intermediate 33.1-66%; advanced 66.1-100%), and calculated the annual deforestation rate for each group. Note, the terms initial, intermediate and advanced refer to the deforested area, and not to how long a municipality has been experiencing deforestation. Grouping the municipalities allowed us to visually identify any micro-regions with similar forest cover and deforestation rates, because spatially-proximate municipalities may have shared histories of occupation. This approach can also help to identify any distinct frontier regions in the Caatinga, as was done in Amazonia (Aguiar, Câmara and Escada, 2007). This grouping was made for all years independently, therefore a municipality can change its deforestation stage, either moving to a lower forest cover stage, or higher forest cover stage through regeneration (Supplementary Figure 1 shows net forest cover change across deforestation groups).

We statistically compared the temporal evolution of development indicators using repeated measures ANOVA for each metric. We used the deforestation stages as “treatments" and the year as within-group variation. Afterwards, we performed a post-hoc test using Tukey adjustment to evaluate any temporal differences in development across the deforestation stages. We built another set of models using lagged deforestation data as a sensitivity analysis to test if temporal changes in deforestation data would change the result of lag 0 models, and hence, potentially alter our results. Considering the years of 1991, 2000 and 2010 as the lag 0, we lagged for one year (i.e., 1990, 1999 and 2009), two years (i.e., 1989, 1998 and 2008), and so on, until five years (i.e., 1986, 1995 and 2005). We chose those lags because beyond five years there would be unacceptably high risks of other unmeasured factors or processes distorting the observed, modelled relationships between municipality-scale deforestation and development outcomes. For instance, changes in environmental governance, and multiscalar political or economic transitions, cycles, or turbulence. Also, the earliest satellite-derived comparable deforestation data for the Caatinga is from 1985, which constrains the ability to use longer lags for the 1991 data. We then performed model averaging to check if the weighted averaged coefficient of each model changed in direction compared with the lag 0 models and hence, potentially changing our empirical findings. For the model averaging we only selected the models that had ΔAICc below 2, a conservative cut-off for selection of plausible models (Grueber et al., 2011).

2.3.2 Propensity scores weighting for cross-sectional 2010 data

The repeated measures ANOVA cannot ascertain causality so we used data only from 2010 to perform a cross-sectional analysis using Covariate Balancing Generalised Propensity Score (hereafter propensity score) weighting within a multiple regression. This is an extension of Imai and Ratikovic's (2014) method for dealing with a continuous treatment variable (Fong, Hazlett and Imai, 2018). We infer from this propensity score analysis that any marginal spatial variation in development outcomes, unexplained by other predictors, is indicative of temporal changes as a result of changing net forest cover. Although using propensity score weighting cannot allow for full confirmation of causality, this method relies on a quasi-experimental approach to improve causal explanation by isolating the effect of a given treatment from other confounding variables (Bärnighausen et al., 2017). Contrary to other approaches used in the literature exploring development and deforestation, the propensity score considers a set of other variables that could affect the relationship between forest cover and socio-economic development relationship (Dyngeland, Oldekop and Evans, 2020). By creating weights for each observation (municipality in a particular year), this method improves the comparability of development outcomes in municipalities with different characteristics (e.g., geographic area, land slope, soil fertility, protected area coverage, human population density) (Fong, Hazlett and Imai, 2018). Accordingly, we were able to evaluate more directly the potential effects of deforestation on development.

Furthermore, the propensity score weighting method helps to minimise endogeneity by self-selecting comparable observations with different levels of treatment, mediated by covariates that influence the outcome (Hall et al., 2022). In our case, the propensity score weighting compared municipalities (i.e., the observations) with different levels of deforested area (i.e., the treatment) that have equivalent characteristics (i.e., the covariates), which could also influence the development indicators (i.e., the outcomes), dealing with potential reverse causality/endogeneity among deforestation and development (i.e., the treatment and outcome) (Dyngeland, Oldekop and Evans, 2020; Hall et al., 2022). Although no method can eliminate the possibility of an endogenous relationship, certain methodological approaches can help minimise this possibility. Indeed, quasi-experimental approaches, including the propensity score, have been applied to alleviate potential endogeneity (Guo, Fraser and Chen, 2020; Hall et al., 2022). Additionally, we ran a Spearman's Rho correlation test in order to check the correlation between the model's residuals and the treatment variable, a test successfully employed by Dyngeland, Oldekop and Evans (2020) and Hall et al. (2022) to reveal a potential endogeneity relation. The Spearman's Rho showed very low correlation between model's residuals and treatment (ranging from -0.01 to 0.09) in all models except for HDI-Education (rho = -0.22) (Supplementary Table 7). This reinforces our assertion that the propensity score approach reduces endogeneity between deforestation and development indicators, and demonstrates that our analysis is robust.

Propensity score weighting cannot eliminate the possibility of our results being confounded by a potential simultaneous causality effect, a common source of endogeneity (Zaefarian et al., 2017). This effect can be remedied by lagging the independent variable. Therefore we lagged the deforestation data from one (2009) to five years (2005) and re-ran the models. We chose to lag only up to five years (i.e., 2005 deforestation data against 2010 development indicators) for the same reasons mentioned above. After re-running all models, we undertook model averaging with the plausible models (those with ΔAICc < 2) and checked if the weighted averaged coefficients and standard errors would change the interpretation of the deforestation and development relationships compared to the lag 0 model (deforestation data from 2010).

We built the propensity scores, using deforestation as a continuous treatment, and a set of 14 potential confounding variables related to the rural and urban economy of municipalities which might affect forest cover and the evaluated development outcomes (Supplementary Table 1). The first step was to build a multiple regression model in order to check which confounding variables were more correlated to the treatment (Supplementary Table 5), and then build the weights to reduce the correlation between the treatment and covariates (Fong, Hazlett and Imai, 2018). The propensity score model results showed a marked reduction in treatment-covariate correlation after weighting, relative to the unmatched initial model (Supplementary Figure 2). Hence, this weighting approach improved our ability to evaluate the potential effects of deforestation on development. Then, we used the propensity score weights within a Generalised Linear Model (GLM) for each development indicator, using deforestation as an explanatory variable with a quadratic term (Equation 1). This was because we expected a non-linear, inverted U-shaped relationship between deforestation and development.

$indicator = \alpha + \beta_1 \stackrel{\text{def}}{=} + \beta_2 \quad ^2 + \varepsilon$ (1)

where indicator is the development outcome of interest (Y), DEF is the total deforestation in the studied year, is the intercept of the regression, ε is the error term and βi are the parameters to be estimate.

2.3.3 Spatial autocorrelation

Spatial correlation is common in land-use datasets and was Weinhold, Reis and Vale's (2015) main reason for rejecting the boom-bust pattern found by Rodrigues et al. (2009). We therefore used Moran's I test to assess potential spatial autocorrelation in the residuals of our propensity score GLMs. We found positive spatial autocorrelation in all of our propensity score GLMs (Supplementary Table 2) meaning that non-modeled variables from neighbouring municipalities with shared territorial borders were influencing the regression estimates to some degree. In order to evaluate how much the autocorrelation was influencing the effect of deforestation on development indicators we performed a robustness test, building a Spatial Autoregressive with Autoregressive Disturbance model (Kelejian and Prucha, 1998) (hereafter spatial model) (Equation 2).

$$indicator = \alpha + \beta_1 \stackrel{\text{def}}{=} + \beta_2 \quad ^2 + \lambda W indicator + u$$

$$u = \rho W u + \varepsilon \qquad (2)$$

where indicator is the development outcome of interest (Y), DEF is the total deforestation occuring in the studied year, ε is the error term, is the intercept of the regression, βi are the parameters to be estimate as in Equation 1, W is the spatial weight matrix, u is the vector of regression disturbance, λ and are the autoregressive parameters.

We then compared the spatial model results with each of the propensity score GLMs in order to check whether there were any changes in significance or direction of the treatment (i.e., deforestation amount) effect. This spatial model accounts for spatial autocorrelation by adding a spatial term to the dependent variable and error term

(Bivand and Piras, 2015). By adding a spatial term to the model's error term, this approach controls for the observed spatial autocorrelation in non-modeled data that we found in our propensity score GLMs. Furthermore, this spatial model also controls for ways in which a municipality's development metrics might be influenced by the development of neighbouring municipalities. For example, the development indicators of a municipality might be positively influenced by strong access to employment, the presence of a public university, or an effective health system in a neighbouring municipality. Thus, if results of the spatial models and propensity score GLMs point in the same direction, the latter are robust to spatial autocorrelation. In order to check for endogeneity issues in the spatial models, we also performed a Spearman's Rho correlation test as we did for the propensity score GLMs. We found very low correlation among model residuals and the treatment, ranging from 0.002 to -0.026 (Supplementary Table 7).

3 Results

3.1 Deforestation and frontier stages

Across the study period, the initial deforestation stage – characterised by high remaining forest cover and a low annual deforestation rate (Figure 1c) - was mostly located in the centre and western areas of the Caatinga, furthest inland (Figure 1a). Advanced deforestation was, not surprisingly, concentrated along the eastern edge of the biome, nearest to the more populated Atlantic coast (with numerous metropolitan areas; albeit the Caatinga biome does not extend to the East coast, which instead would be naturally covered by humid Atlantic Forest). The eastern municipalities experienced the highest deforestation rates throughout the study period, giving rise to a visible “arc of deforestation” (Figure 1a). Intermediate stage deforestation was mostly in the central Caatinga with some clusters (analogous to intra-regional smaller deforestation frontiers) amidst initial stage municipalities (Figure 1a) and had higher deforestation rates than in initial stage municipalities (Figure 1c). Importantly, this spatial patterning in the distribution of deforestation frontier stages went largely unchanged between 1991 and 2010. Based on lag 0 deforestation data (i.e., 1991, 2000 and 2010) most municipalities stayed in the same deforestation stage over the study period (Supplementary Table 3).

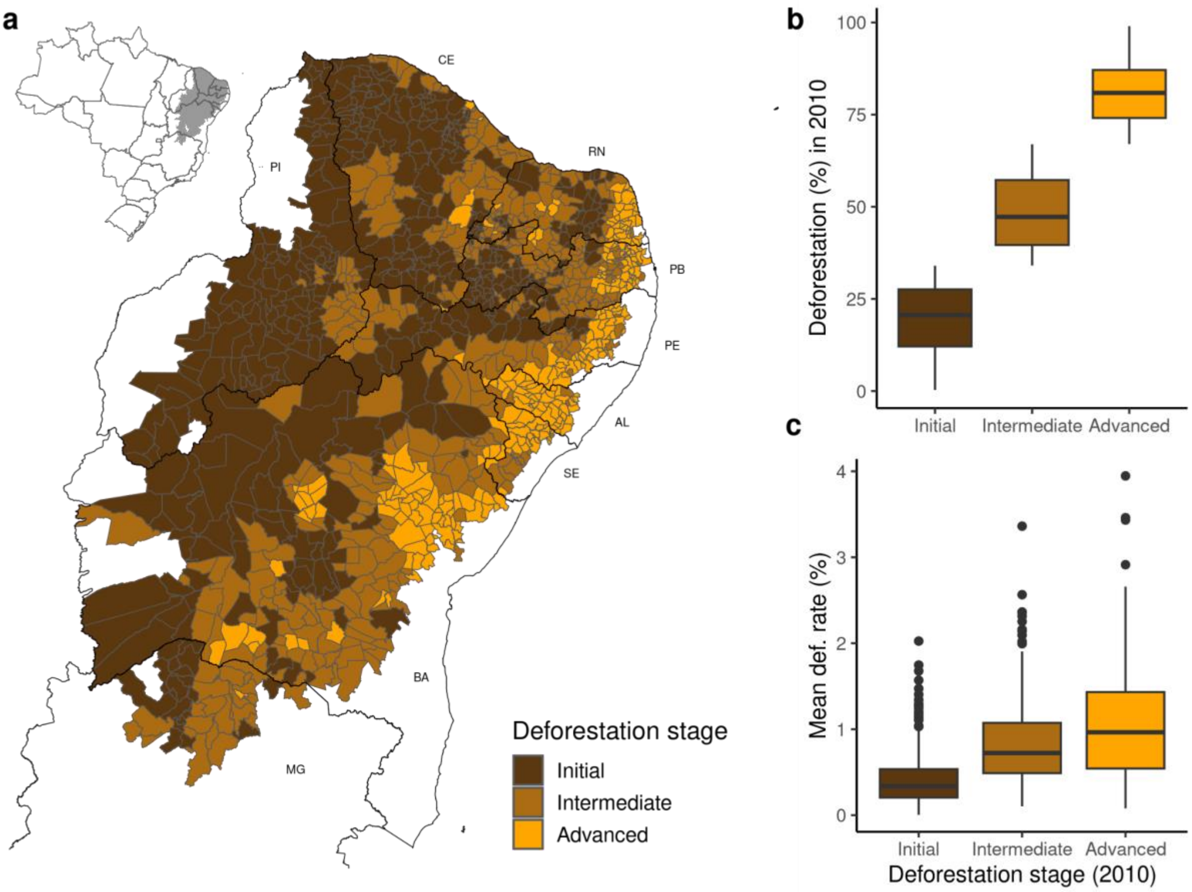


Figure 1. a) Map of deforestation frontier stages in 1207 municipalities in the Caatinga biome in North-East Brazil, based on percentage of original native vegetation cover remaining in 2010. Black lines represent Brazilian States and light grey lines are municipal boundaries. Inset map represents the

location of Caatinga biome in relation to Brazil and its constituent states. **b)** Boxplot of cumulative deforestation (% of original forest lost) by 2010 in municipalities, grouped by deforestation frontier stage. **c)** Boxplot showing mean annual deforestation rate between 1991 and 2010 in municipalities, grouped by deforestation stage.

3.2 Development indicators and vegetation change

3.2.1 Results of longitudinal analysis

There was a remarkable improvement in development metrics over the two decades of our analysis, except for the Gini inequality index, which increased between 1991 and 2000 (Figure 2). For instance, mean HDI-Education increased from 0.11 (± 0.0015 SE) in 1991 to 0.49 (± 0.0018 SE) in 2010. Mean under-five mortality dropped 70% overall, from 97 in 1991 (± 0.58 S.E.), 61.3 in 2000 (± 0.33 S.E.), to 28.5 in 2010 (+- 0.17 S.E.). However, a key finding was that the progression in each indicator varied by deforestation stage (Supplementary Table 4). In general, municipalities in the initial stage showed least improvement in socio-economic metrics across time, while the intermediate stage had the highest gains (Figures 2c-e). This is suggestive of a development 'boom' as deforestation progresses. The exception was education (Figure 2a), for which the improvements in the initial stage were better than the advanced stages, suggesting a slow-down in educational improvements at advanced stage. Overall, the longitudinal analysis showed that the apparent pace of development benefits in intermediate stages is not maintained as deforestation advances. Specifically, municipalities in the advanced stage had the best development indicators in 1991 yet by 2010 their scores were worse, or equal to, intermediate and initial stages. In this sense, the temporal analysis points to a 'stagnation' in the development benefits of deforestation, rather than a 'bust'.

Temporal lags in the deforestation data did not change the direction of the model estimates. In other words, using deforestation data from earlier years did not alter the main findings of our longitudinal analysis in terms of how deforestation affects development indicators. However, these additional sensitivity analyses show that the lag periods between deforestation occuring and any related developmental effects are indicator-specific. For instance, the effects of deforestation on income, extreme poverty and income inequality have between zero and a one year lag (up to two years for income inequality) (Supplementary Table 9). In contrast, we found that deforestation activity has lags of two, three, and four years on educational attainment, under-five mortality, and longevity, respectively. However, we may under-estimate lagged effects because each stages of our treatment variable (initial, intermediate, advanced), are very broad in terms of variation in deforestation extent (i.e., 33%), meaning that even if the total deforestation increased across the five year period of lags we investigated, in only around a tenth of cases this lag causes a municipality to shift to the next deforestation stage (Supplementary Table 10).

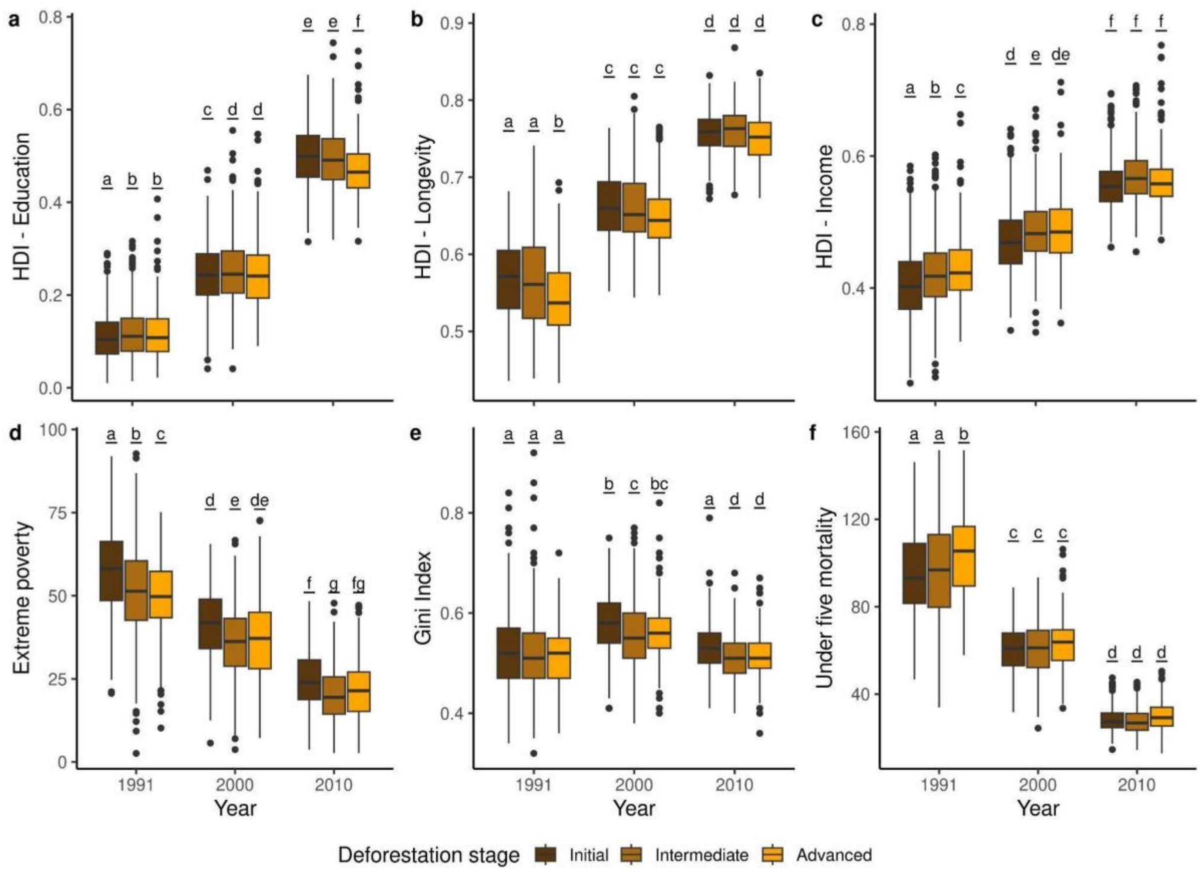


Figure 2. Human development outcomes in municipalities in the Brazilian Caatinga, grouped by deforestation stage (based on annual rate and cumulative extent) between 1991 and 2010, with post-hoc Tukey adjustment after a repeated-measures ANOVA. Letters above the boxplots represent statistical differences between groups and across years. HDI refers to the- Human Development Index, which varies from 0 to 1 and has no units, like the Gini Index. Extreme poverty is the percentage of households in a given municipality living in extreme poverty, and under-five mortality is the number of deaths of children below five-years-old per thousand per year.

3.2.2 Results of cross-sectional analysis

Our cross-sectional analysis revealed a boom-bust development pattern following deforestation in the Caatinga (Figure 3). For 2010, we found a significant quadratic relationship for five indices, but not for education (Table 1). The longevity and income indicators had a U-shaped relationship with native vegetation cover (Figure 3b-c,), whereas the other outcomes had an inverted U-shaped relationship (Figure 3d-f). Hence, municipalities with intermediate deforestation had better development indicators than municipalities with low or high forest cover. For full regression model results of propensity score GLMs see Supplementary Table 6.

The averaged cross-sectional models with lagged deforestation data showed that the direction of the coefficients did not change compared to lag 0 model, and reinforced that the quadratic term for HDI-Education was not significant (in other words, we found no evidence of a 'bust' in education in the advanced stage of deforestation; Supplementary Table 11 and Supplementary Figure 3). Therefore, deforestation in the previous five-years affected the 2010 development indicators in a similar way compared to deforestation in 2010. This indicates that our original models were not affected by endogeneity issues. Nonetheless, the selected lag periods varied across development indicators. The shortest lag was zero for income inequality, whereas the longest lags were for education (5 years) and under five mortality (3-5 years). We also found moderate lags for longevity (2-5 years), and income and extreme poverty (both 1-4 years) (Supplementary Table 11). Our spatial model results indicate that most of the propensity score GLMs (i.e., the regression models explaining development outcomes) were robust to spatial autocorrelation, because there were no changes in the direction or significance of estimates, thus confirming the validity of our results (Table 1).

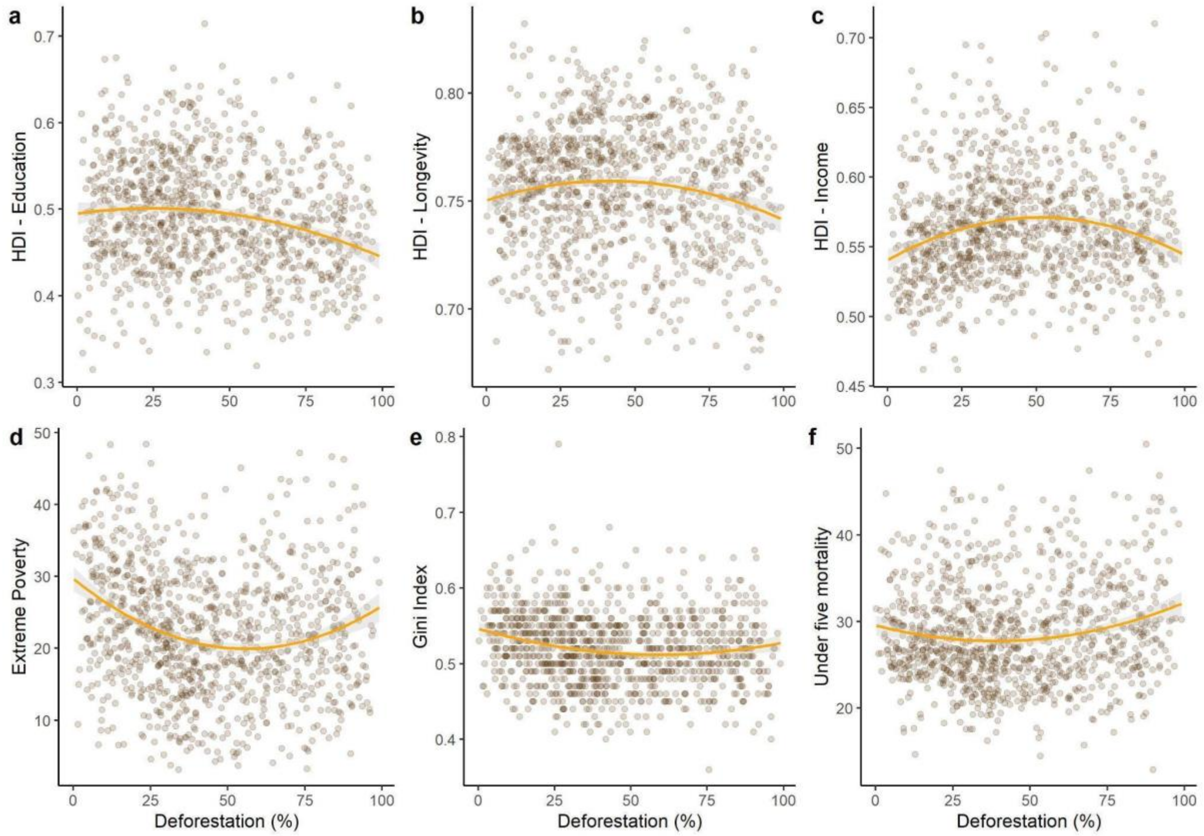


Figure 3. Modelled relationships between municipal-scale deforestation extent (cumulative % of original forest lost) and human development outcomes (a-f) in 2010 (i.e., cross-sectional data), in the Brazilian Caatinga, based on quadratic Generalised Linear Models with Covariate Balancing Generalised Propensity Score. All models are significant (p < 0.01) except for HDI - Education. Dark yellow lines refer to predicted values according to quadratic models and shading indicates 95% confidence intervals. HDI refers to the Human Development Index, which varies from 0 to 1 and has no units, similar to the Gini index. Extreme poverty is the percentage of households in a given municipality living in extreme income poverty and under-five mortality is the annualised number of deaths of children less than five-years-old per thousand.

Table 1. Effects of municipal-scale deforestation (DEF - cumulative percentage of original forest lost) and its quadratic term (DEF^2) on development indicators in the Brazilian Caatinga in 2010, evaluated using propensity score weighting GLMs. Statistically significant quadratic terms demonstrate an apparent boom-bust relationship between deforestation frontier advance and human development, based on cross-sectional data. Numbers shown here are the estimate of each model term (single or quadratic). Only statistically significant terms are shown. Spatial models were built to test the robustness

of GLM models to spatial autocorrelation and can be used to compare the direction and level of significance of the coefficients. Here, the spatial model shows that there were almost no differences and, therefore, the GLM models are robust to spatial autocorrelation . * - p < 0.1; ** - p < 0.05; *** - p < 0.01.

Model	Outcome	def	def^2
GLM with CBPS			
	HDI - Education	6.50e-04*	-
	HDI - Longevity	4.97e-04***	-5.04e-06***
	HDI - Income	8.93e-04***	-7.95e-06***
	Extreme poverty	-2.89e-01***	2.54e-03***
	Gini index	-1.16e-03***	9.88e-06***
	Under five mortality	-1.17e-01***	1.20e-03***
SARAR			
	HDI - Education	-	-
	HDI - Longevity	4.70e-04**	-4.17e-06*
	HDI - Income	6.47e-04***	-5.33e-06***
	Extreme poverty	-1.84e-01***	1.31e-03***
	Gini index	-	-
	Under five mortality	-1.05e-01**	1.01e-03*

4 Discussion

We found evidence suggesting that deforestation fails to support long-term development progress in the Brazilian Caatinga, an under-researched dry forest social-ecological system home to at least 28 million people (Silva et al., 2017a). Average human development metrics are highest in the intermediate phase of deforestation frontier, and there is little-to-no benefit, or even a decline, in advanced (or 'consolidated') frontiers. Although all municipalities showed development improvements from 1991 to 2010, the magnitude of increment differed among categories of frontier stage (i.e., initial, intermediate and late). A boom-bust development phenomenon happened in at least some municipalities of the Brazilian Caatinga. While municipalities in the initial stage seemed to experience a more rapid improvement in their development outcomes, equaling intermediate and advanced stages after 10 or 20 years, improvements for highly deforested municipalities (i.e. advanced stage) happened at a slower pace, suggesting a stagnation. Finally, our cross-sectional analysis revealed a consistent boom-bust pattern of development that suggests a threshold of around 50% forest cover (Figure 3), where development conditions stagnate or even deteriorate. Despite this pattern being better visualised using cross-sectional data, our analytical approach was able to remove most if not all confounding factors and it appears to reflect a pervasive boom-bust phenomenon for the Caatinga.

4.1 Forest poverty and the boom phase

We found strong evidence of so-called ‘forest poverty’ in a specific geographic region (hundreds of municipalities in the western Caatinga; Figure 1a) that we classified as being in the initial deforestation stage. In most under-developed tropical regions, forest poverty remains a persistent and somewhat intractable problem (Castañeda et al., 2018; Sunderlin, Dewi and Puntodewo, 2008). The overall shortcomings of forest-centric sustainable development initiatives have been attributed to factors including disempowerment and marginalisation of forest-dwellers, and governance failures (Eriksson et al., 2021). In many land-abundant countries, forested areas tend to be geographically peripheral frontier spaces for extensive agricultural production, precariously occupied by marginalised people (Barbier, 2020a; Watts, 2018) whose vulnerabilities are often invisible to the state (Scott, 1998). Indeed, we previously found that Caatinga municipalities with the highest forest cover tend to be in the driest areas, least suitable for agriculture (i.e. unsuitable for cattle and staple rain-fed crops) (Alencar et al., 2022). In the Global South, the general absence of States at forest frontiers and lack of opportunities (e.g., education, off-farm jobs) contribute to the persistence of forest livelihoods as safety nets through timber and firewood extraction, and harvesting non-timber forest products (Abebaw et al., 2012; Biland et al., 2021; Mahapatra and Shackleton, 2012). Forest livelihoods can partially alleviate the consequences of poverty by increasing family income (monetary and non-monetary) and improving nutrition and health (Ickowitz et al., 2014; Miller, Mansourian and Wildburger, 2020; Worku et al., 2014). Yet, our finding that development indicators are worst in the initial stage of deforestation may confirm that, alone, forest livelihoods fail to alleviate monetary poverty (Djoudi et al., 2015; Razafindratsima et al., 2021), including in this dry forest.

Our results confirm previous findings that tropical deforestation fails to alleviate long-term poverty (Barbier and Hochard, 2018; Garrett et al., 2017). However the temporary development gains during the intermediate phase of frontier advance in the Caatinga appear to support the notion of a natural resource commodity boom. Research in humid tropical forests shows that logging of valuable hardwoods and charcoal production can, for a time, foster extractive industries which creates jobs and income, and revenue for local governments (Barbier, 2020a; Rodrigues et al., 2009). Land clearance for agricultural production may increase food production, commerce and improve regional income (Tritsch and Arvor, 2016; Weinhold, Killick and Reis, 2013). Thus, consumption of natural resources and agricultural expansion appear to sustain initial rural and urban development, at least in Brazilian Amazonia (Caviglia-Harris et al., 2016). Our cross-sectional analyses reinforce this assertion by showing that intermediate-stage municipalities appeared to experience an advantage in most development indicators in relation to initial or advanced stages. Theories of frontier expansion state that frontier evolution – during the boom phase – attracts capitalist investors (searching for cheap land) and in-migration of wealthier, more educated families (Barbier, 2012). This influx can cause social conflicts, and may dispossess an area's original inhabitants and many of the poor smallholder colonists who arrived in early stages of the frontier (Carr, 2009). Thus, if governments fail to achieve sufficient structural changes during the boom phase (i.e., to ensure broad distribution of social and economic benefits), then the initial cycle of natural resource consumption only perpetuates the social inequities, land concentration (Babigumira et al., 2014; Miyamoto, 2020), and land degradation (Barbier, 2020b). Put eloquently by Brockhaus et al. (2021), forest frontiers in the Global South are not politics-free spaces.

4.2 Overshooting and the bust phase

Although high poverty rates are often associated with high forest cover, it is not uncommon to encounter persistent poverty in mostly-deforested, post-frontier areas (Barbier and Hochard, 2018; Sunderlin, Dewi and Puntodewo, 2008). Related social inequities are typified by poor smallholders – or labourers – living amidst large scale ranches or plantations (Barbier, 2012; Weinhold, Killick and Reis, 2013). The long term encroachment of smallholders may lead to increased poverty and promote more deforestation (Barbier and Hochard, 2018; Eriksson et al., 2021). In the Caatinga, we found that advanced stage municipalities had the highest deforestation rates, while experiencing stagnation or a bust in development.

The lack of forest resources to alleviate poverty can be further aggravated if nearby urban centres are underdeveloped, as in the Caatinga region (e.g., offering limited services or job opportunities), leaving few livelihood options for the poor (Medeiros, Ribeiro and Amaral, 2021; Miller et al., 2021). In the context of the advanced stage of deforestation, the natural capital necessary for forest-related activities (e.g., collecting woodfuel or forest products including food and medicine) is absent or inaccessible (i.e., inside large private properties). Thus, poor urban households may struggle to earn income and, faced with shocks (e.g., climatic), miss out on the natural insurance provided by forests (Pritchard et al., 2020). In rural areas with limited healthcare, for example, reliance on plants as medicine is commonplace and may indeed improve health (Razafindratsima et al., 2021; Santoro et al., 2015). Therefore, forests and the ecosystem services they promote, are likely to be most important to people in remote regions where urban centres have limited access to larger cities and fragile public service provision (Parry et al., 2018; Silva, Prasad, and Diniz-Filho, 2017b).

At an intermediate deforestation stage, delivery of ecosystem services (i.e., benefits from nature) is expected to be higher (Arroyo-Rodríguez et al., 2020; Melo et al., 2013). This is because there should be a sufficient amount of forests, but also more people benefiting from different types of services from more diverse land uses in an (still) ecologically functional landscape. Moreover, intermediate stage frontiers may benefit from better public and private services with higher food production, market integration, and access to education and health services (Araujo et al., 2021; Medeiros, Ribeiro and Amaral, 2021; Smith et al., 2019). Therefore, places with intermediate forest cover may have more livelihood alternatives to help people escape from poverty. However, our analysis can not conclusively establish a causal relationship between deforestation and development. Therefore, policies should not stimulate deforestation as a way to improve social-economic conditions. Such a naïve assumption is surely a misinterpretation of the observed pattern aiming to justify business-as-usual models of land occupancy and frontier advance. Contrarily, a major challenge for initial-to intermediate stages is to improve socioeconomic conditions without losing natural capital.

Finally, our observation of a boom-bust pattern of development in the Caatinga appears robust to spatial autocorrelation, an analytical artefact which is argued to confound many cross-sectional results, including Rodrigues et al.'s (2009) Amazonian research (according to Weinhold, Reis and Vale, 2015). We adopted a quasi-experimental approach (using propensity score weighting), controlled for space and several other socioeconomic variables that could induce a boom-bust pattern, and found the same U-shaped or inverted U-shaped curves (Figure 3), as we predicted. Our longitudinal analyses using three periods of census data affirmed that for many development indicators, initial stage frontiers seem to experience accelerated improvement, in contrast with advanced stage frontiers whose pace of development slowed. Although our lag analysis in both longitudinal and cross-sectional data showed a best response of each development outcome to a different lag in deforestation, there were no significant differences in the main results. It is beyond the scope of this paper and would be too speculative to explain the mechanisms behind the varied lag responses. Future research should try to unveil why and how some development indicators respond to short lags (lag 0 or 1) and others to longer (lag 3 or 4) in order to better understand how deforestation affects development.

5. Conclusion

We found that deforestation in the Brazilian Caatinga dry forest is associated with an apparent boom-bust development pattern when comparing municipalities using cross-sectional data (i.e., a space-for-time substitution), consistent with the pattern observed in Brazilian Amazonia by Rodrigues et al. (2009) and Celentano et al. (2012). This finding was based on propensity score weighting, a pseudo-experimental analytical approach which allows causal inference, and was largely robust to spatial autocorrelation. Nonetheless, our longitudinal analysis (panel data from 1991, 2000 and 2010) demonstrates that human development gains occurred regardless of the deforestation frontier stage; which refutes the hypothesis of an advanced frontier-stage development 'bust', in absolute terms. However, these longitudinal results show that the amount of cumulative deforestation in a given municipality influences the magnitude of long-term development gains. Over the long run, municipalities in advanced stages of deforestation experienced a stagnation in human development, even in the context of high development gains in Brazil between 1991 and 2010. Overall, our findings support the hypothesis that development outcomes are generally highest at intermediate stages of frontier advance. Sustainable development at deforestation frontiers cannot be achieved by depoliticized, technical 'fixes' such as legitimising which drivers are legal or illegal, or through improved measurement and reporting (Brockhaus et al., 2021). Instead, conserving the Caatinga biome and fostering a better future for hitherto oppressed Caatingueiro people requires the State, non-state institutions and Brazilian society at large to confront the entrenched inequalities and uneven power relations at this frontier.

6. References

Abebaw, D., Kassa, H., Kassie, G.T., Lemenih, M., Campbell, B., Teka, W., 2012. Dry forest based livelihoods in resettlement areas of Northwestern Ethiopia. For. Policy Econ. 20, 72–77. https://doi.org/10.1016/j.forpol.2012.02.002

Aguiar, A.P.D., Câmara, G., Escada, M.I.S., 2007. Spatial statistical analysis of land-use determinants in the Brazilian Amazonia: Exploring intra-regional heterogeneity. Ecol. Model. 209, 169–188. https://doi.org/10.1016/j.ecolmodel.2007.06.019

Aide, T.M., Clark, M.L., Grau, H.R., López-Carr, D., Levy, M.A., Redo, D., Bonilla-Moheno, M., Riner, G., Andrade-Núñez, M.J., Muñiz, M., 2013. Deforestation and Reforestation of Latin America and the Caribbean (2001-2010). Biotropica 45, 262–271. https://doi.org/10.1111/j.1744-7429.2012.00908.x

Alencar, L., Rodrigues, A.A., Parry, L., Melo, F.P.L., 2022. Forest security as a fourth dimension of the water-energy-food nexus: empirical evidence from the Brazilian Caatinga, in: The Water-Energy- Food NEXUS: What the Brazilian Research Has to Say. University of São Paulo, São Paulo, Brazil.

Angelsen, A., Wunder, S., 2003. Exploring the forest - poverty link. Center for International Forestry Research (CIFOR). https://doi.org/10.17528/cifor/001554

Antongiovanni, M., Venticinque, E.M., Matsumoto, M., Fonseca, C.R., 2020. Chronic anthropogenic disturbance on Caatinga dry forest fragments. Journal of Applied Ecology 57, 2064–2074. https://doi.org/10.1111/1365-2664.13686

Araujo, H.F.P. de, Machado, C.C.C., Pareyn, F.G.C., Nascimento, N.F.F. do, Araújo, L.D.A., Borges, L.A. de A.P., Santos, B.A., Beirigo, R.M., Vasconcellos, A., Dias, B. de O., Alvarado, F., Silva, J.M.C. da, 2021. A sustainable agricultural landscape model for tropical dry lands. Land Use Policy 100, 104913. https://doi.org/10.1016/j.landusepol.2020.104913

Arroyo-Rodríguez, V., Fahrig, L., Tabarelli, M., Watling, J.I., Tischendorf, L., Benchimol, M., Cazetta, E., Faria, D., Leal, I.R., Melo, F.P.L., Morante-Filho, J.C., Santos, B.A., Arasa-Gisbert, R., Arce-Peña, N., Cervantes-López, M.J., Cudney-Valenzuela, S., Galán-Acedo, C., San-José, M., Vieira, I.C.G., Slik, J.W.F., Nowakowski, A.J., Tscharntke, T., 2020. Designing optimal human-modified landscapes for forest biodiversity conservation. Ecol. Lett. https://doi.org/10.1111/ele.13535

Atlas Brasil, 2020. Atlas do Desenvolvimento Humano no Brasil. URL http://www.atlasbrasil.org.br (accessed 8.10.20).

Babigumira, R., Angelsen, A., Buis, M., Bauch, S., Sunderland, T., Wunder, S., 2014. Forest Clearing in Rural Livelihoods: Household-Level Global-Comparative Evidence. World Dev. 64, S67–S79. https://doi.org/10.1016/j.worlddev.2014.03.002

Barbier, E.B., 2020a. Long run agricultural land expansion, booms and busts. Land Use Policy, Mitigating boom & bust cycles: the roles of land policy and planning 93, 103808. https://doi.org/10.1016/j.landusepol.2019.01.011

Barbier, E.B., 2020b. Is green rural transformation possible in developing countries? World Dev. 131, 104955. https://doi.org/10.1016/j.worlddev.2020.104955

Barbier, E.B., 2012. Scarcity, frontiers and development. Geogr. J. 178, 110–122. https://doi.org/10.1111/j.1475-4959.2012.00462.x

Barbier, E.B., 2004. Agricultural Expansion, Resource Booms and Growth in Latin America: Implications for Long-run Economic Development. World Dev. 32, 137–157. https://doi.org/10.1016/j.worlddev.2003.07.005

Barbier, E.B., Hochard, J.P., 2018. Land degradation and poverty. Nat. Sustain. 1, 623–631. https://doi.org/10.1038/s41893-018-0155-4

Bärnighausen, T., Tugwell, P., Røttingen, J.-A., Shemilt, I., Rockers, P., Geldsetzer, P., Lavis, J., Grimshaw, J., Daniels, K., Brown, A., Bor, J., Tanner, J., Rashidian, A., Barreto, M., Vollmer, S., Atun, R., 2017. Quasi-experimental study designs series—

paper 4: uses and value. J. Clin. Epidemiol. 89, 21–29. https://doi.org/10.1016/j.jclinepi.2017.03.012

Biland, M., Zeb, A., Ullah, A., Kaechele, H., 2021. Why Do Households Depend on the Forest for Income? Analysis of Factors Influencing Households' Decision-Making Behaviors. Sustainability 13, 9419. https://doi.org/10.3390/su13169419

Bivand, R., Piras, G., 2015. Comparing Implementations of Estimation Methods for Spatial Econometrics. J. Stat. Softw. 63, 1–36. https://doi.org/10.18637/jss.v063.i18

Brockhaus, M., Di Gregorio, M., Djoudi, H., Moeliono, M., Pham, T.T., Wong, G.Y., 2021. The forest frontier in the Global South: Climate change policies and the promise of development and equity. Ambio 50, 2238–2255. https://doi.org/10.1007/s13280-021-01602-1

Cardozo, S.A., Martins, H., 2020. New Developmentalism, Public Policies, and Regional Inequalities in Brazil: The Advances and Limitations of Lula's and Dilma's Governments. Lat. Am. Perspect. 47, 147–162. https://doi.org/10.1177/0094582X19888507

Carr, D., 2009. Population and deforestation: why rural migration matters. Prog. Hum. Geogr. 33, 355–378. https://doi.org/10.1177/0309132508096031

Castañeda, A., Doan, D., Newhouse, D., Nguyen, M.C., Uematsu, H., Azevedo, J.P., 2018. A New Profile of the Global Poor. World Dev. 101, 250–267. https://doi.org/10.1016/j.worlddev.2017.08.002

Caviglia-Harris, J., Sills, E., Bell, A., Harris, D., Mullan, K., Roberts, D., 2016. Busting the Boom–Bust Pattern of Development in the Brazilian Amazon. World Dev. 79, 82–96. https://doi.org/10.1016/j.worlddev.2015.10.040

Celentano, D., Sills, E., Sales, M., Veríssimo, A., 2012. Welfare Outcomes and the Advance of the Deforestation Frontier in the Brazilian Amazon. World Dev. 40, 850–864. https://doi.org/10.1016/j.worlddev.2011.09.002

Dawson, N., Martin, A., Camfield, L., 2019. Can agricultural intensification help attain Sustainable Development Goals? Evidence from Africa and Asia. Third World Q. 40, 926–946. https://doi.org/10.1080/01436597.2019.1568190

de Souza, A.A., Mingoti, S.A., Paes-Sousa, R., Heller, L., 2021. Combination of conditional cash transfer program and environmental health interventions reduces child mortality: an ecological study of Brazilian municipalities. BMC Public Health 21, 627. https://doi.org/10.1186/s12889-021-10649-4

de Waroux, Y., Baumann, M., Gasparri, N.I., Gavier-Pizarro, G., Godar, J., Kuemmerle, T., Müller, R., Vázquez, F., Volante, J.N., Meyfroidt, P., 2018. Rents, Actors, and the Expansion of Commodity Frontiers in the Gran Chaco. Ann. Am. Assoc. Geogr. 108, 204–225. https://doi.org/10.1080/24694452.2017.1360761

Djoudi, H., Vergles, E., Blackie, R.R., Koame, C.K., Gautier, D., 2015. Dry forests, livelihoods and poverty alleviation: understanding current trends. Int. For. Rev. 17, 54–69. https://doi.org/10.1505/146554815815834868

Dyngeland, C., Oldekop, J.A., Evans, K.L., 2020. Assessing multidimensional sustainability: Lessons from Brazil's social protection programs. Proc. Natl. Acad. Sci. 201920998. https://doi.org/10.1073/pnas.1920998117

Eriksson, H., Blythe, J.L., Österblom, H., Olsson, P., 2021. Beyond social-ecological traps: fostering transformations towards sustainability. Ecol. Soc. 26, art13. https://doi.org/10.5751/ES-12198-260113

Ewers, R.M., 2006. Interaction effects between economic development and forest cover determine deforestation rates. Glob. Environ. Change 16, 161–169. https://doi.org/10.1016/j.gloenvcha.2005.12.001

Finan, T.J., Nelson, D.R., 2001. Making rain, making roads, making do: Climate Research 19, 97–108. https://doi.org/10.3354/cr019097

Fong, C., Hazlett, C., Imai, K., 2018. Covariate balancing propensity score for a continuous treatment: Application to the efficacy of political advertisements. Ann. Appl. Stat. 12, 156–177. https://doi.org/10.1214/17-AOAS1101

Garrett, R.D., Gardner, T.A., Morello, T.F., Marchand, S., Barlow, J., de Blas, D.E., Ferreira, J., Lees, A.C., Parry, L., 2017. Explaining the persistence of low income and environmentally degrading land uses in the Brazilian Amazon. Ecol. Soc. 22.

Giorgi, G.M., Gigliarano, C., 2017. The Gini Concentration Index: a review of the inference literature. J. Econ. Surv. 31, 1130–1148. https://doi.org/10.1111/joes.12185
Grueber, C.E., Nakagawa, S., Laws, R.J., Jamieson, I.G., 2011. Multimodel inference in ecology and evolution: challenges and solutions. Journal of Evolutionary Biology 24, 699–711. https://doi.org/10.1111/j.1420-9101.2010.02210.x

Guedes, G.R., VanWey, L.K., Hull, J.R., Antigo, M., Barbieri, A.F., 2014. Poverty dynamics, ecological endowments, and land use among smallholders in the Brazilian Amazon. Soc. Sci. Res. 43, 74–91. https://doi.org/10.1016/j.ssresearch.2013.09.002

Guo, S., Fraser, M., Chen, Q., 2020. Propensity Score Analysis: Recent Debate and Discussion. Journal of the Society for Social Work and Research 11, 463–482. https://doi.org/10.1086/711393

Hall, C.M., Rasmussen, L.V., Powell, B., Dyngeland, C., Jung, S., Olesen, R.S., 2022. Deforestation reduces fruit and vegetable consumption in rural Tanzania. Proc. Natl. Acad. Sci. U.S.A. 119, e2112063119. https://doi.org/10.1073/pnas.2112063119

IBGE, 2019. Biomas e sitema Costeiro-Marinho do Brasil. IBGE, Instituto Brasileiro de Geografia e Estatística, Rio de Janeiro.

IBGE , 2017. Classificação e caracterização dos espaços rurais e urbanos do Brasil: uma primeira aproximação, Estudos e pesquisas, informação geográfica. IBGE, Instituto Brasileiro de Geografia e Estatística, Rio de Janeiro.

Ickowitz, A., Powell, B., Salim, M.A., Sunderland, T.C.H., 2014. Dietary quality and tree cover in Africa. Glob. Environ. Change 24, 287–294. https://doi.org/10.1016/j.gloenvcha.2013.12.001

Imai, K., Ratkovic, M., 2014. Covariate balancing propensity score. J. R. Stat. Soc. Ser. B Stat. Methodol. 76, 243–263. https://doi.org/10.1111/rssb.12027

Kelejian, H.H., Prucha, I.R., 1998. A Generalized Spatial Two-Stage Least Squares Procedure for Estimating a Spatial Autoregressive Model with Autoregressive Disturbances. The Journal of Real Estate Finance and Economics 17, 99–121. https://doi.org/10.1023/A:1007707430416

Mahapatra, A.K., Shackleton, C.M., 2012. Exploring the relationships between trade in natural products, cash income and livelihoods in tropical forest regions of Eastern India. Int. For. Rev. 14, 62–73. https://doi.org/10.1505/146554812799973217

MapBiomas, 2022. Projeto MapBiomas – Coleção 5.0 da Série Anual de Mapas de Cobertura e Uso de Solo do Brasil. URL https://plataforma.brasil.mapbiomas.org (accessed 2.2.22).

Marques, R.M., Ximenes, S.B., Ugino, C.K., 2018. Governos Lula e Dilma em matéria de seguridade social e acesso à educação superior. Braz. J. Polit. Econ. 38, 526–547. https://doi.org/10.1590/0101-35172018-2784

Mather, A.S., Needle, C.L., 1998. The forest transition: a theoretical basis. Area 30, 117–124. https://doi.org/10.1111/j.1475-4762.1998.tb00055.x

Medeiros, V., Ribeiro, R.S.M., Amaral, P.V.M. do, 2021. Infrastructure and household poverty in Brazil: A regional approach using multilevel models. World Dev. 137, 105118. https://doi.org/10.1016/j.worlddev.2020.105118

Melo, F.P.L., 2017. The Socio-Ecology of the Caatinga: Understanding How Natural Resource Use Shapes an Ecosystem, in: Caatinga: The Largest Tropical Dry Forest Region in South America. Springer, Cham, Switzerland.

Melo, F.P.L., Arroyo-Rodríguez, V., Fahrig, L., Martínez-Ramos, M., Tabarelli, M., 2013. On the hope for biodiversity-friendly tropical landscapes. Trends Ecol. Evol. 28, 462–468. https://doi.org/10.1016/j.tree.2013.01.001

Miller, D.C., Mansourian, S., Gabay, M., Hajjar, R., Jagger, P., Kamoto, J.F.M., Newton, P., Oldekop, J.A., Razafindratsima, O.H., Shyamsundar, P., Sunderland, T., Wildburger, C., 2021. Forests, trees and poverty alleviation: Policy implications of current knowledge. For. Policy Econ. 131, 102566. https://doi.org/10.1016/j.forpol.2021.102566

Miller, D.C., Mansourian, S., Wildburger, C., 2020. Forests, Trees and the Eradication of Poverty: Potential and Limitations 240.

Miyamoto, M., 2020. Poverty reduction saves forests sustainably: Lessons for deforestation policies. World Dev. 127, 104746. https://doi.org/10.1016/j.worlddev.2019.104746

MMA, 2010. Monitoramento do desmatamento nos biomas brasileiros por satélite 58. Ministério do Meio Ambiente, Brasília - Brasil.

Navarro, M., D'Agostino, A., Neri, L., 2020. The effect of urbanization on subjective well-being: Explaining cross-regional differences. Socioecon. Plann. Sci. 71, 100824. https://doi.org/10.1016/j.seps.2020.100824

Nelson, D.R., Finan, T.J., 2009. Praying for Drought: Persistent Vulnerability and the Politics of Patronage in Ceará, Northeast Brazil. Am. Anthropol. 111, 302–316.

Neufeldt, H., Jahn, M., Campbell, B.M., Beddington, J.R., DeClerck, F., De Pinto, A., Gulledge, J., Hellin, J., Herrero, M., Jarvis, A., LeZaks, D., Meinke, H., Rosenstock, T., Scholes, M., Scholes, R., Vermeulen, S., Wollenberg, E., Zougmoré, R., 2013. Beyond climate-smart agriculture: toward safe operating spaces for global food systems. Agric. Food Secur. 2, 12. https://doi.org/10.1186/2048-7010-2-12

Newton, P., Kinzer, A.T., Miller, D.C., Oldekop, J.A., Agrawal, A., 2020. The Number and Spatial Distribution of Forest-Proximate People Globally. One Earth 3, 363–370. https://doi.org/10.1016/j.oneear.2020.08.016

Parry, L., Davies, G., Almeida, O., Frausin, G., de Moraés, A., Rivero, S., Filizola, N., Torres, P., 2018. Social Vulnerability to Climatic Shocks Is Shaped by Urban Accessibility. Ann. Am. Assoc. Geogr. 108, 125–143. https://doi.org/10.1080/24694452.2017.1325726

Pritchard, R., Grundy, I.M., Horst, D. van der, Ryan, C.M., 2019. Environmental incomes sustained as provisioning ecosystem service availability declines along a woodland resource gradient in Zimbabwe. World Dev. 122, 325–338. https://doi.org/10.1016/j.worlddev.2019.05.008

Pritchard, R., Grundy, I.M., van der Horst, D., Dzobo, N., Ryan, C.M., 2020. Environmental resources as 'last resort' coping strategies following harvest failures in Zimbabwe. World Dev. 127, 104741. https://doi.org/10.1016/j.worlddev.2019.104741

Queiroz, L.P., Cardoso, D., Fernandes, M.F., Moro, M.F., 2017. Diversity and Evolution of Flowering Plants of the Caatinga Domain, in: Caatinga: The Largest Tropical Dry Forest Region in South America. Springer, Cham, Switzerland.

Raudsepp-Hearne, C., Peterson, G.D., Tengö, M., Bennett, E.M., Holland, T., Benessaiah, K., MacDonald, G.K., Pfeifer, L., 2010. Untangling the Environmentalist's Paradox: Why Is Human Well-being Increasing as Ecosystem Services Degrade? BioScience 60, 576–589. https://doi.org/10.1525/bio.2010.60.8.4

Razafindratsima, O.H., Kamoto, J.F.M., Sills, E.O., Mutta, D.N., Song, C., Kabwe, G., Castle, S.E., Kristjanson, P.M., Ryan, C.M., Brockhaus, M., Sunderland, T., 2021. Reviewing the evidence on the roles of forests and tree-based systems in poverty dynamics. For. Policy Econ. 131, 102576. https://doi.org/10.1016/j.forpol.2021.102576

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin, F.S.I., Lambin, E., Lenton, T., Scheffer, M., Folke, C., Schellnhuber, H.J., Nykvist, B., de Wit, C., Hughes, T., van der Leeuw, S., Rodhe, H., Sörlin, S., Snyder, P., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R., Fabry, V., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., Foley, J., 2009. Planetary Boundaries: Exploring the Safe Operating Space for Humanity. Ecol. Soc. 14. https://doi.org/10.5751/ES-03180-140232

Rodrigues, A.S.L., Ewers, R.M., Parry, L., Souza, C., Verissimo, A., Balmford, A., 2009. Boom-and-Bust Development Patterns Across the Amazon Deforestation Frontier. Science 324, 1435–1437. https://doi.org/10.1126/science.1174002

Rodrigues, J.F., 2014. O rural e o urbano no Brasil: uma proposta de metodologia de classificação dos municípios. Análise Soc. 28.

Roser, M., Esteban, O.-O., 2013. Global Extreme Poverty. Our World in Data. Available at: https://ourworldindata.org/extreme-poverty. (Accessed: 24 August 2020).

Rudel, T.K., 1998. Is There a Forest Transition? Deforestation, Reforestation, and Development1. Rural Sociol. 63, 533–552. https://doi.org/10.1111/j.1549-0831.1998.tb00691.x

Rutherford, M.E., Mulholland, K., Hill, P.C., 2010. How access to health care relates to under-five mortality in sub-Saharan Africa: systematic review. Trop. Med. Int. Health 15, 508–519. https://doi.org/10.1111/j.1365-3156.2010.02497.x

Sachs, J.D., Warner, A.M., 1998. The big push, natural resource booms and growth. J. Dev. Econ. 59, 43–76. https://doi.org/10.1016/S0304-3878(99)00005-X

Sagar, A.D., Najam, A., 1998. The human development index: a critical review. Ecol. Econ. 25, 249–264. https://doi.org/10.1016/S0921-8009(97)00168-7

Sampaio, E.V.S.B., 2010. Características e potencialidades, in: Uso sustentável e conservação dos recursos florestais da Caatinga. Serviço Florestal Brasileiro, Brasília, DF, p. 368.

Sampaio, E.V.S.B., Menezes, R.S.C., Sampaio, Y. de S.B., Freitas, A.D.S. de, 2017. Sustainable Agricultural Uses in the Caatinga, in: Caatinga: The Largest Tropical Dry Forest Region in South America. Springer, Cham, Switzerland.

Santoro, F.R., Ferreira Júnior, W.S., Araújo, T.A. de S., Ladio, A.H., Albuquerque, U.P., 2015. Does Plant Species Richness Guarantee the Resilience of Local Medical Systems? A Perspective from Utilitarian Redundancy. PLOS ONE 10, e0119826. https://doi.org/10.1371/journal.pone.0119826

Schröder, J.M., Ávila Rodríguez, L.P., Günter, S., 2021. Research trends: Tropical dry forests: The neglected research agenda? For. Policy Econ. 122, 102333. https://doi.org/10.1016/j.forpol.2020.102333

Scott, J.C., 1998. Seeing Like a State: How Certain Schemes to Improve the Human Condition Have Failed. Yale University Press.

Sena, A., Freitas, C., Feitosa Souza, P., Alpino, T., Pedroso, M., Corvalan, C., Barcellos, C., Carneiro, F., 2018. Drought in the Semiarid Region of Brazil: Exposure, Vulnerabilities and Health Impacts from the Perspectives of Local Actors. PLoS Curr. https://doi.org/10.1371/currents.dis.c226851ebd64290e619a4d1ed79c8639

Seymour, F., Harris, N.L., 2019. Reducing tropical deforestation. Science 365, 756–757. https://doi.org/10.1126/science.aax8546

Silva, J.M.C. da, Barbosa, L.C., 2017. Impact of Human Activities on the Caatinga, in: Caatinga: The Largest Tropical Dry Forest Region in South Americad. Springer, Cham, Switzerland.

Silva, J.M.C. da, Barbosa, L.C., Leal, I.R., Tabarelli, M., 2017a. The Caatinga: Understanding the Challenges, in: Caatinga: The Largest Tropical Dry Forest Region in South America. Springer, Cham, Switzerland.

Silva, J.M.C. da, Prasad, S., Diniz-Filho, J.A.F., 2017b. The impact of deforestation, urbanization, public investments, and agriculture on human welfare in the Brazilian Amazonia. Land Use Policy 65, 135–142. https://doi.org/10.1016/j.landusepol.2017.04.003

Smith, H.E., Ryan, C.M., Vollmer, F., Woollen, E., Keane, A., Fisher, J.A., Baumert, S., Grundy, I.M., Carvalho, M., Lisboa, S.N., Luz, A.C., Zorrilla-Miras, P., Patenaude, G., Ribeiro, N., Artur, L., Mahamane, M., 2019. Impacts of land use intensification on human wellbeing: Evidence from rural Mozambique. Glob. Environ. Change 59, 101976. https://doi.org/10.1016/j.gloenvcha.2019.101976

Souen, J.A., de Souza Campos, G.C., 2019. From Euphoria to Retreat: Formal Employment in Twenty-first-Century Brazil. Lat. Am. Perspect. 46, 66–85. https://doi.org/10.1177/0094582X19845383

Souza Jr, C.M., Shimbo, J.Z., Rosa, M.R., Parente, L.L., Alencar, A.A., Rudorff, B.F.T., Hasenack, H., Matsumoto, M., Ferreira, L.G., Souza-Filho, P.W.M., 2020. Reconstructing Three Decades of Land Use and Land Cover Changes in Brazilian Biomes with Landsat Archive and Earth Engine 27.

Steffen, W., Richardson, K., Rockström, J., Cornell, S.E., Fetzer, I., Bennett, E.M., Biggs, R., Carpenter, S.R., Vries, W. de, Wit, C.A. de, Folke, C., Gerten, D., Heinke, J., Mace, G.M., Persson, L.M., Ramanathan, V., Reyers, B., Sörlin, S., 2015. Planetary boundaries: Guiding human development on a changing planet. Science 347. https://doi.org/10.1126/science.1259855

Sunderland, T., Apgaua, D., Baldauf, C., Blackie, R., Colfer, C., Cunningham, A.B., Dexter, K., Djoudi, H., Gautier, D., Gumbo, D., Ickowitz, A., Kassa, H., Parthasarathy, N., Pennington, R.T., Paumgarten, F., Pulla, S., Sola, P., Tng, D., Waeber, P., Wilmé, L., 2015. Global dry forests: a prologue. Int. For. Rev. 17, 1–9. https://doi.org/10.1505/146554815815834813

Sunderlin, W., Dewi, S., Puntodewo, A., 2008. Poverty and forests: multi-country analysis of spatial association and proposed policy solutions (No. 47), CIFOR Occasional paper. CIFOR, Jakarta.

Tabarelli, M., Leal, I.R., Scarano, F.R., Silva, J.M.C. da, 2017. The Future of the Caatinga, in: Caatinga: The Largest Tropical Dry Forest Region in South America. Springer, Cham, Switzerland.

Thaler, G.M., Viana, C., Toni, F., 2019. From frontier governance to governance frontier: The political geography of Brazil's Amazon transition. World Dev. 114, 59–72. https://doi.org/10.1016/j.worlddev.2018.09.022

Thypin-Bermeo, S., Godfrey, B.J., 2012. Envisioning Amazonian frontiers: place-making in a Brazilian boomtown. J. Cult. Geogr. 29, 215–238. https://doi.org/10.1080/08873631.2012.684948

Timko, J., Le Billon, P., Zerriffi, H., Honey-Rosés, J., de la Roche, I., Gaston, C., Sunderland, T.C., Kozak, R.A., 2018. A policy nexus approach to forests and the SDGs: tradeoffs and synergies. Curr. Opin. Environ. Sustain. 34, 7–12. https://doi.org/10.1016/j.cosust.2018.06.004

Tritsch, I., Arvor, D., 2016. Transition in environmental governance in the Brazilian Amazon: emergence of a new pattern of socio-economic development and deforestation. Land Use Policy 59, 446–455. https://doi.org/10.1016/j.landusepol.2016.09.018

Wan Mahari, W.A., Azwar, E., Li, Y., Wang, Y., Peng, W., Ma, N.L., Yang, H., Rinklebe, J., Lam, S.S., Sonne, C., 2020. Deforestation of rainforests requires active use of UN's Sustainable Development Goals. Sci. Total Environ. 742, 140681. https://doi.org/10.1016/j.scitotenv.2020.140681

Watts, M.J., 2018. Frontiers: Authority, Precarity, and Insurgency at the Edge of the State. World Dev. 101, 477–488. https://doi.org/10.1016/j.worlddev.2017.03.024

Weinhold, D., Killick, E., Reis, E.J., 2013. Soybeans, Poverty and Inequality in the Brazilian Amazon. World Dev. 52, 132–143. https://doi.org/10.1016/j.worlddev.2012.11.016

Weinhold, D., Reis, E.J., Vale, P.M., 2015. Boom-bust patterns in the Brazilian Amazon. Glob. Environ. Change 35, 391–399. https://doi.org/10.1016/j.gloenvcha.2015.09.013

Worku, A., Pretzsch, J., Kassa, H., Auch, E., 2014. The significance of dry forest income for livelihood resilience: The case of the pastoralists and agro-pastoralists in the drylands of southeastern Ethiopia. For. Policy Econ. 41, 51–59. https://doi.org/10.1016/j.forpol.2014.01.001

Zaefarian, G., Kadile, V., Henneberg, S.C., Leischnig, A., 2017. Endogeneity bias in marketing research: Problem, causes and remedies. Industrial Marketing Management 65, 39–46. https://doi.org/10.1016/j.indmarman.2017.05.006

3 ARTIGO 2 – Tradeoffs and synergies between food security and forest cover in Brazilian drylands

Lucas Alencar[1]; Cristina Baldauf[2]; Adriana Pellegrini Manhães[3]; Felipe Pimentel Lopes Melo[1]

[1] Universidade Federal de Pernambuco, Centro de Biociências, Recife-PE Brazil

[2] Universidade Federal Rural do Semi Árido, Mossoró - RN, Brazil

[3] Universidade Federal do Rio de Janeiro, Rio de Janeiro - RJ, Brazil

Abstract

Global food demand is expected to increase in the next decades pushing agricultural expansion and deforestation. However, food production in agricultural lands is just one dimension of food security, to which forest goods and services also contribute. In this paper, we aimed to explore the relationship between forest cover and food security. We hypothesised that food security is improved by both human-made and green infrastructure combined. To test this relationship, we explore the relationships between forest cover and a multidimensional index of food security that included both socioeconomic and natural variables taken from Brazilian official databases for 1,141 municipalities of the Brazilian Caatinga (a seasonally tropical dry forest). The index was formed by 12 principal components axes (12 PCs) and we found that financial poverty (PC 1) and economic inequality (PC 2) were the main determinants of food insecurity in Caatinga. We found that lowest food security values were found in two contrasting contexts: one is represented by poor and unequal municipalities with high forest cover while the other refers to poor and less unequal municipalities but with little forest cover. Municipalities with intermediate levels of forest cover had slightly higher food security, a consistent pattern across time (2006 and 2017). Win-win scenarios where both forest cover and food security increased with time were almost as common as lose-lose situations (25% and 22% respectively). This suggests a sort of balance between forests and human-made land uses and reinforces that natural capital contributes to food security. Zero-hunger is a main issue for sustainable development goals and our results adds to the notion that both sustainable use of forests and socioeconomic improvements must coexist rather than being treated as antagonistic policies.

Keywords: Dry forests, sustainable development goals, social ecological systems, natural resources, poverty, inequality

Introduction

One of the major global challenges for the next decades is to increase food security while preserving natural habitats (Godfray et al., 2010; Latawiec et al., 2015). Despite the technologies to increase agricultural productivity on lands already in use, increase in food production historically comes via the expansion of agriculture over natural habitats (Alexander et al., 2015), placing industrial food systems among the main drivers of land-use change (Foley et al., 2011; IPBES, 2019). More food production, however, does not guarantee access and conversion of natural habitats into agricultural fields has limited impact on food security due to access limitations (FAO, 2020). On the other hand, natural habitats can play a major role on food security for rural populations as sources of plants, fisheries, wild meat and insects, for example

(Baudron et al., 2019). Access to forests can alleviate poverty by allowing many kinds of traditional management practices such as slash-and-burn agriculture, extensive pastoralism and diverse types of extractivism (Jagger et al., 2022). Understanding the relationships between food security and forests is crucial to achieve biodiversity-friendly schemes of food production such as: sustainable intensification of agricultural production (Pretty & Bharucha, 2014) and crop yield improvement (Schütz et al., 2018) that must reduce demand for new lands and therefore, halt deforestation (Lambin et al., 2018). Fighting climate change while the population grows demands conciliating food security and protection of forests (Melo et al., 2021). For this, we need a paradigm shift that places food security in all its dimensions, including not only the access to the products of agricultural fields but the access to forest goods and services.

According to the Food and Agriculture Organisation of the United Nations (FAO, 2013), there are four main dimensions of food security. First, availability, refers to the amount of food available in the system, considering, among others, food production and population size (Burchi & De Muro, 2016). Then, utilisation refers to how people cook, process and store the food available in the system, but this dimension is also related to water and sanitation issues, both affecting food utilisation and health (Ericksen, 2008; FAO, 2020). A third aspect, stability, deals with the capacity of the system to guarantee food supply in the face of several types of disturbances, such as climate, market or political instabilities (Kah, 2017; Tendall et al., 2015). Finally, access is the capacity of people to access the food produced in the system, either buying it or being capable of producing it themselves (FAO, 2013; Klassen & Murphy, 2020). The last is thought to be the main cause of food insecurity because the amount of food needed to meet current basic world population demands is actually produced (Barrett, 2010). Economic poverty precludes access to the food market while utilisation of natural resources can alleviate food insecurity of rural poor (Miller & Hajjar, 2020). Therefore, addressing food insecurity in a broad way demands understanding the nature of the relationship between extent of natural habitats and the several dimensions of food security.

Although the scientific literature highlights the tradeoff between the establishment of new crop fields and the increasing deforestation (Meyfroidt, 2018), forest can help to improve food security (Miller et al., 2020; Rasmussen et al., 2020). Agroforestry systems, for example, can contribute to improve availability, access and stability of food systems at regional scales (Rosenstock et al., 2019). In many poor regions of tropical countries, forests are used as grazing fields for extensive pastoralism practices while helping to maintain forest cover (Alencar et al., 2022; Baudron et al., 2019). Despite this important linkages between forests and food security have been recently recognised (Bahar et al., 2020), the forest-food nexus needs to be better explored (Melo et al., 2021). Important knowledge gaps on the role of natural habitats for poverty alleviation and food security still persist and limit the quantification of forests to food security. Because a diverse set of natural and socioeconomic drivers can affect food security, we need to test and re-create ways of measuring the determinants of access to food.

Human-made or “grey” infrastructure is surely required to improve food security (Devereux, 2016). For example, roads help to guarantee access to markets and water dams to irrigation schemes (de Fraiture & Wichelns, 2010; Khan et al., 2009). The same is true for the “green” infrastructure (sensu Silva and Barbosa 2017) if we consider natural habitats as complementary sources of food items. However, current development models usually replace “green” by “grey” infrastructure and threaten landscapes of crossing a tipping point that compromises the ability of natural habitats to provide the services and goods that may improve food security (Swift & Hannon, 2010). Little grey infrastructure is a signal of underdevelopment that might reduce food availability due to limited access to food markets (Khan et al., 2009). On the other hand, little remaining natural habitats may represent a lack of complementary source of food for people, thus also reducing food security (Vysochyna et al., 2020). If it is true, it is reasonable to expect that a combination of better social indicators and enough natural habitats must provide better food security. At least theoretically, this is in accordance with the concept of “optimal landscapes” that both preserves natural habitats and produce food in a landscape structure that allows the high levels of food production, ecosystem services and biodiversity conservation (Arroyo-Rodríguez et al., 2020).

The Brazilian seasonally dry tropical forest, also known as Caatinga, constitutes an opportunity for assessing the trade-offs and synergies between food security and forests. This region is characterised by high levels of social vulnerability (Hummell et al., 2016) and low food security when compared to the other regions of Brazil (Gubert et al., 2017). Around 60% of its forest cover is preserved, though severely degraded and fragmented (Antongiovanni et al., 2018). People in this region depend largely on small-scale agriculture and extensive pastoralism that are periodically affected by seasonal droughts, thus reducing food availability (Melo, 2017). Natural resources are therefore an important asset for the 28 million people living in this dry forest and are likely to provide both goods and services that contribute to food security. The objective of this work is to understand the nature of the relationship between food security and forests at the regional scale. For this we focused on: 1) identifying the main socioeconomic indicators of food insecurity in the Caatinga through a multidimensional index of food security and; 2) understanding the spatial configuration of food insecurity and deforestation. Our results offer important evidence on the contribution of forests to food security that should be useful to regional landscape management and challenge the notion that current models of development based on land-use change can alleviate poverty and food insecurity.

Methods

The Caatinga socioecological system

The Caatinga dry forest is characterised by an extended dry period in which rainfall is scarce, high evaporation rates and great inter-and intra-annual rainfall variation (Andrade et al., 2017). The rainfall amount goes from 400 to 1500 mm per year across the biome, being considered the wettest seasonally dry forest in the world (Andrade et al., 2017). The vegetation is dominated by small-leaved, thorny trees and several species of succulents (Queiroz et al., 2017). Water for irrigation comes mostly from rivers and wells, but as almost all rivers are intermittent, the largest share of the agriculture is solely rain-fed or relies on rainfall accumulated in the wetter valleys (Sampaio et al., 2017). Water for human consumption comes from diverse sources (e.g. wells, cisterns, water tank trucks), but they are not necessarily of adequate quality, which can impact local food security (Sena et al., 2018). The Caatinga has very diverse soil conditions, but most of the areas present low fertility which reduces the agricultural aptitude (Sampaio et al., 2017). Combined, those environmental characteristics contribute to the low productivity of the agricultural system and to poor socioeconomic conditions of most farmers (Tabarelli et al., 2017).

The set of municipalities comprising the Caatinga have the lowest levels of human development in Brazil (Silva et al., 2017). Most of the municipalities are considered rural (IBGE, 2017), where most farmers produce maize and beans associated with small-scale animal production for their subsistence (Sampaio et al., 2017). Goat production is the main strategy that poor rural farmers use worldwide in arid and semiarid regions (Caatinga included) especially during long droughts because of their tolerance to such climatic conditions and their ability to feed on natural vegetation (Devendra, 1999). Bovine production is mostly developed in a pasture with exotic grasses; however, only capitalised farmers normally engage in cattle production (Sampaio et al., 2017). Furthermore, the Caatinga occurs in one of the most densely populated semi-arid regions in the world (Silva et al., 2017) with millions of people living in rural communities highly dependent on forest products (Albuquerque et al., 2017). Although the Caatinga still has intermediate levels of forest vegetation cover, the forest fragments are not evenly distributed across the biome (Antongiovanni et al., 2018) and have high levels of chronic anthropogenic disturbance linked to the historical use of native vegetation by local people (Antongiovanni et al., 2020).

Food security index and forest cover change

We developed an index of food security for the Caatinga using the four dimensions described in FAO (2013) to select the variables to represent each dimension. All variables were collected from the Brazilian official sources (e.g. IBGE, Ministry of Health) using 1,141 municipalities of the Caatinga as sample units (Figure 1). We selected 38 variables that are summarised in Table 1. The respective premise and sources of each variable are in Supplementary Table 1. It is important to notice that some variables seem to be redundant or misleading. For example, it may look redundant to include variables on both high and low Body Mass Index, but actually they capture two extremes of a healthy weight for adults which has different causes and consequences. Overweight is now considered one of the triple burdens of food insecurity (i.e., undernutrition, hidden hunger and overweight) and is related to a diet rich in ultra processed food, causing many health problems in adulthood (UNICEF, 2019). The lack of nutritious food, on the other hand, could cause undernourishment and low weight, especially if the lack of food occurs during early childhood, which could have negative health consequences for lifetime (FAO, 2013). Another example is the variable related to the social organisation of farmers. It is still debated in the literature if cooperatives always bring benefits to small-farmers, however, in general, they do contribute to rural development and to reduce poverty among small-holder farmers (Johnson & Shaw, 2014). Some studies argue the issue of inclusiveness to be a key point in determining the real contribution of cooperativism to rural development (Bijman & Wijers, 2019). In Brazil, cooperativism is considered an important instrument of rural development and tends to contribute positively to small farmers, despite some farmers being excluded (Stattman & Mol, 2014). Therefore, even if this contribution is not for all farmers or not evenly distributed among them, we believe that the organisation of producers should have a positive effect on food systems and assume a positive impact on food security.

Table 1. Variables used to build a food security index for the Caatinga municipalities and the respective measure and brief definition.

Type of variable	**Measure**	**Definition**
Access	Income	Average per capita income of households in each municipality
	Half minimum wage	Percentage of the population of the municipality with an income below half of the minimum wage
	A quarter of minimum wage	Percentage of the population of the municipality with an income below a quarter of the minimum wage
	Unemployment rate	Percentage of the number of people in the municipality over 16 who are unemployed
	Low Mass Body Index	Percentage of the number of people in the municipality with body mass index below ideal
	Height deficit by age	Percentage of the number of children below height expected for their age
	Weight deficit by age	Percentage of the number of children below expected weight for their age
	Illiteracy rate	Percentage of the population aged 25 and over who is illiterate
	High school	Percentage of the population aged 25 or over who has completed high school
	Graduation level	Percentage of the population aged 25 or over who has completed higher education
	Poorest fifth	Average per capita income of the poorest fifth in the municipality
	Extreme poverty	Percentage of the population of municipalities in extreme poverty
	Poverty	Percentage of the population living in poverty
	Income of the richest	Percentage of total municipal income appropriated by 20% of the population with the highest per capita household income
	Gini index	Gini index of income
	Under-five mortality	Probability of dying between birth and the exact age of 5, per 1000 children born alive
Availability	Agriculture workers	Percentage of the population aged 18 or overworking in the agricultural sector

	Rural population	Percentage of municipality inhabitants in rural areas
	Urban population	Percentage of municipality inhabitants in urban areas
	Pesticides	percentage of establishments using pesticides in the production
	Irrigation	percentage of establishments with any types of irrigation
	Corn production	Municipal average corn production
	Bean production	Municipal average beans production
	Bovine production	Municipal livestock production of cattle
	Caprine production	municipal goat production
	Woman headed establishments	Percentage of establishments headed by women
Stability	Dependency ratio	The proportion of the population under 15 and over 65 related to the population between 15 and 64 years
	Associated producers	Percentage of producers associated with cooperatives and/or class entities
	Protected springs	Percentage of establishments with springs protected by forest
	Springs not protected	Percentage of establishments with springs not protected by forest
	Protected rivers	Percentage of establishments with rivers protected by forest
	Rivers not protected	Percentage of establishments with rivers not protected by forest
	Financing (Rural credit?)	percentage of the total number of establishments with financing (?)
Utilisation	High Body Mass Index	Percentage of the number of people in the municipality with obesity
	Sewage	Percentage of people in households with an inadequate water supply and sanitation
	Organic Agriculture	percentage of establishments with organic production
	Wells	Percentage of households with common wells
	Cisterns	Percentage of households with cisterns

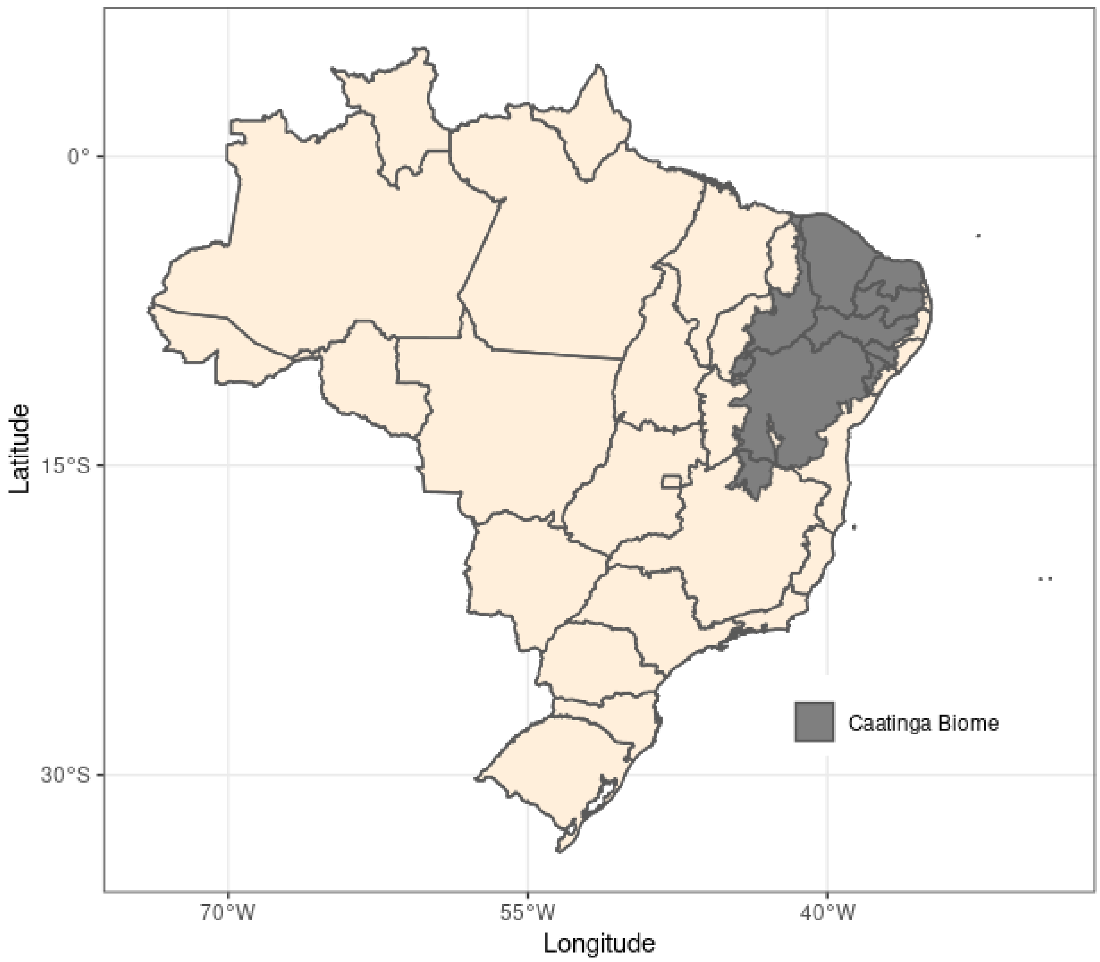


Figure 1. Map of Brazil identifying the Caatinga biome from where the data of 1141 municipalities were used to understand the relationship between forest and food security.

We built a multidimensional food security index for two time-points (2006 and 2017) to understand how it changed over time and how it is related to forest cover change. We built one index using a Principal Component Analysis for each year to reduce a large number of variables into a few dimensions following a method proposed by Hummel et al (2016). We selected only the PCs with an eigenvalue greater than 1 (Cutter et al., 2003) and then we changed the cardinality of each PC depending on whether the variables that compose the PC contribute positively or negatively to food security (Supplementary Table 2). The final food security index for each year was calculated as the sum of the individual scores of each PC for each municipality based on Hummel et al (2016). We estimated the food security change in the municipalities by calculating the difference between the final index score between 2006 and 2017.

We gathered the data of native vegetation cover from the MapBiomas project (MapBiomas, 2020). We considered all categories of forest (forest, savanna, mangrove and forest plantations) and non-forested native vegetation (wetland, grassland, salt flats, rocky outcrops and other non-forest formations) to calculate the forest cover percentage for each municipality. We grouped all types of native vegetation into one class (i.e., native vegetation cover) because people in the Caatinga derive many uses related to food security from all types of native vegetation and not only from forests (Albuquerque et al., 2017). Then, we identified and spatialized the municipalities with synergies (win-win and lose-lose) and trade-offs (win-lose and lose-win) between forest cover and food security change, respectively. We considered all municipalities that gained food security from 2006 to 2017 as a ‘win’ group, irrespective of the scores’ size, as well as ‘lose’ if the food security score was below zero.

Statistical analyses

We first used a Principal Components Analysis to reduce dimensionality of our Multidimensional Food Security Index (MFSI). We used z-transformation to standardise scores of the principal components and calculate the MSFI. Because we changed the cardinality of the dimensions to always increase MFSI, the index is the sum of both positive (increasing security) and negative values (decreasing security). Because the dimensions were formed by different variables across years we provided a table to help readers interpret what drives food insecurity (Table S1). The absolute changes of values of MFSI and forest cover were used to create maps that help to understand spatial distribution of both changes in forest cover and food security. To understand the relationship between forest cover and MFSI and its two most important principal components in 2006 and 2017 we built spatial regression models with a quadratic term of forest cover when testing the effect of forest cover on MFSI and its first principal component (PC-1) and without quadratic term for the second PC. In all models, spatial error was included to check for nonlinear relationships. We used the errorsalm function from the package 'spdep' in the R environment. Moran I test, using the function moran.test was then used to test whether after accounting for spatial error, there was still spatial autocorrelation of the residuals.

Results

Multidimensional Food Security Index

Our proposed Multidimensional Food Security Index (MFSI) was composed of 12 dimensions (principal components) that cover very different aspects of food security and explained up to 70% of the whole variance contained within the 38 variables for both years (2006 and 2017, Table S1 and S2). Economic poverty was by far the most important dimension followed by socioeconomic inequality (see Table2 and Tables S1 and S2). More than half of the variance of the MFSI is attributed to the remaining 10 dimensions but with little contribution of each one, individually. There was a shift in the rank of importance of dimensions and small changes on the components - or variables, of each dimension between years. For example, the third most important dimension in

2006 was made by variables linked to the financial stability in food production alone, however, in 2017 gender inequality (an availability variable) had a more important role in this dimension. Environmental protection was placed at 12th dimension in 2006, increased its importance in 2017 and was ranked as the 4th most important dimension. Many other changes alike happened between years suggesting that food security can be secondarily influenced by many different drivers across time (Table 2).

Table 2. Principal Components of the food security index for 2006 and 2017 and the percentage of explained variance. Numbers refer to the order of importance of each principal component and the names are the author's interpretation of the subset of variables loaded for each principal component.

Name of Principal Component	Year	% of variance explained
1 – Poverty	2006	25.51%
1 – Poverty	2017	26.22%
2 – Inequality	2006	7.11%
2 – Inequality	2017	8.25%
3 – Farmer’s Stability	2006	6.76%
3 – Gender inequality and stability	2017	5.76%
4 – Child Nutrition	2006	4.38%
4 – Forest cover in water bodies	2017	5.59%
5 – Goat herd	2006	4.30%
5 – Farmer’s stability	2017	4.86%
6 – Availability	2006	4.26%

6 – Agricultural Production	2017	4.00%
7 – Agricultural intensification	2006	3.67%
7 – Child Nutrition	2017	3.72%
8 – Agricultural production	2006	3.51%
8 – Goat herd	2017	3.39%
9 – Cattle herd	2006	3.30%
9 – Forest cover in water bodies	2017	3.25%
10 – Protection of water resources	2006	3.16%
10 – Unemployment	2017	2.95%
11 – Utilisation	2006	2.79%
11 – Adult Nutrition	2017	2.78%
12 – Forest cover in water bodies	2006	2.66%
12 – Availability	2017	2.50%

Changes in forest cover and food security

Among the 525 municipalities that experienced net forest loss, deforestation averaged 4.04 ± 6.66% of the total vegetated area from 2006 to 2017. On the other hand, 616 municipalities had an increase in forest cover of 3.34 ± 3.22% in the same period. These values were surprisingly low but suggest small net loss of forest for the whole Caatinga. Only 19 municipalities experienced net forest gains greater than 10% while 39 lost more than 10% of forests between 2006 and 2017 (Fig. 2a). Regarding food security, 561 municipalities registered a decrease in food security while the remaining 580 improved this measure from 2006 to 2017. We found a large variability in food security change, that goes from small changes to three orders of magnitude (>32,000

%) of net gain or net loss (Fig. 2b). Overall, we found that 29% of the municipalities lost forest and gained food security (Fig. 3) between 2006 and 2017. These were followed by the municipalities that gained forest while losing food security (24 %). The worst scenario was registered for 25 % of the municipalities that lost both forest and food security in the time period. However, the optimal combination of gaining both forest cover and food security was registered in almost 22% of the municipalities (Fig. 3).

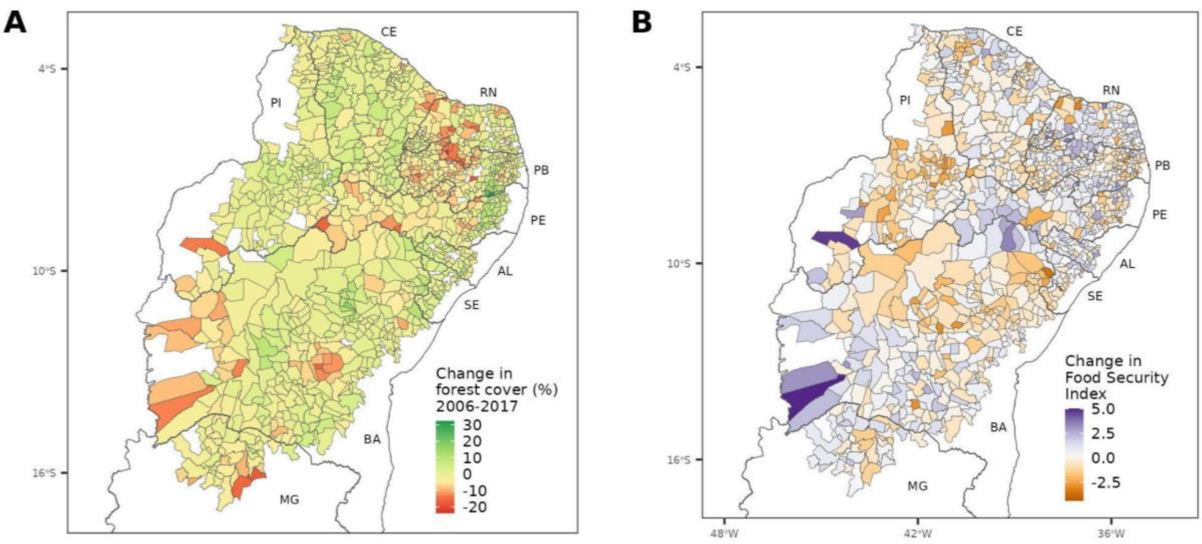


Figure 2. Map of the 1114 municipalities of the Caatinga region showing variation in forest cover (a) and food security index (b) between the years of 2006 and 2017. The food security index was built through a Principal Component Analysis using several variables related to access, availability, stability and utilisation of food. The value for each municipality was calculated as the sum of the individual scores of each PC (with eigenvalue greater than 1) for each municipality, and then we calculated the difference from 2006 to 2017. Therefore, positive values of change means an increase in food security and negative values, a decrease from one year to another.

Also, the correlation between forest cover and food security was complex and spatially structured (all models performed better with spatial error) and, as expected, presented a boom-bust pattern described by a weak but significant quadratic function for both years (for 2006; $z = -2.40$, $p=0.016$ and for 2017; $z = -3.13$, $p = 0.001$). Briefly, food security tends to increase in mid-deforested and reaches its peak around 50% of deforestation when it comes to drop again to similar levels of food security found in highly forested municipalities. A deeper analysis suggests that this pattern is mostly driven by economic poverty that presents a u-shaped curve in response to forest cover for both years (Fig 4.) However, the second most important dimension (PC 2) of food security presented a direct relationship with forest cover suggesting that inequality increases in more forested areas. As a result, low levels of food security can be grouped in two types: 1) poor people living in forested areas with social inequality and; 2) poor people living in deforested areas with smaller social inequality. In our study, the tradeoffs (gain-lose or lose-gain) group the poorest municipalities, but the ones losing forest and gaining food security are more unequal (in terms of income) while the ones gaining forest and losing food security are less unequal.

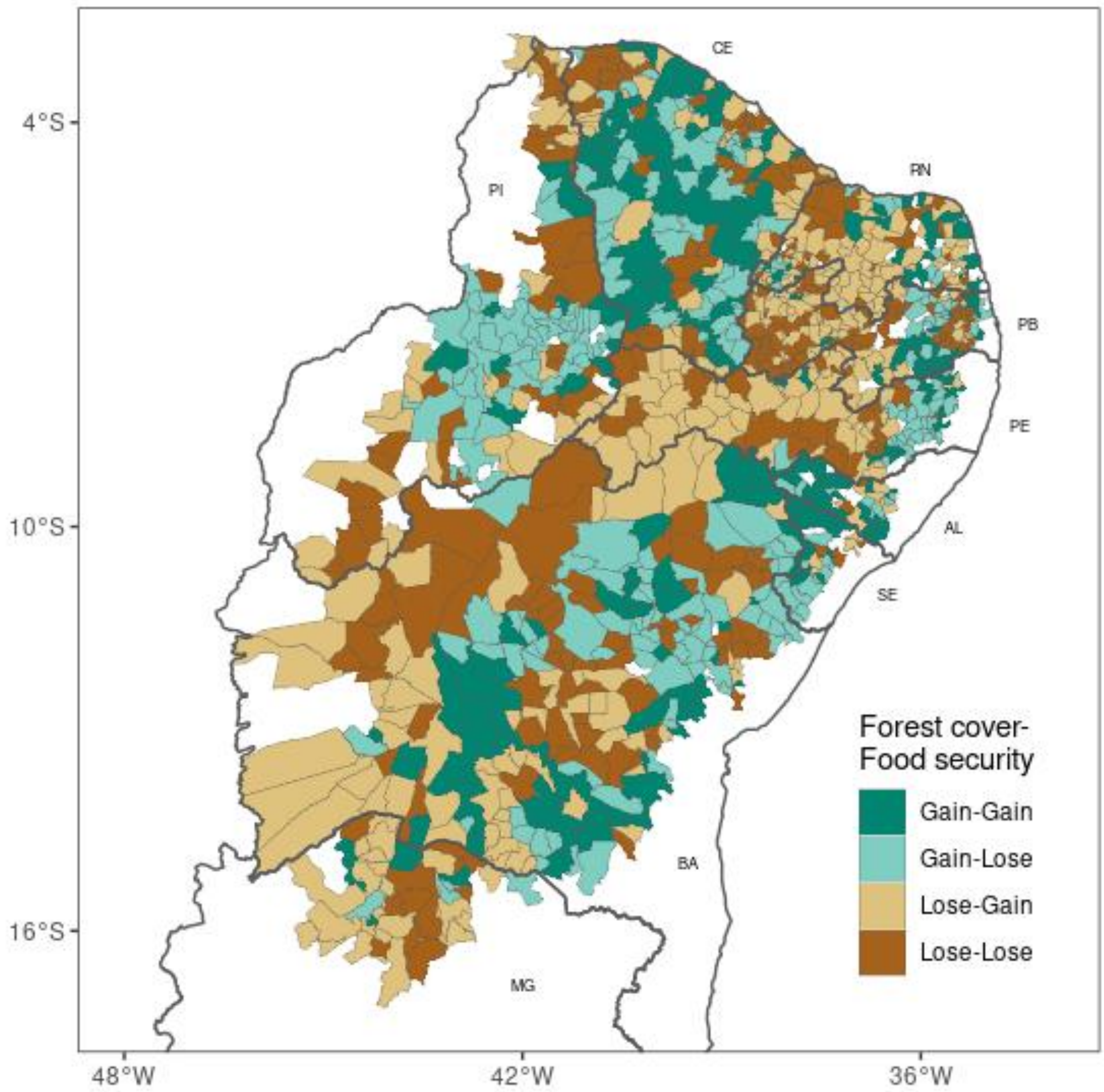


Figure 3. Geographical distribution of municipalities that experienced combinations of either gains or losses of forest cover and food security, between the years of 2006 and 2017. In the legend levels, the first word always refers to forest cover and the second word, after hyphen, refers to food security (either gain or loss).

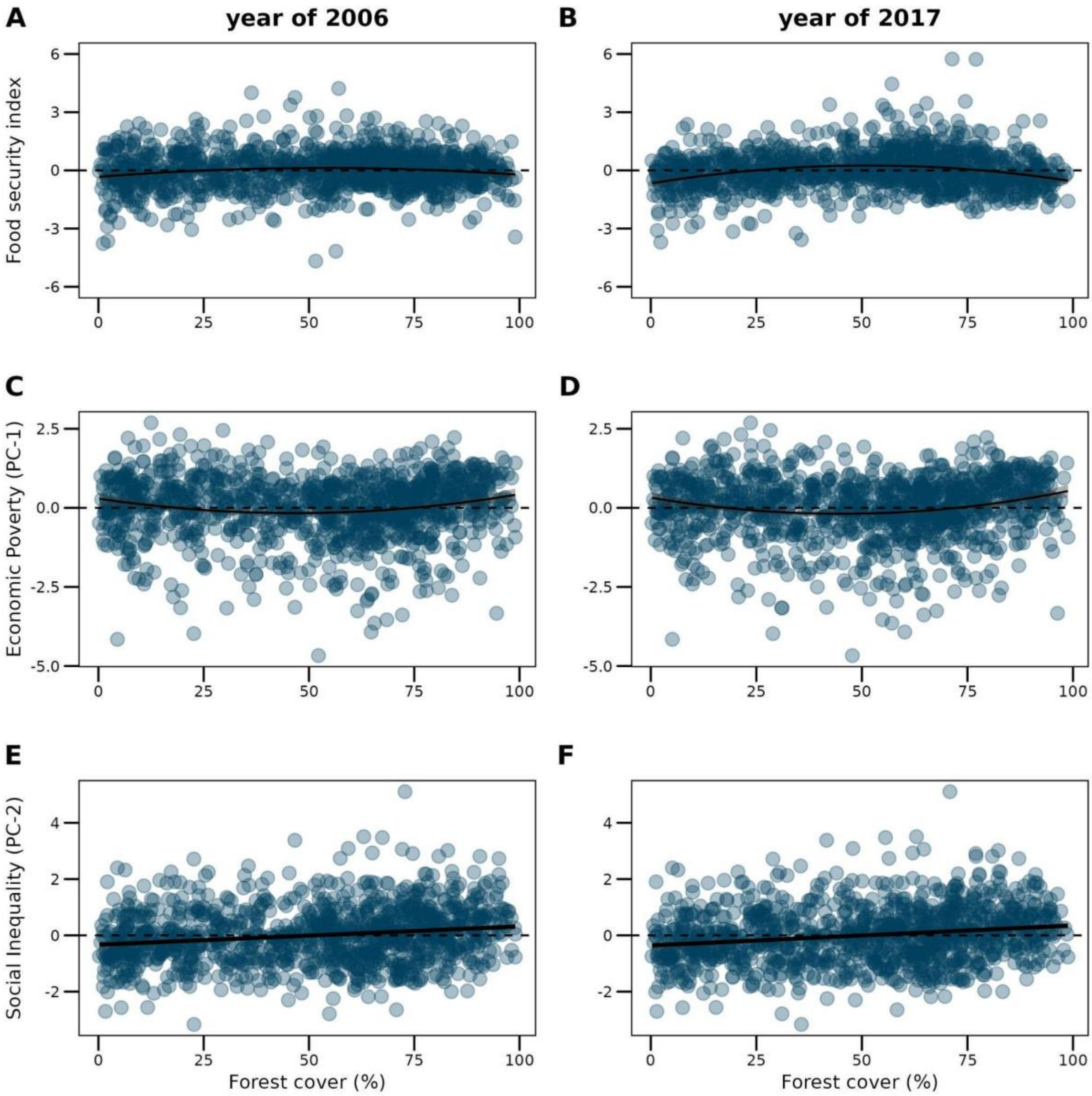


Figure 4. Relationships between the percentage of forest cover per municipality and the values of food security index (A and B) and its main components (C to F) for the years of 2006 (left column) and 2017 (right column).

Discussion

The relationship between our multidimensional index of food security and forest cover is complex but highlights that current models of development based on deforestation do not guarantee, but neither is a cause of reduction in food security. Adopting a multidimensional index of food security brings both challenge and elucidation of hidden relationships between natural vegetation that adds to established measures of food security based on food availability. Although economic poverty and inequality remain constant between years as the main drivers of our proposed index, other dimensions shifted in importance suggesting high dynamism in both environmental and socioeconomic components of the index. All kinds of combinations between forest cover and food security were observed for the 1113 municipalities and both positive (win-win) and negative (lose-lose) synergies, as well as tradeoffs are almost equally likely to take place, however, clumped in space and therefore with strong influence of local context. Ultimately, we found two main types of food insecurity: the first is a sort of “green food insecurity” formed by a group of economically disadvantaged municipalities with high forest cover and low social evensess. The second one is a “grey food insecurity” formed by poor municipalities with low forest cover and more social inequality. In between these extremes, there is a zone of relatively high food security and intermediate forest cover with less poor people and intermediate levels of social inequality.

Multidimensional food security index (MFSI)

Clearly, the access variables were the most important for our MFSI and are mostly related to socioeconomic conditions of the population. Monetary poverty is a historical and widespread problem for the Caatinga population and as well as in other parts of the world, it is among the most important drivers of food insecurity (Fan & Brzeska, 2016; Nahid et al., 2021). On the other hand, the role of socioeconomic inequality is not clear but wealth concentration can also mean generalised low access to food

(Klassen & Murphy, 2020). The interplay of the other dimensions and their variation in the degree of importance between years precludes the search for general patterns and lead to the notion that a great number of factors can shift in importance to determine food security over time (FAO, 2022). One important lesson is the prevalence of access variables within the dimension of the index we created. Stability of food production also had some influence and highlighted the importance of collective association of farmers and the long-term protection of water resources (Parraguez-Vergara et al., 2018). Surprisingly, neither utilisation nor availability variables contributed significantly to our index, reinforcing the rationale that food security is less influenced by food production or capacity to process food (Barrett, 2010). Yet, our results concur that access is far more important than the other dimensions to determine a population's food security (Barrett, 2010; FAO, 2020) even in high-income countries, such as the U.S. (O'Hara & Toussaint, 2021).

The evolution of the index, however, reveals that despite the numerous national policies aimed at combating hunger and poverty, such as the Fome Zero (Zero Hunger) program (Silva et al., 2013), nearly half of the assessed municipalities experienced a decline in food security from 2006 to 2017. This is congruent with recent findings showing that these programs failed to systematically reduce poverty and inequality in the region (Dyngeland et al., 2020), the two most relevant dimensions of food security in this context. Yet, the largest variations in food security were mainly positive, suggesting that some municipalities did benefit from these policies. Enhancing food security for people in the Caatinga and other dry forests is a complex problem and may require additional political solutions addressing other dimensions of food security yet to be considered by governments, such as the agency of people and communities and sustainability of food systems (Clapp et al., 2022).

Food security and forest cover in Caatinga

We found a nuanced scenario with both synergies (win-win and lose-lose) and trade-offs (win-lose and lose-win) between food security and forest cover occurring in nearly similar proportions in the Caatinga. The combination represented by the negative synergy (lose-lose) between forest cover and food security is thought to be the worst scenario where an increasing food insecurity can be worsened by loss of natural resources (Meyfroidt, 2018). Many of these municipalities, but also the lose-gain municipalities, are located in a northeastern-to southwestern axis which is experiencing an expansion of a commodity-driven economy (fruit plantation and soybean) that exerts a pressure of deforestation and concentrates wealth (Weinhold et al., 2013). Double positive scenarios where both forest cover and food security increases may represent development moments when poverty alleviation and politics for reducing social inequality are decoupled from deforestation and must probably rely on services and industry rather than agricultural expansion, combined with forest protection policies (Liu et al., 2017). Municipalities that lost forest and gained food security are probably experiencing the initial phases of commodity-driven development when a rapid increase in socioeconomic indicators derive from the establishment of new agricultural frontiers as already shown for the Amazon deforestation frontier (Rodrigues et al., 2009). The other side of this tradeoff where municipalities present net gains of forest cover but food security decreases may represent the “bust” phase of commodity-driven economy when land abandonment occurs due to the displacement of agricultural frontiers to cheaper lands leaving behind a poor and unequal society (Barbier & Hochard, 2018). Both types of tradeoffs resemble the initial and final phases of “boom-and-bust” development, respectively (Barbier, 2020).

The role of forests for food security

Although our results do not address the role of forest to food security directly, there are some indirect connections that can be made based on the established literature. Forest goods and services are considered an important source of poverty alleviation as they provide vegetables, bushmeat, and work as rangelands for husbandry and provide nutrient cycling needed for shifting agriculture (Baudron et al., 2019). Our index of food security could not grasp such services because this kind of data is hardly available on the scale of our analyses. However, the fact that the highest values of food security were registered in intermediate levels of forest cover suggest that a combination of basic “grey” and enough “green” infrastructure offer more opportunities for poverty alleviation and increase access to natural goods and services that ultimately reduce food insecurity. Our results indicate that improving socioeconomic conditions (i.e. reducing poverty and inequality) can be more effective to increase food security than stimulating food production through agricultural land expansion. Positive associations between forest cover and several nutrition indicators have been reported elsewhere in the literature (Ickowitz et al., 2014; Johnson et al., 2013; Powell et al., 2011) whereas a significant inverted-U relationship between tree cover and fruit and vegetable consumption was registered in a study that compiled data from 21 African countries (Ickowitz et al., 2014). Forests can play an important role as a safety net for low-income people, particularly by providing family income complement (Miller & Hajjar, 2020). In rural Malawi, for example, forests play an important role for poor rural farmers while coping with food shortages during climate shocks, either by providing a direct source of food or through the selling of forest goods (Fisher et al., 2010). In the northern part of Caatinga, forest resources are used by 49% of the rural population to supplement their income, contributing by 10-50% in most families and representing more than 50% of total income for some households (MMA, 2016). Also, the silvopastoral system in the region is known to be a traditional land use system, contributing largely to food security in the region (Pinheiro & Nair, 2018). It is crucial that further studies assess the degree to which local and regional food systems depend on forest goods and services, both within and without the food market.

Future direction

Fifteen percent of the Caatinga municipalities (n = 210) had forest cover lower than 20%. Forest restoration is a legal obligation in rural properties in Caatinga when native vegetation covers less than 20% of the property area, or when the buffer areas of forest around water-resources (i.e., Areas of Permanent Preservation - APP) are degraded (Brasil, 2012). The legal obligation can be seen as an opportunity to promote co-benefits between restoration and food security by the generation of jobs and income related to forest restoration (Mesquita et al., 2010). Forest restoration should directly support food security since it can promote stability for food production through water and soil protection (Soares-Filho et al., 2014) or improve the availability of food resources through agroforestry systems (Chamberlain et al., 2020). Variables related to protection of forests appear as one of the dimensions of our index (PC 10 in 2006 and PC 3 and 4 in 2017 - Table 2 and Supplementary Table 2 and 3) while variables related to the lack of protection contributed negatively to food security (PC 12 in 2006 and PC 9 in 2017). Thus, forest restoration in municipalities can help to increase food security mainly if productive restoration such as agroforests are implemented (Yang et al., 2020). Then, it is possible to restore many deforested areas without competing with food production and also creating opportunities to improve access and stability of food systems. The priority of restoration should be given to low-income farmers and to municipalities with high levels of poverty and deforestation, since these are the places where it should have the greatest positive impact in increasing food security and also as a matter of environmental justice (Cousins, 2021; Reij & Garrity, 2016).

On the other hand, promoting restoration in municipalities with high forest cover could lead to more trade-offs with food security than co-benefits. In this situation, policies aiming to increase food security should attack social inequality (Misselhorn et al., 2012) and take advantage of large tracts of native vegetation to maintain it under legal protection, whereas protected areas can help to improve people's well-being and food security (Naidoo et al., 2019).

Conclusion

We highlighted that native vegetation should be taken into account when thinking about food security and sustainable food systems. Forests play an important role in maintaining the stability and productivity of the local food system (Chamberlain et al., 2020; Melo et al., 2020), but are not usually addressed in food security studies (FAO 2013; Ozturk 2015; Gubert et al. 2017; but see Vysochyna et al. 2020). Our approach shows that there are no intrinsic, unavoidable trade-offs between forest cover increase and food security. In fact, as poverty and inequality were the main source of food insecurity in Caatinga, well-designed ecological restoration programmes can help to alleviate poverty by creating jobs and promoting more income to rural families. Although restoration is an important ally to reduce poverty, this is not a solution for food insecurity in Caatinga and other drylands. There is still an urgent need for social policies that directly aim to reduce poverty in all its dimensions (UNDP & OPHI, 2020) which will greatly improve food security. Those policies should promote ways out of poverty that do not compromise the already over-pressured natural systems (see Chamberlain et al. 2020; Cousins 2021 for examples), they should be focused in the most socially vulnerable and environmentally degraded municipalities, such the ones with low forest cover and low food security or the ones that had negative synergies (lose-lose) between forest and food. Otherwise, those policies might not reduce poverty and food insecurity where it is most needed.

References

Albuquerque UP de, Araújo EL, Castro CC, Alves RRN. 2017. People and Natural Resources in the Caatinga. Caatinga: The Largest Tropical Dry Forest Region in South America. Springer: Cham, Switzerland

Alencar L, Rodrigues, AA, Parry L, Melo FPL. 2022. Forest security as a fourth dimension of the water-energy-food nexus: empirical evidence from the Brazilian Caatinga. The Water-Energy- Food NEXUS: What the Brazilian Research has to Say. University of São Paulo: São Paulo, Brazil

Alexander P, Rounsevell MDA, Dislich C, Dodson JR, Engström K, Moran D. 2015. Drivers for global agricultural land use change: The nexus of diet, population, yield and bioenergy. Global Environmental Change 35: 138–147. DOI: 10.1016/j.gloenvcha.2015.08.011

Andrade EM de, Aquino D do N, Chaves LCG, Lopes FB. 2017. Water as Capital and Its Uses in the Caatinga. Caatinga: The Largest Tropical Dry Forest Region in South America. Springer: Cham, Switzerland

Antongiovanni M, Venticinque EM, Fonseca CR. 2018. Fragmentation patterns of the Caatinga drylands. Landscape Ecology 33: 1353–1367. DOI: 10.1007/s10980-018-0672-6

Antongiovanni M, Venticinque EM, Matsumoto M, Fonseca CR. 2020. Chronic anthropogenic disturbance on Caatinga dry forest fragments. Journal of Applied Ecology 57: 2064–2074. DOI: 10.1111/1365-2664.13686

Arroyo-Rodríguez V, Fahrig L, Tabarelli M, Watling JI, Tischendorf L, Benchimol M, Cazetta E, Faria D, Leal IR, Melo FPL, Morante-Filho JC, Santos BA, Arasa-Gisbert R, Arce-Peña N, Cervantes-López MJ, Cudney-Valenzuela S, Galán-Acedo C, San-José M, Vieira ICG, Slik JWF, Nowakowski AJ, Tscharntke T. 2020. Designing optimal human-modified landscapes for forest biodiversity conservation. Ecology Letters 23: 1404–1420. DOI: 10.1111/ele.13535

Bahar NHA, Lo M, Sanjaya M, Van Vianen J, Alexander P, Ickowitz A, Sunderland T. 2020. Meeting the food security challenge for nine billion people in 2050: What impact on forests? Global Environmental Change 62: 102056. DOI: 10.1016/j.gloenvcha.2020.102056

Barbier EB. 2020. Long run agricultural land expansion, booms and busts. Land Use Policy 93: 103808. DOI: 10.1016/j.landusepol.2019.01.011

Barbier EB, Hochard JP. 2018. Land degradation and poverty. Nature Sustainability 1: 623–631. DOI: 10.1038/s41893-018-0155-4

Barrett CB. 2010. Measuring Food Insecurity. Science 327: 5

Baudron F, Tomscha SA, Powell B, Groot JCJ, Gergel SE, Sunderland T. 2019. Testing the Various Pathways Linking Forest Cover to Dietary Diversity in Tropical Landscapes. Frontiers in Sustainable Food Systems 3. DOI: 10.3389/fsufs.2019.00097

Bijman J, Wijers G. 2019. Exploring the inclusiveness of producer cooperatives. Current Opinion in Environmental Sustainability 41: 74–79. DOI: 10.1016/j.cosust.2019.11.005

Brasil. 2012. Lei de Proteção da Vegetação Nativa.

Burchi F, De Muro P. 2016. From food availability to nutritional capabilities: Advancing food security analysis. Food Policy 60: 10–19. DOI: 10.1016/j.foodpol.2015.03.008

Chamberlain JL, Darr D, Meinhold K. 2020. Rediscovering the Contributions of Forests and Trees to Transition Global Food Systems. Forests 11: 1098. DOI: 10.3390/f11101098

Clapp J, Moseley WG, Burlingame B, Termine P. 2022. Viewpoint: The case for a six-dimensional food security framework. Food Policy 106: 102164. DOI: 10.1016/j.foodpol.2021.102164

Cousins JJ. 2021. Justice in nature-based solutions: Research and pathways. Ecological Economics 180: 106874. DOI: 10.1016/j.ecolecon.2020.106874

Cutter SL, Boruff BJ, Shirley WL. 2003. Social Vulnerability to Environmental Hazards*. Social Science Quarterly 84: 242–261. DOI: 10.1111/1540-6237.8402002

de Fraiture C, Wichelns D. 2010. Satisfying future water demands for agriculture. Agricultural Water Management 97: 502–511. DOI: 10.1016/j.agwat.2009.08.008

Devendra C. 1999. Goats: Challenges for Increased Productivity and Improved Livelihoods. Outlook on Agriculture 28: 215–226. DOI: 10.1177/003072709902800404

Devereux S. 2016. Social protection for enhanced food security in sub-Saharan Africa. Food Policy 60: 52–62. DOI: 10.1016/j.foodpol.2015.03.009

Dyngeland C, Oldekop JA, Evans KL. 2020. Assessing multidimensional sustainability: Lessons from Brazil's social protection programs. Proceedings of the National Academy of Sciences 201920998. DOI: 10.1073/pnas.1920998117

Ericksen PJ. 2008. Conceptualizing food systems for global environmental change research. Global Environmental Change 18: 234–245. DOI: 10.1016/j.gloenvcha.2007.09.002

Fan S, Brzeska J. 2016. Sustainable food security and nutrition: Demystifying conventional beliefs. Global Food Security 11: 11–16. DOI: 10.1016/j.gfs.2016.03.005

FAO (ed). 2013. The multiple dimensions of food security. FAO: Rome

FAO. 2022. The State of Food Security and Nutrition in the World 2022. FAO. DOI: 10.4060/cc0639en

FAO I. 2020. The State of Food Security and Nutrition in the World 2020: Transforming food systems for affordable healthy diets. FAO, IFAD, UNICEF, WFP and WHO: Rome, Italy. DOI: 10.4060/ca9692enAlso Available in:Chinese Spanish Arabic French Russian

Fisher M, Chaudhury M, McCusker B. 2010. Do Forests Help Rural Households Adapt to Climate Variability? Evidence from Southern Malawi. World Development 38: 1241–1250. DOI: 10.1016/j.worlddev.2010.03.005

Foley JA, Ramankutty N, Brauman KA, Cassidy ES, Gerber JS, Johnston M, Mueller ND, O'Connell C, Ray DK, West PC, Balzer C, Bennett EM, Carpenter SR, Hill J, Monfreda C, Polasky S, Rockström J, Sheehan J, Siebert S, Tilman D, Zaks DPM.

2011. Solutions for a cultivated planet. Nature 478: 337–342. DOI: 10.1038/nature10452

Godfray HC, Beddington J, Crute I, Haddad L, Lawrence D, Muir J, Pretty J, Robinson S, Thomas S, Toulmin C. 2010. Food Security: The Challenge of Feeding 9 Billion People. Science 327

Gubert MB, dos Santos SMC, Santos LMP, Pérez-Escamilla R. 2017. A Municipal-level analysis of secular trends in severe food insecurity in Brazil between 2004 and 2013. Global Food Security 14: 61–67. DOI: 10.1016/j.gfs.2017.03.004

Hummell BM, Cutter SL, Emrich CT. 2016. Social Vulnerability to Natural Hazards in Brazil. International Journal of Disaster Risk Science 7: 111–122. DOI: 10.1007/s13753-016-0090-9

IBGE (ed). 2017. Classificação e caracterização dos espaços rurais e urbanos do Brasil: uma primeira aproximação. IBGE, Instituto Brasileiro de Geografia e Estatística: Rio de Janeiro

Ickowitz A, Powell B, Salim MA, Sunderland TCH. 2014. Dietary quality and tree cover in Africa. Global Environmental Change 24: 287–294. DOI: 10.1016/j.gloenvcha.2013.12.001

IPBES. 2019. The global assessment report on biodiversity and ecosystems services. IPBES secretariat: Bonn, Germany

Jagger P, Cheek JZ, Miller D, Ryan C, Shyamsundar P, Sills E. 2022. The Role of Forests and Trees in Poverty Dynamics. Forest Policy and Economics 140: 102750. DOI: 10.1016/j.forpol.2022.102750

Johnson H, Shaw L. 2014. Rethinking rural co-operatives in development introduction to Policy Arena. Journal of International Development 26: 668–682. DOI: 10.1002/jid.3004

Johnson KB, Jacob A, Brown ME. 2013. Forest cover associated with improved child health and nutrition: evidence from the Malawi Demographic and Health Survey and satellite data. Global Health: Science and Practice 1: 237–248. DOI: 10.9745/GHSP-D-13-00055

Kah HK. 2017. 'Boko Haram is Losing, But so is Food Production': Conflict and Food Insecurity in Nigeria and Cameroon. Africa Development 20

Khan S, Hanjra MA, Mu J. 2009. Water management and crop production for food security in China: A review. Agricultural Water Management 96: 349–360. DOI: 10.1016/j.agwat.2008.09.022

Klassen S, Murphy S. 2020. Equity as both a means and an end: Lessons for resilient food systems from COVID-19. World Development 136: 105104. DOI: 10.1016/j.worlddev.2020.105104

Lambin EF, Gibbs HK, Heilmayr R, Carlson KM, Fleck LC, Garrett RD, le Polain de Waroux Y, McDermott CL, McLaughlin D, Newton P, Nolte C, Pacheco P, Rausch LL, Streck C, Thorlakson T, Walker NF. 2018. The role of supply-chain initiatives in reducing deforestation. Nature Climate Change 8: 109–116. DOI: 10.1038/s41558-017-0061-1

Latawiec AE, Strassburg BB, Brancalion PH, Rodrigues RR, Gardner T. 2015. Creating space for large-scale restoration in tropical agricultural landscapes. Frontiers in Ecology and the Environment

Liu J, Liang M, Li L, Long H, De Jong W. 2017. Comparative study of the forest transition pathways of nine Asia-Pacific countries. Forest Policy and Economics 76: 25–34. DOI: 10.1016/j.forpol.2016.03.007

Melo FPL. 2017. The Socio-Ecology of the Caatinga: Understanding How Natural Resource Use Shapes an Ecosystem. Caatinga: The Largest Tropical Dry Forest Region in South America. Springer: Cham, Switzerland

Melo FPL, Parry L, Brancalion PHS, Pinto SRR, Freitas J, Manhães AP, Meli P, Ganade G, Chazdon RL. 2020. Adding forests to the water–energy–food nexus. Nature Sustainability. DOI: 10.1038/s41893-020-00608-z

Melo FPL, Parry L, Brancalion PHS, Pinto SRR, Freitas J, Manhães AP, Meli P, Ganade G, Chazdon RL. 2021. Adding forests to the water–energy–food nexus. Nature Sustainability 4: 85–92. DOI: 10.1038/s41893-020-00608-z

Mesquita CAB, Holvorcem CGD, Lyrio CH, de Menezes PD, da Silva Dias JD, Azevedo JF. 2010. COOPLANTAR: A Brazilian Initiative to Integrate Forest Restoration with Job and Income Generation in Rural Areas. Ecological Restoration 28: 199–207. DOI: 10.3368/er.28.2.199

Meyfroidt P. 2018. Trade-offs between environment and livelihoods: Bridging the global land use and food security discussions. Global Food Security 16: 9–16. DOI: 10.1016/j.gfs.2017.08.001

Miller DC, Hajjar R. 2020. Forests as pathways to prosperity: Empirical insights and conceptual advances. World Development 13

Miller DC, Ordoñez PJ, Brown SE, Forrest S, Nava NJ, Hughes K, Baylis K. 2020. The impacts of agroforestry on agricultural productivity, ecosystem services, and human well-being in low-and middle-income countries: An evidence and gap map. Campbell Systematic Reviews 16. DOI: 10.1002/cl2.1066

Misselhorn A, Aggarwal P, Ericksen P, Gregory P, Horn-Phathanothai L, Ingram J, Wiebe K. 2012. A vision for attaining food security. Current Opinion in Environmental Sustainability 4: 7–17. DOI: 10.1016/j.cosust.2012.01.008

Nahid N, Lashgarara F, Farajolah Hosseini SJ, Mirdamadi SM, Rezaei-Moghaddam K. 2021. Determining the Resilience of Rural Households to Food Insecurity during Drought Conditions in Fars Province, Iran. Sustainability 13: 8384. DOI: 10.3390/su13158384

Naidoo R, Gerkey D, Hole D, Pfaff A, Ellis AM, Golden CD, Herrera D, Johnson K, Mulligan M, Ricketts TH, Fisher B. 2019. Evaluating the impacts of protected areas on human well-being across the developing world. Science Advances 5: eaav3006. DOI: 10.1126/sciadv.aav3006

O'Hara S, Toussaint EC. 2021. Food access in crisis: Food security and COVID-19. Ecological Economics 180: 106859. DOI: 10.1016/j.ecolecon.2020.106859

Ozturk I. 2015. Sustainability in the food-energy-water nexus: Evidence from BRICS (Brazil, the Russian Federation, India, China, and South Africa) countries. Energy 93: 999–1010. DOI: 10.1016/j.energy.2015.09.104

Parraguez-Vergara E, Contreras B, Clavijo N, Villegas V, Paucar N, Ther F. 2018. Does indigenous and campesino traditional agriculture have anything to contribute to food sovereignty in Latin America? Evidence from Chile, Peru, Ecuador, Colombia, Guatemala and Mexico. 17

Powell B, Hall J, Johns T. 2011. Forest cover, use and dietary intake in the East Usambara Mountains, Tanzania. International Forestry Review 13: 305–317. DOI: 10.1505/146554811798293944

Pretty J, Bharucha ZP. 2014. Sustainable intensification in agricultural systems. Annals of Botany 114: 1571–1596. DOI: 10.1093/aob/mcu205

Queiroz LP, Cardoso D, Fernandes MF, Moro MF. 2017. Diversity and Evolution of Flowering Plants of the Caatinga Domain. Caatinga: The Largest Tropical Dry Forest Region in South America. Springer: Cham, Switzerland

Rasmussen LV, Fagan ME, Ickowitz A, Wood SLR, Kennedy G, Powell B, Baudron F, Gergel S, Jung S, Smithwick EAH, Sunderland T, Wood S, Rhemtulla JM. 2020. Forest pattern, not just amount, influences dietary quality in five African countries. Global Food Security 25: 100331. DOI: 10.1016/j.gfs.2019.100331

Reij C, Garrity D. 2016. Scaling up farmer-managed natural regeneration in Africa to restore degraded landscapes. Biotropica 48: 834–843. DOI: 10.1111/btp.12390

Rodrigues ASL, Ewers RM, Parry L, Souza C, Verissimo A, Balmford A. 2009. Boom-and-Bust Development Patterns Across the Amazon Deforestation Frontier. Science 324: 1435–1437. DOI: 10.1126/science.1174002

Rosenstock TS, Dawson IK, Aynekulu E, Chomba S, Degrande A, Fornace K, Jamnadass R, Kimaro A, Kindt R, Lamanna C, Malesu M, Mausch K, McMullin S, Murage P, Namoi N, Njenga M, Nyoka I, Paez Valencia AM, Sola P, Shepherd K, Steward P. 2019. A Planetary Health Perspective on Agroforestry in Sub-Saharan Africa. One Earth 1: 330–344. DOI: 10.1016/j.oneear.2019.10.017

Sampaio EVSB, Menezes RSC, Sampaio, Y de SB, Freitas ADS de. 2017. Sustainable Agricultural Uses in the Caatinga. Caatinga: the largest tropical dry forest region in South America. Springer,: Cham, Switzerland

Schütz L, Gattinger A, Meier M, Müller A, Boller T, Mäder P, Mathimaran N. 2018. Improving Crop Yield and Nutrient Use Efficiency via Biofertilization—A Global Meta-analysis. Frontiers in Plant Science 8

Sena A, Freitas C, Feitosa Souza P, Alpino T, Pedroso M, Corvalan C, Barcellos C, Carneiro F. 2018. Drought in the Semiarid Region of Brazil: Exposure, Vulnerabilities and Health Impacts from the Perspectives of Local Actors. PLoS Currents. DOI: 10.1371/currents.dis.c226851ebd64290e619a4d1ed79c8639

Silva JG da, Del Grossi ME, França CG de, Vasconcelos LM. 2013. The Fome Zero (Zero Hunger) Program: the Brazilian experience.

Silva JMC da, Barbosa LC. 2017. Impact of Human Activities on the Caatinga. Caatinga: the largest tropical dry forest region in South Americad. Springer: Cham, Switzerland

Silva JMC da, Barbosa LC, Leal IR, Tabarelli M. 2017. The Caatinga: Understanding the Challenges. Caatinga: The Largest Tropical Dry Forest Region in South America. Springer: Cham, Switzerland

Soares-Filho B, Rajao R, Macedo M, Carneiro A, Costa W, Coe M, Rodrigues H, Alencar A. 2014. Cracking Brazil's Forest Code. Science 344: 363–364. DOI: 10.1126/science.1246663

Stattman SL, Mol APJ. 2014. Social sustainability of Brazilian biodiesel: The role of agricultural cooperatives. Geoforum 54: 282–294. DOI: 10.1016/j.geoforum.2014.04.001

Swift TL, Hannon SJ. 2010. Critical thresholds associated with habitat loss: a review of the concepts, evidence, and applications. Biological Reviews 85: 35–53. DOI: 10.1111/j.1469-185X.2009.00093.x

Tabarelli M, Leal IR, Scarano FR, Silva JMC da. 2017. The Future of the Caatinga. Caatinga: The Largest Tropical Dry Forest Region in South America. Springer: Cham, Switzerland

Tendall DM, Joerin J, Kopainsky B, Edwards P, Shreck A, Le QB, Kruetli P, Grant M, Six J. 2015. Food system resilience: Defining the concept. Global Food Security 6: 17–23. DOI: 10.1016/j.gfs.2015.08.001

UNDP, OPHI. 2020. Global Multidimensional Poverty Index 2020.

UNICEF. 2019. The state of the world's children 2019: Children, food and nutrition. Growing well in a changing world. United Nations Children's Fund (UNICEF): New York, NY 10017, USA, 24

Vysochyna A, Stoyanets N, Mentel G, Olejarz T. 2020. Environmental Determinants of a Country's Food Security in Short-Term and Long-Term Perspectives. Sustainability 12: 4090. DOI: 10.3390/su12104090

Weinhold D, Killick E, Reis EJ. 2013. Soybeans, Poverty and Inequality in the Brazilian Amazon. World Development 52: 132–143. DOI: 10.1016/j.worlddev.2012.11.016

Yang KF, Gergel SE, Baudron F. 2020. Forest restoration scenarios produce synergies for agricultural production in southern Ethiopia. Agriculture, Ecosystems & Environment 295: 106888. DOI: 10.1016/j.agee.2020.106888

4 ARTIGO 3 - Population-environment transitions in the Brazilian Caatinga dry forest

Lucas Alencar[a]*, Luke Parry[b], Felipe P. L. Melo[a]

[a]Department of Botany, Federal University of Pernambuco, Recife, Brazil 50670-901 and [b]Lancaster Environment Centre, Lancaster University, Lancaster, United Kingdom LA1 4YW

Lucas Alencar, Av. Prof. Moraes Rego, 1235 - Cidade Universitária, Recife - Pernambuco, Brazil 50670-901. Phone: +55 81 988641253

Email: alencar.lucasc@gmail.com

Author's contribution: L.A., L.P., and F.P.L.M. designed research; L.A performed research and analysis; L.A. wrote first draft; L.A., L.P. and F.P.L.M. reviewed and edited the manuscript

Classification: Social Sciences - Environmental sciences; Biological Sciences - Environmental Sciences

Keywords: Sustainability, rural development, Spatial Durbin Error Model

Abstract

Understanding population-environment transitions in the Global South is vital for designing policies that promote poverty alleviation and forest conservation. We evaluate population-environment dynamics and whether this is associated with changes in human development conditions in the Brazilian Caatinga dry forest, the world's most populous dry forest. First, we calculate the number of forest-proximate people inhabiting this biome based on varying minimum thresholds of landscape forest cover. We estimate 9.4 million people living in the Caatinga in landscapes with at least 20% of forest cover. In those landscapes, the forest-proximate population increased by more than 670,00 over the study period, although 35% of landscapes experienced depopulation. Forest growth happened in 55% of the landscapes, with a total positive net cover change of almost 460,000 hectares. We used cluster analysis and spatial modelling to examine the relationship between population and forest transitions at municipality-scales. Analysis revealed strong spatial autocorrelation in the Caatinga's population-environment transitions and no consistent association with development indicators. However, growth in forest-proximate populations was positively correlated with family farming, urban expansion and reduced income inequality. We found that population and environmental transitions in the Caatinga are decoupled and appear not to share municipal-scale socioeconomic drivers and may instead reflect larger-scale processes of change, such as drought exposure. This means that forest change in other dry forests may also be driven by multi-scale processes and not directly by population (de)growth, highlighting the need for a nuanced understanding to better inform policy design for sustainable development.

Significance

Many of the world's one billion forest-proximate people live in poverty. Thus supporting the sustainable development of these communities is vital for eradicating poverty and conserving tropical forests. The connections between forest and population changes in rural landscapes are still unclear and understanding it can have profound implications for the planning of sustainable transitions. We found forest-population transitions to be diverse but geographically clustered, revealing that different kinds of agricultural frontiers can co-exist in the same region. Also, local-scale development conditions in those frontiers are various and more related to population than to environmental changes.

Introduction

The relationship between rural population change and forest cover change is unclear and little is known about the human development condition under which these transitions are happening. Environmental degradation was recently pointed out as a direct consequence of human population growth, however countries with higher GDP per capita tend to have greater environmental footprint, irrespective of population size (1). Whether rural or urban population size has the strongest influence on environmental change is a matter of debate (2). Hence, understanding how rural population growth or decline is connected with forest cover change is important for meeting sustainable development goals. Here, we estimate the number of rural people living in forested landscapes in the Brazilian Caatinga dry forest, how it changes over time and how those numbers vary according to different definitions of a forested landscape. Finally, we measure the joint changes in forest cover and population size in landscapes and explore the human development context in which those changes are happening.

Rural population growth in agricultural frontiers may lead to deforestation if this expands the labour force and food demand, increasing incentives to expand croplands (3). For example, in southwestern parts of Brazilian Amazon, higher rural population density was associated with greater deforestation (4). However, tropical deforestation in 41 countries was mainly driven by urban population growth (2). In other contexts, rural population growth was associated with forest regrowth. In Ghana and Guinea, farming practices (e.g., fire management, agroforestry schemes) and even new settlements (e.g., through actively planting useful tree species) were related to forest cover increase (5), challenging the idea that population growth necessarily causes deforestation. The consequences of rural depopulation on forest cover will depend on what kind of people are occupying the land and how the land is exploited. Remittances from international migration, for example, promote forest regeneration in Nepal, combined with rural depopulation and a declining farm sector (6). Nonetheless, land-grabbing and dispossession of small-holders by large landowners are major drivers of tropical deforestation (7) and can cause rural depopulation as agricultural frontiers advance (8). Broad typologies of population-environment transitions reveal how variable and unpredictable these trajectories can be.

Understanding the development conditions under which population-forest transitions occur may elucidate how environmental change is connected to population change. There are four broad kinds of transitions at the landscape-scale, based on net changes in forest cover and the number of human inhabitants over a given time period. (A) Deforestation and rural population growth. These changes characterise so-called 'populist' farm-forest frontiers based on expansion of small-scale agriculture in land-abundant contexts. This kind of transition may involve in-migration associated with agricultural settlement schemes or spontaneous colonisation (8), and/or internal population growth if there are higher births than deaths, population momentum (reflecting adults entering reproductive age), and limited out-migration. Barbier (9) argues that this initial colonisation process can generate an economic 'boom', reducing poverty rates, however, as the frontier evolves, inequality may remain an untracked issue. In this case, we would expect reducing poverty rates and stable or increasing inequality levels, increase in family farming and stable-yet-low health metrics. B) Reforestation and rural population growth. We expect this to occur in places where people are reducing their dependence on agricultural livelihoods whilst receiving welfare support, and hence avoiding a rural exodus. Instead of relying only, or mainly, on farming to sustain themselves, people may benefit from greater off-farm opportunities, government assistance or remittances from relatives. Dyngeland et al (10) found that increased penetration of cash-transfer programs was correlated with slightly increased forest cover in the Caatinga, however they did not investigate populational changes. We expect declining poverty and inequality, an improvement in health indicators and non-decreasing spatial coverage of family farming. (C) Deforestation and rural depopulation. We expect this transition to occur at so-called commodity or 'capitalist' frontiers, where large land-owners dispossess small-holders in order to invest in commodity production for export (e.g., farming soy-beans) to other regions or nations. Although capital-intensive, agricultural mechanisation reduces labour requirements and hence we encounter rural depopulation and forest loss, either through direct clearance for cropland, or the 'leakage' impact of extensive cattle-raising displaced by increasing land rents (11). We expect decreased presence of family farming, decreased poverty but higher income inequality, and improvement in health conditions. Del Giorgio et al (12), studying commodity frontiers in dry Chaco of Argentina, found increased vulnerability of small-holders together with displacement of many rural families due to the expansion of large-scale soybean plantation. (D) Reforestation and rural depopulation. These simultaneous changes are central to the predictions of Forest Transition Theory (13). In Puerto Rico, for example, the rapid industrialization and changes in political economy supported a forest regeneration growth from 9% to 37% of total island area, together with widespread rural out-migration and increase in remittances to rural families (14). In these situations, we expect decreasing poverty and inequality, declining family farming and improving population health metrics. Finally, It is also possible that a no-change situation happened (i.e., no changes in forest cover area or in number of people living nearby forests), however, tropical rural areas, such as the Brazilian Caatinga dry forest, are at the forefront of global changes, and thus we do not expected no changes in forest cover or population size (or in forest livelihoods) in the time frame of our analysis (2006 to 2017).

Dryland forests across the globe and the rural people living within, tend to be relatively neglected by researchers and policy makers governments compared to tropical rainforests (15). Research in dry forests is underrepresented in several areas and has a distinct focus on disturbance and climate change and less on human-environment interactions (16). Governments often pursue rural development paradigms from rainforest regions to dry lands without much success, locking rural people in a spiral of social and environmental degradation instead of pursuing a new development strategy that considers the specificities of dry land regions (17). Furthermore, the long-standing misbelief that arid regions have low levels of biodiversity resulted in lower levels of biodiversity protection, posing further threats to its forests (18). This resulted in widespread land degradation and low socioeconomic development in most arid lands, globally (17). In northeastern Brazil, the Caatinga dry forest was home to an estimated 28 million people by 2010 (19) and 73% of the municipalities in the region are mainly rural (20). The Caatinga is a highly diverse, but also highly threatened ecosystem due to acute and chronic anthropogenic disturbances (21, 22). Human development metrics in the Caatinga are the lowest in Brazil, with high prevalence of extreme income poverty in rural areas (19). Most small-scale agricultural activities rely on slash-and-burn practice and use native vegetation as forage for cattle (mainly goats) (23). Large-scale farming involves the practice of clear cutting the vegetation and planting exotic grasses adapted to droughts for bovine cattle raising, which is currently the major land-use replacing forest (23, 24).

To better examine the rural population and forest cover connections, we used the term forest-proximate people. Forest-proximate people (FPP) refers to people living within or nearby forests, excluding urban populations, and is gaining traction as a policy-relevant term (25). FPP includes a socially, economically and ethnically diverse group that includes large-scale farmers to small-holders, non-timber forest product harvesters and indigenous communities and other kinds of rural populations. Living close to a forest does not mean direct dependency on forest resources (26), however, many or most of them rely to some degree on forest to maintain their livelihood (e.g. wood extraction, non-timber forest products, food). Forest-proximate populations are defined only by their spatial proximity to forest patches. Consequently, FPP estimates are sensitive to the spatial proximity threshold and to the definition of forests (e.g., canopy cover) or minimal patch size (26). Those rural populations living close to forests are, generally, important agents of local land cover change, either by causing deforestation or allowing forest regeneration to happen (8).

In order to understand the direction and heterogeneity of population-environment transitions in the Caatinga dry forest we asked: 1) what is the size of forest-proximate population of the Caatinga and how is this changing? 2) How sensitive are estimates of the Caatinga's forest-proximate population to the definition of forested landscapes? 3) Is there a consistent relationship between forest cover transitions and population change? And finally 4) how changes in development indicators are related to changes in forest cover and rural population

Methods

Study region

The Caatinga biome retains 63% of its native vegetation and a long-term annualised deforestation rate (1987 through 2020) of 0.37% (SD 0.26%) (24). Nonetheless, the Caatinga dry forest is highly disturbed and fragmented (27, 28). Here, forest resources usage is an integral part of rural livelihoods; and forest-cover is affected by widespread fuelwood collection and timber-harvesting for construction, fencing, etc. (29). Agriculture crops are mainly rain-fed and often rely on slash-and-burn practices (23) large-scale capitalised landowners (or smaller-scale farmers with access to credit) utilise irrigation systems, restricted to areas nearby perennial rivers or hydroelectric dams (30). Many rural households also raise livestock (goats, cattle, sheep) extensively using native vegetation as fodder (23). Accordingly, production of crops and livestock is generally dependent on climatic conditions and native vegetation, which in turn may have an influence in deforestation and/or forest regeneration rates.

Forest cover and population data

We used MapBiomas land cover data, a well-established Brazilian non-governmental remote sensing product that classifies land cover and land cover change yearly, using Landsat imagery, artificial intelligence and field validation for all Brazilian biomes (31). Human population data were obtained from WorldPop (32). We used the unconstrained 1 km² people per pixel data adjusted for United Nations estimates (32). Given we follow Newton et al's (26) approach of excluding urban populations from our definition of forest-proximate people, we used the urban/rural classification of sectors from the Brazilian Institute of Geography and Statistics (IBGE) 2010 census to exclude urban sectors (33). To evaluate temporal changes, we used data from 2006 and 2017 for both people and forests to combine with data from IBGE rural census. We used Bastin et al's (34) sampling points as centroids to build 6193 circular sampling landscapes (from 5km radial buffers) to measure forest cover and count the forest-proximate population. We used sampling points from (34) because they were already used to identify forests in the Caatinga. We later classified landscapes as forested or not, based on minimal native vegetation cover thresholds.

Spatiotemporal variation in forest-proximate people

We measured forest cover in sampled landscapes for 2006 and 2017, and calculated the forest cover change using the 'landscapemetrics' package (35) in R (36). We calculated the number of forest-proximate people inhabiting each sampling of landscapes and re-counted the aggregate population of forest-proximate people using varying minimum forest cover thresholds to include (as forested) or exclude (as non-forested) particular landscapes. We used minimal thresholds of =>10% forest cover and 10% intervals until 100%. Note we did not assess population dynamics in the demographic sense of separating the effects of net migration or from that of birth versus deaths. In Brazil demographic data on age-specific rates of birth and death are only available at the larger, state scale (federal unit).

To visualise population transitions at municipality-scales we selected landscapes (n = 5748) with =>20% forest cover and summed up the number of forest-proximate people for each municipality. To identify the location of potential spatial clusters of changing forest-proximate populations we used the Local Indicator of Spatial Association (LISA), in the R package ‘rgeoda’ (37). LISA identifies locations where a group of neighbouring observations are more similar (or dissimilar) to each other than expected by chance, allowing identification of clusters of change and outliers (38). Each municipality is classified as having Low or High change in its forest-proximate population, relative to the surrounding, local municipalities. Hence, municipalities with positive, yet modest growth in forest-proximate populations may be classified as “low”, depending on the value of change in neighbouring municipalities. There are four possible combinations; “High-High'' indicates places with growth in the forest-proximate population clustered by neighbours with growth. “High-Low” are those with growth surrounded by neighbours with small changes, “Low-Low” are groups of municipalities only with declining population and “Low-High” are municipalities with depopulation surrounded by municipalities with population growth. Finally, municipalities were classified as “Not significant” if not part of any cluster.

Population-environment transitions in forested landscapes

We classified landscapes into four types of population-environment transition: deforestation and rural population growth, reforestation and rural population growth, deforestation and rural depopulation and reforestation and rural depopulation. Afterwards, we aggregated landscapes at the municipality-level of the four types of population-environment transition using mean change in forest cover and sum of FPP number between 2006 and 2017. A linear model to test for a correlation between forest cover change and forest-proximate population change at municipality scale.

We tested for potential residual spatial autocorrelation in this linear model using Moran's I test, and found a significant positive result (Supplementary Table 6). To check the extent to which this spatial autocorrelation affected our linear model, we ran a Spatial Error Model (39), which adds a spatial matrix and a disturbance term to the model's error, accounting for spatial autocorrelations. The Spatial Error Model has the following formula (equation 1):

$$Y = \beta X + u, u = \lambda W u + \varepsilon \quad (1)$$

Where *Y* is the outcome of interest (forest cover change in our case), β is the parameter to be estimate, *X* is the independent variable (forest-proximate people change), u is the vector of regression disturbance, λ is the autoregressive parameter, *W* is the spatial weight matrix and Ɛ is the error term. To build this and the other spatial models we used the 'spdep' package in R environment (40).

Development indicators and diverse population-environment transitions

We choose six development indicators that could reveal if the context under each population-environment transition were happening. Those indicators can also reveal if these changes may provoke welfare concerns. In order to characterise the development context of each type of population-environment transitions we performed two clustering analyses combining the six municipal indicators. The indicators were: percentage of agricultural land occupied by family agriculture (hereafter, family agriculture), urban population size; longevity sub-index of Human Development Index (HDI - Longevity), extreme poverty rates; Gini income inequality index; and number of under five years old children death per thousand children born (hereafter, under five mortality). The family agriculture data was gathered from Brazil's periodic governmental agricultural census (41). Using data from 2006 and 2017, we calculated municipality-scale change in family agriculture. The other development indicators were gathered from 2000 and 2010 national census data (33). Supplementary Table 1 describes the variables in more depth, Supplementary Table 2 shows some descriptive statistics of each indicator for the entire Caatinga, and Supplementary Figure 1 by type of population-environment transition.

To assess if the development indicators were correlated to temporal changes in forest cover and FPP, we employed linear models. We identified several significant correlations, however, Moran I's test revealed spatial autocorrelation in model residuals (Supplementary Table 3) and consequently, we built two additional spatial models to attempt to minimise the effects of spatial autocorrelation on any observed relationships between development metrics and changes in forest cover and forest-proximate populations. We chose the Spatial Durbin Error Model (39) which adds a spatial matrix weight to the independent variables and to the error term, adding the effects of measured (independent) and unmeasured (error) development indicators of neighbouring municipalities on any changes in forest cover and forest-proximate population size within the focal municipality, controlling for spatial autocorrelation. Potentially, changes in development indicators within neighbouring municipalities may affect the forest cover and forest-proximate population in another municipality. For example, better job opportunities in a nearby town could decrease the extreme poverty in a given municipality and may attract people from nearby rural areas or even from very distant places (42). The Spatial Durbin Error Model (hereafter spatial model) has the following formula (Equation 2).

$$Y = \beta_n X + W\theta_n X + u, u = \lambda W u + \varepsilon \quad (2)$$

Where *Y* is the outcome of interest (either forest cover change or forest-proximate population change), β_n and θ_n are the parameters to be estimate of each development indicator, *W* is the spatial weight matrix, *u* is the vector of regression disturbance and λ is the autoregressive parameter.

Results

What is the size of the forest-proximate population of the Caatinga and how is this changing?

Our results show that 9.2 million people were living in forested landscapes (=>20% of forest cover) in the Caatinga dry forest biome in 2017, based on extrapolating from sampled landscapes to all rural areas. The majority of the landscapes experienced a growth in forest-proximate population with some landscapes increasing over 100% (Supplementary Figure 2). The three largest forest-proximate populations were in the states of Ceará (1.36 million), Bahia (1.22 million), and Pernambuco (0.65 million), using =>20% forest cover definition.

From 2006 to 2017, the forest-proximate population grew by 702,274. Considering only the forested (=> 20% cover) subset of sampled landscapes, we counted 4.7 million forest-proximate inhabitants in 2017, an increase of 392,774 people from 2006. Most municipalities (73%) experienced growing forest-proximate populations over this period, but higher growth was concentrated in central and northern areas (Figure 2a). Municipalities with declining forest-proximate populations were concentrated in the south, and others spread across the biome. Analysis of local spatial association (LISA) confirmed two major groupings; one northern clustering of municipalities with rapidly-growing forest-proximate populations, and one southern cluster of rapidly declining forest-proximate populations (Figure 2b). This analysis also revealed smaller clusters of population change spread across the Caatinga. Detailed information on the number of FPP per state is at Supplementary Figure 3 and Supplementary Table 5.

How sensible are the estimates of forest-proximate population size?

Results showed that using conservative definitions of forested landscapes led to large decreases in estimates of the Caatinga's forest-proximate population; considering only the landscapes with =>20% forest cover had approximately 4.7 million people (Figure 1a and Supplementary Table 4), landscapes with >50% forest cover had population of 2.8 million and landscapes with =>70% cover, 1.6 million people.

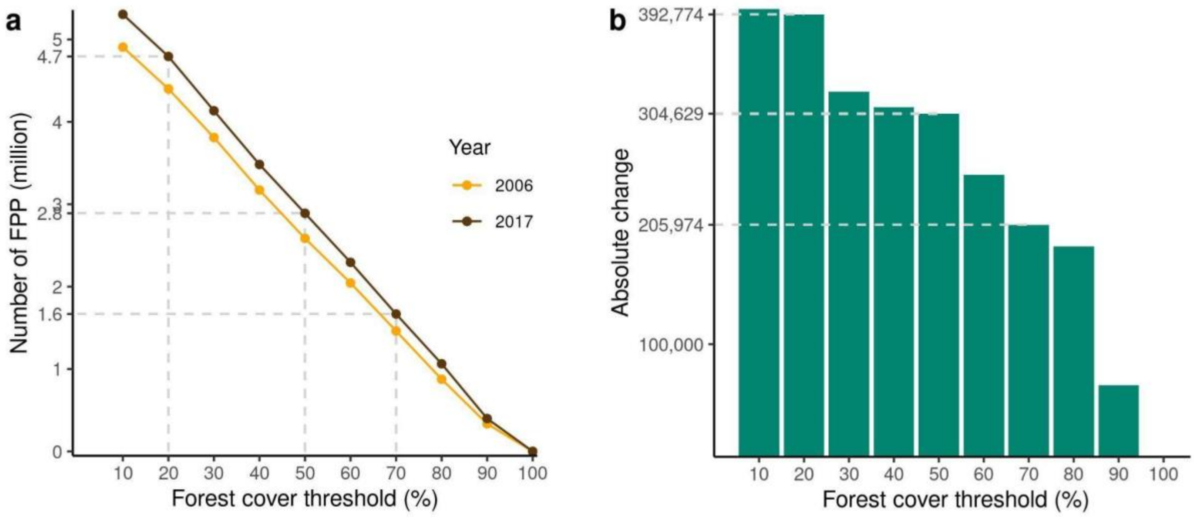


Figure 1. Estimated number of forest-proximate population in the sampled landscapes of Caatinga is sensitive to how forested landscapes are defined. **a)** Absolute number of forest-proximate people per threshold of minimum forest cover. **b)** Change in absolute number of FPP from 2006 to 2017 per threshold. We marked the 20, 50 and 70% forest cover threshold to highlight landscapes we consider as having low, intermediate and high forest covert.

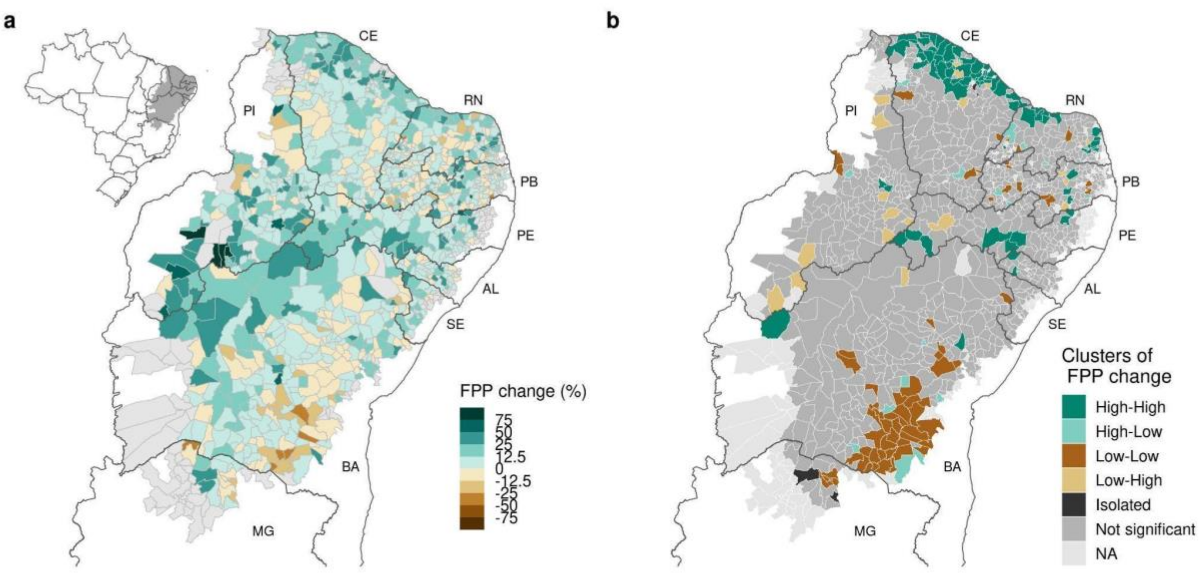


Figure 2. Spatial distribution of forest-proximate population change between 2006 and 2017 in municipalities in the Brazilian Caatinga dry forest. **a)** Percentage changes in the number of forest-proximate people (FPP). **b)** Spatial clusters of growing or declining forest-proximate populations, accounting for spatial autocorrelation (LISA analysis). “High-High” indicates places with growing

population clustered by neighbours also with growing population. “High-Low” are the ones with growing population surrounded by neighbours with depopulation, “Low-Low” are groups of municipalities experiencing depopulation and “Low-High” are municipalities with depopulation neighbouring municipalities with population growth. “Not significant” are municipalities not in core clusters. Black lines delimit Brazilian states, labelled with their acronyms PI, CE, RN, PB, PE, AL, SE, BA, MG are acronyms of Brazilian states.

Is there a consistent relationship between forest cover transitions and population change?

We found all four possible combinations between forest cover change and FPP change (Figure 3a). Nearly two-thirds of landscapes experienced population growth; most common was population growth combined with net forest regrowth (33.0% of all forested landscapes), followed closely by landscapes with deforestation and population growth (31.4%). A fifth (19.7%) of forested landscapes experienced depopulation and forest regrowth, whereas 15.0% experienced depopulation and deforestation (Supplementary Table 7). When aggregated to municipality scale (Figure 3b), we found four large clusters of forest regrowth spread across the biome. Deforestation happened mostly in the eastern part of Caatinga. The spatial structure of our data affected our linear models; We found the simple Generalised Linear Model indicated We found a positive correlation between forest cover and FPP change (GLM) but this correlation disappeared . However, when controlling for the spatial structure of those variables, this correlation disappears (Table 1).

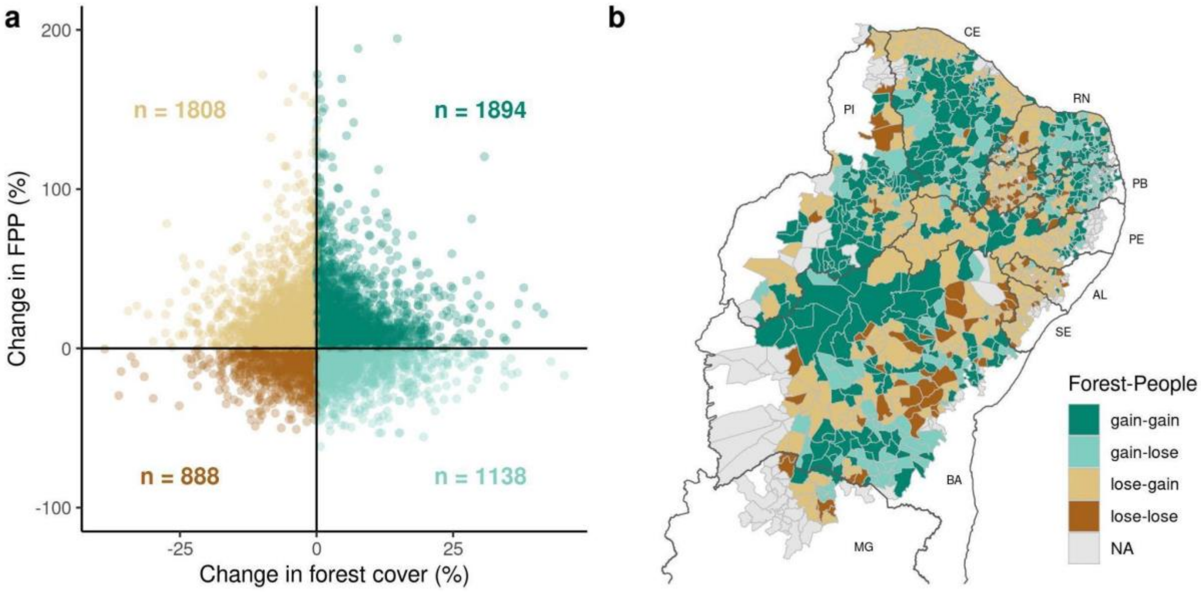


Figure 3. Transitions in forest cover and forest-proximate population size in the Caatinga, between 2006 and 2017. **a)** Simultaneous change in forest cover and forest-proximate people (FPP) at landscape scale (n=5748 forested landscapes) classified as: deforestation and rural population growth, reforestation and rural population growth, deforestation and rural depopulation and reforestation and rural depopulation. Annotations inside the figure are the number of landscapes in each category. **b)** Spatial distribution of the four types of population-environment transitions at municipality scale. PI, CE, RN, PB, PE, AL, SE, BA, MG are acronyms of Brazilian states.

Table 1. Results from a linear and a spatial regression model examining the relationship between municipality-scale forest cover change and forest-proximate population change between 2006 and 2017 . * - z-value for the Spatial Error Model.

Model type	Dependent variable	Independent variable	Estimate	Std. error	t-value*	*p*-value
Generalised Linear Model	Forest cover change (percentage)	population change (number of people)	-0.0439	0.0171	-2.561	0.0106

Spatial Error Model	Forest cover change (percentage)	population change (people)	-0.0045	0.0118	-0.3878	0.6981

How changes in development indicators were related to changes in forest cover and rural population?

We expected that each type of population-environment transition would be associated with distinct development conditions. Spatial models showed that three development indicators (i.e., family farming, urban population size and income inequality) were associated with FPP change, but none to forest cover change (Table 2 and Supplementary Table 10). Expansion of family-farming (estimate = 0.0072; z-value = 2.0773 p-value < 0.05) and urban population growth (estimate = 0.0001; z-value = 4.0314; p-value < 0.001) both had small, positive associations with the number of FPP, controlling for spatial autocorrelation. The Gini index also had a effect on FPP change, albeit inversely correlate (estimate = -12.729; z-value = -1.6992; p-value < 0.1) whereby greater income inequality was associated with depopulation of forested landscapes (Supplementary Table 10). These results show that population trends and forest cover change in forest landscapes are not necessarily linked to the same development processes.

Table 2. Results of Spatial Durbin Error Models assessing the effects of six development indicators and their spatially lagged equivalents on (i) landscape forest cover change and (ii) number of forest-proximate people, between 2006 and 2017 at municipal scale. Spatial lag refers to the effect of the variable from a neighbour on the observed municipality.

Model	**Change in development**	**Estimate**	**Standar**	**z-value**	***p*-value**

	indicator		d Error		
Change in forest cover					
	Family agriculture (% of total farmland)	0.0072	0.0145	0.4998	0.6171
	Urban population size	0.000007	0.00001	0.4460	0.6555
	HDI – Longevity (scaled index)	-16.115	12.336	-1.3064	0.1914
	Extreme poverty (% of households)	-0.0463	0.0294	-1.5717	0.1160
	Gini index (0 through 1)	4.3918	3.3704	1.3030	0.1925
	Under-five-mortality (per 1000 per year)	-0.0171	0.0456	-0.3752	0.7075
	Lagged family agriculture	-0.0165	0.0392	-0.4211	0.6736
	Lagged urban population	0.00005	0.00003	1.5476	0.1217
	Lagged HDI – Longevity	-33.024	25.864	-1.2768	0.2016
	Lagged extreme poverty	-0.1966	0.0804	-2.4429	0.0145
	Lagged Gini index	17.007	9.0434	1.8806	0.0600
	Lagged under five mortality	-0.0027	0.0950	-0.0290	0.9768
Change in forest-proximate people	Family agriculture	0.0679	0.0327	2.0773	0.0377
	Urban population	0.0001	0.00003	4.0314	0.00005

	HDI – Longevity	-21.251	31.644	-0.6716	0.5018
	Extreme poverty	0.0512	0.0667	0.7681	0.4424
	Gini index	-12.729	7.4911	-1.6992	0.0892
	Under five mortality	-0.0613	0.1152	-0.5322	0.5945
	Lagged family agriculture	0.1671	0.0766	2.1809	0.0291
	Lagged urban population	0.0002	0.00007	3.1080	0.0018
	Lagged HDI – Longevity	-46.026	43.042	-1.0693	0.2849
	Lagged extreme poverty	0.2908	0.1563	1.8604	0.0628
	Lagged Gini index	-10.106	17.788	-0.5681	0.5699
	Lagged under five mortality	-0.3365	0.1559	-2.1586	0.0308

Discussion

Rural population change does not necessarily cause environmental changes, however, there is a connection, which is complex and mediated by supra-local factors. This created different transition paths that this relationship can take, especially when considering the entire biome. This indirect relationship was also unveiled in the growth of forest-proximate population in the entire biome, along with many landscapes and municipalities experiencing depopulation. The four types of population-environment transition were not related to any clear pattern of changes in human development conditions, reinforcing the complexity of those transitions. Yet, changes in forest-proximate population size specifically seem influenced by changes in urban and rural contexts, mainly in terms of inequality (of income and access to land).

Increase in forest-proximate population size. Brazil, as many tropical countries in the world, experienced rapid urbanisation in the recent decades along with rapid expansion of export-oriented agriculture (43). Most recent data indicates that 87% of the Brazilian population now lives in cities (44) and that smallholders farms are decreasing in number and in area (41). Contrasting this tendency, rural populations in Caatinga increased, despite the region having a general reduction in number and area of family farming (Supplementary Figure 5). A possible explanation is that off-farm jobs and governmental assistance help to maintain people in rural areas, especially in less remote areas, reducing (or without) the need to engage in agricultural activities, maintaining people in rural areas even with decline in rural activities. In rural Zambia, for example, a cash-transfer program was associated with decrease in farm size, but only to the more educated households who get involved in non-farm activities (45) and in Niger, a cash-transfer program was associated with increased resilience of rural households to climatic shocks, avoiding migration and land abandonment (46). An in-depth understanding of rural livelihood changes in the Caatinga and how it is associated with changes in social and economical conditions is necessary to unveil the mechanisms behind rural population increase with decrease in family farming.

Despite the general increase in FPP, all other combinations of change also happened in the Caatinga. This reinforces the idea that environmental changes are not a direct consequence of population changes (1). Our linear model showed that both are correlated, however, when we control the spatial structure in the models, the correlation disappears, suggesting they only co-occur in the same place, but do not influence each other directly. This correlation is very often mediated by other factors.

In Nepal, for instance, depopulation occurred in the same place with forest cover increase, however this happens due to remittances from international migration and not because reduction in population size per se caused forest cover increase (6). In some cases, reducing local population density has been accompanied by an increase in forest cover through the displacement of rural populations due to ecological restoration projects or the creation of protected areas (47). Although the increase in forest cover may be seen as a positive outcome, this is often overshadowed by the social injustice of forced displacement and the worsening of welfare conditions (48). Furthermore, identifying that forest cover change and population change only co-occur in the same places in Caatinga is interesting because many studies suggest that small-scale agricultural practices is the main driver of deforestation in the region (28), but also in other tropical dry forests of the world (16). As suggested by our spatial models, the change in population is correlated to several changes in development indicators, however, forest cover is not, reinforcing the indirect connection between in population-environment transitions.

Sensitivity of forest-proximate population size. Forest-proximate population is very sensible to the definition of forested landscapes. This definition serves as a fundamental basis for understanding the characteristics and dynamics of the forest-proximate population. Our study showed that changing the minimum threshold of forest cover can change 20%, 50% or even 90% the estimates of forest-proximate population, depending on the threshold used. These sensitivity analyses provide valuable insights into the spatial distribution and dispersion patterns across varying levels of forest cover. In the case of Caatinga dry forest, the forest-proximate population seems to be uniformly distributed across the different levels of forest cover. This should not be the case of most forested regions, since it is expected more people living in less forested places, and the advance of an agricultural frontier is often related to deforestation and population increase, despite this is not always the case (49). Since sustainable development considers the improvement of people's welfare together with the recovery and preservation of forests, this information is extremely relevant to policy makers, because this provides the necessary basis for the promotion of public policies that take into account this dynamic and spatial distribution of people and forests.

Population-environment transitions are not directly connected. Apparently the influence of forest-proximate population size on forest cover is indirect and strongly influenced by spatial patterns of forest and people distribution. Because of the intrinsic characteristics of this type of population (26) and the strong dependency of forest resources by the rural population of Caatinga (29), we expected a strong, direct influence of population change on forest cover, however they seem only indirectly connected.

The indirect relation of people-environment transition suggests that factors affecting rural population change are different from those affecting forest cover change or happen at different spatial and temporal scales. Factors influencing forest cover change in the Caatinga and other dry forests are more related to agricultural expansion and desertification (16) while factors influencing population change are more likely to be related to political and economic processes (8). There is a long-stand understanding that political and economic processes also affect forest cover change, however these are generally classified as underlying causes (50).

The decoupled forest-population transition contradicts many studies pointing to an important role of population growth on deforestation. Rural population growth has tended to cause deforestation in developing countries (3, 4), although future deforestation may be influenced by urban, rather than rural, population changes (2, 8). An alternative explanation is that FPP influence on forest cover change is indirect and that changes in forest cover, in this context, are a secondary effect of development conditions. For example, Rodriguez-Garcia et al., (3) found that rural population growth causes deforestation in middle-income countries, however, this relation disappears when controlling for agricultural area, suggesting a mediating effect of agriculture in environment and population changes. In our analysis, we controlled for family agriculture areas and we could show that family agriculture was correlated to forest-proximate population change, but not to forest cover change. This suggests that family agriculture was not driving deforestation in Caatinga. We did not evaluate the effects of agriculture as a whole, which would include large-scale commodity production and cattle ranching. A pathway analysis, including other agricultural variables (e.g. soil fertility, access to rural credits, type of agricultural production) may help to reveal how change in population size affects forest cover through local development conditions and other mediating factors.

Finally, we may have not evaluated factors that have a mediating effect connecting environmental and population changes. Long and severe droughts still have much negative impact on rural livelihoods and small towns across Caatinga and other dry forests, disrupting food systems and local economies (50, 51). Even with the many social security programs in Brazil, such as construction of cisterns, cash-transfer programs and financial assistance for rural development, thousands of families and cities were affected between 2012 and 2017 during the most recent Grande Seca (long droughts) and several families had to migrate to urban areas (50, 52). Less obvious is the effect of droughts on forest cover change in dry forests. In humid forests, long droughts are associated with fire and forest degradation (53). In dry forest, the native vegetation evolved under regular drought conditions, and the reduction of pressure for planting may result in forest cover increase (51, 54). On the other hand, desertification may be an important cause of forest cover loss (55). Therefore, long droughts could have a role in mediating the observed joint changes of forest and FPP and deserve dedicated research to answer this question.

Development indicators and population-environment transitions. We found a less important role of development conditions in forest cover change than in population change. Neither rural or urban population growth were correlated to deforestation, suggesting again that the dynamics between environmental and population changes in Caatinga is likely to be different than most humid tropical forests.

The increase in rural population size was not related to improvement in health and monetary changes, as our results showed, making it hard to evaluate whether the population increase is happening in a desirable situation or not (i.e., lower poverty, better access to health and education). This demands more specific research to better characterise the socioeconomic context in places with rural population growth. However, we found rural population growth to be directly related to urban population growth, meaning that municipalities in general tend to gain or lose population as a whole. Mobility is an important historical strategy of the Caatinga population to deal with grandes secas (long droughts), migrating in millions to larger and richer cities in southeast Brazil (52). Nowadays, migration in the Caatinga is more common within the region, especially to regional urban centres (56). Our finding that municipalities with urban population growth also had rural population growth suggests that people are moving not only because of better opportunities at urban centres, but maybe due to opportunities in rural areas as well. Rural-rural mobility, therefore, will likely play an important role in population change in Caatinga, as it was in many tropical agricultural frontiers of Latin America (57).

We also found that rural population change is correlated with changes in monetary and land inequality. The positive correlation with family farm area is quite natural; places with increase in family farm areas are either attracting new people to settle (58) or second or third generation of the original settlers are opening new farms (59). The negative correlation with income inequality shows how rural societies, in the context of family agriculture, tend to be more equal (60). On the other hand, in capitalist agricultural frontier context, large scale landowners buy smaller farmers and in a process that the poorest share of the society is pushed to even more marginal lands, reducing their capacity to leave poverty, while leading to economic growth (due to the focus on commodity production), but raising local inequality levels (9, 60). This was observed in places with expansion of large scale soybean plantations in Brazil (61) and in Argentina (12), which resulted in less poor but more unequal societies.

Conclusion. Together our results pointed out that forest cover change only co-occur in the same places of forest-proximate population change and are not directly related to each other. If they are indeed decoupled, it is possible to restore forest cover in degraded areas of Caatinga without the negative consequences to human populations, such as evictions or loss of access to forest resources, as seen in Cambodia (47) for example. None of our development indicators or population change influenced forest cover change, thus more focused studies on understanding the proximate and distal causes of forest cover change in the Caatinga are necessary. This is important because dominant scientific discourse blames small-holders practices (e.g. slash-and-burn agriculture, firewood gathering) as primary causes of disturbance and deforestation in the Caatinga, despite recent research suggest a stronger role of pasture plantation as a major driver of forest cover change (62). However, landscapes under population growth or depopulation experienced forest cover loss alike. In fact, we observed that the majority of the sampled landscapes had forest cover gain and population gain, reinforcing that a smaller population is not necessary to recover forests (1). This also supports the idea that overconsumption should be a stronger factor responsible for biodiversity and ecosystems decline worldwide instead of overpopulation (63). We found that forest-proximate population change is related to changes in equality conditions (i.e., share of family farming and income inequality) and urban population, highlighting the role of rural and urban opportunities (e.g., off-farm jobs in urban centres, access to land) on population change. However, we still need more studies exploring the different geographical scales that the development conditions and population growth may be affecting environmental change in order to promote human development in harmony with recovering forests.

References

1. A. C. Hughes, et al., Smaller human populations are neither a necessary nor sufficient condition for biodiversity conservation. Biol. Conserv. 277, 109841 (2023).

2. R. S. DeFries, T. Rudel, M. Uriarte, M. Hansen, Deforestation driven by urban population growth and agricultural trade in the twenty-first century. Nat. Geosci. 3, 178–181 (2010).

3. V. Rodríguez García, N. Caravaggio, F. Gaspart, P. Meyfroidt, Long- and Short-Run Forest Dynamics: An Empirical Assessment of Forest Transition, Environmental Kuznets Curve and Ecologically Unequal Exchange Theories. Forests 12, 431 (2021).

4. T. Jusys, Fundamental causes and spatial heterogeneity of deforestation in Legal Amazon. Appl. Geogr. 75, 188–199 (2016).

5. M. Leach, J. Fairhead, Challenging Neo-Malthusian Deforestation Analyses in West Africa's Dynamic Forest Landscapes. Popul. Dev. Rev. 26, 17–43 (2000).

6. J. A. Oldekop, K. R. E. Sims, M. J. Whittingham, A. Agrawal, An upside to globalization: International outmigration drives reforestation in Nepal. Glob. Environ. Change 52, 66–74 (2018).

7. K. F. Davis, K. Yu, M. C. Rulli, L. Pichdara, P. D'Odorico, Accelerated deforestation driven by large-scale land acquisitions in Cambodia. Nat. Geosci. 8, 772–775 (2015).

8. D. López-Carr, A Review of Small Farmer Land Use and Deforestation in Tropical Forest Frontiers: Implications for Conservation and Sustainable Livelihoods. Land 10, 1113 (2021).

9. E. B. Barbier, Scarcity, frontiers and development. Geogr. J. 178, 110–122 (2012).

10. C. Dyngeland, J. A. Oldekop, K. L. Evans, Assessing multidimensional sustainability: Lessons from Brazil's social protection programs. Proc. Natl. Acad. Sci., 201920998 (2020).

11. C. Levers, et al., Agricultural expansion and the ecological marginalization of forest-dependent people. Proc. Natl. Acad. Sci. U. S. A. 118, e2100436118 (2021).

12. O. del Giorgio, B. E. Robinson, Y. le Polain de Waroux, Impacts of agricultural commodity frontier expansion on smallholder livelihoods: An assessment through the lens of access to land and resources in the Argentine Chaco. J. Rural Stud. 93, 67–80 (2022).

13. A. S. Mather, C. L. Needle, The forest transition: a theoretical basis. Area 30, 117–124 (1998).

14. T. K. Rudel, M. Perez-Lugo, H. Zichal, When Fields Revert to Forest: Development and Spontaneous Reforestation in Post-War Puerto Rico. Prof. Geogr. 52, 386–397 (2000).

15. B. Wu, et al., Essential dryland ecosystem variables. Curr. Opin. Environ. Sustain. 48, 68–76 (2021).

16. J. M. Schröder, L. P. Ávila Rodríguez, S. Günter, Research trends: Tropical dry forests: The neglected research agenda? For. Policy Econ. 122, 102333 (2021).

17. B. Fu, et al., The Global-DEP conceptual framework — research on dryland ecosystems to promote sustainability. Curr. Opin. Environ. Sustain. 48, 17–28 (2021).

18. L. Miles, et al., A global overview of the conservation status of tropical dry forests. J. Biogeogr. 33, 491–505 (2006).

19. J. M. C. da Silva, L. C. Barbosa, I. R. Leal, M. Tabarelli, “The Caatinga: Understanding the Challenges” in Caatinga: The Largest Tropical Dry Forest Region in South America, (Springer, 2017).

20. IBGE, Ed., Classificação e caracterização dos espaços rurais e urbanos do Brasil: uma primeira aproximação (IBGE, Instituto Brasileiro de Geografia e Estatística, 2017).

21. I. R. Leal, J. M. C. Da Silva, M. Tabarelli, T. E. Lacher, Changing the Course of Biodiversity Conservation in the Caatinga of Northeastern Brazil. Conserv. Biol. 19, 701–706 (2005).

22. E. M. S. Ribeiro, V. Arroyo-Rodríguez, B. A. Santos, M. Tabarelli, I. R. Leal, Chronic anthropogenic disturbance drives the biological impoverishment of the Brazilian Caatinga vegetation. J. Appl. Ecol. 52, 611–620 (2015).

23. E. V. S. B. Sampaio, R. S. C. Menezes, Y. de S. B. Sampaio,, A. D. S. de Freitas, "Sustainable Agricultural Uses in the Caatinga" in Caatinga: The Largest Tropical Dry Forest Region in South America, (Springer, 2017).

24. MapBiomas, Projeto MapBiomas – Coleção 7.0 da Série Anual de Mapas de Cobertura e Uso de Solo do Brasil (2023) (April 12, 2023).

25. P. Newton, A. T. Kinzer, D. C. Miller, J. A. Oldekop, A. Agrawal, The Number and Spatial Distribution of Forest-Proximate People Globally. One Earth 3, 363–370 (2020).

26. P. Newton, et al., "The number of forest- and tree-proximate people" (FAO, 2022) https:/doi.org/10.4060/cc2544en (April 12, 2023).

27. M. Antongiovanni, E. M. Venticinque, C. R. Fonseca, Fragmentation patterns of the Caatinga drylands. Landsc. Ecol. 33, 1353–1367 (2018).

28. M. Antongiovanni, E. M. Venticinque, M. Matsumoto, C. R. Fonseca, Chronic anthropogenic disturbance on Caatinga dry forest fragments. J. Appl. Ecol. 57, 2064–2074 (2020).

29. F. P. L. Melo, “The Socio-Ecology of the Caatinga: Understanding How Natural Resource Use Shapes an Ecosystem” in Caatinga: The Largest Tropical Dry Forest Region in South America, (Springer, 2017).

30. H. Hagel, C. Hoffmann, J. F. Irmão, R. Doluschitz, Socio-economic aspects of irrigation agriculture as livelihood for rural families in Brazil’s semi-arid northeast (2019) https:/doi.org/10.17170/kobra-20191127814 (August 12, 2020).

31. C. M. Souza Jr, et al., Reconstructing Three Decades of Land Use and Land Cover Changes in Brazilian Biomes with Landsat Archive and Earth Engine. 27 (2020).

32. A. J. Tatem, WorldPop, open data for spatial demography. Sci. Data 4, 170004 (2017).

33. IBGE, Atlas do censo demográfico 2010, 1st Ed. (IBGE, Instituto Brasileiro de Geografia e Estatística, 2013).

34. J.-F. Bastin, et al., The extent of forest in dryland biomes. Science 356, 635–638 (2017).

35. M. H. K. Hesselbarth, M. Sciaini, K. A. With, K. Wiegand, J. Nowosad, landscapemetrics : an open-source R tool to calculate landscape metrics. Ecography 42, 1648–1657 (2019).

36. R core Team, R: A language and environment for statistical computing. (2022).

37. X. Li, L. Anselin, rgeoda: R Library for Spatial Data Analysis (2023).

38. L. Anselin, Local Indicators of Spatial Association—LISA. Geogr. Anal. 27, 93–115 (1995).

39. R. Bivand, G. Millo, G. Piras, A Review of Software for Spatial Econometrics in R. Mathematics 9, 1276 (2021).

40. R. Bivand, R Packages for Analyzing Spatial Data: A Comparative Case Study with Areal Data. Geogr. Anal. 54, 488–518 (2022).

41. IBGE, Censo Agropecuário - Resultados definitivos. 109 (2020).

42. C. Y. Liu, G. Painter, Immigrant Settlement and Employment Suburbanisation in the US: Is There a Spatial Mismatch? Urban Stud. 49, 979–1002 (2012).

43. M. Schneider, C. A. Peres, Environmental Costs of Government-Sponsored Agrarian Settlements in Brazilian Amazonia. PLOS ONE 10, e0134016 (2015).

44. UN, World Urbanization Prospects The 2018 Revision (2020).

45. K. Lawlor, S. Handa, B. Davis, D. Seidenfeld, the Z. C. T. E. Team, Poverty-environment relationships under market heterogeneity: cash transfers and rural livelihoods in Zambia. Environ. Dev. Econ. 25, 291–314 (2020).

46. P. Premand, Q. Stoeffler, Cash transfers, climatic shocks and resilience in the Sahel. J. Environ. Econ. Manag. 116, 102744 (2022).

47. A. Scheidel, C. Work, Forest plantations and climate change discourses: New powers of ‘green’ grabbing in Cambodia. Land Use Policy 77, 9–18 (2018).

48. T. Osborne, et al., The political ecology playbook for ecosystem restoration: Principles for effective, equitable, and transformative landscapes. Glob. Environ. Change 70, 102320 (2021).

49. S. Sloan, Fewer People May Not Mean More Forest for Latin American Forest Frontiers. Biotropica 39, 443–446 (2007).

50. J. C. Dantas, R. M. da Silva, C. A. G. Santos, Drought impacts, social organization, and public policies in northeastern Brazil: a case study of the upper Paraíba River basin. Environ. Monit. Assess. 192, 317 (2020).

51. S. Lottering, P. Mafongoya, R. Lottering, Drought and its impacts on small-scale farmers in sub-Saharan Africa: a review. South Afr. Geogr. J. 103, 319–341 (2021).

52. J. A. Marengo, R. R. Torres, L. M. Alves, Drought in Northeast Brazil—past, present, and future. Theor. Appl. Climatol. 129, 1189–1200 (2017).

53. D. M. Lapola, et al., The drivers and impacts of Amazon forest degradation. Science 379, eabp8622 (2023).

54. A. P. M. Cunha, R. C. Alvalá, C. A. Nobre, M. A. Carvalho, Monitoring vegetative drought dynamics in the Brazilian semiarid region. Agric. For. Meteorol. 214–215, 494–505 (2015).

55. R. M. D. S. P. Vieira, et al., Desertification risk assessment in Northeast Brazil: Current trends and future scenarios. Land Degrad. Dev. 32, 224–240 (2021).

56. S. N. de Queiroz, R. Ojima, J. Campos, W. Fusco, Migração em cidades médias do interior nordestino: a atração migratória como elemento distintivo. Rev. Bras. Estud. Urbanos E Reg. (2020) https:/doi.org/10.22296/2317-1529.rbeur.202033pt (October 20, 2022).

57. D. Carr, Population and deforestation: why rural migration matters. Prog. Hum. Geogr. 33, 355–378 (2009).

58. J. L. Caviglia-Harris, E. O. Sills, K. Mullan, Migration and mobility on the Amazon frontier. Popul. Environ. 34, 338–369 (2013).

59. A. F. Barbieri, D. L. Carr, R. E. Bilsborrow, Migration Within the Frontier: The Second Generation Colonization in the Ecuadorian Amazon. Popul. Res. Policy Rev. 28, 291–320 (2009).

60. E. B. Barbier, Long run agricultural land expansion, booms and busts. Land Use Policy 93, 103808 (2020).

61. D. Weinhold, E. Killick, E. J. Reis, Soybeans, Poverty and Inequality in the Brazilian Amazon. World Dev. 52, 132–143 (2013).

62. M. M. do Espírito-Santo, et al., Biophysical and Socioeconomic Factors Associated to Deforestation and Forest Recovery in Brazilian Tropical Dry Forests. Front. For. Glob. Change 3, 569184 (2020).

63. A. R. Green, et al., A response to Cafaro, Hansson & Götmark (2022): Shifting the narrative from overpopulation to overconsumption. Biol. Conserv. 273, 109698 (2022).

5 CONSIDERAÇÕES FINAIS

Em geral, encontramos suporte para a hipótese de que municípios em estágios intermediários de desmatamento apresentam melhores índices de desenvolvimento. Combinado com isso, nosso achado que municípios mais desmatados apresentam uma estagnação ou até mesmo uma retração nos índices de desenvolvimento, dão suporte a ideia do padrão ‘boom-bust’ de desenvolvimento em fronteiras agrícolas. Somado a isso, encontramos também que os municípios com baixa cobertura florestal tendem a ter valores mais baixos de segurança alimentar, principalmente por também serem os municípios mais pobres e desiguais. Por outro lado, nossa análise longitudinal mostrou que os municípios melhoraram em todos os índices avaliados, independente da área total desmatada. Também observamos que os municípios em estágios avançados de desmatamento apresentaram uma melhora mais lenta. Isto enfraquece a ideia de um *bust* inevitável em locais muito degradados, mas também mostra que desmatar além de um certo limiar também não trás benefícios econômicos para o municípios em termos de desenvolvimento humano.

Nossa análise da dinâmica populacional e ambiental da Caatinga revelou que o desmatamento e a regeneração natural não estão diretamente conectados com mudanças no tamanho da população. Além disso, nenhum dos nossos indicadores de desenvolvimento parece estar relacionado à mudanças na cobertura florestal. Portanto, são necessários estudos mais focados na compreensão das causas próximas e distais da mudança da cobertura florestal na Caatinga. Isso é importante porque o discurso científico dominante culpa as práticas dos pequenos proprietários (por exemplo, agricultura de corte e queima, coleta de lenha) como as principais causas de distúrbios e desmatamento na Caatinga, apesar de pesquisas recentes sugerirem um papel mais forte da plantação de pastagens como um dos principais fatores de mudança na cobertura florestal.

Nossas análises também mostraram que as florestas somente não são o suficiente para garantir a segurança alimentar e o desenvolvimento de uma região. A pobreza e a desigualdade são os principais fatores limitantes da segurança alimentar e estão também em níveis altos nos municípios com alta cobertura florestal. Isso também ficou evidente quando observamos que mudanças nas populações rurais da Caatinga estão relacionadas à questões de desigualdade, tanto econômica quanto no acesso à terra. Melhorar a qualidade de vida das pessoas requer que os ganhos obtidos durante a fase inicial do desmatamento sejam mantidos no local e melhor distribuídos, além de um forte controle do desmatamento, de forma a evitar o colapso dos sistemas ecológicos que ajudam a manter o modo de vida dos Caatingueiros. Sendo assim, existe uma urgência de políticas sociais que visem combater diretamente a pobreza e desigualdade, de forma a aumentar diretamente a segurança alimentar, e de incentivos para evitar os extrativismo insustentável dos recursos naturais, evitando o desmatamento desnecessário.

6 REFERÊNCIAS

BARBIER, E. B. Scarcity, frontiers and development. **The Geographical Journal**, v. 178, n. 2, p. 110–122, 1 jun. 2012.

BARBIER, E. B. Long run agricultural land expansion, booms and busts. **Land Use Policy**, Mitigating boom & bust cycles: the roles of land policy and planning. v. 93, p. 103808, 1 abr. 2020.

BAUDRON, F. et al. Testing the Various Pathways Linking Forest Cover to Dietary Diversity in Tropical Landscapes. **Frontiers in Sustainable Food Systems**, v. 3, 2019.

CARAVAGGIO, N. Economic growth and the forest development path: A theoretical re-assessment of the environmental Kuznets curve for deforestation. **Forest Policy and Economics**, v. 118, p. 102259, set. 2020.

DAVIS, K. F. et al. Accelerated deforestation driven by large-scale land acquisitions in Cambodia. **Nature Geoscience**, v. 8, n. 10, p. 772–775, out. 2015.

DEFRIES, R. S. et al. Deforestation driven by urban population growth and agricultural trade in the twenty-first century. **Nature Geoscience**, v. 3, n. 3, p. 178–181, mar. 2010.

DOELMAN, J. C. et al. Afforestation for climate change mitigation: Potentials, risks and trade-offs. **Global Change Biology**, v. 26, n. 3, p. 1576–1591, 2020.

ERIKSSON, H. et al. Beyond social-ecological traps: fostering transformations towards sustainability. **Ecology and Society**, v. 26, n. 1, p. art13, 2021.

FAO, I. **The State of Food Security and Nutrition in the World 2020: Transforming food systems for affordable healthy diets**. Rome, Italy: FAO, IFAD, UNICEF, WFP and WHO, 2020.

GARCIA-POLO, J. et al. Restoring ecosystems and eating them too: guidance from agroecology for sustainability. **Restoration Ecology**, v. 29, n. 8, nov. 2021.

GREEN, A. R. et al. A response to Cafaro, Hansson & Götmark (2022): Shifting the narrative from overpopulation to overconsumption. **Biological Conservation**, v. 273, p. 109698, 1 set. 2022.

HUGHES, A. C. et al. Smaller human populations are neither a necessary nor sufficient condition for biodiversity conservation. **Biological Conservation**, v. 277, p. 109841, 1 jan. 2023.

JAGGER, P. et al. The Role of Forests and Trees in Poverty Dynamics. **Forest Policy and Economics**, v. 140, p. 102750, jul. 2022.

JUSYS, T. Fundamental causes and spatial heterogeneity of deforestation in Legal Amazon. **Applied Geography**, v. 75, p. 188–199, 1 out. 2016.

LEACH, M.; FAIRHEAD, J. Challenging Neo-Malthusian Deforestation Analyses in West Africa’s Dynamic Forest Landscapes. **Population and Development Review**, v. 26, n. 1, p. 17–43, mar. 2000.

LIU, J. et al. Comparative study of the forest transition pathways of nine Asia-Pacific countries. **Forest Policy and Economics**, Forest transition in Asia. v. 76, p. 25–34, 1 mar. 2017.

LÓPEZ-CARR, D. A Review of Small Farmer Land Use and Deforestation in Tropical Forest Frontiers: Implications for Conservation and Sustainable Livelihoods. **Land**, v. 10, n. 11, p. 1113, 21 out. 2021.

MATHER, A. S.; NEEDLE, C. L. The forest transition: a theoretical basis. **Area**, v. 30, n. 2, p. 117–124, jun. 1998.

MELO, F. P. L. et al. On the hope for biodiversity-friendly tropical landscapes. **Trends in Ecology & Evolution**, v. 28, n. 8, p. 462–468, 1 ago. 2013.
MILLENNIUM ECOSYSTEM ASSESSMENT (PROGRAM) (ED.). **Ecosystems and human well-being: synthesis**. Washington, DC: Island Press, 2005.

OLDEKOP, J. A. et al. An upside to globalization: International outmigration drives reforestation in Nepal. **Global Environmental Change**, v. 52, p. 66–74, 1 set. 2018.

RASMUSSEN, L. V. et al. Forest pattern, not just amount, influences dietary quality in five African countries. **Global Food Security**, v. 25, p. 100331, jun. 2020.

RAUDSEPP-HEARNE, C. et al. Untangling the Environmentalist’s Paradox: Why Is Human Well-being Increasing as Ecosystem Services Degrade? **BioScience**, v. 60, n. 8, p. 576–589, set. 2010.

RAZAFINDRATSIMA, O. H. et al. Reviewing the evidence on the roles of forests and tree-based systems in poverty dynamics. **Forest Policy and Economics**, v. 131, p. 102576, out. 2021.

ROCKSTRÖM, J. et al. Planetary Boundaries: Exploring the Safe Operating Space for Humanity. **Ecology and Society**, v. 14, n. 2, 18 nov. 2009.

RODRIGUES, A. S. L. et al. Boom-and-Bust Development Patterns Across the Amazon Deforestation Frontier. **Science**, v. 324, n. 5933, p. 1435–1437, 12 jun. 2009.

RODRÍGUEZ GARCÍA, V. et al. Long- and Short-Run Forest Dynamics: An Empirical Assessment of Forest Transition, Environmental Kuznets Curve and Ecologically Unequal Exchange Theories. **Forests**, v. 12, n. 4, p. 431, abr. 2021.

RUDEL, T. K. et al. Forest transitions: towards a global understanding of land use change. **Global Environmental Change**, v. 15, n. 1, p. 23–31, abr. 2005.

SACHS, J. D.; WARNER, A. M. The big push, natural resource booms and growth. **Journal of Development Economics**, v. 59, n. 1, p. 43–76, 1998.

SCHERER, L. et al. Trade-offs between social and environmental Sustainable Development Goals. **Environmental Science & Policy**, v. 90, p. 65–72, 1 dez. 2018.

TRITSCH, I.; ARVOR, D. Transition in environmental governance in the Brazilian Amazon: emergence of a new pattern of socio-economic development and deforestation. **Land Use Policy**, v. 59, p. 446–455, dez. 2016.

WEINHOLD, D.; REIS, E. J.; VALE, P. M. Boom-bust patterns in the Brazilian Amazon. **Global Environmental Change**, v. 35, p. 391–399, nov. 2015.

WILKES, M. A. et al. Making Way for Trees? Changes in Land-Use, Habitats and Protected Areas in Great Britain under "Global Tree Restoration Potential". **Sustainability**, v. 12, n. 14, p. 5845, 21 jul. 2020.

YANG, K. F.; GERGEL, S. E.; BAUDRON, F. Forest restoration scenarios produce synergies for agricultural production in southern Ethiopia. **Agriculture, Ecosystems & Environment**, v. 295, p. 106888, jun. 2020.

Apêndice A - Supplementary Material for 'Deforestation and human development in the Brazilian Caatinga dry forest'

Supplementary Table 1. List of potential confounding variables used to perform propensity score matching within a Generalised Linear Model (GLM) in order to evaluate the effect of changing native vegetation cover (treatment) on development indicators.

Confounding variable	Reasoning	source
Municipality area	Area is an important aspect of a municipality, since the large ones can hold more forest and other natural resources that could boom economic activities	(IBGE, 2017)
Percentage of population occupied in agricultural activities	As most Caatinga's municipalities are considered rural, more people engage in agricultural activities can denote more economic activity in the municipality and hence higher income and low poverty than municipalities with less agricultural activities	(IBGE, 2020)
Percentage municipality population living in urban areas	Higher population in urban centres can indicate a higher share of the municipality's economy from industries and services. Furthermore, this reduces pressure on forest cover from conversion to plantation or pasture, but higher from urban expansion	(IBGE, 2020)
Percentage of producers associated in cooperatives or unions	This is an important aspect of rural economy and can influence municipality economy	(IBGE, 2020)
Percentage of rural properties with springs protected by forests	Affect directly the total amount of forest cover but also the size of legal rural activities	(IBGE, 2020)
Percentage of rural properties with riversides protected by forests	Affect directly the total amount of forest cover but also the size of legal rural activities	(IBGE, 2020)
Percentage or rural properties with irrigation systems	Irrigation systems implicate higher agricultural productivity, but also denotes the purchasing power of producers. It can also affect the size of arable lands	(IBGE, 2020)
Coefficient of rainfall variation	Staple food production and pasture in Caatinga is mostly rainfed. The rainfall variation can	(Fick and Hijmans,

	indicate places with more or less agricultural production, which in turn can affect the economic activity and total forest cover	2017)
Percentage of rural properties classified as small	Threshold size for being classified as small in Brazil varies according to municipality size. Land structure can affect poverty and inequality in rural places and also forest cover, since large properties are obligated by law to preserve more land to forest protection	(Freitas et al., 2018)
Gross domestic product per capita from agriculture	GDP from agriculture is a direct measure of agricultural activities with implications to forest cover	(IBGE, 2012)
Gross domestic product per capita from industry	GDP from industry is a direct measure of municipality's industrialization level and can liberate land for forests	(IBGE, 2012)
Gross domestic product per capita from public services	GDP from public services is an indirect measure of urban activity and can denote more urbanised municipalities with more diverse economic matrix; with implications to forest cover	(IBGE, 2012)
Charcoal production	Industry in Caatinga is very dependent on charcoal produced with native trees. Therefore, the production of charcoal can indicate higher economic activity and is also directly related to forest cover	(IBGE, 2019)
Firewood production	Firewood consumption is very common among Caatinga's inhabitants. Therefore, production of firewood can be related to socio-economic conditions and is also directly related to forest cover	(IBGE, 2019)

Supplementary Table 2. Spatial autocorrelation test of residuals from GLMs models for each outcome evaluated

Model	Moran's I	*p*-value
HDI - Education	0.42	0

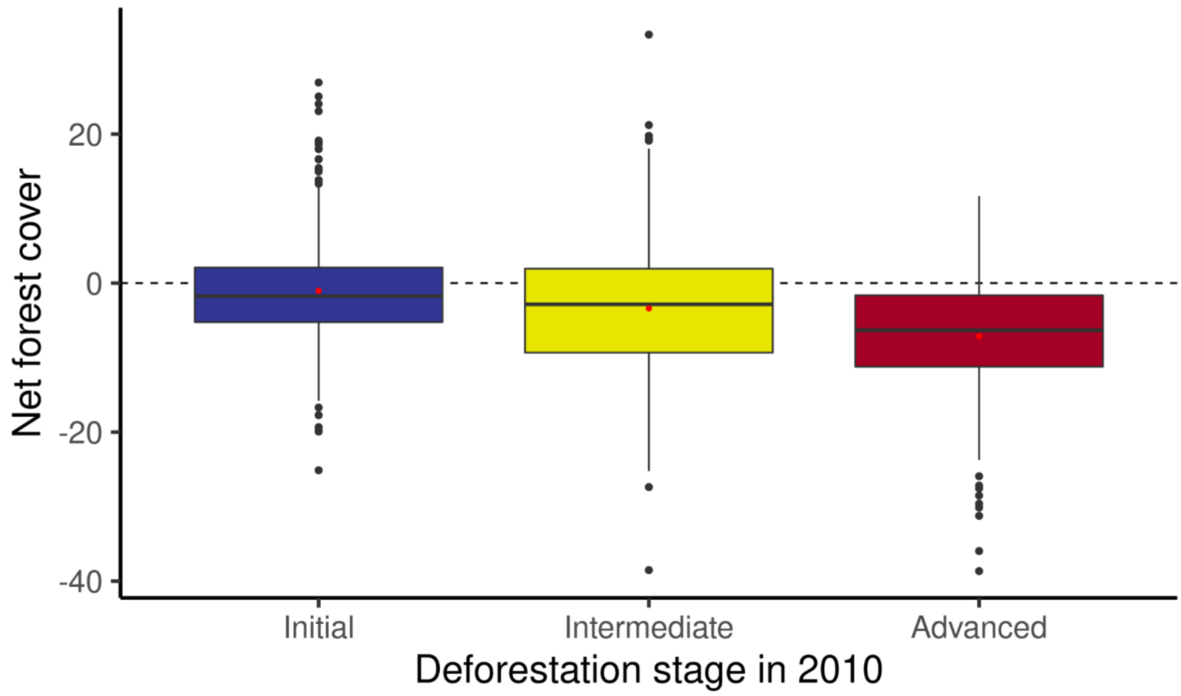
HDI - Longevity	0.18	0
HDI - Income	0.18	0
Extreme poverty	0.35	0
Gini index	0.24	0
Under five mortality	0.21	0

Supplementary Table 3. Municipalities change from one deforestation stage to another between 1991 and 2010 and the respective vegetation cover change and standard deviations for each group of municipalities.

Deforestation Stage 1991	Deforestation Stage 2010	Number of municipalities	% of the total	Vegetation cover change mean (%)	Vegetation cover change SD
Initial	Initial	432	35.79	-2.65	5.38
Initial	Intermediate	76	6.29	-11.94	6.59
Initial	Advanced	0	0		
Intermediate	Initial	58	4.80	11.00	6.04
Intermediate	Intermediate	330	27.34	-1.96	8.09
Intermediate	Advanced	85	7.04	-14.81	7.92
Advanced	Initial	0	0		
Advanced	Intermediate	11	0.91	12.92	8.47
Advanced	Advanced	215	17.81	-4.09	5.66
Total		1207	100	-3.36	8.38

Supplementary Table 4. Results of repeated-measures ANOVA comparing the effect of time and deforestation stage on socio-economic outcomes for municipalities in Caatinga Dry Forest based on Type II Wald chi-square tests. Values are chi-square values. * - *p*-value < 0.1; ** - *p*-values < 0.05; *** - *p*-values < 0.001.

Models	Deforestation stage	Time	Def.stage:Time
HDI - Education	16.48***	97185.56***	141.23***
HDI - Longevity	7.00**	33437.96***	49.35***
HDI - income	15.09***	22004.65***	77.23***
Extreme poverty	47.97***	13511.73***	69.20***
Gini index	31.09***	614.99***	8.28*
Under five mortality	8.41**	29820.78***	46.96***

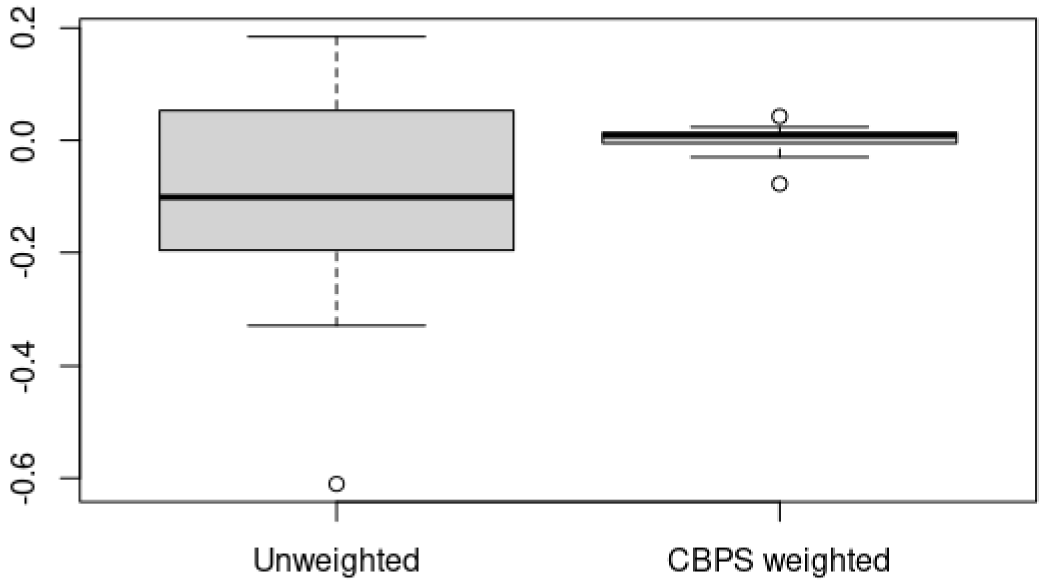


Supplementary Figure 1. Net forest cover change for each municipality from 1991 to 2010, grouped by their respective deforestation stage in 2010. Red dots inside the boxplots are the mean for each group.

Supplementary Table 5. Regression table of covariate balancing generalised propensity score model used to create the weights to reduce the correlation between the treatment (deforestation) and confounding variables.

Variable	Estimate	Standard Error	Z-value	p-value
Municipality area	-4.94	2.22	-2.23	0.0257
Percentage of population occupied in agricultural activities	2.11	0.865	2.44	0.0147
Percentage municipality population living in urban area	5.45	1.13	4.83	1.38e-06
Percentage of producers associated in cooperatives or unions	-0.696	1.03	-0.675	0.5
Percentage of rural properties with springs protected by forests	-1.51	1.39	-1.09	0.278
Percentage of rural properties with riversides protected by forests	-3.04	1.01	-3.01	0.00257
Percentage or rural properties with irrigation systems	1.55	0.816	1.9	0.0576
Coefficient of rainfall variation	-17.4	1.11	-15.7	0.000
Percentage of rural properties classified as small	3.46	1.05	3.3	0.00097

Gross domestic product per capita from agriculture	1.05	1.19	0.888	0.375
Gross domestic product per capita from industry	-0.904	1.34	-0.677	0.499
Gross domestic product per capita from public services	3.72	0.927	4.01	6.04e-05
Charcoal production	-1.47	0.86	-1.71	0.088
Firewood production	-1.4	1.41	-0.988	0.323



Supplementary Figure 2. Correlation differences between the unweighted model of treatment effect (total deforestation) and a set of 14 confounding variables and after CBPS weighting (covariate balancing propensity score) method. This graph shows a reduction in the absolute Pearson correlation among covariates and the treatment which could otherwise mask or create a false correlation between deforestation and development outcomes

Supplementary Table 6. Regression results of deforestation (def) and its quadratic term (def^2) on human development indicators. Note that this regression is weighted by covariate balancing generalised propensity scores.

Model	Variable	Estimate	Std. Error	t value	p-value
HDI - Education					
	Def	6.503e-04	3.635e-04	1.789	0.0739
	Def^2	-5.590e-06	3.959e-06	-1.412	0.1583
HDI - Longevity					
	Def	4.977e-04	1.568e-04	3.174	0.0015
	Def^2	-5.046e-06	1.708e-06	-2.954	0.0032
HDI - Income					
	Def	8.933e-04	2.046e-04	4.366	1.39e-05
	Def^2	-7.956e-06	2.229e-06	-3.570	0.0003
Extreme poverty					
	Def	-0.2892	0.0459	-6.298	4.44e-10
	Def^2	0.0025	0.0005	5.082	4.42e-07
Gini index					
	Def	-1.163e-03	2.789e-04	-4.171	3.29e-05
	Def^2	9.884e-06	3.038e-06	3.254	0.0011
Under five mortality					
	Def	-0.1172	0.0317	-3.688	0.0002
	Def^2	0.0012	0.0003	3.477	0.0005

Supplementary Table 7. Result of Spearman’s Rho correlation test between model residuals and deforestation.

Model	GLM with propensity score	Spatial model
HDI - Education	-0.229	-0.026
HDI - Longevity	-0.066	-0.004
HDI - Income	-0.019	0.005
Extreme poverty	-0.033	-0.010
Gini index	-0.011	-0.024
Under five mortality	0.095	0.002

Supplementary Table 8. Model averaging of repeated measures ANOVA models with lag deforestation data explaining several development indicators (Dev. indicator). Lag 0 is the 1991, 2000 and 2010 deforestation data, lag 1 is the 1990, 1999 and 2009 deforestation data, and so on until lag 5, which is 1986, 1995 and 2005. This table shows only the best models selected by a cut-off of ΔAICc < 2. The column Years of deforestation data represents the series of years used for that lag period. The weights were recalculated after the selection of best models. HDI - Human Development Index.

Dev. indicator	Lag	Years of deforestation data	ΔAICc	Weight
HDI - Education	2	1989-1998-2008	0.00	1.00
HDI - Longevity	4	1987-1996-2006	0.00	1.00
HDI - Income	0	1991-2000-2010	0.00	0.68
	1	1990-1999-2009	1.48	0.32
Extreme poverty	0	1991-2000-2010	0.00	0.57
	1	1990-1999-2009	0.58	0.43
Gini income inequality index	0	1991-2000-2010	0.00	0.51
	1	1990-1999-2009	1.27	0.27
	2	1989-1998-2008	1.75	0.21
Under five mortality	3	1988-1997-2007	0.00	1.00

Supplementary Table 9. Weighted averaged coefficients and Standard Errors of model averaging between a set of repeated measures ANOVA models with lag deforestation data. Lag 0 is the 1991, 2000 and 2010 deforestation data; lag 1 is the 1990, 1999 and 2009 deforestation data; lag 2 is 1989, 1998 and 2008, lag 3 is 1988, 1997 and 2007; lag 4 is 1987, 1996 and 2006; lag 5 is 1986, 1995 and 2005. *Lag of def. data* column represents the set of lagged deforestation data used in the model averaging to weight the coefficients and Standard Error. *Dev. indicator* column is the Development indicator analysed; *Dev. indicator date* is the date of census derived data; x - non-significant; * - p < 0.1; ** - p < 0.05; *** - p < 0.001.

			Comparison between years (weighted coefficient, Standard Errors & significance)			Comparison between deforestation stages (weighted coefficient, Standard Errors & significance)			
			Deforestation stage				Deforestation stage		
Dev. indicator	Dev. indicator date	Lag of def. data	Initial	Intermediate	Advance	Dev. indicator date	Initial vs Intermediate	Initial vs Advanced	Intermediate vs Advanced
HDI - Education	1991 - 2000	2	-0.1352 (0.0020) ***	-0.1329 (0.0019) ***	-0.1280 (0.0025) ***	1991	-0.0195 (0.0028) ***	-0.0220 (0.0037) ***	-0.0024 (0.0032) x
	2000 - 2010	2	-0.2557 (0.0020) ***	-0.2407 (0.0020) ***	-0.2277 (0.0024) ***	2000	-0.0172 (0.0029) ***	-0.0148 (0.0037) ***	0.0023 (0.0032) x
	1991 - 2010	2	-0.3909 (0.0020) ***	-0.3736 (0.0020) ***	-0.3558 (0.0025) ***	2010	-0.0022 (0.0028) x	0.0131 (0.0035) ***	0.0153 (0.0031) ***
HDI - Longevity	1991 - 2000	4	-0.0925 (0.0017) ***	-0.0961 (0.0017) ***	-0.1038 (0.0023) ***	1991	0.0099 (0.0022) ***	0.0198 (0.0028) ***	0.0099 (0.0025) ***
	2000 - 2010	4	-0.0945 (0.0018) ***	-0.0996 (0.0017) ***	-0.1026 (0.0022) ***	2000	0.0063 (0.0023) x	0.0085 (0.0028) *	0.0022 (0.0025) x
	1991 - 2010	4	-0.1871	-0.1958 (0.0017)	-0.2065 (0.0022)	2010	0.0012 (0.0023)	0.0004 (0.0028)	-0.0007 (0.0024)

			(0.0017) ***	***	***		x	x	x
HDI - Income	1991 - 2000	0 and 1	-0.0682 (0.0016) ***	-0.0660 (0.0015) ***	-0.0539 (0.0021) ***	1991	-0.0120 (0.0023) ***	-0.0201 (0.0029) ***	-0.0081 (0.0025) **
	2000 - 2010	0 and 1	-0.0862 (0.0016) ***	-0.0807 (0.0016) ***	-0.0775 (0.0020) ***	2000	-0.0099 (0.0023) ***	-0.0058 (0.0029) x	0.0040 (0.0025) x
	1991 - 2010	0 and 1	-0.1544 (0.0016) ***	-0.1468 (0.0016) ***	-0.1314 (0.0020) ***	2010	-0.0044 (0.0023) x	0.0028 (0.0028) x	0.0072 (0.0024) x
Extreme poverty	1991 - 2000	0 and 1	15.9570 (0.4448) ***	15.6388 (0.4166) ***	11.3321 (0.5714) ***	1991	4.0809 (0.5840) ***	6.7543 (0.7427) ***	2.6734 (0.6613) ***
	2000 - 2010	0 and 1	16.6909 (0.4498) ***	15.3064 (0.4334) ***	15.9043 (0.5387) ***	2000	3.7627 (0.5934) ***	2.1294 (0.7293) *	-1.6332 (0.6462) x
	1991 - 2010	0 and 1	32.6480 (0.4487) ***	30.9452 (0.4328) ***	27.2365 (0.5538) ***	2010	2.3782 (0.5932) ***	1.3429 (0.6969) x	-1.0353 (0.6295) x
Gini income inequality index	1991 - 2000	0, 1 and 2	-0.0528 (0.0035)	-0.0379 (-0.0379) ***	-0.0514 (0.0045) ***	1991	0.0017 (0.0038) x	0.0094 (0.0094) x	0.0077 (0.0044) x

			***						
	2000 - 2010	0, 1 and 2	0.0455 (0.0036) ***	0.0435 (0.0034) ***	0.0482 (0.0043) ***	2000	0.0166 (0.0039) ***	0.0109 (0.0045) x	-0.0057 (0.0043) x
	1991 - 2010	0, 1 and 2	-0.0073 (0.0036) x	0.0056 (0.0034) x	-0.0031 (0.0044) x	2010	0.0146 (0.0039) ***	0.0135 (0.0043) **	-0.0011 (0.0042) x
Under five mortality	1991 - 2000	3	36.0017 (0.6566) ***	39.4287 (0.6322) ***	42.1811 (0.8686) ***	1991	-2.9303 (0.8041) ***	-7.1391 (0.9869) ***	-4.2088 (0.9306) ***
	2000 - 2010	3	29.9545 (0.6785) ***	29.4403 (0.6285) ***	30.1090 (0.8257) ***	2000	0.4966 (0.8136) x	-0.9597 (1.0036) x	-1.4564 (0.9230) x
	1991 - 2010	3	65.9562 (0.6649) ***	68.8691 (0.6473) ***	72.2902 (0.8400) ***	2010	-0.0174 (0.8188) x	-0.8051 (0.9672) x	-0.7877 (0.8956) x

Supplementary Table 10. Number of municipalities (total n = 1207) that change the deforestation stage when comparing the lag 0 deforestation data (years 1991, 2000 and 2010) with lagged deforestation data. Nº of switches includes municipalities that changed to high forested stages through regeneration (e.g. from advanced to intermediate).

year	Lag	nº of municipalities at initial stage	nº of municipalities at intermediate stage	nº of municipalities at advanced stage	Nº of switches	% of municipalities that switches
1991	**0**	**434**	**512**	**261**	**-**	**-**
1990	1	442	511	254	47	4

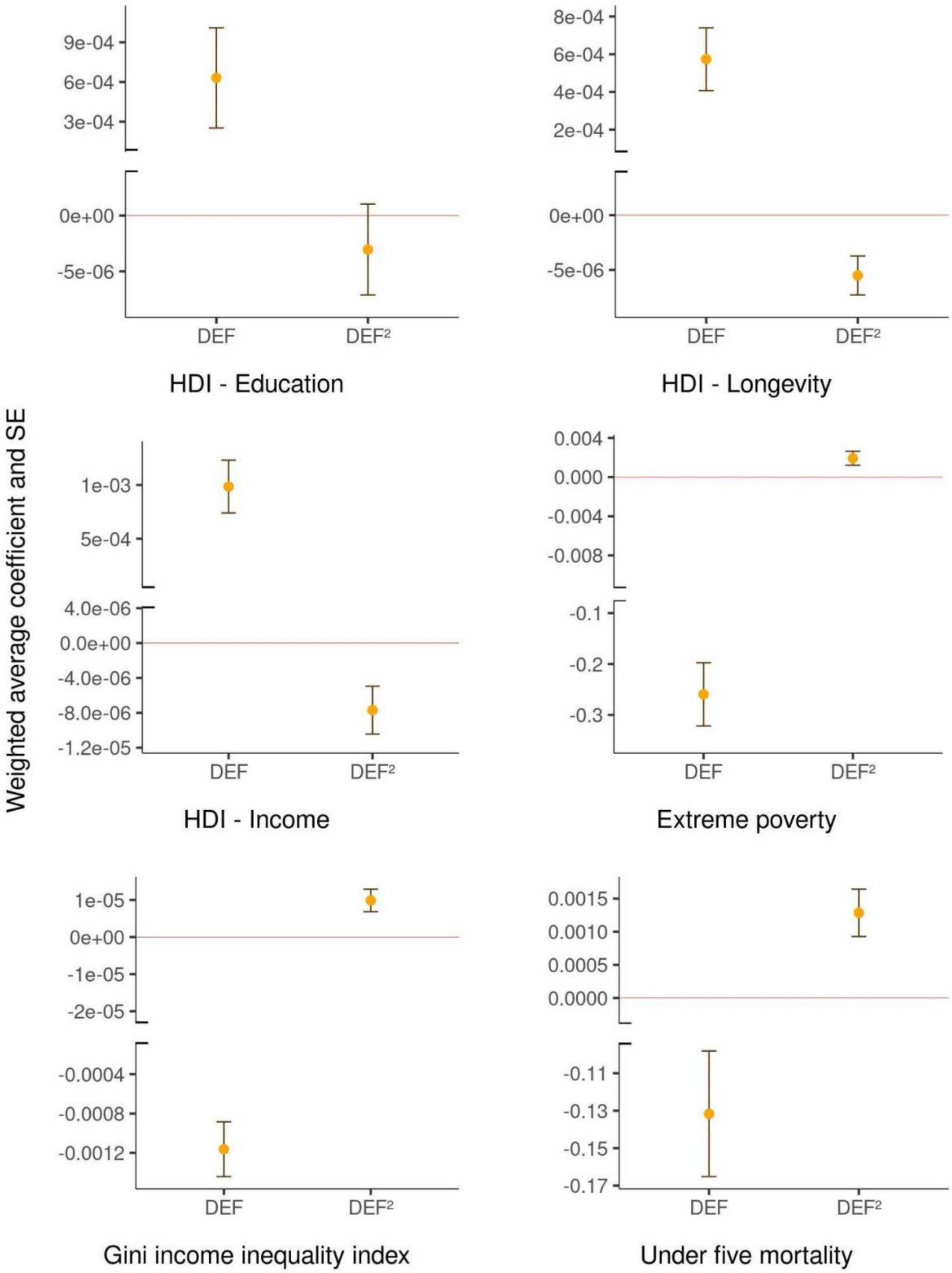
1989	2	465	480	262	90	7
1988	3	469	478	260	110	9
1987	4	480	472	255	134	11
1986	5	494	478	235	160	13
2000	**0**	**425**	**502**	**280**	**-**	**-**
1999	1	428	494	285	66	5
1998	2	426	488	293	96	8
1997	3	426	514	267	96	8
1996	4	439	504	264	127	10
1995	5	442	502	263	137	11
2010	**0**	**430**	**449**	**328**	**-**	**-**
2009	1	425	457	325	56	5
2008	2	421	460	326	77	6
2007	3	418	484	305	123	10
2006	4	419	490	298	121	10
2005	5	425	486	296	125	10

Supplementary Table 11. Results of model averaging on the effect of lagged deforestation data on several development indicators (Dev. indicator), serving as a sensitivity analysis for our cross-sectional analysis using propensity scores weighting. Bold lines are the models with ΔAICc below 2 that were used to calculate the averaged coefficients (Avg. coef.) and Standard Error (Avg. SE). Weights were recalculated after selecting the models with ΔAICc below 2. As we expected a quadratic relationship between deforestation and development (see main text), we built a linear model including the quadratic term of our predictor variable (deforestation). DEF - single term for total deforestation; DEF^2 - quadratic term of total deforestation.

Dev. indicator	Year	Lag	ΔAICc	Weight	Variable	Avg. coef.	Avg. SE	p-value
HDI - Education	**2005**	**5**	**0.00**	**1.0**				
	2008	2	2.21	0.00				
	2009	1	2.95	0.00				
	2006	4	3.87	0.00				
	2007	3	4.85	0.00				
	2010	0	10.24	0.00				
					DEF	6.306e-04	3.785e-04	0.096
					DEF^2	-3.045e-06	-0.747	0.455
HDI - Longevity	**2006**	**4**	**0.00**	**0.39**				
	2007	**3**	**0.54**	**0.30**				
	2005	**5**	**1.73**	**0.16**				
	2008	**2**	**1.96**	**0.15**				
	2009	1	2.40	0.00				
	2010	0	3.46	0.00				
					DEF	5.734e-04	1.663e-04	0.0005
					DEF^2	-5.511e-06	1.776e-06	0.0019
HDI - Income	**2008**	**2**	**0.00**	**0.29**				
	2007	**3**	**0.23**	**0.26**				
	2006	**4**	**0.44**	**0.23**				
	2009	**1**	**0.65**	**0.21**				
	2005	5	2.82	0.00				
	2010	0	15.40	0.00				
					DEF	9.854e-04	2.458e-04	6.18e-05
					DEF^2	-7.696e-06	2.742e-06	0.0050

Extreme poverty	**2008**	**2**	**0.00**	**0.39**				
	2009	**1**	**0.77**	**0.27**				
	2006	**4**	**1.62**	**0.18**				

	2007	**3**	**1.78**	**0.16**				
	2005	5	6.18	0.00				
	2010	0	10.85	0.00				
					DEF	-0.2595	0.0620	2.93e-05
					DEF²	0.0019	0.0007	0.0067
Gini income inequality index	**2010**	**0**	**0.00**	**1.00**				
	2005	5	2.33	0.00				
	2006	4	4.99	0.00				
	2009	1	6.80	0.00				
	2008	2	7.25	0.00				
	2007	3	8.20	0.00				
					DEF	-1.163e-03	2.789e-04	3.29e-05
					DEF²	9.884e-06	3.038e-06	0.0011
Under five mortality	**2006**	**4**	**0.00**	**0.45**				
	2007	**3**	**0.51**	**0.35**				
	2005	**5**	**1.70**	**0.19**				
	2008	2	2.60	0.00				
	2010	0	3.06	0.00				
	2009	1	3.34	0.00				
					DEF	-0.1316	0.0335	8.71e-05
					DEF²	0.0012	0.0003	0.0003



Supplementary Figure 3. Weighted average coefficient and Standard Errors (SE) of models with different lagged deforestation (DEF) and its quadratic term (DEF^2) on several development indicators. Lagged deforestation data goes from 0 (year of 2010) to 5 (year of 2005), but only the models with ΔAICc below 2 were included in this figure. Except for the DEF^2 in HDI (Human Development Index) - Education, all effects were robust to lagged data on deforestation differing from zero.

APÊNDICE B – Material suplementar artigo “Tradeoffs and synergies between food security and forest cover in Brazilian drylands”

Supplementary table 1. Variables used to make the Principal Components Analysis to build the food security index.

	variable	dimension of food security	description	unit	year	premise	Source*
1	renda	access	Average per capita income of households in each municipality	R$ per household	00, 10	The higher the household per capita income, the higher are their monetary conditions to buy food	IBGE
2	meioSal	access	Percentage of the population of the municipality with an income below half of minimum wage	%	00, 10	If a high share of the population lives with income below half of minimum wage, less people in a given municipality should have access to buy food	IBGE
3	quartSal	access	Percentage of the population of the municipality with an income below a quarter of the minimum wage	%	00, 10	If a high share of the population lives with income below a quarter of minimum wage, even less people (compared to half minimum salary) in a given municipality should have access to buy food	IBGE
4	txDesemp	access	Percentage of the number of people in the municipality over 16 who are unemployed	%	00, 10	More unemployed people means less family income, hence lower access to food	IBGE
5	imcBaixo	access	Percentage of the number of people in the municipality with body mass index below ideal	%	08, 17	Body mass index below the ideal at the population level should represent the lack of access to nutritious food	SISVAN
6	imcAlto	utilisation	Percentage of the number of people in the municipality with obesity	%	08, 17	Body mass index above the ideal at the population level should represent the consumption of low quality food	SISVAN
7	dai	access	Percentage of the number of children below height expected for their age	%	08, 17	Stunting is a very common problem in children with chronic undernourishment and represents the lack of access to food during the first childhood	SISVAN
8	dpi	access	Percentage of the number	%	08, 17	Weight deficit is another common problem in children with chronic	SISVAN

			of children below expected weight for their age			undernourishment and represents the lack of access to food during the first childhood	
9	T_ANALF25M	access	Percentage of the population aged 25 and over who is illiterate	%	00, 10	Illiteracy often constrain one's capacity to get good jobs and have access to loans or financing, hence reducing the access to the food systems	IBGE
1	T_MED25M	access	Percentage of the population aged 25 or over who has completed high school	%	00, 10	Completion of high school increases the chances of getting a good job and being better paid	IBGE
1	T_SUPER25M	access	Percentage of the population aged 25 or over who has completed higher education	%	00, 10	Completion of graduation level increases the chances of getting a good job and being better paid	IBGE
1	RAZDEP	stability	Proportion of the population under 15 and over 65 in relation to the population between 15 and 64 years	%	00, 10	high levels of dependency means more people depending on few economically active people of the household to sustain their income and access to food. This means less resilience to economic shocks	IBGE
1	RDPC1	access	Average per capita income of the poorest fifth in the municipality	R$ of 2010	00, 10	The higher the income of the poorest fifth of the population, the better the population's access to food	IBGE
1	expov	access	Percentage of the population of municipalities in extreme poverty	%	00, 10	People under the extreme poverty line are among the most vulnerable people in the world. Thus, this represent people with very little access to the food systems	IBGE
1	pov	access	Percentage of the population living in poverty	%	00, 10	People under the poverty line are very vulnerable and should have compromised access to the food systems	IBGE
1	PREN20RICOS	access	Percentage of total municipal income appropriated by the 20% of the population with the highest per capita household income	%	00, 10	The higher the level of this variable, the higher is the number of people in poverty. Inequality is among the main causes of lack of access to food systems and to the means of producing their own food (lack of access to land).	IBGE
1	gini	access	Gini index of income	%	00, 10	Gini index of income is a very common index that represents differences in income among people living in the same analysed area and, consequently, to the food system	IBGE
18	P_AGRO	availability	Percentage of the	%	00, 10	More people working in the agricultural sector can be translated	IBGE

			population aged 18 or over in the agricultural sector			into a very active sector of the municipality's economy, meaning more food production and a more dynamic food system.	
19	AGUA_ESGOTO	utilisation	Percentage of people in households with inadequate water supply and sanitation	%	00, 10	Access to good quality water and adequate sanitation should mean better utilisation of food and lower risk of water-borne disease.	IBGE
20	MORT5	access	Probability of dying between birth and the exact age of 5, per 1000 children born alive.	0-100	00, 10	Child death is often related to the lack of basic needs of the population (e.g. sanitation, secure food) and is a good proxy for overall population access	IBGE
21	perc_rur	availability	Percentage of municipality inhabitants in rural areas	%	00, 10	Large rural population mean more people demanding food, but also more people engaged in rural activities and partially depending on forest food instead of the food production	IBGE
22	perc_urb	availability	Percentage of municipality inhabitants in urban areas	%	00, 10	Large urban population mean more people demanding food but not producing it	IBGE
23	agrotx	availability	percentage of properties using pesticides in production	%	06, 17	Although food production with agrotoxic produces less health food, it increases the overall productivity of the system, raising food availability	IBGE rural census
24	agroOrg	utilisation	percentage of properties with organic production	%	06, 17	Organic food production produces healthier food, suggesting a better utilisation of the food produced	IBGE rural census
25	prodAss	stability	percentage of producers associated with cooperatives and/or class entities	%	06, 17	Producers often have difficulty getting financing or access to markets individually. Cooperatives or class entities can improve the access and guarantee long and short term stability to the producers, hence to the municipality food system	IBGE rural census
26	chefeMulher	availability	Percentage of properties headed by women	%	06, 17	Rural women often have the burden of being responsible for multiple activities not related to food production (e.g. child caring). When the property is headed by an woman, the food production tend to be reduced	IBGE rural census
27	nascProt	stability	Percentage of properties with springs protected by forest	%	06, 17	Springs are very important for food production, especially in drylands. The native vegetation protecting those springs helps to maintain water quality and quantity and the stability of the municipal food system	IBGE rural census
28	nascNprot	stability	Percentage of properties with springs not protected by forest	%	06, 17	Springs not protected by forest are more unstable and prone to dry during the drought season. Also, they should have lower water quality	IBGE rural census
29	riosProt	stability	Percentage of properties with rivers protected by forest	%	06, 17	Protected rivers help to maintain water quality and quantity and the stability of the municipal food system.	IBGE rural census

30	riosNprot	stability	Percentage of properties with rivers not protected by forest	%	06, 17	Unprotected rivers should have lower water quality and be more prone to silting. This should also result in lower water flow and promote more erosion, compromising the stability of the food system	IBGE rural census
31	pocosComun	utilisation	Percentage of households with common wells	%	06,17	Wells are a very important source of water for consumption, especially for people living in drylands such as the caatinga. Better water access should be translated to a better utilisation and lower health issues related to water	IBGE rural census
32	cist	utilisation	Percentage of households with cisterns	%	06,17	Cisterns are fundamental for guaranteeing water access to rural populations without public water supply. It improve the food utilisation by stocking good quality water for cooking, consumption and even for irrigation during the dry season	IBGE rural census
33	irrig	availability	percentage of properties with any types of irrigation	%	06, 17	Irrigation is very important for food production and improves farm yields. In drylands such as the Caatinga, irrigation are less common but play an important role in areas close to rivers or dams	IBGE rural census
34	finan	stability	percentage of the total number of properties with financing	%	06, 17	Financing is very important for food production because it guarantees the capacity of the producer to plant, harvest and process the food. More properties with financing should promote better stability to the municipal food system	IBGE rural census
35	cornMed	availability	Municipal average corn production	mean per property (tons)	06, 17	Corn is among the main food produced in Caatinga, thus it is a good proxy for overall food production in the region	IBGE rural census
36	beanMed	availability	Municipal average beans production	mean per property (tons)	06, 17	Bean is one of the main crops in Caatinga. Beans are commonly planted interspersed with corn and is frequently found in local markets	IBGE rural census
37	bovMed	availability	Municipal livestock production of cattle	mean per property (tons)	06, 17	Cattle production is a strong cultural practice in Caatinga and can be a good proxy for animal protein production	IBGE rural census
38	capMed	availability	municipal goat production	mean per property (tons)	06, 17	Because it is very drought resistant, goats are raised by many farmers in the region, especially the poorest. Thus, goat production is a good proxy for food security, especially for the poorer share of the population	IBGE rural census

*Link to access and download the data is provided in Supplementary table 4.

Supplementary table 2. Dimensions of food security index in 2006, the respective variables and their sign towards food security.

Dimension	variable	cardinality	explained variation
PC 1 – Poverty			25.51%
	renda	+	
	meioSal	-	
	quartSal	-	
	pov	-	
	expov	-	
	rdpc1	+	
PC 2 – Inequality			7.11%
	pren20ricos	-	
	gini	-	
PC 3 – Stability			6.76%
	prodAss	+	
	finan	+	
PC 4 – Child nutrition			4.38%
	dai	-	
	dpi	-	
PC 5 – Goat production			4.30%
	capMed	+	
PC 6 – Availability			4.26%
	p_agro	+	
	perc_urb	-	
PC 7 – Agricultural intensification			3.67%
	agrotx	+	
	irrig	+	
PC 8 – Agricultural production			3.51%
	beanMed	+	
	cornMed	+	
PC 9 – Cattle production			3.30%

	bovMed	+	
PC 10 – Protection of water resources			3.16%
	nascProt	+	
	riosProt	+	
PC 11 – Utilisation			2.79%
	imcAlto	-	
	agua_esgoto	-	
PC 12 – APP* deficit			2.66%
	nascNprot	-	
	riosNprot	-	

* In Brazil, the federal law of native vegetation protection determines a certain buffer around rivers and other water bodies that should be covered with native forest. These areas are known as Areas of Permanent Preservation (APP)

Supplementary Table 3. Dimensions of food security index in 2017 and the respective variables and their sign towards food security.

Dimension	variable	cardinality	explained variation
PC 1 - Poverty			26.22%
	renda	+	
	meioSal	-	
	quartSal	-	
	pov	-	
	expov	-	
	rdpc1	+	
PC 2 - Inequality			8.25%
	pren20ricos	-	
	gini	-	
PC 3 – Gender driven Inequality and Stability			5.76%
	chefeMulher	-	
	finan	+	
	riosProt	+	
PC 4 – APP* Protection			5.59%
	nascProt	+	
	nascNprot	-	
PC 5 – Producer stability			4.86%
	prodAss	+	
	cist	+	
	irrig	+	
PC 6 – Agricultural production			4.00%
	beanMed	+	
	cornMed	+	
PC 7 – Child nutrition			3.72%
	dpi	-	
	dai	-	
PC 8 – Goat production			3.39%

	capMed		
PC 9 – APP* deficit			3.25%
	riosNprot	-	
PC 10 – Unemployment			2.95%
	txDesemp	-	
PC 11 – Adult Nutrition			2.78%
	imcBaixo	-	
	imcAlto	-	
PC 12 – Availability			2.50%
	p_agro	+	
	perc_urb	-	

* In Brazil the federal law of native vegetation protection determines a certain buffer around rivers and other water bodies that should be covered with native forest. These areas are known as Areas of Permanent Protection (APP)

Supplementary table 4. Description of the data source used to build the Multidimensional Food Security Index and requirements to download and use the data

Source	Description	Requirements	Link
IBGE	Data gathered from IBGE decadal national census. Each variable was download from a different table recovered from SIDRA website and compiled by the authors	There are no requirements to download and use the data	https://sidra.ibge.gov.br/pesquisa/censo-demografico/demografico-2000/amostra-primeiros-resultados **and** https://sidra.ibge.gov.br/pesquisa/censo-demografico/demografico-2010/amostra-resultados-gerais
SISVAN	Data gathered from public access reports on the Health Ministry website. Each variables was downloaded individually and organised by the authors	There are no requirements to download and use the data	https://sisaps.saude.gov.br/sisvan/relatoriopublico/index for both years
IBGE rural census	Data gathered from IBGE decadal rural census. Each variable was download from a different table recovered from SIDRA website and compiled by the authors	There are no requirements to download and use the data	https://sidra.ibge.gov.br/pesquisa/censo-agropecuario/censo-agropecuario-2006/segunda-apuracao **and** https://sidra.ibge.gov.br/pesquisa/censo-agropecuario/censo-agropecuario-2017/resultados-definitivos

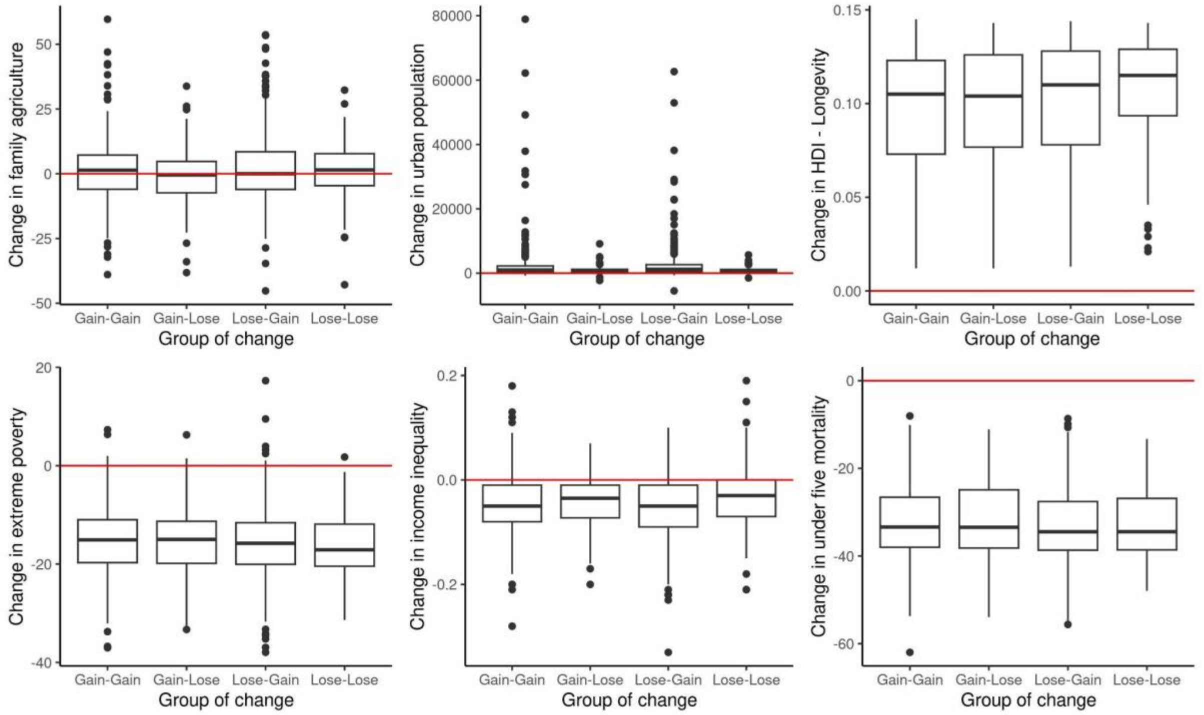
APÊNDICE C – Material suplementar artigo "Population-environment transitions in the Brazilian Caatinga dry forest"

Supplementary Table 1. development indicators used to evaluate the development context of municipalities

Variable	Definition	Unit	Years	Source
Family agriculture share	Percentage of agricultural land occupied by family agriculture	%	2006 and 2017	Rural census (IBGE, 2020)
Urban population size	Total number of people living in the urban area of the municipality	absolute number	2000 and 2010	National census (IBGE, 2013)
HDI - Longevity	Brazilian adaptation of the Human Development Index adapted for municipality-scale analysis. The HDI - Longevity is the person's mean life expectancy at birth	Unitless, varies from 0 to 1	2000 and 2010	Atlas Humano do Brasil (Atlas Brasil, 2020)
Extreme poverty rate	Percentage of households below the extreme poverty line. The line is based on official categories of per capita income, whose thresholds increase over time	%	2000 and 2010	National census (IBGE, 2013)
Gini index	Measures inequality of monetary income distribution within a population	unitless, varies from 0 to 1	2000 and 2010	National census (IBGE, 2013)
Under five mortality rate	number of children death under five years old per thousand children per year	absolute number	2000 and 2010	National census (IBGE, 2013)

Supplementary Table 2. Descriptive statistics of six development indicators for municipalities within the Brazilian Caatinga dry forest biome in 2017.

Development indicator	**Minimum**	**Maximum**	**Mean**	**Standard deviation**
Family agriculture share	4.2%	95.6%	51.0%	19.0%
Urban population size	238	510,635	13,098	30,606
HDI - Longevity	0.6720	0.8320	0.7547	0.0285
Extreme poverty rate	3.1%	48.4%	23.0%	8.8%
Gini index	0.400	0.790	0.522	0.047
Under five mortality rate	12.9	50.5	28.8	5.9



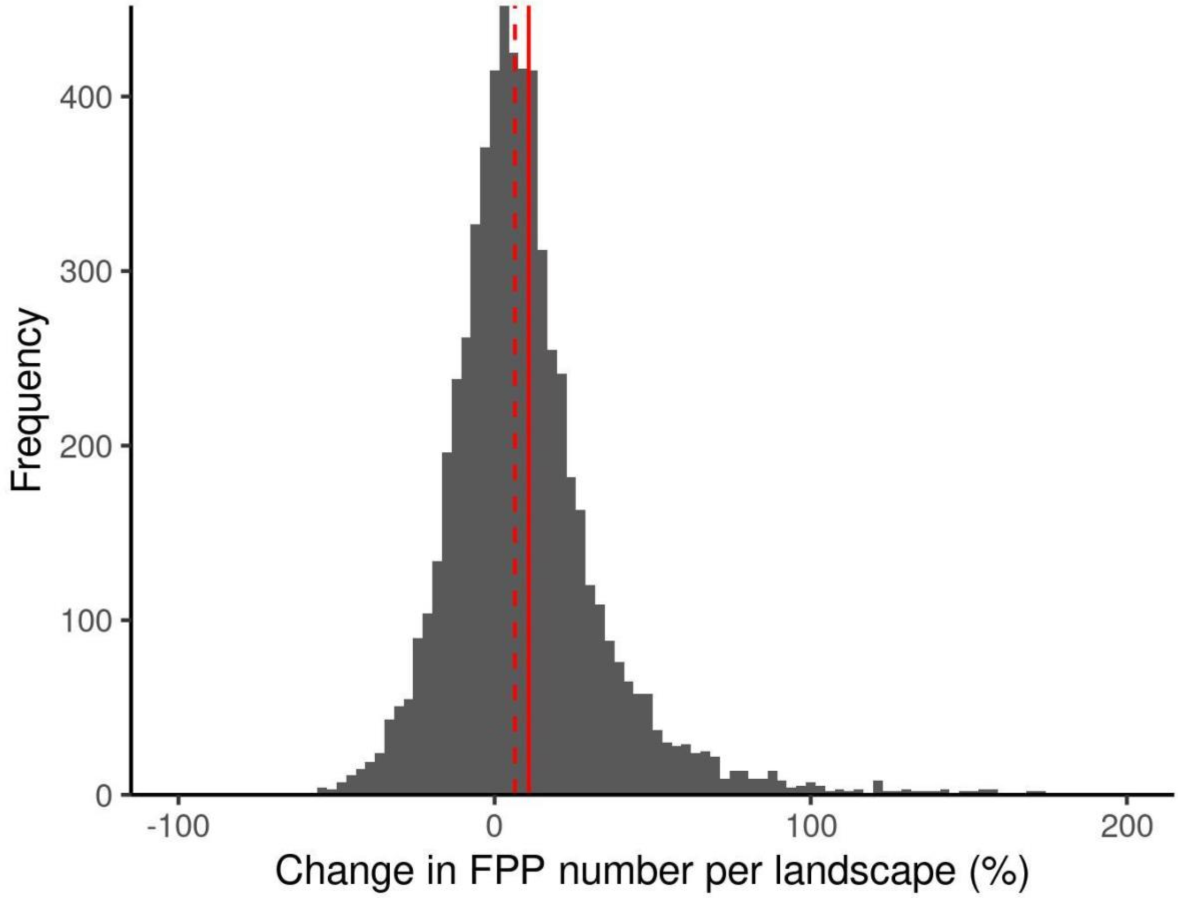
Supplementary Figure 1. Changes in development indicators by group of forest:population change. First term indicate if the municipality experienced regeneration (gain) or deforestation (lose) and the second term if the municipality had population increase (gain) or depopulation (lose).

Supplementary Table 3. Spatial autocorrelation test (Global Moran's I for regression residuals) of GLMs models of forest cover change and forest-proximate people change.

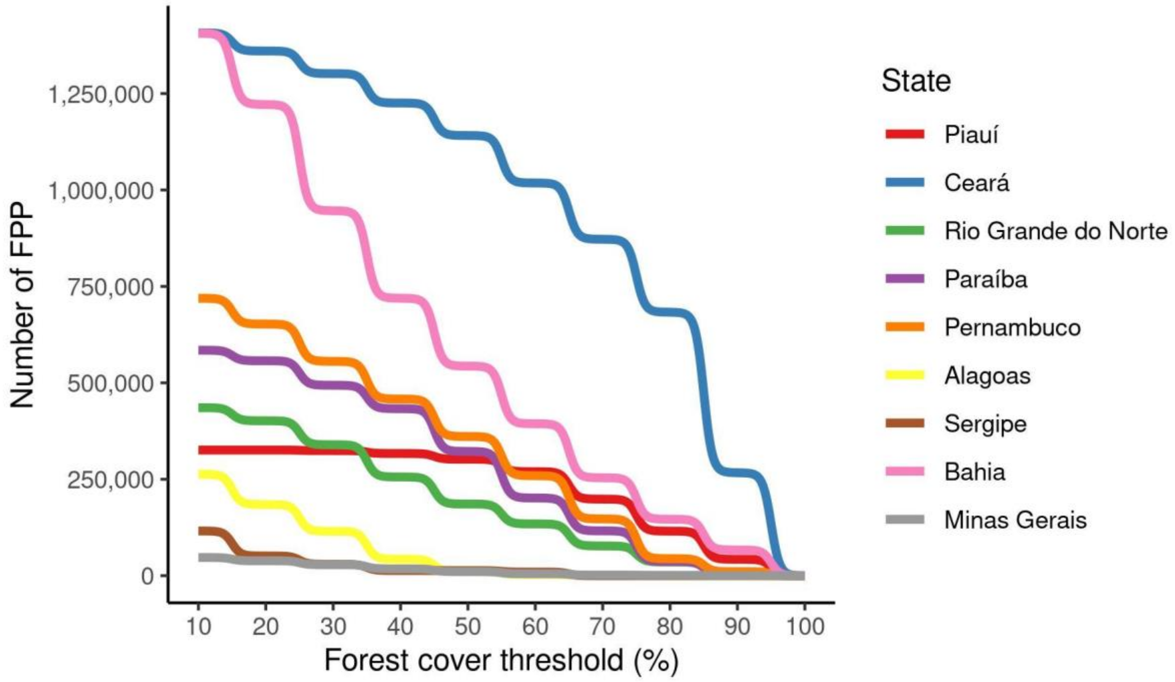
Model	**Moran’s I**	***p*-value**
Forest cover	0.6110	0.00
Forest-proximate people	0.1894	0.00

Supplementary Table 4. Number of forest-proximate people (FPP) in landscapes with different minimum thresholds of vegetation cover.

Threshold (at least %)	Population in 2006	Population in 2017	Absolute change	Mean (SD) change per landscape	n of landscapes (2017)	Mean (SD) forest cover in % for 2017
10	4,910,432	5,304,623	394,191	65.1 (213.6)	6053	64.9 (24.6)
20	4,435,751	4,793,525	357,773	62.2 (207.3)	5748	67.5 (22.4)
30	3,831,309	4,134,340	303,031	56.9 (192.4)	5329	70.9 (19.67)
40	3,231,205	3,482,737	251,531	51.9 (173.5)	4850	74.4 (16.9)
50	2,676,268	2,889,928	213,660	49.3 (165.7)	4337	77.9 (14.3)
60	2,118,434	2,294,548	176,113	47.6 (154.7)	3697	81.9 (11.4)
70	1,537,424	1,667,838	130,414	43.9 (147.7)	2967	86.0 (8.5)
80	971,847	1,063,042	91,193	42.8 (147.7)	2128	90.3 (5.8)
90	359,132	398,462	39,329	35.9 (124.3)	1094	95.2 (3.1)
100	83	77	-6	-0.2 (3.3)	29	100 (0.0)



Supplementary Figure 2. Histogram of the change in percentage of forest-proximate people number from 2006 to 2017 in landscapes with more than 20% of forest cover (n=5478). Red solid line is the mean change and the dashed red line is the median.



Supplementary Figure 3. Number of forest-proximate people (FPP) in 2017 recalculated for the whole Brazilian state constituting the Caatinga Biome based on sampled landscapes. The number of FPP per state are at different minimal thresholds of forest cover within sampled landscapes.

Supplementary Table 5. Size of forest-proximate population per Brazilian state using the sampled landscapes with different levels of forested landscapes, from 20% minimum vegetation cover (least conservative), through 50%, and 70% (most conservative). This table also shows the rank of each state in terms of forest-proximate population size. Note that the rank changes according to the threshold used.

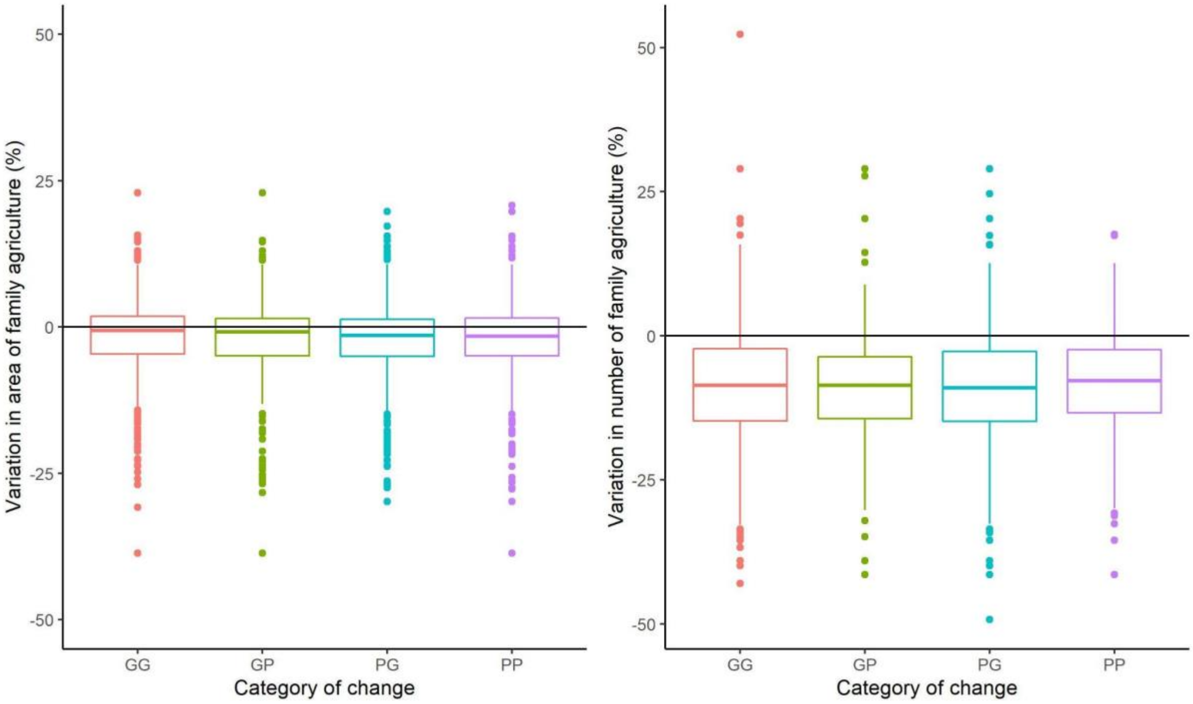
State	FPP Population (20% forest cover)	Rank 20%	Population 50%	Rank 50%	Population 70%	Rank 70%
Ceará	1,360,363	1	1,141,260	1	872,237	1
Bahia	1,221,627	2	543,327	2	253,837	2
Pernambuco	652,417	3	360,868	3	147,123	4
Paraíba	557,232	4	322,126	4	116,241	5
Rio Grande do Norte	401,637	5	185,760	6	77,105	6
Piauí	325,781	6	302,060	5	198,498	3
Alagoas	184,515	7	10,687	8	947	8
Sergipe	51,140	8	13,179	7	0	9
Minas Gerais	38,663	9	10,510	9	1,698	7
Total	4,793,375	-	2,889,777	-	1,667,686	

Supplementary Table 6. Global Moran's I for spatial autocorrelation in GLM regression residuals in a model of forest cover change and forest-proximate people change.

Independent variable	Dependent variable	Observed Moran's I	*p*-value
Change in forest-proximate people population size	Change in forest cover	0.6174	< 2.2e-16

Supplementary Table 7. Variation of forest cover and FPP at landscape level. We consider for this estimation only landscapes with more than 20% of forest cover, resulting in 5748 landscapes. In column “Group of change” the first term refers to forest cover and the second to forest-proximate people.

Type of transition	Number of municipalities	Number of landscapes	FPP in 2017	Mean FPP change (%)	Mean forest cover in 2017 (%)	Mean forest cover change (%)
Reforestation with population growth	393	1894	1,998,418	23.9	61.74	5.7
Reforestation with depopulation	160	1138	579,736	-10.5	57.31	6.6
Deforestation with population growth	353	1808	1,848,515	22.8	54.10	-5.3
Deforestation with depopulation	111	888	315,183	-11.9	51.61	-5.8
Total	1017	5748	4,741,852	-	-	-



Supplementary Figure 5. Variation in area and number of family agriculture in the Caatinga dry forest between 2006 and 2017 separated by category of change at municipal scale (n = 1017). GG refers to the category of municipalities that gained forest and rural population (n = 393); GP is gain forest:lose people (n = 160); PG is lose forest:gain people (n = 353) and PP is lose forest:lose people (n = 111).