Universidade de Brasília

Instituto de Ciências Biológicas

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Dinâmica Sucessional de Áreas em Restauração por Semeadura Direta no Cerrado - desafios e lições aprendidas após 12 anos de pesquisas

Ana Wiederhecker Gabriel

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Dissertação de Mestrado

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Resumo

Comunidades vegetais são naturalmente dinâmicas e suas trajetórias frequentemente imprevisíveis. Por isso, monitoramento e manejo a longo prazo são essenciais para garantir que projetos de restauração sigam uma trajetória desejada, especialmente em ecossistemas que estão presos em estados alternativos. Esse é, frequentemente, o caso de savanas neotropicais degradadas e dominadas por gramíneas exóticas invasoras (IEG, na sigla em inglês). O desenvolvimento de métodos e informações sobre a restauração de ecossistemas de dossel aberto é urgente para a inclusão desses ecossistemas nas prioridades globais e regionais. No Cerrado, a savana mais biodiversa do mundo, os esforços de restauração focados em ecossistemas abertos são raros, tendo aumentado nos últimos 10 anos. Nesse estudo, coletamos dados da comunidade de 22 áreas de restauração no Brasil central e em três áreas nativas de referência em 2022 e 2024. Além disso, compilamos dados de cobertura vegetal, densidade e abundância de indivíduos de espécies lenhosas dos últimos doze anos para avaliar a trajetória das áreas. Observamos que os métodos atuais de restauração são eficazes para o estabelecimento de espécies nativas em uma comunidade que segue uma trajetória desejada, com substituição de espécies de crescimento rápido para lento, nos anos subsequentes à restauração. A semeadura direta foi capaz de estabelecer tanto diversidade de espécies quanto cobertura por gramíneas nativas próximas às dos locais de referência nativos, já nos primeiros anos. O sucesso da restauração variou e foi influenciado pelas condições do solo, pela cobertura da paisagem por IEG e pelo manejo das gramíneas invasoras pós-semeadura. As trajetórias foram prejudicadas pelas IEGs, que recuperaram sua dominância em todos os locais, exceto naqueles com baixa fertilidade do solo e alta acidez ou nas áreas com controle químico das IEGs. Por fim, incêndios tardios não pareceram prejudicar as trajetórias de restauração nem favorecer a dominância das IEGs.

Palavras-chave: Cerrado, Trajetórias de restauração, Gramíneas exóticas invasoras, Savana Neotropical, Restauração de Savana, Turnover de espécies, Semeadura direta e Mix de sementes.

Abstract

Plant community trajectories are naturally dynamic and often unpredictable. Therefore, restoration projects often need long-term monitoring and management to guarantee that the ecosystem is following the desired trajectory, especially for ecosystems that are stuck in alternative stable states. This is often the case for Neotropical Savannahs dominated by invasive exotic grasses (IEG). The development of methods and information on open-canopy ecosystem restoration is, therefore, urgent for the inclusion of these ecosystems into global and regional priorities. In the Brazilian savanna, the most biodiverse savanna in the world, restoration efforts focused on open ecosystems have been virtually absent, but have increased in the last 10 years. In the present study, we collected data for the community trajectory of 22 restoration sites in central Brazil and three native reference sites. To analyse the trajectories of the large-scale experimental restoration sites, we compiled sampled data of vegetation cover and woody density and abundance from the past twelve years. We show that current restoration methods are successful in establishing native species in a community that follows a desired trajectory with fast to slow-growing species turnover in the years following restoration. Additionally, direct seeding was able to establish native species diversity and native grass cover close to those of native reference sites, in early years. Restoration success varied and was affected by soil conditions, IEG landscape cover and post-sowing weeding. Trajectories were hindered by IEG, which quickly recovered dominance in all sites, except for sites with low soil fertility and high acidity or with IEG chemical weeding. Lastly, late fires did not seem to hinder restoration trajectories or aid IEG dominance.

Keywords: Cerrado, Restoration trajectories, Invasive exotic grasses, Neotropical savannas, Savanna restoration, Species turnover and Direct seeding.

Lista de Figuras

Capítulo 1:

Capítulo 2:

Lista de Tabelas

Capítulo 1:

Capítulo 2:

Apêndice B - Supplementary Material Capítulo 2:

Lista de Abreviaturas e Siglas

AG	Gramínea Anual
C/CVNP	Parque Nacional Chapada dos Veadeiros
D/LDPA	Reserva Biológica do Lago Descoberto
E/ERPF	Fazenda Entre Rios
EMMS	Média Marginal Estimada
FGPG	Gramínea perene de crescimento rápido
FGST	Arbusto/árvore de crescimento rápido
GLM	Modelo Linear Generalizado
IEG	Gramínea exótica invasora
NA/NC	Não Classificado
NMDS	Escalonamento Multidimensional Não Métrico
RN	Regenerante Nativa
S	Semeada
SGPG	Gramínea perene de crescimento lento
SGST	Arbusto/árvore de crescimento lento
SLSS	Subarbusto de vida curta

Sumário

Introdução Geral

A conversão e degradação de áreas naturais pela ação humana tem gerado uma grande quantidade de ecossistemas alterados ao redor do globo (HOBBS et al., 2006; ELLIS et al., 2010). Como resultado, tem-se uma crescente fragmentação e perda de habitat para inúmeras espécies, o que ameaça diretamente a conservação da biodiversidade (YUAN; ZHANG; ZHANG, 2024; VIEIRA-ALENCAR et al., 2023; GRANDE; AGUIAR; MACHADO, 2020; MAXWELL et al., 2016). Além disso, a conversão e degradação de ambientes naturais representam um risco à subsistência humana, na medida em que afetam a capacidade dos ecossistemas de fornecer certos serviços (JUNG et al., 2021; MAXWELL et al., 2020). A segurança hídrica, o sequestro de carbono, a oferta de alimentos e a manutenção dos modos de vida de comunidades tradicionais são alguns desses serviços (OVERBECK et al., 2015; BARDGETT et al., 2021). Nesse cenário, intervenções diretas, como a restauração ecológica, podem ser necessárias para recuperar a biodiversidade e os serviços ecossistêmicos dos ambientes alterados. Em décadas recentes a restauração ecológica vem sendo cada vez mais implementada como uma ferramenta para recuperação dos ecossistemas, mitigando os impactos da degradação e da conversão (HOBBS et al., 2011, CARLUCCI et al., 2020, BARROS et al., 2023).

A restauração ecológica pode ser definida como o processo de auxiliar a trajetória de uma comunidade em direção a um estágio autossustentável e semelhante em estrutura e função à comunidade original ou histórica (ecossistema de referência; HOBBS; NORTON, 1996; GANN et al., 2019). Apesar disso, retornar ao ecossistema de referência nem sempre é possível ou desejável. Isso porque a informação da composição original da área pode ter sido perdida ou o estado histórico pode não ser mais viável, devido à alterações na paisagem ou até mesmo às mudanças climáticas globais. Frente a esses obstáculos, objetivos e resultados diferentes do ecossistema de referência foram sendo incluídos e aceitos como restauração ecológica nas últimas décadas (CARLUCCI et al., 2020; HOBBS et al., 2011). Atualmente, entende-se como restauração a busca de um estado mais desejável, seja ele similar ao estado de referência ou não. Recuperar serviços ecossistêmicos importantes, recuperar cobertura vegetal (mesmo que com espécies diferentes), reverter a degradação ou dar um novo uso à área; todos são exemplos de processos de restauração (PRACH et al., 2019; GANN et al., 2019). Ainda assim, em alguns sistemas e ambientes, recuperar o estado histórico é importante. Esse é o caso da restauração em áreas de preservação, que possuem como objetivo intrínseco preservar ecossistemas naturais. Nesses locais a restauração visa a recuperação da composição e da estrutura de um ecossistema o mais próximo possível do ecossistema de referência, garantindo conservação das espécies locais (BALAGUER et al., 2014; WIENS; HOBBS, 2015; JACKSON; HOBBS, 2009).

Restaurar uma comunidade envolve intervenção direta com intensidades variadas. Em condições com alto potencial de regeneração natural, a interrupção do fator de degradação ou conversão basta para recolocar a área em uma trajetória rumo ao estado de referência (CHAZDON et al., 2021). Alternativamente, o retorno de distúrbios endógenos, nativos e característicos do ecossistema, pode ser suficiente para impulsionar o sistema de volta ao estado de referência. Isso geralmente é possível em sistemas que foram degradados pela ausência de tal distúrbio, onde recuperar o regime natural de distúrbio é uma parte essencial dos projetos de restauração (BUISSON et al., 2019; BUISSON et al., 2021; CHAZDON et al., 2021). Isso porque, se o objetivo da restauração é uma comunidade auto sustentável, ela deve ser capaz de resistir aos distúrbios naturais do ambiente em que se encontra (PALMER; AMBROSE; POFF, 1997; GANN et al., 2019). Quando isso não é suficiente, e o potencial de regeneração natural do ecossistema foi perdido, a reintrodução ativa de espécies pode ser necessária para iniciar a trajetória de restauração da área (CHAZDON et al., 2021; MAYER; RIETKERK, 2004; SUDING; GROSS; HOUSEMAN, 2004; HOLL & AIDE 2011).

A intervenção inicial, contudo, nem sempre é suficiente para que o sistema retorne ao estado de referência buscado. Após a interrupção ou recuperação do distúrbio ou até mesmo a reintrodução de espécies, a comunidade em restauração pode seguir a trajetória desejada rumo ao estado de referência mas também pode retornar ao seu estado anterior ou até mesmo atingir a estabilidade em um estado estável alternativo (MAYER; RIETKERK, 2004; SUDING; GROSS; HOUSEMAN, 2004). Isso porque, a alteração de fatores determinantes do equilíbrio de uma comunidade como diversidade funcional, efeitos de prioridade, presença de espécies exóticas invasoras, mudanças nas condições bióticas e abióticas e manejo humano, podem dificultar o retorno ao estágio anterior ao distúrbio (FUKAMI et al., 2005; STANDISH et al., 2014). Tais alterações, frequentemente associadas a atividades de degradação e conversão, podem fazer com que uma comunidade ultrapasse os seus limites naturais de equilíbrio e estabilidade que a mantinham no estágio histórico. A superação desses limites leva a comunidade a se estabilizar em um estágio estável alternativo diferente do original (SUDING; GROSS; HOUSEMAN, 2004;

HOBBS et al., 2006; STANDISH et al., 2014). Nesse cenário, intervenções futuras podem tanto levar o ecossistema a um outro estágio alternativo, diferente do original, como ser insuficiente para retirá-lo do estágio degradado. Isso resulta em trajetórias de restauração não lineares e imprevisíveis, o que torna o entendimento das trajetórias um dos principais desafios para a ecologia da restauração contemporânea (SUDING, 2011; BRUDVIG, 2017).

Outro obstáculo enfrentado pelos projetos de restauração são as espécies exóticas invasoras. Espécies exóticas são uma das principais causas de perda de biodiversidade global (BELLARD et al., 2016; MAXWELL et al., 2016). Além disso, sua presença frequentemente empurra ecossistemas naturais para além de seus limites históricos e em direção a estados estáveis alternativos (BROOKS; SETTERFIELD; DOUGLAS, 2010; SUDING, 2011). Como resultado, ecossistemas invadidos são frequentemente resilientes e resistentes a inúmeros distúrbios, endógenos e exógenos, e tendem a retornar ao estado invadido mesmo após ações de restauração (D'ANTONIO; MEYERSON, 2002; HOBBS et al., 2006; STANDISH et al., 2014). Soma-se a isso o fato de que as espécies exóticas são uma das principais causas de degradação dos ecossistemas. Assim, projetos de restauração, frequentemente, precisam controlar espécies exóticas, além de interromper os distúrbios de degradação e reintroduzir as espécies nativas (WEIDLICH et al., 2019; KETTENRING; ADAMS, 2011; D'ANTONIO; MEYERSON, 2002). Alta resiliência, capacidade de reinvasão, alterações do solo e presença de propágulos em áreas convertidas vizinhas são todos fatores que dificultam o controle permanente das espécies exóticas invasoras (GIORIA et al., 2023). Dessa forma, o controle dessas espécies ainda é um grande obstáculo para a restauração das áreas invadidas (KETTENRING; ADAMS, 2011).

Além das incertezas nas trajetórias e das espécies exóticas invasoras, a ausência de monitoramento e manejo, de projetos a longo prazo, de conhecimento sobre espécies e ambientes nativos e de governança apropriada dificultam o sucesso dos projetos de restauração (SUDING, 2011 BRUDVIG, 2017; PALMA; LAURANCE, 2015; MEDEIROS, 2024; SCHMIDT et al., 2019, MONTENEGRO; URZEDO; SCHMIDT, 2024). Frente às incertezas nas trajetórias e a necessidade de controle de espécies exóticas, o monitoramento e o manejo se tornam partes essenciais do processo de restauração. Apesar disso, poucos projetos monitoram áreas de restauração por mais de dois ou três anos. De forma similar, as trajetórias das comunidades em restauração são frequentemente assumidas a partir de um ponto intermediário, sem o devido conhecimento de seus fatores determinantes (PALMER; AMBROSE; POFF, 1997; SUDING, 2011). Inúmeros projetos não relatam seus resultados ou os presumem, já que os projetos demandam muito tempo para sua conclusão, e o financiamento é muitas vezes escasso, impossibilitando o acompanhamento a longo prazo (SUDING, 2011; BRUDVIG, 2017). Somam-se a isso a falta de conhecimento de espécies e áreas nativas e entraves legais para a realização de projetos de restauração e para a comercialização de sementes encontrados em alguns países (SCHMIDT et al., 2019, MONTENEGRO; URZEDO; SCHMIDT, 2024).

Em ecossistemas abertos esse quadro se agrava. Historicamente negligenciados, e tratados como florestas degradadas, esses ecossistemas estão entre os mais ameaçados do mundo (TÖLGYESI, 2022). Quase 50% dos campos e savanas tropicais e subtropicais em todo o mundo já foram convertidos (ELLIS et al., 2010; BARDGETT et al., 2021; STEVENS et al., 2022), alguns em taxas que superaram o desmatamento de florestas tropicais (a exemplo da savana brasileira, OVERBECK et al., 2015). Os principais motores da conversão desses ambientes são a expansão de pastagens, agricultura e silvicultura, muitas vezes vista como positiva por aumentar cobertura de dossel e/ou sequestro de carbono nestas áreas (ELLIS et al., 2010; NERLEKAR; VELDMAN, 2020). A conservação e restauração de savanas e campos é essencial, não apenas porque são ecossistemas altamente diversos com altas taxas de endemismo, mas também porque oferecem serviços ecossistêmicos importantes (MURPHY; ANDERSEN; PARR, 2016). Ecossistemas graminoides, de dossel aberto, também chamados de *old-growth grasslands* ou *old-growth savanna*s, são essenciais para o abastecimento de água, armazenamento de carbono subterrâneo, bem como para a alimentação e pastejo animal (VELDMAN et al., 2015; STEVENS et al., 2022; OVERBECK et al., 2015; BARDGETT et al., 2021). Apesar de sua importância, os campos e savanas tropicais têm sido historicamente negligenciados (PARR et al., 2014; CARLUCCI et al., 2020). Esses ecossistemas ainda estão atrasados em relação às florestas em termos de conhecimento científico e geral, políticas e investimentos (DUDLEY et al., 2020; SILVEIRA et al., 2022).

Na savana neotropical, a savana mais biodiversa do mundo (MURPHY; ANDERSEN; PARR, 2016), a restauração ecológica é incipiente (SAMPAIO et al., 2019; SILVEIRA et al., 2022) e frequentemente enviesada para o plantio de árvores, desconsiderando a camada herbácea (STEVENS et al., 2022; VELDMAN et al., 2015; GUERRA et al., 2020). Isso é um problema considerando que os ambientes savânicos são caracterizados pela coexistência de árvores, e gramíneas e restaurá-los requer, portanto, a recuperação de todos esses grupos (VELDMAN et al., 2015; SILVEIRA et al., 2022). Além disso, essa savana vem sendo fortemente ameaçada pela conversão de áreas, especialmente para a pecuária e agricultura industriais. A maioria das áreas selecionadas ou disponíveis para restauração na região são pastagens abandonadas (BARROS et al., 2023), previamente cultivadas com gramíneas exóticas que mantêm os ecossistemas em um estado degradado estável. Estas espécies de gramíneas C4 de origem africana, foram introduzidas no Brasil desde o princípio da colonização europeia (PARSONS, 1972), são amplamente utilizadas para o estabelecimento de pastagens e têm capacidade de invadir ambientes alterados (como de agricultura abandonada) e ecossistemas nativos (SAMPAIO; SCHMIDT, 2013; PIVELLO; SHIDA; MEIRELLES 1999). Além disso, muitas delas foram modificadas e selecionadas para sua introdução em pastagens no país (KARIA; DUARTE; ARAUJO, 2006).

A grande capacidade de invasão e permanência de gramíneas exóticas ocorre devido ao seu rápido crescimento e reprodução, associados à sua alta capacidade de rebrota após fogo ou pastejo (CARAMASCHI et al., 2016; GORGONE-BARBOSA et al., 2020). Uma vez estabelecidas, as gramíneas exóticas não apenas resistem ao regime de fogo natural das savanas, mas também o alteram em seu benefício (GORGONE-BARBOSA et al., 2015). Essas espécies conseguem excluir espécies nativas por meio da competição, reduzindo a diversidade das áreas invadidas (HOFFMANN; HARIDASAN, 2008). Além disso, as condições do solo em áreas de restauração frequentemente favorecem IEG devido à fertilização e à calagem realizadas para práticas agrícolas, enquanto a ausência de estruturas vegetativas de reprodução (i.e rizomas, xilopódios e raízes tuberosas; PILON et al., 2021) limita a regeneração de espécies nativas (SAMPAIO et al., 2019; PILON et al., 2023). Portanto, controlar IEG enquanto se mantém a cobertura típica de áreas abertas, não só é essencial, como também representa um dos maiores desafios para a restauração de ecossistemas campestres e savânicos (COUTINHO et al., 2019; SAMPAIO et al., 2019)

Os esforços de pesquisa sobre práticas de restauração na savana neotropical são recentes, especialmente em comparação com os esforços de restauração florestal em grande escala, que remontam ao século XIX (FREITAS; NEVES; CHERNICHARO, 2006). As áreas mais antigas de savana restauradas em escala operacional (>1ha) com introdução de capins remontam a 12 anos atrás, tendo sido implementadas em 2012 (SAMPAIO et al., 2019). Durante a última década, diferentes técnicas e estratégias foram desenvolvidas e testadas para restauração em larga escala de savana usando plantas gramíneas, arbustivas e arbóreas (FERREIRA; WALTER; VIEIRA, 2015; PALMA; LAURANCE, 2015; PILON; BUISSON; DURIGAN, 2018). Dentre essas técnicas, a semeadura direta vêm apresentando o melhor custo-benefício (RAUPP et al., 2020; FERREIRA et al., 2023), sendo viável para estabelecer diferentes formas de vida vegetal em escala operacional com tecnologias prontas para demandas reais (STRAUB, 2015; GIBBS, 2021). Isso se deve principalmente ao seu potencial de mecanização e ao baixo custo das sementes (CAMPOS-FILHO et al., 2013; SAMPAIO et al., 2019; RAUPP et al., 2020; FERREIRA et al., 2023; FERREIRA; VIEIRA 2024). A semeadura direta também promove o engajamento da comunidade local, criando cadeias de restauração economicamente sustentáveis (SCHMIDT et al., 2019, MONTENEGRO; URZEDO; SCHMIDT, 2024).

Em termos de governança, muitos desafios ainda devem ser superados para auxiliar a restauração de savanas. A inclusão da restauração como obrigação legal só foi formalizada em 2012 com a Lei de proteção da vegetação nativa (BRASIL; 2012 - L12651/2012) e ainda carece muito do requerimento de fato por órgãos ambientais competentes. Além disto, não existem determinações legais de indicadores para a restauração em escala nacional (IBAMA; 2024 - IN 14/2024). Atualmente, somente 9 estados possuem normativas que estabelecem valores mínimos de recuperação de áreas degradadas (GILES, não publicado). Além disso, a inclusão de outras formas de vida para a restauração e o reconhecimento, pela legislação, de que biomas abertos devem ser restaurados com diferentes objetivos dos biomas florestais, foi feita pela primeira vez pelo estado de São Paulo em 2014 (CHAVES et al., 2015; MONTENEGRO; URZEDO; SCHMIDT, 2024), seguido por outros estados, o Distrito Federal em 2017 e Goiás em 2024 (i.g. VIEIRA et al., 2017; SEMA-GO, 2024)

Soma-se a isso uma descrença, desconhecimento e resistência popular no uso de sementes nativas para a restauração devido ao uso frequente de mudas (PARR et al., 2014; SILVEIRA et al., 2020; SILVEIRA et al., 2022). Como resultado, projetos de restauração de savanas continuam sendo feitos única e exclusivamente com mudas de espécies arbóreas, frequentemente típicas de ambientes florestais (VELDMAN et al., 2015; SCMIDT et al., 2019; SILVEIRA et al., 2022). Esse problema se agrava quando consideramos que a restauração ecológica vem sendo reconhecida como um processo inclusivo em várias regiões do globo (i.g. Sigman e Elias 2021). Apesar disso, existem limitações legais vigentes para a comercialização de sementes nativas. Atualmente, a legislação que regula a venda de sementes é a mesma para agrícolas e nativas, o que impõe padrões inalcançáveis aos lotes de sementes nativas comercializados para a restauração. A alta rigidez demandada para a qualidade e homogeneidade de sementes dificulta a cadeia produtiva de sementes para a restauração que é atualmente liderada por grupos de base comunitária que coletam e vendem sementes nativas como forma de complementação da renda familiar (SCHMIDT et al., 2019, MONTENEGRO; URZEDO; SCHMIDT, 2024). Atualmente no Brasil existem mais de 20 redes de sementes de base comunitária atuando na oferta de sementes para restauração (REDÁRIO, 2025). A existência dessas redes é essencial para garantir o fornecimento em larga escala das sementes para projetos nacionais de restauração (SCHMIDT et al., 2019, MONTENEGRO; URZEDO; SCHMIDT, 2024). A articulações de todas estas redes produtoras de sementes nativas tem sido importante na busca pela valorização da inclusão na cadeia da restauração, mudanças nas legislações pertinentes em todas as etapas da cadeia produtiva da restauração (GOMES et al., 2024; REDÁRIO, 2025).

A restauração da savana neotropical ainda enfrenta muitos desafios. De maneira geral, há uma falta de conhecimento sobre as espécies nativas e suas formas de propagação, o que dificulta sua inclusão em projetos de restauração. Soma-se a isso a ausência de estudos a longo prazo e da compreensão sobre os efeitos de diferentes variáveis ambientais nos resultados dos projetos de restauração (SUDING, 2011; PALMA; LAURANCE, 2015; BRUDVIG, 2017). Como a restauração de savanas e campos é, em grande parte, recente, as trajetórias de restauração e os fatores determinantes na montagem das comunidades ainda são pouco conhecidos (PILON et al., 2021). Compreender esses processos é imperativo, especialmente no Cerrado brasileiro, onde mais de seis milhões de hectares devem ser restaurados em áreas protegidas públicas e em terras privadas para cumprir as leis nacionais (BRASIL, 2017; GUIDOTTI et al., 2017).

Objetivo

Este estudo resume os resultados de projetos de restauração em escala operacional em savanas neotropicais no Brasil Central ao longo de 12 anos. Os locais amostrados constituem as áreas de savana conhecidas mais antigas e extensas em processo de restauração por meio de semeadura direta no Brasil. Coletamos dados sobre a trajetória da comunidade de várias áreas de restauração e avaliamos se esses locais poderiam restabelecer comunidades vegetais nativas com resiliência estrutural e funcional, prevenindo a reinvasão por espécies exóticas e fornecendo habitat para o recrutamento natural de espécies nativas. Nosso principal objetivo foi compreender as trajetórias das comunidades das áreas de restauração de savanas neotropicais e como elas responderam a diferentes condições de manejo.

Hipóteses

As hipóteses levantadas para este trabalho foram:

i) Ocorre uma substituição de dominância entre espécies semeadas de crescimento rápido e espécies semeadas de crescimento lento;

ii) Dominantes nos primeiros anos, as espécies de crescimento rápido gradualmente deixam o sistema ou reduzem sua cobertura, sendo substituídas por espécies de crescimento lento;

iii) As áreas de restauração alcançam a cobertura de gramíneas e árvores das áreas nativas em poucos anos quando as gramíneas exóticas invasoras (IEG, na sigla em inglês) são manejadas;

iv) A composição de espécies é influenciada pelas condições do solo, pelo manejo pós-semeadura e pelo tempo desde a semeadura, com solos mais pobres em nutrientes e manejo das IEG pós-semeadura, favorecendo as espécies nativas;

v) A presença de IEG na paisagem aumenta sua ocorrência e dominância nos locais de restauração;

vi) O controle inicial de gramíneas exóticas é suficiente para controlar suas populações de forma permanente;

vii) A ocorrência de incêndios tardios (naturais ou antrópicos) aumenta a dominância de IEG nas áreas em restauração invadidas.

Capítulo 1

Ten Years of Directing Seeding Restoration in the Brazilian Savanna: Lessons Learned and The Way Forward

1. Introduction

Savannas are among the most threatened biomes in the world, with extensive degraded areas in need of restoration. Almost 50% of worldwide tropical and subtropical grasslands, savannas and shrublands have been degraded (Bardgett et al., 2021; Stevens et al., 2022), some at rates that recently exceeded that of tropical forest clearing (eg. Brazilian savanna, Overbeck et al., 2015). The main drivers of savanna conversion are pasture expansion, agriculture, and silviculture (Bardgett et al., 2021; Nerlekar and Veldman, 2020). Conservation and restoration of Neotropical Savanna are essential not only because they are highly diverse ecosystems with high endemism rates, but also because they offer important ecosystem services (Murphy et al., 2016). Grassy ecosystems are especially important for water supply, underground carbon storage as well as food and animal forage (Bardgett et al., 2021; Overbeck et al., 2015). Despite their importance, tropical grasslands and savannas have been historically overlooked (Parr et al., 2014). These ecosystems still fall behind forest ecosystems on knowledge, policies and investment (Bardgett et al., 2021; Dudley et al., 2020; Silveira et al., 2022).

Research on restoration practices and efforts in the Neotropical Savanna is recent, especially compared to large-scale forest restoration efforts, which date back to the 19th century (Freitas et al., 2006). In the Brazilian Cerrado, the most diverse savanna in the world (Murphy et al., 2016), ecological restoration is incipient (Sampaio et al., 2019; Silveira et al., 2022) and frequently biased toward tree plantation, disregarding the grassy layer (Guerra et al., 2020; Veldman et al., 2015). During the last decade, different techniques and strategies have been tested for large-scale restoration of the Brazilian savanna using graminous and shrubby plants along with trees (e.g. Ferreira et al., 2015; Pilon et al., 2018; Silva et al., 2023). Direct seeding presents the most cost-benefit method (Raupp et al., 2020), available for establishing different plant life forms (Pellizzaro et al., 2017), at an operational scale, i.e. using technology and inputs that are ready to be applied to real-world demands (considering the definition of 'operational' from the TRL framework: Straub, 2015 eg. Gibbs, 2021). This is mainly due to its potential for mechanization and low seed cost (Campos-Filho et al., 2013; Sampaio et al., 2019; Raupp et al. 2020). Direct seeding also promotes local community engagement, creating economically sustainable restoration chains (Schmidt et al., 2018). Despite the advantages of reintroducing native plants through direct seeding, the restoration of Neotropical Savannas still has many obstacles to overcome.

Invasive exotic grasses (IEG) are among the main causes of degradation in restoration sites and represent the main challenge to restoring open ecosystems (Barros et al., 2023; Coutinho et al., 2019; Silva and Vieira, 2017). These grass species are cultivated globally in pastures and may dominate degraded sites, keeping these systems in alternative stable states (Brooks et al., 2010; Suding, 2011). The invasion process occurs due to high growth and reproductive rates of IEG, associated with their high resprouting ability after fire and/or grazing (Caramaschi et al., 2016; Gorgone-Barbosa et al., 2016). Additionally, soil conditions in restoration sites are often altered due to previous fertilizing and liming for agricultural practices and lack underground vegetative structures (i.e. tuberous roots and rhizomes) due to successive plowing (Sampaio et al., 2019). These altered soil properties, especially higher pH and increased nutrient levels, further favor IEG (re)establishment and dominance, while the absence of underground resprouting structures limits native species regeneration (Brooks et al., 2010; Pilon et al., 2021). Therefore, to successfully restore open ecosystems, it is necessary to control IEG while maintaining open canopy conditions (Coutinho et al., 2019; Sampaio et al., 2019).

Overall, there is a lack of long-term studies and understanding of the effects of different environmental variables on the outcomes of restoration projects (Brudvig, 2017; Palma and Laurance, 2015; Suding, 2011). Since restoration of savannas and grasslands is mostly recent, restoration trajectories and the drivers of community assembly are still unknown (Pilon et al., 2021). Understanding these processes is imperative, especially within the Brazilian savanna where more than 6 Mha should be restored in Public Protected Areas and private lands to comply with national laws (Guidotti et al., 2017; Brasil, 2017).

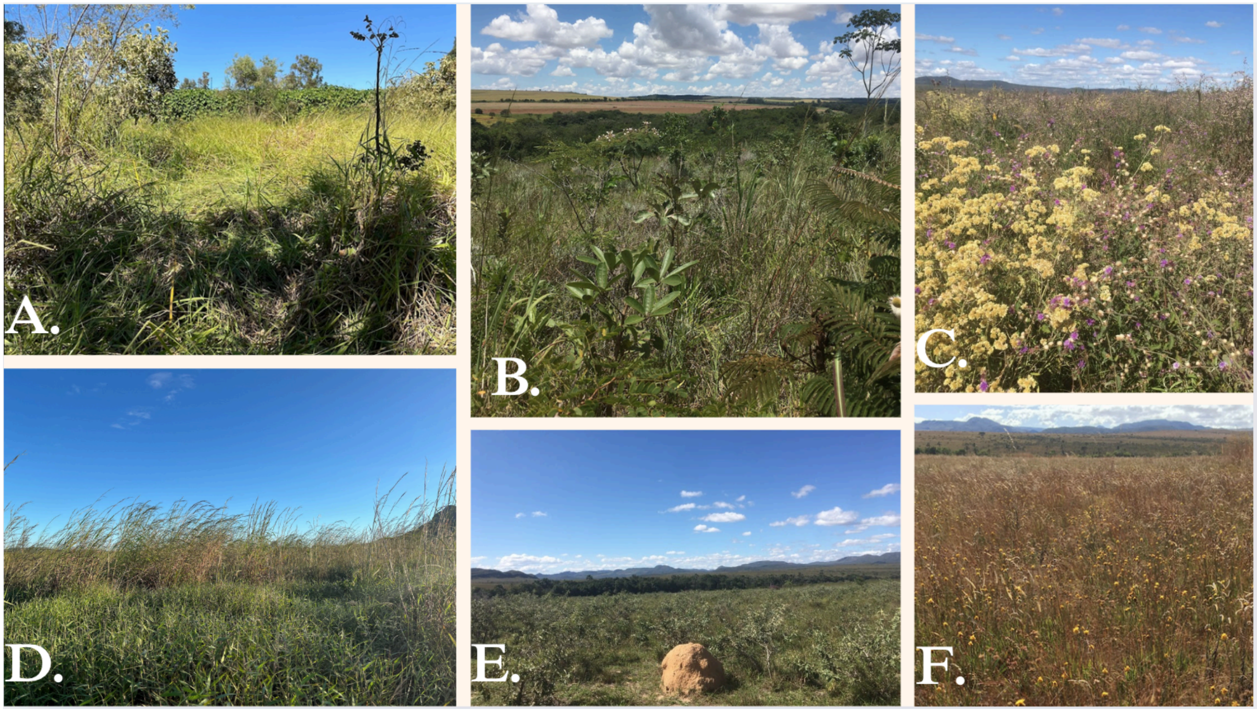
This study summarises the outcomes of operational scale restoration projects in the Neotropical Savanna over a decade. To our knowledge, the sampled sites constitute the oldest and largest savanna areas under restoration through direct seeding in Brazil. We evaluated if the restoration sites could reestablish native vegetation communities that are structurally resilient to reinvasion by exotic species and provide habitat for native species' natural recruitment. We aimed to answer these questions: i) How does the composition of species and functional groups vary over time since intervention in restoration sites? ii) Which abiotic (soil conditions) and management (soil preparation and IEG management) variables influence species composition over time? iii) How does the presence of IEG in the landscape affect their presence in the restoration sites? We hypothesised that: i) Among sowed species, there is a turnover from fast to slow growth throughout the years, ii) Species composition is affected by soil conditions, post-seeding management and time since seeding, with nutrient-poorer soils, and post-seeding management of IEG favoring native species and iii) The occurrence of IEG in the landscape enhances their presence and dominance in restoration sites.

2. Materials and Methods

2.1. Study Sites

We surveyed vegetation cover and soil conditions of 22 restoration sites in the Brazilian savanna (Fig.1), 82 hectares total, located in Chapada dos Veadeiros National Park (CVNP, GO; lat 14°06'43"S long 47°38'14"W), Lake Descoberto Preservation Area (LDPA, DF; lat 15°46'31"W, long 48°13'12"S) and in Entre Rios Private Farm (ERPF, DF; lat 15°57'30"S long 47°27'26"W). The climate is characterized as Aw (Köppen) and has seasonal rainfall from November to April, usually 1,400 to 1,600 mm^3/year (INMET, 2023). Experimental sites were on flat terrain with elevations varying from 1,030 to 1,240 m above sea level. Soils are predominantly Red and Yellow or Red Latosol (IBGE, 2001). Sites were originally within three native physiognomies and their transition states: an open-canopy Savannah with dominance of grasses and the presence of shrubs and trees (*cerrado sensu-stricto*), a similarly open-canopy Savannah but with trees limited to small hills (*campo de Murundu*) and a wet grassland (Ribeiro and Walter, 1998) All sites had been previously converted into agriculture and then pasture or directly to pasture at least 30 years before restoration projects were implemented (direct communications from owners and neighbours).

Figure 1: Restoration sites in Lake Descoberto Preservation Area (A), Entre Rios Private Farm (B) and Chapada dos Veadeiros National Park (C to F).



After pasture conversion and subsequent abandonment, study sites were covered by IEG, especially *Urochloa decumbens*, *Andropogon gayanus* and *Melinis minutiflora*. Restoration interventions were implemented following a common procedure for the Brazilian savanna open ecosystems, which consists of burning and soil plowing, to reduce IEG dominance, and favor native species introduction (Coutinho et al., 2019). However, sites differed from each other (Table 1) in soil characteristics, IEG control method (chemical weeding, grazing by cattle, manual weeding), introduced species, year of restoration intervention (from 2 to 10 years prior to sampling) and landscape cover (native or exotic dominated). In all study sites, the restoration method was direct seeding of grass, herb, shrub and tree species throughout the entire area. However, in two sites, in addition to direct seeding, tree seedlings were planted in low density, aiming to comply with donor goals (Orli 2018; Wiederhecker-Gabriel et al., 2022). In total, 108 native species were seeded in the restoration sites (79 woody and 29 non-woody).

Most sites had no post-seeding exotic grasses control. Before 2019, the use of herbicide for IEG control was not allowed inside Federal Protected Areas. From this year onwards, new restoration sites started to be managed for IEG control with the application of a glyphosate-based solution (concentration 3%) to prevent reproduction (when tussocks were high enough, fully photosynthetic or up to 40 cm tall). Workers performed this using a backpack pump sprayer and wore appropriate personal protective equipment. On average, herbicide was applied in three to four-month intervals for three years in 2019 restoration sites and two years in 2020 sites. Three sites were affected by fires during the sampled time frame. Firstly, in September 2019, the CVNP_2016_2 site was hit by lightning fire. Secondly, in October 2022, both ERPF sites were burned by an anthropogenic fire. Lastly, the two sites from 2015 have been grazed by low-density cattle since 2019. The effects of fire, although recognized, were not considered as management variables in this study.

Table 1: List and characteristics of the 22 restoration areas in Chapada dos Veadeiros National Park (C), Entre Rios Private Farm (E) and Lake Descoberto Preservation Area (D) sampled between February and May 2022. Numbers in the name column stand for site age and id (eg. 4_1 is 4yold site 1). Cited references are listed for a detailed description of the sites and restoration methods. Soil classification from IBGE (2001).

Name	Site (ha)	Seeding (year)	Coordinates	Soil type	Species introduction	IEG management	References
D4_1	1.7	2018	15 46'31"W, 48 13'12"S	Red-Yellow Latosol	Seedlings and seeding in total area		Wiederhecker-Gabriel et al., 2022
D4_3	0.9	2018	15 46'31"W, 48 13'12"S	Red-Yellow Latosol	Seeding in total area		Wiederhecker-Gabriel et al., 2022
D4_4	1.1	2018	15 46'31"W, 48 13'12"S	Red-Yellow Latosol	Seeding in total area		Wiederhecker-Gabriel et al., 2022
D4_6	2.8	2018	15 46'31"W, 48 13'12"S	Red-Yellow Latosol	Seeding in total area		Wiederhecker-Gabriel et al., 2022
E10_1	1.2	2012	15 57'30"S 47 27'26"W	Red Latosol	Seeding in total area	Pre-seeding chemical control with haloxyfop	Pelizzaro et al. 2017
E10_2	1.2	2012	15 57'30"S 47 27'26"W	Red Latosol	Seeding in total area		Pelizzaro et al., 2017
C10_1	1.18	2012	14 06'43"S 47 38'14"W	Red Latosol	Seeding in lines		Alves 2016
C10_2	1.17	2012	14 06'43"S 47 38'14"W	Red Latosol	Seeding in lines		Alves 2016
C10_3	1.38	2012	14 06'43"S 47 38'14"W	Red Latosol	Seeding in lines		Alves 2016
C9	1.39	2013	14 06'43"S 47 38'14"W	Red-Yellow Latosol	Seeding in total area		Pelizzaro 2016
C8	3.34	2014	14 06'43"S 47 38'14"W	Red-Yellow Latosol	Seedlings and seeding in total area		Orli 2018

C7_1	2.64	2015	14 06'43"S 47 38'14"W	Red-Yellow Latosol	Seeding in total area	Post-seeding cattle	Coutinho et al., 2019
C7_2	5.67	2015	14 06'43"S 47 38'14"W	Red-Yellow Latosol	Seeding in total area	Post-seeding cattle	Coutinho et al., 2019
C6_1	6.74	2016	14 06'43"S 47 38'14"W	Red-Yellow Latosol	Seeding in total area	Post-seeding cattle	Liaffa 2020
C6_2	18.6	2016	14 06'43"S 47 38'14"W	Red-Yellow Latosol	Seeding in total area		Liaffa 2020
C6_3	6.91	2016	14 06'43"S 47 38'14"W	Red-Yellow Latosol	Seeding in total area		Liaffa 2020
C3	5.25	2019	14 06'43"S 47 38'14"W	Red-Yellow Latosol	Seeding in total area	Pre and post-seeding chemical control with glyphosate	Silva-Coelho 2021;
C2_1	6.42	2020	14 06'43"S 47 38'14"W	Red-Yellow Latosol	Seeding in total area	Pre and post-seeding chemical control with glyphosate	Semeia Cerrado (personal communication)
C2_2	3.54	2020	14 06'43"S 47 38'14"W	Red-Yellow Latosol	Seeding in total area	Pre and post-seeding chemical control with glyphosate	Semeia Cerrado (personal communication)
C2_3	4.81	2020	14 06'43"S 47 38'14"W	Red-Yellow Latosol	Seeding in total area	Pre and post-seeding chemical control with glyphosate	Semeia Cerrado (personal communication)
C2_4	3.43	2020	14 06'43"S 47 38'14"W	Red-Yellow Latosol	Seeding in total area	Pre and post-seeding chemical control with glyphosate	Semeia Cerrado (personal communication)
C2_5	1.11	2020	14 06'43"S 47 38'14"W	Red-Yellow Latosol	Seeding in total area	Pre and post-seeding chemical control with glyphosate	Semeia Cerrado (personal communication)

2.2. Data Collection

2.2.1. Vegetation Sampling

At each of the 22 sites, we randomly established five plots of 10m² (20m x 0.5m), with a minimum distance of 20m between plots. Throughout the 20m plot line, we conducted vegetation cover sampling using the line-point intercept method (Herrick et al., 2006). For that, we placed a

2m pin perpendicular to the ground every 50cm on the line and recorded all plants that touched the pin and its vertical projection. We identified touching individuals at the species level, whenever possible, or at the genus or family level. We then calculated the relative cover for each taxa. We measured the density and richness of recruiting woody plants (both fast-growing shrubs/trees and slow-growing shrubs/trees) by counting, measuring and identifying all individuals with heights between 30 cm and 2m within each plot, totalling 50m² per site.

2.2.2. Soil analyses

To characterize physical and chemical soil characteristics, we collected a soil sample (0-10cm) in the center of each plot with a shovel, after removing litter. The samples of the five plots were mixed and one sample per site was sent to a laboratory for chemical and physical analyses. We assessed soil pH, organic matter, available P, K, S, Ca, Mg, Al, Mn, Cu, B, Zn and texture (sand, silt, and clay).

We sampled all sites and collected all data during the rainy season of 2022, from February to May.

2.2.3. Functional groups classification

We classified native species into functional groups (Table 2) considering three functional and life-history traits: growth rate (fast or slow), lifespan (annual; short, few years; or perennial), and growth form (graminoids, shrubs, and trees). We generated seven functional groups: annual grass (AG), Short-lived sub-shrub (SLSS), fast-growing perennial grass (FGPG), slow-growing perennial grass (SGPG), fast-growing shrub/tree (FGST), slow-growing shrub/tree (SGST). These groups represent different recovery strategies and functions in active and passive restoration sites (Coutinho et al., 2019; Fensham et al., 2016; Horstmann et al., 2023; Pellizzaro et al., 2017). We defined the IEG category as encompassing all exotic grass species, which are fast-growing perennial grasses regenerating by seeds and rhizomes.

Table 2: Functional Groups characteristics and importance.

		Growth rate	
		Fast	**Slow**
Growth form / lifespan	**Shrub/Tree**	Enhance diversity and niche in the first years producing shade.	Guarantee diversity and persistence long-term.
	Sub-Shrub	Cover the soil in the first three years, reducing erosion and competing with IEG.	Guarantee diversity and persistence long-term.
	Perennial Grass	Cover the soil in the first three years, reducing erosion.	Guarantee diversity and persistence long-term.
	Annual Grass	Cover the soil especially in the first year, reducing seed dragging and competing with IEG.	Guarantee diversity and persistence long-term.
	Invasive Exotic Grass	Compete and exclude native species.	

2.2.4. Landscape Characterisation

To evaluate the influence of the IEG from the surrounding vegetation on the success of restored sites (i.e. high native species and low IEG cover), we averaged the percentage of exotic grass cover within a 100m radius buffer around each site, between the year of sowing interventions and the year of sampling. We used 100m as our buffer distance since it was large enough for landscape characterization but not too much to the point that species present would stop interfering in restoration areas (i.e. propagation limit). As exotic grass cover changes over the years, their propagule pressure also changes, so we used the average of exotic grass cover because they can invade the sites anytime from the start of the restoration. Within the buffers, we identified six categories of IEG cover: pasture sites (100% IEG), sites without IEG (native conserved savanna, roads, water, and others, 0% IEG), restoration sites in LDPA (50% IEG), in ERPF (21% IEG), in CVNP ≤ 3y-old (18% IEG) and in CVNP > 3y-old (67% IEG). For the IEG cover percentages in the restoration sites, we used the mean values found in the surveys of this study. We also used visual characterization for the estimated presence of IEG in native areas and other sites. For each buffer, we calculated the IEG percentage of cover as:

$$\sum (\%\ \text{category } i \text{ in the buffer}) * (\%\ \text{of IEG in the category } i)$$

2.3. Data Analysis

To analyze species composition across the 22 restoration sites, we built a heatmap with the relative cover of each species in each site (Wickham et al., 2016). We included species with more than 2% cover in at least one site. Sites were organized in clusters according to similarity, based on the Bray-Curtis dissimilarity index and with the complete linkage algorithm (Faith et al., 1987; Legendre and Legendre 1998). We assessed the relationship between species composition, soil variables, landscape IEG cover, time since restoration (age) and management variables with a non-metric multidimensional scaling (NMDS) using the Bray-Curtis distance, followed by the "envfit" function that fits environment factor or vector on the ordination (Oksanen et al., 2019). All soil variables were tested in the function, except those excluded for high correlation (>0.70). We considered as management variables: (i) the number of soil plowings, (ii) use (or not) of controlled fire to reduce IEG before restoration, (iii) use of chemical weeding to reduce IEG and (iv) cattle grazing to control IEG after seeding. We used the groups formed in the cluster analysis to identify sites in the woody species density and richness barplots. We analyzed the data in R version 4.2.2 (R Core Team, 2023) using *vegan* and *stats* packages. We classified the buffer cover through visual image interpretation using Qgis software (QGIS, 2023). We followed the botanical nomenclature from Flora e Funga do Brasil (2020).

3. Results

3.1. Species composition and turnover

We classified restoration sites into five groups based on the cluster analysis of species composition (Fig. 2). For the most part, clusters reflected restoration age. The first cluster consisted of 10 y-old sites with shared dominance between IEG and native grass cover (named Native slow-growing grass + exotic grass; Fig. 2). In this group, the slow-growing perennial grass *Echinolaena inflexa* stands out as the native species with the highest mean cover (29%) followed by the short-lived sub-shrub (i.e. forbs) *Achyrocline satureioides* (11%) and two slow-growing trees *Magonia pubescens* and *Plathymenia reticulata* (2 and 0.9% respectively). The IEG cover in these areas was close to native grass cover values, especially *Urochloa decumbens* (23%) and *Andropogon gayanus* (13%).

The second cluster consists of old to intermediate sites (10 to 7 years since restoration) dominated by the exotic grasses *A. gayanus* and *U. decumbens* (29 and 26%, respectively; named

Andropogon gayanus dominance), and fast-growing native trees and shrubs such as *Tachigali vulgaris* (10%) or *Vernonanthura polyanthes* (3%). The third cluster is composed of 4 to 6 y-old sites, characterized by high cover of the exotic grass *U. decumbens* (44%) and low cover for all native species (≤10%) (named: *Urochloa decumbens* dominance). The few important native species were the fast-growing native perennial grass *Schizachyrium sanguineum* (10%) and the fast-growing native shrub *Vernonanthura polyanthes* (1%).

The fourth cluster is composed of more recent sites (3 to 6 year-old) and is dominated by fast and slow-growing native perennial grasses such as *S. sanguineum* (30%) and *Trachypogon spicatus* (8%), respectively (named: Native fast and slow-growing grasses). The last cluster is composed of the younger sites (<2 year-old), and is dominated by native short-lived sub-shrubs *L. aurea* and *Stylosanthes* spp. (21 and 18% respectively), and two fast-growing native grasses, the perennial *S. sanguineum* (9%) and the annual *Andropogon fastigiatus* (7%) (named: Native short-lived sub-shrubs + fast-growing grasses). *Urochloa decumbens* was the only exotic species with an average cover higher than 5% in all groups followed by *A. gayanus* which only fell short of the 5% mark in two groups (with 4 and 0.6% cover in groups three and five respectively).

Fig. 2. Heatmap of the relative species cover (for species with relative cover greater than 2% in at least one site) for 22 restoration sites in Chapada dos Veadeiros National Park (C), Entre Rios Private Farm (E) and Lake Descoberto Preservation Area (D). Numbers in the names of the sites stand for site age and ID (eg. 4_1 is 4 year-old site 1). Sites are organized in the following functional groups: annual grass (AG), short-lived sub-shrub (SLSS), fast-growing

perennial grass (FGPG), slow- growing perennial grass (SGPG), fast-growing shrub/tree (FGST), slow-growing shrub/tree (SGST), invasive exotic grass (IEG) and not classified (NC).

3.2. Plant density and richness

Although native trees and shrubs had a generally low cover (between 3 and 53% on average), they were established in all restoration sites with a high density (4,081 individuals/ha on average) and richness (80 native species in total 44 seeded, Fig. 3A & 3B). Clusters of older sites had both higher density and richness of woody species with the Native slow-growing grass + exotic grass group having the highest number of species and the highest density.

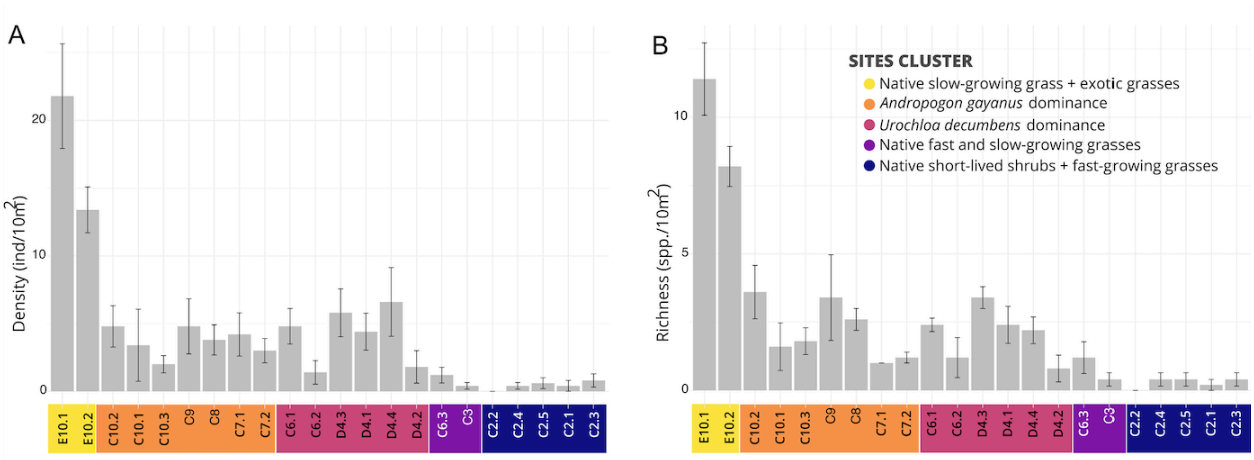


Fig. 3. Density (A) and richness (B) of woody native species (mean ± SE) in restoration sites in Chapada dos Veadeiros National Park (C), Entre Rios Private Farm (E) and Lake Descoberto Preservation Area (D) sampled between February and May 2022. Numbers in the name of the sites stand for site age and ID (eg. 4_1 is 4 year-old site 1).

3.3. Predictors of restoration success

Species composition (Supplementary Table 1) was strongly related to site age and landscape IEG cover. Older sites had more IEG cover in their buffer areas and were dominated by the IEG *A. gayanus* and *U. decumbens* (Fig. 4). *Urochloa decumbens* dominated most sites under restoration, except for younger sites, which were subjected to fire before seeding and IEG chemical control after seeding (Table 1; Fig. 4) and had less landscape IEG cover.

The slow-growing native perennial grass *E. inflexa* shared dominance with IEGs (mainly *U. decumbens* and *A. gayanus*) in only two plots (bottom of the graph, Fig. 4) characterized by low fertility, low pH, and high Aluminum saturation (Supplementary Table 2, 3 and 4). In contrast, native short-lived shrubs and fast-growing grasses dominated the youngest sites (1 to 3 year-old), which had better management of IEGs, before and after seeding. Two sites (3 and 6 year-old) were dominated by fast and slow-growing native grasses, one of them being periodically managed with herbicide for IEG control and the other on low-fertile acidic soil.

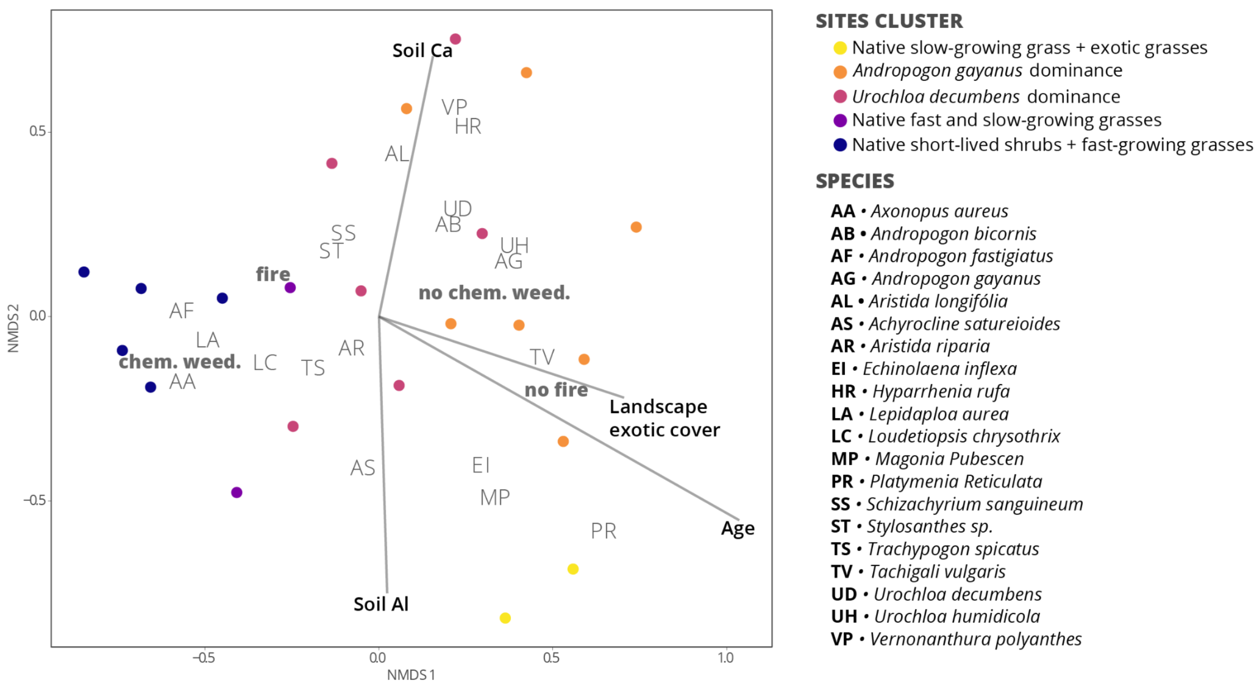


Fig. 4. Non-metric multidimensional scaling (NMDS) ordination of species composition cover in 22 restoration sites of Brazilian savanna, in Central Brazil (Stress = 0.16) sampled between February and May 2022. The vectors (continuous) and bold gray names (categorical) represent soil and management variables significantly related to the vegetation composition cover, as a result of envfit analysis (see Supplementary Table 4).

4. Discussion

During the UN Decade on Ecosystem Restoration, the bias toward forest restoration and tree planting has been strong (Dudley et al., 2020; Silveira et al., 2022). Developing efficient and socially inclusive methods for restoring the Neotropical Savanna is urgent to include the restoration of these ecosystems in global and regional priorities. Former evaluations of savanna restoration with direct seeding evaluated only young (1 to 3 years) restoration sites (Giles et al., 2022) or did not focus on species and functional composition (Sampaio et al., 2019). The myriad of relationships between plant assembly, age and management techniques in restoration sites emphasizes the need for studies to address multiple conditions and long-term experiences to avoid premature conclusions and recommendations. Evaluating older sites is important because restoration trajectories of open-canopy ecosystems are yet unknown and the system could return to its degraded state or reach another alternative state (Suding, 2011). Here we evaluated 22 sites of operational scale restorations up to 10 years (the oldest sites we know of), in three areas, under diverse management schemes and soil types.

We found that direct seeding can simultaneously establish native species of various life forms (trees, shrubs, sub-shrubs/forbs and grasses) and functional groups (fast and slow growing) of the Neotropical Savanna. Fast-growing annual grasses and short-lived sub-shrubs formed a full vegetation cover on 1 to 3 year-old sites. They subsequently senesced with the death of their first cohort and stayed in low densities in the communities (see also Coutinho et al., 2019). After this initial phase, sites (3 to 6 year-old) were dominated by both native and invasive perennial grasses. Invasive perennial grasses, native fast-growing and slow-growing perennial grasses covered older sites (6 to 10 year-old). Native fast-growing trees occupied the emergent strata, whereas slow-growing shrubs and trees were established at high densities, but still had a low contribution to the vegetation cover in any strata, due to their growth rate. In the absence of IEGs, one could expect slow-growing perennial grasses to dominate the herbaceous stratum, whereas slow-growing shrubs and trees would form a crown cover from 5 to 70%, as in characteristic in native cerrado sites (Ribeiro and Walter, 2008), with higher tree cover in moisture climates, fertile and deep soils, under lower fire frequency (Schmidt et al., 2019).

In most sites, due to the absence or insufficient weeding, IEG reinvaded and dominated vegetation cover, except for the sites on infertile soils (cluster Native slow-growing grass + exotic grasses). Although native species, especially trees, were significantly present even under IEG dominance, the success of up to 10 year-old restoration sites was mostly low. The soil seed bank and propagule pressure from the IEG facilitated invasion in the restoration sites, especially given their prevalence in the surrounding areas. The IEG represented around 50% of soil cover after 10 years of restoration interventions, even on infertile soils. Thus, even though reinfestation is lower in poor soil conditions, it does not preclude continued control of IEG in these areas.

4.1. The challenge of IEG control

Reinvasion of restoration sites by IEG represents the main obstacle to restoring the Brazilian savanna. Invasion by IEG has been observed in sites restored with topsoil deposition (Ferreira et al., 2015), and even in old-growth savannas (Assis et al., 2021). We found that landscape IEG cover was strongly related to IEG dominance inside restoration sites probably because it facilitated reinvasion by providing high propagule pressure. Younger sites had less IEG in their buffers since IEG landscape presence decreased with the increase of sites under restoration in the study areas. Although we could not test the effects of age and landscape IEG cover independently, both factors together with chemical weeding have contributed to the low IEG cover found in younger areas. The reduction of IEG landscape cover is important but on its own it is not sufficient to completely exclude the exotic grasses from the areas under restoration due to resprout and germination from the seed bank.

African grasses have been genetically improved to be used as cultivated pastures across the region and can quickly recolonize recently direct seeded sites, especially *Urochloa decumbens* through bud or seed bank, and *Andropogon gayanus* through seed dispersal. The establishment and persistence of IEG is facilitated by calcium addition to the soil through liming, which is a common practice in agriculture and pasture areas due to the natural acidity of soils in the Brazilian savanna (Lambers et al., 2020). In our study, previous soil liming was found to facilitate the establishment and persistence of IEGs and was highly associated with the exotic grass *U. decumbens* cover. Therefore, IEG control methods are the most outstanding challenge for the restoration success in Neotropical Savannas, regardless of the native species' reintroduction methods. Effective IEG control before the reintroduction of native species is important because controlling them afterwards is more challenging and expensive, as they are intermixed with native species. Some companies, however, are successfully doing selective weeding through directed glyphosate application or hoeing (Semeia Cerrado and Tikre Brasil direct communication).

Despite its importance, effective IEG control for ecological restoration is not always possible. Restoration sites were mostly located in areas where the chemical control of IEG was not allowed (restricted protected areas and/or areas close to water reservoirs, e.g. Wiederhecker-Gabriel et al., 2022; Sampaio et al., 2019). Although information on the trajectory of these restoration sites has helped change the national legislation for the use of herbicides in Federal Protected Areas (Brasil, 2019), the development of other control strategies to decrease the prevalence of IEG in restoration sites is necessary. This may involve the use of cattle (Durigan et al., 2022) and/or the manipulation of soil properties to bring nutrient availability and pH closer to original levels in sites that were fertilized and limed (Walker et al., 2004). Adding iron sulfates to the soil helps decrease pH and IEG prevalence in the Brazilian savanna (Lira-Martins et al. submitted), however, the costs might not justify the benefits. Another alternative, that has not been tested for restoration, is the addition of Nitrogen based fertilizer which lowers soil pH and is known to limit at least one exotic grass *Melinis minutiflora* (Bustamante et al., 2012).

4.2. Present and absent functional groups

The past decade saw considerable advances in Brazilian savanna restoration, specifically in species propagation under field conditions (Ferreira et al., 2015; Pellizzaro et al., 2017; Pilon et al., 2018). Assembly models for post-fire recovery (Schmidt et al., 2019), and functional groups related to regeneration strategies (Pilon et al., 2021) have been described but the reassembly of savannas after the seed and bud bank is lost has not. This knowledge is imperative since sites where the bud bank has been extinguished such as mechanized agriculture, pastures, and mining are the primary active-restoration demand of the Brazilian savanna (Strassburg et al., 2017). Furthermore, understanding savanna reassembly is essential to plan optimum species and life-history trait combinations that promote the sustainability of restored communities.

Analyzing the results of the sites under restoration through direct seeding, with a diversity of species, growth forms, lifespan, and growth rates, allowed us to observe the reassembly of the savanna community in a way that can inform the restoration method design (Young et al., 2005). To better understand community assembly, species were classified based on three functional and life-history traits: growth rate, lifespan, and growth form. Fast growth rates were important to cover the soil in the first three years, reducing erosion and seed dragging, longer lifespans were important to maintain native cover stability over the years and diversity of growth form is essential to re-create a savanna structure.

Direct seeding may be biased toward species with low seed prices, high seed availability and high emergence rates (Silva et al., 2022). Nevertheless, it is possible, through this technique, to maintain a savanna structure able to support the colonization of other species and functional groups. There is evidence that the recovery rates of old-growth savannas are low, and it can take decades to recover species dependent on vegetative propagation and with low sexual reproduction rates (Nerlekar and Veldman, 2020; Pilon et al., 2021, 2019). Many forbs, geofits, monocotyledonous shrubs (*Velloziaceae*), and other growth forms were not included in the restoration sites and should be considered in the future (Ferreira et al., 2015; Pilon et al., 2019). Other types of propagation should also be tested in terms of cost-benefit since, for resprouters and rare species the low availability of seeds could result in high costs of direct seeding (Silva et al., 2022).

Although some functional groups characteristic of old-growth areas (such as undergrounders see Pilon et al., 2023) were absent from the restoration sites , others were found in our study sites, represented by some key species. For example, the fast and slow-growing perennial grasses *Schizachyrium sanguineum*, *Echinolaena inflexa*, *Loudetiopsis chrysothrix* and *Trachypogon spicatus* and the slow-growing trees *Dipteryx alata* and *Magonia pubescens* (Flora e Funga do Brasil, 2020). Some of our older restoration sites were dominated by perennial grasses, *E. inflexa* and *L. chrysotrix*, which are indicators of old-growth Brazilian savannas (Amaral et al., 2022), implying that the restoration efforts can trigger plant communities structurally similar to natural states. This study is the first designed for active restoration of Brazilian savannas considering functional groups and providing useful data for species combinations for restoration.

The functional groups used in this study have parallels in subtropical and temperate grassy ecosystems. In the Australian subtropical grasslands, a successional trajectory of functional groups was observed after 60 years of abandonment, with rapid colonization by annual grasses and forbs, followed by a slow recovery of perennial forbs species and a tendency of perennial grasses to increase (Fensham et al., 2016). Similarly, Tognetti et al. (2010) observed a turnover from annual forbs to annual grasses, then to perennial grasses with both exotic and native species in the subtropical Pampa grasslands in South America. Another similar pattern was identified in Mediterranean grasslands, however, the recovery sites still lacked late-successional species 30 years after disturbance (Coiffait-Gombault et al., 2012).

4.3. Study Limitations

The study areas differed in environmental conditions (soil types, climate and land use history) and restoration interventions (soil preparation and post-planting weeding). Some variables such as restoration age, post-sowing weed control and lower soil fertility changed simultaneously, which prevented the study from having a balanced design. These changes in restoration practices were the result of previous evaluations of restoration efforts and joint development of restoration practice and science in the Brazilian savanna, as recommended in many parts of the world (Gann et al., 2019). Another limitation of the present study was the predominance of sites in the CVNP, which might have skewed our analysis towards factors that are characteristic of that region. Finally, although we bring information on a considerable number of sites with known restoration history, we only analyzed the current condition of each site and not their dynamics, which will give more insights into the factors that affect community reassembly.

5. Conclusions and a way forward

By studying the largest and oldest operational scale restoration of Brazilian savannas we showed that direct seeding effectively introduces different functional plant groups. It enabled native plant colonization and community turnover with fast-growing and short-lived sowed species dominating the community until the third year, after which, native slow-growing and long-lived sowed species gradually increased dominance. Management and soil conditions affected community assembly but did not significantly intervene in native species turnover. They did, however, affect exotic grasses' dominance which benefited the native community.

Trees were successfully established from seeds, even with reinvasion by exotic grasses, but other life forms, especially native grasses, were severely reduced in IEG-dominated sites. Establishing native species is largely unsuccessful in preventing IEG reinfestation, especially in areas where IEG dominates the landscape. The effective control of IEG is therefore key to increasing the success of restoration efforts in the Brazilian savanna since establishing native species of different life forms through direct seeding is feasible on an operational scale using techniques and inputs currently available.

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Capítulo 2

Long-Term Trajectories of Direct Seeding in Neotropical Savannahs

1. Introduction

Ecological restoration is the process of altering the state of a community to facilitate a trajectory towards a more desirable state, closer to a reference state (Hobbs and Norton, 1996; Gann et al., 2019). Community trajectories, however, have long been recognized as a naturally dynamic process (Mayer and Rietkerk, 2004; Suding et al., 2004) and trajectory imprevisibility is one of the main challenges for current restoration ecology (Suding, 2011; Brudvig et al., 2017). Even when a clear and feasible goal exists, restoration might fail due to ecological drivers that push communities beyond stability thresholds and towards alternative stable states (Hobbs and Norton, 1996; Mayer and Rietkerk, 2004; Suding, 2011; Fukami et al., 2005). Monitoring and management are, therefore, an important part of the restoration process. Despite that, few restoration projects monitor restoration sites for more than a few years and trajectories are often assumed without proper knowledge of their driving factors (Palmer and Ambrose, 1997; Suding, 2011).

Currently, human activities are the main cause of community degradation and alteration beyond natural thresholds. Once past such thresholds an ecosystem can change to a new alternative stable state (Hobbs et al., 2006; Ellis et al., 2010; Suding, 2011). To restore such systems, stopping the degrading factor is not enough and active reintroduction of species and post-reintroduction management are often needed (Mayer and Rietkerk, 2004; Holl and Aide 2011; Silveira et al., 2020; Pilon et al., 2023). Sometimes, the return of an endogenous disturbance is enough to push the system back to its previous state. This is often the case for systems that were degraded by the interruption of a disturbance regime such as fire and grazing (Buisson et al., 2019). Even when that is not enough, disturbance reintroduction is still important for restoration projects. Especially considering that one of the main goals of restoration is to facilitate the start of a trajectory that will then be kept on its own (Palmer and Ambrose, 1997; Gann et al., 2019).

Among the major threats to self-sustaining restoration trajectories are biological invasions. Exotic invasive species are often the main factor pushing an ecosystem past natural thresholds and towards an alternative stable state. Invaded ecosystems often stabilize beyond natural thresholds becoming resilient and resistant to both endogenous and exogenous disturbances (D'Antonio and Meyerson, 2002; Hobbs et al., 2006). That is the case for exotic grasses invasion on Neotropical Savannahs. Once established, not only do exotic grasses resist the natural savannah fire regime but also alter it to their benefit (Gorgone-Barbosa et al., 2015). These species are able to exclude native species through competition, reducing the diversity of invaded areas (Brooks et al., 2010; Hoffmann and Haridasan, 2008). In such cases, exotic species control is needed to restore the system to its previous condition (D'Antonio and Meyerson, 2002; Coutinho et al., 2019; Wiederhecker et al., 2024).

In light of the above, restoring areas with uncertain trajectories and high resilience of the degraded state may require considerable effort and time. Doing such is even more of a challenge in areas where restoration practice is new and knowledge of species and trajectories is lacking (Hobbs et al., 2006; Suding, 2011). In the Neotropical Savannah, restoration efforts for savannah physiognomies are recent (Sampaio et al., 2019; Pellizzaro et al., 2017). The oldest known large-scale direct seeding restoration sites were seeded in 2012 (Wiederhecker et al., 2024). Restoration trajectories are, therefore, still unknown. Adding to this, most areas selected or available for restoration are abandoned pastures (Barros et al., 2023), where the soil has been fertilized and/or limed, and cultivated with exotic grass species that keep ecosystems in a stable degraded state. Additionally, most restoration sites were implemented inside federally protected areas, where the use of herbicide for chemical IEG control was not allowed until 2019 (Brasil, 2019).

Direct Seeding has been a leading technique to restore open ecosystems due to its cost-efficiency and its capacity for the simultaneous introduction of different life forms through native seed mixes (Raupp et al., 2024; Ferreira et al., 2023; Pellizzaro et al., 2017). Direct seeding is also extremely important for the local economy, often generating jobs and opportunity for the population around restored areas and creating a socially and environmentally balanced supply chain on native seeds (Montenegro et al., 2024; Urzedo et al., 2022; Schmidt et al., 2019). Nevertheless, direct seeding has a diversity limitation to seeder species and often excludes from restoration areas slow-growing species with low seed production, characteristic of these old-growth Savannahs and Grasslands (Pilon et al., 2023; Veldman et al., 2015). In the Neotropical Savannah this technique has been mainly implemented and tested inside federal protected areas in large-scale restoration experiments and operational scale restoration practices (Montenegro et al., 2024; Wiederhecker et al., 2024).

In the present study, we collected data for the community trajectory of 22 restoration sites in central Brazil. Among them are the oldest known Brazilian Savannah direct seeded restoration sites. Our main objective was to understand community trajectories of Neotropical Savannah restoration sites and how they responded to different management conditions. For that we seek to respond to the following questions: i) Is there dominance turnover from fast to slow growth life forms? ii) Do restoration sites approach native reference sites? iii) Is currently implemented IEG control sufficient? We hypothesized that i) Dominant in the first years, fast growing native species would gradually leave the system or reduce their cover being replaced by slow growing species. ii) Restoration sites can reach similar levels of soil cover by native grasses and trees within a few years after direct seeding, if invasive exotic grasses are controlled. iii) The control of exotic grasses during the initial restoration years is sufficient to prevent exotic species' reinvasion, in sites previously dominated by exotic grasses.

Whilst Chapter 1 was focused on the current state of the areas in this chapter we explore more closely the trajectory of the restoration areas in the years following restoration using data from multiple samplings. We also compare the current state of the restoration areas to native reference areas to access restoration success.

2. Materials and Methods

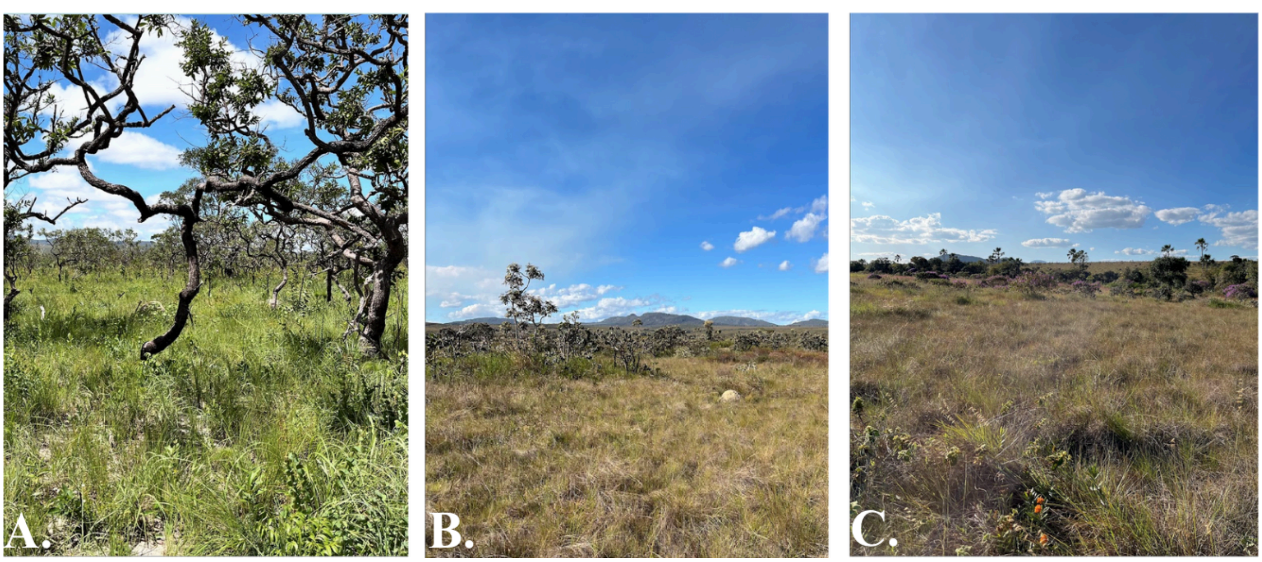
2.1 Study sites

We sampled 22 restoration sites in central Brazil, within the Cerrado Biome, with predominance of Plinthic Soil and Yellow or Red Latosol, climate characterized as Aw (Koppen), elevation between 1000m and 1200m above the sea level and with rainfall of 1600 mm^3/year (INMET, 2024) during the rainy season (October to May). Most of the restoration sites (16 sites) are located in close proximity, inside the Chapada dos Veadeiros National Park (lat 14°06'43"S long 47°38'14"W), two are located in a private farm (lat 15°46'31"S, long 48°13'12"W) and four are in the Lake Descoberto Preservation Area (lat 15°57'30"S, long 47°27'26"W). All restoration sites were limed before IEG introduction for pasture implementation except for those in the private farm, which were seeded with IEG without further soil alterations.

All sites were dominated by invasive exotic grass (IEG) species such as *Andropogon gayanus* and *Urochloa decumbens* and had been abandoned for at least 30 years before restoration implementation. Sites had been restored between four to 12 years before our sampling, by direct seeding of native species seed mixes (different life forms: trees, shrubs, sub-shrubs/forbs and grasses and different functional groups: fast and slow-growing) after burning and soil ploughing; with two sites having received additional seedling planting (Orli, 2018; Wiederhecker-Gabriel et al. 2022). Seed mixes varied in species composition among sites and years (Supplementary Material S1). All sites were originally an open-canopy Savannah with some sites transitioning to wet grasslands.

To analyse the trajectories of large-scale experimental restoration sites in the Brazilian Savannah, we compiled sampled data of vegetation cover and woody abundance of the past twelve years (2012-2024) for these 22 restoration sites. Additionally, we selected and sampled three native physiognomies as reference states (Fig.1): an open-canopy Savannah with dominance of grasses and the presence of shrubs and trees (*cerrado sensu-stricto*), a similarly open-canopy Savannah but with trees limited to small hills (*campo de Murundu*) and a wet grassland (Ribeiro and Walter, 1998). Reference native sites were only found in one of the three study regions (the Chapada dos Veadeiros National Park) due to high degradation of the landscape around other restored sites. We considered these three reference sites because the change in native physiognomies may happen in small areas within the Cerrado (Ribeiro and Walter, 1998; Bueno et al., 2018; Schmidt et al., 2019) and all the study restoration sites were certainly within these three native physiognomies and their transition states.

Figure 1: Native physiognomies sampled as reference states for the restoration sites. *Cerrado sensu-stricto* (**A**), *Campo de Murundu* (**B**) and a wet grassland (**C**).



Restoration sites varied in soil preparation, size of the site restored, date of restoration intervention and management practices after seeding (Wiederhecker et al., 2024). The use of herbicides as a chemical control inside Federal Protected Areas was only recently regulated and therefore, was not implemented in older restoration areas (Brasil, 2019). Since 2019, after the use of herbicide was allowed in restoration sites inside Protected Areas, seeded sites were managed with herbicides to control IEG. Chemical control was implemented on 2019 and 2020 restoration sites by applying a glyphosate-based solution in a concentration of 3% in 10-40cm tall IEG tussocks by workers wearing appropriate personal protective equipment using backpack pump sprayers (WHO, 2001; ICMBio, 2023). The chemical control of IEG in these sites were performed whenever tussocks were high enough to be considered fully photosynthetic active (usually three to four months intervals) for three years in the sites seeded in 2019 and two years in the sites seeded in 2020.

Other sites had mostly no form of exotic species control in the years following restoration. Lastly, three sites were affected by wildfires during the sampled time frame: CVNP_2016_2 site was burned by a lightning fire in Sep. 2019 and both sites from the ERPF_2013 batch were burned by an anthropogenic fire in Sep. 2022. These sites were thus included as case studies for the response of restoration sites to fire. Since fires were unexpected it was not possible to establish an experiment but temporal comparison and comparison with other sites were possible.

2.2 Data compilation and collection

In total we gathered data from 37 sampling efforts in 22 restoration sites joined in nine batches by the year of restoration intervention and implemented management (Tab. 1). Sampling methods changed slightly throughout the years (i.e. Alves, 2016; Pellizzaro, 2016; Pellizzaro et al., 2017; Orli, 2018; Coutinho et al., 2019; Liaffa, 2020; Silva-Coelho, 2022) and each site had between two and seven samplings. In the 2012 sites, first samplings were made using quadrats. From 2014 onwards the line interception method was used and sites had different numbers of sampling lines according to size and sampling year. The seven batches of restoration sites located in the Chapada dos Veadeiros National Park were named CVNP_2012, CVNP_2013, CVNP_2014, CVNP_2015, CVNP_2016, CVNP_2019 and CVNP_2020. And the two batches located in Distrito Federal were named ERPF_2013, in the private farm, and LDPA_2018, in the Preservation Area. The current state of all restoration sites and of the three native reference sites was assessed in

April/May 2024, (by the end of the rainy season) by sampling species cover and woody abundance.

Table 1: Characteristics of the 9 batches of restoration sites in Chapada dos Veadeiros National Park (CVNP), Entre Rios Private Farm (ERPF) and Lake Descoberto Preservation Area (LDPA). Numbers in the name column stand for site location, year and id (e.g. CVNP_2020_1 is CVNP restoration implemented in 2020 site 1). Collected sampling efforts analysed for each group are listed. References are listed for a detailed description of restoration methods.

Batch	Sites	Coordinates	Soil type	Sampling Efforts	References
CVNP_2012	CVNP_2012_1, 2 & 3	14 06'43"S 47 38'14"W	Plinthic	7	Alves 2016
CVNP_2013	CVNP_2013	14 06'43"S 47 38'14"W	Dystrophic Latosol	6	Pelizzaro 2016
CVNP_2014	CVNP_2014	14 06'43"S 47 38'14"W	Plinthic	5	Orli 2018
CVNP_2015	CVNP_2015_1 & 2	14 06'43"S 47 38'14"W	Plinthic	4	Coutinho et al., 2019
CVNP_2016	CVNP_2016_1, 2 & 3	14 06'43"S 47 38'14"W	Plinthic	4	Liaffa 2020
CVNP_2019	CVNP_2019	14 06'43"S 47 38'14"W	Plinthic	2	Silva-Coelho 2021;
CVNP_2020	CVNP_2020_1, 2, 3, 4 & 5	14 06'43"S 47 38'14"W	Plinthic	2	Semeia Cerrado (pers. comm.)
ERPF_2013	ERPF_2013_1 & 2	15 57'30"S 47 27'26"W	Dystrophic Latosol	4	Pelizzaro et al., 2017
LDPA_2018	LDPA_2018_1, 2, 3 & 4	15 46'31"W, 48 13'12"S	Red-Yellow Latosol	3	Wiederhecker-Gabri el et al., 2022

On each site, we sampled five 10m² (20m x 0.5m) plots distant at least 20m from each other. We sampled soil cover following the line-interception method (Herrick et al., 2006) with 40 intercepts placed along the 20m line plot. We performed woody abundance measurements within the 10m² plot by counting all woody individuals. Vegetation sampling was done mostly following the same methodology used in the last sampling made in 2022. However, the previously adopted minimum height of 30cm for woody individuals on density measures was disregarded due to a high number of individuals having been excluded from sampling, especially in younger sites. In the present sampling we considered all woody individuals we could identify as such (Wiederhecker et al., 2024).

2.3 Data Analysis

2.3.1 Restoration trajectories

Since cover data was sampled differently throughout the years, mean values were calculated individually for each sample or in clusters of samplings with the same methods (same number and length of plots and subplots). The mean cover for each restoration batch was calculated as the average cover for all sites within that batch. The average site cover was calculated by averaging, for all plots, the number of occurrences of a functional group on each plot divided by the total number of occurrences in that plot. For that, we classified species into seven functional groups: annual grass (AG), Short-lived sub-shrub (SLSS), fast-growing perennial grass (FGPG), slow-growing perennial grass (SGPG), fast-growing shrub/tree (FGST), slow-growing shrub/tree (SGST) and invasive exotic grasses (IEG) following the classification used by Wiederhecker et al. (2024).

We made a rarefaction-extrapolation analysis to compare the diversity and richness of the different restoration sites throughout the years. We extracted species richness from cover data. Since the number of samples varied over the years, we performed an incidence or sampled-based analysis. This approach allows for standardising data based on the number of samples thus making our diversity values comparable (Chao et al., 2014). We chose to use the value of the Hill number of $q = 2$ (Simpson's diversity) since restoration sites tend to have several rare (low abundance) species. We also calculated richness values for the Hill number of $q = 0$. We analysed the data using the iNEXT package (Hsieh et al., 2016) in the R software for statistical analysis version 4.2.2 (R Core Team, 2024).

2.3.2 Native comparisons

We used cover, diversity, and woody abundance samples of 2024 to compare restoration sites to the three native reference sites. For that, we classified plant cover data in four ways: by genus, by family, by category (native grass, native shrub/tree and invasive exotic grass) and by origin (seeded and natural regeneration). For each classification, we calculated site cover as the average cover of all plots. We extracted diversity values for restoration and native sites from the rarefaction analysis using the 2024 sampling cover data. We calculated woody abundance by dividing the number of individuals from a genus or a family by the total number of individuals in the plot. We then calculated the average abundance by site of each classification (genus and family). Whenever possible, we identified plants at the species level.

To understand how different restoration sites were from native sites, we statistically compared the grass and woody cover of these sites. Since data was not normally distributed, we performed generalized linear models (glm) using quasi poisson distribution and the logarithmic link function. To understand where observed differences lay, we performed a post hoc Tukey's Honest Significant Difference test. All scientific names used followed the botanical nomenclature from Flora e Funga do Brasil (2020).

2.3.3 IEG control and persistence

We compared the cover of IEG through time on three different conditions: chemical control interruption, acidic and infertile soil and wildfire. The mean IEG cover of the CVNP_2019 and 2020 restoration sites were compared between the last rainy season with chemical weeding (2022) and the second rainy seasons after its interruption (2024). We compared both means to the ERPF_2013 (acidic and infertile soil) IEG mean for the same time interval. Since cover was sampled following the same methodology on both years, direct comparisons were possible. Again, since data was not normally distributed, comparisons were made with generalized linear models using quasi poisson distribution and the logarithmic link function. For visualization, we plotted cover data in a boxplot.

Lastly, we compared the cover of the functional groups before and after fire in two batches where wildfires occurred CVNP_2016 and ERPF_2013. For that we also used generalized linear models using quasi poisson distribution and the logarithmic link function, since data was not normally distributed. Additionally, to identify if any functional groups had significantly altered their cover after fire, we calculated estimated marginal means (EMMs) using the Emmeans function (Lenth, 2024). For visualization, we plotted cover data in a boxplot. All analyses were performed in the R software for statistical analysis version 4.2.2 (R Core Team, 2024).

3. Results

3.1 Restoration trajectories

Restoration sites trajectories were highly influenced by the dominance of invasive exotic grasses (IEG) or lack of thereof (Fig. 2). In the CVNP_2012 to 2016 batches, IEG increasingly dominated the community, reaching mean covers higher than 70% after the ninth rainy season. In these sites annual grasses and short-lived sub-shrubs (i.e. forbs) reduced their cover between the 3rd and the 5th rainy season after direct seeding, to less than half of the cover in the first two years after seeding. Fast-growing perennial grasses cover fluctuated between 15 and 1%, reducing as IEG cover increased. Slow-growing perennial grasses had a similar fluctuation, but values were generally lower. Overall woody cover was lower than 10% for both fast and slow-growing, with the latter having less than 5% cover in all years. The CVNP_2012 batch was an exception, with 38% cover of fast-growing trees, mainly *Tachigali vulgaris*.

In the few sites that had limitations to invasive exotic grasses development due to chemical weeding (CVNP_2019 and 2020) or naturally acidic and infertile soils (ERPF_2013) trajectories were vastly different. With IEG covers kept at less than 20% in early years, native grasses were able to thrive. The fast-growing perennial grasses reached covers higher than 20% at their fourth rainy season while slow-growing perennial grasses had covers higher than 10% as early as their third rainy season. In the ERPF_2013 batch, native grass covers were kept high even at the 11th rainy season and slow-growing perennial grass cover surpassed that of fast-growing perennial grasses by almost 20%. Woody cover was kept low in all sites but reached 14% at the 9th rainy season in the ERPF restoration sites. Only annual grasses and short-lived sub-shrubs trajectories were not affected by invasive exotic grasses dominance and kept their reduction tendencies (Fig. 2).

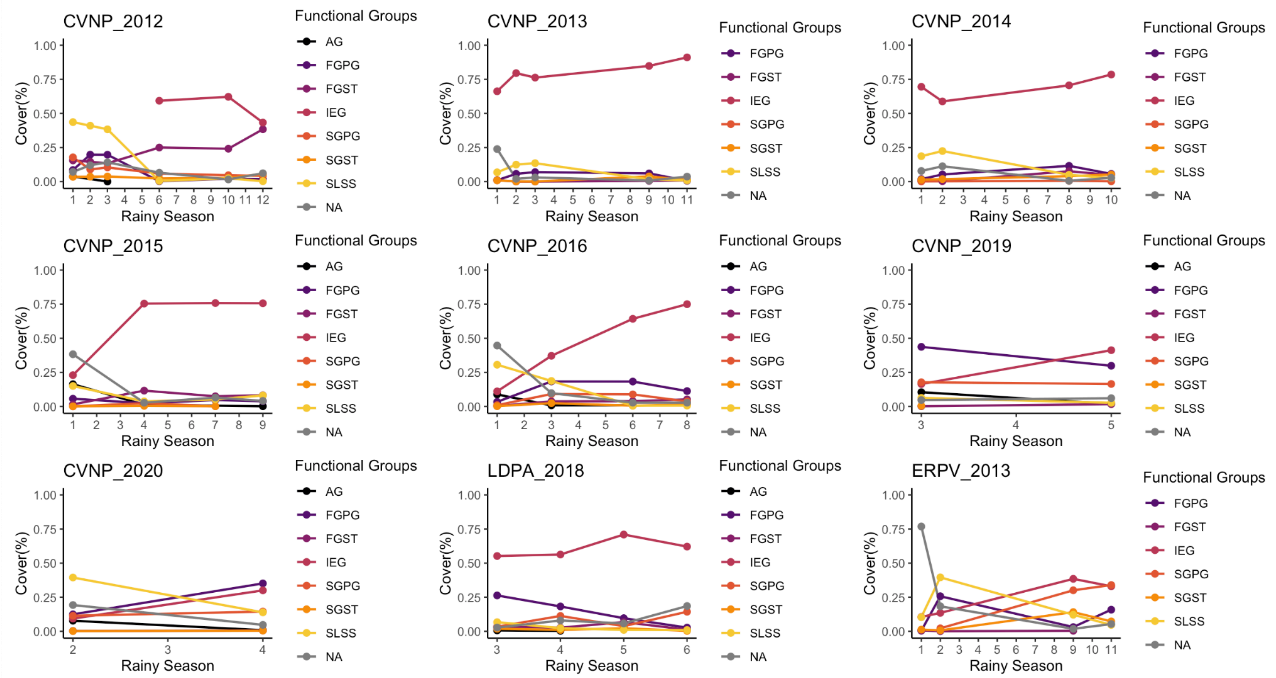


Figure 2: Trajectories of vegetation functional group cover by rainy season of all restoration sites in Chapada dos Veadeiros National Park (CVNP), Entre Rios Private Farm (ERPF) and Lake Descoberto Preservation Area (LDPA) between 2012 and 2024. Sites have been divided in 9 batches by restoration intervention start date and implemented management. Site name consists of its location plus the year of the start of restoration intervention. Species are organized in the following functional groups: annual grass (AG), short-lived sub-shrub (SLSS), fast-growing perennial grass (FGPG), slow-growing perennial grass (SGPG), fast-growing shrub/tree (FGST), slow-growing shrub/tree (SGST), invasive exotic grass (IEG) and not classified (NA).

Richness and diversity values also responded to IEG dominance in restoration sites. Where exotic species were not managed, species richness reduced throughout the years and species diversity mostly followed this reduction (Fig 3). Four batches were exceptions to this pattern: CVNP_2012, 2019, 2020 and ERPF_2013. In the CVNP_2012 batch despite the slight decrease in species richness (from 43 to 39) Simpson's diversity slowly increased, going from 16 to 27 at the end of 12 rainy seasons. CVNP_2020, on the other hand, followed the general trend for richness and diversity reduction over the years after seeding. The species richness and diversity values, however, were higher for this batch. Of the initially established 78 species, 62 remained after the 4th rainy season. During this two-year period diversity went from 54 to 39. The PNCV_2019 and ERPF_2013 were the only batches that showed a tendency for an increase in both richness and diversity, in the years following restoration. PNCV_2019 diversity increased significantly (no overlap for the curve confidence interval) following an increase in richness.

Similarly, ERPF_2013 diversity also increased significantly, going from 12 in the first rainy season to 42 in the last, reaching richness as high as 60 species (Fig. 3).

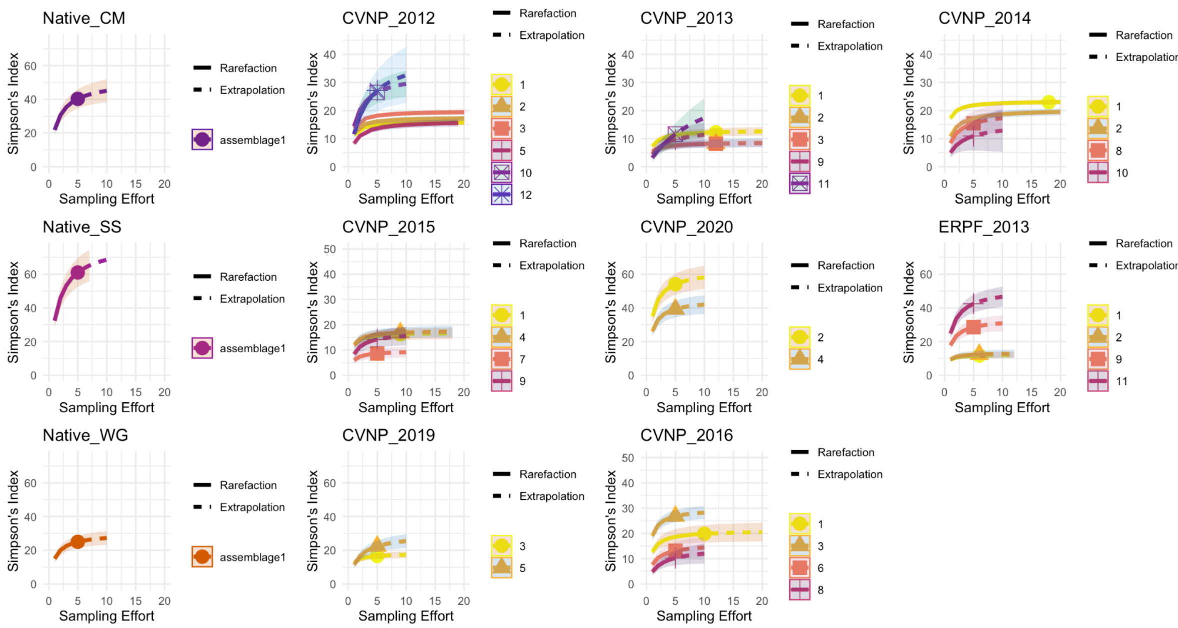


Figure 3: Sample-based rarefaction and extrapolation curves for diversity sampled by the line interception method in all restoration sites located in Chapada dos Veadeiros National Park (CVNP) and Entre Rios Private Farm (ERPF), between 2012 and 2024 and for the native reference areas sampled in 2024. Diversity calculations were based on the Hill number q = 2 (Simpson's diversity). Dots represent reference sample, shaded area 95% confidence interval, solid lines rarefaction and dashed lines extrapolation. Colours and numbers represent the age/number of rainy seasons after restoration. Site name consists of its location plus the year of the start of restoration intervention. Native areas were named: Native_CM (Native Campo de Murundu), Native_SS (Native cerrado *sensu stricto*) and Native_WG (Native Wet Grassland).

3.2 Native comparisons

Beyond keeping high diversity and richness, lower IEG cover allowed the CVNP_2019 and 2020 restoration sites batches to achieve similar levels of native grass cover of a native Cerrado *sensu stricto* reference site, as early as two rainy seasons after restoration ($p<0.05$, Fig. 4). The ERPF_2013 batch kept native cover values similar to reference sites for as long as 11 rainy seasons. Despite that, woody cover of native reference sites was only reached by the CVNP_2012 encroached batch. Overall seeded species cover was higher than native regenerant species cover, mainly ruderal species, in all sites except for the ERPF_2013 batch, where regenerant species cover was higher than seeded species cover by the 11th rainy season (37 and 24%, respectively).

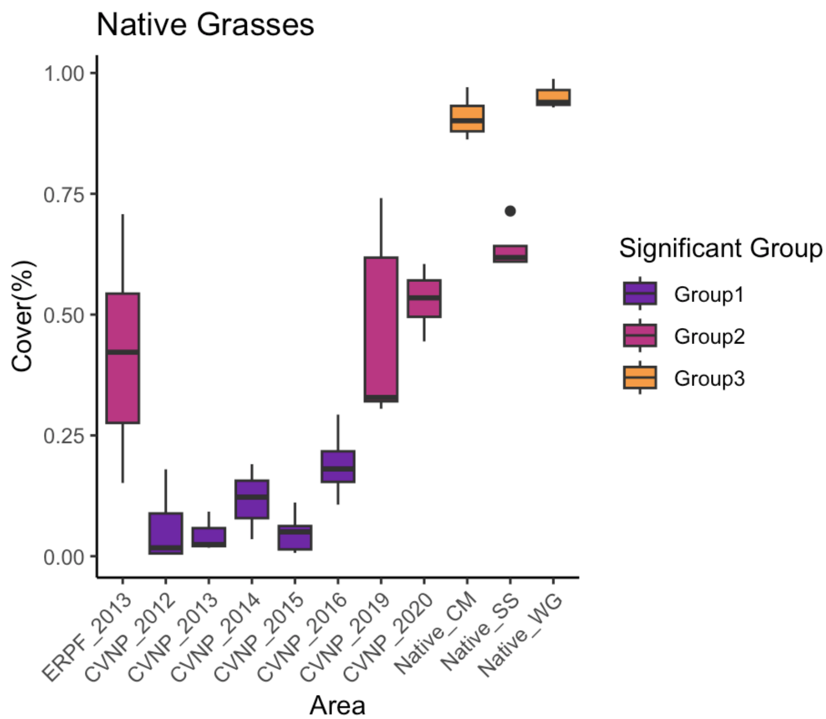


Figure 4: Boxplot of native grass cover in all restoration sites located in Chapada dos Veadeiros National Park (CVNP) and Entre Rios Private Farm (ERPF) and for the native reference areas sampled in 2024. Site name consists of its location plus the year of the start of restoration intervention. Native areas were named: Native_CM (Native Campo de Murundu), Native_SS (Native cerrado *sensu stricto*) and Native_WG (Native Wet Grassland). Rectangles indicate first to third quantile; dots indicate outliers and colours indicate significant difference ($p<0.05$).

Restoration intervention was also able to establish high richness in early years (qD > 15 for a Hill number of q=0). However, only two sites reached richness and diversity values close to that of native reference sites. At its second rainy season the CVNP_2020 batch had a Simpson's diversity value (qD for a Hill number of q=2) as high as 54, whereas the native Cerrado *campo de murundu* and cerrado *sensu stricto,* sampled as references to this site, had diversity values of 25 and 61 respectively (Tab. 2 and 3). Similarly, the CVNP_2019 batch reached a diversity of 22 by the fifth rainy season while the native wet grassland, sampled as reference to this site, had a diversity value of 40. The ERPF_2013 batch was the closest to reference values, reaching a diversity of 42 by the end of its 11th rainy season.

Table 2: Sample-based richness values based on the Hill number q = 0 (Richness). Richness was sampled by the line intersection method in all restoration sites located in Chapada dos Veadeiros National Park (CVNP) and Entre Rios Private Farm (ERPF) between 2012 and 2024, and for the native reference areas sampled in 2024. Group name consists of its location plus the year of the start of restoration intervention.

Group	Age/Rainy season												
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**-**
CVNP_2012	43	32	38			33				37		39	
CVNP_2013	16	13	14						13		13		
CVNP_2014	28	26						23		15			
CVNP_2015	27			28			12		22				
CVNP_2016	38		34			19		14					
CVNP_2019			20		28								
CVNP_2020		78		62									
ERPF_2013	17	18							39		60		
Native_CM													35
Native_SS													93
Native_WG													60

Table 3: Sample-based diversity values based on the Hill number q = 2 (Simpson's diversity). Diversity was sampled by the line interception method in all restoration sites located in Chapada dos Veadeiros National Park (CVNP) and Entre Rios Private Farm (ERPF) between 2012 and 2024, and for the native reference areas sampled in 2024. Site name consists of its location plus the year of the start of restoration intervention.

Group	Age/Rainy season												
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**-**
CVNP_2012	16	17	20			37				26		27	
CVNP_2013	12	8	8						10		12		
CVNP_2014	23	20						15		11			
CVNP_2015	16			17			9		14				
CVNP_2016	20		27			13		10					
CVNP_2019			17		22								
CVNP_2020		54		39									
ERPF_2013	12	12							29		42		
Native_CM													25
Native_SS													61
Native_WG													40

Despite the similarities in diversity, richness and grass cover, species composition was vastly different between restoration and native sites. Restoration through direct seeding was able to establish 274 identified species across all restoration sites, 107 of which were seeded.

(Supplementary material S1). The woody stratum in the restoration sites was mostly dominated by Fabaceae and Asteraceae (39 and 22% mean cover, respectively) while native sites were mostly dominated by Myrtaceae (3% cover) and by Arecaceae (1% cover). On the other hand, Poaceae was the most prominent family (> 90% cover) on both native dry sites and restoration sites with controlled invasive exotic grasses.

A similar pattern was found for genus dominance. While restoration sites were dominated mainly by grasses of the *Schyzachyrium* and *Loudetiopsis* genera (mean cover of 26 and 20% respectively), native dry sites were dominated by the *Paspalum* and *Loudetiopsis* genera (42 and 23%, respectively) while native humid sites were dominated by Xyridaceae of the *Xyrys* genus (84%). The only exception being the ERPF restoration site that was dominated by the species *Echinolaena inflexa* (93%). *Aristida* and *Thrachypogon* genera were the third and fourth most important in restoration sites with covers up to 20 and 17%, respectively.

3.3 IEG control and persistence

Despite the success of chemical control in early years at keeping invasive exotic grasses at low cover levels (<20%) such reduction was not permanent (p<0.05, Fig.5). Two rainy seasons after the interruption of chemical weeding, managed CVNP_2019 and 2020 sites went from 16 to 41% and from 9 to 30% of invasive exotic grasses cover, respectively. Conversely, in the site where acidic and infertile soil conditions limit IEG development, cover seems to have stabilized (p>0.05) at 40%, even 11 years after seeding under no invasive exotic grasses control management (Fig. 5).

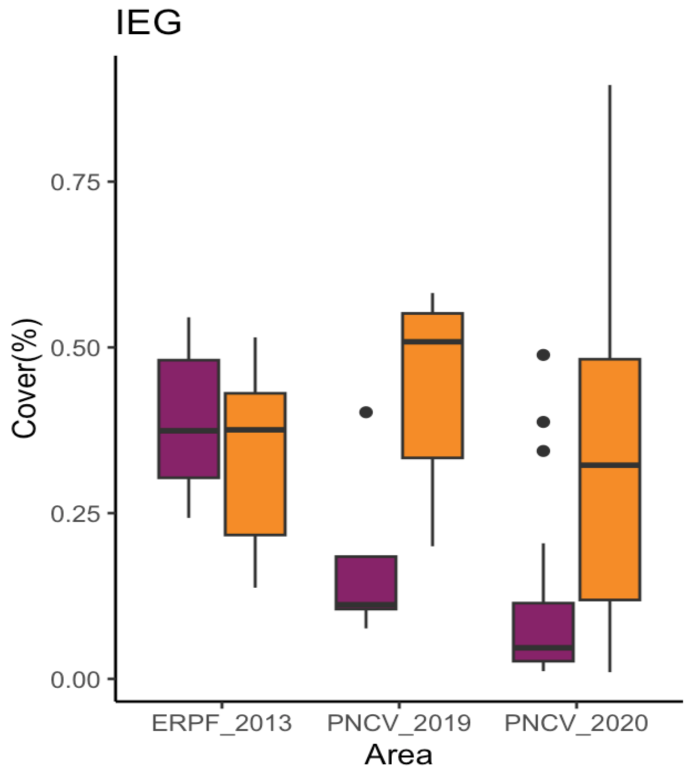


Figure 5: Boxplot of invasive exotic grass (IEG) cover in sites with IEG limitation located in Chapada dos Veadeiros National Park (CVNP) and Entre Rios Private Farm (ERPF). ERPF_2013 had no management alteration between the two samplings (9th and 11th rainy season after seeding). PNCV_2019 and 2020 had interruption of chemical weeding between the two samplings (3rd to 5th and 2nd to 4th rainy season after seeding, respectively). Site name consists of its location plus the year of the start of restoration intervention. Rectangles indicate first to third quantile; dots indicate outliers and colours indicate year of sampling 2022 (purple) and 2024 (orange).

The two late-dry season wildfires did not significantly affect restoration site covers of any functional group presenting the two restoration sites hit by fires (Fig. 6). The invasive exotic grass cover showed a tendency of slightly increase on PNCV_2016 site after the fire, while it slightly decreased in the ERPF_2013 sites after fire; both were non-significant ($p>0.05$) with an average of 10% variation. Native grasses showed a tendency to increase in cover an average of 10% and 28%, respectively, on both sites while sub-shrubs, shrubs and trees decreased in cover after the wildfires.

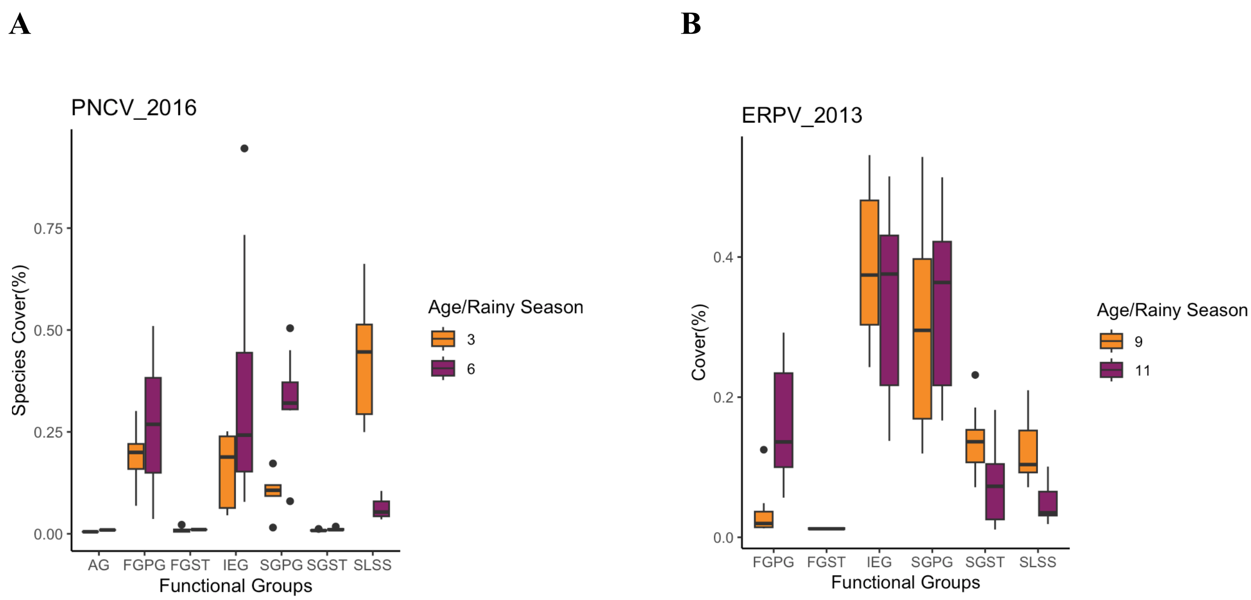


Figure 6: Boxplot functional group cover before (orange) and after fire (purple) in burned sites located in **A:** Chapada dos Veadeiros National Park (CVNP) and **B:** Entre Rios Private Farm (ERPF). Site's name consists of its location plus the year of the start of restoration intervention. Rectangles indicate first to third quantile; dots indicate outliers and colours indicate significantly different groups. Species are organized in the following functional groups: annual grass (AG), short-lived sub-shrub (SLSS), fast-growing perennial grass (FGPG), slow- growing perennial grass (SGPG), fast-growing shrub/tree (FGST), slow-growing shrub/tree (SGST), invasive exotic grass (IEG). Colours represent age/number of rainy seasons since restoration intervention.

4. Discussion

4.1 Restoration trajectories

Restoration trajectories are still poorly understood in restoration ecology of open tropical ecosystems. While species turnover has long been described in forest ecosystems (i.e: *'succession'*), open-canopy ecosystems have no general trajectory model established and are often treated as degraded forests (Veldman et al., 2015; Silveira et al., 2020). In these areas, successional studies are often associated with woody encroachment processes (e.g. Durigan and Ratter, 2006; Raymundo et al., 2023). The understanding of ecosystems assembly trajectories, though vital for restoration ecology, is not easily acquired since extended periods of repetitive sampling are needed (Suding, 2011). In this sense, the present study is an important contribution to the development of such knowledge, by summarizing data for up to 12 years of restoration in a Neotropical Savannah to shed light on restoration trajectories.

We observed that desired restoration trajectories were only present in sites where IEG growth was limited. Although, restoration sites followed mostly similar trajectories (Fig. 2), with annual grasses and sub-shrubs drastically reducing their cover while slow-growing grasses surpassed fast-growing grasses. The dominance of invasive exotic grasses (IEG) hindered this process. In heavily invaded sites, species turnover took longer or did not happen at all. Likewise, diversity and richness values decreased with IEG dominance increase (Fig.3). Such results were expected since IEG are known to exclude native species from sites where they are established due to their higher competitiveness (D'Antonio and Meyerson, 2002). Other studies had already reported failures to restoration interventions due to lack of IEG control both in Neotropical Savannahs (Coutinho et al., 2019; Sampaio et al., 2019) and in other open-canopy ecosystems (Holl et al., 2014; Brooks et al., 2010).

Fast to slow-growing turnover was not observed for shrubs and trees in our sampled sites, in up to 12 rainy seasons after seeding (Fig. 2). Even in the older batch, where high tree cover was achieved (CVNP_2012), established species were mainly fast-growing, planted in high density. This was due to the lack of knowledge on seed germination and seedling establishment of native species during the first year of direct seeding implementation as an experimental restoration technique. As a result, the CVNP_2012 batch had a higher tree density than reference sites and the following seeding efforts were implemented with a significantly smaller quantity of tree seeds in the seed mix. The persistence of fast-growing trees and shrubs, even after 12 rainy seasons, suggests that the Savannah woody strata may change slower than the grassy layer. That is expected since Savannah trees and shrubs are usually slow-growing and concentrate their growth and biomass in the root system during early establishment periods (Veldman et al., 2015; Bond and Keeley, 2005; Buisson et al., 2019; Pilon and Durigan, 2013). Adding to that, many slow-growing species typical of Savannahs are not yet used in direct seeding due to lack or low seed production (Pilon et al., 2023).

4.2 Native comparisons

Reaching native reference values is a daunting task for restoration (Hobbs et al., 2006; Suding, 2011). Past studies on Savannah have estimated the time for recovery of a native Savannah function, structure, and diversity to be more than 1000 years (Nerleklar & Veldman 2020). Despite that, this study showed that some values can be reached rather quickly. In our study, restoration sites were able to reach native reference values for grass cover and diversity in early years and keep it for 11 years (Fig. 3 and 4). The recovery of the grassy layer is an important part of open-canopy biomes restoration. Historically undervalued such biomes have long been restored as forest and their grassy layer ignored, which is a form of degradation through afforestation (Silveira et al., 2022; Veldman et al., 2015). In the Neotropical Savannah of South America, grasses were only included in restoration projects after 2012 (Pellizzaro et al., 2017, Sampaio et al., 2019).

Though our study sites reached the diversity values of native reference sites (Fig. 4), species composition was vastly different (Supplementary table 1). For trees and shrubs, specifically, diversity was low compared with native sites. While restoration sites had high Fabaceae cover of over 30%, the most dominant woody species of native sites was Myrtaceae with a cover of 3%. These results highlight the need to include more woody species in restoration projects, especially slow-growing shrubs and *Undergrounders* (species with great investment in underground tissues, see. Pilon et al., 2021). These species represent an important functional group of the Cerrado helping fire resistance, soil structure and water balance with a highly developed root system (Oliveira et al., 2005, Pilon et al., 2021). The introduction of such species, though important, requires further studies on native species propagation and establishment, since their seed production is generally low, which makes seed harvesting a laborious and non-profitable process (Pilon et al., 2023). Despite this difference from neighbouring reference areas, when compared with average cover for Cerrado open physiognomies, areas did not differ so much. Many of the tree species common in cerrado sensu stricto areas were present in our seed mixes and established in the restoration sites. Similarly, Poaceae and Cyperaceae are both common families in grasslands in the Cerrado region and were dominant in our restoration areas (Ribeiro and Walter, 1998).

We found a similar problem regarding species composition in the herbaceous layer. Though grasses are introduced by direct seeding in high diversity, forbs and other non-woody families important to the Cerrado (such as *Xyridaceae, Eriocaulaceae* and *Verbenaceae,* e.g. de Souza et al., 2021) are underrepresented in restoration sites. Again, the low availability of seeds and the lack of knowledge about species reproduction and establishment is the main constraint to the introduction of these species in restoration initiatives (Pilon et al., 2023; Schmidt et al., 2019). Studies in the African Savannah have highlighted the importance of forbs for the fauna and for the provision of ecosystem services such as herbivore foraging and belowground carbon sequestration (Siebert et al., 2024). In the Neotropical Savannah knowledge on the ecological role of native forbs is still incipient, but the high diversity of this group found on local scale may be a strong indicator of the importance of this group to the structure of native communities (de Souza et al., 2021). Restoration by direct seeding was able to establish a high number of species of important and representative genuses of Neotropical Savannahs and allowed for colonization of restoration sites by other native species. This indicates that restoration sites are able to establish an initial community capable of hosting late-arrivals and increasing biodiversity through time.

4.3 IEG control and persistence

In our study, soil fertility and chemical weeding were both successful in controlling IEG and allowing high native species cover, turnover, richness, and diversity inside restoration sites. However, while acidic and infertile soils kept such values on the long term, IEG quickly reached high cover values after the interruption of management in the chemically weeded sites (Fig. 5). The problem is not the efficiency of the control method, as chemical weeding has been a widespread method to control exotic grass species all over the world (Kettenring and Adams, 2011; Weidlich et al., 2020). This difference is likely because chemical weeding, with post-germination herbicide, controls only IEG plants already established. Lower soil fertility and higher acidity, on the other hand, acted as filters from seed germination to seedling establishment and growth. This filter gives an advantage to native species that have evolved on the acidic and infertile soils common to the Cerrado region and allows for them to compete with IEG for dominance (Sampaio et al., 2019; Silveira et al., 2020). Studies in native areas have had compelling results with the addition of N fertilizer in native areas. Nitrogen was correlated with a reduction in exotic colonization and native slow-growing grasses dominance (Bustamante et.al., 2012). Desired restoration trajectories are, therefore, more easily achieved on unaltered soils that have kept their natural acidity and have not been limed for agricultural use.

The catastrophic effects of liming on restoration trajectories can be highlighted by the difference between the ERPF_2013 and the CVNP_2013 batches. Seeded in the same year and with mostly the same species, one with acidic and infertile soil and the other limed, these batches had vastly different IEG covers after seeding. While in the ERPF batch, exotic cover stabilized at roughly 40%, this was not a pattern observed in the CVNP batch, where IEG cover continued to increase reaching values as high as 90% after nine rainy seasons (Fig. 2). The tendency for exotic grasses to return and increase can be attributed to an acquired resilience of the invaded system. Soil alteration by liming and exotic establishment probably pushed sites past their native threshold (Suding, 2011; Brooks et al., 2010; Hobbs et al., 2006; Silveira et al., 2020). We presented here the first results on the trajectory of Neotropical Savannah restoration with early chemical weeding as a method of IEG control. Other studies have been testing topsoil removal and soil acidification to tip the trajectory back towards a desired native-like state, with some positive results, although not yet applicable in large-scale especially due to logistics and high economic costs (Lira-Martins et al., 2024; Thomas et al., 2019). Shading is used in forest ecosystems but cannot be applied for Savannahs characterized by their open canopies (Veldman et al., 2015). In such conditions, current knowledge is still insufficient for exotic exclusion or permanent control.

This is a problem since funding for restoration projects across the globe are, in general, of short time spans and expect final results as early as two to five years with no additional management accounted for. As a result, restoration is often done over short timespans (Barbosa-Dias et al., 2024; Borgström et al., 2016). In the Neotropical Savannah, it is not much different, since the Brazilian legislation expects results for restoration in four years (Brasil, 2024). Our results, however, show that when IEG are dominant, especially due to previous soil liming or fertilization, managing biological invasions requires more time and continuous management.

Contrary to our expectations, fire did not seem to significantly increase invasive exotic grasses cover both on acidic and infertile and on limed soil (Fig. 6). While the results for ERPF_2013 might have been expected due to the natural limitation for exotic growth provided by soil conditions, the results for the CVNP_2016 burned site were not. A previous study (Giles et al., 2024) on the same site found an increase in exotics after fire, which contrasts with what we observed. The difference is probably due to the natural increase of invasive exotic grasses, measured in other restoration sites not hit by wildfires. By sampling the site one year before the fire and one year after the fire and comparing exotic cover but not comparing the trajectory of this site to other sites under restoration Giles et al. (2024) failed to consider the natural tendency of IEG to increase between different years on any given restoration site in these same soil types. Using samples from the restoration sites a few months before the fire events and considering a larger time spam from a large number of restoration sites, we were able to conclude that the differences previously observed might not be due to the occurrence of fire itself, but a result of the natural trajectory of invasive exotic grasses dominance, which, according to our results, was not significantly affected by the wildfire.

The resilience to fire in Savannah restoration is a necessity for the success of restoration sites. Such ecosystems burn naturally and have evolved with fire as an endogenous disturbance (Miranda et al., 2009; Simon and Pennington, 2012). Therefore, a common goal for the restoration of these sites is to reestablish a vegetation community resilient to the reintroduction of such disturbances (Bond and Keeley, 2005; Buisson et al., 2019; Silveira et al., 2020). Natural savannah fire regime consists of lighting fires that frequently occur during the transition from dry to rainy-season due to the accumulation of dry biomass (Alvarado et al., 2019). Current knowledge is insufficient to testify if the restored ecosystems are resilient to fire. But our data indicate that the current species composition reintroduced via direct seeding might establish plant communities that may acquire resilience to fire events in the first years after restoration interventions, even in the presence of invasive exotic grasses. Corroborating this view, another study on Cerrado restoration sites has shown that seedlings of trees survive fire events as early as two years after direct seeding. Although they lost their aboveground structures to fire, seedlings of 17 woody species were able to resprout from the roots and most recovered their previous height one year after fire (Tinoco, 2017).

The numerous challenges faced by ecosystem restoration highlight the importance of ecosystem conservation. Despite returning selected aspects, services and diversity of historical ecosystems, original species composition and richness is lost and cannot be recovered after ecosystem conversion/degradation. That is especially true considering the widespread of human altered ecosystems and Climate change. Restoration is an amazing tool for aiding recovery and for mitigating degradation effects, but it doesn't reverse the damage that has already been made (Hobbs et al., 2009). Adding to this, restoration is an auxiliary tool and cannot be successful without conservation since conserved areas serve as propagle fonts and seed harvesting areas for restoration projects (Wiens and Hobbs 2015; Montenegro 2024). Currently only 3% of the Cerrado is effectively protected inside protected areas and more than 50% of its original area has already been converted (Françoso et al., 2015; MapBiomas, 2025). National legislation has set an ambitious restoration goal of 5Mha restored till 2030, despite that there are no goals for a proportional increase in conservation areas (Planaveg, 2024). This is a problem because Cerrado is currently under severe conversion pressure especially for pasture conversion with exotic grasses (Durigane Ratter). Adding to this there is a worldwide limitation on knowledge for open ecosystems that are often treated as degraded forests . Therefore, to efficiently conserve these biomes more studies on their ecology and characteristics are needed (BAD, Zaloumis and Bond, 2016; Veldman).

This study monitored the plant community trajectory established through direct seeding in the oldest and largest restoration sites in the Neotropical Savannah. We used data with different numbers of plots and subplots and even different sampling methods but could clearly identify plant community trajectories. Since we sampled sites where restoration and management practices changed over time due to the advances in knowledge on the subject, there were no proper control groups for the herbicide application factor. In the same sense, the fire events were not planned and lacked proper control groups. However, the large time frame and number of study sites allowed for the identification of clear patterns of plant community trajectories in Neotropical Savannahs and their drivers.

4.4 Conclusions

Restoration site trajectories revealed that we are able to successfully recover both diversity and grass cover of native sites of Neotropical Savannahs via direct seeding. Longer-term monitoring efforts are needed to describe the trajectory of woody species since we expect woody cover to reestablish after a few more decades. Invasive grasses were found to be more harmful to ecosystem trajectories than previously thought (Wiederhecker et al., 2024). The control of these species is still a challenge as they continue to recolonize sites from the seedbank even after chemical control was performed in the first years of restoration.

In the opposite direction, the effects of fire were less detrimental to the native species seeded in the study sites than anticipated. This result reinforces the importance of species selection to restore open-canopy ecosystems, which are naturally exposed to wildfires, and that more studies should be done to understand the response of both native and exotic species to different fire regimes.

The challenges to improve restoration success in Neotropical Savannahs are multiple. In this comprehensive sampling of restoration sites trajectories, we highlight that native species reintroduction, and initial succession can be achieved with the species pool available for restoration through direct seeding, presently. However, there is a need to increase species diversity and functional groups, especially considering slow-growing forbs and shrubs (i.e.

Resprouters and *Undergrounders,* Pilon et al., 2021). In addition, the results presented here highlight the importance of invasive exotic grasses management to ensure restoration sites follow a trajectory towards native-dominated plant communities.

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Conclusão Geral

Ao estudar as maiores e mais antigas áreas de restauração em escala operacional de savanas brasileiras conhecidas, mostramos que a semeadura direta introduz de forma eficaz diferentes grupos funcionais de plantas. Essa abordagem permitiu a colonização de espécies nativas e a substituição gradual na comunidade, com espécies de crescimento rápido e ciclo de vida curto dominando até o terceiro ano posterior à semeadura. Após esse período, espécies nativas de crescimento lento e ciclo de vida longo aumentaram gradualmente sua dominância. Condições de manejo e solo influenciaram a montagem da comunidade e a dominância de gramíneas exóticas, o que, por sua vez, impactou a substituição dos grupos funcionais de espécies nativas. Embora árvores tenham se estabelecido com sucesso, outras formas de vida, especialmente gramíneas nativas, foram severamente reduzidas em áreas dominadas por gramíneas exóticas invasoras (IEG, na sigla em inglês).

As trajetórias das áreas de restauração revelaram que é possível recuperar tanto a diversidade quanto a cobertura de gramíneas nativas de savanas neotropicais por meio da semeadura direta. Embora mais estudos sejam necessários, espera-se que a cobertura arbórea também se restabeleça nas próximas décadas. Apesar disso, o estabelecimento de espécies nativas mostrou-se amplamente ineficaz para prevenir a re-infestação por IEG, especialmente em áreas onde essas gramíneas dominam a paisagem. Assim, o controle efetivo de IEG é essencial para aumentar o sucesso dos esforços de restauração no Cerrado brasileiro, uma vez que o estabelecimento de espécies nativas de diferentes formas de vida por meio da semeadura direta é viável em escala operacional utilizando técnicas e insumos atualmente disponíveis. O controle de gramíneas invasoras, contudo, continua sendo um desafio, pois essas espécies persistem recolonizando áreas mesmo após intervenções químicas, já que a recolonização ocorre prioritariamente por germinação do banco de sementes ou pela dispersão de sementes vindas de áreas dominadas por IEG na paisagem.

Ao contrário do que prevíamos, a ocorrência de incêndios foi menos prejudicial para as espécies nativas do que o esperado, indicando a necessidade de mais estudos para entender melhor a resposta tanto de espécies nativas quanto de exóticas em áreas de restauração, aos regimes de fogo das savanas. Esse quadro ressalta a imensa importância da conservação dos ecossistemas naturais. Isso porque, mesmo os projetos de restauração mais bem sucedidos não recuperam a biodiversidade perdida pela degradação ou conversão dos ambientes naturais.

Praticantes e pesquisadores de restauração têm muitos desafios importantes pela frente. Para savanas neotropicais invadidas, o controle de longo prazo ou permanente de gramíneas exóticas, permitindo o restabelecimento de espécies nativas e das trajetórias comunitárias, parece ser a maior fronteira a ser superada.

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Apêndice A - Supplementary Material Chapter 1

Supplementary Table 1: Seeded and Observed species (excluding 26 non-identified species) in all restoration sites in Chapada dos Veadeiros National Park (CVNP), Entre Rios Private Farm (ERPF) and Lake Descoberto Preservation Area (LDPA). Scientific names were written according to Flora do Brasil (2020). Origin column indicates if species were seeded (S), natural generation (NR) or both (S/NR).

Life Form	Family	Scientific Name	Common Name	Origin
Sub-shrub	Asteraceae	*Achyrocline satureioides (Lam.) DC.*	Macela	S/NR
Shrub	Lamiaceae	*Aegiphila verticillata Vell.*	Milho-de-grilo	NR
Tree	Fabaceae	*Amburana cearensis (Allemão) A.C.Sm.*	Amburana	S
Shrub	Anacardiaceae	*Anacardium humile A.St.-Hil.*	Cajuzinho	S
Tree	Fabaceae	*Anadenanthera colubrina (Vell.) Brenan*	Angico	S
Herb	Poaceae	*Andropogon bicornis L.*	Capim-vassoura	S/NR
Herb	Poaceae	*Andropogon fastigiatus Sw.*	Andropoguinho	S
Herb	Poaceae	*Andropogon leucostachyus Kunth*	Capim-mulungu	S/NR
Shrub	Annonaceae	*Annona L.*	Annona sp.	NR
Herb	Poaceae	*Aristida gibbosa (Nees) Kunth*	Capim-carrapato	S
Herb	Poaceae	*Aristida longifolia Trin.*	capim-aristida	S
Herb	Poaceae	*Aristida riparia Trin.*	Capim-rabo-de -burro	S
Tree	Apocynaceae	*Aspidosperma macrocarpon Mart. & Zucc.*	Peroba	S
Tree	Apocynaceae	*Aspidosperma tomentosum Mart. & Zucc.*	Peroba-do-cerrado	S
Tree	Anacardiaceae	*Astronium fraxinifolium Schott*	Aroeira	S/NR

Herb	Poaceae	*Axonopus aureus P. Beauv.*	Capim-pé-de-galinha	S
Herb	Poaceae	*Axonopus pellitus (Nees ex Trin.) Hitchc. & Chase*	Axonopus pellitus	NR
Herb	Poaceae	*Axonopus pressus (Nees ex Steud.) Parodi*	Axonopus pressus	NR
Herb	Poaceae	*Axonopus siccus (Nees) Kuhlm.*	Capim-colonião-nativo	S/NR
Sub-shrub	Asteraceae	*Baccharis dracunculifolia DC.*	Alecrim-do-campo	NR
Tree	Fabaceae	*Bauhinia dumosa Benth.*	Pata-de-vaca	S/NR
Sub-shrub	Fabaceae	*Bauhinia L.*	Bauhinia sp.	NR
Tree	Fabaceae	*Bowdichia virgilioides Kunth*	Sucupira-preta	S
Tree	Moraceae	*Brosimum gaudichaudii Trécul*	Mama-cadela	S/NR
Tree	Combretaceae	*Terminalia corrugata (Ducke) Gere & Boatwr.*	Pau-pilão	S/NR
Shrub	Malpighiaceae	*Byrsonima basiloba A.Juss.*	Murici	NR
Shrub	Malpighiaceae	*Byrsonima coccolobifolia Kunth*	Murici-rosa	NR
Shrub	Malpighiaceae	*Byrsonima Rich. ex Kunth*	Byrsonima sp.	NR
Shrub	Fabaceae	*Cajanus cajan (L.) Huth*	feijão-guandu	NR
Sub-shrub	Fabaceae	*Calliandra dysantha Benth.*	Caliandra	NR
Herb	Fabaceae	Calopogonium mucunoides Desv.	Calopogônio	R
Shrub	Salicaceae	*Casearia sylvestris Sw.*	Caféznho-do-mato	NR
Herb	Poaceae	*Cenchrus echinatus L.*	Cenchrus echinatus	NR
Sub-shrub	Fabaceae	*Centrosema (DC.) Benth.*	Centrosema sp.	NR
Sub-shrub	Fabaceae	*Chamaecrista (L.) Moench*	Chamaecrista sp.	NR

Sub-shrub	Fabaceae	*Chamaecrista (L.) Moench*	Chamaecrista sp.2	NR
Sub-shrub	Asteraceae	*Chresta Vell. ex DC.*	Chresta sp.	NR
Sub-shrub	Asteraceae	*Chresta Vell. ex DC.*	Chresta sp.1	NR
Herb	Commelinaceae	*Commelina L.*	Commelina sp.	NR
Sub-shrub	Asteraceae	*Conyza bonariensis (L.) Cronquist*	Buva	NR
Tree	Fabaceae	*Copaifera langsdorffii Desf.*	Copaíba	S/NR
Sub-shrub	Fabaceae	*Crotalaria L.*	Crotalaria sp.	S/NR
Sub-shrub	Fabaceae	*Crotalaria L.*	Crotalaria sp.1	NR
Sub-shrub	Fabaceae	*Crotalaria L.*	Crotalaria sp.2	NR
Sub-shrub	Euphorbiaceae	*Croton antisyphiliticus Mart.*	Pé-de-perdiz	NR
Tree	Bignoniaceae	*Cybistax antisyphilitica (Mart.) Mart.*	Ipê-verde	S
Herb	Cyperaceae	Cyperaceae Juss.	Cyperaceae sp.	NR
Herb	Cyperaceae	Cyperaceae Juss.	Cyperaceae sp.1	NR
Herb	Cyperaceae	Cyperaceae Juss.	Cyperaceae sp.2	NR
Herb	Cyperaceae	Cyperaceae Juss.	Cyperaceae sp.3	NR
Herb	Cyperaceae	Cyperaceae Juss.	Cyperaceae sp.4	NR
Herb	Cyperaceae	Cyperaceae Juss.	Cyperaceae sp.5	NR
Herb	Cyperaceae	Cyperaceae Juss.	Cyperaceae sp.6	NR
Herb	Cyperaceae	Cyperaceae Juss.	Cyperaceae sp.7	NR
Herb	Cyperaceae	Cyperaceae Juss.	Cyperaceae sp.8	NR
Herb	Cyperaceae	Cyperaceae Juss.	Cyperaceae sp.9	NR
Herb	Cyperaceae	Cyperaceae Juss.	Cyperaceae sp.10	NR
Herb	Cyperaceae	Cyperaceae Juss.	Cyperaceae sp.11	NR

Tree	Fabaceae	*Dalbergia miscolobium Benth.*	Jacarandá-do-Cerrado	S/NR
Tree	Fabaceae	*Dimorphandra mollis Benth.*	Faveiro	S/NR
Tree	Fabaceae	*Dipteryx alata Vogel*	Baru	S
Herb	Poaceae	*Echinolaena inflexa (Poir.) Chase*	Capim-flexinha	S/NR
Tree	Metteniusaceae	*Emmotum nitens (Benth.) Miers*	Sobre	S
Tree	Fabaceae	*Enterolobium gummiferum (Mart.) J.F.Macbr.*	Tamboril-do-cerrado	S/NR
Herb	Poaceae	*Eragrostis Wolf*	Eragostris sp.	NR
Herb	Poaceae	*Eragrostis Wolf*	Eragostris sp.1	NR
Herb	Poaceae	*Eragrostis Wolf*	Eragostris sp.2	NR
Herb	Eriocaulaceae	Eriocaulaceae Martinov	Eriocaulaceae sp.	NR
Herb	Eriocaulaceae	Eriocaulaceae Martinov	Eriocaulaceae sp.1	NR
Tree	Malvaceae	*Eriotheca pubescens (Mart.) Schott & Endl.*	Paineira-do-Cerrado	S/NR
Tree	Myrtaceae	*Eugenia dysenterica (Mart.) DC.*	Cagaita	S/NR
Tree	Malvaceae	*Guazuma ulmifolia Lam.*	Mutamba	S
Tree	Bignoniaceae	*Handroanthus chrysotrichus (Mart. ex DC.) Mattos*	Ipê-amarelo	NR
Tree	Bignoniaceae	*Handroanthus ochraceus (Cham.) Mattos*	Ipê-amarelo	S/NR
Tree	Bignoniaceae	*Handroanthus serratifolius (Vahl) S.Grose*	Ipê-amarelo	NR
Tree	Fabaceae	*Hymenaea courbaril L.*	Jatobá da Mata	S
Tree	Fabaceae	*Hymenaea stigonocarpa Mart. ex Hayne*	Jatobá do Cerrado	S

Herb	Poaceae	*Hyparrhenia bracteata (Humb. & Bonpl. ex Willd.) Stapf*	Jaraguá nativo	NR
Tree	Bignoniaceae	*Jacaranda brasiliana (Lam.) Pers.*	Caroba	S/NR
Sub-shrub	Bignoniaceae	*Jacaranda ulei Bureau & K.Schum.*	Carobinha	S/NR
Tree	Calophyllaceae	*Kielmeyera coriacea Mart. & Zucc.*	Pau-santo	S
Tree	Lythraceae	*Lafoensia pacari A.St.-Hil.*	Pacari	NR
Sub-shrub	Asteraceae	*Lepidaploa aurea (Mart. ex DC.) H.Rob.*	Amargoso	S
Herb	Poaceae	*Loudetiopsis chrysothrix (Nees) Conert*	Capim-brinco-de-princesa	S
Tree	Fabaceae	*Machaerium opacum Vogel*	Jacarandá-do-campo	S
Tree	Sapindaceae	*Magonia pubescens A.St.-Hil.*	Tingui	S/NR
Sub-shrub	Malvaceae	Malvaceae Juss.	Malvaceae sp.	NR
Tree	Euphorbiaceae	*Maprounea guianensis Aubl.*	Marmeleiro-do-campo	NR
Sub-shrub	Melastomataceae	Melastomataceae A.Juss.	Melastomataceae sp.	NR
Shrub	Melastomataceae	*Miconia albicans (Sw.) Steud.*	Canela-de-velho	NR
Shrub	Melastomataceae	*Miconia Ruiz & Pav.*	Miconia sp.	NR
Shrub	Melastomataceae	*Miconia Ruiz & Pav.*	Miconia sp.1	NR
Shrub	Fabaceae	*Mimosa arenosa (Willd.) Poir.*	Mimosa arenosa	NR
Shrub	Fabaceae	*Mimosa claussenii Benth.*	Mimosa	S
Shrub	Fabaceae	*Mimosa invisa Mart. ex Colla*	Mimosa	NR
Shrub	Fabaceae	*Mimosa L.*	Mimosa sp.	NR
Shrub	Fabaceae	*Mimosa L.*	Mimosa sp.1	NR

Shrub	Primulaceae	*Myrsine guianensis (Aubl.) Kuntze*	Capororoca	NR
Sub-shrub	Myrtaceae	Myrtaceae Juss.	Myrtaceae sp.	NR
Herb	Eriocaulaceae	*Paepalanthus Mart.*	Paepalanthus sp.	S/NR
Herb	Poaceae	*Panicum campestre Nees ex Trin.*	Panicum campestre	S/NR
Herb	Poaceae	*Panicum L.*	Panicum sp.	NR
Herb	Poaceae	*Panicum L.*	Panicum sp.1	NR
Shrub	Chrysobalanaceae	*Parinari obtusifolia Hook.f.*	Fruta-de-ema	NR
Herb	Poaceae	*Paspalum L.*	Paspalum sp.	NR
Herb	Poaceae	*Paspalum L.*	Paspalum sp.1	NR
Herb	Poaceae	*Paspalum multicaule Poir.*	Paspalum multicaule	NR
Herb	Poaceae	*Paspalum stellatum Humb. & Bonpl. ex Flüggé*	capim-orelha-de-coelho	S/NR
Sub-shrub	Malvaceae	*Pavonia sidifolia Kunth*	Pavonia sidifolia	NR
Tree	Asteraceae	*Piptocarpha rotundifolia (Less.) Baker*	Cambará-d-campo	NR
Tree	Fabaceae	*Plathymenia reticulata Benth.*	Vinhático	S/NR
Shrub	Melastomataceae	*Pleroma candolleanum (Mart. ex DC.) Triana*	Quaresmeira	S/NR
Herb	Asteraceae	*Praxelis diffusa (Rich.) Pruski*	Praxelis diffusa	NR
Tree	Burseraceae	*Protium ovatum Engl.*	Almécega	NR
Sub-shrub	Myrtaceae	*Psidium L.*	Psidium sp.	NR
Herb	Cyperaceae	*Rhynchospora Vahl*	Rhynchospora sp.	NR
Sub-shrub	Rubiaceae	Rubiaceae Juss.	Rubiaceae sp.	NR
Herb	Poaceae	*Schizachyrium sanguineum (Retz.) Alston*	Capim-roxo	S

Shrub	Fabaceae	*Senna alata (L.) Roxb.*	Fedegoso	S/NR
Sub-shrub	Fabaceae	*Senna Mill.*	Senna sp.	NR
Shrub	Fabaceae	*Senna rugosa (G.Don) H.S.Irwin & Barneby*	Senna rugosa	S/NR
Herb	Poaceae	*Setaria parviflora (Poir.) Kerguélen*	Setaria parviflora	R
Sub-shrub	Malvaceae	*Sida rhombifolia L.*	mata-pasto	NR
Sub-shrub	Malvaceae	*Sida L.*	Sida sp.	R
Sub-shrub	Malvaceae	*Sida L.*	Sida sp.1	R
Sub-shrub	Malvaceae	*Sida L.*	Sida sp.2	R
Tree	Simaroubaceae	*Simarouba Aubl.*	Simarouba sp.	NR
Sub-shrub	Smilacaceae	*Smilax goyazana A.DC.*	Smilax goyazana	NR
Shrub	Solanaceae	*Solanum L.*	Solanum sp.	NR
Shrub	Solanaceae	*Solanum lycocarpum A.St.-Hil.*	Lobeira	S
Tree	Loganiaceae	*Strychnos pseudoquina A.St.-Hil.*	Quina-do-cerrado	NR
Tree	Fabaceae	*Stryphnodendron adstringens (Mart.) Coville*	Barbatimão	S
Sub-shrub	Fabaceae	*Stylosanthes Sw.*	*Stylosanthes spp.	S
Tree	Bignoniaceae	*Tabebuia aurea (Silva Manso) Benth. & Hook.f. ex S.Moore*	Ipê-caraíba	S/NR
Tree	Fabaceae	*Tachigali vulgaris L.G.Silva & H.C.Lima*	Carvoeiro	S/NR
Tree	Combretaceae	*Terminalia argentea Mart. & Zucc.*	Capitão	S
Herb	Poaceae	*Trachypogon spicatus (L.f.) Kuntze*	Capim-fiapo	S/NR
Shrub	Turneraceae	*Turnera L.*	Turnera sp.	NR
Tree	Fabaceae	*Vatairea macrocarpa (Benth.) Ducke*	Angelim	S

Herb	Velloziaceae	*Vellozia squamata Pohl*	Canela-de-ema	NR
Shrub	Asteraceae	*Vernonanthura polyanthes (Sprengel) Vega & Dematteis*	Assa-peixe	S/NR

Supplementary Table 2: Soil variables for restoration sites in Lake Descoberto Preservation Area (L), Entre Rios Private Farm (E) and Chapada dos Veadeiros National Park between 2012 to 2014 (C).

Area	**LDPA_ 2018_1**	**LDPA_ 2018_3**	**LDPA_ 2018_4**	**LDPA_ 2018_6**	**PDAEF_2 013_1**	**PDAEF_ 2013_2**	**CVNP_ 2012_1**	**CVNP_ 2012_2**	**CVNP_2 012_3**	**CVNP_ 2013**	**CVNP_ 2014**
Age	4	3	3	3	10	10	10	10	10	9	8
Plowing	1	6	5	5	2	2	1	1	1	2	3
Harrowing	2	0	1	1	0	0	0	0	0	0	0
Herbicide	0	0	0	0	1	0	0	0	0	0	0
Fire	1	0	1	0	0	0	0	0	0	0	1
Introduction	seed and seedling	seeding	seeding	seeding	seed and seedling	seed and seedling	seeding	seeding	seeding	seed and seedling	seed and seedling
Placing	total area	total area	total area	total area	total area	total area	lines	lines	lines	total area	total area
Cattle	0	0	0	0	0	0	0	0	0	0	0
Fire_ after	0	0	0	0	0	0	0	0	0	0	0
pH_in_ water	4	4.18	4.62	5.1	4.28	4.34	4.62	4.65	4.27	4.72	4.65
pH_in_ CaCl2	3.61	3.86	4.17	4.82	3.87	3.85	4.23	4.18	3.87	4.21	4.2
Organic_ Matter	2.85	2.69	3.27	2.85	1.8	1.62	2.6	2.51	1.75	2.64	2.76
P	0.64	0.61	0.65	0.56	0.67	0.55	0.53	0.84	1.4	0.62	0.79

K	109.18	49.5	93.64	58.6	87.7	71.86	47.39	109.18	42.76	46.57	73.07

Area	**LDPA_2018_1**	**LDPA_2018_3**	**LDPA_2018_4**	**LDPA_2018_6**	**PDAEF_2013_1**	**PDAEF_2013_2**	**CVNP_2012_1**	**CVNP_2012_2**	**CVNP_2012_3**	**CVNP_2013**	**CVNP_2014**
Kcmol	0.2792	0.1266	0.2395	0.1499	0.2243	0.1838	0.1212	0.2792	0.1094	0.1191	0.1869
S	1.67	0.98	1.14	0.73	1.14	0.98	1.14	1.08	2.16	1.17	1.49
Ca	0.45	0.31	0.89	2.4	0.52	0.48	0.83	0.19	0.39	0.25	1.16
Mg	0.14	0.13	0.26	0.66	0.42	0.35	0.4	0.14	0.19	0.13	0.46
Al	0.73	0.68	0.5	0.13	1.42	1.31	0.3	2.05	1.1	0.84	0.34
Potential_Acidity	5.69	6.02	5.86	3.47	4.37	4.21	4.79	7.43	4.21	5.2	4.79
CTC	6.6	6.6	7.2	6.6	5.5	5.2	6.1	8.1	4.9	5.7	6.6
%V	13	8	19	48	21	20	22	8	14	8	28
%m	45	56	27	4	55	56	18	76	62	64	16
Ca/Mg	3.2	2.4	3.4	3.6	1.2	1.4	2.1	1.4	2.1	1.9	2.5
Ca/K	1.6	2.4	3.7	16	2.3	2.6	6.8	0.7	3.6	2.1	6.2
Mg/K	0.5	1	1.1	4.4	1.9	1.9	3.3	0.5	1.7	1.1	2.5
Ca/CTC	7	5	12	36	9	9	14	2	8	4	18
Mg/CTC	2	2	4	10	8	7	7	2	4	2	7
K/CTC	4	2	3	2	4	3	2	3	2	2	3
H+Al/T	87	91	81	52	79	81	77	93	86	92	72
B	0.25	0.32	0.33	0.27	0.22	0.19	0.17	0.28	0.15	0.18	0.14
Zn	0.4	0.37	0.54	0.95	0.63	0.51	1.01	0.46	0.36	0.37	0.44
Fe	51.63	55.29	67.19	25.13	132.53	65.66	45.59	80.23	133.83	79.26	48.4

Area	LDPA_ 2018_1	LDPA_ 2018_3	LDPA_ 2018_4	LDPA_ 2018_6	PDAEF_2 013_1	PDAEF_ 2013_2	CVNP_ 2012_1	CVNP_ 2012_2	CVNP_2 012_3	CVNP_ 2013	CVNP_ 2014
Mn	6.92	13.16	11.25	12.65	4.55	4.68	4.21	3.8	0.86	2.97	3.19
Cu	1.05	1.33	1.24	0.62	1.22	0.89	0.63	0.27	0.3	0.35	0.34
Sant	14	14	8	12	15	18	31	27	52	36	28
Silt	39	38	42	43	44	45	30	38	27	29	35
Clay	47	48	50	45	41	37	39	35	21	35	37

Supplementary Table 3: Soil variables for restoration sites in Chapada dos Veadeiros National Park (2015 to 2020).

Area	CVNP_ 2015_1	CVNP_ 2015_2	CVNP_ 2016_1	CVNP_ 2016_2	CVNP_ 2016_3	CVNP_ 2019	CVNP_ 2020_1	CVNP_ 2020_2	CVNP_ 2020_3	CVNP_ 2020_4	CVNP_ 2020_5	CVNP_ 2020_6
Age	7	7	6	6	3	3	2	2	2	2	2	2
Plowing	3	3	2	2	6	2	4	4	4	4	4	4
Harrowing	0	0	4	4	0	3	1	1	1	1	1	1
Herbicide	0	0	0	0	0	1	1	1	1	1	1	1
Fire	1	1	1	1	1	1	1	1	1	1	1	1
Introduction	seeding	seeding	seeding	seeding	seeding	seeding	seeding	seeding	seeding	seeding	seeding	seeding
Placing	total area	total area	total area	total area	total area	total area	total area	total area	total area	total area	total area	total area
Cattle	1	1	1	0	0	0	0	0	0	0	0	0
Fire_ after	0	0	0	1	1	0	0	0	0	0	0	0
pH_in_ water	4.69	4.39	4.49	4.8	4.26	4.39	3.95	4.02	3.99	3.99	4.47	4.99

Area	CVNP_ 2015_1	CVNP_ 2015_2	CVNP_ 2016_1	CVNP_ 2016_2	CVNP_ 2016_3	CVNP_ 2019	CVNP_ 2020_1	CVNP_ 2020_2	CVNP_ 2020_3	CVNP_ 2020_4	CVNP_ 2020_5	CVNP_ 2020_6
pH_in_ CaCl2	4.13	3.71	4.07	4.33	3.74	3.82	3.66	3.6	3.69	3.51	3.92	4.15
Organic_ Matter	2.2	2.22	2.29	3.28	2.07	2.69	3.08	1.98	2.85	1.53	1.71	2.76
P	2.09	0.92	0.62	1.3	1.58	0.88	0.92	1.15	0.78	1.06	1.22	1.77
K	47.18	126.51	48.18	55.36	61.66	95.99	48.11	91.64	70.69	30.74	43.77	35.67
Kcmol	0.1207	0.3236	0.1232	0.1416	0.1577	0.2455	0.123	0.2344	0.1808	0.0786	0.1119	0.0912
S	1.41	1.34	0.88	1.28	1.18	1.31	2.4	2.18	1.37	1.42	1.18	2.65
Ca	0.81	1.09	0.55	0.98	0.12	1.1	0.45	0.2	1.22	0.42	0.59	0.1
Mg	0.36	0.52	0.23	0.44	0.09	0.38	0.11	0.09	0.45	0.15	0.18	0.09
Al	0.88	0.58	0.69	0.46	1.33	0.89	0.79	1.16	0.53	1.07	0.69	1.11
Potential_ Acidity	3.8	3.55	4.54	6.35	4.7	5.36	6.52	4.95	5.69	4.13	3.8	7.1
CTC	5.1	5.5	5.4	7.9	5.1	7	7.2	5.5	7.6	4.8	4.7	7.4
%V	25	35	16	19	8	24	9	9	25	14	19	4
%m	41	23	44	23	76	35	54	70	22	61	44	79
Ca/Mg	2.3	2.1	2.4	2.2	1.2	2.9	4.1	2	2.7	2.8	3.3	1
Ca/K	6.7	3.4	4.5	6.9	0.8	4.5	3.7	0.9	6.7	5.3	5.3	1.1
Mg/K	3	1.6	1.9	3.1	0.6	1.5	0.9	0.4	2.5	1.9	1.6	1.1
Ca/CTC	16	20	10	12	2	16	6	4	16	9	13	1
Mg/CTC	7	9	4	6	2	5	2	2	6	3	4	1
K/CTC	2	6	2	2	3	4	2	4	2	2	2	1

Area	CVNP_ 2015_1	CVNP_ 2015_2	CVNP_ 2016_1	CVNP_ 2016_2	CVNP_ 2016_3	CVNP_ 2019	CVNP_ 2020_1	CVNP_ 2020_2	CVNP_ 2020_3	CVNP_ 2020_4	CVNP_ 2020_5	CVNP_ 2020_6
H+Al/T	75	65	84	80	93	75	90	90	76	86	81	97
B	0.13	0.14	0.16	0.2	0.16	0.27	0.22	0.16	0.2	0.12	0.11	0.15
Zn	0.46	0.38	0.39	0.63	0.68	0.65	0.58	0.56	0.98	0.5	0.48	0.72
Fe	97.44	64.41	103.81	78.75	86.24	140.32	113.35	86.12	88.72	163.72	101.81	69.17
Mn	1.33	1.68	2.47	4.25	1	5.18	3.37	2.08	3.89	1.01	1.27	0.78
Cu	0.4	0.26	0.4	0.37	0.09	0.35	0.18	0.19	0.53	0.22	0.14	0.09
Sant	65	58	40	33	54	6	14	49	27	63	68	27
Silt	14	16	25	32	21	43	39	20	32	14	9	32
Clay	21	26	35	35	25	51	47	31	41	23	23	41

Supplementary Table 4: Values of the *envfit* analysis, correlating soil, landscape and management variables with cover composition in 22 restoration sites in the Brazilian savanna.

Vectors	r2	Pr (>r)
Age	**0.7463**	**0.001 *****
Number of plowing times	0.1973	0.124
Superficial plowing	0.1690	0.175
Organic matter	0.1507	0.209
Phosphorous	0.1462	0.215
Potassium	0.0819	0.440
Kcmol	0.0820	0.439
Sulfur	0.2354	0.071
Calcium	**0.2831**	**0.036 ***
Magnesium	0.2030	0.122
Aluminum	**0.3053**	**0.033 ***
Potential acidity	0.0523	0.606

Vectors	**r2**	**Pr (>r)**
CTC	0.0090	0.904
Base saturation (V%)	**0.2766**	**0.046 ***
Aluminium saturation	0.2503	0.078
Ca/Mg	**0.5370**	**0.003 ****
Ca/K	0.1976	0.115
Mg/K	0.0947	0.404
Ca/CTC	**0.3214**	**0.021 ***
Mg CTC	0.2009	0.128
K CTC	0.0420	0.675
H. Al T	**0.2845**	**0.041 ***
Boron	0.0447	0.665
Zinc	0.0085	0.932
Iron	0.2515	0.065
Manganese	0.0897	0.412
Copper	0.1782	0.163
Sand	0.0737	0.464
Silt	0.1698	0.155
Clay	0.0177	0.846
IEG cover	**0.2950**	**0.043 ***
pH in water	**0.3841**	**0.012 ***
pH in $CaCl_2$	**0.3350**	**0.017 ***
Factors	**r2**	***p value***
Herbicide	**0.2538**	**0.006 ****
Fire	**0.2908**	**0.001 *****
Seedling introduction	0.1424	0.048
Planting form	0.0943	0.132
Cattle	0.0894	0.169
Fire after planting	0.0698	0.237

Apêndice B - Supplementary Material Chapter 2

Supplementary Material S1: Seeded and observed species (excluding 46 non-identified species) in all restoration sites in Chapada dos Veadeiros National Park (CVNP), Entre Rios Private Farm (ERPF) and Lake Descoberto Preservation Area (LDPA). Site name consists of its location plus the year of the start of restoration intervention. Scientific names were written according to Flora do Brasil (2020). Origin column indicates if species was seeded (S), established through natural generation (NR) or both (S/NR) and Sites column indicates where species occurred.

Life Form	Family	Scientific Name	Common Name	Origin	Sites
Herb	Asteraceae	*Acanthospermum hispidum* DC.	Carrapicho de carneiro	NR	ERPF_2013
Herb	Asteraceae	*Praxelis diffusa (Rich.) Pruski*	Praxelis diffusa	NR	PNCV_2012, PNCV_2014, ERPF_2013
Herb	Commelinaceae	*Commelina* L.	Commelina sp.	NR	PNCV_2016, LDPA_2018
Herb	Cyperaceae	Cyperaceae Juss.	Cyperaceae sp.	NR	PNCV_2014, PNCV_2015, PNCV_2020
Herb	Cyperaceae	Cyperaceae Juss.	Cyperaceae sp.1	NR	PNCV_2020, ERPF_2013
Herb	Cyperaceae	Cyperaceae Juss.	Cyperaceae sp.2	NR	PNCV_2014, PNCV_2020, ERPF_2013
Herb	Cyperaceae	Cyperaceae Juss.	Cyperaceae sp.3	NR	PNCV_2015, PNCV_2020, ERPF_2013
Herb	Cyperaceae	Cyperaceae Juss.	Cyperaceae sp.4	NR	PNCV_2016, PNCV_2019, PNCV_2020, LDPA_2018
Herb	Cyperaceae	Cyperaceae Juss.	Cyperaceae sp.5	NR	PNCV_2015, PNCV_2019, PNCV_2020
Herb	Cyperaceae	Cyperaceae Juss.	Cyperaceae sp.6	NR	PNCV_2020
Herb	Cyperaceae	Cyperaceae Juss.	Cyperaceae sp.7	NR	PNCV_2020
Herb	Cyperaceae	Cyperaceae Juss.	Cyperaceae sp.8	NR	PNCV_2020
Herb	Cyperaceae	Cyperaceae Juss.	Cyperaceae sp.9	NR	PNCV_2020
Herb	Cyperaceae	Cyperaceae Juss.	Cyperaceae sp.10	NR	PNCV_2020
Herb	Cyperaceae	Cyperaceae Juss.	Cyperaceae sp.11	NR	PNCV_2020
Herb	Cyperaceae	Cyperaceae Juss.	Cyperaceae sp.12	NR	PNCV_2020
Herb	Cyperaceae	*Rhynchospora consanguinea* (Kunth) Boeckeler	Rhynchospora consanguinea	NR	ERPF_2013
Herb	Cyperaceae	*Rhynchospora rugosa (Vahl) Gale*	Rhynchospora rugosa	NR	PNCV_2015, PNCV_2016
Herb	Cyperaceae	*Rhynchospora* Vahl	Rhynchospora sp.	NR	PNCV_2015, PNCV_2016, PNCV_2019, PNCV_2020, ERPF_2013
Herb	Eriocaulaceae	Eriocaulaceae Martinov	Eriocaulaceae sp.	NR	PNCV_2020
Herb	Eriocaulaceae	Eriocaulaceae Martinov	Eriocaulaceae sp.1	NR	PNCV_2016, LDPA_2018

Herb	Eriocaulaceae	Eriocaulaceae Martinov	Eriocaulaceae sp.2	NR	PNCV_2020
Herb	Eriocaulaceae	Eriocaulaceae Martinov	Eriocaulaceae sp.3	NR	PNCV_2020
Herb	Eriocaulaceae	Eriocaulaceae Martinov	Eriocaulaceae sp.4	NR	PNCV_2020
Herb	Eriocaulaceae	*Paepalanthus* Mart.	Paepalanthus sp.	S/NR	PNCV_2015, PNCV_2020
Herb	Eriocaulaceae	*Paepalanthus* Mart.	Paepalanthus sp.1	S	PNCV_2020
Herb	Poaceae	*Andropogon bicornis* L.	Capim-vassoura	S/NR	PNCV_2012, PNCV_2014, PNCV_2015, PNCV_2019, PNCV_2020, ERPF_2013
Herb	Poaceae	*Andropogon fastigiatus* Sw.	Andropoguinho	S	PNCV_2012, PNCV_2015, PNCV_2016, PNCV_2019, PNCV_2020, LDPA_2018
Herb	Poaceae	*Andropogon* L.	Andropogon sp.	NR	PNCV_2015, PNCV_2016
Herb	Poaceae	*Andropogon leucostachyus* Kunth	Capim-mulungu	S/NR	PNCV_2019, ERPF_2013
Herb	Poaceae	*Aristida gibbosa (Nees) Kunth*	Capim-carrapato	S	PNCV_2012, PNCV_2013, PNCV_2014, PNCV_2015, PNCV_2016, PNCV_2019, PNCV_2020, ERPF_2013, LDPA_2018
Herb	Poaceae	*Aristida* L.	Aristida sp.	NR	PNCV_2012, PNCV_2015, PNCV_2016, PNCV_2019, PNCV_2020, ERPF_2013, LDPA_2018
Herb	Poaceae	*Aristida longifolia* Trin.	capim-aristida	S	PNCV_2015, PNCV_2019, PNCV_2020, ERPF_2013
Herb	Poaceae	*Aristida riparia* Trin.	Capim-rabo-de -burro	S	PNCV_2012, PNCV_2013, PNCV_2014, PNCV_2015, PNCV_2016, PNCV_2019, PNCV_2020, ERPF_2013, LDPA_2018
Herb	Poaceae	*Axonopus aureus P. Beauv.*	Capim-pé-de-galinha	S	PNCV_2012, PNCV_2015, PNCV_2016, PNCV_2019, PNCV_2020, LDPA_2018
Herb	Poaceae	*Axonopus P. Beauv.*	Axonopus sp.	NR	PNCV_2012, PNCV_2016, PNCV_2020, ERPF_2013, LDPA_2018
Herb	Poaceae	*Axonopus P. Beauv.*	Axonopus sp.1	NR	PNCV_2020
Herb	Poaceae	*Axonopus P. Beauv.*	Axonopus sp.2	NR	PNCV_2020
Herb	Poaceae	*Axonopus P. Beauv.*	Axonopus sp.3	NR	PNCV_2020

Herb	Poaceae	*Axonopus pellitus* (Nees ex Trin.) Hitchc. & Chase	Axonopus pellitus	NR	PNCV_2012
Herb	Poaceae	*Axonopus pressus (Nees ex Steud.) Parodi*	Axonopus pressus	NR	PNCV_2016, LDPA_2018
Herb	Poaceae	*Axonopus siccus* (Nees) Kuhlm.	Capim-colonião-nativo	S/NR	PNCV_2012, PNCV_2014, PNCV_2015, PNCV_2016, PNCV_2019, PNCV_2020, ERPF_2013, LDPA_2018
Herb	Poaceae	*Cenchrus echinatus* L.	Cenchrus echinatus	NR	PNCV_2020
Herb	Poaceae	*Echinolaena inflexa (Poir.) Chase*	Capim-flexinha	S/NR	PNCV_2012, PNCV_2013, PNCV_2015, PNCV_2016, ERPF_2013, LDPA_2018
Herb	Poaceae	*Eragrostis* Wolf	Eragostris sp.	NR	PNCV_2015, PNCV_2020
Herb	Poaceae	*Eragrostis* Wolf	Eragostris sp.1	NR	PNCV_2020
Herb	Poaceae	*Eragrostis* Wolf	Eragostris sp.2	NR	PNCV_2020
Herb	Poaceae	*Eragrostis* Wolf	Eragostris sp.3	NR	PNCV_2020
Herb	Poaceae	*Hyparrhenia bracteata* (Humb. & Bonpl. ex Willd.) Stapf	Jaraguá nativo	NR	PNCV_2016, PNCV_2019, PNCV_2020, ERPF_2013, LDPA_2018
Herb	Poaceae	*Loudetiopsis chrysothrix (Nees) Conert*	Capim-brinco-de-princesa	S	PNCV_2012, PNCV_2013, PNCV_2014, PNCV_2015, PNCV_2016, PNCV_2019, PNCV_2020, LDPA_2018
Herb	Poaceae	*Megathyrsus maximus (Jacq.) B.K.Simon & S.W.L.Jacobs*	Capim-mombaça	NR	PNCV_2016, LDPA_2018
Herb	Poaceae	*Panicum campestre Nees ex Trin.*	Panicum campestre	S/NR	PNCV_2012, PNCV_2016, PNCV_2019, PNCV_2020, LDPA_2018
Herb	Poaceae	*Panicum* L.	Panicum sp.	NR	PNCV_2016, PNCV_2020, LDPA_2018
Herb	Poaceae	*Panicum* L.	Panicum sp.1	NR	PNCV_2012, PNCV_2020
Herb	Poaceae	*Panicum* L.	Panicum sp.2	NR	PNCV_2016, LDPA_2018
Herb	Poaceae	*Panicum peladoense* Henrard	Panicum peladoense	NR	PNCV_2016
Herb	Poaceae	*Panicum trichanthum* Nees	Panicum trichanthum	NR	PNCV_2016
Herb	Poaceae	*Paspalum* L.	Paspalum sp.	NR	PNCV_2016, PNCV_2020, ERPF_2013, LDPA_2018
Herb	Poaceae	*Paspalum* L.	Paspalum sp.1	NR	PNCV_2016, ERPF_2013, LDPA_2018

Herb	Poaceae	*Paspalum multicaule* Poir.	Paspalum multicaule	NR	PNCV_2019, PNCV_2020
Herb	Poaceae	*Paspalum stellatum Humb. & Bonpl. ex Flüggé*	capim-orelha-de-coelho	S/NR	PNCV_2020, ERPF_2013
Herb	Poaceae	Poaceae Barnhart	Poaceae sp.	NR	PNCV_2012, PNCV_2013, PNCV_2014, PNCV_2016, ERPF_2013
Herb	Poaceae	Poaceae Barnhart	Poaceae sp.1	NR	PNCV_2012
Herb	Poaceae	Poaceae Barnhart	Poaceae sp.11	NR	PNCV_2020
Herb	Poaceae	Poaceae Barnhart	Poaceae sp.12	NR	ERPF_2013
Herb	Poaceae	Poaceae Barnhart	Poaceae sp.14	NR	PNCV_2020
Herb	Poaceae	Poaceae Barnhart	Poaceae sp.15	NR	PNCV_2020
Herb	Poaceae	Poaceae Barnhart	Poaceae sp.16	NR	PNCV_2020
Herb	Poaceae	Poaceae Barnhart	Poaceae sp.17	NR	PNCV_2020
Herb	Poaceae	Poaceae Barnhart	Poaceae sp.18	NR	PNCV_2020
Herb	Poaceae	Poaceae Barnhart	Poaceae sp.2	NR	PNCV_2012
Herb	Poaceae	Poaceae Barnhart	Poaceae sp.20	NR	PNCV_2013
Herb	Poaceae	Poaceae Barnhart	Poaceae sp.3	NR	PNCV_2012
Herb	Poaceae	Poaceae Barnhart	Poaceae sp.4	NR	PNCV_2012, PNCV_2016, LDPA_2018
Herb	Poaceae	Poaceae Barnhart	Poaceae sp.5	NR	PNCV_2016, LDPA_2018
Herb	Poaceae	Poaceae Barnhart	Poaceae sp.6	NR	PNCV_2016, LDPA_2018
Herb	Poaceae	Poaceae Barnhart	Poaceae sp.7	NR	PNCV_2016, LDPA_2018
Herb	Poaceae	Poaceae Barnhart	Poaceae sp.8	NR	PNCV_2016, LDPA_2018
Herb	Poaceae	Poaceae Barnhart	Poaceae sp.9	NR	PNCV_2019
Herb	Poaceae	*Schizachyrium* Nees	Schizachyrium sp.	S	PNCV_2019, PNCV_2020
Herb	Poaceae	*Schizachyrium sanguineum* (Retz.) Alston	Capim-roxo	S	PNCV_2012, PNCV_2013, PNCV_2014, PNCV_2015, PNCV_2016, PNCV_2019, PNCV_2020, ERPF_2013, LDPA_2018
Herb	Poaceae	*Setaria* P. Beauv.	Setaria sp.	NR	PNCV_2020
Herb	Poaceae	*Setaria parviflora* (Poir.) Kerguélen	Setaria parviflora	NR	PNCV_2019
Herb	Poaceae	*Trachypogon spicatus (L.f.) Kuntze*	Capim-fiapo	S/NR	PNCV_2012, PNCV_2013, PNCV_2014, PNCV_2015, PNCV_2016, PNCV_2019, PNCV_2020, ERPF_2013, LDPA_2018
Herb	Velloziaceae	*Vellozia squamata* Pohl	Canela-de-ema	NR	LDPA_2018

Herb	Xyridaceae	*Xyris Gronov. ex L.*	Xyris sp.	NR	PNCV_2020
Shrub	Annonaceae	*Annona coriacea* Mart.	Araticum	NR	PNCV_2012
Shrub	Annonaceae	*Annona* L.	Annona sp.	NR	PNCV_2012, PNCV_2020, LDPA_2018
Shrub	Annonaceae	*Annona* L.	Annona sp.2	NR	LDPA_2018
Shrub	Asteraceae	*Baccharis* L.	Baccharis sp.	NR	PNCV_2019, PNCV_2020, LDPA_2018
Shrub	Asteraceae	*Lessingianthus bardanoides* (Less.) H.Rob.	*Lessingianthus bardanoides*	NR	LDPA_2018
Shrub	Asteraceae	*Vernonanthura* H.Rob.	Vernonanthura sp.	NR	PNCV_2012, PNCV_2014, PNCV_2016, PNCV_2019, PNCV_2020, LDPA_2018
Shrub	Asteraceae	*Vernonanthura polyanthes* (Sprengel) Vega & Dematteis	Assa-peixe	S/NR	PNCV_2012, PNCV_2014, PNCV_2015, PNCV_2016, PNCV_2019, PNCV_2020, LDPA_2018, PNCV_2013, ERPF_2013
Shrub	Bignoniaceae	*Zeyheria montana* Mart.	Bolsa-de-pastor	S/NR	PNCV_2012, LDPA_2018
Shrub	Bixaceae	*Cochlospermum regium* (Mart. ex Schrank) Pilg.	Algodão-bravo	NR	LDPA_2018
Shrub	Burseraceae	*Protium Burm.f.*	Protium sp.	NR	PNCV_2012
Shrub	Celastraceae	*Salacia crassifolia (Mart. ex Schult.) G.Don*	Bacupari-do-cerrado	NR	PNCV_2016
Shrub	Celastraceae	*Tontelea* Miers	Tontelea sp.	NR	PNCV_2015
Shrub	Chrysobalanaceae	*Parinari obtusifolia* Hook.f.	Fruta-de-ema	NR	ERPF_2013
Shrub	Dilleniaceae	*Davilla elliptica* A.St.-Hil.	Lixeirinha	NR	PNCV_2015
Shrub	Ebenaceae	*Diospyros* L.	Diospyros sp.	NR	PNCV_2016
Shrub	Ebenaceae	*Diospyros lasiocalyx (Mart.) B.Walln.*	Caqui-do-cerrado	NR	PNCV_2016
Shrub	Erythroxylaceae	*Erythroxylum daphnites* Mart.	Chapadinho	NR	PNCV_2012
Shrub	Erythroxylaceae	*Erythroxylum* P.Browne	Erythroxylum sp.	NR	PNCV_2012
Shrub	Euphorbiaceae	*Sapium* Jacq.	Sapium sp.	NR	LDPA_2018
Shrub	Euphorbiaceae	*Sapium obovatum* Klotzsch ex Müll.Arg.	Sapium obovatum	NR	PNCV_2012

Shrub	Fabaceae	*Cajanus cajan (L.) Huth*	feijão-guandu	NR	PNCV_2019
Shrub	Fabaceae	Fabaceae Lindl.	Fabaceae sp.	NR	PNCV_2012, PNCV_2013, PNCV_2015, PNCV_2016, PNCV_2020, ERPF_2013, LDPA_2018
Shrub	Fabaceae	Fabaceae Lindl.	Fabaceae sp.1	NR	PNCV_2015
Shrub	Fabaceae	Fabaceae Lindl.	Fabaceae sp.2	NR	PNCV_2015
Shrub	Fabaceae	*Mimosa arenosa (Willd.) Poir.*	Mimosa arenosa	NR	PNCV_2014
Shrub	Fabaceae	*Mimosa claussenii* Benth.	Mimosa	S	PNCV_2016, ERPF_2013, LDPA_2018, PNCV_2013, PNCV_2014, PNCV_2015, ERPF_2013
Shrub	Fabaceae	*Mimosa invisa* Mart. ex Colla	Mimosa	NR	PNCV_2015, ERPF_2013
Shrub	Fabaceae	*Mimosa* L.	Mimosa sp.	NR	PNCV_2012, PNCV_2013, PNCV_2014, PNCV_2015, PNCV_2016, ERPF_2013
Shrub	Fabaceae	*Mimosa* L.	Mimosa sp.1	NR	PNCV_2016, LDPA_2018, PNCV_2012
Shrub	Fabaceae	*Periandra* Mart. ex Benth.	Periandra	NR	PNCV_2012
Shrub	Fabaceae	*Senna alata (L.)* Roxb.	Fedegoso	S/NR	PNCV_2013, PNCV_2014, PNCV_2016, PNCV_2020, LDPA_2018
Shrub	Fabaceae	*Senna rugosa (G.Don)* H.S.Irwin & Barneby	Senna rugosa	S/NR	PNCV_2012, PNCV_2013, PNCV_2016, PNCV_2020
Shrub	Krameriaceae	*Krameria tomentosa A.St.-Hil.*	Krameria tomentosa	NR	PNCV_2020
Shrub	Lamiaceae	*Aegiphila integrifolia (Jacq.) Moldenke*	Tamanqueiro	NR	LDPA_2018
Shrub	Lamiaceae	*Aegiphila verticillata* Vell.	Milho-de-grilo	NR	PNCV_2016, LDPA_2018
Shrub	Malpighiaceae	*Banisteriopsis* C.B.Rob. ex Small	Banisteriopsis sp.	NR	ERPF_2013
Shrub	Malpighiaceae	*Byrsonima basiloba* A.Juss.	Murici	NR	ERPF_2013
Shrub	Malpighiaceae	*Byrsonima coccolobifolia* Kunth	Murici-rosa	NR	ERPF_2013
Shrub	Malpighiaceae	*Byrsonima* Rich. ex Kunth	Byrsonima sp.	NR	ERPF_2013, PNCV_2015, PNCV_2016, LDPA_2018

Shrub	Malpighiaceae	*Byrsonima verbascifolia* (L.) DC.	Murici-de-tabuleiro	NR	PNCV_2016
Shrub	Melastomataceae	*Miconia albicans* (Sw.) Steud.	Canela-de-velho	NR	ERPF_2013
Shrub	Melastomataceae	*Miconia* Ruiz & Pav.	Miconia sp.	NR	PNCV_2012, ERPF_2013
Shrub	Melastomataceae	*Miconia* Ruiz & Pav.	Miconia sp.1	NR	PNCV_2012
Shrub	Melastomataceae	*Pleroma candolleanum* (Mart. ex DC.) Triana	Quaresmeira	S/NR	PNCV_2020, ERPF_2013
Shrub	Myrtaceae	*Campomanesia adamantium* (Cambess.) O.Berg	Gabiroba	NR	PNCV_2012
Shrub	Myrtaceae	*Campomanesia* Ruiz et Pav.	Campomanesia sp.	NR	PNCV_2012
Shrub	Myrtaceae	*Eugenia bimarginata* DC.	Eugenia bimarginata	NR	ERPF_2013
Shrub	Myrtaceae	*Eugenia punicifolia* (Kunth) DC.	Cereja-do-cerrado	NR	PNCV_2012
Shrub	Ochnaceae	*Ouratea* Aubl.	Ouratea sp.	NR	PNCV_2020
Shrub	Ochnaceae	*Ouratea hexasperma* (A.St.-Hil.) Baill.	Vassoura-de-bruxa	NR	LDPA_2018
Shrub	Primulaceae	*Myrsine guianensis (Aubl.) Kuntze*	Capororoca	NR	PNCV_2012, PNCV_2014, PNCV_2020, ERPF_2013
Shrub	Rubiaceae	*Alibertia* A.Rich. ex DC.	Alibertia sp.	S	ERPF_2013
Shrub	Rubiaceae	*Palicourea officinalis* Mart.	Bate-caixa	NR	PNCV_2012, PNCV_2013
Shrub	Rubiaceae	*Palicourea rigida* Kunth	Bate-caixa	NR	LDPA_2018
Shrub	Salicaceae	*Casearia sylvestris* Sw.	Caféznho-do-mato	NR	PNCV_2012, PNCV_2013, PNCV_2019, PNCV_2020, LDPA_2018
Shrub	Solanaceae	*Solanum* L.	Solanum sp.	NR	PNCV_2012, PNCV_2019, LDPA_2018
Shrub	Solanaceae	*Solanum* L.	Solanum sp.2	NR	LDPA_2018
Shrub	Solanaceae	*Solanum lycocarpum* A.St.-Hil.	Lobeira	S	PNCV_2012, PNCV_2013, PNCV_2014, PNCV_2015, PNCV_2016, PNCV_2020, ERPF_2013, LDPA_2018
Shrub	Turneraceae	*Turnera* L.	Turnera sp.	NR	PNCV_2012

Sub-shrub	Annonaceae	*Annona monticola* Mart.	Annona monticola	NR	PNCV_2012, PNCV_2015, LDPA_2018
Sub-shrub	Apocynaceae	*Apocynaceae* Juss.	Apocynaceae sp.	NR	PNCV_2015
Sub-shrub	Asteraceae	*Achyrocline alata* (Kunth) DC.	Macela	S	PNCV_2015
Sub-shrub	Asteraceae	*Achyrocline satureioides* (Lam.) DC.	Macela	S/NR	PNCV_2012, PNCV_2013, PNCV_2014, PNCV_2019, PNCV_2020, ERPF_2013
Sub-shrub	Asteraceae	*Aldama* La Llave & Lex.	Aldama sp.	S	ERPF_2013
Sub-shrub	Asteraceae	*Aspilia* Thouars	Aspilia sp.	NR	PNCV_2012
Sub-shrub	Asteraceae	Asteraceae Bercht. & J.Presl	Asteraceae sp.	NR	PNCV_2015, ERPF_2013, PNCV_2012, LDPA_2018
Sub-shrub	Asteraceae	Asteraceae Bercht. & J.Presl	Asteraceae sp.1	NR	PNCV_2020, PNCV_2016
Sub-shrub	Asteraceae	Asteraceae Bercht. & J.Presl	Asteraceae sp.2	NR	PNCV_2020
Sub-shrub	Asteraceae	Asteraceae Bercht. & J.Presl	Asteraceae sp.3	NR	PNCV_2014
Sub-shrub	Asteraceae	*Baccharis dracunculifolia* DC.	Alecrim-do-campo	NR	PNCV_2012, ERPF_2013, PNCV_2014, PNCV_2016, PNCV_2019, ERPF_2013, LDPA_2018
Sub-shrub	Asteraceae	*Chresta* Vell. ex DC.	Chresta sp.	NR	PNCV_2019
Sub-shrub	Asteraceae	*Chresta* Vell. ex DC.	Chresta sp.1	NR	PNCV_2020
Sub-shrub	Asteraceae	*Conyza bonariensis* (L.) Cronquist	Buva	NR	PNCV_2015, PNCV_2019, PNCV_2020
Sub-shrub	Asteraceae	*Cosmos* Cav.	Cosmos sp.	NR	PNCV_2012
Sub-shrub	Asteraceae	*Grazielia intermedia* (DC.) R.M.King & H.Rob.	*Grazielia intermedia*	NR	PNCV_2020
Sub-shrub	Asteraceae	*Lepidaploa aurea* (Mart. ex DC.) H.Rob.	Amargoso	S	PNCV_2012, PNCV_2013, PNCV_2014, PNCV_2015, PNCV_2016, PNCV_2019, PNCV_2020, ERPF_2013, LDPA_2018
Sub-shrub	Bignoniaceae	*Anemopaegma arvense* (Vell.) Stellfeld ex de Souza	Catuaba	NR	PNCV_2012
Sub-shrub	Bignoniaceae	*Jacaranda ulei* Bureau & K.Schum.	Carobinha	S/NR	PNCV_2012, PNCV_2013, PNCV_2014, PNCV_2015, PNCV_2016, PNCV_2020, LDPA_2018

Sub-shrub	Celastraceae	*Peritassa* Miers	Peritassa sp.	NR	PNCV_2012
Sub-shrub	Euphorbiaceae	*Croton antisyphiliticus* Mart.	Pé-de-perdiz	NR	PNCV_2013, ERPF_2013
Sub-shrub	Euphorbiaceae	*Croton* L.	Croton sp.	NR	PNCV_2012, ERPF_2013
Sub-shrub	Fabaceae	*Arachis* L.	Arachis sp.	NR	ERPF_2013
Sub-shrub	Fabaceae	*Bauhinia* L.	Bauhinia sp.	NR	PNCV_2012, PNCV_2013, LDPA_2018, PNCV_2016, ERPF_2013
Sub-shrub	Fabaceae	*Calliandra dysantha* Benth.	Caliandra	NR	PNCV_2012, PNCV_2016, LDPA_2018, PNCV_2020
Sub-shrub	Fabaceae	*Centrosema* (DC.) Benth.	Centrosema sp.	NR	PNCV_2016
Sub-shrub	Fabaceae	*Chamaecrista* (L.) Moench	Chamaecrista sp.	NR	PNCV_2015, PNCV_2020, PNCV_2013
Sub-shrub	Fabaceae	*Chamaecrista* (L.) Moench	Chamaecrista sp.2	NR	PNCV_2015
Sub-shrub	Fabaceae	*Crotalaria* L.	Crotalaria sp.	S/NR	PNCV_2012, PNCV_2014, PNCV_2016
Sub-shrub	Fabaceae	*Crotalaria* L.	Crotalaria sp.1	NR	PNCV_2020, PNCV_2012
Sub-shrub	Fabaceae	*Crotalaria* L.	Crotalaria sp.2	NR	PNCV_2020
Sub-shrub	Fabaceae	*Desmodium* Desv.	Desmodium sp.	NR	PNCV_2013
Sub-shrub	Fabaceae	*Eriosema* (DC.) Desv.	Eriosema sp.	NR	PNCV_2016, PNCV_2020, LDPA_2018
Sub-shrub	Fabaceae	*Mimosa* L.	Mimosoideae sp.	NR	LDPA_2018
Sub-shrub	Fabaceae	*Mimosa* L.	Mimosoideae sp. 2	NR	LDPA_2018
Sub-shrub	Fabaceae	*Mimosa L.*	Mimosoideae sp. 3	NR	LDPA_2018
Sub-shrub	Fabaceae	*Senna* Mill.	Senna sp.	NR	PNCV_2020, PNCV_2016
Sub-shrub	Fabaceae	*Stylosanthes* Sw.	*Stylosanthes spp.	S	PNCV_2012, PNCV_2013, PNCV_2014, PNCV_2015, PNCV_2016, PNCV_2019, PNCV_2020, ERPF_2013, LDPA_2018
Sub-shrub	Malpighiaceae	Malpighiaceae Juss.	Malpighiaceae sp.	NR	LDPA_2018
Sub-shrub	Malvaceae	Malvaceae Juss.	Malvaceae sp.	NR	PNCV_2012
Sub-shrub	Malvaceae	*Pavonia sidifolia* Kunth	Pavonia sidifolia	NR	PNCV_2012
Sub-shrub	Malvaceae	*Sida glaziovii* K.Schum.	Sida glaziovii	NR	PNCV_2012, PNCV_2013
Sub-shrub	Malvaceae	*Sida* L.	Sida sp.	NR	PNCV_2012, PNCV_2014, PNCV_2015, PNCV_2020, ERPF_2013
Sub-shrub	Malvaceae	*Sida* L.	Sida sp.1	NR	PNCV_2020, ERPF_2013

Sub-shrub	Malvaceae	*Sida* L.	Sida sp.2	NR	PNCV_2020
Sub-shrub	Malvaceae	*Sida rhombifolia* L.	mata-pasto	NR	PNCV_2012, PNCV_2015, PNCV_2019, ERPF_2013
Sub-shrub	Melastomataceae	Melastomataceae A.Juss.	Melastomatacea sp.2	NR	PNCV_2015
Sub-shrub	Melastomataceae	Melastomataceae A.Juss.	Melastomataceae sp.	NR	PNCV_2012, PNCV_2020, ERPF_2013
Sub-shrub	Myrtaceae	*Eugenia* L.	Eugenia sp.	NR	ERPF_2013
Sub-shrub	Myrtaceae	Myrtaceae Juss.	Myrtaceae sp.	NR	PNCV_2012, LDPA_2018
Sub-shrub	Myrtaceae	Myrtaceae Juss.	Myrtaceae sp.1	NR	PNCV_2012
Sub-shrub	Myrtaceae	Myrtaceae Juss.	Myrtaceae sp.2	NR	LDPA_2018
Sub-shrub	Myrtaceae	*Psidium* L.	Psidium sp.	NR	PNCV_2012, PNCV_2013, PNCV_2020, PNCV_2015
Sub-shrub	Rubiaceae	Rubiaceae Juss.	Rubiaceae sp.	NR	PNCV_2012
Sub-shrub	Rubiaceae	*Sabicea brasiliensis* Wernham	Sangue-de-cristo	NR	ERPF_2013, LDPA_2018
Sub-shrub	Smilacaceae	*Smilax goyazana* A.DC.	Smilax goyazana	NR	PNCV_2012, PNCV_2020
Sub-shrub	Verbenaceae	Lippia L.	Lippia sp.	NR	PNCV_2014
Sub-shrub	Verbenaceae	*Stachytarpheta* Vahl	Stachytarpheta sp.	NR	PNCV_2015
Tree	Anacardiaceae	*Anacardium humile* A.St.-Hil.	Cajuzinho	S	PNCV_2013, PNCV_2014, PNCV_2015, PNCV_2016, LDPA_2018, PNCV_2020
Tree	Anacardiaceae	*Astronium fraxinifolium* Schott	Aroeira	S/NR	PNCV_2012, PNCV_2014, PNCV_2013, PNCV_2015, ERPF_2013
Tree	Anacardiaceae	*Astronium urundeuva* (M.Allemão) Engl.	Aroeira	S/NR	PNCV_2012, LDPA_2018
Tree	Anacardiaceae	*Astronium urundeuva* (M.Allemão) Engl.	Aroeira	S	PNCV_2012
Tree	Anacardiaceae	*Schinus terebinthifolia* Raddi	Aroeira-pimenteira	NR	PNCV_2012
Tree	Annonaceae	*Annona crassiflora* Mart.	Marolo	S	PNCV_2012
Tree	Annonaceae	*Annona montana* Macfad.	Jaca-de-pobre	NR	ERPF_2013,
Tree	Annonaceae	*Cardiopetalum calophyllum* Schltdl.	Imbira	NR	LDPA_2018

Tree	Apocynaceae	*Aspidosperma macrocarpon* Mart. & Zucc.	Peroba	S	ERPF_2013
Tree	Apocynaceae	*Aspidosperma* Mart. & Zucc.	Aspidosperma sp.	NR	ERPF_2013
Tree	Apocynaceae	*Aspidosperma tomentosum Mart. & Zucc.*	Peroba-do-cerrado	S	ERPF_2013, PNCV_2016
Tree	Asteraceae	*Eremanthus glomerulatus* Less.	Candeia	S	PNCV_2016, PNCV_2012, PNCV_2014
Tree	Asteraceae	*Eremanthus* Less.	Eremanthus sp.	NR	PNCV_2016
Tree	Asteraceae	*Eremanthus* Less.	Eremanthus sp.2	NR	PNCV_2016
Tree	Asteraceae	*Eremanthus uniflorus* MacLeish & H.Schumach.	Candeia	S	PNCV_2016, PNCV_2020
Tree	Asteraceae	*Piptocarpha rotundifolia* (Less.) Baker	Cambará-do-campo	NR	PNCV_2014, PNCV_2015, PNCV_2016, PNCV_2020, ERPF_2013, LDPA_2018, PNCV_2012
Tree	Bignoniaceae	*Cybistax antisyphilitica* (Mart.) Mart.	Ipê-verde	S	ERPF_2013
Tree	Bignoniaceae	*Handroanthus chrysotrichus* (Mart. ex DC.) Mattos	Ipê-amarelo	NR	PNCV_2013, PNCV_2019, ERPF_2013
Tree	Bignoniaceae	*Handroanthus ochraceus (*Cham.) Mattos	Ipê-amarelo	S/NR	PNCV_2016, LDPA_2018, PNCV_2013, PNCV_2014
Tree	Bignoniaceae	*Handroanthus serratifolius* (Vahl) S.Grose	Ipê-amarelo	NR	LDPA_2018
Tree	Bignoniaceae	*Jacaranda brasiliana* (Lam.) Pers.	Caroba	S/NR	PNCV_2012, PNCV_2014, ERPF_2013, PNCV_2013, PNCV_2020, LDPA_2018
Tree	Bignoniaceae	*Tabebuia aurea* (Silva Manso) Benth. & Hook.f. ex S.Moore	Ipê-caraíba	S/NR	PNCV_2015, PNCV_2014, PNCV_2016, PNCV_2020
Tree	Burseraceae	*Protium ovatum* Engl.	Almécega	NR	PNCV_2012, LDPA_2018
Tree	Calophyllaceae	*Kielmeyera coriacea* Mart. & Zucc.	Pau-santo	S	PNCV_2015, ERPF_2013
Tree	Combretaceae	*Terminalia argentea* Mart. & Zucc.	Capitão	S	PNCV_2012, PNCV_2014, PNCV_2015

Tree	Combretaceae	*Terminalia corrugata* (Ducke) Gere & Boatwr.	Pau-pilão	S/NR	PNCV_2012, PNCV_2014, PNCV_2015, PNCV_2016, ERPF_2013
Tree	Combretaceae	*Terminalia fagifolia* Mart.	Orelha-de-cachorro	S	PNCV_2015
Tree	Combretaceae	*Terminalia* L.	Terminalia sp.	S	PNCV_2016
Tree	Connaraceae	*Connarus suberosus* Planch.	Arauta-do-campo	NR	PNCV_2016
Tree	Cordiaceae	*Cordia alliodora* (Ruiz & Pav.) Cham.	Freijó	S	PNCV_2014
Tree	Euphorbiaceae	*Maprounea guianensis* Aubl.	Marmeleiro-do-campo	NR	ERPF_2013
Tree	Fabaceae	*Acacia* Mill.	Acacia sp.	S/NR	PNCV_2013
Tree	Fabaceae	*Amburana cearensis* (Allemão) A.C.Sm.	Amburana	S	PNCV_2014, PNCV_2013
Tree	Fabaceae	*Anadenanthera colubrina* (Vell.) Brenan	Angico	S	PNCV_2013, PNCV_2014, ERPF_2013
Tree	Fabaceae	*Bauhinia dumosa* Benth.	Pata-de-vaca	S/NR	PNCV_2012, PNCV_2013, PNCV_2016
Tree	Fabaceae	*Bowdichia virgilioides* Kunth	Sucupira-preta	S	ERPF_2013
Tree	Fabaceae	*Copaifera langsdorffii* Desf.	Copaíba	S/NR	PNCV_2014, PNCV_2012, PNCV_2013, PNCV_2016, ERPF_2013
Tree	Fabaceae	*Dalbergia* L.f.	Dalbergia sp.	NR	PNCV_2012
Tree	Fabaceae	*Dalbergia miscolobium* Benth.	Jacarandá-do-Cerrado	S/NR	PNCV_2012, ERPF_2013, LDPA_2018
Tree	Fabaceae	*Dimorphandra mollis* Benth.	Faveiro	S/NR	ERPF_2013, PNCV_2015, PNCV_2016
Tree	Fabaceae	*Dipteryx alata* Vogel	Baru	S	PNCV_2012, PNCV_2013, PNCV_2014, PNCV_2015, PNCV_2020, ERPF_2013, LDPA_2018
Tree	Fabaceae	*Enterolobium contortisiliquum* (Vell.) Morong	Tamboril	NR	PNCV_2012, PNCV_2013
Tree	Fabaceae	*Enterolobium gummiferum* (Mart.) J.F.Macbr.	Tamboril-do-cerrado	S/NR	PNCV_2012, PNCV_2013, ERPF_2013
Tree	Fabaceae	*Enterolobium* Mart.	Enterolobium sp	NR	PNCV_2016
Tree	Fabaceae	*Hymenaea courbaril* L.	Jatobá da Mata	S	PNCV_2014, PNCV_2012, PNCV_2016, ERPF_2013

Tree	Fabaceae	*Hymenaea stigonocarpa* Mart. ex Hayne	Jatobá do Cerrado	S	PNCV_2012, PNCV_2014, ERPF_2013, PNCV_2020
Tree	Fabaceae	*Inga* Mill.	Inga sp.	NR	PNCV_2016
Tree	Fabaceae	*Leptolobium dasycarpum* Vogel	Chapadinha	NR	PNCV_2012, PNCV_2015, PNCV_2016, LDPA_2018
Tree	Fabaceae	*Machaerium opacum* Vogel	Jacarandá-do-campo	S	PNCV_2015, ERPF_2013, LDPA_2018
Tree	Fabaceae	*Plathymenia reticulata* Benth.	Vinhático	S/NR	PNCV_2012, ERPF_2013
Tree	Fabaceae	*Stryphnodendron adstringens* (Mart.) Coville	Barbatimão	S	PNCV_2013, PNCV_2015, ERPF_2013, PNCV_2016
Tree	Fabaceae	*Tachigali vulgaris* L.G.Silva & H.C.Lima	Carvoeiro	S/NR	PNCV_2012, PNCV_2013, PNCV_2014, PNCV_2016, PNCV_2019, PNCV_2020, LDPA_2018
Tree	Fabaceae	*Vatairea macrocarpa* (Benth.) Ducke	Angelim	S	PNCV_2013, PNCV_2016
Tree	Loganiaceae	*Strychnos pseudoquina* A.St.-Hil.	Quina-do-cerrado	NR	ERPF_2013, LDPA_2018
Tree	Lythraceae	*Lafoensia pacari* A.St.-Hil.	Pacari	NR	PNCV_2012
Tree	Malpighiaceae	*Byrsonima crassifolia* (L.) Kunth	Muric	S	ERPF_2013
Tree	Malvaceae	*Eriotheca pubescens* (Mart.) Schott & Endl.	Paineira-do-Cerrado	S/NR	ERPF_2013, PNCV_2015
Tree	Malvaceae	*Guazuma ulmifolia* Lam.	Mutamba	S	LDPA_2018
Tree	Metteniusaceae	*Emmotum nitens* (Benth.) Miers	Sobre	S	PNCV_2013
Tree	Moraceae	*Brosimum gaudichaudii* Trécul	Mama-cadela	S/NR	LDPA_2018, ERPF_2013
Tree	Myrtaceae	*Blepharocalyx salicifolius* (Kunth) O.Berg	Cambuí	S	PNCV_2016
Tree	Myrtaceae	*Eugenia dysenterica* (Mart.) DC.	Cagaita	S/NR	ERPF_2013, PNCV_2012
Tree	Sapindaceae	*Magonia pubescens* A.St.-Hil.	Tingui	S/NR	PNCV_2012, PNCV_2013, PNCV_2014, PNCV_2016, PNCV_2019, ERPF_2013, LDPA_2018, PNCV_2015

Tree	Simaroubaceae	*Simarouba* Aubl.	Simarouba sp.	NR	PNCV_2020
Tree	Vochysiaceae	*Qualea grandiflora* Mart.	Pau-terra	NR	PNCV_2015
Tree	Vochysiaceae	*Qualea parviflora* Mart.	Pau-terra	NR	ERPF_2013
Vine	Convolvulaceae	Ipomoea L.	Ipomoea sp.	NR	PNCV_2020
Vine	Loranthaceae	*Struthanthus flexicaulis (Mart.) Mart.*	Erva-passarinho	NR	PNCV_2012
Palm Tree	Arecaceae	*Butia archeri* (Glassman) Glassman	Butia archeri	NR	LDPA_2018
Palm Tree	Arecaceae	*Syagrus oleracea* (Mart.) Becc.	Gueroba	NR	PNCV_2012
Palm Tree	Arecaceae	*Syagrus romanzoffiana* (Cham.) Glassman	Jerivá	S	PNCV_2012