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Restauração ecológica em campos invadidos por *Urochloa decumbens* nos Campos Sulinos

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RESUMO

Ecossistemas campestres encontram-se fortemente impactados por conversão de hábitat e por espécies exóticas invasoras. É necessário restaurar os ecossistemas campestres ao redor do globo. Entretanto, para os ecossistemas campestres brasileiros há poucas experiências de restauração e precisamos testar a viabilidade de técnicas normalmente empregadas em outros lugares. O objetivo deste estudo foi testar diferentes combinações de técnicas para restauração de campos invadidos por *Urochloa decumbens* nos Campos Sulinos, sul do Brasil. Combinaram-se duas técnicas de controle da espécie invasora (aplicação de herbicida e remoção superficial de solo) e duas técnicas de introdução de espécies nativas (transposição de feno e semeadura direta). Foram estabelecidos oito blocos em um experimento bifatorial no Morro Santana, Porto Alegre, combinando dois fatores e cada um com três tratamentos (duas técnicas mais o controle). A cobertura de vários grupos de espécies, riqueza de espécies e composição de espécies foram avaliadas por análises de variância, e então também por análise de coordenadas principais. Adicionalmente, a relação entre a cobertura da invasora e das espécies nativas foi investigada. As técnicas de controle da invasora mostraram-se eficientes tanto na redução da cobertura da espécie, como em permitir a entrada de espécies nativas na comunidade. Comparando as duas técnicas, aplicação de herbicida pareceu ser melhor do que a remoção superficial de solo, pois parcelas que tiveram a aplicação tiveram menor cobertura da invasora e maior riqueza de espécies. Já as técnicas de introdução de espécies mostraram-se insuficientes para adicionar espécies nativas na comunidade para competir com *U. decumbens*. Padrões de composição de espécies diferiram entre os tratamentos. Técnicas de controle do invasor diferiram grandemente do seu controle, que foi dominado por *U. decumbens*. Uma clara relação existe entre a cobertura da invasora e a presença e cobertura de espécies nativas. Então o controle da espécie invasora é fundamental para uma maior recuperação da vegetação. Entretanto, os resultados aqui apresentados correspondem a apenas oito meses após a finalização da implementação do experimento, e ações futuras de manejo na área deverão combinar novamente o controle da invasora com introdução de espécies nativas.

PALAVRAS-CHAVE: Ecossistemas campestres subtropicais, remoção de *topsoil*, herbicida, transposição de feno, semeadura, riqueza de espécies, recuperação da vegetação

ABSTRACT

Grasslands ecosystems are strongly impacted by land use and invasive species. It is necessary restore these ecosystems around the world. However, there are few experiences with ecological restoration for the Brazilian grasslands and we need to test the viability of techniques normally used in others grasslands ecosystems. The aim of this study was to test different combinations of techniques to the ecological restoration of grasslands invaded by *Urochloa decumbens* in *Campos*, Southern Brazil. We combine two techniques to control the invasive species (herbicide application and topsoil removal) and two techniques to introduce native species (hay transfer and direct sowing). We established nine blocks in a bi-factorial experiment on Morro Santana, Porto Alegre, Rio Grande do Sul, combining two factors and each one with three treatments (two techniques plus the control). The coverage of various groups of species, species richness, and species composition were evaluated by variance analyses, and the later also by a principal ordination analysis. Additionally, the relation between invader coverage and native species was investigated. Both techniques to control invasive species have shown to be efficient to reduce the coverage of the invasive species, as well as to allow the arrival of native species. Specifically comparing them, the herbicide application seems to be a better treatment than the topsoil removal, once plots with herbicide had lower invasive species coverage and higher species richness. However, the species introduction techniques failed efficiently to add native species to the community composition and to compete with *U. decumbens*. Species composition patterns differed among the treatments. The invader control techniques greatly differed from their control, which was dominated by *U. decumbens*. A clear relationship exists between the invader coverage and the presence and coverage of native species. Thus, the control of the invasive species is fundamental to further vegetation recovery. Nevertheless, results here presented correspond to just eight months from the experiment implantation and future monitoring and management actions on the area should combine once more the control of the invasive species and the introduction of native species.

KEY WORDS: Subtropical grasslands, topsoil removal, herbicide, hay transfer, sowing, species richness, vegetation recovery

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INTRODUÇÃO GERAL

RESTAURAÇÃO ECOLÓGICA

Nos últimos séculos, e mais acentuadamente a partir do século XX, a população mundial e o consumo per capita de recursos aumentaram consideravelmente. Tais crescimentos resultaram em enorme pressão sobre os ecossistemas e a biodiversidade, pondo-os em risco (MEA 2005; Hilderbrand et al. 2005). Tamanho é o impacto sobre a biodiversidade, que resultou no que se considera ser a sexta extinção em massa da história do planeta, e alterações nas funções ecossistêmicas e na provisão de serviços ecossistêmicos ameaçam milhões de pessoas. Chegamos a tal grau de degradação que se compreende que proteger áreas conservadas já não é suficiente, é necessário restaurar as que foram degradadas (Hilderbrand et al. 2005; Gann & Lamb 2011). Diante desse quadro, a restauração ecológica torna-se cada vez mais importante para restabelecer o funcionamento dos ecossistemas, a prestação de serviços ecossistêmicos e a conservação das espécies (Hilderbrand et al. 2005; Funk et al. 2008; ONU 2010; Harris & van Diggelen 2008).

Com o aumento dos impactos humanos, aumentou também o interesse por restaurar os danos que provocamos no meio ambiente (Choi et al. 2008). Entretanto esse interesse não é novo. Em 1861, D. Pedro II ordenou reflorestar a Floresta da Tijuca no Rio de Janeiro, para garantir abastecimento de água na cidade. Ao redor do mundo e ao longo do tempo, esforços de restauração de ambientes degradados tiveram muitas origens e motivações (Galatowitsch 2012), fato que resultou em uma variedade de termos e definições para tais atividades. Desse modo, a Sociedade para Restauração Ecológica (*Society for Ecological Restoration* – SER) definiu “restauração ecológica” como o processo que auxilia a recuperação de um ecossistema que foi degradado, danificado ou destruído. E “ecologia da restauração” é a ciência por trás da prática da restauração ecológica (SER 2004). Ehrenfeld (2000) define que há três tipos de objetivos em restauração ecológica: restauração de espécies, restauração das funções de um ecossistema (ex.: estocagem de carbono, ciclagem de nutrientes) e restauração de serviços ecossistêmicos (ex.: fornecimento de água, controle de erosão).

O uso do termo “restauração” pode sugerir que um projeto de restauração ecológica objetive recriar um estado antigo de um ecossistema, normalmente anterior a uma degradação (Galatowitsch 2012). Porém surgem problemas a partir dessa ideia. Normalmente não sabemos exatamente como o local era antes da degradação (Rodrigues 2013). Há também problemas em definir o que é “natural” devido ao histórico de ocupação humana, tanto no Velho Mundo (Rodrigues 2013), como no Novo e Novíssimo Mundo (Jackson & Hobbs 2009). Por fim, muitas vezes o impacto humano sobre o ambiente é tão profundo que se considera impossível retornar a um estado semelhante ao passado, como em minerações a céu aberto (Howell et al. 2012; Jackson & Hobbs 2009). Pode-se pensar então que uma área conservada próxima da área degradada representaria um estado anterior à degradação, sendo esse o ponto ao qual pretendemos chegar em um projeto de restauração (Rodrigues 2013). Mas há de se ter em mente que essa área conservada está refletindo apenas um momento da sua trajetória e também não sabemos seu histórico. Ela pode apresentar toda uma amplitude de características devido à sua própria dinâmica (SER 2004; Hiers et al. 2011).

Fica claro que manter uma visão estática das comunidades, com composição particular de espécies em abundância, tempo e espaço determinados é irreal (Hobbs et al. 2009). Desse modo, não se deve buscar atingir uma comunidade pré-definida em restauração ecológica. Ao escolher uma área conservada ou um estado passado (quando o conhecemos) devemos usá-los como uma referência para o planejamento e avaliações de projetos de restauração (SER 2004; Howell et al. 2012). Aceitando que não precisamos atingir uma comunidade exata e pré-estabelecida, aumentamos nossas chances de alcançar resultados ainda valiosos poupando boa quantidade de recursos (Hobbs et al. 2009), principalmente quando pretendemos restaurar serviços ecossistêmicos.

A ecologia da restauração tem forte base teórica na ecologia de comunidades, pois esforços de restauração geralmente focam recriar em conjuntos multi-espécies de comunidades (Palmer et al. 1997). Por isso, para ecologia da restauração progredir, além de tratar cada caso isoladamente, é importante saber quais e como funcionam as regras que regulam a estruturação das comunidades, as denominadas regras de montagem (do inglês, *assembly rules*) (Temperton & Hobbs 2004). As regras de montagem definem qual subconjunto de espécies, dado um *pool* regional total, irá ocorrer em um hábitat específico (Keddy 1992). Deve-se ter em mente que projetos de restauração ecológica tentam (re)criar em poucos anos (ou até meses) sistemas que, sob condições naturais, levaram muito tempo para se formar (Hilderbrand *et al.* 2005). A partir disso, manipulando as regras de montagem das comunidades podemos atingir mais rapidamente nossos objetivos (Hulvey & Aigner 2004).

Outro sério problema ambiental decorrente das atividades humanas é a invasão por espécies exóticas (MEA 2005). Espécies invasoras ameaçam a diversidade por predação, competição e alterando condições abióticas (Guido & Guadagnin 2015). Espécies invasoras e áreas degradadas estão intimamente ligadas. Uma área que sofreu ou sofre com algum tipo de degradação ambiental é muito vulnerável a espécies exóticas (Primack 2002; Ziller 2010; Elorza et al. 2004). Tais exóticas podem fácil e rapidamente dominar as comunidades e os recursos disponíveis, atingindo elevada cobertura e restringindo a ocorrência de outras espécies. Assim, áreas degradadas invadidas por exóticas necessitam ações de remoção e/ou controle da abundância das mesmas para que possam se estabelecer processos de restauração ecológica. Porém, ações de remoção de espécies invasoras geralmente criam condições para o estabelecimento da mesma espécie ou de outra espécie exótica (D’Antonio & Meyerson 2002). Por tais motivos, espécies exóticas são um dos maiores desafios em projetos de restauração ecológica (D’Antonio & Meyerson 2002; Suding et al. 2004; Packard & Ross 2005; Funk et al. 2008), e o seu controle deve ser a principal prioridade no manejo desses projetos (D’Antonio & Meyerson 2002).

OS CAMPOS SULINOS

Na região sudeste da América do Sul há vastas áreas de uma paisagem campestre que ajudaram a moldar características comuns na história, economia e cultura entre Argentina, Uruguai e o Sul do Brasil (Fonseca et al. 2013; Vélez-Martin et al. 2015). A figura do gaúcho, típica da região, está associada ao trabalho nesses campos. Entretanto, apenas recentemente esses ecossistemas começaram a receber estudos buscando melhor entendê-los, conservá-los e preservá-los (Behling et al. 2009).

No sul e oeste do Rio Grande do Sul esses campos fazem parte do bioma Pampa, inseridos nos 750.000 km² de *Pastizales Del Río de la Plata*, que se estendem também pela Argentina e Uruguai (Bilenca & Miñarro 2004). Na porção norte do Rio Grande do Sul, em Santa Catarina e no Paraná os campos são classificados como parte do bioma Mata Atlântica, apresentando-se em mosaicos com suas formações florestais (Overbeck et al. 2015). Apesar de diferenças na composição florística entre os campos do Pampa e da Mata Atlântica, um grande número de espécies, principalmente as mais abundantes, ocorrem em ambos. Por esse motivo, segundo Overbeck et al. (2015), é empregado o termo "Campos Sulinos" para referir-se em conjunto aos campos desses dois biomas no sul do Brasil.

Os Campos Sulinos são considerados por Veldman et al. (2015) como campos naturais primários (do inglês, *old-growth grasslands*), por apresentarem alta diversidade de espécies herbáceas e alto grau de endemismo, e podem ser considerados um dos ecossistemas campestres mais diversos do mundo (Bond 2016). Somente para o Rio Grande do Sul, a diversidade de plantas é estimada em 2600 espécies campestres, sendo 2150 no Pampa e 1620 na Mata Atlântica (Boldrini et al. 2015). Uma característica marcante dos Campos Sulinos é a coexistência de gramíneas de metabolismo C3 (típicas de climas mais frios, como da região do rio da Prata) e C4 (de climas mais quentes, como do Cerrado) (Overbeck et al. 2007). Os Campos Sulinos também apresentam uma considerável riqueza de espécies de animais. Há registros de 84 espécies de anfíbios (Iop et al. 2015), 158 espécies de répteis (Verrastro & Borges-Martins 2015), mais de 95 espécies de aves campestres (Fontana & Bencke 2015) e 25 espécies de mamíferos não-voadores (Bencke 2009).

O clima na região dos Campos Sulinos é subtropical úmido, com a temperatura média anual em torno de 16 a 22°C. A precipitação na borda leste do Planalto Brasileiro, onde estão os campos da Mata Atlântica, atinge média anual de 2000 mm e diminui em direção ao sul e interior do continente até a 1300 mm anuais (Overbeck et al. 2015). Não há uma estação de seca definida, mas curtos períodos de estiagem podem ocorrer principalmente na parte sul e oeste do Rio Grande do Sul (Overbeck et al. 2015).

Os Campos Sulinos desenvolveram-se sobre condições climáticas mais frias e secas do que as atuais (Behling & Pillar 2007; Behling et al. 2009). Entretanto, hoje o clima quente e úmido suporta campos de alta produtividade (Bond 2016), e juntamente com as condições edáficas, permite o desenvolvimento e avanço de formações florestais sobre os campos (Pillar & Vélez 2010). Nos Campos Sulinos e demais ecossistemas campestres subtropicais e tropicais de alta produtividade do planeta, a manutenção da estrutura de vegetação campestre é associada ao fogo e à herbivoria (Bond & Keeley 2005; Bond 2016). O crescimento das plantas é mais limitado pelo consumo da biomassa por esses distúrbios do que pela competição por recursos (Bond & Keeley 2005). Ao reduzir a biomassa e a altura das plantas dominantes, os distúrbios mantêm a estrutura aberta dos ecossistemas e a diversidade de espécies herbáceas (Bond & Keeley 2005).

O uso de fogo e a pecuária são tradicionais nos Campos Sulinos, desempenhando importante papel na sua ecologia (Overbeck et al. 2007). Acredita-se que o fogo utilizado pelos ameríndios, tenha sido fundamental ao evitar um avanço mais expressivo das formações florestais sobre as formações campestres (Behling & Pillar 2007). Atualmente, o fogo normalmente é usado no fim do inverno para renovar as pastagens para o gado nos campos do bioma Mata Atlântica (Overbeck et al. 2007). Quanto à presença de herbívoros, os Campos Sulinos parecem ter tido um longo histórico de coevolução com herbívoros pastadores, porém de baixa abundância e que foi interrompido há 8000 anos com a extinção desses (Bencke 2009). A presença de pastadores só foi retomada no século XVII, com a introdução de gado doméstico pelos colonizadores europeus (Pillar & Vélez 2010).

Estudos demonstram que áreas dos Campos Sulinos quando excluídas de distúrbios são dominadas por gramíneas cespitosas (ex.: *Andropogon lateralis* Nees), por arbustos (ex.: *Baccharis* spp e *Calea phyllolepis* Baker) e por arbóreas pioneiras (Boldrini & Eggers 1997; Oliveira & Pillar 2004). Com isso, a fisionomia campestre pode transformar-se em arbustiva ou florestal (Veldman et al. 2015), perde-se a diversidade de espécies herbáceas (Overbeck et al. 2007) e aumenta-se o risco de incêndios catastróficos devido ao acúmulo de biomassa seca (Pillar & Vélez 2010). Tais distúrbios não são importantes somente para a manutenção da diversidade de espécies herbáceas. Bencke (2009) afirma que a conservação da avifauna nestes campos também depende da existência de algum nível de distúrbios nos campos e fisionomias distintas na escala da paisagem. Nos campos planálticos de Córdoba, Argentina, García et al. (2008) constataram que a riqueza e a densidade da avifauna é maior em pastagens pastoreadas do que em áreas com exclusão do gado.

Degradação e atividades de restauração nos Campos Sulinos

Estima-se que na época da chegada dos europeus (séc. XVI), os Campos Sulinos ocupassem 217.819 km^2 (Vélez-Martin et al. 2015). A partir de sensoriamento remoto, calcula-se que 60% dos campos no Rio Grande do Sul tenham sido convertidos (Andrade et al. 2015), principalmente para agricultura e silvicultura (Pillar et al. 2009). A criação de unidades de conservação é o único meio de garantir que estes campos não sejam convertidos em áreas para agricultura e silvicultura. No entanto, devido à sua relação com fogo e pastejo, os Campos Sulinos não podem ser mantidos por áreas de conservação de proteção integral, onde tais distúrbios não são legalmente permitidos (Overbeck et al. 2007). Overbeck et al. (2016) e Pillar & Velez (2010) argumentam que precisamos incluir distúrbios para conservar a estrutura aberta e a diversidade de espécies herbáceas nos Campos Sulinos.

Contudo, embora 60% de conversão de campos já seja um número expressivo, a proporção de áreas degradadas pode ser ainda maior, pois a avaliação por sensoriamento remoto não possibilita perceber degradações em escalas menores, como a invasão de gramíneas exóticas. Espécies de gramíneas africanas de metabolismo C4 foram introduzidas para servir de forragem para o gado (Matthews 2005) e hoje ameaçam a biodiversidade e a produção pecuária, principalmente no Rio Grande do Sul. As principais ameaças são *Eragrostis plana* Nees, *Urochloa* P. Beauv., e *Melinis* P. Beauv. (SEMA 2013). A invasão por *Pinus* spp. e *Ulex europaeus* também preocupam (SEMA 2013).

Para garantir, e inclusive melhorar, a provisão de bens e serviços ecossistêmicos prestados pelos ecossistemas campestres é preciso restaurá-los quando degradados (Gibson 2009). Projetos de restauração de ecossistemas campestres normalmente combinam restaurar o regime de distúrbios e reintroduzir espécies. As técnicas mais comuns para restaurar distúrbios são reintrodução de herbívoros pastadores, roçadas e queimadas controladas. Já as principais técnicas de reintrodução de espécies são semeadura de mistura de sementes, transferência de *topsoil*, transplante de leivas ou transposição de feno (Török et al. 2011; Packard & Ross 2005; Kiehl et al. 2010). Poucos projetos de restauração foram realizados em ecossistemas campestres subtropicais e tropicais (Bond & Parr 2010).

No Brasil, há experiências recentes de restauração no Cerrado e nos Campos Rupestres (Stradic et al. 2013; Stradic et al. 2014; Ferreira et al. 2015). No entanto, nos Campos Sulinos não há experiências de restauração ecológica (Overbeck et al. 2013). É necessário desenvolver técnicas de restauração ecológica a partir do que conhecemos sobre restauração de ecossistemas campestres em outras partes do mundo (Overbeck et al. 2013; Vieira & Overbeck 2015). A restauração ecológica pode ser útil tanto em áreas convertidas, como em áreas invadidas por espécies exóticas.

Nesse contexto, para a conservação dos Campos Sulinos são necessárias medidas que evitem a conversão das áreas remanescentes, a manutenção de fogo e pastejo nas áreas conservadas, a reintrodução de fogo e pastejo naquelas onde foram excluídos, o controle de espécies invasoras e o desenvolvimento de técnicas de restauração ecológica. Os Campos Sulinos têm forte relação com a figura do gaúcho, e conservá-los, além de conservar a biodiversidade e os serviços ecossistêmicos, também é conservar a história e a cultura de milhões de pessoas que os habitam.

Esta dissertação tem como objetivo geral contribuir com o conhecimento de técnicas de restauração de campos invadidos por *Urochloa decumbens*. Para tanto, estamos especificamente testando duas formas de controle da dominância da invasora, uso de Herbicida e retirada dos cinco centímetros superficiais do solo; e duas formas de introdução de espécies nativas, adição de feno fresco de áreas de campo nativo e semeadura de três gramíneas nativas. Os resultados apresentados refletem o monitoramento de oito meses após a finalização da implementação do experimento. Estudos de mais longo prazo, com monitoramento e manejo adaptativo certamente irão contribuir ainda mais para o avanço da restauração ecológica dos Campos Sulinos.

CAPÍTULO 1

Controlling *Urochloa decumbens*: subsides for ecological restoration in invaded Campos grasslands of Southern Brazil

1. INTRODUCTION

Grassy ecosystems around the world have been strongly impacted by habitat conversion and exotic invasive species, affecting biodiversity and ecosystem services (MEA, 2005; Gibson 2009). Hence, restoration of these ecosystems is important to ensure biodiversity conservation, ecosystem functioning and ecosystem services (Clewell & Aronson 2006; Funk et al. 2008; Gibson 2009; ONU 2010). However, degraded areas and exotic species often are closely related, and invasion by exotics is one of the major problems of ecological restoration projects (D’Antonio & Meyerson 2002; Funk et al. 2008). Therefore, invasive species control is a priority in execution and management of restoration projects (D’Antonio & Meyerson 2002).

Restoration of grasslands invaded by exotic grasses is difficult due to several reasons. Invasive grasses can affect the ecological organization from population to the ecosystem level, as they are strong competitors and influence the disturbance regime (D’Antonio & Vitousek 1992). For example, invasion of the neotropical Savanna *Cerrado* by *Urochloa brizantha* alters frequency and intensity of natural fires (Gorgone-Barbosa et al. 2015), which in turn benefits the invaders *U. brizantha* and *U. decumbens* (Gorgone-Barbosa et al. 2016). Another problem is that exotic and native grasses may have similar growth forms, so both respond in a similar way to eradication treatments (Keeley 2015). Thus, eradication treatments normally create conditions for a new invasion by the same or another exotic species (D'Antonio & Meyerson 2002), once disturbances and degradation normally release resources and provide opportunities to the invaders (Davis et al. 2000). Moreover, exotic invasive grasses normally present advantages over native species that allow their spread, such as higher germination potential and fast seedling growth (Baruch & Bilbao 1999), as well as more efficient resource use (Baruch & Bilbao 1999; D'Antonio & Vitousek 1992; Williams & Baruch 2000). Consequently, it is necessary to improve ecological restoration techniques in a way that at the simultaneously control exotic species and favor native species.

To advance in Restoration Ecology it is necessary to advance from treating each case in isolation to understanding the mechanisms and processes that structure biological communities, the *assembly rules* (Temperton & Hobbs 2004; Hulvey & Aigner 2014). Under this perspective, three distinct groups of filters for community assembly can be identified: (1) dispersal filter – limitations that avoid species to reaching a site (Funk et al. 2008; Öster et al. 2009); (2) abiotic filter – environmental conditions that differently interfere the establishment and survive of the species on the site (Cleland et al. 2012); and (3) biotic filter – interspecific interactions that differently interfere the persistence and abundance of the species on the community (Funk et al. 2008; Cleland et al. 2012). To understand the importance of these filters for plant communities in a restoration context helps us to implement management actions that will favor desired species (Thomsen & D'Antonio 2007; Funk et al. 2008; Hulvey & Aigner 2014), as well introduce species that have greater probability to establish itself in the community (Gramn et al. 2015). Understanding assembly rules in restoration projects thus can help to manipulate community development in a way that community resistance to invasive species is increased and development of these species themselves reduced, and so reach quickly our objectives (Funk et al. 2008; Hulvey & Aigner 2004).

However, knowledge on ecological restoration on grasslands are available mainly for temperate grasslands of Europe and North America (Stradic et al. 2013). There are few studies about ecological restoration of subtropical and tropical grasslands, such as the South Brazilian grasslands *Campos* (Overbeck et al. 2013). Differently than temperate grasslands, *Campos* are a high productivity ecosystem (Veldman et al. 2015) associated with high herbaceous diversity, and especially characterized by be compost by a mix of C3 and C4 species (Overbeck et al. 2009). By absence of knowledge and differences to temperate grasslands, we need test the applicability and the long-term efficiency of these techniques to know how restore tropical and subtropical grasslands, such as those that occur in southern Brazil (Overbeck et al. 2013; Vieira & Overbeck 2015).

Campos grasslands in southern Brazil are considered as “old-growth grasslands”, and are among the most biodiverse grassy ecosystems of the world (Veldman et al. 2015). Despite their inestimable ecological and cultural value and besides the potential use for cattle production (Valls et al. 2009), about 60% of the original area of *Campos* has been converted to other land uses (Andrade et al. 2015). Moreover, in the remaining grasslands, exotic species invasions are a major problem, even though few studies have attempted to quantify exotic plant invasions (e.g. Guido et al. 2016). On the invasive species, Campos again differs to temperate grasslands. While C3 grasses invade temperate grasslands, mainly C4 African grasses invade *Campos* C4 African grasses that were initially introduced for forage purposes are among the major threat to the *Campos* ecosystems (Matthews 2005). The principal exotic invasive grasses in the region are *Eragrostis plana* Nees, *Urochloa decumbens* (Stapf) R.D. Webster, and *Melinis minutiflora* P. Beauv. (SEMA 2013).

The aim of this study was to test different techniques of ecological restoration, manipulating dispersal, abiotic and biotic filters, to restore plant species composition in an area invaded by *Urochloa decumbens* in the *Campos*, in southern Brazil. We had two specific objectives: (1) to evaluate the efficiency of two different techniques of *U. decumbens* control in controlling the coverage of *U. decumbens* and in further allow the establishment of native species: application of herbicide and topsoil removal; and (2) to evaluate the efficiency of two techniques to reintroduce native species for grassland plant recover: hay transfer and sowing of native grass species.

2. MATERIAL AND METHODS

2.1 Study area

Fieldwork was carried on Morro Santana hill, Porto Alegre, Rio Grande do Sul, Brazil (30°3'58.27" S, 51°7'46.82" W, 170 m a s l). Climate is subtropical humid (Cfa type according to Köppen classification), with an annual mean temperature of 19.5 °C, cold winter (15.5 °C) and hot summer (24.2 °C) (Inmet 2016). Annual mean precipitation is 1350 mm (Inmet 2016) and there is no dry period. Vegetation on Morro Santana hill is a mosaic of natural grasslands and forests, since the area is located in the transition between the Pampa and the Atlantic Forest biomes. Grassland patches of the hill present species-rich communities, with 430 plants species identified in an area of about 220 ha (Overbeck et al. 2005).

The studied degraded area had been used for farming and livestock grazing probably from the 17th or 18th century, but activities have been suppressed about three decades ago; exact land use history is not known (Fidelis et al. 2012). Today, the area is abandoned and invaded by *Urochloa decumbens*, which likely spread after having been planted and cover approximately 60% of area. Anthropogenic fires still occurs frequently, in intervals of 1 or 2 years. In the experimental area, the last fire occurred 2 months before starting the experiment.

2.2 Experimental design

We established eight blocks with 9 m x 9 m, subdivided in nine plots with 3 m x 3 m, for a two-factorial experiment (Fig. 1). Factor 1 was “Invader control” and Factor 2, “Species introduction”. Each factor has correspondence in each specific objective, respectively. Between February and May 2016, all blocks were mowed and the experiment was carried on.

The “Invader control” factor had three treatments: (1) topsoil removal, (2) herbicide application, and (3) control (only mowing). Topsoil removal treatment consisted on removal of five centimeters of the topsoil to reduce the rhizomes and the seed bank of the invasive *U. decumbens*. To herbicide application treatment, we applied 30 ml of herbicide for each 9 m² mixed with water, at a concentration of 1:10. Herbicide used was glyphosate (*Mademato Dipil Herbicida Glifosato*). Topsoil removal and herbicide application was done in early autumn (April 2016).

The “Species introduction” factor had three treatments: (1) hay transfer, (2) sowing of a mix three native grass species (*Paspalum notatum* Flüggé, *P. guenoarum* Arechav. and *Axonopus affinis* Chase), and (3) control (no species introduction). Hay was collected in two nearby native grassland patches in the Morro Santana hill, on late summer (March 2016). Hay was cut, dried and homogenized. We transferred the equivalent to 600 g/m² of hay in each 9 m² plot. Seeds of *P. guenoarum* and *P. notatum* were obtained from researches of the UFRGS Agronomy Faculty, and seeds of *A. affinis* from a commercial package. We sowing each plot at two times. First time was in early autumn (April 2016). On the four days after this first sowing rained a total of 200 ml, thus we decided to do another sowing in early spring (September 2016). We sowed 1.5 g/m² of *P. guenoarum* and *P. notatum*, and 0.75 g/m² of *A. affinis* at each of the two sowings.

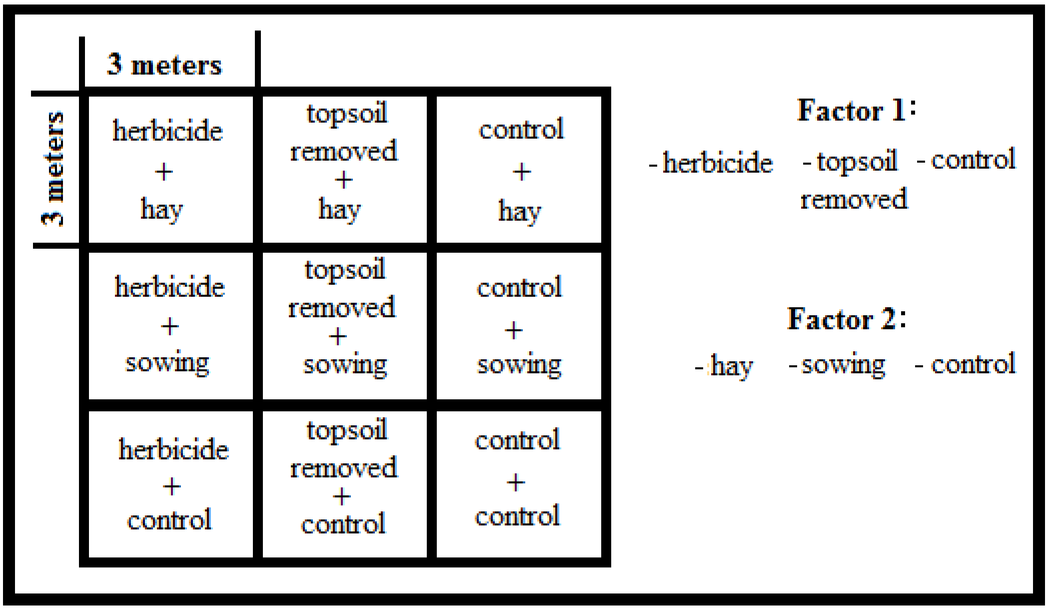

Figure 1. Schematic drawing of a block combining the three treatments of both factors of experimental design.

2.3 Monitoring of vegetation development in experimental plots

Vegetation development was analyzed in three permanent quadrats of 0.5 m x 0.5 m within each plot. Sampling was carried on in late spring (December 2016). All species had their aboveground coverage estimated on a decimal scale (Londo 1976). We also recorded height of vegetation (5 measures per quadrat), percentage of bare soil, standing dry biomass and litter. Species were also classified within in groups based on you origin: “native”, “invasive” (*U. decumbens*) and “exotic” (another’s exotic species that are not invasive in the area).

2.4 Data analyses

Analysis of Variance (ANOVA) with permutation followed by Tukey tests compared the treatment effects on *U. decumbens* coverage, species group coverage and richness. By using the abundance date of the species composition matrix, we tested for differences in species composition with a Multivariate Analysis of Variance (MANOVA), based on the Cord distance as the dissimilarity index and a randomization test with 1000 iterations. All ANOVAS and the MANOVA considered the block level, both factors (‘Invader control’ and ‘Species introduction’), and the interaction between factors. We also evaluated the relations between the invader coverage (predictor) and species richness and coverage of native species (responses) through linear regression modeling. Ordination by principal coordinates analysis (PCoA) was applied to species data sets using chord distance as a resemblance measure. Bootstrap resampling (Pillar 2004) tested ordination axes.

Analyses were done with `vegan` (Oksanen et al. 2017) package on the R platform (R Core Team, 2016), and Multiv (Pillar 2004).

3. RESULTS

3.1 Plant coverage

Eight months after implementation of the treatments, total plant coverage was significantly lower in the herbicide and topsoil removal treatments when compared to the control (only mowing) treatment, while the former two did not differ (Table 1). Species introduction (Factor 2) did not have a significant effect on total plant coverage, and neither did the interaction between both factors (p = 0.14 and 0.13 respectively). These results are clearly related to the *U. decumbens* coverage (Table 1), which had 14% of average coverage. Next species with higher average coverage were *Eryngium ciliatum* (9.60%), *Baccharis psiadioides* (4.43%), *Gamochaeta americana* (2.20%) and *E. horridum* (1.66%), all of them native species (Full species table with average coverage values in Appendix 1).

Native species coverage was high under all treatments and had significance only to invader control factor (Table 1). A decrease of *U. decumbens* coverage leads an increase on native species coverage (Fig. 2). No significant differences occurred between treatments of factor 2, neither the interaction between factors (p = 0.13 and 0.14 respectively).

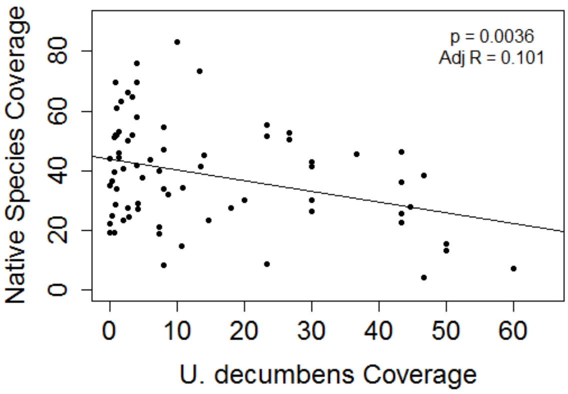

Figure 2. Linear relation between native species coverage in function of the coverage of the invader *Urochloa decumbens*.

The mean species number across all treatments was 17.36 per quadrat of 0.5 m². Species introduction treatments failed to affect species richness, and interaction among factors was not significant too ($p = 0.52$ and 0.22 respectively).

Table 1. Mean values of response variables per treatment according to each factor. Exotic species did not count *U. decumbens* coverage. Different letters mean statistical significance among treatments. ^{ns} $p > 0.05$; * $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$

Factors and treatments	Total plant coverage (%)	Invasive species coverage (%)	Native species coverage (%)	Exotic species coverage (%)	Species Richness (#)
Invader control	***	***	*	ns	***
Herbicide	42.08 ^{a}	2.76 ^{a}	44.64 ^{a}	1.16	19.21 ^{a}
Topsoil removal	37.71 ^{a}	5.83 ^{b}	34.72 ^{b}	1.38	18.04 ^{a}
Control	66.89 ^{b}	33.47 ^{c}	37.19 ^{ab}	2.43	14.96 ^{b}
Species introduction	ns	ns	ns	ns	ns
Hay transfer	46.18	13.47	34.64	1.79	16.41
Sowing	50.42	12.92	42.56	2.39	16.65
Control	52.08	15.67	39.34	0.78	18.87
Interaction	ns	ns	ns	ns	ns
Overall coverage	49.56	14.02	38.85	1.66	17.36

3.2 Structural variables

All structural variables, except litter, responded to the treatments of the invader control factor (Tab. 2). Percentage of bare soil was significantly higher and vegetation height was significantly lower on plots with topsoil removal and herbicide application than on control plots (only mowed). Percentage of dry biomass was significantly lower on topsoil removal plots than on herbicide application and on control plots. Litter and bare soil were also significant for the species introduction factor. Litter percentage was higher on plots with hay transfer and lowest on sowing plots, in counterpoint to the bare soil, which was lower on hay transfer plots (Tab. 2).

Table 2. Mean values to structural variables per treatment according each factor. Different letters means statistical significance among treatments. ^{ns} $p > 0.05$;* $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$

Factors and treatments	Bare Soil (%)	Dry Biomass (%)	Vegetation Height (cm)	Litter (%)
Invader control	***	*	***	ns
Herbicide	31.43 ^{a}	5.95 ^{a}	9.25 ^{a}	19.7
Topsoil removal	40.67 ^{a}	3.42 ^{b}	6.85 ^{a}	18.81
Control	9.17 ^{b}	6.70 ^{a}	15.13 ^{b}	13.53
Species introduction	***	ns	ns	***
Hay transfer	5.85 ^{a}	4.89	9.48	41.26 ^{a}
Sowing	36.70 ^{b}	6.75	11.80	4.90 ^{b}
Control	38.72 ^{b}	4.43	9.95	5.87 ^{b}
Interaction	**	ns	ns	***
Average overall	27.09	5.35	10.41	17.34

3.4 Species Richness

A decrease of *U. decumbens* coverage leads an increase on species richness (Fig. 3) Nevertheless, total plant coverage (R adj = 0.007, p = 0.29) and none of the measured structural variables were related to species richness ($p > 0.05$).

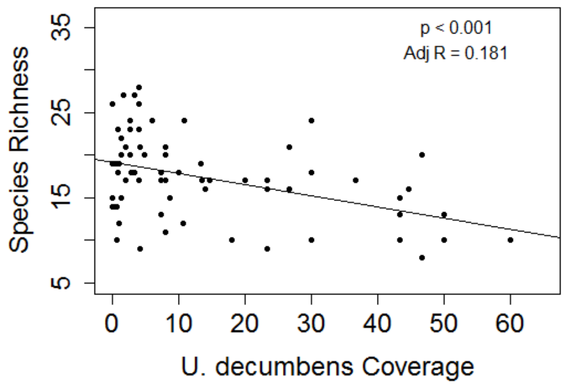

Figure 3. Linear relation between species richness in function of the coverage of the invader *Urochloa decumbens*.

3.6 Species composition

In all plots, a considerable portion of plant coverage was contributed by *U. decumbens* itself, as stated above. Sowed species *P. notatum* and *A. affinis* had low mean coverage values (0.02% and 0.04% respectively), meanwhile *P. guenoraum* was not registered on the plots, i.e. establishment success was very low. Invader control and species introduction factors had a significant effect on species composition ($p < 0.001$), considering all levels, however their interaction did not ($p = 0.18$). Because the interaction between factors was not significant, we opted show two diagram ordination, each one focusing in a factor (Fig. 4 and Fig. 5).

The ordination diagram (Fig. 4) indicates a clear separation according to the invader control treatments, which explained the most variation of the data (axis 1: 30.55%). Control plots are mainly on the left part of the diagram, while plots with herbicide are on the right and topsoil removal are situated in between, with considerable overlap. *U. decumbens* clearly characterize the control plots and *B. psiadioides* was more abundant on plots with herbicide.

These plots and principally those of the topsoil removal also presented many other species, as shown in the diagram (Fig. 4). Meanwhile there is no clear correlation on ordination diagram to introduction species factor (Fig. 5).

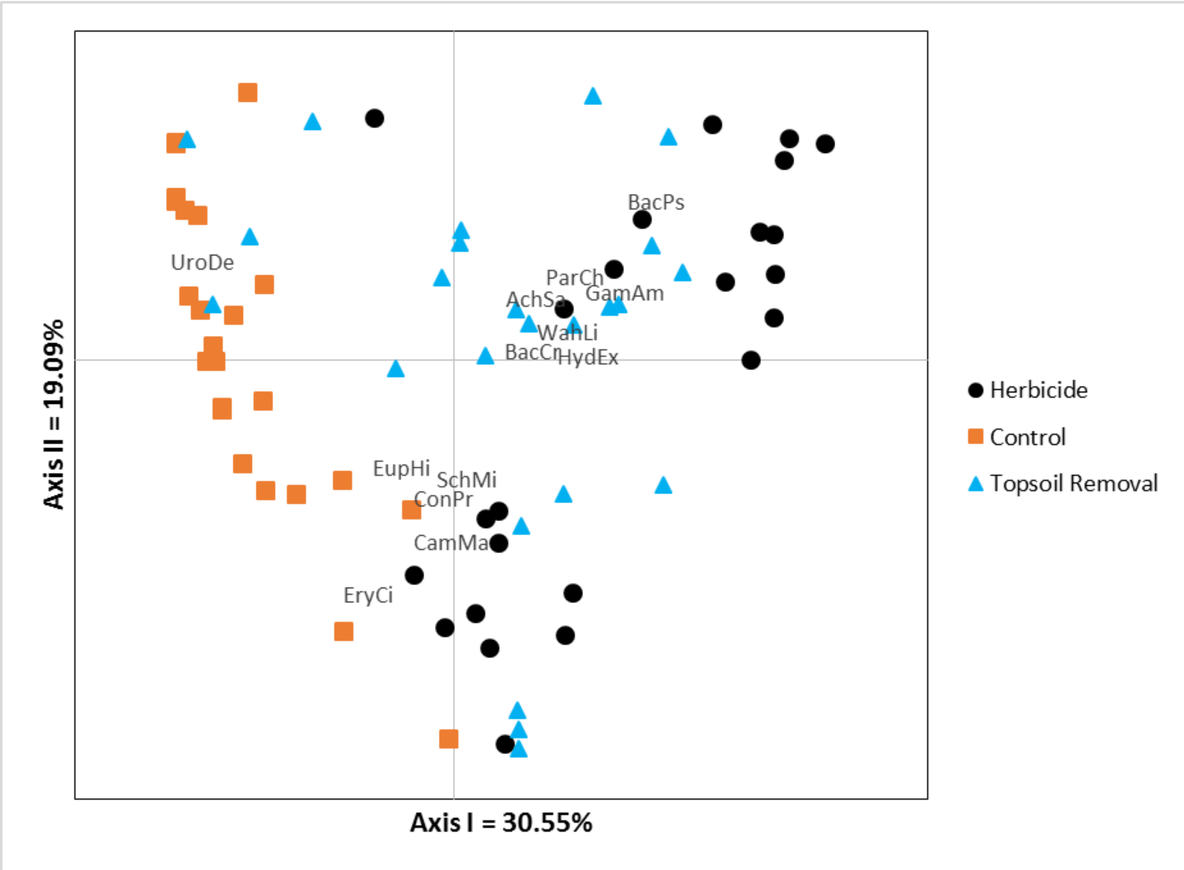

Figure 4. Ordination diagram of the Principal Coordinate Analyses, considering the species abundance and the Cord distance as the measure of plots similarity. The black circles represent sampling units with herbicide application, while orange squares identify control plots (only mowed) and blue tringle topsoil removal plots. Species with 0.30 or more of correlation with one of the two axes are shown: AchSa, *Achyrocline satureioides*; BacCr, *Baccharis crispa*; BacPs, *B. psiadioides*; CamMa, *Campuloclinium macrocephalum;* ConPr, *Conyza primunifolia*; EryCi, *Eryngium ciliatum*; EupHi, *Eupatorium hirsutum*; GamAm, *Gamoachaeta americana*; HydEx, *Hydrocotyle exigua*; ParCh, *Paronichia chilensis*; SchMi, *Schizachyrium microstachyum*; UroDe, *Urochloa decumbens*; WahLi, *Wahlenbergia linarioides*.

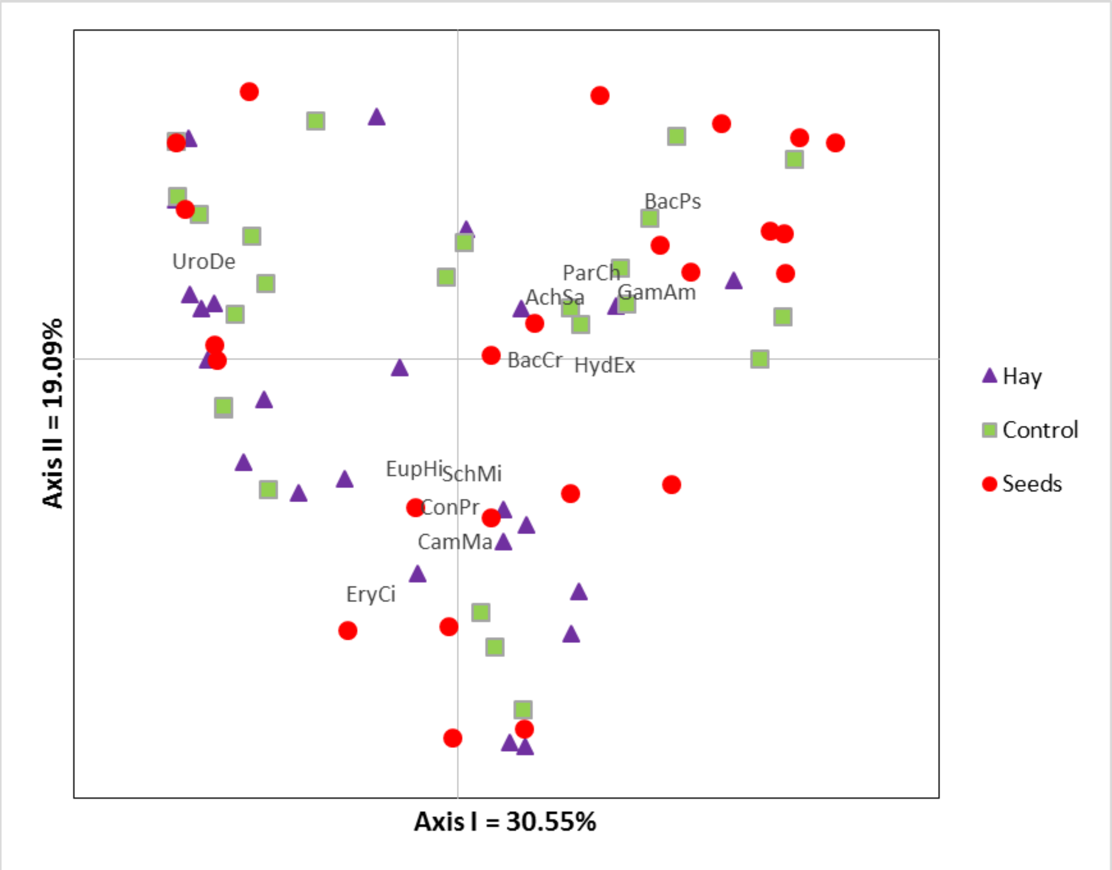

Figure 5. Ordination diagram of the Principal Coordinate Analyses, considering the species abundance and the Cord distance as the measure of plots similarity. Purple triangles correspond to hay transfer plots; Green squares represent control plots (no one action to introduce species); and red circles identify plot with species sowing. Species with 0.30 or more of correlation with one of the two axes are shown: AchSa, *Achyrocline satureioides*; BacCr, *Baccharis crispa*; BacPs, *B. psiadioides*; CamMa, *Campuloclinium macrocephalum;* ConPr, *Conyza primunifolia*; EryCi, *Eryngium ciliatum*; EupHi, *Eupatorium hirsutum*; GamAm, *Gamoachaeta americana*; HydEx, *Hydrocotyle exigua*; ParCh, *Paronichia chilensis*; SchMi, *Schizachyrium microstachyum*; UroDe, *Urochloa decumbens*; WahLi, *Wahlenbergia linarioides*.

4. DISCUSSION

4.1 Invasive species control

Compared to the control plots, both herbicide application and topsoil removal showed some efficiency in reducing coverage of the invasive grass *U. decumbens* (Tab. 1). It is important to keep in mind that we here present only results from the first eight months of study, however initially topsoil removal and herbicide seem had a positive effect on *U. decumbens* control.

Machado et al al. (2012) consider that a single herbicide application was not efficient to control *U. decumbens* and *M. minutiflora* on *Cerrado* neotropical savanna 100 days after application. On the other hand, using topsoil removal too, but in different conditions of degradation in a Mediterranean steppe, Jaunatre et al. (2013) also found good results to improve species richness and avoid seed bank of non-target species with topsoil removal. At the same time, low coverage of *U. decumbens* on topsoil removal plots show that the most part of rhizomes and seeds of this invasive species are present within the first centimeters of soil.

As the removal of the invasive species releases resources, especially space, these management actions could allow a reinvasion (Davis et al. 2000; D'Antonio & Meyerson 2002). Topsoil removal and herbicide could create opportunities for the establishment or enhanced growth of the invasive species due high proportions of bare soil and low height vegetation (Table 2). In our study, the period between experiment implementation and vegetation survey was rather short, thus, despite the conditions for reinvasion by invasive species, we did not observed that. Moreover the short time, the atypical strong winter could explain that, because low temperatures and frosts limit *U. decumbens*. This means that we can only evaluate the initial effects, and probably more management actions will be required to ensure reduction of *U. decumbens* coverage. Keeping in mind that topsoil removal hardly could be applied again, herbicide application seems to be the best management action to control *U. decumbens*.

4.2 Species introduction

At to introduce species by hay transfer and sowing, we expect that the establishment of them help us to control *U. decumbens*. On the other hand, actions to control the invasive species would create conditions to the establishment of introduced species, once conditions with low competition in early stages of succession seem to be essential for establishment of transferred species (Muller et al. 2013). In addition, plots with topsoil removal and herbicide seem to give these conditions, as they had a higher percentage of bare soil and lower vegetation height than the control plots (Table 2).

A review by Hedberg & Kotowski (2013) showed that hay transfer and sowing are the most satisfactory techniques to reintroduce species in grasslands. Many studies report goods results regarding target community establishment by hay transfer (Kiehl & Wagner 2006; Klimkowska et al. 2009; Knut et al. 2010; Schmiede et al. 2011) or combining hay transfer and sowing (Török et al. 2012). However, these studies are from temperate grasslands, and in our experiment, both techniques were not significant to improve species richness of the degraded community.

The grasses species observed with ripe seeds in the reference community at the time of hay collection were *Eragrostis airoides*, *Saccharum angustifolium, Schizachyrium microstachyum, Paspalum. polyphyllum, P. urvillei*, *Andropogon selloanus*. Only the two latter were not registered on the plots after hay transfer, and it becomes clear that this component is difficult to restore, as also found in other studies in tropical grasslands. For instance, testing hay transfer for restoration of *Campos Rupestres* grasslands, Stradic et al. (2013) also did not achieve good results. Low germination rate from species on hay can be due to unfavorable site conditions, the lack of seeds in the hay or the presence of non-viable and dormant seeds on hay (Stradic et al. 2013). How we did not a control of germination of seeds contained on hay, so we cannot say which of these three reasons could be the probable cause of the low germination rate from hay.

Sowing of *P. notatum*, *P. guenoarum* and *A. affinis* was not successful. Seeds of *P. notatum* and *P. guenoarum* used had germination rates of 80% and 70%, respectively, under laboratory conditions (Dall'Agnol, personal communication). Probably the lack of rusticity of the seeds may have influenced the results. The three species seeds were developed to germination in laboratory with specific conditions; therefore, abiotic filters, such as moisture availability, may have impeded germination and/or seedling establishment. Soil preparations might improve efficiency for this technique. There is no information about germination rate of *A. affinis*.

Comparing different weights of seed additions, Goldblum et al. (2013) found bests results adding 5.6 g/m² of seeds. Jones et al. (2013), by sowing 3 g/m², showed that sowing native species is a good technique to restore grasslands. We used a total of 3.75 g/m² of seeds (1.5 g/m² of *P. notatum* and *P. guenoarum*, and 0.75 g/m² of *A. affinis*) and had bad results. However, a lack of commercial native seeds on our region (see Overbeck et al. 2013) avoids the application of this technique with larger seed quantities or another species.

4.3 Richness and species composition

We could increase species richness by *U. decumbens* removal. Despite means species richness were do not differ between herbicide and topsoil removal treatment, herbicide application results in a higher native species coverage and lower *U. decumbens* coverage (Tab. 1). At the same time, less invasive coverage leads a higher native species coverage (Fig. 2) and richness (Fig. 3). Topsoil removal treatment remove too bud and seed banks of native species. Therefore, herbicide application seems be the best technique to control *U. decumbens* and allow the establishment of native species.

However, the application of herbicides may have negative side effects. In native grasslands in the Flooding Pampa region, in Argentina, herbicide had negative impacts on the seed bank, reducing diversity and richness and changing composition (Rodriguez & Jacobo 2013). In their study, herbicide appears to be responsible for the local extinction of several perennial species. Cornish & Burgin (2005) studying possible effects of glyphosate residues in soil when applied as a spray in ecological restoration, found that glyphosate residues in soil might be taken up through roots.

Clear differences of the invader control treatments were observed in the ordination diagram (Fig. 4). This main difference is due to the proportion of *U. decumbens* in the control plots, while many other species (right portion of the diagram) start to increase in abundance on plots where some treatment was applied (herbicide and topsoil removal). Plots where the invader was less dominant in the community already presented many species (mostly Asteraceae) that commonly are considered ruderal, but that are also pioneer in the process of restoration.

How species introduced by hay and sowing had low establishment and coverage, on the ordination diagram to species introduction factor (Fig. 5) we cannot see a clear correlation between introduced species and treatments, neither a clear separation between treatments. So we believe that the principal effect of introduction species treatment to your significant ($p < 0.0001$) is the alteration of biotic conditions by hay transfer and effects of the blocks.

5. CONCLUSION

Our experiment aimed test two distinct ways of re-establishing species of grassland vegetation, besides control an invasive species. We expected that management actions to control the invasive species would create conditions for the establishment of native species; as well, the establishment of native species would help to control the invader by competition. Among the tested techniques, herbicide application showed better results in the short-term for the control of *U. decumbens*. Taking care with native species, new topic applications can be repeated to reduce the coverage of the invasive species until native vegetation can compete with the invasive species. However results found in others studies demonstrate that we need to be careful before suggesting herbicide application. Thus, more studies need be carried out, such as studies of the effects of the herbicide on soil seeds bank and soil biota, before widely recommend the use of herbicide.

Species introduction of the native community seems to be the major challenge. Hay transfer and sowing, two of the techniques easier to do and with best results show to have little efficiency to control *U. decumbens* and to introduce species. Even though we cannot expect that the community will re-establish within a year, recruitment of typical and dominant species – especially grasses – was very low, and the larger part of species that did establish were ruderals (or pioneers). Additional management actions and more time are necessary to increase species establishment. On the other hand, other techniques, such as topsoil transfer and turf transplant might have better results in the area because, beyond introducing species, we are giving better conditions to the establishment (part of the reference soil is transferred as well). Moreover, the collection of ripe seeds from specific species of source sites can also be an alternative technique of introducing desired species.

Due to the little time between management actions and vegetation survey that generated the results here presented, we recognize that continuous vegetation surveys are necessary to better evaluate the treatment efficiencies. This study evaluates only the first restoration phase and probably further management actions for *U. decumbens* control will be necessary.

We improve the technical and scientific knowledge of restoration ecology to *Campos* grasslands. Transposition of biotic and abiotic filters is hard to be effective with the techniques here applied (hay transfer and sowing of grasses). Future ecological restoration projects can focus in overcoming these filters to favor desired species (Thomsen & D'Antonio 2007; Funk et al. 2008; Hulvey & Aigner 2014). A better understanding of filters will allow for more appropriate methods and thus higher restoration success. From applied perspectives, it seems important to focus studies on those species that showed greater establishment even under the initial harsh conditions (Gramn et al. 2015).

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CONSIDERAÇÕES FINAIS

O objetivo geral dessa dissertação era contribuir com o conhecimento de técnicas de restauração de campos invadidos por *Urochloa decumbens* e creio que esse objetivo foi alcançado. Enquanto a aplicação de herbicida e a remoção de 5 centímetros superficiais de solo se mostraram eficientes no controle da espécies invasora e no aumento da riqueza da comunidade, as técnicas de transposição de feno e semeadura de espécies nativas mostraram sem resultados expressivos para ambos objetivos específicos. Entretanto apresentamos aqui resultados de apenas 8 meses após a implementação do experimento. Mais monitoramentos e ações de controle da invasora e adição de nativas serão necessários para que possamos atingir resultados satisfatórios, principalmente quanto à recomposição de uma comunidade campestre baseada em áreas de referência. Outras técnicas e a adaptação dessas aqui utilizadas podem ser testadas para chegarmos a esse objetivo.

Mas o grande objetivo, o de contribuir para o avanço da restauração ecológica dos Campos Sulinos, depende de muito mais. Depende de projetos em longo prazo, o que por sua vez, depende de uma política de estado que dê valor ao meio ambiente e a ciência. É triste constatar que nas esferas estadual e federal, ambas as áreas estão muito longe de ter qualquer importância considerada. Especificamente para o Rio Grande do Sul, o desmonte do Estado e a opção político-econômica trilhados pelas lideranças desvalorizam os Campos Sulinos, ignorando o seu potencial e importância na história e cultura gaúchas.

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Apêndice

Appendix 1: coverage average (%) for each treatment and total coverage average (%) of the species found in grassland plots on Morro Santana hill, Porto Alegre, RS, Brazil. Species marked with * are exotic.

		Life	Factor: Invader control			Factor: Species introduction			
Family	**Species**	**Form**	**Control**	**Topsoil**	**Herbicide**	**Control**	**Hay**	**Seeds**	**Average**
Amaranthaceae	*Pfaffia tuberosa* (Spreng.) Hicken		0.18	0.31	0.49	0.31	0.31	0.38	**0.33**
Apiaceae	*Cyclospermum leptophyllum* (Pers.) Sprague		0.01	0.33	0.67	0.21	0.69	0.10	**0.33**
	Eryngium ciliatum Cham. & Schltdl.		17.72	3.58	7.52	8.40	10.00	10.42	**9.60**
	Eryngium horridum Malme		2.22	0.75	2.01	1.86	1.97	1.15	**1.66**
Araliaceae	*Hydrocotyle exigua* Malme		0.35	1.42	2.03	1.38	1.42	1.00	**1.27**
Arecaceae	*Butia odorata* (Barb.Rodr.) Noblick		0.06	0.00	0.00	0.06	0.00	0.00	**0.02**
Asteraceae	*Achyrocline satureioides* (Lam.) DC.		0.04	0.38	0.59	0.19	0.09	0.72	**0.34**
	Aspilia montevidensis (Spreng.) Kuntze		0.03	0.00	0.00	0.03	0.00	0.00	**0.01**
	Asteraceae 1		0.00	0.06	0.33	0.19	0.06	0.14	**0.13**
	Asteraceae 2		0.03	0.06	0.03	0.06	0.05	0.00	**0.04**
	Asteraceae 3		0.01	0.00	0.03	0.00	0.04	0.00	**0.01**
	Asteraceae 4		0.01	0.00	0.00	0.00	0.00	0.01	**0.00**
	Asteraceae 5		0.00	0.00	0.03	0.00	0.00	0.03	**0.01**
	Asteraceae 6		0.00	0.01	0.01	0.00	0.01	0.01	**0.00**

Asteraceae 7	0.00	0.00	0.01	0.01	0.00	0.00	**0.00**
Asteraceae 8	0.00	0.08	0.00	0.03	0.06	0.00	**0.03**
Baccharis crispa Spreng.	0.05	0.07	0.40	0.17	0.13	0.22	**0.17**
Baccharis dracunculifolia DC.	0.03	0.14	0.11	0.22	0.00	0.06	**0.09**
Baccharis pentodonta Malme	0.01	0.00	0.01	0.00	0.00	0.01	**0.00**
Baccharis psiadioides (Less.) Joch.Müll.	0.92	4.90	7.47	5.35	1.40	6.54	**4.43**
Baccharis spicata (Lam.) Baill.	0.00	0.00	0.03	0.00	0.00	0.03	**0.01**
Calea uniflora Less.	0.00	0.00	0.01	0.00	0.00	0.01	**0.00**
Campuloclinium macrocephalum (Less.) DC.	0.25	0.22	0.69	0.14	0.81	0.22	**0.39**
Chaptalia exscapa (Pers.) Baker	0.01	0.01	0.00	0.01	0.00	0.01	**0.01**
Chaptalia sp.	0.01	0.00	0.00	0.00	0.00	0.01	**0.00**
Chevreulia acuminata Less.	0.01	0.00	0.01	0.01	0.01	0.00	**0.01**
Chromolaena sp.	0.03	0.00	0.00	0.00	0.03	0.00	**0.01**
Chrysolaena cognata (Less.) M. Dematteis	0.00	0.00	0.19	0.14	0.06	0.00	**0.06**
Conyza bonariensis (L.) Cronquist	0.06	0.03	0.56	0.22	0.17	0.25	**0.22**
Conyza floribunda Kunth	0.00	0.00	0.14	0.00	0.00	0.14	**0.05**
Conyza primulifolia (Lam.) Cuatrec. & Lourteig	0.51	0.44	1.18	0.26	1.67	0.20	**0.71**
Conyza sp.	0.00	0.13	0.35	0.40	0.00	0.08	**0.16**

Disynaphia ligulifolia (Hook. & Arn.) R.M.King & H.Rob.	0.13	0.01	0.19	0.13	0.06	0.14	**0.11**
Elephantus mollis Kunth	0.00	0.06	0.06	0.03	0.06	0.03	**0.04**
Eupatorium hirsutum Hook. & Arn.	0.83	0.08	0.72	0.63	0.56	0.45	**0.55**
Eupatorium sp.	0.10	0.03	0.11	0.05	0.12	0.07	**0.08**
Gamochaeta americana (Mill.) Wedd.	0.41	2.35	3.84	1.92	1.81	2.86	**2.20**
Gamochaeta coarctata (Willd.) Kerguélen	0.00	0.01	0.00	0.01	0.00	0.00	**0.00**
Gamochaeta purpurea (L.) Cabrera	0.00	0.06	0.17	0.08	0.01	0.14	**0.08**
Gyptis pinnatifida Cass.	0.03	0.00	0.03	0.00	0.00	0.06	**0.02**
Hierarcium commersonii Monnier	0.00	0.01	0.08	0.08	0.00	0.01	**0.03**
Hypochaeris albiflora (Kuntze) Azevêdo-Gonç. & Matzenb.	0.00	0.00	0.14	0.00	0.14	0.00	**0.05**
Orthopappus angustifolius (Sw.) Gleason	0.98	1.26	0.40	1.03	0.88	0.73	**0.88**
Plantago tomentosa Lam.	0.01	0.15	0.08	0.09	0.03	0.12	**0.08**
Porophyllyum lanceolatum DC.	0.03	0.03	0.06	0.00	0.11	0.00	**0.04**
Pterocaulon polystachyum DC.	0.08	0.23	0.56	0.44	0.17	0.26	**0.29**
Pterocaulon sp.	0.01	0.00	0.01	0.00	0.01	0.00	**0.00**
Senecio heterotrichius DC.	0.00	0.00	0.01	0.00	0.00	0.01	**0.00**
Senecio sellowii (Spreng.) DC	0.00	0.01	0.00	0.00	0.00	0.01	**0.00**
Stenachaenium macrocephalum Benth. ex	0.00	0.00	0.03	0.00	0.03	0.00	**0.01**

	Benth. & Hook.f							
	Vernonia nudiflora Less.	1.17	0.06	0.93	0.76	0.70	0.70	**0.72**
Campanulaceae	*Wahlenbergia linarioides* (Lam.) A.DC.	0.33	0.68	0.87	0.76	0.60	0.52	**0.63**
Caryophyllaceae	*Paronychia chilensis* DC.	0.00	0.08	0.14	0.06	0.02	0.15	**0.07**
	Silene gallica L.	0.29	0.44	0.72	0.31	0.22	0.92	**0.48**
Commelinaceae	*Commelina erecta* L.	0.00	0.00	0.06	0.00	0.06	0.00	**0.02**
Convolvulaceae	*Dichondra sericea* Sw.	0.39	0.42	0.86	0.90	0.65	0.12	**0.56**
	Evolvulus sericeus Sw.	0.00	0.19	0.00	0.03	0.11	0.06	**0.06**
Cyperaceae	*Bulbostylis capillaris* (L.) Kunth ex C.B.Clarke	0.83	1.63	0.19	1.11	0.24	1.31	**0.89**
	Carex sp.	0.00	0.00	0.01	0.01	0.00	0.00	**0.00**
	Cyperaceae 1	0.01	0.01	0.00	0.00	0.01	0.01	**0.00**
	Cyperus aggregatus (Willd.)	1.08	0.63	0.62	0.88	0.89	0.56	**0.78**
	Cyperus reflexus Vahl	0.62	0.11	0.03	0.15	0.22	0.40	**0.25**
	Kyllinga vaginata Lam.	0.03	0.23	0.01	0.09	0.03	0.15	**0.09**
	Rhynchospora sp.	0.00	0.00	0.03	0.00	0.03	0.00	**0.01**
Fabaceae	*Aeschynomene falcata* (Poir.) DC.	0.41	0.02	0.00	0.22	0.22	0.00	**0.14**
	Chamaecrista nictitans (L.)Moench	0.49	0.00	0.00	0.14	0.11	0.24	**0.16**
	Desmanthus tatuhyensis Hoehne	0.77	0.19	0.20	0.26	0.42	0.49	**0.39**
	Desmodium incanum DC.	0.47	0.03	0.00	0.01	0.00	0.49	**0.17**

	Desmodium uncinatum (Jacq.)DC.	0.00	0.00	0.14	0.00	0.14	0.00	**0.05**
	Fabaceae 1	0.00	0.00	0.03	0.03	0.00	0.00	**0.01**
	Macroptilium atropurpureum (DC.)Urb.	0.11	0.00	0.14	0.00	0.11	0.14	**0.08**
	Zornia sp.	0.09	0.00	0.00	0.00	0.00	0.09	**0.03**
Hypoxidaceae	*Hypoxis decumbens* L.	0.00	0.00	0.08	0.03	0.06	0.00	**0.03**
Iridaceae	Iridaceae 1	0.00	0.00	0.03	0.00	0.00	0.03	**0.01**
	Sisyrinchium micranthum Cav.	0.17	0.31	0.44	0.40	0.16	0.37	**0.31**
	Sisyrinchium palmifolium L.	0.32	0.67	0.14	0.28	0.22	0.63	**0.38**
	Sisyrinchium vaginatum Spreng.	0.00	0.03	0.07	0.00	0.01	0.10	**0.03**
Malvaceae	*Sida rhombifolia* L.	0.01	0.09	0.00	0.07	0.03	0.00	**0.03**
	Waltheria communis A.St.-Hil.	0.15	0.12	0.11	0.15	0.23	0.00	**0.13**
Melastomataceae	*Tibouchina gracilis* (Bonpl.) Cogn.	0.28	0.00	0.08	0.36	0.00	0.00	**0.12**
Orobanchaceae	*Agalinis communis* (Cham. & Schltdl.) D'Arcy	0.00	0.00	0.03	0.03	0.00	0.00	**0.01**
Poaceae	*Andropogon leucostachyus* Kunth	0.14	0.00	0.00	0.14	0.00	0.00	**0.05**
	Axonopus affinis Chase	0.00	0.01	0.01	0.00	0.01	0.01	**0.01**
	Calamagrostis viridiflavenscens (Poir.) Steud.	0.01	0.13	1.11	0.40	0.22	0.63	**0.42**
	Chascolytrum subaristatum (Lam.) Desv.	0.11	0.33	0.44	0.28	0.58	0.01	**0.29**
	Chascolytrum uniolae (Nees) L. Essi,	0.00	0.01	0.25	0.11	0.01	0.14	**0.09**

Longhi-Wagner & Souza-Chies							
Dichantelium sabulorum (Lam.) Gould & C.A. Clark	0.06	0.19	0.03	0.06	0.17	0.06	**0.09**
Digitaria eriantha Steud. *	0.25	0.17	0.00	0.06	0.14	0.22	**0.14**
Eleusine indica (L.) Gaertn *	0.19	0.42	0.00	0.31	0.00	0.31	**0.21**
Eragrostis airoides Nees	0.17	1.13	0.99	0.99	0.17	1.12	**0.76**
Eragrostis neesii Trin.	0.06	2.35	0.33	0.98	0.09	1.67	**0.91**
Eragrostis polytricha Nees	0.19	0.40	0.24	0.42	0.13	0.28	**0.28**
Eragrostis sp.	0.00	0.03	0.00	0.00	0.00	0.03	**0.01**
Melinis minutiflora P.Beauv. *	0.70	0.11	0.06	0.08	0.26	0.53	**0.29**
Panicum aquaticum Poir.	0.00	0.01	0.05	0.03	0.01	0.01	**0.02**
Paspalum notatum Flüggé	0.00	0.00	0.06	0.06	0.00	0.01	**0.02**
Paspalum polyphyllum Nees ex Trin.	0.00	0.03	0.08	0.06	0.00	0.06	**0.04**
Phalaris angusta Nees ex Trin. *	0.00	0.08	0.27	0.08	0.00	0.28	**0.12**
Piptochaetium montevidensis (Spreng.) Parodi	1.10	0.95	0.25	0.59	0.78	0.93	**0.77**
Piptochaetium stipoides (Trin. & Rupr.) Hack. & Arechav.	0.00	0.01	0.01	0.01	0.00	0.00	**0.00**
Poaceae 1	0.01	0.00	0.00	0.00	0.00	0.01	**0.00**
Poaceae 2	0.00	0.01	0.03	0.03	0.00	0.01	**0.01**

	Poaceae 3	0.03	0.00	0.01	0.00	0.03	0.00	**0.01**
	Poaceae 4	0.01	0.00	0.00	0.00	0.01	0.00	**0.00**
	Poaceae 5 *	0.12	0.22	0.28	0.06	0.08	0.49	**0.21**
	Poaceae 6	0.00	0.03	0.00	0.00	0.03	0.00	**0.01**
	Saccharum angustifolium (Nees) Trin.	0.20	0.00	0.00	0.00	0.00	0.20	**0.07**
	Schizachyrium microstachyum Desv. ex Ham.) Roseng., B.R. Arrill. & Izag.	0.53	0.57	0.59	0.17	0.92	0.60	**0.56**
	Schizachyrium tenerum Nees	0.00	0.03	0.00	0.00	0.03	0.00	**0.01**
	Setaria parviflora (Poir.) M.Kerguelen	0.25	0.17	0.22	0.25	0.25	0.14	**0.21**
	Setaria vaginata Spreng.	0.00	0.06	0.00	0.02	0.01	0.03	**0.02**
	Steinchisma hians (Elliot) Nash	0.00	0.08	0.01	0.05	0.00	0.04	**0.03**
	Urochloa decumbens (Stapf.) R.D. Webster *	33.47	5.78	2.76	15.63	13.47	12.92	**14.00**
	Urochloa sp. *	1.17	0.01	0.00	0.01	1.17	0.01	**0.39**
Primulaceae	*Anagallis arvensis* L. *	0.00	0.68	0.56	0.50	0.15	0.58	**0.41**
Rubiaceae	*Borreria eryngioides* Cham. & Schltdl.	0.00	0.03	0.06	0.00	0.06	0.03	**0.03**
	Diodia sp.	0.00	0.00	0.01	0.01	0.00	0.00	**0.00**
	Galium hirtum Lam.	0.46	0.72	0.81	0.53	0.90	0.55	**0.66**
	Galium richardianum (Gillies ex Hook. & Arn.) Endl. ex Walp.	0.13	0.06	0.12	0.07	0.13	0.10	**0.10**

	Richardia humistrata (Cham. & Schltdl.) Steud.	0.01	1.41	0.06	0.49	0.41	0.58	**0.49**
	Spermacoce verticilata L.	0.06	0.96	0.11	0.42	0.71	0.00	**0.38**
Solanaceae	*Calibrachoa excellens* (R.E.Fr.) Wijsman	0.17	0.40	0.20	0.35	0.03	0.38	**0.25**
	Petunia integrifolia (Hook.) Schinz & Thell.	0.06	0.40	0.38	0.53	0.01	0.29	**0.28**
	Solanum americanum Mill.	0.00	0.03	0.03	0.00	0.06	0.00	**0.02**
Undertermined	Spp 1	0.03	0.00	0.00	0.00	0.03	0.00	**0.01**
	Spp 2	0.00	0.03	0.00	0.00	0.03	0.00	**0.01**
	Spp 3	0.00	0.03	0.00	0.03	0.00	0.00	**0.01**
Verbenaceae	*Glandularia peruviana* (L.) Small	0.00	0.43	0.01	0.22	0.01	0.21	**0.15**
	Verbenaceae 1	0.00	0.06	0.08	0.00	0.14	0.00	**0.05**