UFRGS
UNIVERSIDADE FEDERAL
DO RIO GRANDE DO SUL

PPGBAN
PROGRAMA DE PÓS-GRADUAÇÃO EM BIOLOGIA ANIMAL

INSTITUTO DE BIOCIÊNCIAS
PROGRAMA DE PÓS-GRADUAÇÃO EM BIOLOGIA ANIMAL

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INFLUÊNCIA DA SILVICULTURA DE *PINUS* NA ÁREA DE OCUPAÇÃO DO LAGARTINHO-PINTANDO *CONTOMASTIX VACARIENSIS* (SQUAMATA: TEIIDAE): UMA RELAÇÃO ENTRE A DISTÂNCIA E A CONSERVAÇÃO.

PORTO ALEGRE
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Dissertação apresentada ao Programa de Pós-Graduação em Biologia Animal, Instituto de Biociências da Universidade Federal do Rio Grande do Sul, como requisito parcial à obtenção do título de Mestre em Biologia Animal.

Área de concentração: Biologia e Comportamento Animal

Orientador(a): Prof. Dra. Laura Verrastro

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Aprovada em _____ de _____ de _____.

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"A felicidade só é verdadeira se for compartilhada"

Christopher McCandless

AGRADECIMENTOS

Às vezes não nos damos conta de quantas pessoas especiais nos rodeiam, e que de alguma forma acabam contribuindo para nosso crescimento. Estas pessoas e as suas várias formas de pensar consolidaram o meu aprendizado para esta dissertação, foi através das "mãos" de vocês que esta conquista foi construída.

Em especial, preciso primeiramente agradecer ao meu amor Nathalia Matias, que no processo da aleatoriedade do destino o mestrado nos colocou lado a lado. Obrigado pelas longas discussões sobre os trabalhos, pela parceria nos campos, e por ter contruibuído tanto nesta dissertação e em minha vida.

Aos meus pais e irmã pelo suporte e tempo que dedicam a mim, vocês são muito importantes! Obrigado por sempre acreditarem em mim. Aos loucos da 127, que me lembram que a normalidade é uma ilusão imbecil e sáfara. Vocês me proporcionaram os momentos de maior alegria e diversão nestes dois anos.

Agradeço a minha orientadora Laura Verrastro, pois sem ela não teria realizado esta dissertação e por nunca deixar de me incentivar. Ao meu co-orientador Murilo Guimarães que me apresentou aos modelos, me ajudou nas análises e colaborou muito para que esta pesquisa fosse realizada.

Aos amigos, Rafael Balestrin e Martin Schosler, que sempre me inspiraram na herpetologia, que me apresentaram as áreas, colaboraram com o campo e tornaram possível este trabalho. A todos que foram a campo para me ajudar: Vinicius Ferri, Victor Zucchetti, Nathalia Matias, Marluci Rebelato, Avelino Miguel, Diogo Reis, Gabriel Armiliato, Rhaysa Trindade, Samuel Gohlk, Andrée Solomom, Rafael Balestrin e Martin Schosler. Obrigado por dividirem o tempo de vocês comigo.

A família toda do Antônio Nunes e da dona Márcia Padilha, que sempre receberam todos muito bem, proporcionando conforto, comida boa e uma imesa hospitalidade. Ao seu "Branco" por permitir a entrada em sua fazenda.

A Pacha 90 pelas imagens capturadas de todos os sítios amostrais através do Drone pilotado pelo Gabriel Armiliato, o trabalho ficou bem mais interessante. Pela ajuda fornecida no "R" pela Débora Bordignon, Stephanie Silveira, Karoline Gilioli e Murilo Guimarães, sem vocês a análise não teria ficado tão boa quanto acho que ficou.

Ao Ismael Brack, que me ajudou no processamento de imagens no Arcgis, teve paciência de me ensinar tudo passo-a-passo. Ao Matias Malleret e ao Diego Álvares pela ajuda na confecção dos mapas. A banca de acompanhamento que acrescentou ideias a este trabalho, obrigado a todos.

 A Costelinha, pela companhia durante as longas tardes de dedicação no computador enquanto me fitava do sofá aguardando arduamente para passearmos.

SUMÁRIO

RESUMO

A invasão de plantas exóticas em ambientes naturais é uma das principais ameaças à biodiversidade. Entre as atividades humanas que mais contribuem para introdução de espécies exóticas em ambientes naturais está a silvicultura. Entender as dimensões e até onde os efeitos desta atividade podem afetar o ambiente é de grande importância para a conservação. O objetivo deste trabalho foi verificar se a silvicultura influencia negativamente a distribuição do lagartinho-pintado (*Contomastix vacariensis*). Consideramos esta espécie endêmica um ótimo modelo para responder aos efeitos negativos da plantação de exóticas. Para testarmos nossa hipótese selecionamos 69 sítios com diferentes distanciamentos da silvicultura. Utilizamos o modelo de ocupação Single-Season com abordagem bayesiana para avaliar a ocorrência da espécie nos sítios, considerando variáveis abióticas na probabilidade de detecção. Nós encontramos uma relação positiva entre o distanciamento da silvicultura e a ocupação pelo lagartinho-pintado, nossos dados evidenciam uma baixa probabilidade de ocupação pela espécie com um distanciamento inferior a 1000 m dos sítios em relação à silvicultura, e que os efeitos negativos podem atingir até 2000 m. A detecção foi influenciada pelo vento. Este é o primeiro trabalho a demonstrar um efeito gradual negativo da silvicultura de exóticas, onde ocorre com maior intensidade na área plantada e com menor intensidade conforme o distanciamento da plantação exótica.

Palavras-chave: silvicultura; pinheiros; impactos; lagarto; efeito gradual; ocupação; distribuição; extinção; endêmica.

ABSTRACT

The invasion of exotic plants in natural environments is one of the main threats to biodiversity. Forestry is one of the human activities that most contribute to the introduction of exotic species in natural environments. Understanding the dimensions and extent to which the effects of this activity can affect the environment is very important for conservation. The objective of this study was to verify if the silviculture negatively affects the distribution of the painted-lizard (*Contomastix vacariensis*). We consider this endemic specie a great model to test the negative effects of exotic planting. To test our hypothesis we selected 69 sites with different distances from silviculture. We used the single-season occupation model with Bayesian approach to evaluate the occurrence of the species into the sites, considering abiotic variables in the probability of detection. We found a positive relationship between the distance from silviculture and the occupation of the painted-lizard. Our data show a low probability of occupation by the specie within a distance of 1000m between sites and forestry, and that the negative effects can reach up to 2000m of distance. Detection was influenced by wind. This is the first work to demonstrate a gradual negative effect of exotic forestry to biodiversity which occurs with more intensity closer to the planted area and with less intensity as far as the exotic plantation gets from the occupation site of lizards.

INTRODUÇÃO GERAL

A introdução de espécies exóticas invasoras nos ecossistemas é um dos problemas mais sérios de conservação no mundo (Walker & Steffen 1997; Wilcove *et al*., 1998; Wittenberg & Cock 2001). Considerada como a segunda maior ameaça à biodiversidade, a seguir da fragmentação e destruição do habitat, as espécies exóticas invasoras podem causar danos significativos aos ecossistemas naturais (Wilcove *et al*.,1998). Entre as várias formas de ameaça relatadas, está a redução da variabilidade genética das populações nativas, extinção de espécies endêmicas, propagação de doenças e as alterações do habitat e do funcionamento dos ecossistemas (Hulme *et al*., 2007; Mack *et al*., 2000).

De maneira geral, mais espécies exóticas são conhecidas por impactos econômicos do que ecológicos, já que no primeiro caso, são mais facilmente percebidas e imediatamente relatadas (Pimentel *et al*., 2001; Born *et al*., 2004; Pimentel *et al*., 2005; Vilà *et al*., 2009). Entre os principais setores que sofrem com os impactos econômicos estão a agricultura, a silvicultura, a aquicultura e a saúde (Williamson, 2002; Vilà *et al*., 2009). O custo de invasões biológicas nos Estados Unidos para 79 espécies foi estimado em US $ 120 bilhões por ano (Pimentel *et al*., 2005). No Canadá, os prejuízos de 11 espécies invasoras foram estimados entre US $ 13 e 34 bilhões anuais (Colautti *et al*., 2006). Para a Europa o custo de 10 espécies invasoras foi de aproximadamente € 19 milhões por ano (Vilà *et al*., 2009). Desta forma, a magnitude dos impactos ocasionados pelas espécies exóticas vem despertando maior interesse sobre as ameaças das invasões biológicas, tanto à saúde humana, à produção econômica, ao ecossistema e principalmente à biodiversidade (Millennium Ecosystem Assessment, 2005).

Contudo uma espécie invasora pode ter impactos preocupantes em algumas áreas e insignificantes em outras. Atkinson (1985) demonstrou que em ilhas com roedores nativos, os ratos introduzidos causaram menos extinções de aves do que em ilhas que antes eram livres de predadores naturais. Provavelmente este fato está relacionado com o comportamento eficaz de escape de predadores já desenvolvido pelas aves. Além disso, estudos com a riqueza nativa e o estabelecimento de espécies de anfíbios invasores em diferentes regiões podem apresentar uma relação positiva (Poessel *et al*., 2013; Both *et al*., 2014). Relação que também já foi observada em plantas nativas e exóticas em grandes escalas espaciais (e. g., Stohlgren *et al*., 1999; Sax, 2002).

Essa relação positiva na riqueza de plantas nativas e exóticas deu origem a hipótese de aceitação biótica (Stohlgren *et al*., 2006), a qual afirma que a riqueza de espécies não nativas e nativas está positivamente relacionada (Stohlgren *et al*., 2006). Por outro lado, a hipótese de resistência biótica argumenta que comunidades complexas de plantas e animais são mais resistentes à invasão do que comunidades mais simples (Elton, 1958). Relacionamentos negativos de riqueza foram encontrados em pequenas escalas geográficas com invertebrados marinhos (Stachowicz *et al*., 1999) e répteis em ilhas (Case & Bolger, 1991). Já as relações positivas foram mostradas em escalas maiores com invertebrados (Burger *et al*., 2001), mamíferos (Jeschke & Genovesi, 2011), répteis e anfíbios (Poessel *et al*., 2013). O que sugere que o tipo de relação entre espécies nativas e exóticas não está ligada ao

grupo taxonômico, mas que esta relação pode ser positiva em escalas espaciais maiores.

Contudo, a maior diversidade de espécies nem sempre é o melhor resultado para a conservação da vida silvestre, uma vez que a composição de conjuntos faunísticos particulares se mostra mais importante (Burgman & Lindenmayer, 1998). Na literatura há diversos exemplos de casos em que espécies invasoras levam espécies nativas locais à extinção (Case & Bolger, 1991; Ricciardi *et al.*, 1998; Rodda *et al.*, 1997), um dos casos mais críticos é o da cobra-marrom (*Boiga irregularis*) sobre a biota de Guam, onde a serpente invasora causou a extinção de aves, répteis e mamíferos da ilha (Rodda *et al.*, 1997). Da mesma forma, Whitaker (1973) descobriu que em pequenas ilhas em que ocorriam roedores invasores, havia menos espécies de lagartos nativos. Já a extinção da lagartixa (*Phelsuma edwardnewtonii*), endêmica da ilha de Rodrigues, foi ocasionado pelo intenso assentamento europeu e pela introdução de ratos e gatos. Além disto, sabe-se que ilhas com perturbações humanas apresentam maiores taxas de extinção de répteis do que ilhas relativamente intactas (Richman *et al.*, 1988). Pregill (1986) e Case & Bolger (1991) correlacionam o período das ocupações humanas em ilhas com o período de extinção de vários répteis insulares. De maneira geral, a chegada de humanos a uma ilha está intimamente associada ao aumento das taxas de extinção de répteis, especialmente de espécies endêmicas (Case & Bolger, 1991).

Entre as principais atividades humanas responsáveis pela introdução de espécies de plantas exóticas em ambientes naturais está a silvicultura (Simberloff *et al.*, 2010; Kull *et al.*, 2011; Zenni, 2014), pois as espécies mais propensas a terem sucesso em determinado local são selecionadas para o plantio (Culley *et al.*, 2011). Outro fator que contribui é a quantidade de espécies transportadas e liberadas pelos incentivos governamentais (Overbeck *et al.*, 2007; Essl *et al.*, 2010; Culley *et al.*, 2011;). Entre os anos 1990 a 2015 estima-se um aumento de 66% nas áreas de florestas plantadas, de 168 milhões de hectares para 278 milhões de hectares no mundo (Keenan *et al.*, 2015). Contudo, a maioria compreende espécies nativas, com apenas 18 a 19% da área total sendo de espécies exóticas introduzidas (Payn *et al.*, 2015). Espécies introduzidas foram dominantes nos países do hemisfério sul da América do Sul, Oceania e África Oriental e Austrália, onde a silvicultura industrial é dominante (Payn *et al.*, 2015).

A maioria dos estudos realizados em plantações exóticas observou que estes, são menos diversos que os ambientes naturais (Lindenmayer *et al.*, 2002; Kikkawa, 1968; Cowley, 1971; Gepp, 1976; Driscoll, 1977; Friend, 1982; Smith, 1982; Bonham *et al.*, 2002; Saccol *et al.*, 2017). O plantio de exóticas em grande escala causa a perda de habitats nativos alterando a estrutura das comunidades em níveis de microclima, disponibilidade de recursos, capacidade de dispersão e interações entre espécies (Vitt & Caldwell, 2014; Wilson *et al.*, 2016). Estes impactos estão principalmente relacionados a características de crescimento e cobertura da planta, além do tamanho da área de plantio e o quanto a plantação se difere taxonomicamente da estrutura nativa do local (Martin & Muray, 2011).

Dentre as espécies mais utilizadas na silvicultura estão as do gênero *Pinus*, sendo consideradas as mais importantes do ponto de vista ecológico e econômico no mundo (GISD, 2015). O gênero possui pelo menos 19 espécies que podem ser consideradas invasoras (Richardson, 1998; GISD, 2015), devido a ampla gama de adaptações e sua excelente capacidade de colonização (Richardson, 1998; GISD, 2015). Natural do hemisfério norte, os pinheiros (*Pinus spp.*) foram cultivados em muitas partes do mundo, formando a base de empresas florestais exóticas em muitos países do hemisfério sul. Em muitas dessas áreas, os pinheiros invadiram a vegetação natural e estão agora entre as árvores exóticas invasoras mais difundidas e prejudiciais (Richardson, 1998). Os principais impactos dos pinheiros invasores resultam do aumento da abundância de árvores em habitats onde antes estavam ausentes ou menos comuns (Richardson, 1998). Grandes áreas podem ser rapidamente colonizadas por pinheiros selvagens como resultado de um evento de dispersão significativo, como ventos fortes enquanto as árvores estão em formação (Richardson, 1998).

Uma das regiões fisiográficas que vem apresentando uma alta taxa de expansão de *Pinus spp.* são os Campos de Altitude da Serra Geral. Na década de 1970 com a proibição do corte de araucária no Brasil (Brasil – Lei 4771/1965) surge um novo ciclo econômico através do plantio de *Pinus spp.* na região. O cultivo passa a ser subsidiado, tornando-se uma atividade bastante atrativa, tanto para pecuaristas quanto madeireiros (Theodoro, 2003; EMBRAPA, 2005). Deste modo, a silvicultura de *Pinus spp.* vem causando a redução de matas com araucária, bem como de campos e banhados associados da região (Fontana *et al*., 2003; Bilenca & Miñarro, 2004).

Os Campos de Altitude da Serra Geral apresentam uma alta taxa de endemismo de plantas quando comparados a outros ecossistemas neotropicais (Iganci *et al*., 2011). Uma das espécies endêmicas que ocorrem nesta região é o lagartinho-pintado: *Contomastix vacariensis* (Feltrim & Lema, 2000). A espécie vive associada a afloramentos rochosos de áreas abertas e de altitude (acima de 900 m), construindo suas tocas sob as pedras (Caruccio *et al*., 2010). O *C. vacariensis* é uma espécie diurna terrícola, ovípara e que se alimenta de artrópodes (Rezende-Pinto *et al*., 2009; Schossler, 2007). Através de estudo de marcação e recaptura foi verificado que os indivíduos são fiéis aos afloramentos rochosos, não se deslocando entre eles. Atualmente, *C. vacariensis* encontra-se ameaçado de extinção, na categoria vulnerável (VU) (Machado *et al*., 2008; MMA, 2014). Dentre as principais ameaças estão as monoculturas e plantações de espécies arbóreas exóticas (*Pinus spp.* e *Eucaliptus spp.*), atividades que vêm descaracterizando o hábitat da espécie (Di-Bernardo *et al*., 2003; Bérnils *et al*., 2004).

Para os lagartos, as modificações na estrutura do habitat causada pelas plantas exóticas invasoras podem alterar o alcance, a frequência e a distribuição espacial das temperaturas, impedindo assim que estes mantenham temperaturas corporais adequadas ou aumentem os custos energéticos para termorregulação (Clusella-Trullas & Chown 2011; Sears *et al*., 2016). Mudanças na composição de espécies de plantas também podem gerar acidificação do solo e consequentemente comprometer a qualidade dos ovos dos lagartos (Cannell, 1999; Jobbágy & Jackson, 2003; Marco *et al*., 2005).

Ademais, a silvicultura também pode causar redução na abundância e riqueza de espécies de presas de lagartos, especialmente artrópodes (Cunningham, 2005; Martin & Murray, 2011). Em um estudo de revisão bibliográfica, Martin e Murray (2011) verificaram que seis características preditivas em espécies de lagartos podem determinar a intensidade e o tempo de resposta destes impactos: (1) tamanho corporal; (2) tempo de vida; (3) tamanho de área de vida; (4) especialização de habitat; (5) dieta e (6) estratégia reprodutiva. O tamanho corporal reduzido, a área de vida relativamente pequena, o hábitat restrito, a dieta insetívora e a oviparidade, tornam o *C. vacariensis* uma espécie potencialmente sensível aos impactos da silvicultura, que são mais facilmente detectáveis e perceptíveis em menor escala de tempo (Martin & Murray, 2011).

Neste estudo, construímos um modelo de ocupação Single Season (Mackenzie *et al*., 2002) para verificar se *Pinus sp.* pode influenciar a ocupação de *C. vacariensis*. O modelo considera a detecção imperfeita reduzindo o viés amostral, com uma maior precisão e qualidade nas estimativas populacionais (Mackenzie *et al*., 2006), especialmente tratando-se do grupo herpetofauna (Guimarães *et al*., 2014). A detectabilidade (*p*) é influenciada por fatores que variam no tempo e local específico, tais como umidade e temperatura, enquanto a ocupação (ψ) utiliza características do ambiente como por exemplo o tipo de vegetação (Mackenzie & Bailey, 2004). Neste estudo utilizaremos o distanciamento dos sítios amostrais em relação a silvicultura de *Pinus spp.* para verificar se esta influência negativamente a distribuição do *C. vacariensis*.

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CAPÍTULO 1

HOW CLOSE IS DANGER? RELATIONSHIP BETWEEN THE DISTANCE FROM AN EXOTIC TREE PLANTATION AND OCCUPANCY OF AN ENDEMIC LIZARD.

Artigo formatado para publicação no periódico Austral Ecology.

Normas aos autores:

https://onlinelibrary.wiley.com/page/journal/14429993/homepage/forauthors.html

Abstract

The cultivation of exotic plants in natural environments is one of the main threats to biodiversity. Silviculture is among the human activities that contribute most to fragmentation and habitat loss. Understanding the dimensions and how far the effects of this activity can affect the environment is important for conservation. The objective of this study was to verify if silviculture negatively influences the distribution of *Contomastix vacariensis*, an endemic lizard species that is a good model of response to the negative effects of exotic trees planting. We used a single-season occupancy model with a Bayesian approach in order to assess the occurrence of the species in 69 selected sites with different distances from silviculture areas. In this model, we considered abiotic variables in the probability of detection. We found a positive relationship between the distance from silviculture and occupancy by *C. vacariensis*. Our data show a low probability of occupancy by the species within a distance inferior to 1000 m from silviculture areas, and such negative effects can reach up to 2000 m. Detection was influenced by the wind. This is the first work to demonstrate a gradual negative effect of exotic silviculture, which takes place with greater intensity inside the planted area and decreases its intensity as the distance from the exotic plantation increases.

Keywords: environmental impact; plantations; cultivation; monoculture; endemism, extinction; occupancy model; anthropization.

Introduction

Fragmentation and habitat loss are considered to be the main causes of biodiversity loss and ecosystem degradation worldwide (Meffe *et al.* 2006; Wu 2013). Habitat fragmentation can be defined as a process in which a continuous habitat is transformed into smaller remnants, and it is linked to changes in the natural landscape, while habitat loss usually occurs along with fragmentation (Franklin *et al.* 2002; Carvalho *et al.* 2009; Collinge and Forman 2009). Although fragmentation and habitat loss can occur naturally (Clark *et al.* 1999), human disturbance is its most common cause, for example, through the expansion of agriculture, silviculture, pasture lands, and urbanization (How and Dell 2000; Lovejoy and Laurance 2013; Almeida *et al.* 2019).

Silviculture is among the main human activities responsible for changes in the natural landscape (Pauchard *et al.* 2010; Simberloff *et al.* 2013; Kull *et al.* 2011; Zenni 2014). It is estimated that, between 1990 and 2015, there was a 66% increase in planted forest areas, from 168 million ha to 278 million ha worldwide (Keenan *et al.* 2015). However, most of these planted forests are made up of native species, and about 18% of the total area is formed by introduced exotic species (Payn *et al.* 2015). Introduced plant species are predominant in countries of the southern hemisphere, where industrial silviculture is dominant (Payn *et al.* 2015). The pines of the *Pinus* genus, native to the northern hemisphere, are among the main species used in silviculture and are considered the most widespread and harmful invasive exotic trees in the world (Richardson 1998).

Changes in natural landscapes cause changes in biodiversity at the population level through changes in the number, distribution, reproduction, survival and recruitment of individuals (Wolff *et al.* 1997; Fahrig 2002). Specialist species tend to be more sensitive to these changes, presenting an increase in mortality and a decrease in reproductive rates when favorable environments are no longer available (Fahrig 2002; Martin and Murray 2011). In addition, the replacement of the original vegetation by new forms of land use can cause negative changes in environmental heterogeneity and, consequently, in the supply of resources available to the species (Lion *et al.* 2016).

Due to government incentives, silviculture activity in Brazil is quite attractive, thus a significant increase in planting areas has been observed (Pillar 2003; Theodoro *et al.* 2003). In the state of Rio Grande do Sul, southern Brazil, the prohibition of cutting the native Paraná pine (*Araucaria angustifolia*) generated a considerable increase in the silviculture of *Pinus spp.*, an activity that continues to grow (Hasenack 2017). Studies indicate an increase of 112% in the planted area from 2002 to 2017, with 98% of this expansion taking place on native grasslands (Hasenack 2017). The negative effects caused by the conversion of grasslands into exotic silviculture are even more worrying, since there is a structural disfiguring of the vegetation form, making silviculture the main threat to natural grasslands (Miñarro 2004; Gerhard and Overbeck *et al.* 2007; Martin and Murray 2011).

Most studies that demonstrate the impacts of silviculture on the fauna mainly evaluate the presence or absence of species within plantations, investigating the impacts on the environment that

has been directly altered (Martin and Murray 2011). On the other hand, the effects of planted forests on adjacent areas and from the perspective of the landscape are still poorly known, especially in open areas, where plantations can act as barriers to native species, causing habitat loss and fragmentation (Mortelliti *et al.* 2015).

In reptiles, the impacts caused by invasive exotic plants can occur through three mechanisms: alteration of habitat structure, changes in interaction dynamics, and changes in reproductive success (Martin and Murray 2011). The impacts of monocultures on reptile communities include decreased species richness and diversity (Loehle *et al.* 2005; Gardner *et al.* 2007; Mott *et al.* 2010), although positive relationships have already been recorded (Vonesh 2001). The magnitude of these impacts on reptiles depends on the characteristics of the invasive plant (growth form, coverage area, taxonomic distinction in relation to the native ones) and on the characteristics of the impacted species, such as body size, lifespan, home range size, habitat specialization, diet and reproductive strategy (Martin and Murray 2011).

In this study we evaluated whether the presence of *Pinus spp*. forests influences the occurrence of *Contomastix vacariensis* beyond the area directly planted. *C. vacariensis* is an endemic species threatened with extinction (ICMBio 2018), potentially susceptible to the impact of silviculture due to its life history characteristics, which include restricted distribution, small body size and oviparity. In addition, the species has a high degree of habitat specificity, occupying rocky outcrops in grassland areas. Because of these characteristics, *C. vacariensis* could respond to the negative effects of the silviculture of exotic species in a faster and more noticeable way than generalist lizards (Martin and Murray 2011). Therefore, we believe that the occupancy probability of *C. vacariensis* is directly influenced by the distance from *Pinus spp.* plantations, with most occupied sites further away from silviculture areas.

Material and methods

Study area and data collection

Our study was developed in the Atlantic Forest biome, in the physiographic region called Highland Grasslands of the Serra Geral. The sampling was carried out in the municipalities of Bom Jesus and Vacaria, both located in the state of Rio Grande do Sul, southern Brazil, approximately 70 km away from each other. The Highland Grasslands of the Serra Geral are the coldest and highest region of Rio Grande do Sul, the climate of the region being subtropical with average annual temperatures of 15.2°C and average temperature of the coldest month of 10.6°C (Maluf 2000). The annual precipitation varies from 1,500 mm to 2,000 mm and it is well distributed throughout the year (Moreno 1961). The landscape of the Highland Grasslands region is composed of mosaics of open areas interspersed with forests, with herbaceous and/or shrubby formations, which usually occur on the ridges of mountain ranges with high altitudes. The grassland flora is characterized by many endemisms at a specific level (Klein 1984; Overbeck *et al.* 2007).

In order to assess the occurrence of *C. vacariensis,* we selected 69 sample sites, distributed in five groups: four located in the municipality of Bom Jesus and one in Vacaria. The distance among groups was greater than the distances among sampling sites within a group. The smaller group contained three sites, while the largest one contained 27 sampling sites (Fig. 1). We believe that each sampling area corresponds to a distinct population, given the low dispersal capacity of the species, which has already been described in studies about phylogeny and micro-habitat use (Caruccio *et al.* 2010; Zanotelli 2010). The sampling sites were selected in order to present different distances from the forest patches of *Pinus spp.* monocultures, with sizes ranging between 374 and 800 ha each. The sites presented mean distance of 1344 m from *Pinus spp.* areas, being 0 m the shortest distance (inside the plantation), and 6500 m the largest.
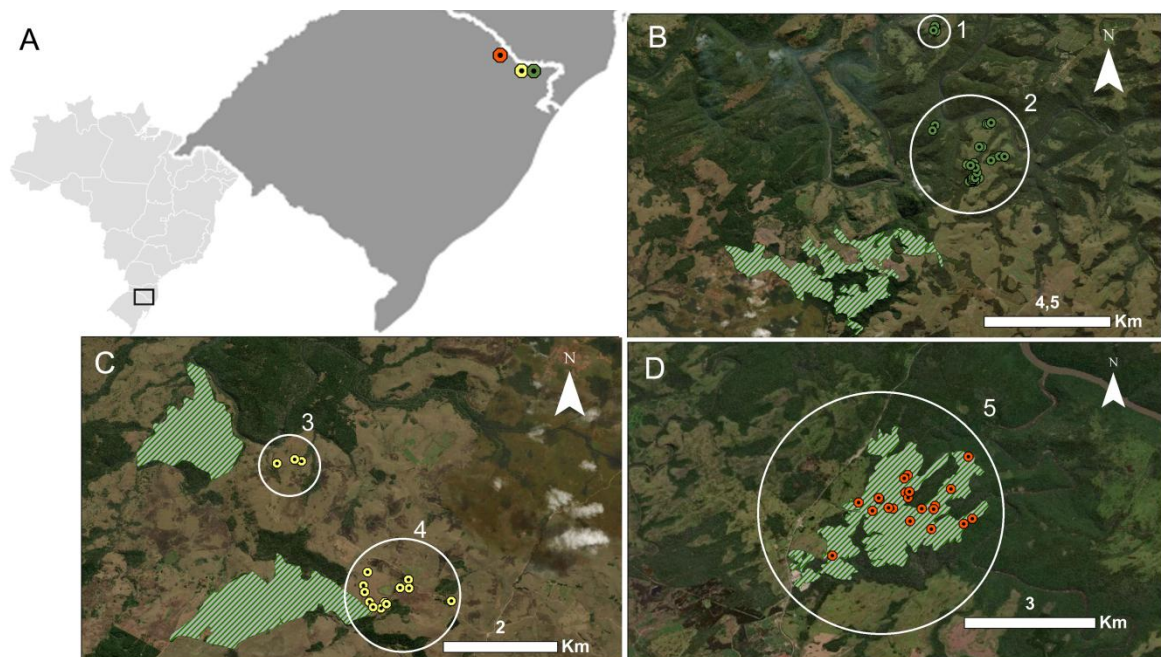

Figure 1 Study areas located in Rio Grande do Sul, Brazil. (a) Location of the sampling areas: Bom Jesus (Green and yellow dots) and Vacaria (red dot). (b), (c) and (d) Sampling sites and their respective sampling groups (white circles) at different distances from the highlighted silviculture spots (striped spots) (adapted from Google Earth, 2003).

The Vacaria sampling area (Fig. 1D) corresponded to the sampling sites (rocky outcrops) that had their surroundings converted from native grasslands into *Pinus spp.* silviculture (MDP = 0m) (Figure 2). After the growth of the *Pinus spp.* plantation, the sites became partial or total shaded throughout their extension, and also exposed to a shorter period of sunlight during the day. Before the conversion of the natural grasslands into *Pinus spp.*, this area had a historical record of occurrence of the species, according to records in the Reptile Collection of the Federal University of Rio Grande do Sul (Table 1 - supplementary material), in addition to being the type locality of *C. vacariensis* (Feltrim and Lema 2000). The other areas presented similar environments, where rocky outcrops could be found in open areas along with native arboreal vegetation nearby, forming a mosaic.

Each sample site corresponded to a rocky outcrop, the habitat of *C. vacariensis*, and they were

individualized when they were not interconnected. Sampling took place between October 2017 and May 2018, totaling four campaigns for each of the five areas. The employed method was active sampling (Parmelee *et al.* 1995; McDiarmid *et al.* 2012), with three researchers looking for the target species under all accessible rocks of each sampling site. Surveys were conducted during daytime, between 8:00 a.m. and 05:00 p.m.
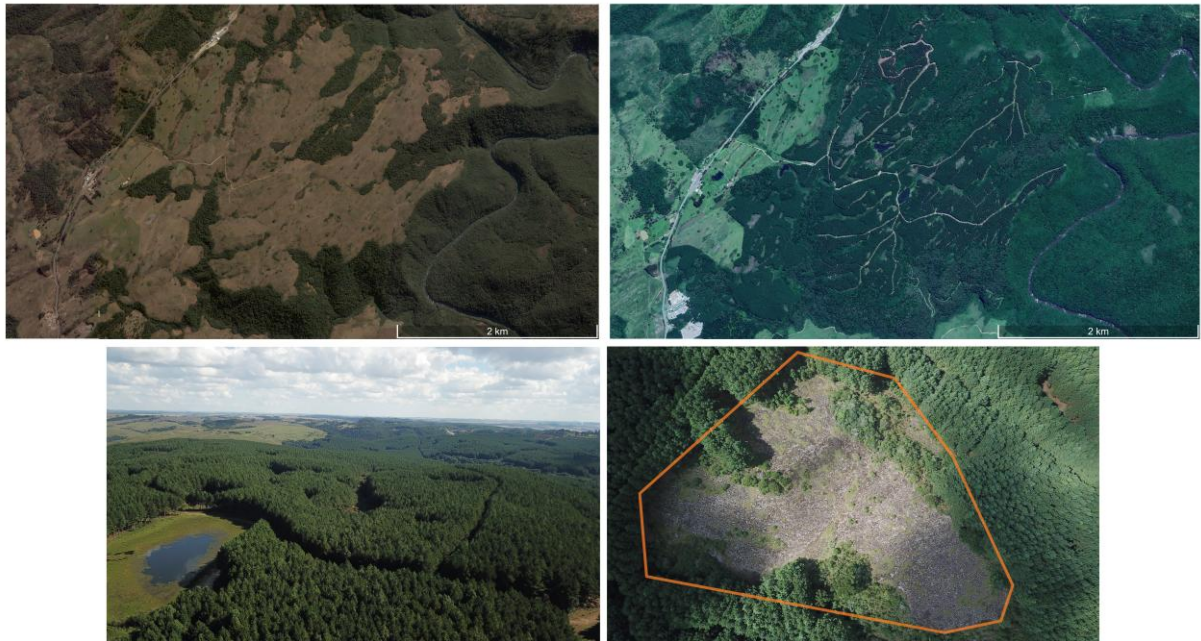

Figure 2 Study area located in Vacaria, Rio Grande do Sul, Brazil. (a) Satellite image before silviculture (Google Earth, 2003); (b) Satellite image after silviculture (Google Earth, 2018); (C) Aerial image of silviculture areas during the study; (d) Example of an isolated rocky outcrop after silviculture. The dotted orange line represents the delimitation of a sampling unit.

Temporal and spatial variables

In order to evaluate the probability of occurrence of *C. vacariensis*, we registered the size of the rocky outcrop in each sampling site (Area), the number of loose rocks (nRock), the minimum distance between outcrops and the *Pinus spp.* plantation (MDP), the minimum distance between outcrops and the native forest (MDN), slope (Slope) and the type of coverage of the sample site according to the amount (%) of vegetation and rocks. The distance values of MDP and MDN were obtained using the ArcGis program. In order to determine the size of the sampling sites, we used the Minimum Convex Polygon (MCP), considering the outermost rocks to determine the area. For that we used aerial photographs obtained with a drone (Mavic Pro – DJI), standardizing the altitude in 100 m, besides using a scale of 1 m x 1 m in each site as reference for measurement. Subsequently, the images were processed in the ArcGis program in order to obtain the area and the coverage type of each sampling site. To obtain the coverage type, we performed supervised color classification, which uses training samples indicated by the user, differentiating the coverage pixels by Maximum Likelihood. The levels of the coverage type variable were: rocky (cRock), herbaceous vegetation (cHerbaceous) and shrubby/arboreal vegetation (cShrubby) for each of the sampling sites. The number of loose rocks

was obtained by the average total number of accessible rocks in all visits. Slope was measured with a clinometer in the central area of each sampling site, considering the highest part in relation to the lowest.

As temporal variables we included air temperature (temperature), quadratic air temperature (temperature^2), air humidity (humidity) and wind speed (wind), which were measured at the beginning of the surveys for each of the sites. The sum of surveyed rocks (nvRock) at each site visit was also used as a temporal variable, as a way to include the sampling effort per site and per visit.

Statistical analysis

Herbaceous vegetation (cHerbaceous) and shrubby/arboreal vegetation (cShrubby) were negatively correlated (Spearman correlation - 0.77), and thus, to avoid multicollinearity, we chose to use herbaceous vegetation, since the studied species prefers open areas (Caruccio *et al.* 2010). The other variables showed no correlation among themselves.

We used the closed population model (Single-Season Occupancy Model) *C. vacariensis* in the sampling sites. Occurrence is defined as the probability of a species being present at site *i*, and detection as the conditional probability of, given that the species is present, it being found at the sampling site *i* and time *t* (Bailey *et al.* 2004). By incorporating the probability of detection in the models, we assume that detection is imperfect, and reduce bias in the estimated parameters (MacKenzie *et al.* 2017). The presence/absence of the species in site *i* is modeled as a binary occurrence indicator (zi), treated as a Bernoulli random variable governed by the parameter ψ (probability of occupancy). The measurement is performed considering detection or not of the species at site *i* during visits *j* ($y_{i, j}$). The two parameters (ψ_i and $p_{i, j}$) are estimated separately by replicating visits.

All variables were rounded to two decimal places and subsequently standardized. Our model for occupancy (a) in each site *i*, and for detection (b) in each site *i* and visit *j*:

(a) logit (ψ_i) = β0 + β1*Area + β2*nRock + β3*MDP + β4*slope + β4*cRock + β5*cHerbaceous

(B) logit ($p_{i, j}$) = α0 + α1*temperature + α2*temperature^2 + α3*humidity + α4*wind + α5*nvRocha + eps.pi

We estimated the parameters employing Bayesian inference with Markov Chains - Markov Chain Monte Carlo (MCMC) (Kéry and Royle 2015). For that, we used three parallel chains with 100,000 iterations each, burn-in period of 50,000, and a thinning rate of 100. The number of occupied sites among the 69 sampled was also estimated. We implemented the model in the BUGS (Lunn *et al.* 2000) language in JAGS (Plummer 2003), using the R program (R Core Team 2016) and the jagsUI package (Kellner 2014). Chain convergence was evaluated visually and using the Brooks-Gelman-

Rubin statistic (Brooks and Gelman 1998), accepting Rhat values <1.1 for all parameters. We present averages and 95% credibility intervals of the posterior parameter distributions to assess uncertainty in the parameters of interest. We inserted the random effect to control a possible pseudo-replication effect among the five sample groups in which the 69 sampling sites were distributed.

Results

We had a total of 60 records of *C. vacariensis* in 14 of the 69 sampled sites. Five individuals were observed foraging or moving in rocky outcrops (8%), and 55 lizards were found under rocks, in burrows or shelters (92%). The probability of occurrence of the species was estimated at 0.22 (CRI 0.19 – 0.25), where 15 sites (CRI 14 – 18) were occupied, according to the model. The covariate coverage of herbaceous vegetation had a positive and significant effect on the probability of occupancy by the species ($\text{ßMc}_{Herbaceous}$ = 4.26; CRI 0.28 – 10.10), as well as the distance from *Pinus spp.* plantations ß_{MDP} = 9.98 (CRI 4.50 – 16.88) (Fig. 3), and we only found the species at a minimum distance of 1983 m from the plantation areas. The effect of distance from native forest was also positive, although not significant and lower in magnitude (ßM_{MDN} = 2.27, CRI -1.14 – 5.8).

The remaining covariates, number of rocks (ßM_{nRock} = 3.75; CRI -0.83 – 10.86), site size (ßM_{Area} = 0.25; CRI -1.85 – 3.02), rock coverage (ßM_{cRock} =1.56; CRI -2.10 – 5.74) and slope (ßM_{Slope} = 1.16; CRI -0.40 – 3.09), had minor positive effects on the probability of occurrence of the species and were not significant.

The effect of spatial auto-correlation indicated significant variation between the five sampling units 7.26 (CRI = 2.13 – 9.87).
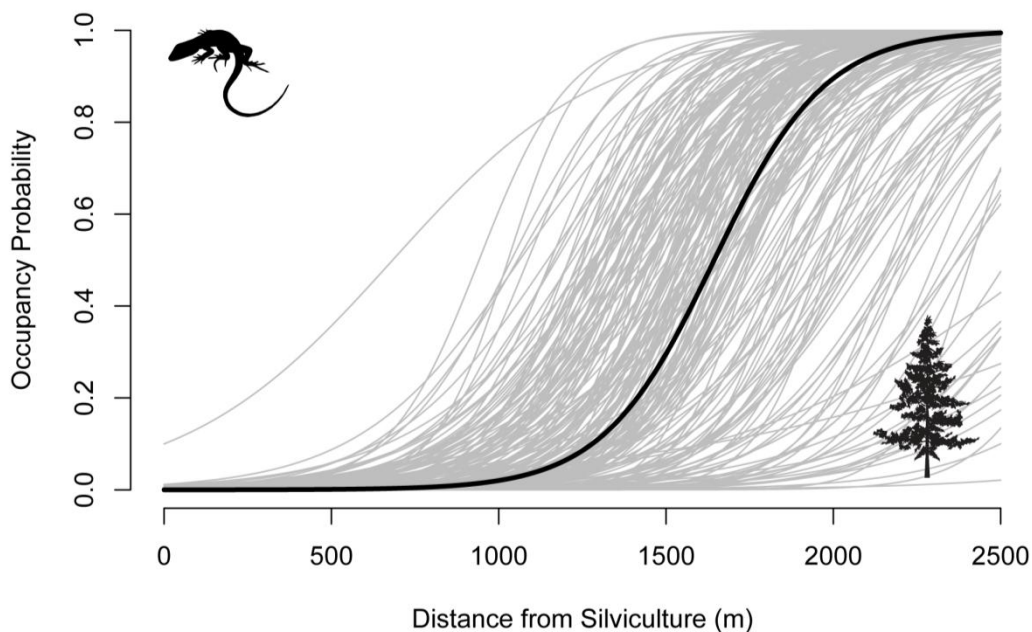

Figure 3 Probability of occurrence of *Contomastix vacariensis* as a function of distance from

silviculture sites. The black line represents the average prediction, while the grey lines represent 200 random draws from the posterior distribution, describing the uncertainty of the prediction.

Figure 4 Probability of occurrence of *Contomastix vacariensis* as a function of distance from arboreal vegetation. The black line represents the average prediction, while the grey lines represent 200 random draws from the posterior distribution, describing the uncertainty of the prediction.
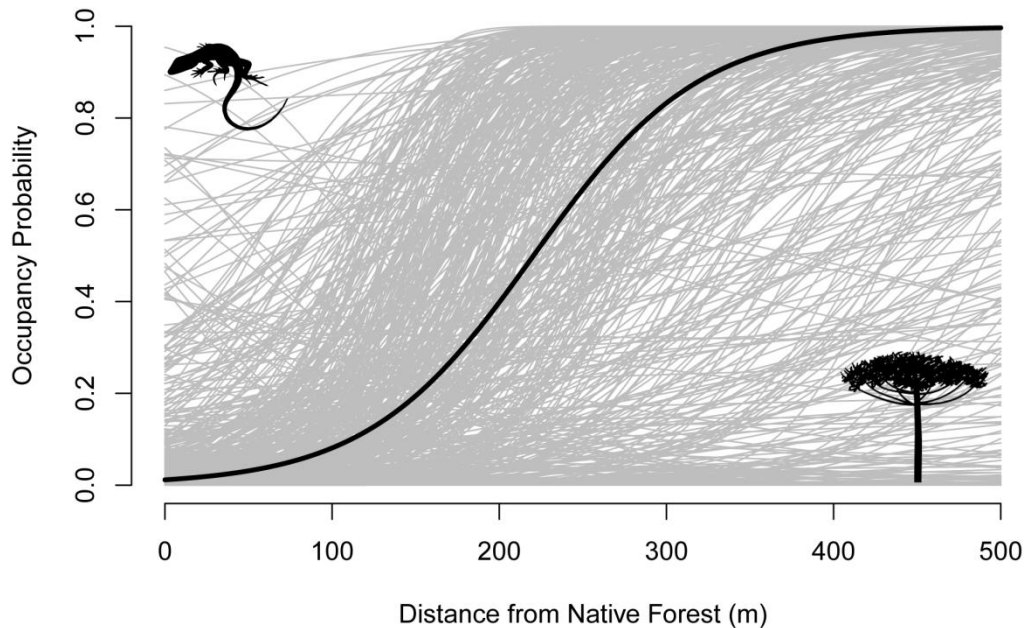
The probability of detection (p) was estimated at 0.70 (CRI from 0.51 to 0.86). Only wind speed had a negative effect on the detection of the species in the areas (α_{wind} = -0.72; CRI -1.45 – -0.04). The other variables, temperature ($\alpha_{temperature}$ = 0.64; CRI -0.29 – 1.68), its quadratic effect ($\alpha_{temperature2}$ = 0.29; CRI -0.59 – 1.42), humidity ($\alpha_{humidity}$ = 0.08; CRI -0.57 – 0.80) and effort (α_{nvRock} = -0.83; CRI -2.53 – 1.04) showed no significant effects on the detection of the species.

Discussion

Our estimates resulting from the Single-Season Occupancy Model corroborate the preference of *C. vacariensis* for rocky outcrops with herbaceous vegetation, since this covariate (cHerbaceous) presented a positive relationship with the occupancy by the species. This result was expected, since *C. vacariensis* is specialized in rocky outcrops of open areas (Caruccio *et al.* 2011, 2010). Caruccio *et al.* (2010) observed 99% of individuals under rocks in their study on the use of micro-habitat by this species, and a similar result was found in the present study, given that the majority of individuals (92%) was recorded sheltered under rocks.

Species of lizards adapted to open areas often have their occupancy restricted by tree species, such as the sand lizard (*Lacerta agilis*), which due to thermoregulation difficulties, occupies only open areas or edges of vegetative areas (House and Spellerberg 1983). It is known that substrates shaded by

arboreal vegetation absorb less amount of thermal energy, making it impossible for species that thermoregulate to reach optimal temperatures (Mott *et al.* 2010; Singh *et al.* 2002; Schreuder and Clusella-Trullas 2016; Downes and Hoefer 2007). In this case, *C. vacariensis* depends on the rocky substrate to thermoregulate, and shading the rocks would impair thermoregulation (Caruccio *et al.* 2011). Besides that, it has already been demonstrated that the thermal condition of the environment determined habitat selection in two sympatric species of lizards (*Carlia vivax* and *Lygisaurus foliorum*) in Australia (Singh *et al.* 2002).

Our estimates of occupancy probability for *C. vacariensis* in sites less than 1000 m away from the *Pinus spp.* plantations were very low, close to zero (Fig 3). Since the species has a limited dispersion rate between rocky outcrops – even when these are relatively close to each other (Caruccio *et al.* 2010), we suspect that a local extinction happened in the Vacaria sampling group, which had its surrounding rocky outcrops converted into silviculture sites (MDP = 0m). The outcrops closest to this area that did not have their surroundings converted into *Pinus spp.* plantations are more than 2400 m away, which supports our hypothesis. In addition, it is known that plantations of *Pinus spp.* can act as a barrier for lizards inhabiting open areas (Mortelliti *et al.* 2015).

Changes in habitat characteristics, added to the environmental homogeneity caused by silviculture, were responsible for changes in the composition and structure of reptilian assemblages (Heatwole & Taylor, 1987). Studies have shown that lizards are often associated with specific habitats based on fine-scale characteristics of vegetation structure (Vitt *et al.* 1998; Mott *et al.* 2010; Garda *et al.* 2013). Furthermore, specialist lizards may not occur in the interior of tree plantations (Gainsbury and Colli 2014). Other studies have already shown that silviculture can reduce the abundance and richness of lizard species that occupy a given area (Mott *et al.* 2010; Stellatelli *et al.* 2013; Mortelliti *et al.* 2015). Habitat loss, for example, has already been observed negatively influencing the occupancy of the European green lizard (*Lacerta viridis*) (Prieto-Ramirez *et al.* 2020). The severity of the impact of silviculture is strongly influenced by the difference between coverage and height of the introduced species compared to the native ones (Hejda *et al.* 2009). This makes the conversion of native grasslands into *Pinus spp.* plantations even more worrying since their area and height of canopy cover are much higher than the native tree species, exerting more severe negative effects on biological diversity (Hejda *et al.* 2009). *C. vacariensis* is currently threatened with extinction due to the fact that its subpopulations are isolated and have restricted habitat selection, silviculture of exotic trees being one of the main threats (ICMBio 2018).

The effects of native forests on the occurrence of *C. vacariensis* were lower than the effects of the *Pinus spp.* plantation, and were not significant. However, native forests have greater structural complexity and heterogeneity (Alamgir *et al.* 2016), and even species which typically inhabit open areas, such as lizards, can benefit from such environments, which can be a source of food resources and allow the maintenance of specific microclimates (Sax 2002; Rolim *et al.* 2010). Our results show a significant increase in the probability of occupancy by *C. vacariensis* 200 m away from the native

forest, while the same probability increase could only be observed 1500 m away from *Pinus spp.* plantations (Figs. 3 and 4). The observed discrepancy between native and exotic forest vegetations indicates that the negative effects of silviculture are not merely associated with mechanical shading, given that these two kinds of arboreal vegetation impose different impacts when compared at different distances.

In addition to habitat change, two other mechanisms are known to impact lizard species in exotic plantations: reproductive success and availability of food resources (Martin and Murray 2011). Reproductive success can be influenced by exotic planting, since incubation temperatures can alter growth and development of embryos (Wells 2007; Martin and Murray 2011). Furthermore, the conversion of native grasslands into silviculture can lead to soil acidification (Jobbágy and Jackson 2003), damaging eggs and affecting size, mass and fitness of individuals (Marco *et al.* 2005). Exotic plants have already been observed acting as a barrier to the oviposition of an endangered species of Australian turtle (*Elusor macrus*), besides the direct destruction of eggs by their roots (Tucker 1999; Van Kampen *et al*. 2003).

Prey availability (arthropods) for *C. vacariensis* may also have been significantly reduced, since herbivores can respond strongly and quickly to the incursion of exotic plants (Sax 2002). Changes in invertebrate assemblages caused by invasion of exotic plants can have impacts on vertebrates, altering the availability and composition of prey species (Herrera and Dudley 2003; Greenwood *et al.* 2004). Such changes in vegetation structure have already been observed causing changes to the diet of other endemic lizards, such as *Ameivula nigrigula (Xavier et al. 2019).* In Northern Australia, reduced reptile species richness has been associated with a decline in arthropod abundance in weed-infested habitats (Griffin *et al.* 1989). Native forests are generally richer in invertebrates, which would corroborate the importance of the presence of native forest patches in the vicinity of the occurrence of *C. vacariensis*, since it would provide a greater diversity of food resources.

We believe that the negative effects decrease as distance from silviculture sites increases, therefore forming a gradient. Canopy modification caused by silviculture, which causes shading, affects with greater intensity the area directly planted. However, changes in the microclimate may occur beyond this area, an effect that has already been observed in plantations of tree species (Theodoro *et al.* 2003). Soil acidification caused by *Pinus spp.* plantations is closely linked to the deposit of pine needle biomass, thus acidification is expected to have a greater intensity inside plantation sites, where there is greater deposit of organic matter (Lilienfein *et al.* 2000). Arthropod availability is also ought to be higher in areas further away from silviculture, and studies show that the interior of these plantations presents lower insect diversity than their edges (Wilson and Belcher 1989; Herrera and Dudley 2003; Martin and Murray 2011). This gradient formed by one or more of the impacts mentioned above would cause an observational biological response in the probability of occupancy by *C. vacariensis*, which increases along with distance from *Pinus spp.* plantations (Figure

3). This effect is analogous to the so-called "road-effect zone", so that the impacted area can be expanded beyond the converted area itself (Richard *et al.* 2000).

The estimated detection of *C. vacariensis* was relatively high (0.70; CRI 0.51 – 0.85) when compared to other studies involving lizards (*e.g.* Roughton and Seddon 2006; Smolensky and Fitzgerald 2010; Molina-Zuluaga *et al.* 2013). Among the temporal variables tested, only wind speed had a negative influence on the detection of *C. vacariensis*, probably because a higher wind index accelerates the heat loss of rocks that are warmer than the environment itself, causing individuals to shelter under larger stones, which keep the heat longer and may be inaccessible to observers. The selection of suitable micro-habitats for thermoregulation has a profound impact on their thermal physiology and consequently on individual survival (Huey 1991). These rocks present a valuable resource for ectothermic animals of open areas, as they have high conductivity and heat storage capacity (Sabo 2003). Thinner and smaller rocks are more susceptible to variations in environmental temperatures (Huey *et al.* 1989). The temporal variable related to the number of rocks (nvRock) showed a negative effect on detection. This result may have a relationship with the fact that, with a greater number of shelters, researchers may involuntarily end up not surveying some rocks at a given sampling site, decreasing detectability. However, since this relationship was not significant, the adopted survey method proved to be efficient.

Implications for conservation

Our study has great impact on conservation, since pines are the most widespread tree species in plantations around the world (Brockerhoff *et al.* 2008). Being considered an invasive exotic species in countries of the southern hemisphere, the silviculture of *Pinus spp.* is dominant in South America, Africa and Oceania (Payn *et al.* 2015). Although our study was conducted with only one species of exotic tree and a single species of lizard, it is likely that the negative effects of exotic silviculture are similar for other taxonomic groups, affecting with higher intensity the planted area itself and with progressively lower intensity the surrounding areas as the distance from the plantation increases. The magnitude of these effects in native species may, however, depend on a variety of factors, including the scale of silviculture (size, extent and density), the region, and the taxonomic group affected (Kumschick *et al.*, 2015; Ricciardi *et al.*, 2013).

The impacts caused by the silviculture of *Pinus spp.* can be greater in ectotherms (reptiles, amphibians and arthropods), since, for these species, the energy cost is more influenced by the environment, besides their dispersal capacity being generally lower than mammals and birds, for example (Endler 1977; Clusella-Trullas and Garcia 2017). However, bird species are generally less recorded on pine plantations (Cowley 1971; Kloeden 1973; Bevege 1974). Pine trees are pollinated by the wind, and there are usually few species of flowering plants in these plantations, making nectar and fruits scarce food resources, consequently decreasing the attractiveness for some insects. Kloeden (1973) and Bevege (1974) observed more bird species on plantations that had native understory

vegetation and greater food availability. Mammal species richness is also lower in pine forests (Frith 1973), a fact that may also be linked to a lower availability of food resources. We highlight the importance of understanding the relationship between the local distribution of native species and the presence of invasive plants in order to understand the impacts caused by silviculture of exotic species. Our contribution sheds light on the importance of assessing the distance up to which planted forests may influence species, and can subsidize planning and actions to reduce the impoverishment of native fauna and flora.

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Supplementary Material

Table 1. Collection of individuals of *C. vacariensis* in the sampling unit of the Vacaria municipality - RS, Brazil. Material deposited in the UFRGS collection.

Number	Collector	Collection date	UTM
3684	André, Denise, Juliana, Rita, Isabel, Martin, Ernesto, Carlos	21/08/2004	22J 0522870/6871701
3685	André, Denise, Juliana, Rita, Isabel, Martin, Ernesto, Carlos	21/08/2004	22J 0522951/6871714
3686	André, Denise, Juliana, Rita, Isabel, Martin, Ernesto, Carlos	21/08/2004	22J 0522951/6871714
3687	André, Denise, Juliana, Rita, Isabel, Martin, Ernesto, Carlos	21/08/2004	22J 0522951/6871714
3688	André, Denise, Juliana, Rita, Isabel, Martin, Ernesto, Carlos	21/08/2004	22J 0522951/6871714
3689	André, Denise, Juliana, Rita, Isabel, Martin, Ernesto, Carlos	21/08/2004	22J 0523016/6871801
3690	André, Denise, Juliana, Rita, Isabel, Martin, Ernesto, Carlos	21/08/2004	22J 0523000/6871781
3692	André, Denise, Juliana, Rita, Isabel, Martin, Ernesto, Carlos	21/08/2004	22J 0523030/6871818
3693	André, Denise, Juliana, Rita, Isabel, Martin, Ernesto, Carlos	21/08/2004	22J 0523030/6871818
4720	Schossler, M.; Turcati, A.	11/04/2008	22J 0523076/6871844
4721	Schossler, M.; Turcati, A.	11/04/2008	22J 0522930/6871706
4724	Schossler, M.; Turcati, A.	11/04/2008	22J 0522930/6871706

Table 2. Sampling sites according to the distance from *Pinus spp* patches (MDP) and from native forest (MDN), with the respective municipality.

Sampling Site Code	Sampling Group	Size of Influence Silviculture (ha)	MDP (m)	MDN (m)	Municipality
B21	1	820	6498,34	43,10	Bom Jesus
B22	1	820	6378,62	117,20	Bom Jesus
B20	1	820	6333,18	98,30	Bom Jesus
B19	2	820	3936,93	50,10	Bom Jesus
B18	2	820	3832,93	32,20	Bom Jesus
B17	2	820	3831,15	2,00	Bom Jesus
B41	2	820	3275,50	31,00	Bom Jesus
B42	1	820	3220,18	82,00	Bom Jesus
B40	2	820	3150,91	26,00	Bom Jesus
B15	2	820	3063,21	35,10	Bom Jesus
B39	2	820	3014,52	15,00	Bom Jesus
B16	2	820	2980,12	73,40	Bom Jesus
B23	2	820	2875,39	20,70	Bom Jesus

| Sampling Site Code	Sampling Group	Size of Influence Silviculture (ha)	MDP (m)	MDN (m)	Municipality
B38	2	820	2858,56	2,00	Bom Jesus
B24	2	820	2834,07	62,40	Bom Jesus
B14	2	820	2427,58	28,70	Bom Jesus
B12	2	820	2322,72	32,30	Bom Jesus
B13	2	820	2276,76	81,20	Bom Jesus
B10	2	820	2268,82	19,10	Bom Jesus
B11	2	820	2237,12	18,00	Bom Jesus
B09	2	820	2083,57	57,70	Bom Jesus
B01	2	820	2073,79	143,73	Bom Jesus
B02	2	820	2034,21	85,25	Bom Jesus
B08	2	820	2030,58	37,20	Bom Jesus
B07	2	820	1983,12	65,50	Bom Jesus
B03	2	820	1953,66	64,50	Bom Jesus
B04	2	820	1912,59	89,40	Bom Jesus
B05	2	820	1878,15	31,10	Bom Jesus
B25	4	820	1253,30	13,00	Bom Jesus
B33	4	341	862,65	162,00	Bom Jesus
B32	4	341	765,89	255,70	Bom Jesus
B43	4	341	677,27	115,00	Bom Jesus
B31	4	341	655,19	343,30	Bom Jesus
B44	4	341	533,70	53,00	Bom Jesus
B30	4	341	496,33	260,00	Bom Jesus
B45	4	341	490,54	80,00	Bom Jesus
B29	4	341	293,16	340,00	Bom Jesus
B36	3	341	282,75	371,10	Bom Jesus
B37	3	341	275,25	139,10	Bom Jesus
B34	3	341	201,99	320,80	Bom Jesus
B28	4	341	165,00	198,18	Bom Jesus
B27	4	341	124,65	248,02	Bom Jesus
B26	4	341	88,76	441,70	Bom Jesus
V01	5	510	0,00	48,60	Vacaria
V02	5	510	0,00	41,20	Vacaria
V03	5	510	0,00	383,70	Vacaria
V04	5	510	0,00	187,70	Vacaria
V05	5	510	0,00	298,65	Vacaria
V06	5	510	0,00	403,30	Vacaria
V07	5	510	0,00	210,10	Vacaria
V08	5	510	0,00	136,30	Vacaria
V09	5	510	0,00	154,50	Vacaria
V10	5	510	0,00	42,20	Vacaria
V11	5	510	0,00	65,50	Vacaria
V12	5	510	0,00	177,30	Vacaria
V13	5	510	0,00	135,39	Vacaria
V14	5	510	0,00	151,81	Vacaria
V15	5	510	0,00	291,48	Vacaria
V16	5	510	0,00	245,60	Vacaria
V17	5	510	0,00	188,30	Vacaria
V18	5	510	0,00	76,20	Vacaria
V19	5	510	0,00	327,20	Vacaria
V20	5	510	0,00	168,30	Vacaria
V21	5	510	0,00	143,41	Vacaria
V22	5	510	0,00	36,60	Vacaria
V23	5	510	0,00	72,96	Vacaria
V24	5	510	0,00	152,2	Vacaria
V25	5	510	0,00	86,73	Vacaria
V26	5	510	0,00	18,33	Vacaria
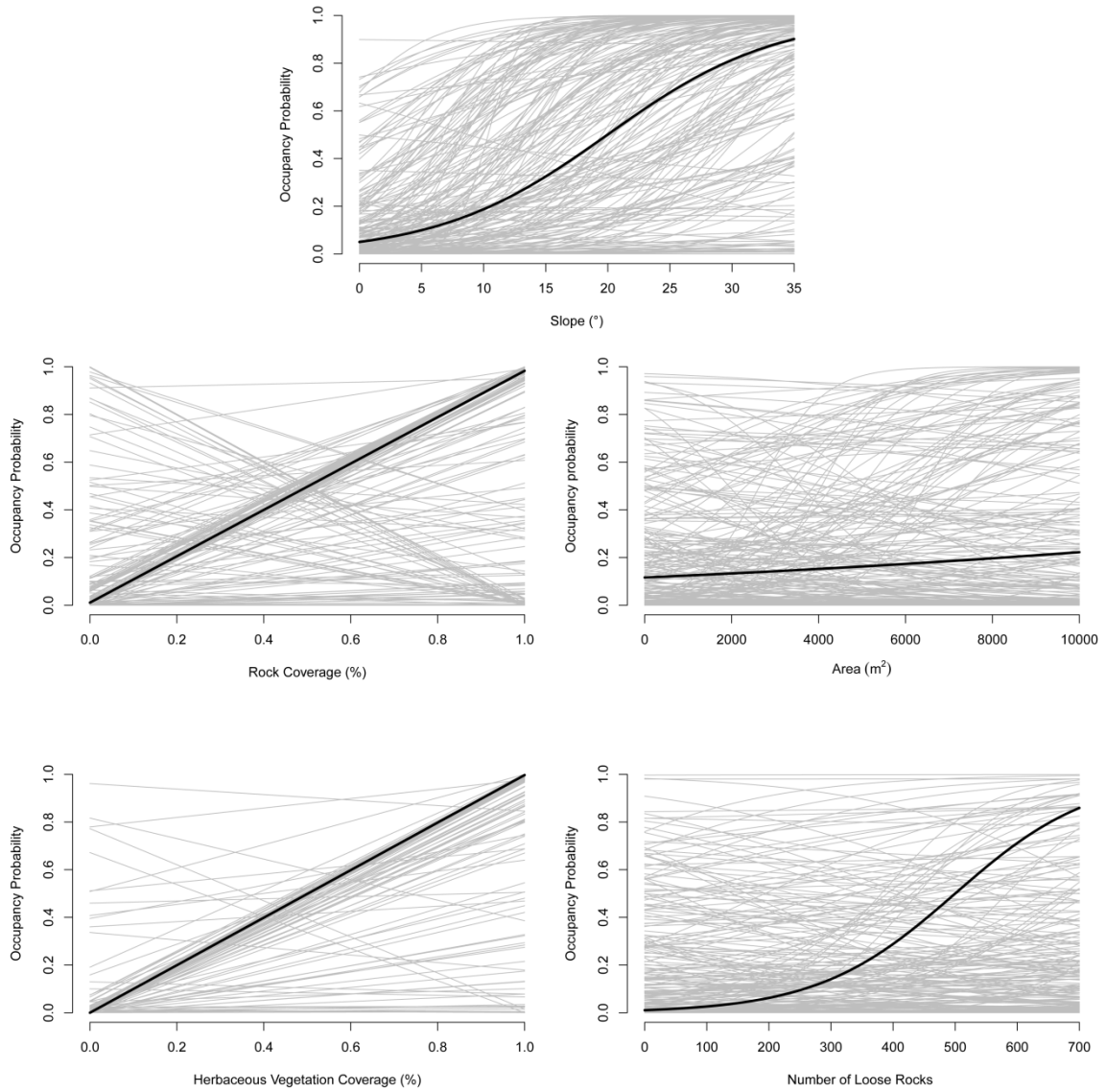
Figure 4 Efects of covariates on the probability of occurrence of *Contomastix vacariensis*, The black line represents the average prediction, while the grey lines represent 200 random draws from the posterior distribution, describing the uncertainty of the prediction,

CONCLUSÃO GERAL

As exigências biológicas, térmicas, nutricionais e ambientais do *C. vacariensis*, bem como os impactos ocasionado pela silvicultura de *Pinus sp.*, fazem com que essa espécie sofra um alto grau de ameaça por esta atividade. Fica evidente que os impactos ocasionados pelas silviculturas de exóticas, especialmente quando há conversão de áreas abertas naturais em ambientes florestados causam impactos consideráveis a espécies adaptadas a este tipo de ambiente. Impactos que não se limitam a área diretamente alterada, mas que podem afetar na distribuição das espécies em distanciamentos mais longos.

A capacidade de dispersão e a facilidade de colonizar áreas abertas tornam o *Pinus sp.* uma ameaça aos campos nativos e a sua conservação. Os Campos de Altitude da Serra apresentam uma ampla gama de espécies endêmicas, inclusive ameaçadas, como é o caso do *C. vacariensis*. Este estudo enfatiza a necessidade de se ter um maior controle desta atividade nesta região, e que as áreas prioritárias para conservação, bem como áreas protegidas, estejam afastadas das silviculturas, a fim de evitar seus efeitos negativos. O distanciamento neste estudo demonstra um efeito negativo de até 2.000m, contudo, é provável que o efeito seja gradual, com maior intensidade na área plantada e menos perceptível em áreas mais distantes. Além disto, os impactos à longa distância podem ser diferentes devido a características específicas do ambiente (e.g matas nativas próximas, zonas de amortecimento para impactos).

É importante destacar que lagartos endêmicos já foram extintos devido a atividades humanas, e que neste estudo constatamos a extinção de uma população de *C. vacariensis* por influência da silvicultura. Sugerimos que o distanciamento de no mínimo 2.000m de silviculturas seja uma referência para criação de áreas de proteção. Além disto, os órgãos ambientais poderiam adotar como norma regulatória para o licenciamento da atividade de silvicultura, uma vez que se constadado a presença da espécie *C. vacariensis* respeitando o distanciamento mínimo de 2.000m.